

# The impact of cattle access on ecological water quality in streams: Examples from agricultural catchments within Ireland

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# **Cattle access, an additional pressure on ecological water quality in streams in agricultural catchments**

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## **Abstract**

Surface water quality is potentially impacted by catchment scale and localised pollutant inputs. Unrestricted cattle access to rivers and streams represent a potentially significant localised pressure on freshwater systems. However there is no consensus in the literature on the occurrence and extent of impact and limited research has examined the effects on aquatic biota in the humid temperate environment examined in the present study. Furthermore, for the first time our research has considered the potential for impact in streams that differ in water quality. We investigated the effects of cattle access on macroinvertebrate communities and deposited fine sediment levels, in four high/good and four moderate water quality status rivers draining, low gradient, calcareous grassland catchments in Ireland. We assessed the temporal variability in macroinvertebrates communities across two seasons. Site specific impacts were evident which appeared to be influenced by water status and season. Downstream impacts were found across all four of the high/good water status rivers but in just two of the four moderate water status rivers. These two moderate water status rivers had high or prolonged livestock activity. In view of these findings, the potential for some of these sites to achieve at least high/good water quality status, as set out in the EU Water Framework Directive, may be compromised. The results presented highlight the need for additional research to further define the site specific factors and livestock management practices that increase the risk of impact on aquatic ecology due to these cattle-river interactions.

**Key words:** deposited sediment, impact, Ireland, water status.

## **1. Introduction**

The EU Water Framework Directive (WFD, 2000/60/EC) stipulates the need to identify and mitigate against pressures in freshwater systems. Regulatory authorities are also required to

develop integrated catchment management plans which are supported by robust scientific data. Agricultural grasslands represent one of the most significant land uses, accounting for approximately 68% of the total land farmed globally (Anon, 2009a). In the Republic of Ireland, agriculture accounts for 67% of the total land-use area (DAFM, 2013) compared to a European average of *c.* 40% (Eurostat, 2014). Agricultural land-use intensification has been recognized as one of the principle drivers of ecological degradation in freshwater systems (Malmqvist and Rundle, 2002; Strayer and Dudgeon, 2010) and, as the main land-use in Ireland and parts of Northern Europe, agriculture has been identified as a key contributor of anthropogenic sediment inputs to surface waters (Harrod and Theurer, 2002; Evans *et al.*, 2006). Livestock access to rivers is also considered to pose a threat to water quality (e.g. (Line *et al.*, 2000; Davies-Colley *et al.*, 2004; Evans *et al.*, 2006; Lefrancois *et al.*, 2007) and may be adding to the pollution pressure, particularly where there are multiple access points within a relatively short length of stream.

Research has shown that cattle access can result in a range of impacts on aquatic ecology, geomorphology and water quality as well as the integrity of riparian zones (e.g. (Avery *et al.*, 2004; Bilotta *et al.*, 2007; Bond, 2012). Riparian zones represent an important aquatic-terrestrial interface for regulating instream temperatures through shading, providing allochthonous food inputs to aquatic food webs (Clary and Kinney, 2002), stabilising river channels and controlling bank erosion. They also serve as traps for overland run-off from surrounding areas (Fitch and Adams, 1998). However, high grazing intensities can result in reduced plant cover and vitality in these zones, leading to increased soil erosion and elevated sediment inputs (Trimble and Mendel, 1995).

Although cattle spend less than 2% of their time in streams (Haan *et al.*, 2010; Bond, 2012), they will preferentially defecate in watercourses where they have unrestricted access (Oudshoorn *et al.*, 2008; Bond, 2012). Direct deposition of animal wastes within rivers or indirectly through runoff, including during floods, can pose a threat to human health, because of the presence of harmful pathogens such as *Escherichia.coli* 0157 (*E. coli*), *Salmonella* sp. and *Cryptosporidium* sp. (Nagels *et al.*, 2002; Avery *et al.*, 2004; Davies-Colley *et al.*, 2004; Collins *et al.*, 2007). Deposited sediment, stored nutrients and bacteria can also be re-suspended by cattle trampling on river banks and in riverbeds (Terry *et al.*, 2014).

In addition, cattle encroachment in channels can result in geomorphic impacts commonly known as ‘poaching’ (Bilotta *et al.*, 2007). Poaching can cause soil compaction, reduced soil infiltration rates and increased soil detachment and transport (Mulholland and Fullen, 1991;

Evans *et al.*, 2006; Sharrow, 2007). During periods of high soil moisture content and high stream flow, cattle access can increase risks of bank slope failure (slumping and topples) and the subsequent release of large quantities of sediment to channels (Kauffman and Krueger, 1984; Trimble and Mendel, 1995; Magner *et al.*, 2008). Research also indicates that localised sediment influxes can degrade water quality, alter instream and riparian habitats and reduce biodiversity of macroinvertebrates with potential knock-on consequences for fish populations (Wohl and Carline, 1996; Strand and Merritt, 1999; Harrod and Theurer, 2002; Scrimgeour and Kendall, 2003; Greig *et al.*, 2005).

Although a fair body of research has reported the impact of livestock access on freshwaters the majority relate to large rangelands across North America (e.g. (Belsky *et al.*, 1999; Braccia and Voshell, 2007) and on dairy farms in Australia and New Zealand (e.g. (Amy and Robertson, 2001; Davies-Colley *et al.*, 2004) where stock densities, management practices and hydroclimatic conditions are not generally comparable to those found in northwest Europe. In this paper, we examine the impact of cattle access in the humid temperate environment of Ireland where 80% of the agricultural land area is devoted to pasture (DAFM, 2013). Furthermore, for the first time our research has considered the potential for impact in streams that differ in water quality. This study represents a timely investigation of the potential effects of cattle access on ecological water quality, as regulatory authorities in many countries strive to address pollution pressures from agriculture in a policy environment promoting intensification of agriculture (e.g. Food Harvest 2020 (DAFF, 2010b). It also explores the recognised challenges associated with linking ecological responses to localised disturbances, primarily associated with elevated sediment concentrations. It was hypothesised that cattle access would result in (i) a downstream reduction in macroinvertebrate abundance, richness and sediment-sensitive taxa, (ii) downstream changes in sediment related traits, (iii) rivers at high/good status would show the greatest impact of cattle access, and (iv) that elevated inputs of deposited fine sediment would be observed downstream of cattle access drinking points, compared to upstream sites.

## **2. Material and Methods**

### *2.1 Site selection*

Cattle access points were initially identified using aerial photography and ordnance survey maps. These were subsequently ground-truthed in site visits which resulted in eight rivers

been included in this study (Fig. 1, Table 1). The WFD typologies of the sites were type 31 (calcareous geology with low slope) (Dodkins *et al.*, 2005) and agricultural grassland was the dominant land use upstream of each sampling location (mainly intensive pasture for beef in the north east and a mix of dairying and non-dairying in the midlands region). Site selection was informed by water quality status upstream of each cattle access drinking point as reported online by the Environmental Protection Agency (EPA) in the annual macroinvertebrate Q value river reports (<http://www.epa.ie/QValue/webusers/>). Four of the rivers, the Barrow, Clodiagh, Douglas and Glenlahan Rivers are classified as having ‘high/good’ WFD water quality status (Q4-5, Q4) while four of the rivers, the Boycetown, D’arcy, Dee and Erkina Rivers are classified as having ‘moderate’ WFD water quality status (Q3-4, Q3), (

Fig 1). Where possible instream habitats and riparian shading were matched between the upstream and downstream sampling locations, but the proportion of shading did vary in the Clodiagh and Glenlahan Rivers as shown in Table 1.

## 2.2 Field sampling and laboratory analysis

Sampling was conducted upstream and downstream of each cattle access point in spring (April/May) and autumn (September/October) 2013. The sampling locations were approximately 20 m upstream and downstream of the access points and included the first riffle area in each direction. An additional upstream control site located *c.* 50 m upstream of the cattle access point was sampled on four of the rivers (Clodiagh, Erkina, Glenlahan Rivers and D’arcy’s crossroad stream) in autumn 2013 to account for any natural variability between sampling locations. A habitat survey conducted at each site prior to sampling included visual estimates of percentage riffle, pool, glide, sand/silt cover and riparian shading, together with measurement of mean wetted width, depth, and flow (Table 2).

Patch scale macroinvertebrate sampling, using a 1 mm mesh Surber sampler, was conducted at the same location as the environmental data. Six replicate Surber samples were taken within the mid-channel and margins at each sampling location. Macroinvertebrate samples were preserved in 70% Industrial Methylated Spirits (IMS). In the laboratory, macroinvertebrates were washed through a 0.5 mm sieve and sorted. All taxa were identified to the lowest practicable taxonomic level (species where possible) using Freshwater Biological Association (FBA) identification keys (Hynes, 1977; Macan and Cooper, 1977;

Elliott and Mann, 1979; Elliott *et al.*, 1988; Wallace *et al.*, 1990; Edington and Hildrew, 1995; Nilsson, 1996; Nilsson, 1997).

Prior to macroinvertebrate sampling patch scale visual estimations of % deposited fine sediment (< 2 mm) (Zweig and Rabení, 2001; Rabení *et al.*, 2005; Matthaei *et al.*, 2006; Larsen *et al.*, 2009) were made within the Surber sampler frame. In addition, turbidity and resuspendable sediment from the stream bed were measured adopting a method developed by (Lambert and Walling, 1988). An open-ended cyclinder was pressed into the stream bed within the frame of the Surber sampler and water depth within the cyclinder was measured. Following agitaion of the top 5 cm of the bed substratum for 30 seconds, a manual grab sample of water containing resuspendable sediment was taken. A known volume of water was filtered through pre-weighed 0.45µm Whatman Glass Microfibre GF/C filters, dried at 103 °C for 2 hrs and cooled in a desiccator for 1 hr and reweighed. Re-suspendable solids was calculated as mg/l and converted to g m<sup>-2</sup> using the water volume within the container. Turbidity (NTU) of these grab samples was also measured using a HACH 2100NIS turbidity meter.

On-site measurements of pH, conductivity, temperature and dissolved oxygen were taken at the same time as the biological sampling using WTW automatic field probes. Manual grab water column samples were collected in clean polypropylene bottles that had been triple rinsed with deionised water and river water prior to sampling. These water column samples were analysed using standard methods to determine suspended solids concentrations (SS, mg/l) (APHA, 1995) and turbidity (NTU) using a HACH 2100NIS turbidity meter. Additional water samples were collected upstream and downstream of each cattle access drinking point during base (28<sup>th</sup> July, 2014) and storm-flow (29<sup>th</sup> August and 6<sup>th</sup> October, 2014) conditions and analysed for SS (mg/l) and turbidity (NTU), together with ammonia/ammonium (NH<sub>4</sub><sup>+</sup>: salicylate method using Lachat QuikChem following 0.45µm filtration), total oxidised nitrogen (TON: cadmium reduction method using Lachat QuikChem following 0.45µm filtration), nitrite (NO<sub>2</sub><sup>-</sup>: colorimetric method using Lachat QuikChem following 0.45µm filtration), nitrate (NO<sub>3</sub><sup>-</sup>: calculated as difference between TON and NO<sub>2</sub><sup>-</sup>), total phosphorus (TP: manually using ascorbic acid following digestion) and soluble reactive phosphorus (SRP: ascorbic acid method using Lachat QuikChem following 0.45µm filtration) using standard methods (Clesceri *et al.*, 2001).

Base flow benthic sediment samples were taken (28<sup>th</sup> July, 2014) upstream and downstream of each cattle access point at all sites and placed in labelled zip-locked plastic bags. Samples

were air dried, subsequently sieved (< 2 mm) and analysed for Total Kjeldahl Nitrogen (as ammonia using Lachat QuikChem following acid digestion), TP (absorbance measured using SHIMADZU UV-160A following acid digestion) and organic matter (%) (loss on ignition method) as prescribed in standard methods (Clesceri *et al.*, 2001).

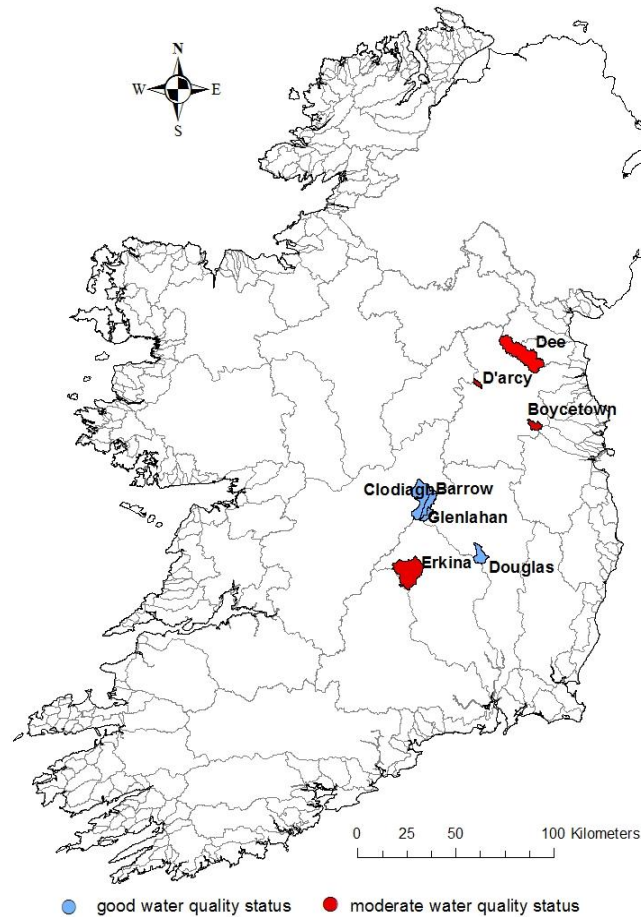


Fig 1 Locations of the eight study river catchments in the midlands and east of Ireland



Table 1 Characteristics of each river sampled including GPS co-ordinates, catchment geology and riparian type, u/s = upstream; d/s = downstream

River	Location	GPS co-ordinates	Catchment geology	Riparian vegetation
Barrow	u/s	53°10'54" -7°25'25"	Limestone/ sandstone	Bramble ( <i>Rubus</i> sp.), gorse scrub ( <i>Ulex europaeus</i> )
	d/s	53°10'53" -7°25'24"		Bramble ( <i>Rubus</i> sp.), nettles ( <i>Urtica</i> sp.)
Glenlahan	u/s	53°8'46" -7°29'47"	Limestone/ sandstone	Ash ( <i>Fraxinus excelsior</i> ), hawthorn ( <i>Crataegus</i> sp.), sycamore ( <i>Acer pseudoplatanus</i> )
	d/s	53°8'49" -7°29'16"		Ash ( <i>Fraxinus excelsior</i> ), bramble ( <i>Rubus</i> sp.), hawthorn ( <i>Crataegus</i> sp.),
Clodiagh	u/s	53°16'9" -7°33'52"	Limestone/ sandstone	Willow ( <i>Salix</i> sp.), Alder ( <i>Alnus</i> spp.),
	d/s	53°16'9" -7°33'57"		Willow ( <i>Salix</i> sp.), Alder ( <i>Alnus</i> spp)
Douglas	u/s	52°55'5" -7°2'8"	Limestone/ sandstone/shale	Alder ( <i>Alnus</i> sp.), ash ( <i>Fraxinus excelsior</i> ), hawthorn ( <i>Crataegus</i> sp.),
	d/s	52°55'5" -7°2'5"		Alder ( <i>Alnus</i> sp.), ash ( <i>Fraxinus excelsior</i> ), hawthorn ( <i>Crataegus</i> sp.),
D'arcy	u/s	53°31'50" -6°42'24"	Limestone/ sandstone	Ash ( <i>Fraxinus excelsior</i> ), bramble ( <i>Rubus</i> sp.)
	d/s	53°31'51" -6°42'25"		Alder ( <i>Alnus</i> sp.), ash ( <i>Fraxinus excelsior</i> ), hawthorn ( <i>Crataegus</i> sp.), bramble ( <i>Rubus</i> sp.)
Boycetown	u/s	53°31'50" -6°42'24"	Limestone/ sandstone/shale	Willow ( <i>Salix</i> sp.), bramble ( <i>Rubus</i> sp.), nettles ( <i>Urtica</i> sp.)
	d/s	53°31'51" -6°42'25"		Willow ( <i>Salix</i> sp.), bramble ( <i>Rubus</i> sp.), nettles ( <i>Urtica</i> sp.)
Dee	u/s	53°50'3" -6°38'35"	Limestone/ sandstone	Alder ( <i>Alnus</i> sp.), hawthorn ( <i>Crataegus</i> sp.)
	d/s	53°50'2" -6°38'35"		Alder ( <i>Alnus</i> sp.)
Erkina	u/s	52°51'6" -7°31'56"	Limestone/ shale	Bramble ( <i>Rubus</i> sp.)
	d/s	52°51'4" -7°31'52"		Bramble ( <i>Rubus</i> sp.), nettles ( <i>Urtica</i> sp.)

Table 2 In-stream site characteristics upstream and downstream of cattle access points on the sampled rivers, u/s = upstream; d/s = downstream

River	Location	Mean wetted width (m)	Mean Depth (m)	% Shading	% Run	% Riffle	% Glide	% Pool	% Sand/Silt	Mean flow (m/s)
Barrow	u/s	9.7	0.18	0	10	90	0	0	15	0.44
	d/s	6.7	0.16	0	60	30	10	0	10	0.29
Glenlahan	u/s	5.2	0.15	75	20	75	5	0	5	0.35
	d/s	4.7	0.12	30	20	30	50	0	3	0.29
Clodiagh	u/s	7.8	0.30	10	0	15	80	5	15	0.3
	d/s	7.4	0.24	30	15	0	75	10	5	0.47
Douglas	u/s	3.5	0.11	80	0	50	40	10	10	0.17
	d/s	5.0	0.12	80	70	15	15	0	20	0.17
D'arcy	u/s	2.3	0.17	0	80	20	0	0	3	0.41
	d/s	3.2	0.12	10	50	50	0	0	75	0.35
Boycetown	u/s	3.2	0.19	10	0	90	10	0	5	0.34
	d/s	3.7	0.22	0	0	90	7	3	10	0.32
Dee	u/s	10.2	0.32	0	0	0	100	0	30	0.21
	d/s	8.4	0.28	0	35	60	0	5	25	0.37
Erkina	u/s	9.7	0.25	0	90	0	10	0	20	0.4
	d/s	7.9	0.35	0	15	10	75	0	5	0.45

### 3. Data Analysis

The Asterics 3.3 programme was used to calculate a range of univariate metrics including taxon richness, % Ephemeroptera, Plecoptera and Trichoptera richness (% EPT richness), taxon abundance, Ephemeroptera (E) abundance, % Ephemeroptera, Plecoptera and Trichoptera abundance (% EPT abundance), Coleoptera (C) abundance and Biological Monitoring Working Party (BMWP) scores. Species traits including functional feeding groups, locomotion and microhabitat preference taxa (% feeding, % locomotion and % microhabitat) were also calculated using this programme.

Univariate metrics were compared across sites using a 4-factor [time (spring and autumn), site (eight rivers), location (upstream and downstream) and habitat (mid-channel and margin)] permutational multivariate analysis of variance (ANOVA/PERMANOVA) in PRIMER 6 (Clarke and Gorley, 2006; Anderson *et al.*, 2008) based on Euclidean distances. This four factor design was also applied to test for differences in community structure and species traits using PERMANOVA in PRIMER 6 based on Bray-Curtis distances for community structure and Zero-adjusted Bray-Curtis distances for species traits. The data were initially tested for homogeneity of variance using Permdisp, and transformed as appropriate so as to meet the requirements of homogeneity. Post-hoc analysis was conducted using pair-wise tests when significant differences were detected. As multiple tests were undertaken a more stringent  $p$  value of  $< 0.01$  was used in order to avoid Type 1 errors (i.e. an increased probability of rejecting the null hypothesis). Differences in community structure were visualised on non-metric multidimensional scaling (nMDS) plots using Bray-Curtis similarities of untransformed data. SIMPER (Similarity of Percentages) analysis was conducted to indicate the taxa responsible for any detected differences in community structure (Clarke, 1993).

Deposited sediment levels, measured as re-suspendable sediment, turbidity and % surface cover, were compared across sites with the 4-factor design (as above) using PERMANOVA analysis based on Euclidean distances of  $\log(x+1)$  transformed data in PRIMER 6. A 2-factor design [(location (upstream and downstream) and flow (base and high))] was used to assess differences in turbidity, suspended solids concentration (SS), ammonia, soluble reactive phosphorus (SRP), total phosphorus (TP), nitrite and nitrate during base and storm-flow conditions across the study rivers using PERMANOVA analysis based on Euclidean distances of square root transformed data in PRIMER 6.

## 4. Results

### 4.1 Macroinvertebrates

The two upstream control sites showed no significant differences in any of the metrics apart from mid-channel community structure ( $t=1.93$ ,  $P<0.01$ ) results in the Clodiagh and % functional feeding groups ( $t=2.21$ ,  $P<0.01$ ) in the margin samples taken from the D'arcys.

In terms of the comparisons between sites upstream and downstream of the cattle access points there were significant interactions between site and location for taxon richness ( $F_{7,320}=13.26$ ,  $P<0.001$ ) and between time, site and location for % EPT richness ( $F_{7,320}=2.88$ ,  $P<0.01$ ), taxon abundance ( $F_{7,320}=2.99$ ,  $P<0.001$ ), % EPT abundance ( $F_{7,320}=3.08$ ,  $P<0.01$ ), % E abundance ( $F_{7,320}=4.78$ ,  $P<0.001$ ) and % C abundance ( $F_{7,320}=3.34$ ,  $P<0.01$ ) (Table 3).

Pairwise comparisons indicated taxon richness decreased downstream of cattle access in the Barrow ( $t=5.66$ ,  $P<0.001$ ) and Clodiagh ( $t=3.8$ ,  $P<0.001$ ) while taxon richness increased at the downstream sites in the Douglas ( $t=3.56$ ,  $P<0.001$ ), Dee ( $t=4.06$ ,  $P<0.001$ ) and Glenlahan sites ( $t=3.60$ ,  $P<0.001$ ) (Fig. 2a). Similarly, taxon abundance showed variable results across sites with downstream reductions in taxon abundance for the Barrow in both seasons (spring:  $t=3.31$  and autumn:  $t=4.75$ ,  $P<0.001$ ) while significant downstream increases were evident for autumn samples in the Douglas ( $t=4.41$ ,  $P<0.001$ ) and Glenlahan ( $t=4.20$ ,  $P<0.001$ ) and spring samples in the Dee sites ( $t=4.38$ ,  $P<0.001$ ) (Fig. 2b).

Spring samples in the Clodiagh ( $t=3.29$ ,  $P<0.001$ ) and Glenlahan ( $t=2.70$ ,  $P<0.01$ ) and autumn samples in the Barrow ( $t=4.87$ ,  $P<0.001$ ) showed significant reductions in %EPT richness downstream of the cattle access points (Fig. 2c). In terms of % EPT abundance, significant downstream reductions were evident for the Boycetown ( $t=3.10$ ,  $P<0.01$ ) and Glenlahan ( $t=3.59$ ,  $P<0.01$ ) spring samples together with the autumn Barrow ( $t=3.23$ ,  $P<0.01$ ). In contrast, the Dee showed significant downstream increases in %EPT richness ( $t=4.61$ ,  $P<0.001$ ; Fig. 2b) and %EPT abundances ( $t=4.39$ ,  $P<0.001$ ; Fig. 2d) in autumn.

Ephemeroptera (E) abundance decreased downstream of cattle access point in the Clodiagh ( $t=3.98$ ,  $P<0.001$ ) and Boycetown ( $t=3.14$ ,  $P<0.01$ ) in spring and in the Barrow ( $t=6.53$ ,  $P<0.001$ ) in autumn. In contrast, significant downstream increases were evident in the Dee ( $t=3.18$ ,  $P<0.01$ ) in spring (Fig. 2e). Coleoptera (C) abundance decreased downstream of cattle access in spring and autumn in the Barrow ( $t=3.51$  and  $t=2.80$ ,  $P<0.01$  respectively)

and in spring for the Boycetown ( $t=3.42$ ,  $P<0.01$ ). The autumn downstream Glenlahan site showed a significant increase ( $t=6.29$ ,  $P<0.001$ ) in C abundance (Fig. 2f).

Table 3 Summary of the significant differences found in univariate and multivariate mid-channel metrics detected downstream of cattle access. +/- indicates direction of change in univariate metrics at downstream location.

	Taxon Richness	% EPT richness	Taxon abundance	% EPT abundance	E abundance	Community structure	% Microhabitat	% Feeding	% Locomotion
<i>Interaction term</i>	<i>Site location</i>	<i>Time x site x location</i>	<i>Time x site x location</i>	<i>Time x site x location</i>	<i>Time x site x location</i>	<i>Time x site x location x habitat</i>	<i>Time x site x location x habitat</i>	<i>Time x site x location x habitat</i>	<i>Time x site x location</i>
Barrow	-	Autumn -	Spring - Autumn -	Autumn -	Autumn -	Autumn	ns	ns	ns
Douglas	+	ns	Autumn +	ns	ns	ns	ns	Autumn	ns
Clodiagh	-	Spring -	ns	ns	Spring -	Autumn	Autumn	Autumn	Spring
Glenlahan	+	Spring -	Autumn +	Spring -	ns	Spring Autumn	ns	Spring Autumn	Spring
D'arcys	ns	ns	ns	ns	ns	ns	ns	ns	ns
Boycetown	ns	ns	ns	Spring -	Spring -	Spring Autumn	ns	Spring	Spring
Dee	+	Autumn +	Spring +	Autumn +	Spring +	Spring Autumn	Spring	Spring Autumn	Spring
Erkina	ns	ns	ns	ns	ns	ns	ns	ns	Spring

ns: not significant

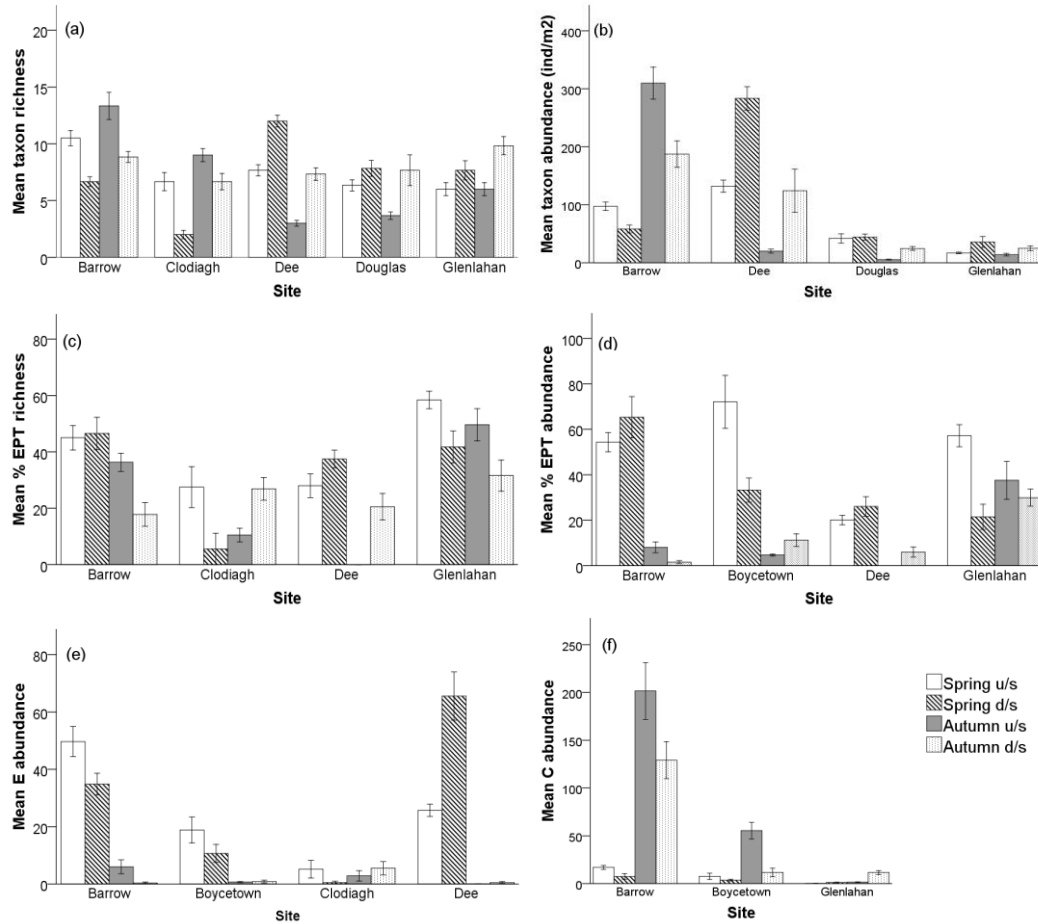


Fig. 2 Mean (+/- standard error) for a) taxon richness, b) taxon abundance, c) %EPT richness, d) %EPT abundance, e) Ephemeroptera (E) abundance and f) Coleoptera (C) abundance for mid-channel spring and autumn sampling at sites where significant differences were found, u/s = upstream, d/s = downstream.

The results of the species trait analysis highlighted significant interactions between time, site, location and habitat for % microhabitat groups ( $F_{7,320}=1.88$ ,  $P<0.01$ ) and % functional feeding groups ( $F_{7,320}=2.72$ ,  $P<0.001$ ) while a significant interaction between time, site and location was evident for % locomotion groups ( $F_{7,320}=3.85$ ,  $P<0.001$ ) (Table 3). The % microhabitat preference results differed significantly between the upstream and downstream mid-channel locations for the spring Dee ( $t=8.01$ ,  $P<0.01$ ; Fig 3a) and autumn Clodiagh sites ( $t=2.70$ ,  $P<0.01$ ; Fig 3b). SIMPER analysis indicated downstream increases in taxa associated with all microhabitat groups, with the exception of % CPOM and % algae/moss groups, for the Dee spring sites while downstream increases in taxa associated with all microhabitat groups were evident for the Clodiagh (autumn) sites (Table 4).

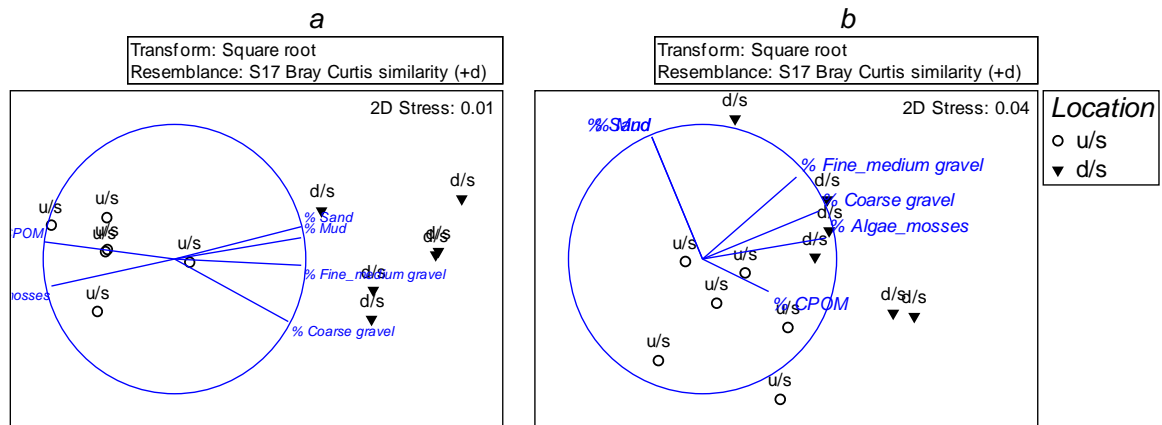


Fig 3 nMDS plot, with vector graph overlaid for a) spring Dee and b) autumn Clodiagh mid-channel microhabitat groups. us = upstream; ds = downstream

The % functional feeding groups differed significantly between the upstream and downstream locations for spring mid-channels locations in the Boycetown ( $t=3.32$ ), Dee ( $t=3.60$ ) and Glenlahan ( $t=2.46$ ) rivers while differences in the autumn mid-channel sampling locations were evident for the Douglas ( $t=1.66$ ), Clodiagh ( $t=3.33$ ), Dee ( $t=2.32$ ) and Glenlahan ( $t=2.96$ ) at  $P<0.01$  (Table 4). This study hypothesised that shredders would be favoured over filter feeders and grazers/scapers downstream of cattle access points. SIMPER analysis partially supported this hypothesis in relation to shredders in the Boycetown (spring) and the Clodiagh and Dee (autumn) sites (Table 4). In contrast, downstream decreases in shredders were evident in the Dee (spring) and Douglas and Glenlahan autumn samples (Table 4). We had expected a downstream reduction in filter feeders and while our findings partially supported this hypothesis, results were variable across rivers (Table 4). Contrary to our expectations with regard to grazers/scapers, where significant downstream differences were found, most sites recorded downstream increases (autumn samples in the Clodiagh, Douglas, Glenlahan together with both spring and autumn samples in Dee) although two spring sites (Glenlahan and Boycetown) did show downstream decreases in grazers/scapers (Table 4).

Significant differences in % locomotion groups were evident between spring upstream and downstream mid-channel locations in the Clodiagh ( $t=2.75$ ), Glenlahan ( $t=2.04$ ), Boycetown ( $t=2.27$ ) and Erkina ( $t=2.50$ ) at  $P<0.01$  and Dee ( $t=3.26$ ) at  $P<0.001$ . In contrast, no significant differences were detected for autumn mid-channel samples. SIMPER analysis indicated downstream decreases in sprawling/walking and swimming/diving taxa in the spring samples from the Clodiagh, Dee and Erkina, but this trend was reversed in the Glenlahan sites with downstream increases in both sprawling/walking and swimming/diving



taxa. Analysis of the spring data from the Boycetown indicated a downstream decrease in sprawling/walking while swimming/diving taxa showed a slight downstream increase. No significant differences were detected for autumn mid-channel samples (Table 4).

Table 4 Summary of the significant differences detected in species traits (microhabitat preference, functional feeding and locomotion groups) downstream of cattle access points. +/- indicates the direction of change in groups at downstream locations

	Clodiagh (spring)	Glenlahan (spring)	Boycetown (spring)	Dee (spring)	Erkina (spring)	Clodiagh (autumn)	Douglas (autumn)	Glenlahan (autumn)	Boycetown (autumn)	Dee (autumn)
<i>Microhabitat preferences</i>										
% Sand	ns	ns	ns	+	ns	+	ns	ns	ns	ns
% Fine/medium gravel	ns	ns	ns	+	ns	+	ns	ns	ns	ns
% Coarse gravel	ns	ns	ns	+	ns	+	ns	ns	ns	ns
% Mud	ns	ns	ns	+	ns	+	ns	ns	ns	ns
% Algae/mosses	ns	ns	ns	-	ns	+	ns	ns	ns	ns
% CPOM	ns	ns	ns	-	ns	+	ns	ns	ns	ns
<i>Feeding groups</i>										
% Shredders	ns	na	+	-	ns	+	-	-	ns	+
% Passive filter feeders	ns	na	-	+	ns	+	na	-	ns	na
% Active filter feeders	ns	+	na	na	ns	-	+	na	ns	-
% Grazers and scrapers	ns	-	-	+	ns	+	+	+	ns	+
% Gatherers/collectors	ns	-	+	+	ns	+	-	+	ns	+
<i>Locomotion groups</i>										
% sprawling/walking	-	+	-	-	-	ns	ns	ns	ns	ns
% swimming/diving	-	+	+	-	-	ns	ns	ns	ns	ns

ns not significant; na not applicable

In terms of community structure a significant interaction between time, site, location and habitat was evident ( $F_{7,320}=1.49$ ,  $P<0.01$ ). Pairwise comparisons indicated a significant difference between sampling locations for spring mid-channels in the Boycetown ( $t=2.20$ ,  $P<0.01$ ), Dee ( $t=3.80$ ,  $P<0.01$ ) and Glenlahan ( $t=2.10$ ,  $P<0.01$ ) while autumn mid-channel samples indicated differences in the Barrow ( $t=2.25$ ,  $P<0.01$ ), Clodiagh ( $t=2.31$ ,  $P<0.01$ ), Boycetown ( $t=2.12$ ,  $P<0.01$ ), Dee ( $t=3.40$ ,  $P<0.01$ ) and Glenlahan ( $t=1.38$ ,  $P<0.01$ ).

In autumn, *Elmis aenea* (Müller), a coleopteran, was the main driver of the differences between locations in a number of rivers. *Elmis aenea* was more abundant upstream in the Barrow contributing 14.5% to the differences between locations, while this taxon was more abundant downstream in the Glenlahan and Dee, contributing 15.5% and 30%, respectively to the differences (

Table 5). Another coleopteran species, *Limnius volckmari* (Panzer) made the largest contribution (24.7%) to differences between locations for the Boycetown autumn locations, being more abundant upstream (

Table 5). The upstream dominance of Chironomidae in the autumn mid-channel samples from the Clodiagh contributed 24.3% to the dissimilarity between locations (**Error! Reference source not found.**).

The spring results were more variable, with higher downstream abundances of a number of taxa driving the differences between locations in the Glenlahan (Culicidae), Boycetown (*Gammarus dubeni* (Lilljeborg)) and Dee (Oligochaeta) contributing 14.3, 18.6 and 25%, respectively to observed differences (**Error! Reference source not found.**).

SIMPER analysis also revealed some seasonal differences in rivers. For example, the upstream spring Dee sampling location was associated with higher abundances of the shredder *Gammarus dubeni* and Chironomidae while the downstream location was associated with higher abundances of Oligochaeta, *Seratella ignita* (Poda) and *Elmis aenea*.(Fig 4a). In contrast, the Dee autumn mid channel upstream sampling locations were associated with the grazer/predator Chironomidae while the collector/gatherer *Elmis aenea*, the shredder

*Gammarus duebeni*, Oligochaeta and Hydropsyche sp. predominated at the downstream sampling location (Fig 4b). No significant differences in community structure or composition were detected in any of the study sites from the margin samples in either season.

Table 5: SIMPER analysis identifying the main taxa, where significant differences were found that made the largest contributions in mid-channel community structure upstream and downstream of cattle access points. The mean counts presented are the average of the square root transformed abundances.

River/ season	Taxa	Average abundance u/s	Average abundance d/s	% Contribution
Glenlahan spring	<i>Culicidae</i>	0.00	1.96	14.25
	Chironomidae	2.38	3.98	13.38
	Baetis spp.	1.94	0.74	9.36
	<i>Rhithrogena semicolorata</i>	1.72	0.76	8.86
Boycetown spring	<i>Gammarus duebeni</i>	1.05	3.97	18.58
	Oligochaeta	1.65	2.48	14.80
	Hydropsyche spp.	2.43	0.81	10.39
	Baetis spp.	3.86	2.87	9.34
	<i>Rhithrogena semicolorata</i>	1.49	0.74	6.89
Dee spring	<i>Limnius volckmari</i>	1.97	1.59	5.74
	Oligochaeta	1.34	9.53	25.19
	<i>Elmis aenea</i>	1.39	4.54	9.43
	<i>Gammarus duebeni</i>	9.26	6.65	8.61
	Chironomidae	2.96	2.85	8.54
	<i>Serretella ignita</i>	4.70	7.33	8.10
Barrow autumn	Baetis spp.	1.13	2.58	5.95
	<i>Elmis aenea</i>	10.93	8.39	14.47
	Chironomidae	5.53	2.57	12.27
	<i>Leuctra</i> spp.	2.84	0.57	9.70
	Oligochaeta	4.86	4.06	8.09
Clodiagh autumn	Baetis spp.	1.80	0.24	7.03
	<i>Esolus parralelepipedus</i>	3.42	2.37	6.39
	Chironomidae	4.76	0.24	24.26
	<i>Helius Longirostis</i>	2.09	0.00	10.64
	<i>Dixidae pupae</i>	1.84	0.00	9.46
	Baetis spp.	0.97	1.88	9.25
Glenlahan autumn	Oligochaeta	1.70	1.86	8.04
	<i>Gammarus duebeni</i>	1.62	1.93	5.84
	<i>Elmis aenea</i>	0.57	2.59	15.44
	<i>Esolus parralelepipedus</i>	0.00	1.18	8.98
	Chironomidae	2.13	1.42	7.63
	<i>Gammarus duebeni</i>	0.83	0.64	6.95
Boycetown autumn	<i>Limnius volckmari</i>	0.64	1.34	6.69
	Hydropsyche spp.	1.19	1.02	6.41
	<i>Limnius volckmari</i>	5.68	1.75	24.70
Boycetown autumn	<i>Elmis aenea</i>	4.56	2.25	15.93
	Oligochaeta	1.77	2.47	11.58

	<i>Gammarus duebeni</i>	5.16	3.69	9.44
	Hydropsyche spp.	0.86	0.29	5.15
	<i>Dicranota robusta</i>	1.09	0.71	4.81
	<i>Elmis aenea</i>	0.00	6.85	30.00
Dee autumn	<i>Gammarus duebeni</i>	1.83	4.60	13.97
	<i>Limnius volckmari</i>	0.00	2.48	11.58
	Oligochaeta	1.03	2.11	9.77
	Hydropsyche spp.	0.00	2.05	9.45
	Chironomidae	3.56	2.08	8.99

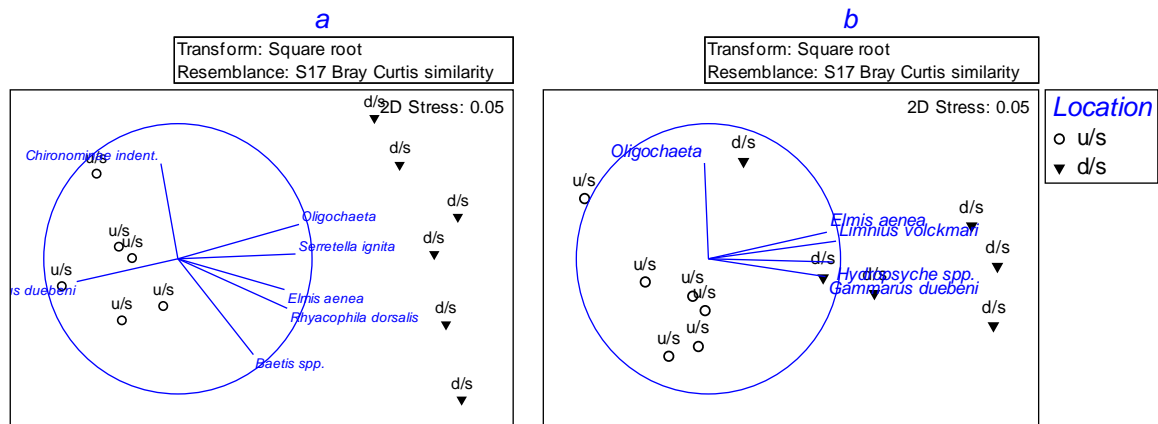


Fig 4 nMDS plot, with a vector graph overlaid, for community structure a) spring and b) autumn mid-channel locations in the river Dee.

#### 4.2 Sediment cover and chemistry

No upstream-downstream differences were detected for % surface cover while significant interaction between time, site and location was detected for re-suspendable sediment ( $F_{7,320}=2.88$ ) and turbidity ( $F_{7,320}=3.18$ ) at  $P<0.01$ . Pairwise comparisons revealed significantly higher downstream re-suspendable sediment in the Barrow ( $t=4.31$ ,  $P<0.001$ ) and Erkina ( $t=4.38$ ,  $P<0.001$ ) together with Glenlahan autumn samples ( $t=4.10$ ,  $P<0.001$ ). Significantly higher downstream turbidity was also evident in the Barrow ( $t=5.01$ ,  $P<0.001$ ), Erkina ( $t=2.72$ ,  $P<0.01$ ) and Glenlahan spring samples ( $t=2.79$ ,  $P<0.01$ ) together with autumn Boycetown samples ( $t=3.72$ ,  $P<0.01$ ).

Although the differences in sediment TN, TP or % organic matter between the upstream and downstream sites were not statistically significant, there was a general downstream increase in these across all sites (Fig 5) with the exception of slight downstream reductions in TP in the Dee and TN in the Douglas and Clodiagh rivers (Table 6).

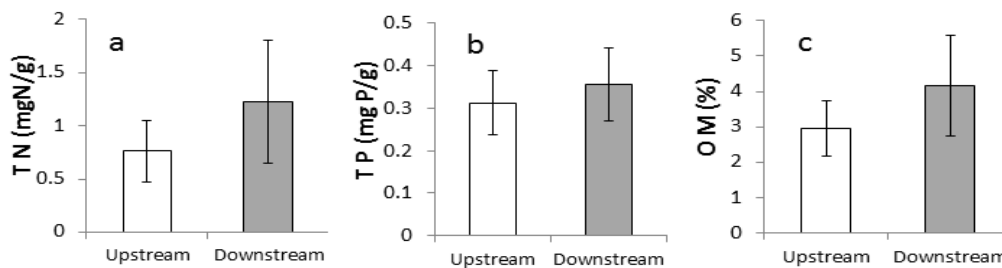


Fig 5 Mean (+/- standard error) benthic sediment chemistry for all rivers sampled a) total Kjeldahl nitrogen, b) total phosphorous and c) % organic matter.

Table 6 Sediment chemistry for all eight rivers

River/site	TN (mgN/g)		TP (mgP/g)		Organic matter (%)	
	upstream	downstream	upstream	downstream	upstream	downstream
Barrow	0.26	0.31	0.08	0.10	1.00	1.23
Douglas	0.74	0.65	0.28	0.39	4.10	4.10
Clodiagh	0.19	0.16	0.08	0.10	1.00	1.20
Glenlahan	0.09	0.12	0.07	0.11	0.90	1.05
D'arcys	2.78	5.23	0.64	0.79	8.00	13.57
Boycetown	0.36	0.95	0.32	0.54	2.10	3.40
Dee	0.53	0.73	0.52	0.37	2.40	2.70
Erkina	0.55	2.23	0.29	0.66	2.90	7.27

#### 4.3 Base and high flow water chemistry

No significant upstream-downstream effects were found during base or high flows for any of the measured determinants. However, significant differences were found in turbidity ( $F_{2,42}=13.11$ ), suspended solids ( $F_{2,42}=14.41$ ) and ammonia ( $F_{2,42}=9.16$ ) at  $P<0.001$  between base-and high-flows but not between the two high flows or between the upstream and downstream sites.

### 5. Discussion

Freshwater systems and associated ecological processes are closely connected to their drainage basins and therefore are under pressure from a range of land-use activities (Richardson *et al.*, 2009). Agricultural land-use intensification has been recognized as one of the main drivers of ecological degradation in freshwater systems (Malmqvist and Rundle,

2002; Strayer and Dudgeon, 2010) and a key contributor of anthropogenic sediment into surface waters (Harrod and Theurer, 2002; Evans *et al.*, 2006).

Several studies have linked cattle access, and poaching of riverbanks, to a range of direct effects as outlined previously. Fewer studies have examined the more subtle, indirect, structural changes that may result from in-stream cattle activities, particularly on benthic macroinvertebrate communities. Rivers are naturally heterogeneous systems and any alteration in habitat conditions will induce changes in macroinvertebrate community structure and composition, at different spatial scales. Substrate, flow and food resources have been shown to be particularly important in structuring macroinvertebrate assemblages at patch scale (Rabeni and Minshall, 1977). The majority of macroinvertebrates have specific substrate requirements and favour patches which met those conditions while avoiding unfavourable patches (Culp *et al.*, 1983; Sarriquet *et al.*, 2007). Cattle access reduces vegetation cover and results in poached, unstable river banks which can contribute significant amounts of inorganic sediment to rivers (Evans *et al.*, 2006) with a large variety of effects on taxa depending on their specific habitat requirements (Braccia and Voshell, 2006). Increased deposited sediment levels cause the infilling of interstitial spaces and embeddedness, as the average size of sediment particles diminishes (Kaufmann *et al.*, 2009), reducing the availability of refugia for a range of taxa e.g. clingers and crawlers (Lancaster and Hildrew, 1993). These taxa maintain their position on clean substrates using a range of behavioural and morphological adaptations (e.g. claws, silks and suction). However, increased deposited fine sediment interferes with these apparatus (Braccia and Voshell, 2006). Furthermore, elevated sand and silt causes bed instability (Kaufmann *et al.*, 2009) that can have a negative impact on crawlers although burrowing taxa may be favoured as they have a preference for habitats with fine sediment (Wood and Armitage, 1997).

Our results suggest that for the most part, macroinvertebrates in mid-channel habitats rather than channel margins were most sensitive to the added pressure of cattle access points. Under normal flow conditions channel margin habitats generally have higher rates of sedimentation, due to lower flow velocity and reduced resuspension (Tipping *et al.*, 1993). As most taxa have specific habitat requirements, those taxa inhabiting channel margins may be more sediment tolerant than those found in mid-channels, so any additional pressure on macroinvertebrate community structure from cattle access points *per se* may be difficult to detect in stream margins.

There was clear evidence of some seasonal differences and site specific impacts on macroinvertebrate communities across both high/good and moderate status rivers. Overall downstream changes in high/good status rivers (Barrow, Clodiagh, Douglas and Glenlahan) were more frequently observed in autumn than spring samples. Autumn samples from these rivers revealed changes in community structure and functional feeding groups in three of the four high/good status rivers. Significant changes in at least two univariate metrics (total richness and EPT richness together with taxon, E and EPT abundance) were observed in both seasons for all four high/good status rivers. This seems to indicate more pronounced effects of cattle access during the summer months. Cattle are primarily grass fed and tend to concentrate around water sources (Robertson, 1997; James *et al.*, 1999) during these months in Ireland. This livestock activity coincides with periods of low flow during which sediment may accumulate on river beds clogging interstitial spaces (Wood and Petts, 1999; Braccia and Voshell, 2007) potentially reducing habitat heterogeneity. (Lefrancois *et al.*, 2007) investigated the effects of cattle access on suspended sediment concentrations (SSC) in two small agricultural streams and found that elevated levels in early autumn coincided with periods where cattle access and low flows were prevalent in the catchments. In this study SSC decreased over winter months, although flows were high, only to increase once more in spring when cattle were reintroduced to the catchments.

It was hypothesised that moderate status rivers would be less susceptible to impacts from cattle access points compared to high/good status rivers, as cattle access points were unlikely to cause any further detectable deterioration in moderate status rivers over and above impacts from other stressors e.g. nutrients (Madden *et al.*, 2011). Results generally support this hypothesis with significant downstream impacts detected in only two of the four moderate status rivers; the Boycetown and Dee rivers. The other two rivers showed either no response (D'arcy) or a change in one metric at most (Erkina). The two rivers that showed responses had the highest usage of the access points. The access point on the river Dee is used on a regular basis as a crossing point for a large dairy herd and results showed downstream mid-channel impact in both seasons. These changes were mainly driven, in spring, by downstream increases in the burrowing, collector/gatherers oligochaetes (+25.2%) and the collectors/gatherers, detritus feeding *Elmis aenea* (+9.4%) while autumn samples showed downstream increases in *Elmis aenea* (+30%) and the shredder *Gammarus duebeni* (+13%). This supports our hypothesis that detritus and burrowing taxa would be favoured over other taxa. These changes may be related to increased nutrient levels, as a result of cattle defecating



close to or within the river system. The observed downstream increases in *Gammarus duebeni* (+18.6%) and the burrowing taxa, *Oligochaetes* (+14.8%) in the Boycetown stream in spring further support this hypothesis. These findings may be partly explained by the overwintering of cattle on a steeply sloped field bordering this river. Heavy poaching of this field resulted in the complete removal of vegetation cover, exposing a very large area of bare soil and increasing its vulnerability to erosion during high flow events. It was observed during the field visits that livestock movement in the immediate vicinity of the access point produced a downstream plume of sediment-laden, turbid water. This suggests that outdoor overwintering of cattle in the vicinity of watercourses is not best practice in terms of water quality protection. A study of the River Bush in Northern Ireland apportioned 60% of the annual bedload and 2% of the annual suspended sediment load to livestock poaching and bank erosion during high flows (Evans *et al.*, 2006).

There is no consensus in the available literature on the effects of cattle access on water quality. A study by (Madden *et al.*, 2011) on a number of small streams in the south-east of Ireland indicated no significant effect on water chemistry and biotic indices commonly used in Ireland (Q-value and Small Stream Risk Score (SSRS)). These streams had moderate water quality, which potentially masked any further deterioration in water quality due to cattle access. Similarly other studies found no significant differences in a range of macroinvertebrate metrics (e.g. taxon richness, EPT abundance, % E, % Chironomidae, % scrapers) between grazed and ungrazed reaches (e.g. (Wohl and Carline 1996; (Nerbonne and Vondracek, 2001; Ranganath *et al.*, 2009). In contrast, studies reviewing the benefits of cattle exclusion measures, including streamside fencing, observed higher EPT scores (Galeone *et al.*, 2006) and increased macroinvertebrate abundances, although diversity remained unchanged, following cattle exclusion (Carline and Walsh, 2007). Rapid improvements in water clarity and channel stability coupled with variable responses in nutrient and faecal contamination following cattle exclusion were reported in New Zealand, however significant changes in macroinvertebrate metrics were not apparent (Parkyn *et al.*, 2003).

In terms of sediment the present study detected no significant upstream/downstream differences in % visual estimations of deposited sediment cover although a number of rivers revealed significant differences in turbidity and deposited sediment (measured as re-suspended sediments). Ranganath *et al.* (2009) attributed the lack of differences between reaches in their study to upstream sediment sources overwhelming the impact of localised sediment inputs from cattle access. Furthermore, sediment, in combination with nutrients are

two of the key in-stream stressors arising from cattle access (Muenz *et al.*, 2006; Vidon *et al.*, 2008) which may impact stream integrity (Townsend *et al.*, 2008). Non-linear relationships and complex interactions between these stressors, present challenges in unravelling the individual and combined effects of such multiple pressures in rivers (Lemly, 1982; Townsend *et al.*, 2008; Matthaei *et al.*, 2010). In the present study, nutrient levels and organic matter were generally higher downstream of cattle access across all rivers. In rivers that already have moderate water status at the upstream sampling location, due in part to the presence of excessive nutrients, it may be more challenging to detect any further changes in community structure and composition due to cattle access or to disentangle cattle access effects from upstream stressors. The presence of these two stressors may be a contributing factor in this study and explain some of the seasonal and site-specific responses found.

It is also likely that the magnitude of the risk posed by cattle access, and thus the potential for detectable impacts, depends on a number of biotic and abiotic factors including landform features (e.g. soil type, texture and structure, geology and topography), biophysical factors (e.g. flow velocity, riparian and field vegetation cover), climate (precipitation rates and storm frequency) and agricultural management practices (Clark, 1998; Kurz *et al.*, 2006; Bilotta *et al.*, 2007). Stocking density, species and age of animals are also known to be some of the key factors in determining cattle access impacts and grazing effects (Bilotta *et al.*, 2007). Livestock management practices must be also taken into account when reviewing the risk from cattle access. Clearly as this study has highlighted the combination of factors that increase the risk of impact remain to be quantified by further research.

## **6. Conclusion and management implications**

In this study of site-specific impacts on macroinvertebrate communities, effects were more evident for high/good status rivers, although two moderate status rivers showed impacts where livestock activity was heavy (e.g. at river crossing points) or continuous (overwintering of cattle). Cattle access may limit the future potential of some sites to achieve at least good status, as required by the EU Water Framework Directive. The implications of the new GLAS agri-environmental scheme (DAFM, 2015), part of the Rural Development Programme 2014-2020, in Ireland should go somewhere towards mitigating against potential impacts of cattle access points in high status rivers, however this may be dependent on the farmer's participate rates in this voluntary scheme. However, an opportunity may have been missed here with regard to moderate status rivers which are not deemed to be a priority under this scheme. Furthermore, meeting ambitious agriculture growth targets (e.g. Food harvest

2020), will put additional pressures on freshwater systems and may cause further deterioration in water quality or compromise efforts to reach water quality targets. Effective catchment management plans need to take an integrated approach across the entire catchment taking account of not only diffuse pollution sources but also point sources, such as cattle access points. The results presented highlight the need for additional research to further define the site specific factors and livestock management practices that increase the risk of impact on aquatic ecology due to these cattle-river interactions.

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