Mitigation of impacts of cattle access on stream ecosystems – efficacy of fencing

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Abstract

Headwater streams can constitute up to 80% of river channel length and are vulnerable to anthropogenic pressures due to their high connectivity to adjacent land, large relative catchment size and low dilution capacity. In these environments unrestricted cattle access is a potential significant cause of water quality deterioration, resulting from increases in stream bank erosion, riparian damage and sediment deposition among others. Several studies have reported improvements in physico-chemical and hydromorphological conditions of streams following elimination of cattle access; few, however, have focussed on the ecological impacts of such management practices. Here, such impacts are assessed. We look at the short-term effects by comparing habitat condition, sediment deposition, and instream macroinvertebrate communities upstream and downstream of cattle access points prior to, and one year following exclusion via fencing. The long-term effects are also measured by reassessing a small stream catchment entirely fenced off from cattle access in 2008 under a concerted management effort. In the short term, cattle exclusion led to reduction in deposited sediment downstream of cattle access points and a related homogenisation of macroinvertebrate community structure between upstream and downstream sampling points. Increased abundances of specific indicator taxa (*Ancylus fluviatilis*, Glossosomatidae and Elmidae) in the fenced catchment following 9 years of exclusion highlight the long-term ecological benefits of such mitigation practices. These findings highlight the importance of incentivised agri-environment schemes in reducing the negative impacts of cattle access to these vulnerable ecosystems.

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ABSTRACT

Headwater streams can constitute up to 80% of river channel length and are vulnerable to anthropogenic pressures due to their high connectivity to adjacent land, large relative catchment size and low dilution capacity. In these environments unrestricted cattle access is a potential significant cause of water quality deterioration, resulting from increases in stream bank erosion, riparian damage and sediment deposition among others. Several studies have reported improvements in physico-chemical and hydromorphological conditions of streams following elimination of cattle access; few, however, have focussed on the ecological impacts of such management practices. Here, such impacts are assessed. We look at the short-term effects by comparing habitat condition, sediment deposition, and instream macroinvertebrate communities upstream and downstream of cattle access points prior to, and one year following exclusion via fencing. The long-term effects are also measured by reassessing a small stream catchment entirely fenced off from cattle access in 2008 under a concerted management effort. In the short term, cattle exclusion led to reduction in deposited sediment downstream of cattle access points and a related homogenisation of macroinvertebrate community structure between upstream and downstream sampling points. Increased abundances of specific indicator taxa (*Ancylus fluviatilis* , Glossosomatidae and Elmidae) in the fenced catchment following 9 years of exclusion highlight the long-term ecological benefits of such mitigation practices. These findings highlight the importance of incentivised agri-environment schemes in reducing the negative impacts of cattle access to these vulnerable ecosystems.

Key Words:

Cattle Access; Freshwater; Fencing; Macroinvertebrates

Introduction

Agriculture, particularly in intensively managed systems, can pose a risk to water quality via both diffuse and point sources. Unrestricted cattle access to streams and rivers is among these sources and can have potentially negative impacts on stream water quality as has been demonstrated both in the temperate European setting (Conroy et al., 2016a; O'Sullivan et al., 2019a & b) and internationally (Braccia and Voshell, 2007; Vidon et al., 2008; O'Callaghan et al. 2019, Raymond and Vondracek, 2011). Facilitating animal access to watercourses provides farmers with a cheap, low-maintenance source of water for their livestock.

Cattle access to streams has been linked with increases in levels of deposited sediment and habitat homogenisation (Herbst et al., 2012), hydromorphological changes (Belsky et al., 1999), degraded riparian conditions (Grudzinski et al., 2016), physico-chemical degradation (Arnaiz et al., 2011), alterations in biological communities such as aquatic macroinvertebrates (Braccia and Voshell, 2007; Conroy et al. 2016a) and increases in sediment concentrations of *Escherichiacoli* (Bragina et al., 2017).

Cattle access mediated changes in macroinvertebrate communities are driven primarily by changes in stream hydromorphology and riparian habitat conditions, and in elevated nutrient and sediment levels in streams (Li et al., 1994; Stone et al., 2005; Arnaiz et al., 2011; Miserendino et al., 2011; O'Callaghan et al. 2019). Increases in limiting nutrients such as phosphorus and nitrogen, as a result of cattle defecation both directly in streams and in the adjacent riparian zone, can lead to concentrations that are detrimental in themselves (Sarriquet et al., 2006), but can also result (in association with reductions in riparian cover/shade) in proliferations of instream algal communities (Braccia and Voshell, 2007). Damage to stream banks by the erosive action of cattle hooves as they enter and exit streams, and as they consume and trample streamside vegetation, contributes significantly to stream suspended sediment inputs and bed loads of fine sediment (Scrimgeour and Kendall, 2003; Braccia and Voshell, 2007; Ranganath et al., 2009; Schulte et al., 2009; O'Sullivan et al., 2019a,b; Rice et al., 2021). Habitat change as a result of algal proliferation and sediment induced stream bed homogenisation seriously alters macroinvertebrate communities in affected streams (Stone et al., 2005; Sarriquet et al., 2006; Braccia and Voshell, 2007). The environmental impact of cattle access to watercourses have been recognised in agri-environmental policy, with cattle exclusion being included in various schemes under Pillar I and Pillar II of the Common Agricultural Policy (see Kilgarriff et al., 2020).

Fenced riparian buffer measures have been included in most European agri-environment schemes (Dworak et al., 2009), including Ireland's, and are amongst the commonest mitigation measures to prevent cattle access to watercourses.

Various methods of limiting livestock access to waterbodies have been used and empirically tested. Miner et al. (1992), Bagshaw et al. (2008) and Rawluk et al. (2014) studied the use of alternative water sources to encourage cattle congregation away from streams and riparian areas. Legrand et al. (2011) examined the use of "cow showers" as a means of attracting cattle away from shaded riparian areas in the arid Californian climate. The impacts of varying grazing practices have been tested by Raymond and Vondracek (2011), where the benefits of reduced grazing intensity on stream ecosystems via rotational grazing methods were demonstrated. More commonly, however, has been the complete exclusion of cattle from watercourses via fencing (Li et al., 1994; Bewsell et al., 2007; Collins et al., 2010).

Various studies have cited how restriction of cattle access to streams has positively affected factors such as instream and riparian habitat (Weigel et al., 2000); stream hydromorphology (Agouridis et al., 2005), reduced sediment loads (Collins et al., 2010) and physico-chemical conditions (Batchelor et al., 2015; Line, 2003). Fewer studies have examined the response of stream biota to restricted cattle access and most of these studies are from North America (e.g. Raymond and Vondracek, 2011 (macroinvertebrates), Derlet et al., 2017 (microbial)). The efficacy of cattle exclusion as a means of improving stream water ecological quality has not been widely assessed in the context of the temperate northwest of Europe where pastoral farming predominates.

This study aimed to assess the environmental benefits of cattle exclusion via fencing in the short-term (one-year post-fencing), by examining changes in levels of deposited sediment, habitat condition and macroinvertebrate communities in streams in Ireland. Additionally, a case study on the longer-term responses to fencing (9 years) is also presented. Here, 2008 macroinvertebrate data from two sub-catchments (one which was entirely fenced as a part of a group water scheme in the following year), are compared to data collected nine years following exclusion. The results will help inform further development of programmes of measures to improve water quality in agricultural catchments.

2. Methods

2.1 Sites

Eight cattle access points located in five catchments in the east and south of Ireland were sampled for macroinvertebrates and deposited sediment prior to, and one year following exclusion of cattle from streams via fencing (as a part of the Green Low Carbon Agri-environment Scheme (GLAS)) in October 2016 and October 2017, respectively (Figure 1). Three sites (GLAS Sites 1, 2 and 3) were located along the moderate status, second order Blacklion stream in Co. Carlow. Two sites (GLAS Sites 4 and 5) were located on separate, good-high status tributaries of the Munster Blackwater in Co. Cork. GLAS Site 6 was located on the poor status Commons stream, a tributary of the White river in Co. Louth. GLAS Site 7 was located on a good status, first order tributary of the Big river in Co. Louth. GLAS Site 8 was on a moderate status tributary of the Milltown Lake catchment in Co. Monaghan. Geology was variable (see supplementary information) and across sites while catchment land use was principally grassland. Stream substrates at GLAS Sites 1, 2, 4 and 7 were predominately gravel whereas coarse substrates, mainly cobble, dominated at all other sites. Stream widths ranged from 0.72m to 3.60 m (supplementary information).

The study on the long-term efficacy of fencing was carried out in the Milltown Lake catchment of Co. Monaghan (Figure 1). Here 13 sites were sampled in October 2008 (see Wynne and Linnane, 2008). Seven sites were located in a sub-catchment fed from the Carnagh lake in Tievenamara (TV), that was entirely fenced off from cattle following sampling in 2008, as part of remediation works for a group water scheme. Six sites were located in a control catchment in the Tullycaghney (TH) that was not fenced as a part of a concerted effort (although some ad-hoc, localised fencing has taken place since sampling in 2008). All sites were re-sampled in October 2017, nine years post fencing.

2.2 Macroinvertebrate sampling

For the short-term fencing study, twelve macroinvertebrate samples were collected at each site prior to fencing (in October 2016) and one-year post fencing (in October 2017) using a standard 0.9m^2Surber sampler. Sampling points were located in riffle geomorphic units immediately upstream (x 6) and downstream (x 6) of cattle access points, hereafter referred to as *control* and *pressure* points, respectively. A total of 96 samples (six Surbers per sampling point) were collected in each year. The majority of insect taxa were identified to species level; Coleoptera and Diptera were identified to family level and oligochaetes were left at order level.

For the long-term fencing study, samples were collected (in 2008 and 2019) by 2-minute kick sampling from riffle habitats at each sampling point. To enable comparisons with the pre-fencing study (Wynne and Linnane, 2008) the sampling methods of Wynne and Linnane were replicated as closely as possible and the macroinvertebrates were identified to family level.

2.3 Sediment sampling

Deposited sediment sampling was undertaken as part of the short-term study only. The samples were taken from riffle habitats at control and pressure points at the same time as macroinvertebrate sampling, using the 'Quorer' resuspendable sediment sampling technique (Lambert and Walling, 1988; Conroy et al., 2016b). Six samples were collected at each sampling point resulting in a total of 96 deposited sediment samples from each sampling period.

The deposited sediment samples were filtered using pre-ashed Whatman glass Microfibre GF/C filters following standard methods for suspended sediment as set out by the American Public Health Association (APHA, 1995). They dried at 1030C to a constant weight and weighed to give sediment mass (g). Organic matter mass was determined by weights of filter papers upon loss on ignition of samples dried at 5500C to a constant weight (APHA, 1995). Estimated resuspended sediment was calculated as mg l^{-1} and converted to g m^{-2} using the water volume within the stilling well (Conroy et al., 2016b). Percent organic matter (%OM) was calculated as a percentage of overall sediment mass.

2.4 Habitat assessment

Habitat assessment was carried out as part of the short-term study prior to fencing (2016) and following one year of exclusion (2017). It consisted of both quantitative measures of stream bank parameters and an assessment of qualitative parameters, carried out at the reach scale. A total of 13 qualitative sub-indices were calculated and were summed to produce a Total Habitat Index (THI) and a Riparian Habitat Index (RHI) (see O'Sullivan et al., 2019a). Each habitat assessment sub-index was scored on an ordinal scale from 0-20 or 0-10 where stream banks were assessed individually, with higher scores equating to better habitat condition.

2.5 Data analysis

Short-term study

A before, after, control and impact (BACI) study design was used to test whether short-term exclusion of cattle via fencing affected levels of deposited stream sediment metrics and macroinvertebrate community structure. The analyses of macroinvertebrate data tested the hypothesis that differences in community structure between control and pressure points prior to fencing would be greater than those differences following fencing and one year of cattle exclusion. This hypothesis was tested in Primer using the PERMDISP function (Anderson, 2006).

Here, *Site* (levels: 1-8) and *Fencing* (levels: Pre and Post) factors were combined to create the grouping factor for the PERMDISP procedure. Thus, samples from control and pressure points were combined for each site and the homogeneity of dispersions between samples (taking in both control and pressure samples) prior to and following fencing were tested via permutations, as is the protocol with the PERMDISP procedure. The hypothesis, thus states that homogeneity of dispersions will increase following fencing, or conversely, the within group variation will be reduced following fencing (Figure 2).

Pairwise comparisons of PERMDISP results and mean z scores were assessed to identify any convergence or divergence of samples between pre-fencing and post-fencing time periods at each site.

Potential differences in the multivariate locations (centroids) of macroinvertebrate samples (community structure) based on control v pressure differences before and after fencing were tested using PERMANOVA. In these analyses *Fencing* and *Treatment* were combined to create a fixed factor with four levels (Pre-fencing control, pre-fencing pressure, post-fencing control and post-fencing pressure).*Site* was a random factor with eight levels (GLAS Sites 1, 2, 3, 4, 5, 6, 7 and 8).

SIMPER analysis was used to identify taxa that contributed to dissimilarities in community structure between four combinations of*Fencing/Treatment* levels:

1. pre-fencing and post-fencing control points
2. pre-fencing and post-fencing pressure points
3. pre-fencing control and pre-fencing pressure points
4. post-fencing control and post-fencing pressure points.

These SIMPER comparisons were made using data from across all eight sites to identify taxa that demonstrated consistent changes to exclusion.

Univariate metrics (Ephemeroptera Plecoptera Trichoptera (EPT) abundance, % EPT, richness indices, functional feeding group (FFG) metrics and fine sediment sensitivity rating (FSSR) metrics) were also analysed using this study design. Pairwise tests for pairs of the*Fencing/Treatment* term within levels of the *Site* term were carried out to investigate any interaction effects in relation to univariate metrics.

Mean deposited sediment values were calculated for each Treatment level (control and pressure) at each site for the pre-fencing and post-fencing periods. PERMANOVA analysis was carried out using a two-factor design with Fencing and Treatment as two fixed factors, each with two levels (pre-fencing and post-fencing, and control and pressure, respectively).

With respect to habitat, qualitative habitat assessment scores were analysed as univariate data using a two factor PERMANOVA (based on Euclidean distance) with *Fencing* and *Treatment* as fixed factors. Habitat scores were square root transformed prior to analysis.

2.6 Long-term study

A blocked study design was used to assess for effects of long-term fencing on macroinvertebrate community structure, using a three factor PERMANOVA. The three factors were *Fencing* and *Catchment*(both fixed with two levels; pre-fencing and post-fencing and fenced (TV) and control (TH) respectively) and *Site* nested in*Catchment* .

Blocked study designs as carried out in Primer v.6 are akin to repeated measure PERMANOVAs and proceed with the exclusion of the 'highest level' interaction term (*Site(Catchment)* x *Fencing* in this case) from the model (Anderson et al., 2008). Blocking of un-replicated samples leads to greater power of detection (Fisher, 1935).

3. Results

3.1 Short-term study

3.1.1 Sediment mass and organic matter content: GLAS sites

Mean deposited sediment levels in the pre-fencing period ranged from 19.96± 5.29 gm^{-2} (SE) at the GLAS site 11 control point to 538.15± 473.74 gm^{-2} (SE) at the GLAS site 3 pressure point. In the post fencing period, mean deposited sediment levels ranged from 40.48± 11.94 gm^{-2} (SE) at the GLAS site 7 control point to 458.76± 197.88 gm^{-2} (SE) at the GLAS site 1 control point.

PERMDISP analysis detected a marginally significant difference in homogeneity of dispersions among samples for levels of deposited sediment mass (z scores) at sampling points prior to fencing compared to sampling points following fencing ($F_{15,188}$= P=0.06*), but none in relation to %OM. No reductions were found, however a pattern of reductions in dispersions (z scores) following fencing was observed at several sites (Figure 3).

A significant interaction effect was identified between *Fencing* and *Treatment* in relation to deposited sediment mass ($F_{1,28}$= 5.17, P=0.03). Pairwise results demonstrated significantly greater masses of deposited sediment at pressure points compared to control points, prior to fencing ($t_{1,14}$=2.38, P=0.04) but no significant difference following fencing. It should be noted however, that there was also a significant increase in levels of deposited sediment at control points in year 2 compared to year 1 ($t_{1,14}$=3.09, P=0.01) (Figure 4).

There were no significant effects in relation to %OM data for either PERMDISP or PERMANOVA analyses.

3.1.2 Macroinvertebrate

A significant difference was detected in the magnitude of multivariate dispersions among samples (across control and pressure points) between pre-fencing and post-fencing periods ($F_{15,176}$=3.81, P=0.01). Pairwise comparisons indicated that at Site 1 ($t_{1,10}$=2.34, P=0.04) and Site 3 ($t_{1,10}$=3.58, P=0.01) (both on the Blacklion stream) the magnitude of multivariate dispersion between samples was significantly less following fencing. Additionally, a marginally significant decrease in sample dispersions was found in relation to Site 4 ($t_{1,10}$=2.07, P=0.06) (Figure 5).

Principle coordinate ordination (PCO) plots demonstrate (in most cases) closer grouping of macroinvertebrate data between control and pressure points post fencing (Figure 6).

SIMPER analysis highlighted twelve taxa that consistently contributed to dissimilarities between the *Treatment/Fencing* levels (Table 1).*A. fluviatilis, R. semicolorata, S. pallipes* and *Agapetus*sp. had higher abundance at control points than at pressures points in the pre-sampling period (Pre-Pressure v Pre-Control) and greater in the post-fencing period compared to the pre-fencing period at both control (Pre-Control v Post-Control) and pressure (Pre-Pressure v Post-Pressure) points. The grazing riffle beetles, *Elmis aenea* and *Limnius volckmari* occurred in higher abundance at pressure points following fencing. Three taxa (*B. rhodani/atlaniticus* , Simuliidae and Chironomidae) were less abundant in the post-fencing period at both control and pressure points, and less abundant at control points in both the pre-fencing and post-fencing periods.

In terms of the univariate metrics, there was a significant interaction effect (*Fencing/Treatment* x *Site*) in relation to total richness ($F_{21,157}$=5.10, P=0.01) and EPT richness ($F_{21,157}$=5.96, P=0.01). Pairwise results highlighted significantly higher values at the pre-fencing control point relative to the pre-fencing pressure point at Site 3 ($t_{1,10}$=5.17, P=0.01), Site 6 ($t_{1,10}$=2.63, P=0.04) and Site 7 ($t_{1,10}$=4.15, P=0.01), while at Site 1 total richness values were greater at the pressure point ($t_{1,10}$=3.92, P=0.01) (Figure 7). Following fencing none of these significant differences in total richness persisted and there were significant increases in total richness at the pressure points following fencing compared to prior to fencing at Site 2 ($t_{1,10}$=4.13, P=0.01), Site 3 ($t_{1,10}$=6.29, P=0.01) and Site 6 ($t_{1,10}$=2.79, P=0.03)

In relation to EPT richness (Figure 8), there were higher values at the control points relative to the pressure points at Site 3 ($t_{1,10}$=6.76, P=0.01) and Site 7 ($t_{1,10}$=3.80, P=0.01) in the pre-fencing period. At Site 1, EPT richness values were greater at the pressure point in the pre-fencing period ($t_{1,10}$=3.39, P=0.02). Here again, none of these differences persisted following fencing and there were also significant increases in EPT richness at pressure points following fencing at Site 2 ($t_{1,10}$=5.26, P=0.01) and Site 3 ($t_{1,10}$=5.26, P=0.01).

3.1.3 Habitat assessment: short-term study

Univariate PERMANOVA analysis of habitat index scores (THI and RHI) did not highlight any difference between pre- and post-fencing periods. Multivariate analysis of habitat score sub-indices and stream substrate cover, geomorphic unit representation, physico-chemical measurements and stream dimensions, similarly did not detect any differences. However, RHI scores (available in supplementary information) did show an increase at pressure points following fencing, but RHI scores were typically higher at control points.

In relation to the univariate analyses, the *ground cover*sub-index of the RHI and THI was the only habitat parameter that showed a difference between pre- and post-fencing periods. A significant*Fencing* and *Treatment* interaction was observed ($F_{1,28}$=6.44, P=0.02) and pairwise results highlighted a significantly ($t_{1,14}$=2.65, P = 0.05) higher metric value at post-fencing pressure points (9.75 ±0.66 SE) compared to pre-fencing pressure points (7±0.2.65 SE). Significant main term *Treatment*effects with no *Fencing* interaction were detected for longitudinal connectivity ($F_{1,28}$=9.29, P=0.01), canopy cover ($F_{1,28}$=7.58, P=0.01) and shrub layer cover ($F_{1,28}$=8.82, P=0.01). For each of these RHI sub-indices, mean values at control points were greater than for those at pressure points.

3.2 Long term study

There was a significant interaction between the factors *Fencing*and *Catchment* in relation to community structure data in the Milltown Lake catchment ($F_{1,11}$=2.78, P = 0.02). Significant differences were detected in community structure in both the fenced ($t_{1,6}$=3.09, P=0.01) and the control ($t_{1,5}$=2.51, P = 0.01) catchments between pre- and post-fencing periods. Pairwise results also showed that prior to fencing there was a significant difference in community structure between the 'to be fenced' catchment and control catchment ($t_{1,11}$=2.31, P = 0.01) that did not persist following the fencing period.

Ordination based on principle components (PCO) illustrated a clear separation between pre- and post-fencing sites in the fenced catchment along the first PCO axes which accounted for 34.8% of total variation in the community structure (Figure 9). There is also a separation between pre-and post-fencing samples in the control catchment, although this appears not to be as pronounced as in the fenced catchment.

SIMPER analysis on the fenced catchment data showed that increased abundances of Simuliidae, *Ancylus fluviatilis,* Glossomatidae, Elmidae and Baetidae, and reduced abundance of Asellidae, in the post fencing period accounted for 41% of the dissimilarity in community structure between the pre- and post-fencing period. In the control catchment 48% of the dissimilarity in community structure between the two periods was due to increased abundances of Gammaridae and Simuliidae in the post-fencing period, and reduced abundances of Baetidae,*Hydropsyche siltalai* and Elmidae.

In relation to univariate metrics, there was a significant interaction effect (*Fencing* x *Catchment*) for % ephemeropteran abundance (%E) ($F_{1,11}$=19.62, P=0.01) and EPT abundance ($F_{1,11}$=14.69, P=0.01). Pairwise results for both, showed significant differences in values between pre- and post-fencing periods in both the fenced and control catchments. In the fenced catchment values for both metrics generally increased following fencing while in the control catchment values for both generally decreased (Figure 10).

4. Discussion

Degradation of freshwater resources with nutrients and sediment as a result of agricultural practices is a key global challenge (Novotny, 1999; Schulte et al., 2009; EEA 2018; EPA 2020). Direct cattle access to streams also represents a point source of nutrients, sediment and pathogens (Godwin and Miner, 1996; Belsky et al., 1999; Davies-Colley et al., 2004; Conroy et al., 2016a; Bragina et al., 2017, Rice et al., 2021). In this study the efficacy of fencing as a means of reducing in-stream sediment deposition and improving stream ecological quality in both the short and longer term has been demonstrated.

Cattle exclusion over the short-term impacted instream deposited sediment and aquatic macroinvertebrate communities

Significantly greater mean masses of deposited sediment were observed at pressure points (relative to control points) prior to fencing, however, following fencing there was no difference between control and pressure points, despite a general increase in background levels of deposited sediment. Patch scale deposits of fine sediment (as assessed here) reflect stream bank erosion in the immediate upstream vicinity of the affected site (Larsen et al., 2009). Habitat assessment results here did not highlight any improvement in bank stability following exclusion of cattle access, although increased ground cover in riparian areas was observed. Reductions in the extent of bare ground following cattle exclusion via fencing were also observed in Alberta, Canada by (Miller et al., 2010a) and in the north of Ireland by Rice et al. (2021) and were linked to improved water quality (Miller et al., 2010a). These authors also found no improvement in stream bank stability following four years of cattle exclusion but did suggest that increased ground cover could also contribute to reduced rates of stream bank erosion. Collins et al. (2010) concluded that stream bank erosion was significantly reduced in all of six catchments studied in the UK following fencing and eight years of exclusion, leading to reduced siltation of salmonid spawning gravels. Hansen and Budy (2011) however did observe significant improvement in stream bank stability after only one to two years of exclusion of cattle via fencing in Utah. The time-frame of the current study may have been too short to allow for improvements in bank stability following fencing.

Elevated levels of deposited sediment have been shown to drive changes in macroinvertebrate communities affected by cattle access (Braccia and Voshell, 2006; Conroy et al. 2018). The results here have highlighted the elimination of pre-fencing disparities in masses of deposited sediment between control and pressure points as a result of one year of exclusion via fencing. Additionally, reductions in multivariate dispersions (macroinvertebrate data) between control and pressure samples following fencing have been highlighted at several sites. Warwick and Clarke (1993) found that multivariate dispersion among marine community samples collected from impacted areas may be greater than from un-impacted areas, thus, here, higher dispersion among (control and pressure) samples prior to fencing compared to post-fencing may indicate reduced community stress following restoration via fencing. Other authors (see White et al., 2017) have shown increases in the magnitude of multivariate dispersion following restoration of streams and instream habitats due to improved habitat heterogeneity. It should be noted however that multivariate dispersions in the current study were measured across the gradient of impact (i.e. upstream and downstream of cattle access points) and thus reductions in dispersion here represent a convergence of communities towards *control* conditions following the removal of the cattle access impact. Results for EPT richness and total richness also suggested that improvements in ecological quality had occurred following one year of cattle exclusion at certain sites.

Cattle exclusion had a significant impact on grazing ephemeropteran, plecopteran and trichopteran taxa

SIMPER results highlighted consistency in the taxa that contributed to dissimilarities between the communities in the pre and post fencing periods. The grazing EPT taxa, *R. semicolorata, B. rhodani, Silo pallipes* , *Agapetus* sp. all contributed significantly to these dissimilarities and these taxa generally showed consistent responses (except *B. rhodani*). During the post-fencing sampling season,*R. semicolorata* , *S. pallipes* and *Agapetus* sp. increased in abundances at both control and pressure points (across all sites) compared to the pre-fencing period. Pre-fencing, these taxa occurred in greater abundances at control than pressure points. Interestingly however, this pattern was reversed in the post-fencing period with abundances at pressure points increasing beyond those seen at control points. Such a response may indicate that affected streams are in a state of intermediate disturbance (Connell, 1978), with community niches developing due to an abundance of food resources (algae) for grazing taxa following a reduction of the main community stressor, sediment. Cattle access points have been shown to negatively affect grazers and EPT fauna such as *Agapetus* sp. and *R. semicolorata* with sediment highlighted as the most likely stressor (O'Sullivan unpublished). Similar impacts have also been highlighted by Braccia and Voshell (2006) and Burdon et al. (2013). Elevated deposited sediment can affect periphytic algal biomass and reduce its palatability (Wood and Armitage, 1997; Schofield et al., 2004; Jones et al., 2012). Higher fine sediment deposits downstream of cattle access points have been detected in first and second order streams in the Irish setting (O'Sullivan et al., 2019a) and were demonstrated here in the pre-fencing sampling period. Such deposits however are transient in nature and can be removed by high flows (Gomi et al., 2005). Nonetheless, macroinvertebrate community composition reflects prevailing stream conditions (Metcalfe, 1989) and sediment inputs from recurrent entry of cattle to streams may represent a continuous stressor for these communities.

Recovery time in macroinvertebrate communities from stressor effects is variable and dependent on the specific stressor or stressors shaping affected communities and the aspects of the communities assessed (Laasonen et al., 1998; Muotka et al., 2002). Several authors have highlighted similarly rapid responses in macroinvertebrate communities to restoration efforts although not specifically relating to the exclusion of cattle via fencing. Miller et al. (2010b) concluded from a meta-analysis of a range of restoration projects that improved macroinvertebrate richness can occur in communities within one year of the commencement of restorative works. Contrastingly, Laasonen et al. (1998) demonstrated a delayed recovery of shredder macroinvertebrates in streams channelized for logging activities, relating the delay to poor retention of coarse particulate organic matter (CPOM) in such streams. In relation to cattle exclusion studies, recovery times are equally variable with this variability reflecting, again, the main stressor influencing communities and the scale of exclusionary measures. Herbst et al. (2012) reported a relatively rapid recovery of macroinvertebrate communities (in streams where bank erosion and sediment deposition were significant drivers) following fencing (4 years), however with active restoration (involving stream channel adjustment), even quicker responses (2 years) were observed (Herbst and Kane, 2009). Where significant depletion of riparian vegetation occurs however, greater recovery times can be expected, owing to the substantial time required for woody vegetation to re-establish and grow (Belsky et al., 1999; Braccia and Voshell, 2007; Ranganath et al., 2009; Herbst et al., 2012).

Rapid responses of macroinvertebrates to restorative efforts are indicative of communities that possess good elasticity (Halpern, 1988). Greathouse et al. (2005) state that, recovery is driven by nearby sources of macroinvertebrates for colonisation and the mobility of dominant taxa. This point is reiterated by Diaz Villanueva et al. (2017) who state that the distribution of invertebrates in disturbed environments is shaped by the dispersal capabilities of the resident taxa and the changes in environmental conditions along a spatial gradient. Wallace (1990) cites the importance of drifting taxa and aerially dispersing taxa (particularly in headwater streams) for ecological recovery. Here eight out of the twelve taxa listed in Table 1 have an adult life stage that disperses aerially. Also, the downstream extent of impacts of cattle access points in headwater streams is limited (O'Sullivan et al., 2019a), thus the rapid restoration of communities observed here is likely aided by both drifting taxa from upstream control sites and from aerial dispersal of adults from un-impacted downstream sites.

Fencing over the long-term results in improvements in macroinvertebrate communities

In the present study, significant improvements in stream macroinvertebrate community structure were seen in response to the long-term fencing of a sub-catchment of the Milltown Lake in Co. Monaghan. This was attributed to increases in the abundances of taxa such as *A. fluviatilis* and *Glossomatidae* (taxa that responded positively to fencing in the short term) and Elmidae (a taxon that is shown to be sensitive to cattle access (Braccia and Voshell, 2007) and was more abundant downstream of access points following short term fencing) in the post fencing sampling period of the fenced catchment. These results are more emphatic given the apparent decrease in ecological water quality in the control stream, where cattle access was still permitted over the nine-year period. The lower ecological quality scores in the post-fencing period were driven by increases in taxa such as Simuliidae and Gammaridae and reductions in Baetidae,*Hydropsyche siltalai* and Elmidae compared to the pre-fencing period. Despite Simuliidae and Gammaridae being highly sensitive and moderately sensitive to sediment respectively, they are also moderately tolerant of organic pollution and increases in their abundances may reflect such pollution (Herbst et al., 2012; Burdon et al., 2013). Furthermore, previous work by O'Sullivan et al. (2019a) demonstrated that deposited sediment levels were unaffected by cattle access at sites located in the control catchment, while a detailed characterisation of the catchment by Wynne and Linnane (2008) concluded that bare, coarse substrates were dominant in the catchment. Also work by Bragina et al. (2017) in 2012-2013 found significantly higher *Escherichia coli*levels in the stream sediments for the unfenced (control) tributary (TH) compared to the fenced tributary (TV), suggesting higher loading with organic matter from animal and/or human wastes.

The results from the univariate analyses also highlighted ecological improvements in the fenced catchment and deteriorations in the control catchment following the fencing period. The scope for inferences in relation to the impact of fencing however, are limited due to the lack of temporal replication over the course of the nine years between fencing and the post-fencing sampling period.

Limitations in the study design are acknowledged by the authors, particularly in relation to the failure of the analyses to account for fluctuations in environmental conditions (e.g. sediment) and resultant macroinvertebrate responses over time. Replication of sampling in relation to the *Fencing* factor (i.e. replicate sampling of sediment and macroinvertebrate populations at multiple times prior to fencing and after fencing, both at control and pressure points) would provide a more robust basis on which conclusions could be drawn (Stewart-Oaten et al., 1986; Miller et al., 2010b). Macroinvertebrate populations, however, respond to the prevailing environmental conditions and as such changes in macroinvertebrate metrics here are considered to reflect changes in sediment deposition in the longer term.

Multiple targeted mitigation measures should be incorporated and incentivised in catchment management plans

Results from this study support fencing as an effective method of improving the ecological integrity of headwater streams. However, such measures can be costly (Byers et al., 2005; Bayley and Li, 2008; Rawluk et al., 2014), impractical (Sheffield et al., 1997) and unattractive to farmers and other land managers (Graz et al., 2012). To avoid a one-size-fits-all approach, Schulte et al. (2009) promoted targeting fencing schemes to critical source areas, while Kilgarriff et al (2020) found that fencing areas according to agricultural intensity to be a cost-effective solution. Similarly, fencing and cattle exclusion alone may not be sufficient to restore the ecological condition of impacted watercourse and thus should not be considered a cure-all, but rather one measure in a suite of mitigation measures that could be integrated into wider management plans (Brannan et al., 2000; Schulte et al., 2009; Collins et al., 2010; Smolders et al., 2015).Further research on the socio-economic costs of fencing and costs associated with provision of an alternative water supply (if necessary) as part of integrated catchment management plans is required.

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Figure Legends

Figure 1 Site locations for sites used to assess the short-term (site names with the GLAS prefix) and long-term (TH (unfenced) and TV (fenced) site name prefixes) efficacy of cattle exclusion.

Figure 2 Experimental design demonstrating how the PERMDISP routine was applied in this study. PERMDISP tests the hypothesis that dispersion among triangles (pre-fencing) is greater than dispersion among squares (post-fencing). The black lines (a) represent an approximation of the dispersion/z-scores. The barbed wire represents fencing (b).

Figure 3 Measures of dispersions (z scores) between masses of deposited sediment samples among control and pressure points during the pre-fencing and post-fencing period.

Figure 4 Mean deposited sediment levels (+- SE) at control and pressure points prior to and following fencing across all sites. Letters (A and B) identify significant *Treatment* effects within a given *Fencing*period. Asterisks (* and **) identify significant *Fencing* effects within a given *Treatment* .

Figure 5 Measures of dispersions (z scores) among control and pressure macroinvertebrate samples during the pre-fencing and post-fencing period. Differing letters highlight where significant differences occur

Figure 6 PCO plots for macroinvertebrate community structure data (based on Bray-Curtis dissimilarity matrices) for Site 1-8 (labelled a-h).PERMANOVA analysis of macroinvertebrate community abundance (structure) highlighted a significant *Fencing/Treatment* x *Site* interaction ($F_{21,157}$=4.80, P=0.0002).

Figure 7 Total richness values (+- SE) at control and pressure points during the pre-fencing and post-fencing period Significant differences between levels of the Treatment factor are designated by differing letters (A and B), while differences between levels of the Fencing factor are designated by differing numbers of asterisks' (* and **).

Figure 8 EPT richness values (+- SE) at control and pressure points during the pre-fencing and post-fencing period. Significant differences between levels of the Treatment factor are designated by differing letters (A and B), while differences between levels of the *Fencing* factor are designated by differing numbers of asterisks' (* and **).

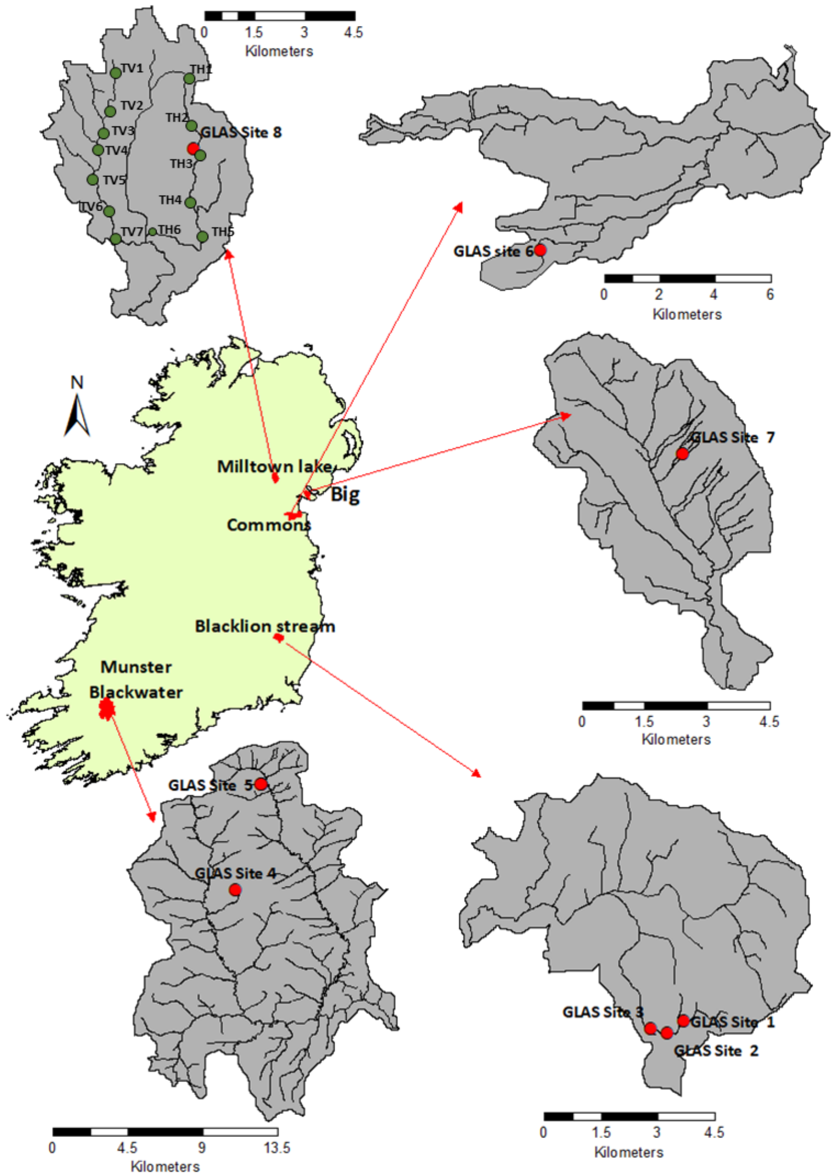
Figure 9 PCO plot of macroinvertebrate community structure data showing greater separation of pre-fencing control and pressure samples compared to post-fencing control and pressure samples.

Figure 10. Values of %E abundance and EPT richness for samples in the fenced (a and c) and control catchments (b and d).

Table

Table 1. SIMPER results for selected taxa. Each column indicates the relative change in abundance (indicated by directional arrow) of the specified taxon at the second listed*Fencing/Treatment* level compared to the first. The percentage contribution of each taxa to dissimilarities between the given*Fencing/Treatment* levels is given next to the directional arrow. Control and Pressure refer to the sampling points.

	Pre-Control v Post-Control	Pre-Control v
Ancylus fluviatilis	—	**5.91%**
Rhithrogena semicolorata	—	**4.77%**
Baetis rhodani	—	**4.28%**
Silo pallipes	—	**3.74%**
Agapetus sp.	—	**4.04%**
Hydropsychidae sp.	—	**3.89%**
Simuliidae	—	**5.42%**
Chironomidae	—	**6.37%**
Pediciidae	—	**3.17%**
Limnius volckmari	—	**5.21%**
Elmis aenea	—	**4.23%**
Gammarus duebeni	–	**5.00%**
Cumulative % contribution to dissimilarities of all taxa for each comparison	56.03%	56.03%


0 1.5 3 4.5
Kilometers
TV1
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TH6
TH5
GLAS site 6
0 2 4 6
Kilometers
N
Milltown lake
Big
Commons
GLAS Site 7
Blacklion stream
Munster
Blackwater
0 1.5 3 4.5
Kilometers
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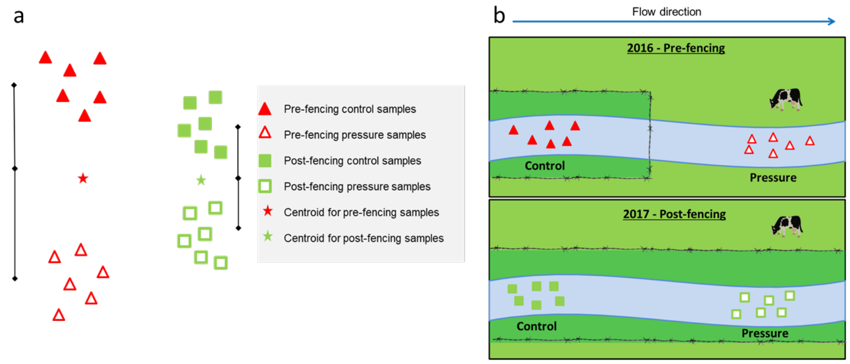
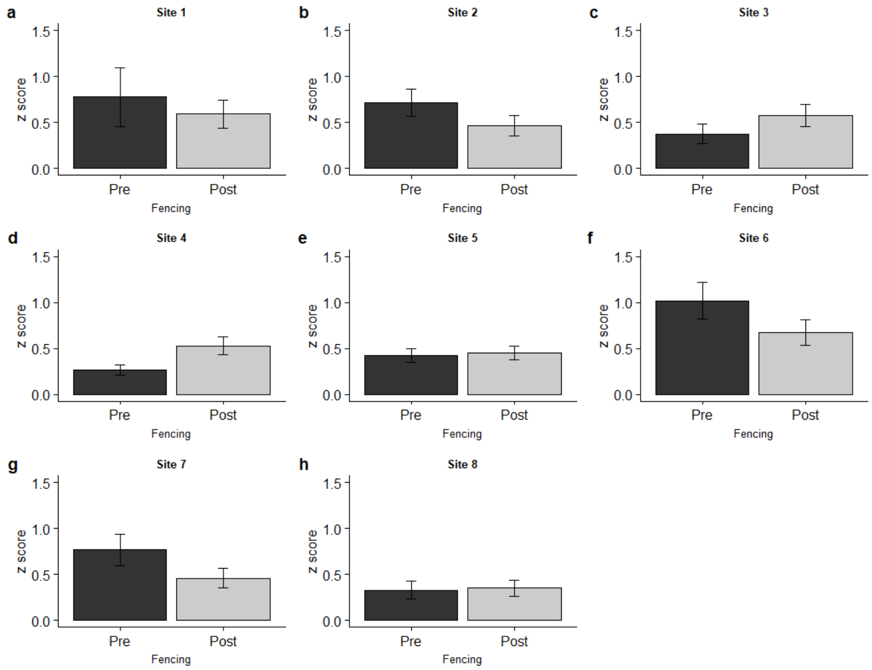
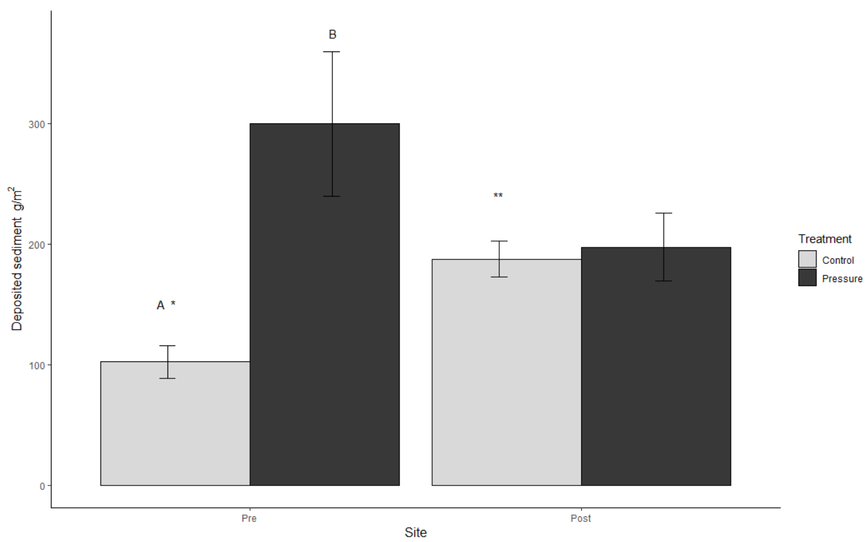
Kilometers

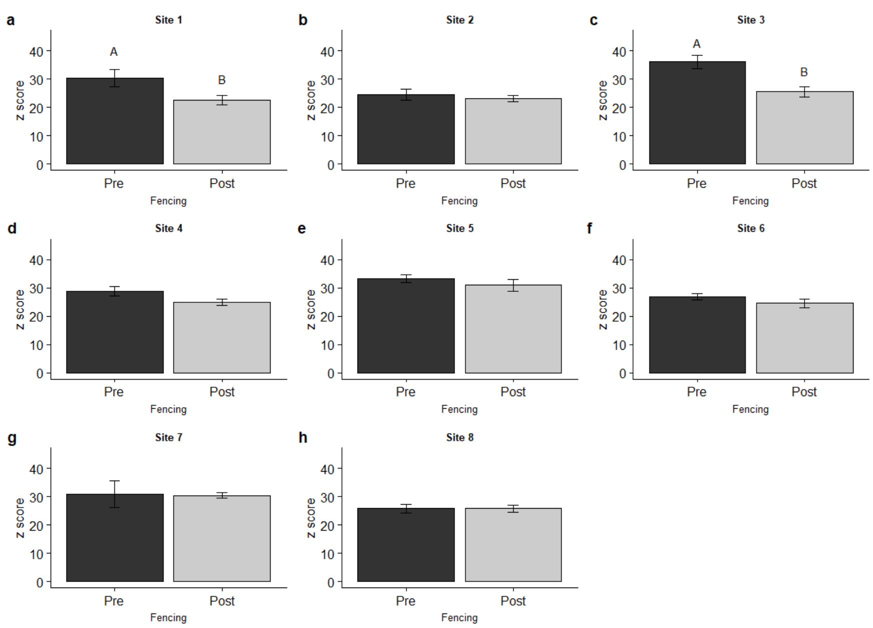


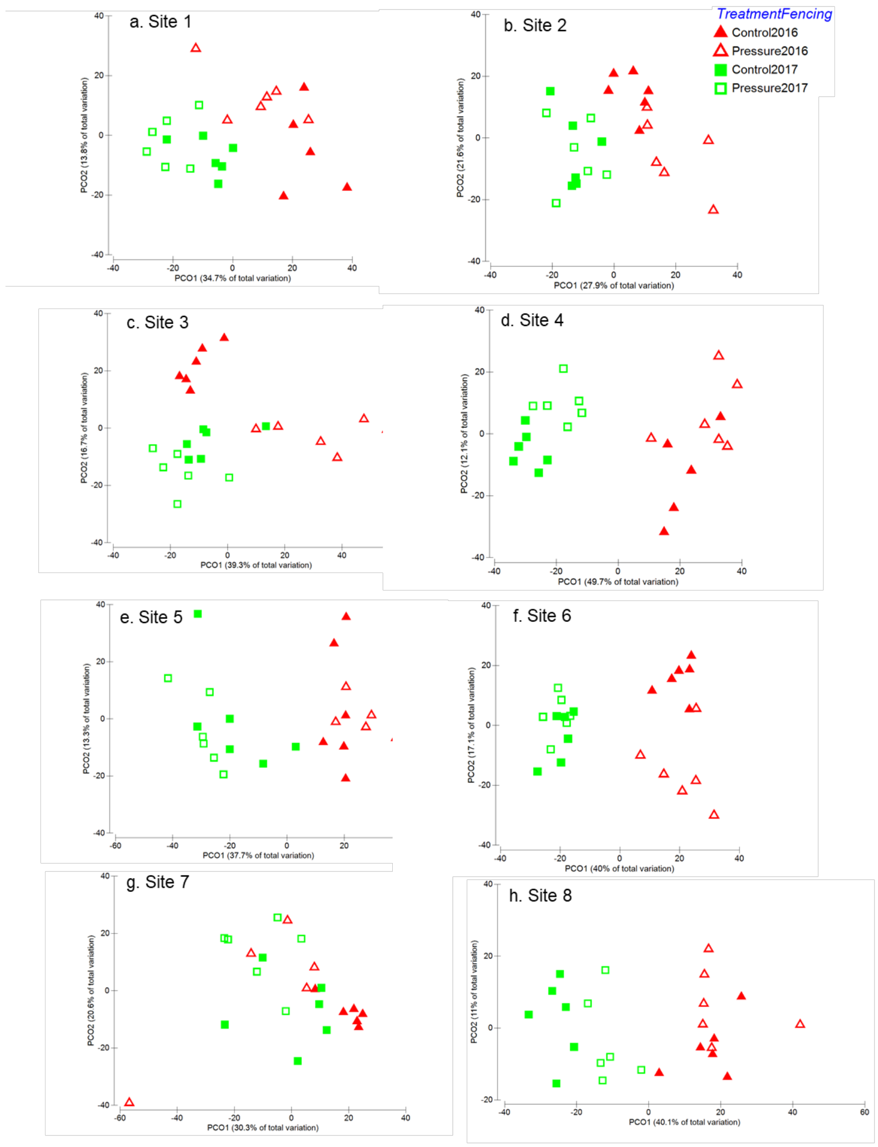
a
Pre-fencing control samples
Pre-fencing pressure samples
Post-fencing control samples
Post-fencing pressure samples
Centroid for pre-fencing samples
Centroid for post-fencing samples
b
Flow direction
2016 - Pre-fencing
Control
Pressure
2017 - Post-fencing
Control
Pressure

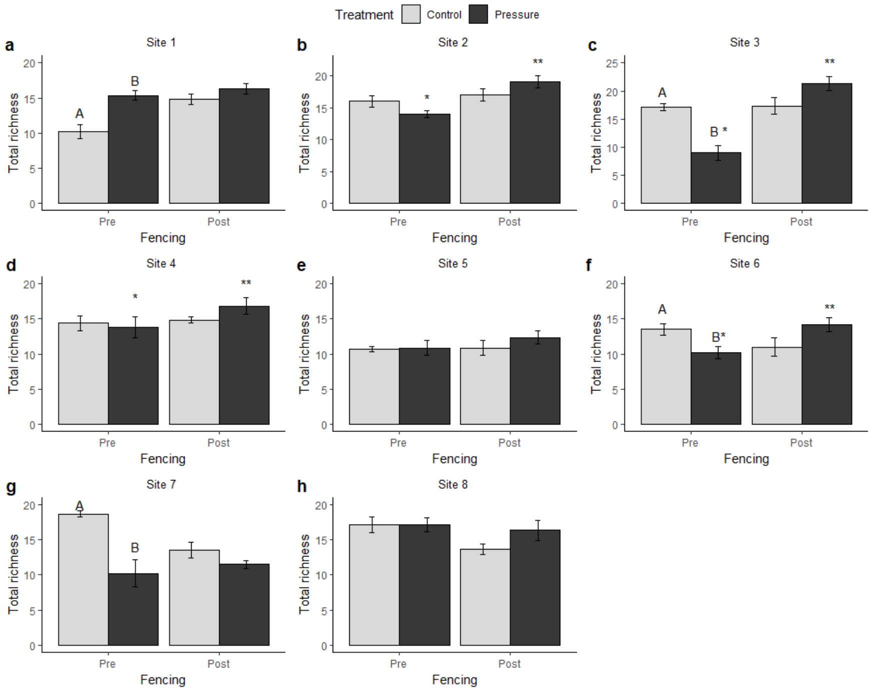


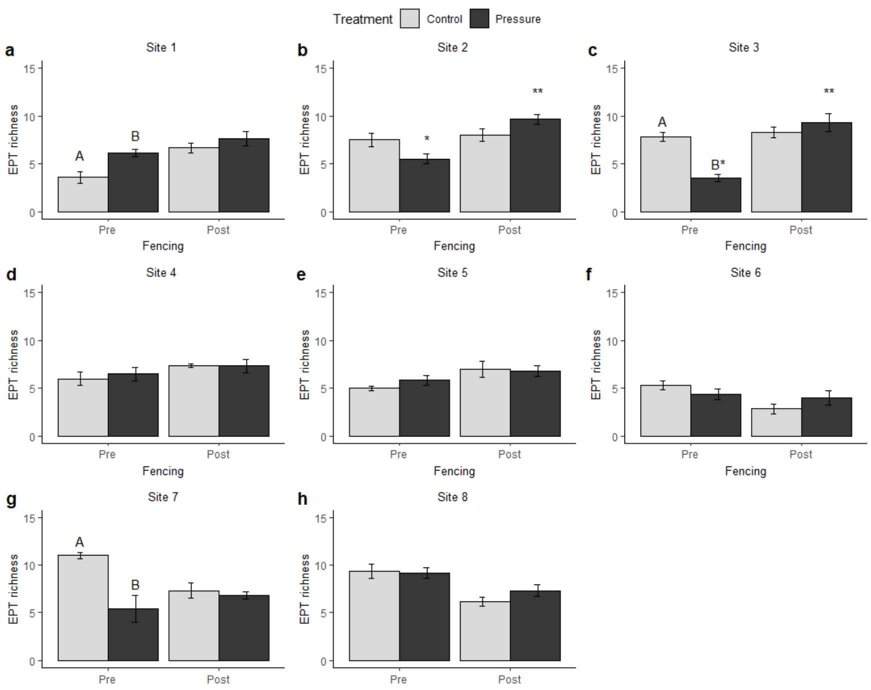
a Site 1
b Site 2
c Site 3
d Site 4
e Site 5
f Site 6
g Site 7
h Site 8
z score
0.0
0.5
1.0
1.5
Pre
Post
Fencing

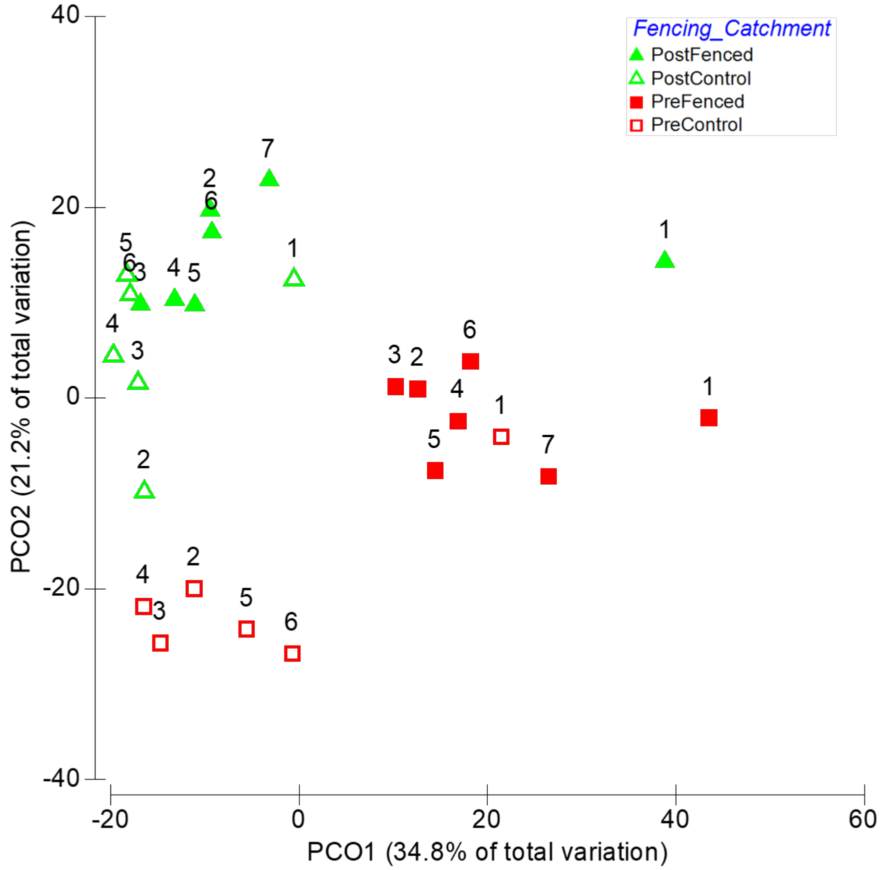

B
A *
**
300
200
100
0
Deposited sediment g/m^2
Pre
Post
Site
Treatment
Control
Pressure

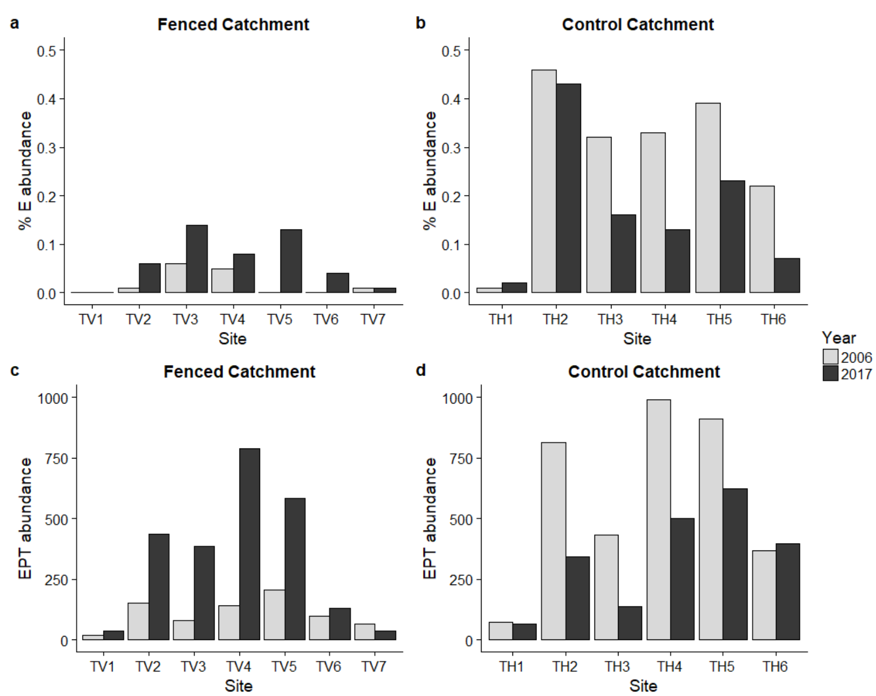

a
Site 1
b
Site 2
c
Site 3
d
Site 4
e
Site 5
f
Site 6
g
Site 7
h
Site 8
z score
Pre
Post
Fencing
A
B


a. Site 1
PCO1 (34.7% of total variation)
PCO2 (13.8% of total variation)
b. Site 2
PCO1 (27.9% of total variation)
PCO2 (21.6% of total variation)
TreatmentFencing
Control2016
Pressure2016
Control2017
Pressure2017
c. Site 3
PCO1 (39.3% of total variation)
PCO2 (16.7% of total variation)
d. Site 4
PCO1 (49.7% of total variation)
PCO2 (12.1% of total variation)
e. Site 5
PCO1 (37.7% of total variation)
PCO2 (13.3% of total variation)
f. Site 6
PCO1 (40% of total variation)
PCO2 (17.1% of total variation)
g. Site 7
PCO1 (30.3% of total variation)
PCO2 (20.6% of total variation)
h. Site 8
PCO1 (40.1% of total variation)
PCO2 (11% of total variation)


Treatment
Control
Pressure
a
Site 1
b
Site 2
c
Site 3
d
Site 4
e
Site 5
f
Site 6
g
Site 7
h
Site 8
Total richness
Fencing
Pre
Post
A
B
B *
B*
*
**


Treatment
Control
Pressure
a
Site 1
b
Site 2
c
Site 3
d
Site 4
e
Site 5
f
Site 6
g
Site 7
h
Site 8
EPT richness
Fencing
Pre
Post
A
B
B*
*
**


Fencing_Catchment
PostFenced
PostControl
PreFenced
PreControl
PCO2 (21.2% of total variation)
PCO1 (34.8% of total variation)


a
Fenced Catchment
b
Control Catchment
c
Fenced Catchment
d
Control Catchment
% E abundance
EPT abundance
Site
Year
2006
2017

Hosted file

`Table 1.docx` available at https://authorea.com/users/588330/articles/625577-mitigation-of-impacts-of-cattle-access-on-stream-ecosystems-efficacy-of-fencing