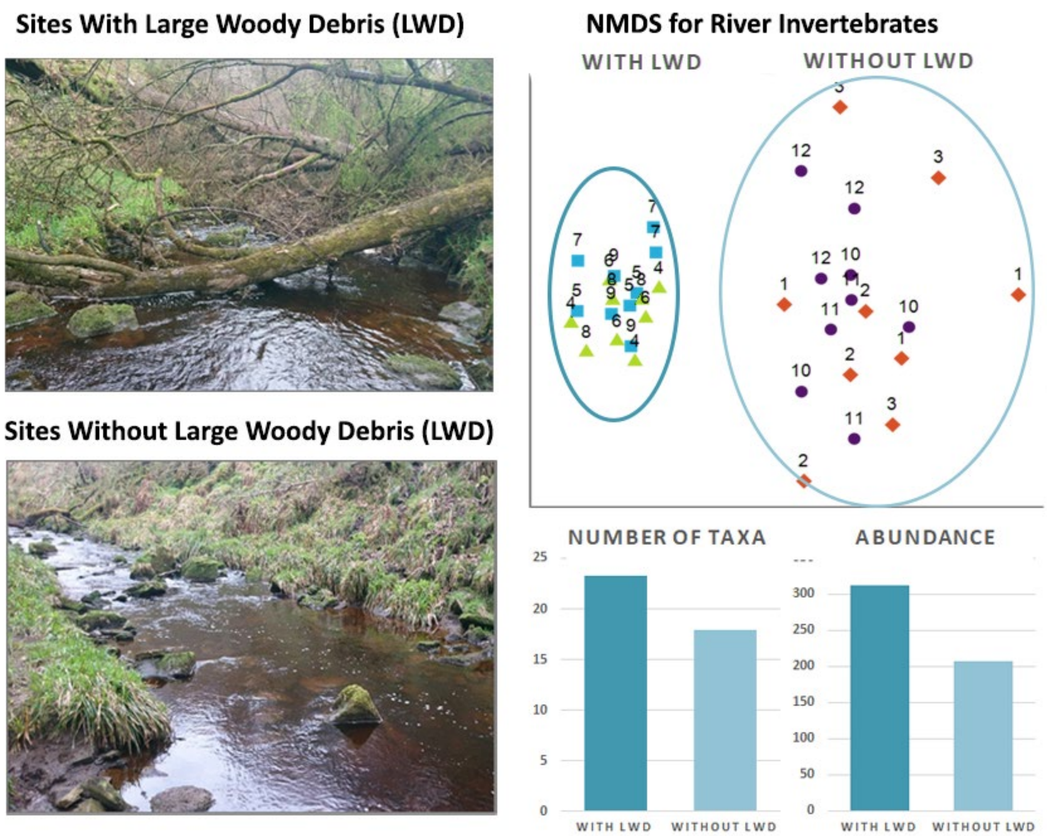
Riverine Large Woody Debris introduced for Natural Flood Management leads to rapid improvement in aquatic macroinvertebrate diversity

Deane, A[1,2]., Norrey J[1]., Coulthard, E[1]., McKendry, D.C[1]. and Dean, A.P.[*,1]

[1]Department of Natural Sciences, Faculty of Science and Engineering, Manchester Metropolitan University, Oxford Road, Manchester M1 5GD, UK; [2]Cheshire Wildlife Trust, Bickley Hall Farm, Malpas, Cheshire, SY14 8EF

* Corresponding author. E-mail address: andrew.dean@mmu.ac.uk; JDE431 John Dalton Building, Department of Natural Sciences, Faculty of Science and Engineering, Manchester Metropolitan University, Oxford Road, Manchester M1 5GD

Graphical Abstract



Abstract

Natural flood management interventions, such as Large Wood Debris (LWD) or engineered log jams, are being increasingly deployed throughout the UK and elsewhere. In addition to alleviating flood risk, it is anticipated that they may influence the ecology of freshwater river systems, including macroinvertebrate populations. This study explores macroinvertebrate assemblages, water quality parameters, and sediment size distribution in a headwater stream following the addition of LWD as part of a natural flood management scheme. The study area consists of 6 sites within the intervention zone where LWD was implemented, with comparative control sites upstream and downstream (3 sites each). Macroinvertebrate communities, sediment size distribution, and water chemistry and were sampled 3 and 10 months following the addition of LWD. Our findings revealed increased macroinvertebrate abundance and taxa richness in LWD intervention zone versus control, with an increased BMWP score reflecting the increased taxa richness. Average Score Per Taxon, and water chemistry showed no change, revealing invertebrate changes to be independent of water quality. NMDS and hierarchical clustering analysis on invertebrate data showed a clear separation of communities where LWD was present from those with no LWD while SIMPER analysis showed that LWD addition led to the rapid establishment of taxa (Hydraenidae, Rhyacophilidae, Scirtidae, and Elmidae) that were otherwise absent. Ten months after LWD addition, improved biodiversity was also found in areas below the intervention zone, suggesting the positive impacts of LWD extend downstream. LWD also altered sediments, with sites immediately upstream of LWD dams have a greater percentage of fine sediment than those immediately downstream. These results suggest that biological complexity and niche availability increased within the in-channel zone as a result of introduced LWD, thus

revealing wider aquatic habitat improvement potential of LWD for natural flood management. The use of LWD as an intervention for flood management is recommended for its additional benefits for ecosystem health.

Key words: Benthic Macroinvertebrates; Freshwater ecology; Large Woody Debris; LWD; Natural Flood Management; NFM

1. Introduction

Healthy river systems provide invaluable ecosystem services in the form of clean water, sediment transport, biodiversity and natural flood management (Thorne, 2014). Threats to these ecosystems include poor land management, non-native species, environmental pollution, dredging, draining, and channel modification (Carpenter et al., 2011; Everall et al., 2017; Holmes and Raven, 2014; Mainstone and Holmes, 2010; Thorne, 2014). These factors alter hydrology and reduce ecosystem function and diversity (Raven et al., 1998; Sear et al., 2000). Channel modification and dredging are often carried out to manage flood risk (Dadson et al., 2017). Flooding is one of the most pressing concerns relating to river system management (Pitt, 2008; Thorne, 2014; Wilkinson et al., 2019), and the management of flood risk, while also maintaining healthy biodiverse river systems is therefore difficult. Exploring alternative strategies to manage flood risk, while maintaining riverine biodiversity, should therefore be considered a priority.

Management of flood risk is however, undergoing a paradigm shift, with less emphasis on solely structural defences and channel engineering, and more towards the inclusion of catchment-based measures which attenuate flood risk (Lavers and Charlesworth, 2018; Wilkinson et al., 2019; Wingfield et al., 2019). Catchment-based schemes for flood alleviation include those that utilise natural flood management (NFM) approaches (Dadson et al., 2017;

Lane, 2017; Nicholson et al., 2012). NFM interventions can be broadly split into two categories. The first involves catchment wide measures (out of channel) such as woodland creation, hedgerows, soil de-compaction and *Sphagnum* inoculation. The second type is direct river network restoration (in-channel) such as grip blocking, diverter logs, floodplain reconnection, and online storage created through leaky dams and Large Woody Debris (LWD) restoration (SEPA, 2016).

Restoration of river channels using LWD involves the artificial reintroduction of woody material into the watercourse in the form of whole trees and/or large limbs. LWD is conventionally defined as woody material >0.1 m in diameter and >1 m in length (Gippel et al., 1996). Naturally occurring LWD has many benefits including the formation of gravel bars, flood regulation, increased hydraulic roughness of the channel, and increased habitat heterogeneity (Gurnell et al., 2005; Janes et al., 2017; Osei et al., 2015). These changes can lead to greater macroinvertebrate biodiversity (Gregory et al., 2003; Johnson et al., 2003; Magliozzi et al., 2019; Pilotto, et al., 2014) as well as wider biological benefits, including fish populations (Howsen et al., 2012). Artificially introduced LWD, for example, in woody engineered stream revetments for erosion control (Everall et al., 2012), or when woody debris is added directly to rivers (Elosegi et al., 2016, Flores et al., 2017, Kail and Hering, 2005) has also shown these hydrological and biologic benefits. Addition of LWD to streams can also replace organic material that has been depleted and lost from river systems due to the historic clearance of LWD from margins (Gurnell et al., 2005).

The introduction of LWD for flood management, via the felling or winching bankside trees into the water course, aims to achieve hydrological benefits though the reduction in downstream peak flows. The studies quoted above on natural LWD, engineered woody river revetments, and addition of woody debris for river restoration, suggest that cross-channel LWD dams for flood management will also lead to increased richness and abundance of the aquatic macroinvertebrate assemblages, though this is not certain, as not all restoration practices that increase habitat heterogeneity lead to biodiversity improvements (Palmer et al., 2010). Despite these potential biological benefits, research looking at the biological impact of engineered LWD log jams for flood alleviation is limited. More research is therefore needed to fully understand the biological impacts of LWD interventions that are specifically implemented for flood management purposes, as opposed to those implemented primarily for general habitat improvements. A wider understanding of the impacts of LWD, including any biodiversity improvements, will assist those at the forefront of delivery in securing funding and political capital for such works (Dadson et al., 2017).

This study aims to quantify the impacts on benthic aquatic invertebrates of LWD, in the form of cross-river dams that have been introduced to a river for flood management. Benthic invertebrates are commonly used for water quality assessment purposes, and are used to assess the success of stream restoration, and inform water quality management decisions (Kenney et al., 2009). It is hypothesised that the impacts on invertebrates and the river system will be similar to those seen in LWD additions designed specifically for biodiversity and habitat improvement. Reaches of the streams with flood alleviation measures, in the form of LWD, will therefore have higher species richness, abundance, biodiversity and improved biometric scores, than streams where LWD is absent.

2. Methods

2.1 Study Site

Black Brook is a headwater stream of the River Dane and is situated in the South West of the Peak District National Park, UK (Figure 1, Supplementary Figs. S1, S2). As part of the 'Slowing the Flow' flood alleviation project carried out by Cheshire Wildlife Trust (CWT) and the South West Peak Landscape Partnership Scheme (Cheshire Wildlife Trust, 2017), Black Brook underwent restorative habitat improvement works in January 2018, including the introduction of LWD to fulfil NFM and habitat enhancement objectives. This work consisted of selective felling of large bankside trees directly into the channel to create 19 engineered log jams. There were made from a total of 59 felled trees, with 2-4 trees making up each log jam, with a distance between each log jam of 20-50m. The LWD was installed to span the width of the channel and in some instances felled parallel with flow direction to create an interlinked mass of large wood within the active flow of the river.

A 2km stretch of Black Brook was selected for sampling. This stretch included six sites (Fig. 1: S4-S9) where LWD dams was present in the form artificially engineered log jams (intervention zone (IZ)), and three control (LWD absent) sites upstream (Fig. 1: S10-S12), and three sites downstream (Fig. 1: S1-S3) of the intervention zone. Grid references for sampling sites are provided in Table S1. Three replicate samples were obtained at each.

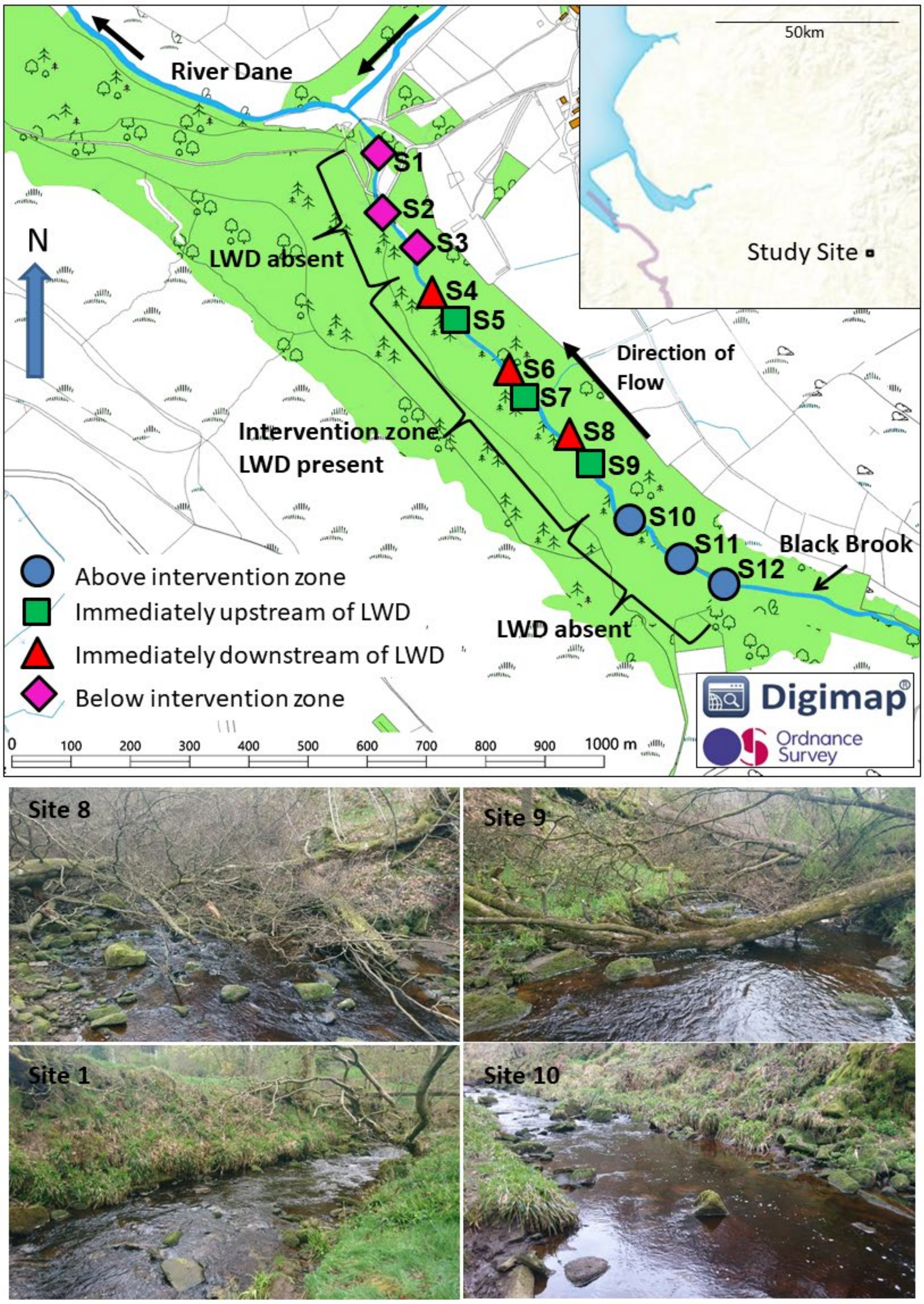


Figure 1: Sample sites within the Black Brook, River Dane, UK. The intervention zone (IZ) is located within the centre of the sampling reach and consisted of a suite of LWD engineered log jams. Six sites (S4-S9) were sampled within the intervention zone, with sites situated upstream and downstream of LWD dams. Three sites (S1-S3) were located below the intervention zone, and three sites above (S10-S12). Woodland areas shaded in green. Representative photographs of the sample sites are shown. Sites 8 and 9 are within the intervention zone and are situated upstream and downstream of the large woody debris clearly shown in the photograph. Sites 1 and 10 are outside the intervention zone where no LWD was present. Map created in Edina Digimap © Crown copyright and database rights 2020 Ordnance Survey.

2.2 Field Sampling

Sampling took place 3 months after addition of LWD (26th April 2018) and 10 Months after LWD addition (12th November, 2018). At each site three replicate samples were taken across the breadth of the channel. Sampling was carried out following a five-day period of no/low rainfall to ensure the catchment was not exhibiting spate conditions with high flows.

2.2.1 Invertebrate Sampling

Benthic macro-invertebrate samples were collected from the riverbed using a standardised Surber sampler (quadrat size 330mm x 310mm, fitted with 250µm mesh net and screw-thread collecting tub) obtaining fully quantitative sampling size of $0.1m^2$ (Ghani et al., 2016; Everall et al., 2017). Benthic (<5cm deep) and partial hyporheic (>5cm deep) substrate (Magliozzi et al., 2019) was agitated to dislodge organisms into the net. Large stones were held within the net whilst removing organisms attached to the surface. Invertebrate samples were preserved in 70% industrial denatured alcohol (IMS) and transported to the laboratory for identification.

2.2.2 Chemical and Physical Sampling

Dissolved oxygen (mg/l and %), temperature (C), pH and conductivity (µS/cm) were measured using a YSI Profession Plus multimeter. Water samples were filtered through a 0.2µm pore membrane filter for subsequent analysis of water chemistry. Sediment samples were collected to a depth of 5cm from the riverbed using a small metal hand trowel, stored in plastic sealable bags and transported to the laboratory for analysis of particle size. Stream flow rate (m/s) was measured using a GeoPacks Flow Meter1.

2.2 Laboratory Analysis

2.2.1 Invertebrate Identification

Benthic invertebrates were stored in 70% Ethanol at 4°C until processing. Samples were separated through a 250µm sieve, large debris removed, and transferred into a white sorting tray. Invertebrates were removed and preserved in IMS. Identification was carried out to family level (with the exception of Oligocheata) using the identification keys Croft (1986) and Pawley et al., (2011). The Biological Monitoring Working Party (BMWP) and Average Score Per Taxon (ASPT) (Hawkes, 1998), Proportion of Sediment-sensitive Invertebrates (PSI) (Turley et al., 2015) and Lotic-invertebrate Index for Flow Evaluation (LIFE) (Extence et al., 1999) indices were calculated as well as average taxa richness and abundance.

2.2.2 Environmental Laboratory Analysis

Water chemistry (Cl-, NO_2^-, SO_4^{2-}, NO_3^-, PO_4^{3-}, Na, NH_4^+ and K) was measured using ion chromatography (Thermo Scientific Dionex ICS5000+ DC). Sediment samples were transferred into foil trays and dried in an oven at 60^{o}C for 72 hours to remove all residual moisture. Dried samples were then separated into 8 particle size classes (>4mm, >2mm, >1mm, >500µm, >250µm, >125µm, >63µm, <63µm) using an Endecotts Automated Sieve Shaker MINOR 200 for 15 minutes. Each fraction was then individually weighed and converted to a percentage of the overall sample.

2.3 Data analysis

Sampling sites were categorised into 4 groups for analysis - above intervention zone, below intervention zone, upstream of debris dams, and downstream of debris dams - the latter two both situated within the intervention zone where LWD was present. Biometric indices

(BMWP, ASPT, LIFE, PSI) were calculated using SAFIS_v30.0 (Chalkey, 2016). Site/zone differences in biotic indices, abundance and taxa richness were calculated using one-way ANOVA with a post-hoc Tukey HSD in R (R Core Team, 2017). Multivariate analyses were carried out in PRIMER-e (Clarke and Gorley, 2006). Environmental data (excluding sediments) was normalised prior to spatial ordination via Principal Component Analysis (PCA) using R. Macroinvertebrate analyses were carried out at the family level. Family level analyses are commonly used for water quality assessment, while multivariate analysis of family level data can be more interpretable at higher taxonomic levels, without large departures in sensitivity when compared with lower taxonomic levels (Bailey et al., 2001). Invertebrate abundances were square-root transformed and a resemblance matrix created using Bray-Curtis distance. Non-metric multidimensional scaling (nMDS) and hierarchical cluster analysis (group average) were used to graphically analyse the patterns of invertebrate community structure and identify site groupings. Differences between sampling groups identified using nMDS were tested using analysis of similarities (ANOSIM). Taxa driving the dissimilarity of statistically different (using ANOSIM) groups were determined using SIMPER.

3. Results

3.1 Biodiversity and water quality metrics

3.1.1. Taxa Richness and Abundance

A total of 36 samples (12 sites, each with 3 replicate samples) were collected on each sampling occasion. The number of individual invertebrates for each sample ranged between 183 and 452, with a mean number of individuals per sample of 301. These came from a total of 25 taxa.

Figure 2A shows taxa richness across the 4 zones in both sampling occasions. For the April sampling (3 months after LWD addition) there was a highly significant difference (ANOVA, F= 46.12, df= 3, p<0.001) in taxa richness between zones with a Tukey post-hoc test showing a significant increase (p<0.001) between sites outside the intervention zone where no LWD was present (Above IZ = 18 .4 ± 0.4 taxa, Below IZ = 17.3 ± 0.5 taxa), and those where LWD was present (Upstream of LWD = 23.9 ± 0.4 taxa, Downstream of LWD = 22.7 ± 0.6 taxa). The November sampling (10 months after LWD addition) also showed statistically significant differences between sampling zones (p<0.001, F= 55.76, df= 3), with a Tukey post-hoc revealing a significant (p<0.001) increase in richness between sites within the LWD zone (Upstream of LWD= 22.4 ± 0.2, Downstream of LWD= 22.3 ± 0.2 taxa) compared with those above the intervention zone (Above IZ = 18.8 ± 0.4 taxa), with a further significant increase (p<0.001) in richness below the intervention zone (Below IZ = 24.3 ± 0.4 taxa). In both April and November there was no significant differences in taxa richness between sites situated upstream and downstream of LWD within the IZ.

Abundance (Fig. 2B) also showed significant differences between zones in both April (F= 80.5, df= 3, p <0.001) and November (F= 157.6, df= 3, p<0.001). Tukey post hoc analysis showed

April average taxa abundance where LWD was present (Upstream of LWD= 306.4 ± 9.3, Downstream of LWD= 319.1 ± 6.0 individuals) was significantly higher ($p<0.001$) than where LWD was absent (Above IZ=206.3 ± 3.3, Below IZ=207.1 ± 7.3 individuals). In November, abundance within the intervention zone (Upstream of LWD= 398, Downstream of LWD= 432 individuals) was approximately double, and significantly higher than ($p<0.001$, Tukey post-hoc test) above the intervention zone (213.4 ± 2.3 individuals). Although abundance below the intervention zone (328.9 ± 2.6 individuals) was lower than the intervention zone, it was significantly higher ($p<0.001$, Tukey post-hoc test) than those sites situated above the intervention zone. In both April and November there was no significant differences in abundance between sites situated upstream and downstream of LWD within the IZ.

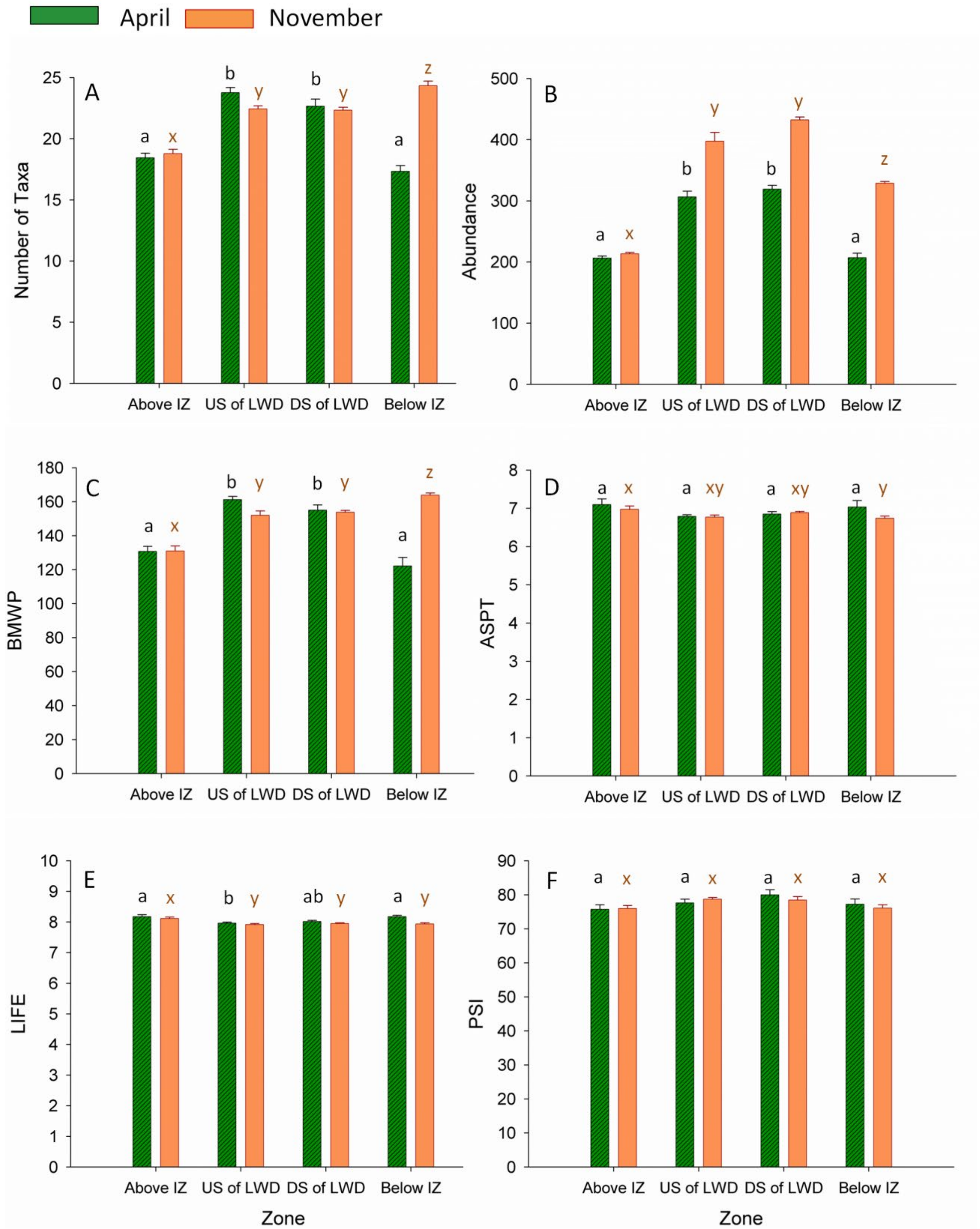


Figure 2: Number of Taxa (A), Total Abundance of Invertebrate pH (B), BMWP (Biological Monitoring Working Party) (C), Average Score Per Taxon (D) and LIFE (Lotic-invertebrate Index for Flow Evaluation) (E) and PSI (Proportion of Sediment-sensitive Invertebrates) (F). Data is presented for each zone – Above the Intervention zone (IZ), and Below the intervention zone where no artificial large woody debris was introduced to the watercourse, and upstream and downstream of LWD dams within the intervention zone. Each bar consists of 3 sampling sites, with each site having 3 replicate samples. All values are means±1SE. Bars that do not share lowercase letters are significantly different ($p < 0.05$) as determined by one-way ANOVA.

3.1.2. Biological Monitoring Working Party Scores (BMWP)

The BMWP score (Fig. 2C) assesses the overall biological quality of the assemblage (Hawkes, 1998). Variability in BMWP scores across the sites showed the same pattern as taxa richness and abundance with scores significantly different across the sampling zones in both April ($F = 30.60$, $df = 3$, $p < 0.001$) and November ($F = 42.65$, $df = 3$, $p < 0.001$). Scores were higher than 130 on all sampling occasions putting them in bracket 'A' of the BMWP ranges (very good biological quality) with the exception of downstream of the intervention zone in April, which was in bracket 'B' (good biological quality). In April, scores within the intervention zone (Upstream of LWD = 161.3±1.8, Downstream of LWD = 155.0 ±3.0) were significantly higher ($p<0.001$, Tukey post-hoc test) than those outside (Above IZ = 130.8 ± 3.0, Below IZ = 122.2 ± 5.0). In November, scores were again significantly higher ($p < 0.001$, Tukey post-hoc test) in the intervention zone (Upstream of LWD = 152.01 ± 2.5; Downstream of LWD = 153.8 ± 1.09) than above (Above IZ = 131.0 ± 3.0). However, a significantly increased BMWP ($p < 0.001$, Tukey post-hoc test) was observed at sites below the intervention zone (163.9 ±1.2) when compared to sites above the intervention zone (131.0±3.0). Overall, the results reveal an improved BMWP score in association with LWD.

3.1.3. Average Score Per Taxon (ASPT)

ASPT ratings (Fig. 2D) across all sites in both sampling seasons were between 6.0-6.9, indicating 'good water quality', except sites upstream of the intervention zone in April, which had a rating of >7 indicating 'very good water quality'. In April, there were no significant differences detected in ASPT scores across the sampling zones. In November the ANOVA did show a significant difference between zones ($F = 3.29$, $df = 3$, $p < 0.05$) though Tukey post-hoc analysis showed the only significant ($p < 0.05$) difference was a higher ASPT above the intervention zone (7.0±0.1) compared with below (6.7±0.06). The increases in the BMWP were therefore driven by increases in taxa richness, rather than increased ASPT scores for those taxa present.

3.1.4. Lotic-invertebrate Index for Flow Evaluation (LIFE)

The LIFE metric (Fig. 2E) ranks assemblages based on the individual taxa preferences for differing flow regimes (Turley et al., 2015). High LIFE scores are linked to a fast flow rate. LIFE scores across all sampling zones in both seasons were greater than 7.5 which indicates the invertebrate assemblages are typical of fast flowing lotic systems. In each sampling zone 22 taxa contributed to the LIFE biometric given their known flow rate requirements; of which 6 typify very fast flows, 9 moderate-fast flow and 7 slow flow conditions.

LIFE Scores were found to be significantly different across the sampling zones in both April ($F = 5.98$, df = 3, $p < 0.01$) and November ($F = 12.01$, df = 3, $p < 0.001$). In April sites within the intervention zone, and upstream of LWD have significantly lower LIFE scores (8.0±0.03) than sites above (8.2±0.06) and below (8.2±0.05) the intervention zones ($p < 0.05$, Tukey post-hoc test), suggesting a reduced flow rate within the intervention zone is causing compositional differences within the macroinvertebrate assemblages. LIFE scores in November were also significantly different across the sampling zones ($F = 12.01$, df = 3, $p < 0.001$), with sites above the intervention zone (8.1±0.05) significantly higher ($p < 0.001$, Tukey post-hoc test) than other zones where LIFE scores were ~7.9.

3.1.5. Proportion of Sediment-sensitive Invertebrates (PSI)

The Proportion of Sediment-sensitive Invertebrates (PSI) is used as a proxy to describe the extent to which the riverbed is impacted by sedimentation from fine silts (Extence et al., 1999). PSI scores (Fig. 2F) were higher than 75 in all sampling zones with a maximum of 80, on both sampling occasions, indicating that the riverbed is slightly impacted by sedimentation. There were 22 taxa which contributed to the PSI calculation, 73% of which are sensitive to sediment (10 taxa highly sensitive, 6 moderately sensitive) whilst 23% are insensitive (6 taxa moderately sensitive, 3 highly insensitive). PSI scores in April showed no significant differences across the sampling zones ($F = 1.59$, $df = 3$, $p > 0.05$). In November however a significant difference was detected ($F = 2.91$, $df = 3$, $p < 0.05$), although a Tukey post-hoc test was insignificant.

3.2. Macroinvertebrate community analysis

NMDS ordination plots showing macroinvertebrate assemblages at the four sampling zones in April are shown in Fig. 3A. In April, macro-invertebrate assemblages from the 4 zones were clustered into two distinct groupings. The first group consisted of sites within the intervention zone (both upstream and downstream of LWD engineered log jams), which clustered closely together, and could not be statistically separated (analysis of similarities, ANOSIM). Sites where LWD was absent (upstream of IZ and downstream of IZ) also showed no significant difference, and together formed a separate, more disparate, grouping from the IZ sites. The resulting two clusters were significantly different (ANOSIM, $R = 0.849$, $p < 0.001$), indicating significant differences in taxa composition where LWD is present. Hierarchical cluster analysis (Figure 3C) also showed a clear separation of sites between those within the intervention zone, and those outside.

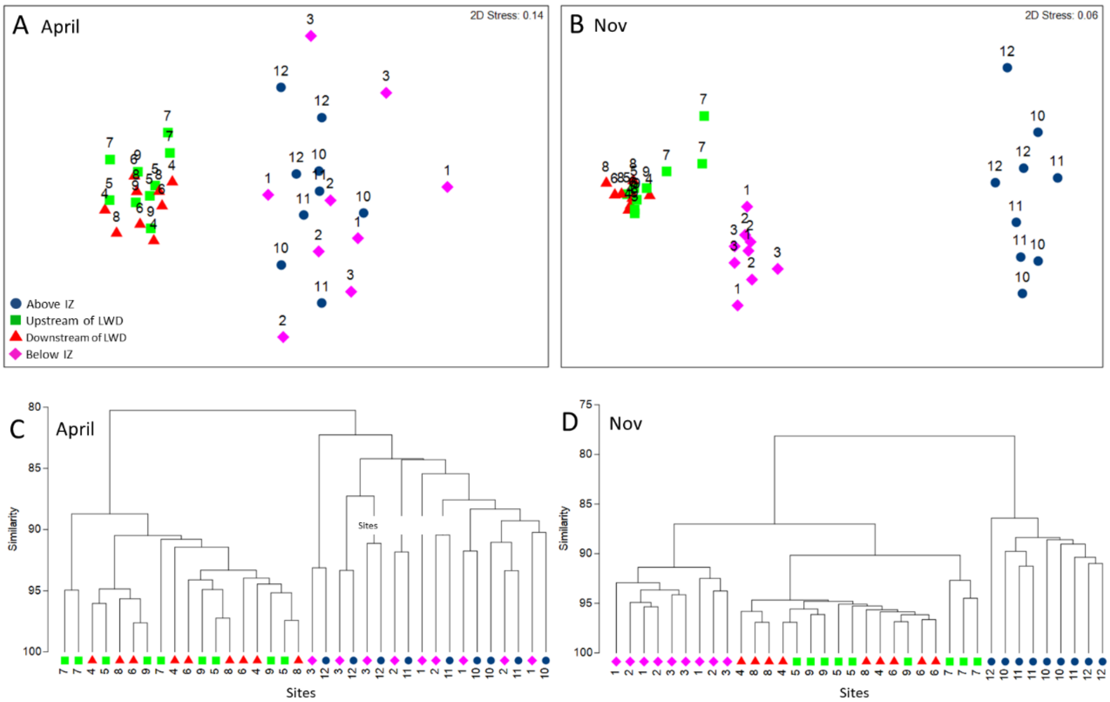


Figure 3: Discrimination of sites on the basis of bacterial community structure. (A) Two-dimensional NMDS plot and (C) Hierarchical clustering of sites in April, based on macroinvertebrates in April, showing a separation between sites within the intervention zone (S4-S9), and those outside the intervention zone (1-3, 1-12); (B) and (D) show the analysis based on the November sampling, showing a separation of sites downstream of the intervention zone. All macroinvertebrate data was √ transformed prior to multivariate analysis.

SIMPER analysis (Table 1) of the April data suggests that differences between the LWD intervention zone and non-intervention zone (above and below IZ) were associated with an increased abundance of all taxa where LWD was present. Leuctridae, Hydraenidae, Simuliidae, Baetidae and Rhyacophilidae were the top 5 taxa contributing to dissimilarity between the sampling zones, which cumulatively contributed 35.5% towards the dissimilarity between the intervention and non-intervention reach. Particularly notable was Hydraenidae, Rhyacophilidae, Scirtidae, and Elmidae which were absent in the non-intervention zone but present in the intervention zone.

Table 1: SIMPER analysis of April Data contributing to % dissimilarity in macroinvertebrate assemblage composition between intervention (where LWD is present (u, d)) and non-intervention (where LWD is absent (a, b)) sampling zones. The % contribution each taxa has on differentiation of sites is shown in column 4 with cumulative % value in column 5. Mean abundance are shown as square-root-log transformed values. Average dissimilarity = 19.72

Taxa	**Mean abundances**			
	Intervention zone	**Non-intervention zones**	**Cont. %**	**Cumul %**
Leuctridae	8.36	6.50	8.30	8.30
Hydraenidae	1.84	0.00	7.88	16.19
Simuliidae	3.01	1.27	7.49	23.68
Baetidae	7.81	6.36	6.53	30.21
Rhyacophilidae	1.25	0.00	5.33	35.54
Ephemeridae	7.02	5.82	5.26	40.80
Scirtidae	1.05	0.00	4.49	45.29
Pediciidae	1.82	0.81	4.37	49.66
Elmidae	0.99	0.00	4.21	53.87
Glossosomatidae	4.31	3.42	3.94	57.82
Chloroperlidae	1.91	1.14	3.58	61.39
Perlodidae	2.20	1.58	3.48	64.87
Odontoceridae	1.67	0.89	3.4	68.27
Tipulidae	1.37	1.07	3.25	71.53

In the November sampling (Fig. 3B), the four zones showed 3 distinct clusters on the NMDS plot, with the previously overlapping non-intervention sites (Upstream of IZ and downstream of IZ) now significantly different from each (ANOSIM, $R = 0.994$, $p < 0.001$). Sites below the intervention zone now appear to be more similar to those within the intervention zone. SIMPER analysis (Table 2) showed that the differences in sites below the intervention zone was likely driven by increases in Baetidae, Leuctridae, Simuliidae, Hydraenidae and Ephemeridae, which cumulatively accounted for 35.3% of dissimilarity. In addition, a number of families that were absent above the intervention zone were now present below.

Table 2: SIMPER analysis of the November data contributing to % dissimilarity in macroinvertebrate assemblage composition between assemblages from sampling zones above the LWD interventions, and below the intervention zone (both zones where the LWD was absent). The % contribution each taxa has on differentiation of sites is shown in column 4 with cumulative % value in column 5. Mean abundance are shown as square-root-log transformed values. Average dissimilarity = 18.09.

Taxa	Mean abundances			
	Above intervention	Below intervention	Contribution %	Cumulative %
Baetidae	6.40	8.17	7.94	7.94
Leuctridae	6.74	8.46	7.70	15.64
Simuliidae	1.31	2.83	6.82	22.46
Hydraenidae	0.11	1.55	6.54	29.00
Ephemeridae	5.89	7.29	6.27	35.27
Rhyacophilidae	0.00	1.35	6.03	41.30
Elmidae	0.00	1.27	5.66	46.95
Scirtidae	0.00	1.15	5.15	52.10
Heptgeniidae	0.95	1.93	4.41	56.51
Pediciidae	1.15	2.06	4.07	60.58
Perlidae	0.76	1.44	3.34	63.93
Odontoceridae	1.07	1.76	3.26	67.18
Planorbidae	0.00	0.71	3.17	70.35

In November, although the sites immediately upstream and downstream of LWD appeared to group closely in the NMDS, ANOSIM showed a minor but significant difference (ANOSIM, R-statistic = 0.158, $p < 0.01$). This difference (Table 3) was largely driven an increase in abundance of Oligochaeta and Tipulidae upstream of LWD log jams, whilst Dytiscidae, Leuctridae and Baetidae were all in greater abundance immediately downstream. Differences between zones amongst these five taxa cumulatively accounted for 34.57% of dissimilarity. Hierarchical Cluster Analysis (Fig. 3D) showed that of the sites immediately upstream of LWD, the statistical difference was largely driven by site 7 which showed the slowest flow, and finest particle size. It is possible that other sites immediately upstream of LWD will develop in a similar way, with finer particulate size and slower flow, leading to further niche availability and habitat heterogeneity within the intervention zone.

Table 3: SIMPER analysis of the November data comparing macroinvertebrate composition between assemblages immediately upstream and downstream of large woody debris dams. The % contribution each taxa has on differentiation of sites is shown in column 4 with cumulative % value in column 5. Mean abundance are shown as square-root-log transformed values. Average dissimilarity = 6.73.

Taxa	Mean Abundances Downstream of Dam(d)	Mean Abundances Upstream of Dam (u)	Contribution%	Cumulative %
Dytiscidae	3.25	2.70	8.03	8.03
Leuctridae	8.91	8.29	7.09	15.12
Oligochaeta	0.71	1.24	6.60	21.72
Baetidae	9.42	9.06	6.49	28.20
Tipulidae	0.79	1.20	6.36	34.57
Perlidae	1.27	1.16	6.03	40.60
Hydraenidae	5.19	4.81	5.28	45.89
Taeniopterygidae	5.59	5.06	5.11	51.00
Nemouridae	1.53	1.15	4.82	55.81
Gammaridae	3.02	3.20	4.53	60.34
Scirtidae	3.04	3.05	4.13	64.47
Chironomidae	3.27	3.13	3.67	68.14
Perlodidae	2.34	2.14	3.29	71.43

3.3. Analysis of abiotic factors

3.3.1. Water quality

On each sampling occasion, chemical water quality parameters remained relatively consistent between each sampling site. Dissolved oxygen was between 100-110% saturation, pH varied between 7.2 and 7.6, and conductivity between 100 and 150µS/cm. Phosphate was 0.01mg/l, and Nitrate ~2.2mg/l with ammonium and nitrite undetectable. These values showed little variation between sites, and there was little clustering of the different sampling zones (Fig. 4, Tables S2, S3), though the different sampling occasions did form separate grouping on the PCA, indicating seasonal changes in water quality.

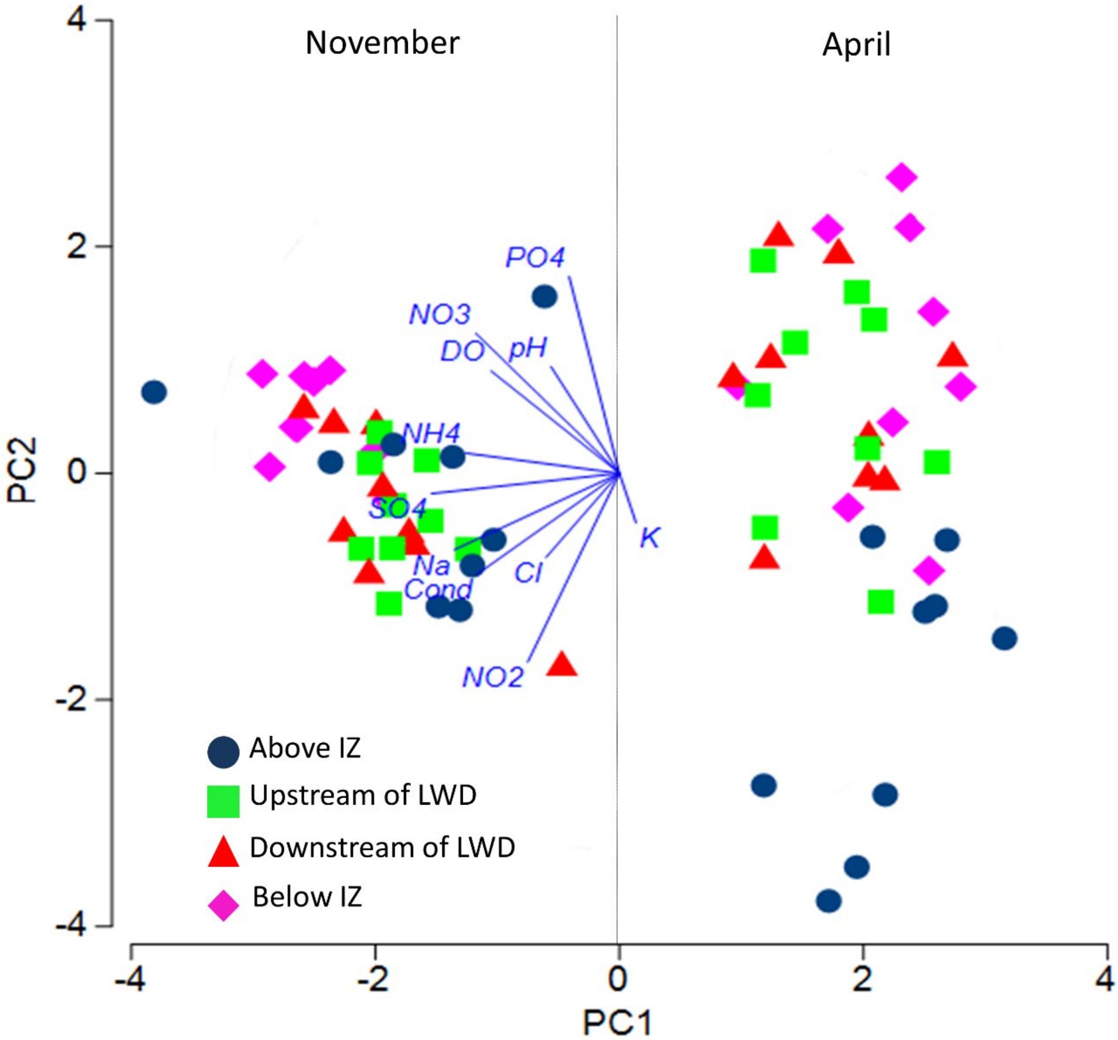


Figure 4: Discrimination of sites on the basis of physicochemical parameters using PCA. Clustering reveals clear seasonal differences with April samples situated positively on PC1, and November samples negatively. Little discrimination was observed between sites and zones. Data was normalised prior to PCA ordination. PC1 (Table 10), which accounts for 41.5% of variance. PC2 accounted for 17% of variance

3.3.2. Sediment Analysis

PCA analysis of the sediment dataset (Fig. 5) showed a gradient along PC1 from fine silt (<63µm) to course gravels and pebbles (>4mm), with sites containing higher percentages of fine sediment (<63µm) positively loaded on PC1, whilst sites with higher percentages of larger sediment (>4mm) are positively loaded. Sites immediately upstream of LWD dams contained a greater percentage of fine silts and sediments.

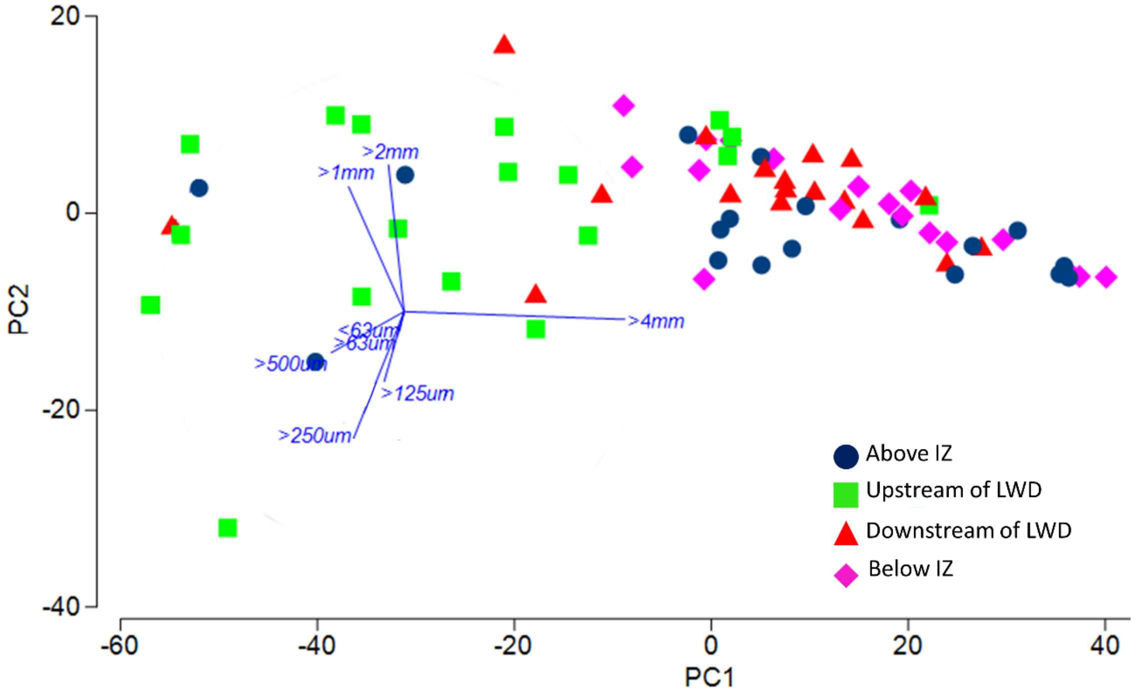


Figure 5: Discrimination of sites on the basis of sediment size classes using PCA, with both sampling occasions combined. Sites immediately upstream of LWD dams have a greater percentage of fine sediment than those immediately downstream of LWD, and sites above and below the intervention zone. Figure shows data from both sampling occasions. PC1 accounts for 90.3% of variance, PC2 accounts for 7% of variance.

4. Discussion

The use of LWD for natural flood management is designed to alter hydrological processes to reduce downstream peak flows; however, the introduction of LWD will also alter habitat heterogeneity, which in turn may impact macroinvertebrates. Macroinvertebrates are also impacted by water quality, however in this study although seasonal differences in water chemistry were observed, on each sampling occasion water quality remained homogenous across sites and sampling zones, indicating that water chemistry was not driving the observed changes in biological communities, and the observed differences were due to the addition of LWD.

Previous research has established that LWD improves hydraulic roughness and complexity of stream systems, enabling the natural dynamics of sediment mobilisation, transport and deposition to function efficiently (Gurnell, 2007). Pilotto et al. (2014) also found that LWD altered channel depth, width and velocity regimes. This is also reflected here in an increase in fine silts immediately upstream of LWD. This pattern of sediment drop-out is comparable with known observations of sediment accumulations in slow flow regimes designed into NFM features (Janes et al., 2017), and comparable to natural LWD accumulation in lotic systems (Gurnell et al., 2005). Flow data supports this by indicating a reduced flow within the intervention reach; slower flowing eddies and pools allow suspended solids to drop out of the water column (Johnson et al., 2003). The improved sediment transfer function of the stream allows spatially variable sediment deposition.

These changes in the physical conditions of stream systems can lead to changes in the macroinvertebrate communities. Previous research by Pilotto et al. (2014) has shown that areas with LWD have higher organic matter content, but also increased taxonomic richness and diversity. Similar results have been found in terms of the overall invertebrate community composition in areas with LWD, which is attributed to increased heterogeneity of habitats (Osei et al., 2015). In this study, the changes in habitat diversity and hydrological dynamics in intervention zones have led to positive changes in the biotic communities surveyed. We observed significant differences in macroinvertebrate abundance, taxonomic richness and BMWP scores, as well as the community assemblages. Castro and Thorne (2019) proposed that these changes in the biotic components of stream systems can lead to subsequent further alterations in geomorphology and hydrology; a ‘stream evolution triangle’ where all elements are interlinked. Thus, natural flood management interventions may have long-term sustained benefits for steam ecosystems, something which is key for successful restoration of river systems (Gilvear et al., 2013).

In this study the macroinvertebrate communities associated in the intervention zone have changed significantly. NMDS and cluster analysis indicated distinct macroinvertebrate assemblages, the first associated with areas where LWD was present, and the second in areas without these interventions. These findings suggest that the changes in invertebrate communities are similar to those found in association with areas with naturally occurring LWD (Gurnell et al., 2005). Indeed, river surveys by Johnson et al. (2003) showed that around 90% of all aquatic invertebrate taxa recorded were associated with woodland habitats in their survey areas, suggesting that such habitats support far more species that those systems adjacent to non-wooded areas. In this study the study site was situated within a woodland habitat, yet the addition of LWD resulted in increased macroinvertebrate numbers and richness. Similar interventions (addition of LWD for flood management) in non-woody areas may therefore give rise to even greater improvements in macroinvertebrate diversity than those seen here.

Although on the first sampling occasion (3 months after LWD addition) no significant differences in macroinvertebrate composition were observed immediately above and below LWD dams, by November (10 months after LWD added) there was a differentiation between macroinvertebrates at sites immediately upstream and downstream of LWD. These results indicate a temporal change in the habitat and associated macroinvertebrates after LWD addition. Taxa which prefer faster following riffles, such as Batidae and Leuctridae, were found in greater numbers immediately downstream of LWD compared with upstream, whilst taxa adapted to slower flows, such as Tipulidae and Oligocheata, were found in great abundance immediately upstream, evidencing possible niche diversification within the channel as a result of LWD.

These changes in invertebrate community composition are therefore likely due to the engineered LWD enhancing the previously uniform habitat structure by creating a ‘pooling’ effect on the upstream side of a log jam and a ‘riffle’ effect on the downstream side, where flow is temporarily increased enabling greater surface mixing (Johnson *et al*., 2003). These pool and riffle niches were interspersed with ‘runs’ where water flowed unimpeded to the next LWD dam leading to enhanced habitat complexity and a more diverse flow regime. Evidence for the impact of changing flow (and sedimentation) can be seen from analysis of the LIFE metric (Turley et al., 2015) which ranks assemblages based on the individual taxa preferences for differing flow regimes and the Proportion of Sediment-sensitive Invertebrates (PSI) metric (Extence et al., 1999) which is used as a proxy to describe the extent to which the riverbed is impacted by sedimentation from fine silts. The LIFE biometric was >7.5 in all zones, typical of fast flowing headwater streams with small but significant decreases where LWD is present, suggestive of a macroinvertebrate response to reduced flow rate due to LWD. PSI scores show macroinvertebrate assemblages are indicative of slightly sedimented riverbed habitat. Although no significant difference in PSI scores between sites immediately above and below LWD dams was observed in the April sample, there was in the second sampling (November), suggesting that over time fine silts and sediment settle out more readily where LWD is present. Conclusions inferred by the results of PSI analysis are supported by changes in relative percentages of sediment fractions, with smaller sediment sizes present immediately above LWD dams.

In addition to changes within the intervention zone, it was notable that on the second sampling (November) sites downstream of the intervention zone also showed a marked improvement despite no LWD being present. Both community composition and richness, BMWP and abundance became similar to the LWD sites. This downstream improvement has also been observed by Pilotto et al. (2014) who found that LWD used in river restoration significantly affected macroinvertebrate communities, sediment deposition and organic matter downstream of the additions.

The overall findings from this study on the use of LWD for natural flood management support previous research demonstrating the positive affect of natural and introduced woody debris on macroinvertebrates and water quality (Everall et al., 2012; Janes et al., 2017, Pilotto et al., 2014; Flores et al., 2017). This study explicitly demonstrate the impacts of artificially introduced channel-spanning LWD log jams on macroinvertebrate communities, showing increased richness and abundance of invertebrates and strengthening the ecological integrity of the water course (Everall et al., 2012; Spänhoff and Arle, 2007). Although the findings observed during this study reflect short-term changes arising following the reintroduction of LWD, it is likely that benefits will persist over a longer timescale due to the increase in biocomplexity of the habitat and niche creation as a result of LWD in the watercourse (Gilvear et al., 2013).

In this study, only the effect on macroinvertebrates was studied, and although the wider ecosystem effects of the LWD interventions have not been assessed, other studies have shown biodiversity benefits are not limited to the invertebrate communities, and LWD intervention can positively impact fish populations and food web connectivity, helping to restore human-impacted river ecosystems across multiple trophic levels (Howsen et al., 2012; Thompson et al., 2018). Aside from the benefits to ecosystems, the addition of LWD also has evidenced benefits for the enhancement of ecosystem services including flood alleviation, reducing soil loss into water systems, as well as providing additional ecosystem services by means of improved water quality and carbon sequestration (Iacob et al., 2014; Walling, 2006).

5. Conclusions

In this study LWD was introduced into an upland stream for the purpose of flood management. Within 12 months of introducing LWD positive benefits on macroinvertebrates abundance and taxa richness, and overall water quality biometrics, were observed in comparison to control areas with no addition of LWD. The results presented here support those findings where LWD interventions were specifically implemented for habitat improvements, and are also comparable to those detailing the ecological benefits of naturally occurring LWD. These benefits observed in this study are in addition to those relating to changing hydrological flow regimes and reducing peak downstream flow, which, in this case, was the principal rationale for the introduction of LWD debris. While this study utilised a headwater stream in the Peak District, UK, results may be comparable to catchments of similar land use, hydrology and geology, though further research is needed to determine if the results are repeated at a wider geographical scale. Further research would also determine longer terms changes to the ecosystem, and the impact of LWD intervention on the wider riverine ecosystem. Overall, this study demonstrate that biological complexity and niche availability increased within the in-channel zone as a result of introducing LWD for flood management, revealing the wider aquatic habitat improvement potential of such natural flood management approaches. The use of LWD as an intervention for flood management is recommended for its additional benefits for ecosystem health and biodiversity enhancement.

Declaration of Competing Interests

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

Thank you to Staffordshire Wildlife Trust for access to study site and to Cheshire Wildlife Trust and the South West Peak Landscape Partnership funding the restoration work, and Mercia Tree Care for carrying out the LWD dam creation through tree felling.

Appendix A. Supplementary data

Supplementary data to this article can be found online.

References

Bailey, R.C., Norris, R.H. and Reynoldson, T.B., 2001. Taxonomic resolution of benthic macroinvertebrate communities in bioassessments. Journal of the North American Benthological Society, 20(2), pp.280-286.

Carpenter, S, R., Stanley, E, H., Vander, Z., Jake, M., 2011. State of the world's freshwater ecosystems: Physical, chemical, and biological changes. Annual Review of Environment and Resources, 36, pp. 75-99.

Castro, J.M. and Thorne, C.R., 2019., The stream evolution triangle: Integrating geology, hydrology, and biology. River Research and Applications, 35, pp.315-326.

Chalkey, A., 2016. SAFIS: Site Analysis for Freshwater Invertebrate Surveys. Version 30.0. [Software] Boxvalley Aqua Surveys software available via the author at safis@boxvalley.co.uk. [Accessed May 2019]

Cheshire Wildlife Trust., 2017. Slowing the Flow Project Plan. Unpublished.

Clarke, K, R., Gorley, R, N., 2006. PRIMER-E v6: User Manual/ Tutorial. Version 6. [Software] Plymouth, England: PRIMER-E Ltd. [Accessed May 2019]

Croft, P, S., 1986. A key to the major groups of British freshwater invertebrates. Field Studies Council Publications.

Dadson, S., Hall, J. H., Murgatroyd, A., Acreman, A., Bates, P., Beven, K., Heathwaite, L., Holden, J., Holman, I.P., Lane, S. N., O' Connell, E., Penning-Rowsell, E., Reynard, N., Sear, D., Thorne, C., Wilby, R., 2017. A restatement of the natural science evidence concerning catchment-based ‘natural’ flood management in the UK. Proceeding of the Royal Society A, 473, 20160706.

Elosegi, A., Elorriaga, C., Flores, L., Martí, E. and Díez, J., 2016. Restoration of wood loading has mixed effects on water, nutrient, and leaf retention in Basque mountain streams. Freshwater Science, 35(1), pp.41-54.

Everall, N, C., Farmer, A., Heath, A, F., Jacklin, T, E., Wilby, R, L., 2012. Ecological benefits of creating messy rivers. Area, 44(4), pp. 470–478.

Everall, N, C., Johnson, M, F., Wood, P., Farmer, A., Wilby, R, L., Meesham, N., 2017. Comparability of macroinvertebrate biomonitoring indices of river health derived from semi-quantitative and quantitative methodologies. Ecological Indicators, 78, pp. 437-448.

Extence, C.A., Balbi, D.M., and Chadd, R.P., 1999. River flow indexing using British benthic macroinvertebrates: a frame- work for setting hydroecological objectives. Regulated Rivers: Research & Management, 15, pp. 545–574

Flores, L., Giorgi, A., González, J.M., Larrañaga, A., Díez, J.R. and Elosegi, A., 2017. Effects of wood addition on stream benthic invertebrates differed among seasons at both habitat and reach scales. Ecological Engineering, 106, pp.116-123.

Ghani, W.M.H.W.A., Rawi, C.S.M., Hamid, S.A., Al-Shami, S.A., 2016. Efficiency of different sampling tools for aquatic macroinvertebrate collections in Malaysian streams. Tropical life sciences research, 27(1), pp. 115-133.

Gilvear, D, J., Spray, C, J., Casas-Mulet, R., 2013. River rehabilitation for the delivery of multiple ecosystem services at the river network scale. Journal of Environmental Management, 126, pp. 30–43.

Gippel, C, J., Neill, I, C., Finlayson, B, L., Schnatz, I., 1996. Hydraulic guidelines for the re-introduction and management of large woody debris in lowland rivers. *Regulated Rivers-Research & Management* 12, pp 223–236.

Gregory, S., Boyer, K., Gurnell, A, M. (Eds)., 2003. *The ecology and management of wood in world rivers.* Bethesda Maryland, American Fisheries Society.

Gurnell, A., 2007. Analogies between mineral sediment and vegetative particle dynamics in fluvial systems. Geomorphology, 89, pp. 9-20.

Gurnell, A., Tockner, K., Edwards, P., Petts, G., 2005. Effects of Deposited Wood on Biocomplexity of River Corridors. Frontiers in Ecology and the Environment, 3(7), pp. 377-382.

Hawkes, H.A., 1998. Origin and development of the biological monitoring working party score system. Water Research. 32, pp. 964–968.

Holmes, N., Raven, P. (2014) Rivers. Baydon, Wiltshire: D & N Publishing.

Howsen, T, J., Robson, B, J., Mathews, T, G., Mitchell, B, D., 2012. Size and quantity of wood debris effects fish populations in a sediment-disturbed lowland river. Ecological Engineering, 40, pp. 144-152.

Iacob, O., Rowan, J, S., Brown, I., Ellis, C., 2014. Evaluating wider benefits of natural flood management strategies: an ecosystem-based adaptation perspective. Hydrology Research, 45, pp. 774–787.

Janes, V. J., Grabowski, R. C., Mant, J., Allen, D., Morse, J. L., Haynes, H., 2017. The Impacts of Natural Flood Management Approaches on In-Channel Sediment Quality. River Research and Applications, 33, pp. 89–101.

Johnson, L, B., Breneman, D, H., Richards, C., 2003. Macroinvertebrate community structure and function associated with large wood in low gradient streams. River Research and Applications, 19, pp. 211–18.

Kail, J. and Hering, D., 2005. Using large wood to restore streams in Central Europe: potential use and likely effects. Landscape ecology, 20(6), pp.755-772.

Kenney, M.A., Sutton-Grier, A.E., Smith, R.F. and Gresens, S.E., 2009. Benthic macroinvertebrates as indicators of water quality: The intersection of science and policy. Terrestrial Arthropod Reviews, 2(2), p.99.

Lane, S., 2017. Natural Flood Management. WIREs Water, 4(3), p.e1211.

Lavers, T., Charlesworth, S., 2017. Opportunity mapping of natural flood management measures: a case study from the headwaters of the Warwickshire-Avon. Environmental Science and Pollution Research, 25(20), pp. 19313-19322

Magliozzi., C., Usseglio-Polatera, P., Meyer, A., Grabowsk, R, C., 2019. Functional traits of hyporheic and benthic invertebrates reveal importance of wood-driven geomorphological processes in rivers. Functional Ecology, 33, pp. 1758–1770.

Mainstone, C., Holmes, P., 2010. Embedding a strategic approach to river restoration in operational management processes: experiences in England. Aquatic Conservation: Marine and Freshwater Ecosystems, 23, pp. 82–95.

Nicholson, A., Wilkinson, M., O’Donnell, G., Quinn, P., 2012. Runoff attenuation features: a sustainable flood mitigation strategy in the Belford catchment, UK. Area, 44(4), pp. 463–469.

Osei, N, A., Gurnell, A., Harvey, G, L., 2015. The role of large wood in retaining fine sediment, organic matter and plant propagules in a small, single-thread forest river. Geomorphology, 235, pp. 77-87.

Palmer, M.A., Menninger, H.L. and Bernhardt, E., 2010. River restoration, habitat heterogeneity and biodiversity: a failure of theory or practice? Freshwater Biology, 55, pp.205-222.

Pawley, S., Dobson, M., Fletcher, M., 2011. Guide to British freshwater macroinvertebrates for biotic assessment. Freshwater. No 67. Ambleside: The Freshwater Biological Association.

Pilotto, F., Bertoncin, A., Harvey, G, L., Wharton, G., Pusch, M, T., 2014. Diversification of stream invertebrate communities by large wood. Freshwater Biology, 59, pp. 2571–2583.

Pitt, M., 2008. ‘The Pitt Review: Lessons learned from the 2007 floods. https://webarchive.nationalarchives.gov.uk/20100812084907/http://archive.cabinetoffice.gov.uk/pittreview/_/media/assets/www.cabinetoffice.gov.uk/flooding_review/pitt_review_full%20pdf.pdf [Accessed May 2019].

Raven, P, J., Holmes, N, T, H., Dawson, F, H., Fox, P, J, A., Everard, M., Fozzard, I, R., Rouen, K, J., 1998. River Habitat Quality: The Physical Character of Rivers and Streams in the UK and Isle of Man. Environment Agency: Bristol.

R Core Team., 2017. R: A language and environment for statistical computing. R Foundation for Statistical Computing. [Software] Vienna, Austria. http://www.R-project.org/ [Accessed Jun 2019]

Sear, D, A., Wilcock, D., Robinson, M, R., Fisher, K, R., 2000. Channel modifications and impacts. In The Changing Hydrology of the UK. Acreman, M, C. (ed.). London: Routledge.

SEPA., 2016. Natural flood management handbook. Edinburgh: Scottish Environment Protection Agency

Spänhoff, B., Arle, J., 2007. Setting attainable goals of stream habitat restoration from a macroinvertebrate view. Restoration Ecology, 15, pp. 317–20.

Thompson, M.S., Brooks, S.J., Sayer, C.D., Woodward, G., Axmacher, J.C., Perkins, D.M. and Gray, C., 2018. Large woody debris “rewilding” rapidly restores biodiversity in riverine food webs. Journal of Applied Ecology, 55(2), pp.895-904.

Thorne, C., 2014. Geographies of UK flooding in 2013/4. The Geographical Journal, 180, pp. 297–309.

Turley, M.D., Bilotta, G.S., Krueger, T., Brazier, R.E. and Extence, C.A., 2015. Developing an improved biomonitoring tool for fine sediment: combining expert knowledge and empirical data. Ecological Indicators, 54, pp.82-86.

Walling, D, E., 2006. Human impact on land–ocean sediment transfer by the world's rivers. Geomorphology, 79, pp. 192-216.

Wilkinson, M, E., Addy, S., Quinn, P, F., Stutter, M., 2019. Natural flood management: small-scale progress and larger-scale challenges. Scottish Geographical Journal, 135(1-2), pp.23-32.

Wingfield, T., Macdonald, N., Peters, K., Spees, J., Potter, K., 2019. Natural Flood Management: Beyond the evidence debate. Area, 51(4), pp.743-751.

Appendix A: Supplementary Tables and Figures

Table S1: Locations of sampling sites (river and wetland) on Black Brook, Staffordshire, UK. A map of all sites is shown in Figure 1.

Site	Description	Grid Reference
1	Downstream of Intervention Zone	SJ9903165797
2	"	SJ9908265632
3	"	SJ9911465606
4	Immediately below LWD dam	SJ9916065511
5	Immediately above LWD dam	SJ9917765511
6	Immediately below LWD dam	SJ9917265487
7	Immediately above LWD dam	SJ9916865474
8	Immediately below LWD dam	SJ9923565406
9	Immediately above LWD dam	SJ9925765407
10	Upstream of intervention Zone	SJ9967665012
11	"	SJ9968464993
12	"	SJ9969264989

Table S2: Water Chemistry and Flow for April Sampling. All data are means (each 3 sites, 3 replicates, n=9) ±1SE.

Site	DO (%)	DO (mg/L)	Temp (^{o}C)	pH	Cond (µS)	NO_3 (mg/L)	NO_2 (mg/l)	PO_4 (mg/L)	Na (mg/l)	NH_4 (mg/L)	K (mg/L)	Cl (mg/L)	SO_4 (mg/L)	Flow (m/s)
Site 1	105.83 ±0.12	12.76 ±0.03	7.20 ±0.00	7.20 ±0.06	102.00 ±0.00	2.17 ±0.02	0.00 ±0.00	0.00 ±0.00	8.52 ±0.24	0.00 ±0.00	1.34 ±0.56	14.81 ±0.10	8.05 ±0.07	0.57 ±0.06
Site 2	105.60 ±0.17	12.69 ±0.05	7.27 ±0.07	7.27 ±0.03	102.67 ±0.67	2.16 ±0.00	0.00 ±0.00	0.00 ±0.00	8.41 ±0.16	0.00 ±0.00	1.33 ±0.56	14.68 ±0.07	8.00 ±0.06	0.64 ±0.02
Site 3	104.97 ±0.49	12.60 ±0.06	7.33 ±0.07	7.30 ±0.00	102.67 ±0.67	2.15 ±0.01	0.00 ±0.00	0.00 ±0.00	8.20 ±0.22	0.00 ±0.00	0.74 ±0.04	14.61 ±0.11	8.00 ±0.05	0.86 ±0.21
Site 4	104.93 ±0.47	12.59 ±0.05	7.40 ±0.00	7.33 ±0.03	103.33 ±0.67	2.14 ±0.01	0.00 ±0.00	0.00 ±0.00	8.34 ±0.23	0.00 ±0.00	0.76 ±0.04	14.44 ±0.08	7.92 ±0.04	0.99 ±0.18
Site 5	105.03 ±0.52	12.57 ±0.05	7.47 ±0.07	7.38 ±0.04	103.00 ±0.58	2.15 ±0.02	0.00 ±0.00	0.00 ±0.00	8.17 ±0.19	0.00 ±0.00	0.72 ±0.04	14.54 ±0.17	7.96 ±0.08	0.90 ±0.26
Site 6	105.23 ±0.32	12.56 ±0.05	7.53 ±0.07	7.43 ±0.02	103.33 ±0.33	2.18 ±0.03	0.00 ±0.00	0.01 ±0.00	8.52 ±0.23	0.00 ±0.00	0.75 ±0.04	14.79 ±0.26	8.06 ±0.11	0.61 ±0.21
Site 7	103.67 ±1.46	12.34 ±0.17	7.60 ±0.00	7.47 ±0.02	103.67 ±0.67	2.21 ±0.01	0.00 ±0.00	0.01 ±0.00	8.34 ±0.30	0.00 ±0.00	0.71 ±0.03	15.02 ±0.08	8.19 ±0.04	0.32 ±0.08
Site 8	103.87 ±1.60	12.32 ±0.16	7.83 ±0.23	7.53 ±0.06	104.33 ±0.67	2.16 ±0.05	0.00 ±0.00	0.01 ±0.00	8.67 ±0.36	0.00 ±0.00	0.79 ±0.07	15.27 ±0.18	8.14 ±0.09	0.29 ±0.07
Site 9	104.20 ±1.71	12.30 ±0.15	8.07 ±0.23	7.58 ±0.04	104.67 ±0.33	2.11 ±0.05	0.00 ±0.00	0.01 ±0.00	8.43 ±0.35	0.00 ±0.00	0.76 ±0.08	15.45 ±0.22	8.09 ±0.08	0.28 ±0.06
Site 10	106.03 ±0.22	12.46 ±0.03	8.30 ±0.00	7.62 ±0.02	104.33 ±0.33	2.06 ±0.00	0.00 ±0.00	0.01 ±0.00	8.81 ±0.30	0.00 ±0.00	0.78 ±0.08	15.60 ±0.06	7.99 ±0.05	0.33 ±0.01
Site 11	105.47 ±0.52	12.33 ±0.13	8.43 ±0.13	7.57 ±0.03	105.00 ±1.00	2.06 ±0.01	0.00 ±0.00	0.01 ±0.00	8.81 ±0.31	0.00 ±0.00	0.73 ±0.03	15.64 ±0.08	8.00 ±0.05	0.29 ±0.06
Site 12	105.47 ±0.52	12.30 ±0.12	8.57 ±0.13	7.55 ±0.04	106.00 ±1.00	2.05 ±0.00	0.00 ±0.00	0.01 ±0.00	9.17 ±0.06	0.00 ±0.00	0.76 ±0.00	15.66 ±0.08	7.97 ±0.03	0.30 ±0.06

Table S3: Water Chemistry and Flow for November Sampling. All data are means (each 3 sites, 3 replicates, n=9) ±1SE.

Site	DO (%)	DO (mg/L)	Temp (°C)	pH	Cond (µS)	NO_3 (mg/L)	NO_2 (mg/l)	PO_4 (mg/L)	NH_4 (mg/L)	Na (mg/l)	K (mg/L)	Mg (mg/L)	Ca (mg/L)	Cl (mg/L)	SO_4 (mg/L)	Flow (m/s)
Site 1	108.00 ±0.61	13.16 ±0.06	6.93 ±0.03	7.60 ±0.00	143.10 ±0.00	2.50 ±0.04	0.01 ±0.00	0.01 ±0.00	0.04 ±0.01	10.04 ±0.20	0.87 ±0.07	2.85 ±0.02	10.96 ±0.07	15.17 ±0.24	23.72 ±0.40	0.37 ±0.03
Site 2	108.30 ±0.40	13.16 ±0.05	6.97 ±0.03	7.63 ±0.03	143.20 ±0.10	2.51 ±0.04	0.01 ±0.00	0.01 ±0.00	0.06 ±0.00	10.03 ±0.19	0.88 ±0.07	2.87 ±0.04	11.04 ±0.14	15.23 ±0.23	23.78 ±0.39	0.37 ±0.03
Site 3	108.10 ±0.20	13.17 ±0.06	7.00 ±0.00	7.63 ±0.03	143.30 ±0.10	2.51 ±0.04	0.01 ±0.00	0.01 ±0.00	0.05 ±0.00	9.97 ±0.21	0.84 ±0.04	2.87 ±0.04	11.02 ±0.14	15.25 ±0.21	23.82 ±0.36	0.40 ±0.06
Site 4	108.43 ±0.29	13.21 ±0.05	7.00 ±0.00	7.63 ±0.03	143.40 ±0.00	2.48 ±0.01	0.01 ±0.00	0.01 ±0.00	0.05 ±0.00	10.06 ±0.17	0.85 ±0.05	2.91 ±0.03	11.17 ±0.11	15.10 ±0.06	23.51 ±0.13	0.53 ±0.09
Site 5	108.80 ±0.15	13.24 ±0.02	7.00 ±0.00	7.60 ±0.00	142.63 ±0.77	2.47 ±0.01	0.01 ±0.00	0.01 ±0.00	0.05 ±0.00	9.97 ±0.10	0.90 ±0.01	2.90 ±0.03	11.15 ±0.10	15.10 ±0.06	23.40 ±0.09	0.53 ±0.09
Site 6	108.03 ±0.92	13.10 ±0.13	7.00 ±0.00	7.60 ±0.00	141.87 ±0.77	2.51 ±0.03	0.01 ±0.00	0.01 ±0.00	0.06 ±0.00	9.96 ±0.11	0.90 ±0.02	2.90 ±0.03	11.12 ±0.12	15.29 ±0.14	23.79 ±0.32	0.53 ±0.09
Site 7	106.33 ±1.50	12.87 ±0.20	7.00 ±0.00	7.60 ±0.00	141.10 ±0.00	2.50 ±0.03	0.01 ±0.00	0.01 ±0.00	0.06 ±0.00	9.94 ±0.10	0.90 ±0.02	2.90 ±0.03	11.14 ±0.13	15.27 ±0.15	23.75 ±0.33	0.40 ±0.06
Site 8	100.37 ±4.68	12.12 ±0.58	7.13 ±0.13	7.40 ±0.20	141.77 ±0.67	2.48 ±0.05	0.01 ±0.00	0.01 ±0.00	0.06 ±0.00	9.91 ±0.08	0.90 ±0.02	2.90 ±0.03	11.14 ±0.13	15.25 ±0.16	23.75 ±0.34	0.30 ±0.12
Site 9	99.27 ±4.09	11.98 ±0.51	7.17 ±0.12	7.33 ±0.18	142.43 ±0.67	2.46 ±0.03	0.01 ±0.00	0.01 ±0.00	0.06 ±0.00	9.98 ±0.02	0.90 ±0.02	2.89 ±0.04	11.14 ±0.13	15.30 ±0.21	23.83 ±0.42	0.23 ±0.07
Site 10	100.53 ±4.91	12.14 ±0.61	7.20 ±0.10	7.30 ±0.15	143.10 ±0.00	2.44 ±0.04	0.01 ±0.00	0.01 ±0.00	0.05 ±0.01	10.28 ±0.30	0.79 ±0.11	2.92 ±0.05	11.14 ±0.13	15.27 ±0.22	24.07 ±0.39	0.30 ±0.12
Site 11	106.33 ±1.74	12.86 ±0.21	7.13 ±0.03	7.47 ±0.03	143.67 ±0.57	2.44 ±0.04	0.01 ±0.00	0.01 ±0.00	0.05 ±0.00	10.27 ±0.30	0.79 ±0.11	2.92 ±0.05	11.15 ±0.14	15.26 ±0.23	24.05 ±0.42	0.33 ±0.09
Site 12	107.27 ±0.83	12.92 ±0.15	7.17 ±0.03	7.50 ±0.00	144.23 ±0.57	2.44 ±0.04	0.01 ±0.00	0.01 ±0.00	0.04 ±0.01	10.22 ±0.33	0.79 ±0.11	2.93 ±0.04	11.21 ±0.08	15.26 ±0.23	24.01 ±0.39	0.27 ±0.12

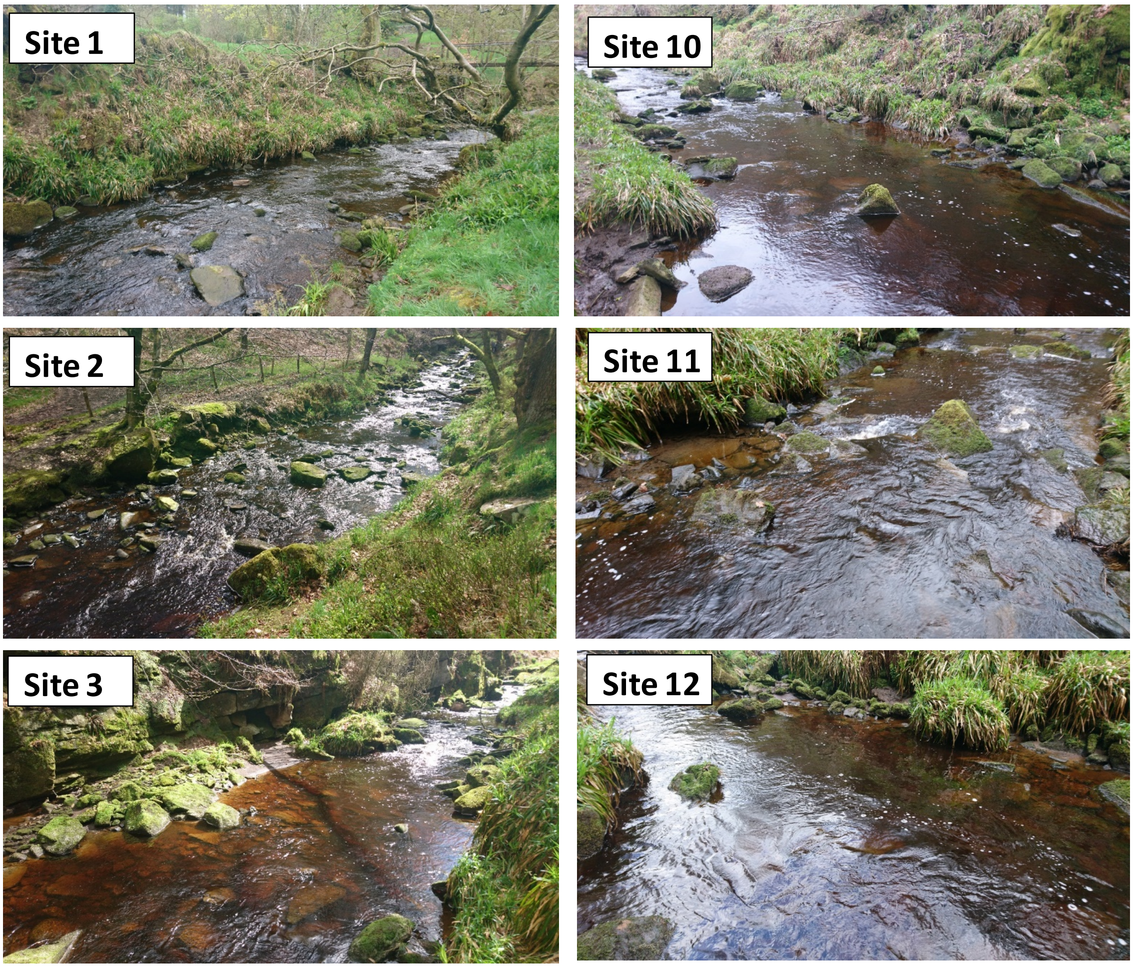


Figure S1: sampling sites below the intervention zone (S1-3) and above the intervention zone (S10-S12). The river width was between 4 and 7 m, with a mean of 5.1m, with a mean depth of 0.14m, and maximum depth of 0.40m.

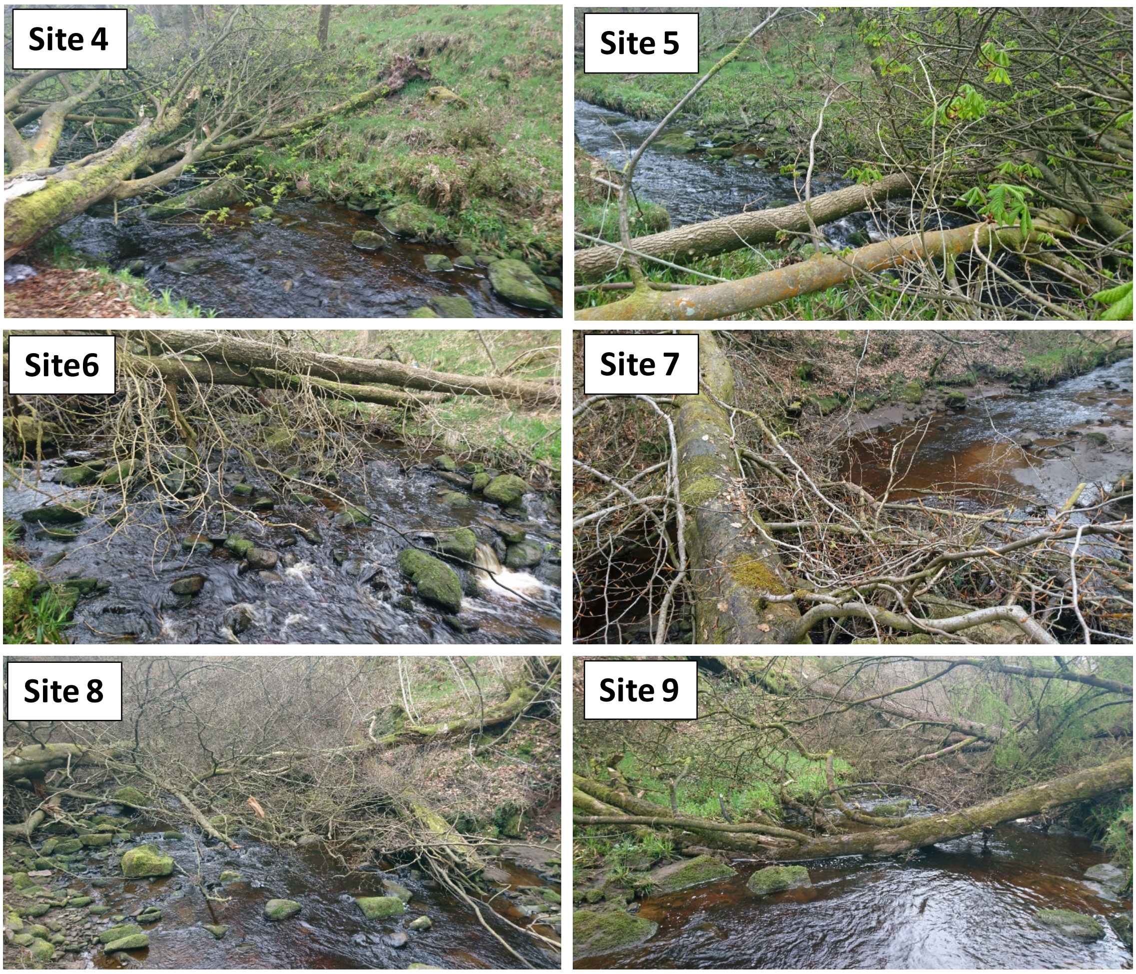


Figure S2: Sampling sites within the intervention zone. Sites 4, 6 and 8 are downstream of a LWD intervention, and Sites 5, 7, and 9 are immediately above the same intervention. The river width was between 4 and 8 m, with a mean of 6.2m, with a mean depth of 0.14m, and a maximum depth of 0.44m.