



# The influence of aquatic buffer zone vegetation on river macroinvertebrate communities



Cormac Mc Conigley<sup>a,\*</sup>, Heather Lally<sup>a,b</sup>, Declan Little<sup>c</sup>, Philip O'Dea<sup>d</sup>, Mary Kelly-Quinn<sup>a</sup>

<sup>a</sup> School of Biology and Environmental Science, University College Dublin, Belfield, Dublin 4, Ireland

<sup>b</sup> Marine and Freshwater Research Centre, Galway-Mayo Institute of Technology, Dublin Road, Galway City, Ireland

<sup>c</sup> Woodlands of Ireland, Suite 2, Wicklow Enterprise Park, The Murrough, Wicklow, Ireland

<sup>d</sup> Coillte, Dublin Road, Newtownmountkennedy, Co Wicklow, Ireland

## ARTICLE INFO

### Article history:

Received 20 March 2017

Received in revised form 8 June 2017

Accepted 19 June 2017

Available online 30 June 2017

### Keywords:

Riparian buffer zones

Conifer plantations

Forestry

Sitka spruce

## ABSTRACT

Aquatic buffer zones (ABZs) are areas of the riparian zone managed to protect streams from the potential impacts of commercial conifer forests and associated forestry operations in Ireland. Previous research identified that the dominant vegetation communities in ABZs are wet grassland and scrub habitats and that the vegetation communities are broadly determined by the soil type (peaty or mineral). One of the primary management options available to forest managers is whether to plant native broadleaves in the ABZ or leave them to be naturally colonised. The present study sought to determine the degree to which the vegetation of ABZs influences the composition and structure of associated aquatic macroinvertebrate communities and how this might inform their future management in particular the decision of whether to afforest the ABZ with broadleaf trees. The macroinvertebrate communities from stream on peaty and mineral soils with ABZs dominated by four vegetation communities were sampled, streams bordered by grassland habitats, scrub habitats, no ABZ i.e. conifer trees planted to the streams edge and control streams with no conifer forests in the vicinity. The results of this study showed that the composition and structure of the macroinvertebrate community varied significantly between streams on peaty and mineral soils, but was largely unaffected by the ABZ type. The differences between communities across soil type can be attributed to the lower abundance and richness of Ephemeroptera and conversely the higher abundance and richness of Plecoptera in streams on peaty soils. The lack of a consistent effect of ABZ type on the macroinvertebrate community illustrates the more dominant effect of water chemistry driven by soil type particularly episodic acidity.

© 2017 Elsevier B.V. All rights reserved.

## 1. Introduction

The riparian zone is the area along a water body that marks the interface between aquatic and terrestrial ecosystems. In the case of rivers, it is generally considered to include the river banks and the area of land which is influenced by inundation during times of flooding (Gregory et al., 1991; Naiman and Décamps, 1997; Little et al., 2008). In headwater streams, the riparian zone may be small, in the order of a couple of meters (Hagan et al., 2006), while in large lowland rivers it extends to the maximum extent of the floodplain (Gregory et al., 1991). Changes to riparian zones vegetation arising from anthropogenic activities such as forestry, agriculture, urbanisation and mining, alter the chemical and physical conditions of rivers as well as the dynamics, structure and functioning of aquatic communities (Hynes, 1975; Gregory et al., 1991;

Sweeney, 1993; Naiman and Décamps, 1997; Sovell et al., 2000; Allan, 2004; Dosskey et al., 2010). Therefore, in many countries, areas of the riparian zone, often termed riparian buffer zones (RBZs) or aquatic buffer zones (ABZs), are established and maintained for the protection of water quality from the impacts of adjacent land-use. Water quality protection is among the many benefits from riparian vegetation that are becoming increasingly important in the context of payments for ecosystem services.

In Ireland, ABZs were made a mandatory feature in all forests planted or replanted since 1991 (Forest Service, 2000). The ABZs range from 10 to 25 m wide, depending on local soil conditions and slope. Their primary function is to minimise diffuse pollution inputs from forestry operations, in particular, sediment and nutrients, which can arise during potentially high impact forest operations, such as road construction, harvesting and ground preparation for replanting (Leeks and Roberts, 1987; Kreutzweiser et al., 2005; Finnegan et al., 2012; Clarke et al., 2015).

\* Corresponding author.

E-mail address: [cormacmconigley@gmail.com](mailto:cormacmconigley@gmail.com) (C. Mc Conigley).

Much of the research to date on riparian buffer zones has focussed on their capacity to attenuate nutrient and sediment pollution (Vought et al., 1994; Lee et al., 2003; Nieminen et al., 2005; Finnegan et al., 2012), with less focus on the links between the vegetation and the aquatic communities. Planting native broadleaved tree species and management might optimize beneficial interactions. The management strategy for ABZs in Ireland to date has been to leave them undisturbed and allow vegetation to naturally colonise from local seed sources. Thus, the vegetation composition and structure of ABZs is comparable to that of natural riparian communities (Mc Conigley et al., 2015). Future management options will need to consider the connectivity between riparian zones and adjacent water bodies and to question whether active management, in terms of native tree planting or maintenance of grassy swards and herbaceous vegetation has measurable effects on aquatic communities.

The presence of deciduous trees and shrubs in the riparian zone can influence aquatic macroinvertebrate communities though altering the availability of food sources. Trees for example can reduce the amount of light reaching the stream surface through shading, depending on the extent of overhang and the width of the aquatic zone. This can limit primary production (Vannote et al., 1980; Sweeney, 1993; Ghermandi et al., 2009) with consequences for secondary production and instream trophic structures (Kiffney et al., 2003; Lecerf et al., 2012) and thermal regimes, in particular, reducing maximum water temperatures (Broadmeadow et al., 2011; Ryan et al., 2013). Deciduous trees also provide large quantities of organic matter in the form of leaves and woody debris which contribute to the streams energy budget, forming the primary energy source in forested headwater streams (DeLong and Brusven, 1994; Wallace et al., 1997) as well as influencing *in situ* macroinvertebrate communities. Inputs of coarse particulate organic matter, such as leaves have been shown to increase the abundance of shredder species (Drake, 1984; Richardson, 1991; Wallace et al., 1997) and yield increased metabolic activity, arising from the high-energy input (Sweeney et al., 2004; Bott et al., 2006). Leaves also make up one of the main food sources for many detritivores invertebrates (Sweeney, 1993; Heino, 2005) and the exclusion of allochthonous leaf material has been found to reduce abundances and biomass of macroinvertebrates in general (Wallace et al., 1999). Furthermore, woody debris aids in retaining other organic matter within stream reaches where it can be utilised by micro- and macro organisms (Speaker et al., 1984; Wiley et al., 1990; Gregory et al., 1991). The occurrence of feeding groups such as shredders are important in processing allochthonous detritus particularly in a forested catchment. This can potentially increase the food resource available and influence higher trophic levels.

Grass-dominated bordered streams, on the other hand, receive more direct sunlight allowing for higher water temperatures during the summer, and greater primary production when compared with forested streams (Sweeney, 1993; Broadmeadow et al., 2011; Ryan et al., 2013). However, these streams would be expected to receive less allochthonous organic matter than those with a wooded riparian zone (Wiley et al., 1990), which may affect overall macroinvertebrate productivity. It could be expected therefore that streams with grass dominated riparian zones would have a macroinvertebrate community dominated by taxa that utilise autochthonous food sources such as grazers.

The present study sought to determine the degree to which the vegetation of ABZs influences the composition and structure of associated aquatic macroinvertebrate communities and how this might inform their future management. Soil type is an important factor influencing both riparian vegetation (Mc Conigley et al., 2015) and stream chemistry (Bluth and Kump, 1994), and therefore was considered in the study design.

It was hypothesised that:

Macroinvertebrate communities in streams with an ABZ would more closely resemble communities in control streams (no conifer forestry in the vicinity) than the community in streams with no ABZ.

Macroinvertebrate communities in streams with scrub ABZs would contain a larger proportion of shredders and associated filter feeders due to the availability of leaf litter, while grazers would predominate in ABZ types without trees in the riparian zone because increased light would shift food availability towards more benthic algae and less allochthonous inputs from the surrounding forest.

## 2. Materials and method

### 2.1. Site Selection

Sites were selected using aerial photography and ordnance survey maps (OSI Discovery map series 1:50,000 scale) to identify river sections of suitable length within conifer forest. Two soil types were chosen for comparison following on from the findings of Mc Conigley et al. (2015), that the primary separation in plant communities was between peaty (i.e. organic) and mineral soils. The most commonly encountered habitats in ABZs were wet grassland and scrub habitats (Mc Conigley et al., 2015). ABZs dominated by these habitats were compared with control sites which had no conifers present and streams where conifers were planted to the streams edge (i.e. no ABZ). This resulted in four ABZ types, summarised below, across two soil types, combined to give eight categories. Excluding the control sites, all had conifer forests on both sides of the stream. In almost all cases, the forests were dominated by *Picea sitchensis* Bongard (Sitka spruce) with occasional *Pinus contorta* Douglas ex Loudon (Lodgepole pine).

- **Streams bordered by grassy ABZs:** grassland habitats dominated the ABZ, in most cases wet grasslands dominated by *Molinia caerulea* (L.) Moench (purple moor-grass).
- **Streams bordered by scrub ABZs:** scrub habitats dominated the ABZ with species such as *Fraxinus excelsior* (L.), *Salix* spp., *Betula* spp., *Alnus glutinosa* (L.) and *Ulex europaeus* (L.) most common.
- **No ABZ:** streams with conifer trees planted to the streams edge. In all cases *P. sitchensis* was the dominant tree species.
- **Control streams:** streams with no conifer forests in the catchment, on peaty soils the streams were in open heathland with *Molinia caerulea* and *Calluna vulgaris* (L.) Hull, while on mineral soils, native woodland occurred with typical species including *Quercus petraea* (Mattuschka) Liebl, *F. excelsior* and *Betula* spp.

Four replicate streams within each combination of soil type and ABZ type (Table 1, 32 streams in total) were sampled over a four week period in spring (April–May) 2012 and again over a three week period in autumn (October) 2012. There is considerable variation in rainfall across Ireland, the west coast receives approximately 2000 mm in 250 days of rain annually while on the east coast 700 mm of rain falls in 190 days. In order to take account of the variations in climate particularly rainfall across the country half of the streams in each category were located in the west of the country and half in the east, with the exception of scrub buffered streams on peat, of which three of the sites were in the west (Fig. 1). Site location and characteristics are given in Table 1.

**Table 1**

Location and description of the 32 sites across soil types and four different ABZ vegetation types. Coordinates on the Irish national grid. O = stream order Tr. = tributary Grazing: Low = evidence of the presence of grazers (faeces, animal tracks), Medium = evidence of grazing taking place with obvious nipping of plants and some light poaching along crossings. Buffer slope; medium = 0–15%, steep = 15–30%.

Stream	O	Soil	ABZ type	Easting	Northing	Site bedrock	Channel slope	Buffer slope	Valley slope	Grazing	Width (m)	Depth (m)
Glencullen	3	Mineral	Control	321164	218273	Schist	Moderate	Steep	Moderate	Medium	8	0.2
Tr. Avonmore	3	Mineral	Control	318846	191264	Quartzite	Moderate/Steep	Steep	Moderate	Low	6.1	0.15
Owenwee	5	Mineral	Control	97444	280235	Quartzite	Moderate	Moderate/Steep	Moderate	Medium	6	0.35
Tr. Lough conn	2	Mineral	Control	116720	307072	Sandstone	Moderate	Moderate	Moderate	Low	3.8	0.08
Inchavore	3	Mineral	Grass	313300	204650	Granite	Moderate/Steep	Steep	Moderate	Low	12.6	0.22
Skerdagh	1	Mineral	Grass	101521	301312	Quartzite	Moderate	Moderate/Steep	Moderate	Low	5	0.21
Tr. Slaney	2	Mineral	Grass	294947	194784	Schist	Moderate	Steep	Moderate	Low	3.4	0.23
Tr. Arigna	1	Mineral	Grass	186596	319483	Sandstone	Steep	Steep	Steep	Low	1.5	0.1
Tr. Killaclogher	2	Mineral	No Buffer	155263	237481	Limestone	Moderate	Steep	Moderate	Low	1.17	0.1
Tr. Derrybawn	2	Mineral	No Buffer	317045	193276	Schist	Steep	Steep	Steep	Low	1.4	0.16
Tr. Delour	3	Mineral	No Buffer	230155	201730	Sandstone	Moderate	Steep	Moderate	Low	3.7	0.19
Arigna	2	Mineral	No Buffer	186400	320955	Sandstone	Steep	Steep	Steep	Low	1.2	0.2
Owenaher	3	Mineral	Scrub	142011	317369	Schist	Moderate/Steep	Moderate/Steep	Moderate	Low	5	0.3
Tr. Camcor	2	Mineral	Scrub	223284	204259	Sandstone	Moderate	Steep	Moderate	Low	4.5	0.27
Gorragh	2	Mineral	Scrub	231077	207977	Sandstone	Moderate	Moderate/Steep	Moderate	Low	6.3	0.17
Tr. Moy	1	Mineral	Scrub	140634	318205	Schist	Steep	Moderate/Steep	Steep	Low	0.9	0.07
Tr. Liffey	3	Peaty	Control	309579	211962	Granite	Moderate	Moderate/Steep	Moderate	Low	3	0.24
Cloghoge	1	Peaty	Control	312675	207732	Granite	Steep	Steep	Steep	Low	2.3	0.22
Owengarve	2	Peaty	Control	71843	256412	Schist	Steep	Steep	Steep	Low	0.6	0.15
Tr. Owenboliska	3	Peaty	Control	111292	234556	Granite	Moderate	Moderate	Moderate	Medium	3	0.22
Tr. King	2	Peaty	Grass	305665	200942	Granite	Steep	Steep	Steep	Medium	6.9	0.16
Glennamore	3	Peaty	Grass	302166	201293	Granite	Steep	Steep	Steep	Medium	8	0.33
Tr. Yellow	2	Peaty	Grass	133490	309964	Granite	Moderate	Moderate/Steep	Moderate	Low	5.2	0.3
Glendaturk	1	Peaty	Grass	91091	300934	Quartzite	Moderate	Moderate/Steep	Moderate	Low	1.67	0.22
Tr. Aghavannagh	1	Peaty	No Buffer	306339	186904	Granite	Moderate	Steep	Moderate	Low	6.5	0.25
Unnamed	3	Peaty	No Buffer	86451	303027	Schist	Moderate	Moderate	Moderate	Low	2.7	0.15
Unnamed	2	Peaty	No Buffer	107245	233991	Granite	Steep	Steep	Steep	Low	1.2	0.17
Glasnagollam	1	Peaty	No Buffer	306719	201907	Granite	Moderate	Steep	Moderate	Low	2.55	0.16
Tr. Carrowmore	3	Peaty	Scrub	86272	328654	Quartzite	Moderate	Steep	Moderate	Low	4.5	0.23
Tr. Crumpaun	2	Peaty	Scrub	104614	302141	Sandstone	Steep	Steep	Steep	Medium	3	0.2
Owenwee	2	Peaty	Scrub	90937	278832	Quartzite	Moderate	Steep	Moderate	Low	2.4	0.22
Tr. Little Brosna	2	Peaty	Scrub	210351	194209	Limestone	Moderate	Steep	Moderate	Low	3.5	0.32

## 2.2. Sampling methodology

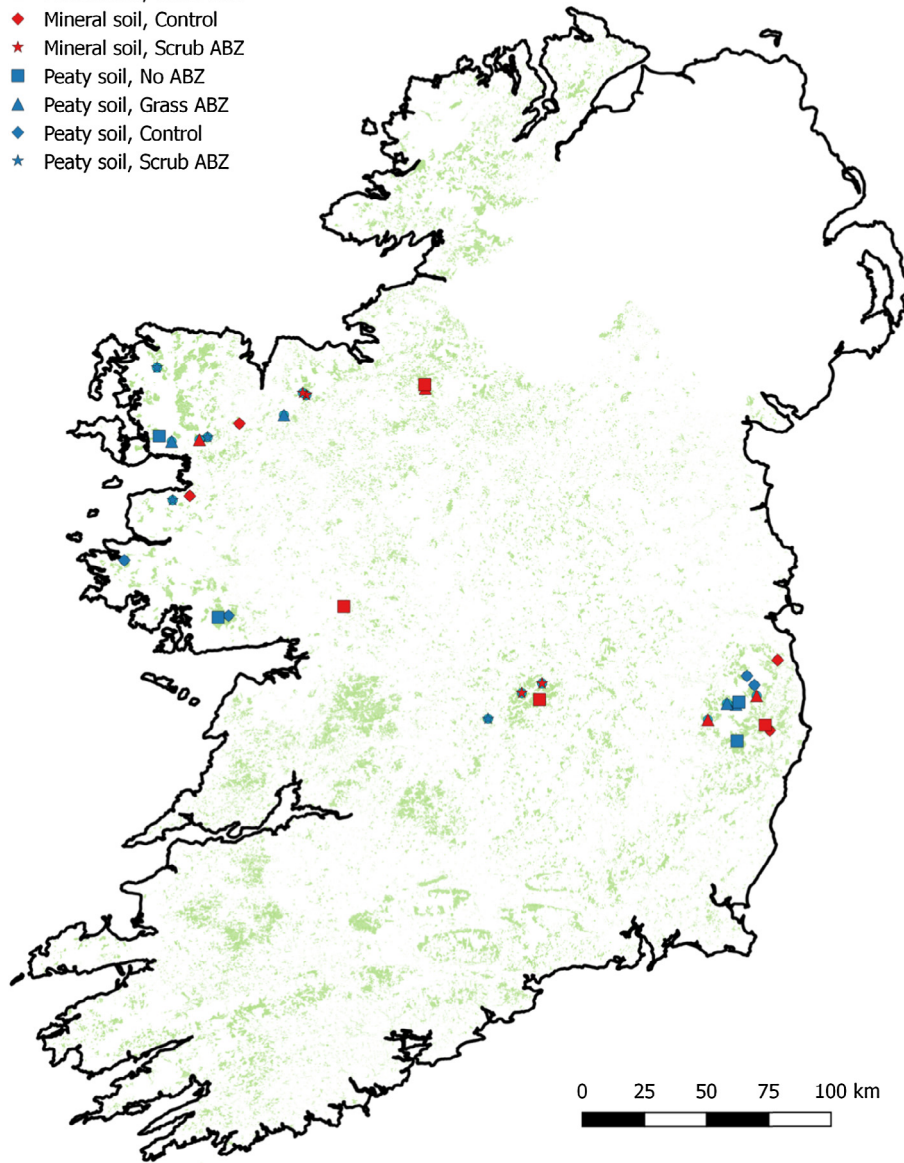
Streams were sampled at the end of the conifer forest to avoid potential effects of immediate shading, and at an equivalent position in the control streams. At each sampling point, six replicate Surber samples (500 µm mesh size with area 0.0625 m<sup>2</sup> (0.25 m × 0.25 m)) were collected in riffle and margins (streams edge) within a 50 m stream reach. The location of each Surber was determined using a random number table, placing half in the riffle and half in the margins. Surbers were collected by gently disturbing the sediment to a depth of approximately 100 mm with a trowel to ensure consistency across sites and seasons. In the laboratory, macroinvertebrates were removed and identified to

species level for the Ephemeroptera, Plecoptera and Trichoptera (EPT), and Elmidae. Diptera, Mollusca and other Coleoptera were identified to family or genus level, while Chironomidae, Simuliidae and Oligochaeta were not classified any further. All identification was carried out using standard keys (Hynes, 1977; Elliott et al., 1988; Wallace et al., 1990; Edigton and Hildrew, 1995; Nilsson, 1996, 1997).

Water samples were collected from each stream at each sampling time in spring and in autumn and analysed for: sodium (Na<sup>+</sup>) potassium (K<sup>+</sup>), magnesium (Mg<sup>2+</sup>), calcium (Ca<sup>2+</sup>), and aluminium (Al<sup>3+</sup>) (VARIAN SpectraAA 300), ammonia/ammonium (NH<sub>4</sub><sup>+</sup>), chloride (Cl<sup>-</sup>), and sulphate (SO<sub>4</sub><sup>2-</sup>) (Lachat Quick-Chem) using standard methods (Clesceri et al., 2001). Dissolved organic

### Legend

- Mineral soil, No ABZ
- ▲ Mineral soil, Grass ABZ
- ◆ Mineral soil, Control
- ★ Mineral soil, Scrub ABZ
- Peaty soil, No ABZ
- ▲ Peaty soil, Grass ABZ
- ◆ Peaty soil, Control
- ★ Peaty soil, Scrub ABZ



**Fig. 1.** Distribution of sampling sites across the country. Green areas represent forested areas identified in the Forest Inventory and Planning System (FIPS) GIS database (Forest Service, Department of Agriculture, Food and the Marine). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

carbon (DOC) was determined using high temperature combustion (SHIMADZU UV-160A).

### 3. Data preparation and analysis

#### 3.1. Biological data

A standardised taxon list was prepared for comparisons across all sites. In cases where individuals could not be identified to species level taxonomic adjustment was undertaken. If 80% or more were identified the unidentified portion was divided proportionally among the species of the genus present at that site. When a larger portion (>20%) was unidentified all species were brought to genus level. Rare taxa were removed from the dataset used for analysis of community structure and composition to reduce noise

in the dataset. A taxon was classed as rare if its abundance never exceeded five percent in any sample and it occurred in fewer than five percent of samples.

Spring and autumn data were analysed separately as there was significant variation across season.

Analysis comparing community structure and composition were run using a nested hierarchical design with four factor, stream order, soil (peaty or mineral), ABZ type and site. Soil type and ABZ type were both fixed factors while stream order and site were both random and site was nested in the other factors. Stream width and depth were included as continuous covariable to take account of the influence they may have on the community. Type I sum of squares (SS) (sequential) was used. Using type I SS the order that terms are placed in the model is important, as the SS of each term is calculated given the previous terms. By altering



the order the variance attributable to each term can be investigated, this allows for the variation across treatments to be investigated after accounting for the variances due to covariates.

Statistical analysis to test for significant differences between treatments was carried out using the PERMANOVA routine in PRIMER 6 & PERMANOVA + 1.0.2 (Clarke and Gorey, 2006; Anderson et al., 2008). PERMANOVA is a permutational multivariate analysis of variance, analogous to a multi factorial ANOVA but with the p value calculated through permutations. The routine calculates an F statistic, then randomly reassigns the labels and recalculates the F statistic with the new label arrangement. The p value can be calculated based on the portion of the permutations with an F value greater than the original arrangement. The rationale being that if rearranging the labels at random produces a higher F value than the true arrangement, then, the true arrangement is no more significant than random. PERMANOVA has the advantage of being non-parametric and can be run on any similarity or dissimilarity matrix.

A Bray-Curtis dissimilarity matrix was used to compare the community structure across categories. Species abundances were averaged per site and square root transformed before the matrix was calculated. Square root transforming the abundances reduces the effect of the most abundant species.

For comparisons of community composition between categories, abundance data were transformed to presence/absence. PERMANOVA was then run on a resemblance matrix calculated using Jaccard's index, which is a measure of the percentage of unshared species between two samples.

Similarity Percentage (SIMPER) analysis was carried out to identify the key taxa contributing to any significant differences between categories which were identified by PERMANOVA. SIMPER returns the mean abundance of each taxon in the categories under comparison and their contribution to the differences observed as a percentage of the total dissimilarity.

Results for community structure and composition were also illustrated using non-metric multi-dimensional (MDS) ordination. There is no absolute scale for the axes in MDS ordinations, points (treatments) are placed so as to preserve their rank similarity; points closest together are most similar while those furthest apart are least similar. How well the 2 dimensional ordinations represent the high dimensional relationship between samples is indicated by a stress value. The lower the stress the more accurately the MDS representation of the rank similarities of the data.

### 3.2. Functional feeding groups

The percentage of the community made up of the various feeding groups was determined using the ASTERICS computer program (version 3.1) (available from AQEM <http://www.asterics.de/en/download/berechnung/>). ASTERICS assigns taxa to functional feeding groups proportionately to their diet using figures available in Moog and Österreich (1995). Many taxa are split between several feeding groups. For example *Leuctra inermis* (Kempny) are divided 30% to grazers, 30% to shredders and 40% to gatherers.

### 3.3. Environmental data

Where pairs of environmental variables were strongly correlated (i.e.  $r^2 > 0.7$  and  $r^2 < -0.7$ ), one variable was eliminated to avoid redundancy (Haidekker and Hering, 2007). The remaining variables were normalised to have a mean of 0 and standard deviation of 1 and a resemblance matrix based of Euclidian distance calculated. This was compared to the Bray-Curtis dissimilarity matrix calculated from square root transformed abundance data

from the same season using the RELATE routine (Anderson et al., 2008). RELATE is a non-parametric form of a Mantel test, which tests for correlation between two matrices, in this case the triangular matrices of the biological and environmental data. Initially the measure of agreement in the rank correlation coefficient in calculated ( $\rho$ ), where  $\rho = 0$  is the null hypothesis of no relationship. The routine then randomly permutes the set of labels of one matrix relative and recalculates  $\rho$ . The p value of the correlation is determined by comparing the original value  $\rho$  to a histogram of the random  $\rho$  values.

DistLM (Distance Linear Modelling) was used to find the most parsimonious set of environmental variables that had the strongest correlation with the community data (Anderson et al., 2008). DistLM is analogous to a regression (or multiple regression) model.

## 4. Results

### 4.1. Biological data

Across all sites over 30,000 individual macroinvertebrates were identified, yielding 84 taxa. There were no significant differences in taxon richness or mean abundance in either season across soil or ABZ type.

There was no significant effect of ABZ type on the macroinvertebrate community composition (presence/absence data) within a soil type in either spring ( $F_{3,31} = 1.1$   $p = 0.38$ ) or autumn ( $F_{3,31} = 0.89$   $p = 0.66$ ) (Fig. 2). The communities did however differ between the two soil types in autumn ( $F_{1,31} = 3.05$   $p = 0.02$ ) but not in spring ( $F_{1,31} = 2.2$   $p = 0.08$ ) though  $p = 0.08$  is near significance. In autumn the difference between the communities across soils was driven by a greater occurrence of the mayflies *Rhithrogena semicolorata* (Curtis) and *Ecdyonurus* spp. in streams on mineral soil (Table 2). In the spring the freshwater shrimp *Gammarus duebeni* (Liljeborg) was more common in streams on mineral soils while the stoneflies *Leuctra inermis*, *Amphinemura sulciollis* (Stephens) and *Siphonoperla torrentium* (Pictet) were more often found in streams on peaty soil (Table 2). In autumn,

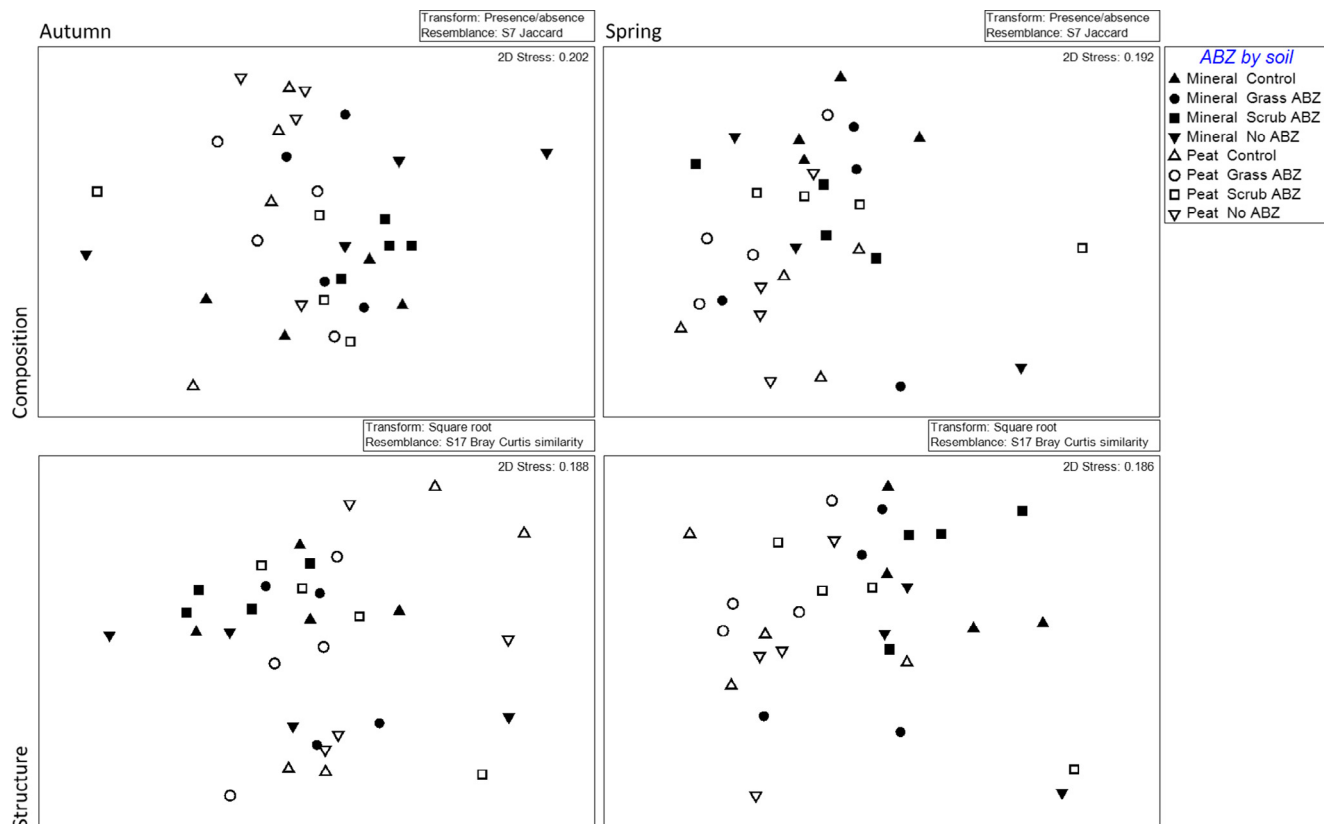
Similar to composition, there was no significant effect of ABZ type on the macroinvertebrate community structure in either spring ( $F_{3,31} = 0.95$   $p = 0.56$ ) or autumn ( $F_{3,31} = 0.94$   $p = 0.58$ ) (Fig. 2). There was an effect of soil type in both spring ( $F_{1,31} = 5.7$   $p < 0.01$ ) and autumn ( $F_{1,31} = 3.37$   $p = 0.02$ ) (Fig. 2). In spring, the main drivers were the greater abundances of Chironomidae (non-biting midge larvae), *L. inermis*, and *B. rhodani*, on peat and higher abundance of *G. duebeni* and *R. semicolorata* at sites on mineral soil (Table 3). While in autumn, *R. semicolorata* made the largest contribution to the dissimilarity between streams on the two soil types, being on average approximately five times as abundant at sites on mineral soil, while *P. meyeri* and *Oligochaeta* were more abundant at sites on peaty soil (Table 3).

### 4.2. Functional feeding groups

In autumn streams on peaty soil with a grass ABZ had significantly more grazers than those with no ABZ ( $t = 2.5621$   $p(\text{mc}) = 0.0446$ ) (Fig. 3) primarily due to higher numbers of *Silo pallipes* and *B. rhodani*.

There was no significant difference in the proportion of other functional feeding groups between the various ABZs on either soil type in spring ( $F_{3,155} = 0.9542$   $p = 0.427$ ) (Fig. 4).

Notably grazers constituted a significantly higher percentage of the macroinvertebrate community in streams on mineral soil than those on peaty soils in both spring ( $F_{1,155} = 4.9415$   $p = 0.034$ ) and autumn ( $F_{1,157} = 4.6088$   $p = 0.0412$ ) (Figs. 3 and 4).



**Fig. 2.** Multi-Dimensional Scaling (MDS) plot of macroinvertebrate community composition and structure in the spring and autumn. Buffer type denoted by symbol, peaty soil is denoted by hollow points and mineral soil is denoted by solid point.

**Table 2**  
SIMPER results identifying taxa with the largest contribution to the significant difference between the community composition in streams on peaty and mineral soils (ignoring ABZ type as it had no significant effect).

		Spring			Autumn		
		Mineral	Peat	% Contribution	Mineral	Peat	% Contribution
Ephemeroptera	<i>Rhithrogena semicolorata</i>	0.73	0.31	2.58	0.69	0.38	2.9
	<i>Ecdyonurus</i> spp.	0.53	0.19	2.2	0.56	0.25	2.67
	<i>Baetis rhodani</i>	0.93	0.88	0.91	0.94	0.56	2.47
Plecoptera	<i>Isoperla grammatica</i>	0.47	0.75	2.36	0.44	0.44	2.53
	<i>Amphinemura sulcicollis</i>	0.47	0.81	2.39	0.31	0.5	2.52
	<i>Siphonoperla torrentium</i>	0.8	0.94	1.23	0.56	0.44	2.67
	<i>Leuctra inermis</i>	0.47	0.75	2.36	0.63	0.5	2.59
Coleoptera	<i>Limnius volckmari</i>	0.87	0.69	1.81	0.5	0.75	2.64
	<i>Elmis aenea</i>	0.53	0.5	2.24	0.63	0.44	2.77
Tricoptera	<i>Silo pallipes</i>	0.8	0.44	2.49	0.75	0.44	2.87
Amphipoda	<i>Gammarus duebeni</i>	0.93	0.38	2.77	0.81	0.56	2.52

#### 4.3. Environmental data

Nutrient concentrations were generally low across sites with little variation in concentrations of P, nitrite or ammonia between treatments and are not presented. Nitrate correlated strongly with total N and was removed from the analysis. Potassium and sodium were also removed as it correlated strongly with magnesium and chloride respectively.

In spring, streams on peaty soil had significantly lower pH than those on mineral soils ( $F_{1,23} = 4.1055$   $p = 0.042$ ) (Table 4). There were no significant difference in the pH of streams between ABZ types within a soil type ( $F_{3,23} = 1.2437$   $p = 0.32$ ) (Table 4). In autumn, there was a significant interaction between soil and buffer type ( $F_{3,24} = 4.2339$   $p = 0.012$ ). Pairwise comparison found that on

mineral soils there were no significant differences in pH between the various ABZ types. However, on peaty soils streams with no buffer were significantly more acidic than those with a scrub buffer ( $t = 2.856$   $p(\text{mc}) = 0.027$ ) and grass buffer ( $t = 2.4058$   $p(\text{mc}) = 0.045$ ) but not to control streams ( $t = 2.2106$   $p(\text{mc}) = 0.069$ ) (Table 4).

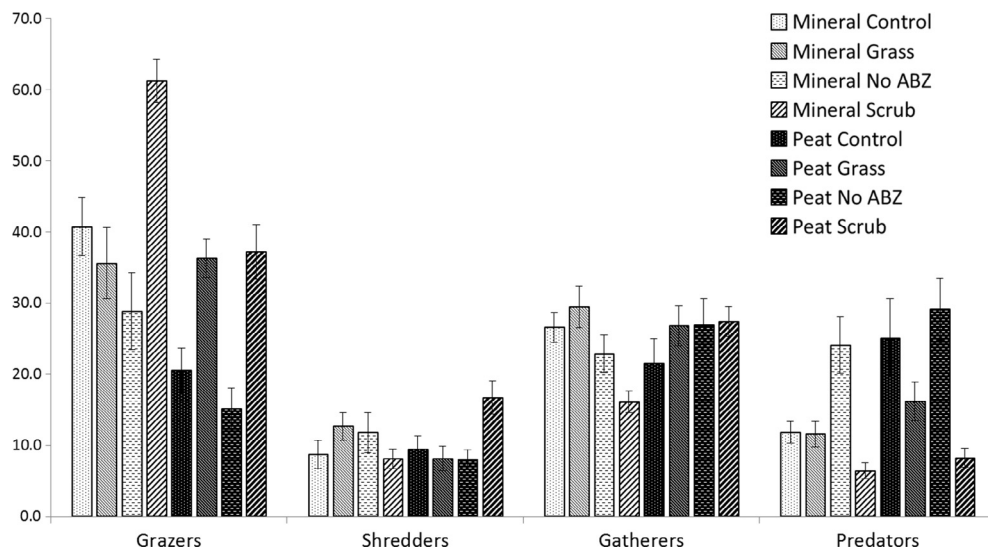
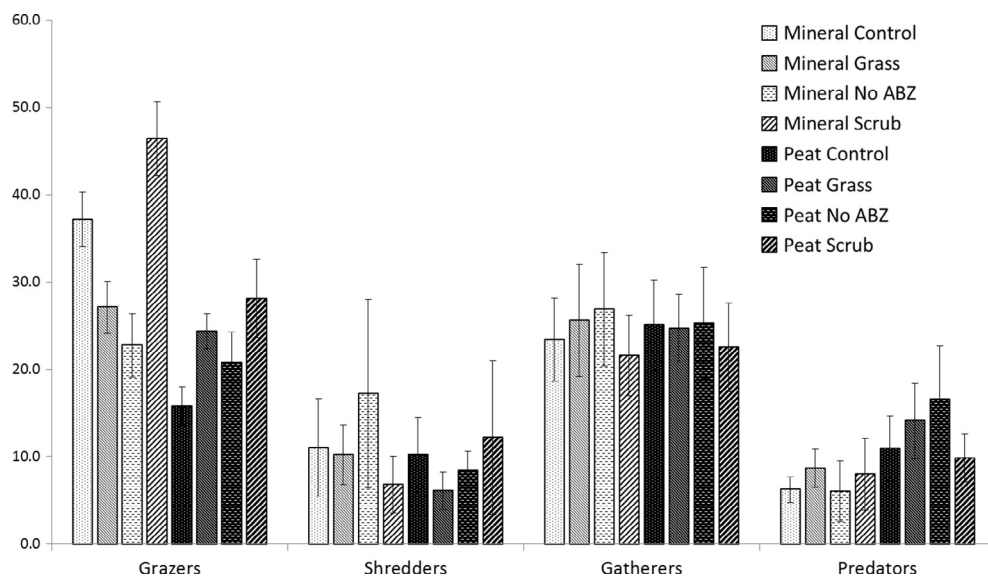
#### 4.4. Abiotic drivers of community structure

The abiotic variables used in analysis are listed in Table 4. In spring there was a significant correlation of 0.422 ( $p = 0.001$ ) between the environmental and biological data. pH and conductivity as the most important contributors to the correlation accounting for 14 and 11 percent of the variability respectively.

**Table 3**

SIMPER results identifying taxa with the largest contribution to the significant difference in community structure between sites on peat and mineral soil (ignoring buffer type).

		Spring			Autumn		
		Mineral	Peat	% Contribution	Mineral	Peat	% Contribution
Ephemeroptera	<i>Baetis rhodani</i>	1.36	2.24	6.03	1.41	1.01	5.65
	<i>Rhithrogena semicolorata</i>	1.8	0.55	5.14	2	0.39	7.3
Plecoptera	<i>Leuctra inermis</i>	0.83	2.45	6.67	0.51	0.49	3.15
	<i>Siphonoperla torrentium</i>	0.67	1.63	4.37	0.19	0.25	1.21
	<i>Amphinemura sulciollis</i>	0.21	1.45	3.89	0.12	0.31	1.05
Tricoptera	<i>Silo pallipes</i>	0.84	0.67	2.87	0.79	1.04	4.54
Diptera	<i>Chironomidae</i> spp.	2.45	4.33	9.2	1.03	1.72	5.89
	<i>Simuliidae</i> spp.	1.21	1.23	4.58	0.65	0.86	3.98
Coleoptera	<i>Limnius volckmari</i>	0.81	0.79	3.06	0.98	1.03	4.75
Oligochaetes	<i>Oligochaetes</i> spp.	0.83	0.76	3.04	1.02	1.8	5.92
Amphipoda	<i>Gammarus duebeni</i>	2.03	0.83	6.87	1.4	0.98	6.49

**Fig. 3.** Percentage abundance ( $\pm$ standards error) of four functional feeding groups at sites in the eight treatments sampled in autumn.**Fig. 4.** Percentage abundance ( $\pm$ standards error) of four functional feeding groups at sites in the eight treatments sampled in spring.

**Table 4**  
Selection of abiotic variables used in the RELATE and DistLM analysis.

	Soil	ABZ type	Ph	Cond. ( $\mu\text{S cm}^{-1}$ )	Aluminium ( $\mu\text{g Al l}^{-1}$ )	TON ( $\text{mg N l}^{-1}$ )	DOC ( $\text{mg C l}^{-1}$ )	Magnesium ( $\text{mg Mg l}^{-1}$ )	Calcium ( $\text{mg Ca l}^{-1}$ )	Chloride ( $\text{mg Cl}^{-1} \text{ l}^{-1}$ )
Spring	Min	Control	7.54 $\pm$ 0.23	169.75 $\pm$ 39.74	18.58 $\pm$ 10.00	0.94 $\pm$ 0.50	5.31 $\pm$ 2.18	2.66 $\pm$ 0.38	16.71 $\pm$ 6.17	16.56 $\pm$ 2.93
	Min	Grass	7.01 $\pm$ 0.30	76.25 $\pm$ 19.29	27.70 $\pm$ 14.61	0.71 $\pm$ 0.35	4.94 $\pm$ 1.39	1.29 $\pm$ 0.39	5.70 $\pm$ 2.25	9.36 $\pm$ 2.22
	Min	No Buffer	7.40 $\pm$ 0.27	261.33 $\pm$ 178.72	9.10 $\pm$ 3.01	6.27 $\pm$ 5.23	4.01 $\pm$ 1.44	2.69 $\pm$ 1.44	30.86 $\pm$ 22.63	10.23 $\pm$ 4.09
	Min	Scrub	7.62 $\pm$ 0.25	115.00 $\pm$ 29.08	26.55 $\pm$ 9.60	0.18 $\pm$ 0.06	8.75 $\pm$ 1.15	1.54 $\pm$ 0.26	15.99 $\pm$ 3.08	8.21 $\pm$ 1.58
	Peat	Control	6.42 $\pm$ 0.42	44.25 $\pm$ 7.56	93.18 $\pm$ 31.68	4.75 $\pm$ 4.10	7.19 $\pm$ 2.38	0.88 $\pm$ 0.36	2.30 $\pm$ 1.21	10.08 $\pm$ 3.31
	Peat	Grass	7.08 $\pm$ 0.36	88.75 $\pm$ 40.08	9.68 $\pm$ 1.98	0.22 $\pm$ 0.14	3.28 $\pm$ 0.26	0.97 $\pm$ 0.32	7.36 $\pm$ 5.96	10.55 $\pm$ 3.11
	Peat	No Buffer	5.95 $\pm$ 0.25	44.75 $\pm$ 14.48	70.40 $\pm$ 33.34	0.67 $\pm$ 0.16	5.80 $\pm$ 1.69	0.74 $\pm$ 0.21	1.23 $\pm$ 0.26	10.68 $\pm$ 2.97
	Peat	Scrub	7.36 $\pm$ 0.17	188.8 $\pm$ 83.98	15.25 $\pm$ 5.60	0.29 $\pm$ 0.21	8.37 $\pm$ 2.82	3.74 $\pm$ 0.98	25.40 $\pm$ 17.71	19.49 $\pm$ 2.91
Autumn	Min	Control	6.63 $\pm$ 0.29	143 $\pm$ 28.12	22.50 $\pm$ 8.47	0.75 $\pm$ 0.38	6.02 $\pm$ 1.83	2.07 $\pm$ 0.14	11.30 $\pm$ 3.34	14.38 $\pm$ 1.47
	Min	Grass	6.50 $\pm$ 0.22	58.25 $\pm$ 10.66	58.98 $\pm$ 49.55	0.48 $\pm$ 0.41	6.73 $\pm$ 2.51	0.98 $\pm$ 0.24	3.24 $\pm$ 1.60	7.90 $\pm$ 1.21
	Min	No Buffer	7.15 $\pm$ 0.26	224.5 $\pm$ 150.24	15.55 $\pm$ 6.87	0.49 $\pm$ 0.21	5.71 $\pm$ 2.80	3.07 $\pm$ 1.11	20.37 $\pm$ 12.97	9.30 $\pm$ 1.87
	Min	Scrub	6.68 $\pm$ 0.06	118.25 $\pm$ 11.92	18.23 $\pm$ 3.91	0.10 $\pm$ 0.02	7.46 $\pm$ 1.10	1.63 $\pm$ 0.13	12.21 $\pm$ 2.63	8.80 $\pm$ 1.33
	Peat	Control	6.15 $\pm$ 0.32	54.5 $\pm$ 9.56	105.55 $\pm$ 35.04	0.03 $\pm$ 0.01	10.51 $\pm$ 2.23	0.76 $\pm$ 0.15	1.53 $\pm$ 0.47	9.08 $\pm$ 2.39
	Peat	Grass	6.65 $\pm$ 0.55	40.25 $\pm$ 8.12	25.98 $\pm$ 5.40	0.04 $\pm$ 0.01	5.67 $\pm$ 0.58	0.60 $\pm$ 0.12	1.87 $\pm$ 0.95	6.99 $\pm$ 2.23
	Peat	No Buffer	5.23 $\pm$ 0.24	44.75 $\pm$ 5.54	111.4 $\pm$ 50.55	0.16 $\pm$ 0.05	8.83 $\pm$ 0.61	0.56 $\pm$ 0.12	0.87 $\pm$ 0.13	8.05 $\pm$ 2.07
	Peat	Scrub	6.46 $\pm$ 0.10	206 $\pm$ 143.44	31.23 $\pm$ 17.04	0.68 $\pm$ 0.55	13.92 $\pm$ 6.17	3.31 $\pm$ 2.04	24.05 $\pm$ 21.19	15.86 $\pm$ 1.39

In autumn, there was weaker but still significant correlation of 0.28 ( $p = 0.008$ ) between the environmental and biological data. Chloride, aluminium and conductivity were the most useful combination of abiotic factors.

## 5. Discussion

Field experiments, such as the study presented here, provide an opportunity to test a hypothesis under the relevant natural settings, including the myriad of chemical and biological interactions present, rather than controlled laboratory experiments, which may lack ecological realism. However, these conditions complicate the interpretation of results as the influences of individual factors are difficult to isolate, particularly in complex and heterogeneous environments.

The results of this study showed that the composition and structure of the macroinvertebrate community varied significantly between streams on peaty and mineral soils, but was largely unaffected by the ABZ type. The differences between communities across soil type can be attributed to the lower abundance and richness of Ephemeroptera and conversely the higher abundance and richness of Plecoptera in streams on peaty soils. The main abiotic driver of this difference was likely the generally more acidic conditions in streams draining peaty soils.

Ephemeroptera are sensitive to acid conditions ( $\text{pH} < 5.5$ ) (Ormerod et al., 1987; Sutcliffe and Hildrew, 1989; Kowalik and Ormerod, 2006; Petrin et al., 2007) which may be common in streams draining peaty soils and unlikely to be ameliorated by ABZs, regardless of type. As expected, Plecoptera, which are more acid-tolerant (Davy-Bowker et al., 2005; Feeley and Kelly-Quinn, 2014) were more abundant in the streams on peaty soils, particularly *L. inermis*, *A. sulcicollis*, *P. meyeri* and *S. torrentium* (Feeley et al., 2011; Feeley and Kelly-Quinn, 2014). The influences of acidity on the community override that of the vegetation in the ABZ. The expected grazing Ephemeroptera in streams with open canopied ABZ vegetation are excluded by acidity on peaty soils (Rutt et al., 1990; Ormerod et al., 2004). On mineral soils where acidity from the soil is not limiting, grazers made up a smaller proportion of the community in streams with no ABZ than any other ABZ type. Though the difference was not significant in this instance it points

toward the influence of ABZ vegetation. In the absence of ABZ, conifers planted to the streams edge may limit availability of suitable food for grazers.

Noteworthy, at the time of sampling, most streams had a circumneutral pH with only a few (peaty sites with no ABZ) below pH 5.5. Sampling occurred at times of low flow and therefore would have missed episodic acidity typical of sites draining peaty soils. Stream pH may drop during episodes of increased discharge due to processes such as base-cation dilution, interception of sulphate and nitrate from anthropogenic emissions, release of organic acids, and oxidation of sulphur in organic soils, particularly peat (Wigington et al., 1996; Laudon and Bishop, 1999; Ormerod and Durand, 2009). In Ireland, the effect of acidification has been associated most strongly with peaty soils on acid-sensitive geology and areas with higher cover of conifer forests (Kelly-Quinn et al., 2008, 2016; Feeley et al., 2013). Episodic acidity driven by base cation dilution and organic acidity occur most during the winter months and the effects are most obvious in spring (Feeley et al., 2013; Feeley and Kelly-Quinn, 2014). This may explain the lower proportions of grazers in streams with scrub or grass ABZs on peaty soils in spring.

It was hypothesised that shredding taxa would represent a greater proportion of the community in streams with scrub ABZs due to a greater availability of leaf litter in those streams. It was the case on peaty soils that streams with scrub ABZs had a higher percentage of shredders in both seasons but the difference was not significant. The reasons for the lack of significant effect of ABZ type on the proportion of shredders may be due to the uncertainty in the availability of leaf litter in the stream or the plasticity of macroinvertebrate feeding strategies, possibly both these factors play a role.

The feeding habitats of taxa may vary at different larval stages and many taxa are highly plastic in their feeding strategies and are considered to be generalist (Moog and Österreich, 1995; Mihuc, 1997; Dangles, 2002; Moog, 2002). This is addressed to some extent by dividing taxa abundance proportionally based on estimates of their percentage utilisation of each feeding strategies, as was done in the current study. However, the lack of certainty in what food source a taxon is consuming in any given time period may obscure small changes in the resources utilized by the community. Large differences in available resources such as the



presence or absence of leaf litter would be expected to be obvious in the community composition (Vannote et al., 1980). If however the difference in resource availability was small or of short duration it may be obscured as taxa shift their feeding habits.

Streams with a scrub ABZ may not receive as large an input of leaf litter or as little sunlight as expected. The contribution of leaf litter from adjacent riparian zones depends on where the trees and shrubs are located relative to the stream. Inputs of allochthonous litter to streams likely declines as the distance of the tree from the stream edge increases (Brooks et al., 2012). Similarly the degree of shade cast on the stream will also decrease with distance or increasing stream width. The trees and shrubs in the scrub ABZs in the present study were at various distances from the stream bank and their potential inputs are difficult to estimate. Deciduous trees in the scrub ABZs were also generally young (<2 m tall) as ABZs have only been a feature of forestry since 1991. Future research when trees within these ABZs mature may find that the community more typical of deciduous woodland streams with greater abundances of shredding taxa.

## 6. Conclusions and implications for management

The lack of a consistent effect of ABZ type on the macroinvertebrate community illustrates the more dominant effect of water chemistry, particularly episodic acidity which challenges forest managers to maximise the benefits from deciduous tree planting in ABZs to positively affect aquatic communities. Tree species are not a feature on peaty soils and the planting of trees on this soil type is not advised unless to ameliorate projected elevated water temperatures in the future due to climate change, or for enhancement of instream production through increased leaf litter input. Strategic planting to facilitate inputs of leaf material to low-productivity streams draining peat could potentially increase the production of shredding Plecoptera and Trichoptera (Dobson et al., 1995), particularly Plecoptera which dominate episodically acidic streams. They could also provide food resources (e.g. pollen) for adult Plecoptera if appropriate species are planted.

Tree planting would be a more valuable and feasible management option for ABZs on mineral soils. Planting mixtures should match site type and should comprise native species such as *Salix* spp., *Betula* spp., *Alnus glutinosa* (L.). However, it is unclear if at the catchment or sub-catchment scale (entire stream systems) the benefits of native tree planting would be as obvious at a reach scale (Ormerod et al., 1993; Pretty and Dobson, 2004). Further research is required to address this knowledge gap.

While tree planting may increase allochthonous inputs to streams a further challenge is retention of that material within adjacent stream reaches. Observations on most of the sites studied highlighted the lack of obvious leaf material and large woody debris instream. The spaty nature of these streams clearly limits retention. Options to improve leaf litter retention in appropriate locations should be explored, e.g. small, low gradient tributary streams or drainage channels. This may allow the benefits of tree planting in ABZs to be reflected in the macroinvertebrate communities of adjacent streams (Pretty and Dobson, 2004).

## Acknowledgements

This project was funded by COFORD, the Department of Agriculture, Food and the Marine. We acknowledge Woodlands of Ireland and Coillte, Technical Services for their assistance and technical input into the study as well as Coillte forest management for facilitating site access. We would also like to thank referees and editors of the journal for their valuable input to the manuscript.

## References

- Allan, J.D., 2004. Landscapes and riverscapes: The influence of land use on stream ecosystems. *Annu. Rev. Ecol. Evol. Syst.* 35, 257–284.
- Anderson, M.J., Gorley, R.N., Clarke, K.R., 2008. PERMANOVA + for PRIMER: Guide to Software and Statistical Methods. PRIMER-E Ltd, Plymouth, UK.
- Bluth, G.J.S., Kump, L.R., 1994. Lithologic and climatologic controls of river chemistry. *Geochim. Cosmochim. Acta* 58, 2341–2359.
- Bott, T.L., Newbold, J.D., Arscott, D.B., 2006. Ecosystem metabolism in piedmont streams: Reach geomorphology modulates the influence of riparian vegetation. *Ecosystems* 9, 398–421.
- Broadmeadow, S.B., Jones, J.G., Langford, T.E.L., Shaw, P.J., Nisbet, T.R., 2011. The influence of riparian shade on lowland stream water temperatures in southern England and their viability for brown trout. *River Res. Appl.* 237, 226–237.
- Brooks, R.T., Nislow, K.H., Lowe, W.H., Wilson, M.K., King, D.I., 2012. Forest succession and terrestrial-aquatic biodiversity in small forested watersheds: A review of principles, relationships and implications for management. *Forestry* 85, 315–327.
- Clarke, J., Kelly-Quinn, M., Blacklocke, S., Bruen, M., 2015. The effect of forest windrowing on physico-chemical water quality in Ireland. *Sci. Total Environ.* 514, 155–169.
- Clarke, K.R., Gorley, R.N., 2006. PRIMER v6: User Manual/ Tutorial. PRIMER-E Ltd, Plymouth, UK.
- Clesceri, L., Greenberg, A., Eaton, A., 2001. Standard Methods for the Examination of Water and Wastewater. APHA, Washington, DC.
- Dangles, O., 2002. Functional plasticity of benthic macroinvertebrates: implications for trophic dynamics in acid streams. *Can. J. Fish. Aquat. Sci.* 59, 1563–1573.
- Davy-Bowker, J., Murphy, J.F., Rutt, G.P., Steel, J.E., Furse, M.T., 2005. The development and testing of a macroinvertebrate biotic index for detecting the impact of acidity on streams. *Archiv für Hydrobiologie* 163, 383–403.
- Delong, M.D., Brusven, M.A., 1994. Allochthonous inputs of organic matter from different riparian habitats of an agriculturally impacted stream. *Environ. Manage.* 18, 59–71.
- Dobson, M., Hildrew, A.G., Orton, S., Ormerod, S.J., 1995. Increasing litter retention in moorland streams: ecological and management aspects of a field experiment. *Freshw. Biol.* 33, 325–337.
- Dosskey, M.G., Vidon, P., Gurwick, N.P., Allan, C.J., 2010. The role of riparian vegetation in protecting and improving chemical water quality in streams. *J. Am. Water Resour. Assoc.*, 1–18.
- Drake, J.A., 1984. Species aggregation: the influence of detritus in a benthic invertebrate community. *Hydrobiologia* 112, 109–115.
- Edginton, J., A. Hildrew, 1995. Caseless caddis larvae of the British Isles, a key with ecology notes, Freshwater Biological Association.
- Elliott, J.M., Humpesch, U.H., Macan, T.T., 1988. Larvae of the British Ephemeroptera: a key with ecological notes, Freshwater Biological Association.
- Feeley, H.B., Bruen, M., Blacklocke, S., Kelly-Quinn, M., 2013. A regional examination of episodic acidification response to reduced acidic deposition and the influence of plantation forests in Irish headwater streams. *Sci. Total Environ.* 443, 173–183.
- Feeley, H.B., Kelly-Quinn, M., 2014. Re-examining the effects of episodic acidity on macroinvertebrates in small conifer-forested streams in Ireland and empirical evidence for biological recovery. *Biol. Environ.: Proc. Roy. Irish Acad.* 114, 205–218.
- Feeley, H.B., Kerrigan, C., Fanning, P., Hannigan, E., Kelly-Quinn, M., 2011. Longitudinal extent of acidification effects of plantation forest on benthic macroinvertebrate communities in soft water streams: evidence for localised impact and temporal ecological recovery. *Hydrobiologia* 671, 217–226.
- Finnegan, J., Regan, J.T., de Eyto, E., Ryder, E., Tiernan, D., Healy, M.G., 2012. Nutrient dynamics in a peatland forest riparian buffer zone and implications for the establishment of planted saplings. *Ecol. Eng.* 47, 155–164.
- Ghermandi, A., Vandenbergh, V., Benedetti, L., Bauwens, W., Vanrolleghem, P.A., 2009. Model-based assessment of shading effect by riparian vegetation on river water quality. *Ecol. Eng.* 35, 92–104.
- Gregory, S.V., Swanson, F.J., McKee, W.A., Cummins, K.W., 1991. An ecosystem perspective of riparian zones. *Bioscience* 41, 540–551.
- Hagan, J.M., Pealer, S., Whitman, A.A., 2006. Do small headwater streams have a riparian zone defined by plant communities? *Can. J. For. Res.* 36, 2131–2140.
- Haidekker, A., Hering, D., 2007. Relationship between benthic insects (Ephemeroptera, Plecoptera, Coleoptera, Trichoptera) and temperature in small and medium-sized streams in Germany: A multivariate study. *Aquat. Ecol.* 42, 463–481.
- Heino, J., 2005. Functional biodiversity of macroinvertebrate assemblages along major ecological gradients of boreal headwater streams. *Freshw. Biol.* 50 (9), 1578–1587.
- Hynes, H.B.N., 1975. Edgardo Baldi memorial lecture. The stream and its valley. *Verhandlungen der Internationalen Vereinigung für theoretische und angewandte Limnologie* 19, 1–15.
- Hynes, H.B.N., 1977. A key to the adults and nymphs of the British stoneflies (Plecoptera).
- Kelly-Quinn, M., Bruen, M., Harrison, S., Healy, M., Clarke, J., Drinan, T., Feeley, H., Finnegan, J., Graham, C., Regan, J., Blacklocke, S., 2016. Assessment of the impacts of forest operations on the Ecological Quality of Water (HYDROFOR). Environmental protection Agency 47.
- Kelly-Quinn, M., Cruikshanks, R., Johnson, J., Matson, R., Baars, J.R., Bruen, M., 2008. Forestry and Surface-water Acidification (FORWATER), pp. 81–81.

- Kiffney, P.M., Richardson, J.S., Bull, J.P., 2003. Responses of periphyton and insects to experimental manipulation of riparian buffer width along forest streams. *J. Appl. Ecol.* 40, 1060–1076.
- Kowalik, R.A., Ormerod, S.J., 2006. Intensive sampling and transplantation experiments reveal continued effects of episodic acidification on sensitive stream invertebrates. *Freshw. Biol.* 51, 180–191.
- Kreutzweiser, D.P., Capell, S.S., Good, K.P., 2005. Effects of fine sediment inputs from a logging road on stream insect communities: a large-scale experimental approach in a Canadian headwater stream. *Aquat. Ecol.* 39, 55–66.
- Laudon, H., Bishop, K.H., 1999. Quantifying sources of acid neutralisation capacity depression during spring flood episodes in Northern Sweden. *Environ. Pollut.* 105, 427–435.
- Lecerf, A., Baudoin, J.-M., Besson, A.A., Lamothe, S., Lagrue, C., 2012. Is smaller necessarily better? Effects of small-scale forest harvesting on stream ecosystems. *Annales de Limnologie – Int. J. Limnol.* 48, 401–409.
- Lee, K.H., Isenhardt, T.M., Schultz, R.C., 2003. Sediment and nutrient removal in an established multi-species riparian buffer. *J. Soil Water Conserv.* 58, 1–8.
- Leeks, G.J.L., Roberts, G., 1987. The effects of forestry on upland streams – with special reference to water quality and sediment transport.
- Little, D., Collins, K., Cross, J., 2008. Native Riparian Woodlands – A guide to identification, design, establishment and management. Native Woodland Scheme Information Note No. 4, Woodlands of Ireland. Dublin. No. 4.
- Mc Conigley, C., Lally, H., O'Callaghan, M., O'Dea, P., Little, D., Kelly-Quinn, M., 2015. The vegetation communities of unmanaged aquatic buffer zones within conifer plantations in Ireland. *For. Ecol. Manage.* 353, 59–66.
- Mihuc, T.B., 1997. The functional trophic role of lotic primary consumers: generalist versus specialist strategies. *Freshw. Biol.* 37 (2), 455–462.
- Moog, O., 2002. Functional feeding guilds–family/ genus level. Part IV. Fauna Aquatica Austriaca–A Comprehensive Species Inventory of Austrian Aquatic Organisms with Ecological Notes (eds O. Moog). Wasserwirtschaftskataster, Bundesministerium für Land-und Forstwirtschaft, Umwelt und Wasserwirtschaft, Vienna.
- Moog, O., Österreich, W., 1995. Fauna Aquatica Austriaca: A Comprehensive Species Inventory of Austrian Aquatic Organisms With Ecological Notes. Bundesministerium für Land-und Forstwirtschaft, Wasser-Wirtschaftskataster.
- Naiman, R.J., Décamps, H., 1997. The ecology of interfaces: riparian zones. *Annu. Rev. Ecol. Syst.* 28, 621–658.
- Nieminen, M., Ahti, E., Nousiainen, H., 2005. Capacity of riparian buffer zones to reduce sediment concentrations in discharge from peatlands drained for forestry. *Silva Fennica* 39, 331–339.
- Nilsson, A.N., 1996. Aquatic Insects of North Europe: A Taxonomic Handbook. Vol 1: Ephemeroptera, Plecoptera, Heteroptera, Neuroptera, Megaloptera, Coleoptera, Trichoptera, Lepidoptera, Apollo Books.
- Nilsson, A.N., 1997. Aquatic Insects of North Europe: A Taxonomic Handbook. Vol 2: Odonata-Diptera, Apollo Books.
- Ormerod, S., Mawle, G., Edwards, R., 1987. The influence of forest on aquatic fauna. *Environmental Aspects of Plantation Forestry in Wales* 22.
- Ormerod, S.J., Durance, I., 2009. Restoration and recovery from acidification in upland Welsh streams over 25 years. *J. Appl. Ecol.* 46, 164–174.
- Ormerod, S.J., Jones, M.E., Jones, M.C., Phillips, D.R., 2004. The effects of riparian forestry on invertebrate drift and brown trout in upland streams of contrasting acidity. *Hydrol. Earth Syst. Sci.* 8, 578–588.
- Ormerod, S.J., Rundle, S.D., Lloyd, E.C., Douglas, A.A., 1993. The influence of riparian management on the habitat structure and macroinvertebrate communities of upland streams draining plantation forests. *J. Appl. Ecol.* 30, 13–24.
- Petrin, Z., Laudon, H., Malmqvist, B., 2007. Does freshwater macroinvertebrate diversity along a pH-gradient reflect adaptation to low pH? *Freshw. Biol.* 52, 2172–2183.
- Pretty, J.L., Dobson, M., 2004. The response of macroinvertebrates to artificially enhanced detritus levels in plantation streams. *Hydrol. Earth Syst. Sci.* 8, 550–559.
- Richardson, J.S., 1991. Seasonal food limitation of detritivores in a montane stream: an experimental test. *Ecology* 72, 873–887.
- Rutt, G.P., Weatherley, N.S., Ormerod, S.J., 1990. Relationships between the physicochemistry and macroinvertebrates of British upland streams: the development of modelling and indicator systems for predicting fauna and detecting acidity. *Freshw. Biol.* 24, 463–480.
- Ryan, D.K., Yearsley, J.M., Kelly-Quinn, M., 2013. Quantifying the effect of semi-natural riparian cover on stream temperatures: implications for salmonid habitat management. *Fish. Manage. Ecol.* 20, 494–507.
- Service, F., 2000. Forestry and Water Quality Guidelines. Forest Service 49. 833–833.
- Sovell, L.A., Vondracek, B., Frost, J.A., Mumford, K.G., 2000. Impacts of rotational grazing and riparian buffers on physicochemical and biological characteristics of Southeastern Minnesota, USA, streams. *Environ. Manage.* 26, 629–641.
- Speaker, R., Moore, K., Gregory, S., 1984. Analysis of the process of retention of organic matter in stream ecosystems. *Verhandlungen der Internationalen Vereinigung für theoretische und angewandte Limnologie* 22, 1835–1841.
- Sutcliffe, D.W., Hildrew, A.G., 1989. Invertebrate communities in acid streams. *Acid Toxicity Aquatic Animals* 34, 13.
- Sweeney, B.W., 1993. Effects of streamside vegetation on macroinvertebrate communities of White Clay Creek in eastern North America. *Proc. Acad. Nat. Sci. Philadelphia* 144, 291–340.
- Sweeney, B.W., Bott, T.L., Jackson, J.K., Kaplan, L.A., Newbold, J.D., Standley, L.J., Hession, W.C., Horwitz, R.J., 2004. Riparian deforestation, stream narrowing, and loss of stream ecosystem services. *Proc. Natl. Acad. Sci. U.S.A.* 101, 14132–14137.
- Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R., Cushing, C.E., 1980. The river continuum concept. *Can. J. Fish. Aquat. Sci.* 37, 130–137.
- Vought, L.B.M., Dahl, J., Pedersen, C.L., Lacoursiere, J.O., 1994. Nutrient retention in riparian ecotones. *Ambio* 23, 342–348.
- Wallace, I.D., Wallace, B., Philipson, G., 1990. A key to the case-bearing caddis larvae of Britain and Ireland.
- Wallace, J.B., Eggert, S.L., Meyer, J.L., Webster, J.R., 1997. Multiple trophic levels of a forest stream linked to terrestrial litter inputs. *Science* 277, 102–104.
- Wallace, J.B., Eggert, S.L., Meyer, J.L., Webster, J.R., 1999. Effects of resource limitation on a detrital-based ecosystem. *Ecol. Monogr.* 69, 409–442.
- Wigington, P.J., Baker, J.P., DeWalle, D.R., Kretser, P.S., Murdoch, P.S., Simonin, H.A., Van Sickle, J., McDowell, M.K., Peck, D.V., Barchet, W.R., 1996. Episodic acidification of small streams in the northeastern United States: Episodic Response Project. *Ecol. Appl.* 6, 374–388.
- Wiley, M.J., Osborne, L.L., Larimore, R.W., 1990. Longitudinal structure of an agricultural prairie river system and its relationship to current stream ecosystem theory. *Can. J. Fish. Aquat. Sci.* 47, 373–384.