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RESEARCH ARTICLE

Adding large woody material into a headwater stream has immediate benefits for macroinvertebrate community structure and function

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Abstract

1. Hydromorphological rehabilitation through installing large woody material (LWM) is increasingly being used to reverse degradation of stream ecosystems. There have been many criticisms of stream rehabilitation projects, because many have not met their goals and many others have not been monitored well enough to assess whether their goals were met.
2. In a before–after–control design (with samples collected 1 year before and two successive years after LWM installation), instream biotopes and their macroinvertebrate assemblages were used as structural and functional units to assess the effectiveness of LWM installed at the Rolleston Brook, a headwater tributary of the River Welland in Leicestershire, UK.
3. The project was successful in enhancing the coefficient of variation of channel water depth and width, wetted surface area, number of instream biotopes, and the biotope diversity in the rehabilitated reach.
4. LWM installation led to significant increases in macroinvertebrate total density, total biomass, and taxon richness. Macroinvertebrate community composition was also enhanced, so that it became more similar to that of the control reach.
5. Small increases in the number of instream biotopes (appearance of gravel and leaf litter) and changes in biotope proportions (decreasing percentage of silt) were significantly related to changes in the macroinvertebrate community metrics in the rehabilitated reach.
6. The results show that using macroinvertebrate community composition is more effective than only using taxon richness and/or diversity metrics for understanding the relationship between LWM installation and macroinvertebrate community responses. To be effective, samples must also be collected in a predefined sampling protocol stratified at the instream biotope level. This approach would be of great benefit in evaluating biodiversity conservation value, and could be incorporated into the advice provided by Natural England concerning restoration and protection of English rivers that are designated as Sites of Special Scientific

Interest under UK legislation or Special Areas of Conservation under European legislation.

KEYWORDS
biodiversity, hydromorphology, invertebrates, restoration, river, sedimentation, stream

1 | INTRODUCTION

For several centuries, anthropogenic changes have reduced the hydromorphological complexity of streams and rivers, and their biotic communities have been strongly altered — in most cases depleted (Sparks, 1995), or made more uniform with lower biodiversity (Wallace, Webster & Meyer, 1995).

Large woody material (LWM) is increasingly used in rehabilitation projects to improve the hydromorphological and ecological status of physically degraded streams and rivers (Kail et al., 2007). LWM seems to have an important effect on channel structure and functioning, acting as an 'ecosystem engineer' to increase habitat heterogeneity through alteration of geomorphological, hydraulic, and sediment retention processes (Thorp & Covich, 2001; Corenblit et al., 2007). Although installation of LWM can increase local flood risk by redirecting water onto floodplains, this can reduce flood peaks downstream (Gippel et al., 1996).

Many studies have reported positive effects of LWM installation on channel morphology, current velocity, sediment retention, pool creation, leaf litter retention, and nutrient dynamics (Smock, Metzler & Gladden, 1989; Hilderbrand et al., 1997; Larson, Booth & Morley, 2001; Roni et al., 2006; Lester, Wright & Jones-Lennon, 2007). However, biological improvement has often been limited by other broader scale constraints acting at the catchment scale (Palmer, Menninger & Bernhardt, 2010; Bernhardt & Palmer, 2011), including dispersal constraints that limit the recolonization of lost biota (Langford et al., 2009) and poor water quality (Kail, Arle & Jähnig, 2012).

The National River Restoration Inventory of the UK and the Rivers: Engaging, Supporting and Transferring knowledge on River Restoration (RESTORE) project in Europe were reviewed by Thompson (2015). He found that the main aim of 91% of the 649 projects for which information was available was ecological rehabilitation; but 70% of projects provided no ecological monitoring information. Only 0.7% had used a rigorous (before–after–control–impact, BACI) study design to demonstrate that ecological changes in the rehabilitated site were not due simply to natural variation. Cashman et al. (2019) searched the National River Restoration Inventory during March 2018 and found that only 276 of 912 individual LWM rehabilitation projects provided details concerning post-rehabilitation monitoring approaches, and these were mostly restricted to photographic records. Macroinvertebrates were used as a monitoring approach in only 20 projects.

The introduction of the European Water Framework Directive (WFD; Council of the European Communities, 2000) required EU countries to achieve at least 'Good Ecological Status' (GES) of streams and rivers by 2015, and introduced a new phase of managing European rivers (Muhar et al., 2016). However, many EU member states failed to achieve these initial goals, so the target dates were shifted to 2021 or 2027. The latter date is the final deadline by which all member states must meet GES (European Commission, 2012). In the UK, leaving the EU at the end of 2020 is promised to result in 'no-deterioration' of environmental standards, something that seems unlikely to be achieved when the Environment Agency made 5,000 (or 33%) fewer inspections in 2019 than in 2014 because of financial cutbacks (Everett, 2020).

1.1 | Project planning based on previously published work

An extensive review of the existing literature was conducted, focusing on peer-reviewed literature and readily available unpublished literature, such as dissertations, theses, and case study reports (Al-Zankana, Matheson & Harper, 2020). The search was not restricted to particular journals. Web of Science, Google Scholar, and SCOPUS were searched using the following keywords: (Restore* OR rehabilit* OR enhance* OR mitigate* OR reconfigurat* OR re-meander*) AND (aquatic habitat* OR reach* OR channel* OR stream* OR river*) AND (heterogeneity* OR LWD* OR habitat* OR instream*) AND (macroinvertebrate* OR invertebrate*). The British Library eTheses Online (EThOS) database was searched using the terms 'Restoration and macroinvertebrates', 'rehabilitation and macroinvertebrates', 're-meandering and macroinvertebrates', 'stream restoration', 'river restoration', 'stream rehabilitation', 'river rehabilitation', 'heterogeneity and macroinvertebrates', 'habitat and macroinvertebrates', 'LWD and macroinvertebrates', 'boulder addition and macroinvertebrates', or 'channel reconfiguration and macroinvertebrates'. These searches were conducted from 15 March to 15 April, 2016.

Each paper was examined to determine whether the study included an evaluation of a rehabilitation project that added LWM as a sole rehabilitation measure. Four criteria determined inclusion: (1) The paper must have evaluated a physical rehabilitation project designed to enhance habitat heterogeneity, involving addition of LWM as a sole rehabilitation measure. (2) The paper must have quantified macroinvertebrate community responses, such as community composition, density, richness, diversity, and/or biomass. (3) Macroinvertebrate responses must have been quantified at the reach scale, not within a single habitat (e.g. macroinvertebrate density recorded on only marginal plants or only on gravels, with no information about the rest of the stream). (4) The study must have included a before–after (BA), a control–impact (CI), or a BACI design. Some papers were eliminated based on their abstracts; all other papers were read in full. Related literature cited in every paper, including former meta-analyses, was also searched. Thirteen published papers that evaluated the outcomes of 49 independent projects were located (Table 1). Out of the 49 projects, only 11 recorded an increase in taxon richness or diversity within 1–4 years. Fifteen projects recorded an increase in macroinvertebrate density within 1 year. Eleven projects assessed macroinvertebrate biomass as a response variable, but only four of these found that it increased within 1–4 years (Smock, Metzler & Gladden, 1989; Wallace, Webster & Meyer, 1995; Entrekin et al., 2009).

Failure of LWM treatment even to improve the physical habitat and channel complexity was recorded as the limiting factor in six rehabilitation projects by Leal (2012) and Thompson (2015). Entrekin et al. (2009) measured macroinvertebrate secondary production 2 years after the addition of LWM. They suggested that rehabilitated reaches are likely to require more than 2 years to achieve measurable changes in channel geomorphology, organic matter retention, and macroinvertebrate community.

An appropriate study design that is able to partition the effects of treatment from natural sources of variation (e.g. seasonal and inter-annual variability) is still uncommon. There is a dearth of pre-installation data, which has pushed researchers to use a surrogate for pre-installation data: a control–impact study design. Although this was used in 26 projects, it can be misleading (Miller, Budy & Schmidt, 2010) and 'render [supposed] impacts on macroinvertebrates questionable' (Feld et al., 2011). This approach might confound responses to rehabilitation activities with differences between macroinvertebrate communities (Laasonen, Muotka & Kivijärvi, 1998; Negishi & Richardson, 2003) because macroinvertebrate community metrics vary naturally at small spatial scales for reasons unrelated to rehabilitation activities (Negishi & Richardson, 2003; Miller, Budy & Schmidt, 2010). Monitoring rehabilitation outcomes needs to consider the direction of biotic community changes, not just change itself (Downes et al., 2002). Assessing the effectiveness of LWM installation needs rigorous study design using pre-installation data to assess inherent differences between control and rehabilitated reaches. Only nine projects of the 49 used undisturbed control reaches to represent the target state of rehabilitation (Smock, Metzler & Gladden, 1989; Gerhard & Reich, 2000; Thompson, 2015), to account better for other confounding sources of variance, and therefore to provide more conclusive evidence for the significance of rehabilitation treatment on the macroinvertebrate community (Miller, Budy & Schmidt, 2010; Feld et al., 2011). Macroinvertebrate taxon richness and diversity were commonly used as monitoring metrics, but density and biomass (which showed some significant responses) were rarely used. A key aim of many LWM projects is to enhance morphological complexity and physical heterogeneity by increasing leaf-litter retention within the restored reach. Increases in leaf litter (as an organic instream biotope) should provide food and shelter for benthic invertebrates, thus affecting their density and biomass. These metrics are therefore very useful for evaluating the outcomes of this kind of restoration. In two low-gradient headwater streams on the Coastal Plain of Virginia, USA, macroinvertebrate abundance and biomass increased significantly with an increase in the number of woody dams that led to the collection of organic matter and increased food availability (Smock, Metzler & Gladden, 1989).

The present study addresses these major limitations of previous studies by using stratified pre-identified sampling in a before–after–control (BAC) study design to evaluate the short-term ecological effects of LWM installed in a small rural stream (we were unable to identify a suitable degraded 'impact' site to create an ideal BACI design). Instream biotopes and their macroinvertebrate assemblages were used as a tool to assess ecological effectiveness. These have been shown to be a useful way of linking macroinvertebrate ecology and river hydromorphology (Demars et al., 2012). The specific hypotheses were as follows: (i) LWM installation will increase morphological complexity and physical heterogeneity; (ii) macroinvertebrate community structure and functional metrics will change significantly in the rehabilitated reach compared with similar measures made before rehabilitation; and (iii) the macroinvertebrate community metrics in the rehabilitated reach will resemble those of a nearby control reach. A less degraded semi-natural tributary of the same stream order about 200 m away was the theoretical goal of the rehabilitation. Data from it were used as benchmarks for the level of recovery of the rehabilitated reach (following Hughes, Larsen & Omernik, 1986).

2 | METHODS

2.1 | Study sites and LWM installation

The study sites and rehabilitation measures applied have been described in detail by Al-Zankana (2018), so only basic information is given here. Rolleston Brook is a headwater tributary of the Welland River, at Rolleston, Leicestershire, England (52.603564° N, −0.913059° W). The stream has a catchment of approximately 7.7 km^2; this is largely rural, with a mixture of both arable and improved grassland. The landscape is moderately sloping and creates substantial runoff risks for the water environment. The soils in this study area are mainly medium to heavy clay. The average daily flow was 0.1–0.6 m^3 s^{-1}. The channel widths ranged from 0.7 m to 2.85 m (mean channel width 1.3 m). Artificially straightening and over-deepening were the two main types of physical degradation on this stream, leading to an imbalance of instream biotope appearance and frequency. The entire reach was covered by fine-grained sediments, with an apparent low retention capacity of leaf litter and small woody material, owing to low amounts of instream LWM. The longitudinal connectivity of the channel was disrupted by the presence of four concrete structures (Supporting Information Figure S1). The reach lacked hydromorphological variability and biodiversity. A 220 m stretch of this stream (the 'rehabilitated reach') was compared with a 220 m stretch of a nearby, less modified tributary (the 'control reach').

TABLE 1 Summary of published studies addressing the effects of LWM rehabilitation projects on habitat heterogeneity and macroinvertebrate community structure and function. The findings of individual but closely related studies are listed together for coherence, but are itemized separately (where necessary) in the Key findings column

Reference, location (no. of projects)	Study design (project age)[a]	Key findings
Smock, Metzler & Gladden (1989), Virginia, USA (2)	CI (1)	Macroinvertebrate abundance and biomass increased with an increase in the amount of woody material because there was increased collection of organic matter and increased food availability. The contribution of shredders to biomass increased with increasing the abundance of dams.
Wallace, Webster & Meyer (1995), North Carolina, USA (1)	BA (4)	At sites with LWM addition, stream depth and organic matter increased, current velocity decreased, and sand and silt covered the cobble substratum. Macroinvertebrate abundance, biomass, and secondary production increased significantly after rehabilitation. Abundance, biomass, and secondary production of scrapers and filterers decreased; collectors and predators increased; no change in overall shredder biomass.
Hilderbrand et al. (1997), West Virginia, USA (2)	BA (2)	Systematic placement had a lower effect on erosion and score rates than random placement did. There were no changes in total abundance of macroinvertebrates in either stream. Ephemeroptera abundance increased significantly with increasing pool area.
Gerhard & Reich (2000), Germany (2)	CI (4)	Rehabilitated reaches had more functional habitat patches than the non-rehabilitated reaches. Macroinvertebrate abundance, species richness, and diversity increased in Joseklein stream, but not in Lude stream.
Larson, Booth & Morley (2001), Washington, USA (6)	CI (2–10)	Channel complexity increased significantly. There was no change in benthic IBI.
Pretty & Dobson (2004), UK (3)	BA (2)	Log addition enhanced detrital standing stocks. Total abundance and taxon richness of macroinvertebrates were significantly increased in rehabilitated reaches. The response was most marked for detritivores.
Roni et al. (2006), Oregon, USA (13)	CI (1–20)	Pool area, amount of LWM, and number of boulders and pools were significantly higher in rehabilitated sites than in the control sites. There were no changes in macroinvertebrate species abundance, richness, EPT%, FFGs%, or IBI.
Lester, Wright & Jones-Lennon (2007), Australia (8)	BACI (1)	Wood increased storage of organic matter and sediments and improved bed and bank stability. Macroinvertebrate density and richness increased significantly. Treated streams had greater family richness and greater richness of all functional feeding groups. Richness increased in all wood, benthic, and edge habitats.
Entrekin et al. (2009), Michigan, USA (3)	BACI (2)	Significant increase (22%) of macroinvertebrate biomass and secondary production was recorded in one rehabilitated reach, whereas there were no significant changes in two other reaches in comparison with values before log addition.
Coe et al. (2009), Washington, USA (2)	CI (2)	Macroinvertebrate density was significantly higher on woody material than on cobbles. Wood substrate increased density of invertebrates at reach level.
Testa, Shields & Cooper (2011), Mississippi, USA (1)	BACI (2)	Woody substrate tripled after rehabilitation, but there were no significant changes in macroinvertebrate density and family richness.
Leal (2012), California, USA (1)	CI (1)	Smaller substrate particle sizes were found across the rehabilitated site. There were no significant changes in other habitat features, such as canopy cover, algae, tree roots, and emergent vegetation. Lower invertebrate abundance and diversity were associated with LWM in several months of the first year after rehabilitation. There was no significant improvement of macroinvertebrate density or richness.

TABLE 1 (Continued)

Reference, location (no. of projects)	Study design (project age)[a]	Key findings
Thompson (2015), UK (5)	BACI (1)	There were no significant changes in reach-scale geomorphology. Macroinvertebrate abundance and biomass were significantly higher within LWM habitat. At reach scale, biomass was significantly higher in rehabilitated reaches than in non-rehabilitated reaches, but there were no significant changes in density and richness, diversity, and FFG composition.

CI: control–impact; BA: before–after; BACI: before–after–control–impact; LWM: large woody material; EPT%: Ephemeroptera, Plecoptera, Trichoptera index; FFGs%: functional feeding groups index; IBI: index of biotic integrity.
[a]Project age is given as the age in years at the time of monitoring by the study authors.

The control reach was a suitable target because it had a more natural configuration characterized by meanders, LWM dams, riffle–pool sequences, and a wide range of organic and inorganic biotopes (Supporting Information Figure S2).

Large natural wood pieces (branches and logs >10 cm diameter and 1 m in length) were installed in five sections of the rehabilitated reach during summer 2014. LWM was installed (Supporting Information Figure S3) as follows: (a) parallel to the flow (from one or both sides) to narrow the channel, reduce ponding upstream of the obstructions, and enhance the water flow; (b) perpendicular (70–90°) to the channel to create meander patterns and promote riffle–pool sequences, increase hydraulic roughness; (c) downstream facing (30°) as deflectors to kick flow over to one side and promote bank scour for outer meander bend development; or (d) as wing deflectors from both sides spanning the stream channel to create steps along the channel profile, regulate sediment movements through the channel system, and enhance leaf litter retention. The distance between each LWM installation was 6.5–9.1 m, which was equal to five to seven times the average channel width. Deflectors stretched out from the bank to at least half-way across the low-flow channel.

2.2 | Channel hydromorphological survey

The 220 m reach of each stream was surveyed and mapped in spring 2014 (Sp.14; before LWM installation). Channel depth, channel width, instream biotope number, and the area covered by each available biotope were recorded. After the LWM had been installed, changes in these hydromorphological metrics were recorded in spring 2015 (Sp.15) and spring 2016 (Sp.16). Channel width and depth were measured every 5 m; depth was measured to the nearest centimetre at the centre of each 5 m cross-section. Instream biotopes were visually identified and named according to Demars et al. (2012). Instream biotopes were cobbles, gravel, sand, clay, soft silt (with organic matter), tree roots, marginal plants, and leaf litter. Their cover was estimated using lateral transects spaced every 5 m following Entrekin et al. (2009). All transect measures were summed to give reach-level parameters and total wetted surface area for each study reach. Instream biotope diversity (‘SWI-biotope’) characterized by the Shannon–Wiener diversity index (SWI; Shannon & Weaver, 1949) was calculated following Kemp (1999) and Poppe et al. (2015). The coefficients of variation (defined as the ratio of the standard deviation to the mean) of channel water depth and width (CV-depth, CV-width) were calculated from all measurements made along the reach.

2.3 | Sampling and processing of macroinvertebrates

In Sp.14 (the middle of March and again in May), before the installation process started, macroinvertebrate samples were collected from both reaches. Three replicate samples were taken from each biotope (defined as covering ≥1% area of the river bed) within the two reaches. The replicates were taken from up to three different patches per reach to represent the entire study reach. Samples were taken from a random location within each patch. Biotope-specific samples were collected from all available habitat patches in the reaches. The sampling locations accounted for the fact that the degraded reach had larger patches. Samples were collected using a Surber sampler (500 μm mesh size and area of 0.09 m^2). The area within the frame was disturbed for 30 s to dislodge all animals in the substrate, with the animals subsequently being swept by the water into the net. After LWM installation, samples were collected in Sp.15 (the middle of March and again in May) and in Sp.16 (the middle of March and again in May).

Macroinvertebrates were extracted from the samples, placed in 50 ml sealable plastic sample tubes containing 75% ethanol, and kept separately for later taxonomic identification and counting. Specimens were identified to the lowest possible taxonomic level (either species or genus), with the exception of Oligochaeta, Coleoptera, Diptera, and early Limnephilidae instars, which were identified to family level. Chironomidae were identified to sub-family level. The number of individuals per sample of each identified taxon was recorded for each biotope and is hereafter referred to as ‘biotope-specific count per sample’.

Macroinvertebrate population biomass (mg dry mass (DM) per sample) was estimated according to the published size-specific mass regressions in the literature, in addition to direct estimation for worms and some insect larvae — following Rodriguez & Verdonschot (2002) and Benke & Huryn (2006). Population biomass of each macroinvertebrate species (mg DM per sample) was calculated using both population density (number of individuals per sample), and dry mass (mg) of each individual organism within the population.

At biotope level, the population of each species was divided into different size classes based either on body length (BL) or head-capsule width (HW). BL was measured to the nearest 0.5 mm and HW to the nearest 0.1 mm, and size classes were defined by this measurement accuracy. Length was measured by using an ocular micrometer for small specimens, or a sheet of 1 mm graph paper placed directly on the dissecting microscope stage for the large specimens. HW was measured across the widest part of the head. BL was measured as the distance between the anterior of the head to the posterior of the last abdominal segment (after Poepperl, 1998). In addition, for two species, other linear body dimensions were used: for *Gammarus pulex*, the length of the first thoracic segment, and for *Asellus* spp. the length of the pleotelson. Trichoptera larval HW at eye level was measured to the nearest 0.2 mm, after Ross & Wallace (1983).

The DM of each individual organism was estimated using

$$\log M = \log a + b \log L$$

where M (mg) is organism DM, L (mm) is any linear dimension, and a and b are constants.

Size-specific biomass (mg DM per sample) for each size group was calculated by multiplying the size-specific dry mass (mg) by the density (number of individuals per sample) of the size group, then the biomasses of all the size groups were summed to obtain the population biomass per sample (mg DM per sample), which is referred to as the biotope-specific taxon biomass per sample. This approach properly accounts for seasonal growth because each taxon-specific biomass for each sample is derived directly from the measured sizes of each individual (of that taxon) in that sample. The computation of taxon-specific biomass used one common taxon-specific size-to-weight relationship for all individuals of that taxon, regardless of season of collection. The list of these regressions and length parameters, and relevant references, are available in Supporting Information Table S1.

Reach-level values of taxon count per sample and taxon biomass per sample were calculated according to the relative area of each instream biotope in the given reach. The given reach-level variable lists were created by summing biotope-specific list values that were weighted by their percentage availability, following Huryn & Wallace (1987), Lugthart & Wallace (1992), Kedzierski & Smock (2001), Pedersen et al. (2007), and Jähnig et al. (2010).

The reach-level taxon count per sample data lists were used to generate eight community structure and diversity metrics: total density (individuals per sample), taxon richness, evenness, taxon diversity, Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness, EPT diversity, EPT count%, and Chironomidae count%. Taxon biomass per sample data lists were used to calculate total biomass (mg DM per sample), EPT biomass%, and Chironomidae biomass%.

2.4 | Data analysis

Macroinvertebrate total density and total biomass were pooled to show values per square metre before the data analysis. Reaches were compared before and after LWM installation using 11 univariate metrics (total density, total biomass, taxon richness, evenness, taxon diversity, EPT richness, EPT diversity, EPT count%, EPT biomass%, Chironomidae count%, and Chironomidae biomass%) as response variables. The Euclidean distance matrix was used to calculate distances between samples for each metric separately. Metrics were transformed before the analysis to normalize the data distribution and satisfy the permutation analysis of variance (ANOVA) test requirement, where applicable. A two-way permutation ANOVA design with reach type (fixed factor, two levels: control, rehabilitated) and period (fixed factor, three levels: Sp.14, Sp.15, Sp.16) was used for running a BAC design test and all possible pairwise tests. Since the design was unbalanced, a type III sum of squares was used. All ANOVA tests used 9,999 random permutations under a reduced model. When there were too few possible permutations (<100) to obtain a reasonable test, a P value was calculated using 9,999 Monte Carlo draws from the appropriate asymptotic permutation distribution (Anderson & Robinson, 2003).

When a permutation ANOVA gave a significant overall interaction (reach × period), all pairwise comparisons were made to examine which elements contributed to the overall interaction. If there was no overall effect but there were only reach or period effects, all the pairwise comparisons related to that reach effect or period effect were examined because the aim of the study was to capture all changes.

Initial spatial differences between the reaches were assessed by contrasting control-before and rehabilitated-before data. The effect of rehabilitation was assessed by contrasting rehabilitated-before and rehabilitated-after data. To assess the direction of the change in the rehabilitated reach (i.e. towards or away from the control reach, which was the target), control-after data were contrasted with rehabilitated-after data. Temporal variation, unrelated to the restoration, was assessed by contrasting control-before and control-after data.

To examine differences in temporal change between the two reaches, the differential changes in the measured metrics that were significantly affected were also calculated using BACI contrasts (Smith, 2002) using the emmeans package v.1.2.2 in R v.3.4.0 (R Core Team, 2017). BACI contrasts with 95% confidence intervals (CIs) provide information about the effect size of the treatment (e.g. rehabilitation) on the response variable (e.g. taxon richness). Positive values indicate that the variable has increased more in the rehabilitated reach relative to the control reach, across the time period ranging from before to after the rehabilitation.

The BACI contrast is represented as follows:

$$(\mu_{RA} - \mu_{RB}) - (\mu_{CA} - \mu_{CB})$$

where μ_{RA} is the mean of the measured metrics (e.g. taxon richness) after the rehabilitation in the rehabilitated reach, μ_{RB} is the mean of the same metric before rehabilitation in the rehabilitated reach, μ_{CA} is the mean of the measured metrics after the rehabilitation in the control reach, and μ_{CB} is the mean of the same metric before rehabilitation in the control reach.

Data matrices for the taxonomic composition of the macroinvertebrate community (taxon count per sample and taxon biomass per sample) were pooled to show values per square metre before data analysis and are referred to as count/m^2, and biomass/m^2 hereafter. Both count/m^2 and biomass/m^2 data matrices were then fourth-root transformed prior to the analysis to downweight the influence of numerically dominant taxa and prevent masking of less abundant taxa. The Bray–Curtis dissimilarity matrix was used to calculate distances between samples for each metric separately. Reaches were compared before and after LWM installation by performing the same ANOVA design. Two-dimensional non-metric multidimensional scaling (nMDS) ordination plots with Bray–Curtis dissimilarity coefficients were used to visualize significant differences.

A similarity percentage procedure was used to determine which family or taxonomic group accounted for the dissimilarities in any significant spatial or temporal measures, with exclusion of family or taxonomic groups that contributed less than 30% of the dissimilarity, following Johnson et al. (2010) and Gosch et al. (2014). These analyses were visualized using nMDS and shade plots.

The relationships between channel morphological variables and macroinvertebrate metrics were analysed using distance-based linear modelling following Eddy & Roman (2016). These analyses were performed after normalizing the morphological variables. Euclidean distance matrices of all metrics used for the ANOVAs were used separately. Sequential tests were used to determine which combinations of morphological variables best explained variability in the response variable. Each sequential test was performed with a stepwise selection procedure using Akaike's information criterion. This analysis partitions the variability of the macroinvertebrate community metrics along best-fit axes and then tests the morphological variables that are most closely related to these axes. The significance of the relationships between morphological variables and biological metrics were determined using Spearman's rank correlation ρ.

BIOENV analysis (Clarke & Ainsworth, 1993) was used to investigate relationships between patterns in macroinvertebrate community taxonomic composition (count/m^2 and biomass/m^2) and morphological variables. The test selected a maximum of five morphological variables from Euclidian distance resemblance matrices that contributed the best Spearman's rank correlation ρ with each of count/m^2, and biomass/m^2 data Bray–Curtis similarity matrices. All analyses were carried out using PRIMER v.7 software (Clarke & Gorley, 2015) and the PERMANOVA+ add-on package (Anderson, Gorley & Clarke, 2008).

3 | RESULTS

3.1 | Channel hydromorphology and instream biotope composition

During the first post-rehabilitation year, LWM installation enhanced the hydromorphological complexity of the rehabilitated reach. The diversity of current velocity was enhanced because LWM installation created steps along the channel profile, and these had sequences of riffles and pools (Figure 1a, Sp.15). Coefficient of variation of channel water depth and width (CV-depth and CV-width), wet surface area (m^2), number of instream biotopes, and the biotope diversity (SWI-biotope) increased (Supporting Information Table S2 and Figure S4). At the beginning of Sp.16 (during the second post-rehabilitation year), most, if not all, of the LWM installation was washed downstream of the rehabilitated reach by a flood, where it was trapped by the concrete structures and blocked the channel (causing further longitudinal disconnection to the reach; Figure 1 and Supporting Information Figure S5). Trapped silty materials were dispersed so that they covered former cobble and gravel patches (Supporting Information Figure S6a), and formerly retained leaf litter was washed out to the channel banks or downstream, meaning that the morphological metrics and instream biotope composition of the rehabilitated reach declined (Supporting Information Figure S4).

3.2 | Macroinvertebrate community metrics

The permutation ANOVA results (Supporting Information Table S3) show that there were statistically significant interactions between reach and period for total density (pseudo-F = 7.4747, P < 0.003) and total biomass (pseudo-F = 22.17, P < 0.0002). They also indicated a significant reach effect (pseudo-F = 6.0363, P < 0.002 and pseudo-F = 176.36, P < 0.0002 respectively) and a significant period effect (pseudo-F = 6.0835, P < 0.006 and pseudo-F = 25.034, P < 0.0002 respectively). There were statistically significant (P < 0.0002) reach effects for taxon richness, taxon diversity, evenness, EPT richness, EPT diversity, EPT count%, EPT biomass%, Chironomidae count% and Chironomidae biomass%. Chironomidae count% also had a significant period effect (P < 0.02).

Results from the pairwise tests between reaches showed that before the LWM installation process, in Sp.14, the control reach had two to 10 times higher values for total density (individual/m^2), total biomass (mg DM m^{-2}), taxon richness, taxon diversity, evenness, EPT richness, EPT diversity, EPT count%, and EPT biomass% compared with the degraded reach. The degraded reach had about five times higher Chironomidae count% and nine times higher Chironomidae biomass% (Table 2). Reaches differed significantly (P < 0.05) in all macroinvertebrate community metrics measured (Supporting Information Table S4).

During the first post-rehabilitation year, LWM installation led to significant increases (P < 0.007) in the macroinvertebrate total density (individuals/m^2) of the rehabilitated reach, and the BACI contrast was 692, 95% CI (419.2, 1169.3). Total biomass (mg DM m^{-2}) increased significantly (P < 0.003), and the BACI contrast was 497, 95% CI (88.5, 1071.2). Taxon richness also increased significantly (P < 0.02), and the BACI contrast was 1, 95% CI (0.9, 5.6). Other metrics measured did not show any significant responses to the rehabilitation process (Supporting Information Table S5). The lack of any temporal difference in the metrics measured in the control reach was also a good

FIGURE 1 Two sections of the rehabilitated reach, before (2014) and after installation of large woody material (LWM; 2015 and 2016). (a) Upstream of the first concrete obstruction, the LWM installation created steps along the channel in a riffle–pool sequence, and increased leaf-litter retention in spring 2015. (b) Downstream of the first concrete obstruction, installed LWM increased the complexity of the channel morphology by increasing the instream biotope mosaic and enhancing leaf-litter retention in spring 2015. In spring 2016, the LWM installation washed away and the rehabilitated reach lost most of the positive hydromorphological benefits it had brought

indicator that the significant changes in the rehabilitated reach (e.g. total density, total biomass, and taxon richness) were induced by the morphological effects of the LWM installation applied to that reach only (Supporting Information Table S5). Total density in the rehabilitated reach became more similar to that in the control reach in Sp.15 (Supporting Information Table S4).

During the second post-rehabilitation year (Sp.16), after the flood event, the positive influences of the LWM installation were lost (Figure 1), with total density, total biomass, and taxon richness declining significantly ($P < 0.05$). Even though macroinvertebrate total biomass had declined in Sp.16 compared with Sp.15, it was still higher than in Sp.14 ($P < 0.005$). Chironomidae count% and Chironomidae biomass% also decreased significantly (both $P < 0.005$), and the rehabilitated reach had lower Chironomidae count% and Chironomidae biomass% values than in Sp.14.

3.3 | Taxonomic composition of the macroinvertebrate community

According to the PERMANOVA results for count/m^2 and biomass/m^2 (Supporting Information Table S3), in addition to the reach effect (pseudo-F = 58.852, $P < 0.0002$ and pseudo-F = 51.237, $P < 0.0002$ respectively) and period effect (pseudo-F = 2.6771, $P < 0.002$ and pseudo-F = 2.5516, $P < 0.0008$ respectively), there were statistically significant interactions between reach and period (pseudo-F = 2.9882, $P < 0.001$ and pseudo-F = 2.4075, $P < 0.002$ respectively). Results from the pairwise tests between reaches showed that, before the LWM installation (Sp.14), reaches differed significantly ($P < 0.005$) for both their count/m^2 and biomass/m^2 values (Supporting Information Table S6).

By Sp.15 the taxonomic composition of the macroinvertebrate community had changed significantly (count/m^2, $P < 0.005$, and biomass/m^2, $P < 0.005$; Supporting Information Table S7), but those changes were not enough to make the rehabilitated reach similar to the control reach. Thus, between-reach differences in community taxonomic composition remained significant after the process over the two successive seasons sampled (Supporting Information Table S6). The difference was demonstrated by the clear separation of repeated samples according to the study reaches on the nMDS ordination plots (Figure 2). Similarity percentage analyses on count/m^2 data indicated that after the LWM installation (Sp.15) the average dissimilarity between the two reaches decreased from 68.7 to 58.3 (Sp.14; Supporting Information Table S8). This shows that, although the differences between the reaches in community taxonomic composition were significant, the macroinvertebrate community composition of

TABLE 2 Mean (±SD) of macroinvertebrate community structural and functional metrics

	Spring 2014		Spring 2015		Spring 2016	
Metrics	C	R	C	R	C	R
Total density (individuals/m^2)	1,050 ± 111	571 ± 119	**1,072 ± 146**	**1,285 ± 922**	1,060 ± 204	662 ± 86
Total biomass, dry mass (mg m^{-2})	1,070 ± 141	118 ± 30	994 ± 129	539 ± 146	1,122 ± 138	376 ± 88
Taxon richness	7.1 ± 1.1	2.1 ± 0.5	7.0 ± 0.6	3.0 ± 0.5	7.4 ± 0.5	2.3 ± 0.3
Taxon diversity	24.9 ± 5.2	4.8 ± 1.2	20.2 ± 3.5	5.2 ± 2.6	23.9 ± 4.4	4.8 ± 0.8
Evenness	0.99 ± 0.002	0.97 ± 0.009	0.98 ± 0.002	0.96 ± 0.014	0.98 ± 0.003	0.96 ± 0.006
EPT richness	3.30 ± 0.38	1.14 ± 0.41	3.04 ± 0.36	1.54 ± 0.42	3.31 ± 0.41	1.24 ± 0.38
EPT diversity	10.60 ± 2.87	3.98 ± 1.06	8.18 ± 2.46	4.58 ± 1.90	9.89 ± 3.28	3.25 ± 0.86
EPT count%	41 ± 4	7 ± 3	42 ± 5	12 ± 6	45 ± 6	7 ± 3
EPT biomass%	44 ± 7	19 ± 8	40 ± 8	24 ± 17	42 ± 10	23 ± 13
Chironomidae count%	14 ± 6	71 ± 6	19 ± 10	50 ± 23	13 ± 3	37 ± 11
Chironomidae biomass%	2 ± 2	18 ± 8	2 ± 2	14 ± 17	2 ± 1	3 ± 1

C: control reach; EPT: Ephemeroptera, Plecoptera, Trichoptera; R, rehabilitated reach.
Bold type indicates that reaches became similar in the given metrics during the given season.

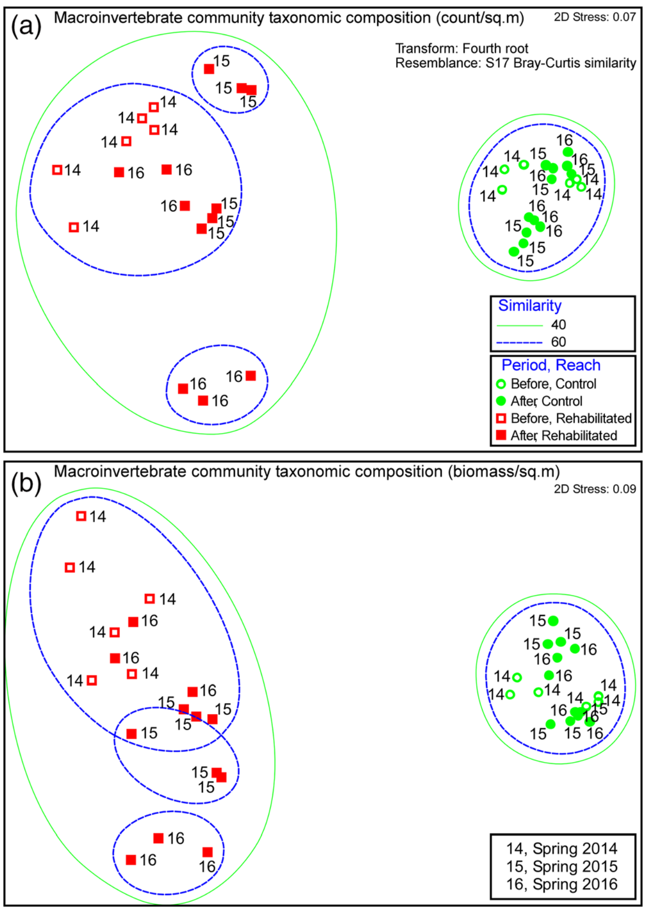

FIGURE 2 Non-metric multidimensional scaling ordination plots, based on the fourth-root-transformed Bray–Curtis similarities of macroinvertebrate taxonomic composition: (a) count/m^2; (b) biomass/m^2. Clusters at 40% and 60% of similarity. Repeated data collected before the rehabilitation process in spring 2014, then over two successive spring seasons after the rehabilitation process

the rehabilitated reach was enhanced during the first post-rehabilitation spring and moved towards the goal state of the rehabilitation (i.e. the control reach). Decreases in average dissimilarity between the two reaches in Sp.15 are visualized in Figure 3. Macroinvertebrate taxa that contributed to the between- and/or within-reach dissimilarities are visualized in a shade plot (Figure 4), where darker colours indicate higher density. The macroinvertebrate taxa are shown in presence/absence form in Supporting Information Table S9.

In Sp.16 (after the flood event), however, average dissimilarity between reaches increased to become 62.3. Significant changes in Sp.16 compared with Sp.14 (count/m^2, $P < 0.02$; Supporting

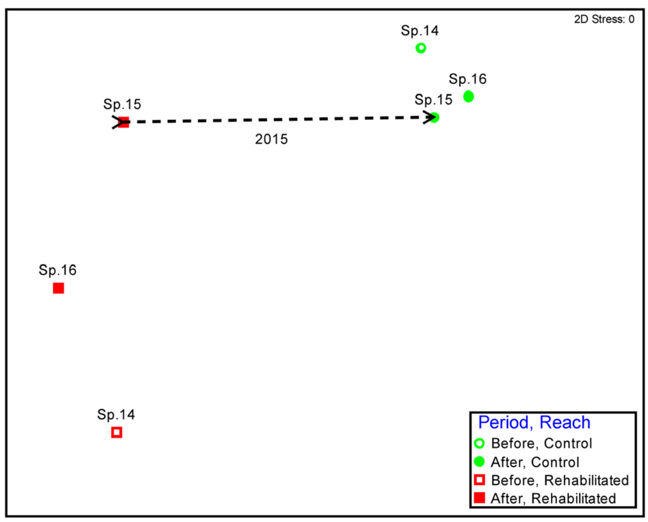

FIGURE 3 Non-metric multidimensional scaling ordination plot of macroinvertebrate taxonomic composition (count/m^2) showing that dissimilarity between the control and rehabilitated reach decreased after large woody material LWM installation in spring 2015. Sp.14: spring 2014; Sp.15: spring 2015; Sp.16: spring 2016

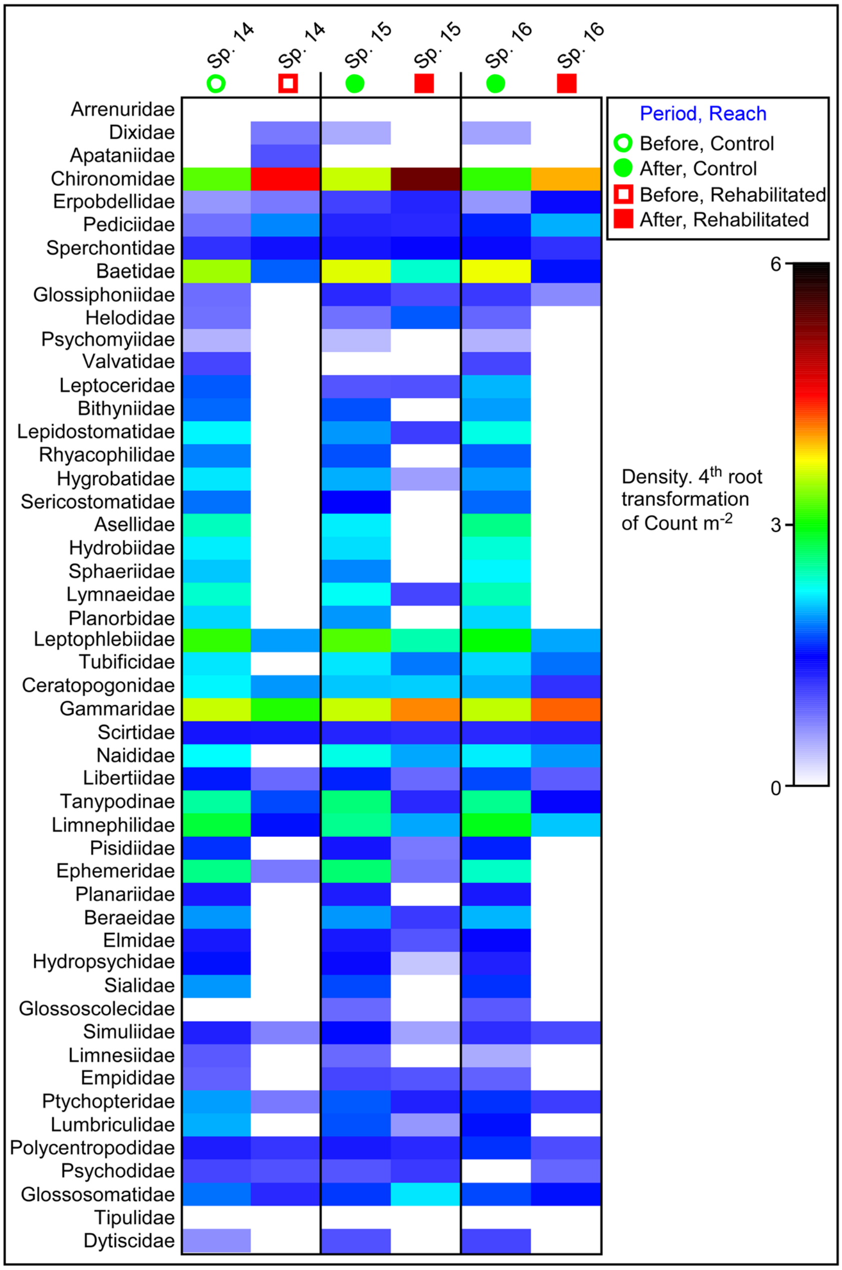

FIGURE 4 Shade plot of macroinvertebrate community taxonomic composition data matrix, showing contribution of families to seasonal dissimilarities between the control and rehabilitated reach. The depth of colour shading is linearly proportional to a fourth-root transformation of the count/m^2; darker colours indicate higher density. Sp. 14: spring 2014; Sp. 15: spring 2015; Sp. 16: spring 2016

Information Table S8) were not, however, in the direction of the control reach, which was the goal state of the rehabilitation.

Before the LWM installation, Chironomidae (non-Tanypodinae) contributed 26% of the dissimilarity between the two reaches. They were the dominant family in the degraded reach (402 individuals/m^2) but were less dominant in the control reach (110 individuals/m^2). Baetidae (12% of dissimilarity), Leptophlebiidae (7%), Gammaridae (7%), and Limnephilidae (5%) also had significantly higher average density in the control reach than in the degraded reach (Supporting Information Table S8). After rehabilitation, average densities of Chironomidae (non-Tanypodinae) and Gammaridae responded significantly to the morphological changes ($P < 0.05$; Supporting Information Table S10). They contributed more than 70% of the before–after changes in macroinvertebrate community composition of the rehabilitated reach. In Sp.15, average density of Chironomidae (non-Tanypodinae) doubled (relative to Sp.14) in the rehabilitated reach to 820 individual/m^2, and contributed 54% of the pre-/post-rehabilitation changes (Table 3). Average density of Gammaridae tripled in Sp.15 to 274 individuals/m^2, from 89 individuals/m^2 in Sp.14. It contributed 24% of the pre-/post-rehabilitation changes. Average density of Chironomidae declined significantly ($P < 0.02$) in Sp.16 compared with Sp.15 to 245 individuals/m^2 (Table 3; Supporting Information Table S10).

The control reach had higher biomass than the degraded reach before the LWM installation (Supporting Information Table S11). The caddisfly family Limnephilidae, average biomass (259 mg m^{-2} DM in the control reach, but only 12 mg m^{-2} DM in the degraded reach) contributed to 25% of between-reach dissimilarity. Lymnaeidae (16%

TABLE 3 Similarity percentages (SIMPER) analysis based on macroinvertebrate community taxonomic composition (count/m^2) data (at family level), identifying those families most affected by morphological changes and contributing at least 70% of the temporal dissimilarity in community composition of the rehabilitated reach

	Average abundance in spring (individuals/m^2)				
Family	2014	2015	2016	Contribution (%)	Cumulative (%)
Spring 2014 vs. 2015					
Average dissimilarity: 49.4					
Chironomidae (non-Tanypodinae)	402	820*		54	54
Gammaridae	89	274***		24	77
Spring 2014 vs. 2016					
Average dissimilarity: 42.2					
Gammaridae	89		311**	43	43
Chironomidae (non-Tanypodinae)	401		245*	33	76
Spring 2015 vs. 2016					
Average dissimilarity: 38.0					
Chironomidae (non-Tanypodinae)		820	245*	60	60
Gammaridae		274	311	11	71

$^*P < 0.05$; $^{**}P < 0.005$; $^{***}P < 0.0005$.

of dissimilarity), Baetidae (8%), Planorbidae (6%), and Gammaridae (6%) also had significantly higher biomass in the control reach than in the degraded reach.

After LWM installation, the average biomass of Gammaridae, Limnephilidae, Chironomidae (non-Tanypodinae), and Erpobdellidae responded significantly ($P < 0.05$) to the morphological changes (Table 4; Supporting Information Table S10). In Sp.15, average biomass of Gammaridae increased to 218 mg m^{-2} DM, compared with 57 mg m^{-2} DM in Sp.14, and contributed 34% of pre-/post-rehabilitation changes in community biomass (Table 4). Average biomass of Limnephilidae increased from 12 mg m^{-2} DM to 82 mg m^{-2} DM to contribute 18% of changes; Chironomidae (non-Tanypodinae) increased from 21 mg m^{-2} DM to 93 mg m^{-2} DM and contributed 16% of changes; and Erpobdellidae increased from 7 mg m^{-2} DM to 48 mg m^{-2} DM and contributed 10% of changes. After the flood event, the average biomass of Chironomidae (non-Tanypodinae) declined significantly ($P < 0.02$) to become 10 mg m^{-2} DM in Sp.16 compared with Sp.15 (Supporting Information Table S10).

3.4 | Relationships between morphological variables and macroinvertebrate community metrics

The total density and total biomass of macroinvertebrate were related positively to temporal variations in leaf litter% ($\rho > 0.75$; Table 5). SWI-biotope, CV-width, and gravel% were also related positively to temporal variations in total density of macroinvertebrates, but in a weak correlation (ρ = 0.11, 0.09, and 0.08 respectively). Taxon richness had a positive correlation with gravels%, SWI-biotope, and number of biotopes. Chironomidae count% had a negative relationship with number of biotopes (ρ = 0.73). Chironomidae biomass% had a negative relationship with gravel% (ρ = 0.93).

Temporal variations in the taxonomic composition of the macroinvertebrate community based on abundance of taxa (count/m^2) were related to variations in wetted surface area and leaf litter% (ρ = 0.61), whereas variations in composition based on taxon biomass (biomass/m^2) were related to variations in CV-depth and leaf litter% (ρ = 0.59) (Table 6).

4 | DISCUSSION

4.1 | Heterogeneity and diversity

Instream habitat enhancement projects that rely on the installation of LWM are based on the hypothesis that changes enhancing substrate heterogeneity will result in increased biodiversity (species richness and diversity). This assumption has been questioned by many (Table 1). The results of the present study partially supported the first hypothesis by showing how the Rolleston Brook LWM rehabilitation project was initially successful in enhancing reach-level channel morphology and instream biotope heterogeneity. The installation of wood pieces was successful in reducing downstream transport of leaf litter, and it was particularly important in increasing leaf-litter biotope proportion at the reach level, especially during leaf fall in autumn. The biotic community is heavily dependent on detritus that enters the stream during the autumn season. LWM reduces the distance travelled by newly fallen dry leaves as much leaf litter is dry when it first enters a stream (Smock, Metzler & Gladden, 1989). The amount of leaf litter that enters the reach is not as important as the ability of the reach to retain it within the stream (Cummins et al., 1989).

Installation of woody material was also successful in dissipating flow energy, enhancing the stability of the stream bed through controlling the distribution of silt, and the appearance of coarse biotopes

TABLE 4 Results of similarity percentages (SIMPER) analysis based on macroinvertebrate community taxonomic composition (biomass/m^2) data (at family level), identifying most affected families by morphological changes and contributing at least 70% of the temporal dissimilarity in community composition of the rehabilitated reach

	Average biomass in spring (mg m^{-2}, dry mass)				
Family	2014	2015	2016	Contribution (%)	Cumulative (%)
Spring 2014 vs 2015					
Average dissimilarity: 69.8					
Gammaridae	57	218**		34	34
Limnephilidae	12	82*		18	52
Chironomidae (non-Tanypodinae)	21	93*		16	68
Erpobdellidae	7	48*		10	78
Spring 2014 vs 2016					
Average dissimilarity: 63.4					
Gammaridae	57		194**	43	43
Erpobdellidae	7		73*	23	65
Limnephilidae	12		78*	22	87
Spring 2015 vs 2016					
Average dissimilarity: 44.4					
Gammaridae		218	194	24	24
Chironomidae (non-Tanypodinae)		93	10*	18	42
Erpobdellidae		48	73	17	59
Limnephilidae		82	78	17	75

*$P < 0.05$; **$P < 0.005$; ***$P < 0.0005$.

for different species, thus increasing the instream biotope diversity. These results are in line with those of previous studies that found installation of LWM increased storage of organic matter and sediments, improved bed and bank stability, and enhanced the appearance of new instream biotopes (Gregory et al., 1991; Maohua, Tarmi & Helenius, 2002; Lester, Wright & Jones-Lennon, 2007).

Silt is often the key driver in the instream biotope composition of a rehabilitated reach. The rehabilitated stream had a high sediment load, which was not addressed by LWM installation. Therefore, it is likely that the LWM played a secondary role, controlling only the distribution of silt locally, as was also observed in the study of lowland streams by Thompson (2015). The scale of this was demonstrated when in-stream biotope composition is compared between the first and the second post-rehabilitation spring seasons: when LWM washed away, gravel patches became covered by silt again. This resulted from catchment-scale factors, such as erosion from agricultural land and an upstream village. This silt deposition adversely affected the instream biotope composition by covering coarser biotopes (cobbles and gravels) in the absence of LWM sequestering it behind dams.

Chironomids are generally tolerant of pollution and silt, so they are not a good target indicator for rehabilitation (Thompson, 2015), but their relative abundance and biomass responded uniquely to instream biotope changes. Significant increases in the proportions of their density and biomass during the first post-rehabilitation spring samples were related to increases in organic matter on trapped sediments behind the woody dams, as the stepped channel profile created by dams reduced water velocity by dissipating the flow energy (Heede, 1972; Keller & Swanson, 1979). This slowed suspended particles throughout the reach and thus facilitated their retention and settling behind dams (Smock, Metzler & Gladden, 1989), in addition to increasing the stability of the silty patches.

There was good support for the hypothesis that macroinvertebrate communities were enhanced by LWM installation at the reach scale. The significant increases in total density, total biomass, and taxon richness of macroinvertebrates in this study are contrary to the growing body of literature, which reports only minor effects of stream rehabilitation processes on macroinvertebrates (Hilderbrand et al., 1997; Roni et al., 2006; Testa, Shields & Cooper, 2011; Leal, 2012; Pinto et al., 2019), but they are consistent with some studies that compared rehabilitation outcomes with nearby natural systems and/or the status before rehabilitation. Smock, Metzler & Gladden (1989) found that macroinvertebrate abundance and biomass increased with an increase in the number of woody dams that led to the collection of organic matter and increased food availability in two low-gradient, headwater streams on the Coastal Plain of Virginia, USA. Entrekin et al. (2009), in a study of the effects of LWM installation in three forested headwater streams in the Upper Peninsula of Michigan, USA, found a significant increase in macroinvertebrate biomass in one rehabilitated reach, with no changes in two other reaches, compared with values before LWM installation. Thompson (2015) also found that

TABLE 5 Summary of sequential tests, obtained from distance-based linear models, seeking relationships between temporal variations in macroinvertebrate metrics and channel morphological variables. Values displayed indicate the proportion of variability explained by each morphological variable, and the cumulative variability explained by the models

Macroinvertebrate community data	Morphological variables added to model	Proportion	Cumulative	Relationship
Total density				
Sp.14:Sp.15	+Leaf litter%	0.77*	0.77	Positive
	+CV-width	0.08*	0.85	Positive
	+Gravels%	0.09*	0.94	Positive
Sp.14:Sp.16	+Leaf litter%	0.09	0.09	Positive
Sp.15:Sp.16	+Leaf litter%	0.83*	0.83	Positive
	+SWI-biotope	0.11*	0.94	Positive
Total biomass				
Sp.14:Sp.15	+Leaf litter%	0.92*	0.92	Positive
	+CV-width	0.03	0.95	Positive
Sp.14:Sp.16	+Leaf litter%	0.85*	0.85	Positive
Sp.15:Sp.16	+Leaf litter%	0.75*	0.75	Positive
Taxon richness				
Sp.14:Sp.15	+Gravels%	0.65*	0.65	Positive
Sp.14:Sp.16	+Number of biotopes	0.22*	0.87	Positive
	+Silt%	0.08	0.95	Negative
Sp.15:Sp.16	+SWI-biotope	0.08	0.08	Positive
	+SWI-biotope	0.45*	0.45	Positive
Chironomidae count%				
Sp.14:Sp.15	+Cobbles%	0.09	0.09	Negative
	+Wet surface area	0.07	0.16	Negative
	+CV-depth	0.07	0.23	Negative
Sp.14:Sp.16	+Number of biotopes	0.73*	0.73	Negative
Sp.15:Sp.16	+Leaf litter%	0.07	0.07	Positive
Chironomidae biomass%				
Sp.14:Sp.15	+Cobbles%	0.09	0.09	Negative
	+Gravels%	0.09	0.18	Negative
Sp.14:Sp.16	+Gravels%	0.93*	0.93	Negative
Sp.15:Sp.16	+Cobbles%	0.09	0.09	Negative

*$P < 0.05$. Sp.14: spring 2014; Sp.15: spring 2015; Sp.16: spring 2016; CV: coefficient of variation; SWI: Shannon–Wiener diversity index.

TABLE 6 Optimal BIOENV selected morphological variables with total Spearman's rank correlation coefficient (ρ) for temporal variations in macroinvertebrate community composition

	Sp.14:Sp.15		Sp.14:Sp.16		Sp.15:Sp.16	
Macroinvertebrate community data	Variable	(ρ)	Variables	(ρ)	Variables	(ρ)
Taxonomic composition (count/m^2)	Wet surface area	(0.61)	Leaf litter%	(0.31)	Leaf litter%	(0.39)
	Leaf litter%		Silt%		Wet surface area	
			Gravels%			
Taxonomic composition (biomass/m^2)	CV-depth	(0.59)	SWI-biotope	(0.22)	Leaf litter%	(0.43)
	Leaf litter%		Gravel%		Wet surface area	

Sp.14: spring 2014; Sp.15: spring 2015; Sp.16: spring 2016; ρ: Spearman's rank correlation; CV: coefficient of variation; SWI: Shannon–Wiener diversity index.

macroinvertebrate biomass increased significantly after LWM installation in rehabilitated reaches of five chalk streams in England, and significant increases in total density and richness were observed in studies by Gerhard & Reich (2000), Pretty & Dobson (2004), and Lester et al. (2007).

4.2 | Methodology of assessing rehabilitation effectiveness

Using macroinvertebrate community composition as a response variable was effective in gauging outcomes towards the project goals. Macroinvertebrate community composition (depending on taxon density and biomass) was more effective than only taxon richness and/or diversity metrics (the most commonly used metrics in the literature) for understanding the relationship between LWM installation and macroinvertebrate community responses. Changes in the community composition of the rehabilitated reach led to greater similarity to the condition of the semi-natural reach. Macroinvertebrate community composition was an effective response variable to gauge the outcome of integrated catchment management effects on macroinvertebrates in four streams of Waikato, New Zealand (Quinn et al., 2009), and in rehabilitation of former sewage channels in the Emscher River (right tributary of the River Rhine, Germany), towards 'Good Ecological Potential' (Winking, 2015). Enhanced macroinvertebrate community composition in the present study was a result of increased space (wetted surface area), instream biotope heterogeneity (changes in biotope %, especially gravel and silt proportion), and/or increased resources (detritus or leaf litter).

These results strengthen the call for a broader range of macroinvertebrate metrics (structural and functional) to be used to understand the ecological effects of LWM installation, rather than depending only on diversity and/or richness measures. Thus, monitoring rehabilitation outcomes must not focus solely on the structural attributes (taxon richness, diversity) of macroinvertebrate communities (Muhar et al., 2016); it needs to focus more on assessing natural processes and changes in their dynamic shifts towards the targeted rehabilitation endpoint through inclusion both of structural and functional measures (Palmer et al., 2005; Feld et al., 2011).

Collecting samples in a BAC study design proved to be an effective way of comparing before–after ecological changes induced by installation of LWM, and gauging the direction of the changes towards the conditions of the semi-natural reach which was used as an ecological baseline. The lack of changes in the measured macroinvertebrate structural and functional metrics of the semi-natural reach (before *vs* after) provided a good indicator that the positive changes in the rehabilitated reach were specifically induced by the morphological effects of the LWM installation. Gauging the direction of the changes towards the semi-natural reach was a key to determining successes or failures of the project. It is important to monitor the direction and not only the extent of change in biotic communities.

The overall changes in the morphological parameters of the rehabilitated reach did not explain a significant proportion of the changes in its macroinvertebrate metrics. At the instream biotope-level, small increases in the number of instream biotopes (appearance of gravel and leaf litter), and changes in biotope proportions (decreasing silt%) were significantly related to changes in the macroinvertebrate community metrics of the rehabilitated reach. This shows that the effects on macroinvertebrates could be related to changes in the cover of those specific biotope types. Verdonschot et al. (2015) claimed that 'the effects on macroinvertebrates could be related to changes in the cover of specific substrate (here biotope) types in the rehabilitated section'. It is necessary, therefore, to sample all available instream biotopes to collect biotic data representative of the study reach and to enable quantification of the changes. As macroinvertebrate species often have specific and changing instream biotope requirements throughout their life, all these habitats must be present and of sufficient quality to guarantee recolonization and the development of sustainable populations (Verdonschot et al., 2015). Our finding thus confirms that inadequate sampling could be a factor behind the negative outcomes of many previous studies. For example, evaluation studies often collect biotic samples in riffles, and yet these habitats are less likely to change as a result of most habitat enhancement projects (Brooks et al., 2002; Palmer, Menninger & Bernhardt, 2010), but improvements can be more evident when sampling is stratified by biotope type (Nakano & Nakamura, 2008; Sundermann, Stoll & Haase, 2011; Winking, 2015).

4.3 | Implications for river conservation

This study was very small — testing the effectiveness of a single hydromorphological improvement added to a small headwater stream. Its findings may be somewhat limited by catchment processes outside the rehabilitation scale, as well as the limited temporal scale of the evaluation. The upstream reach of the rehabilitated site was severely degraded, however, as are many headwater streams in the developed world. Such streams are often the subject of severe agricultural alteration, such as straightening to run as field boundaries (as in this case) and subject to sewage input (in this case at its source). The condition of the stream might have limited the rehabilitation potential (because that depends on available taxonomic diversity), but the results show that simple rehabilitation by addition of LWM can produce a substantial improvement. The temporal scale of this study may have been short (it included one before and two successive spring seasons following LWM installation), which limited the ability of the study to detect responses in some taxa or longer term trends. Importantly, however, it did show how rapidly biodiversity improvements can begin. This latter result is an important guide to post-project monitoring, shown by a recent review to be necessary but infrequently implemented (Al-Zankana, Matheson & Harper, 2020).

The location of the study — a tributary of the Stonton Brook, itself a tributary of the Welland River — is one of the headwater catchments of the Water Friendly Farming project in the UK, a long-running project that seeks to quantify the extent to which modern farming can increase landscape-scale biodiversity by implementing water-retention devices, such as ponds (Biggs et al., 2016). LWM is equivalent to Water Friendly Farming 'leaky dams', which are increasingly also proposed as floodwater retention features in intensively managed landscapes in which rainwater is lost downstream too quickly, causing flooding and economic damage. To be most effective, 'leaky dams' are necessary on a much larger scale than was investigated here, but the evidence provided by this study for their success in terms of biodiversity, as well as the evidence for impermanence, broaden and strengthen the case for their construction in streams. It is not surprising that the evidence for landscape-scale benefits of the reintroduction of beavers into the UK (Wilson et al., 2020) centres around their construction of 'leaky dams' on a large scale.

These results indicate how simple measures of physical heterogeneity are effective in assessing biodiversity in rivers, regardless of whether or not they are specifically applied to a rehabilitated project, as they are here. The EU WFD requires measures of 'hydromorphology' (Boon, Holmes & Raven, 2010), which underpin the identification of 'water bodies'—the units of ecological quality assessment. Several approaches have been made to use effective measures of hydromorphology, such as those of Shuker, Moggridge & Gurnell (2015) for heavily modified urban rivers and the earlier developed UK River Habitat Survey (RHS) in rural areas (Clews, Vaughan & Ormerod, 2010). The biotopes used here are incorporated into RHS (Environment Agency, 2003) but offer a much more simple assessment of physical diversity, identifying the river channel in a 'jig-saw' fashion (Kemp, Harper & Crosa, 2000).

Specific assessment of rivers for their biodiversity conservation value under the European Habitats Directive (Council of the European Communities, 1992) — as opposed to classifying all water bodies by their ecological status under the WFD — has been recommended by agencies in the British Isles, often in partnership with other agencies. For example, restoration of the rivers that form the Natura 2000 sites in England (part of the European network under the Habitats Directive) has been recommended by Natural England (Wheeldon et al., 2015) and both conservation and restoration of rivers in the UK and the Republic of Ireland recommended by the International Union for Conservation of Nature (IUCN) in partnership with the four constituent countries' water agencies (Addy et al., 2016). In England, there are 338 Natura 2000 sites covering 2,076,875 ha. These are protected under European legislation for their important wildlife and habitats. Forty-four rivers in England (~2,500 km) are legally protected as Sites of Special Scientific Interest (SSSIs; Wheeldon et al., 2015). These provide a useful platform for demonstrating large-scale strategic approaches to river conservation (Mainstone, 2008). Approximately 1,684 km of SSSI rivers are designated as Special Areas of Conservation under the Habitats Directive (Council of the European Communities, 1992) and therefore require concentrated action under European law. There are also domestic objectives to restore SSSI condition, most recently as part of the UK Government's Biodiversity 2020 agenda (Department of Environment, Food and Rural Affairs, 2011). Restoration of SSSIs is assigned to relevant parties and tracked by Natural England — the government's advisory body for the natural environment, sponsored by the Department for Environment, Food and Rural Affairs — to ensure that schemes are implemented adequately (Mainstone & Wheeldon, 2016).

The UK conservation agencies, recognizing the importance of physical habitat in underpinning biodiversity richness and ecosystem functioning, use broad habitat divisions, such as 'backwaters', 'aquatic macrophyte beds', 'tree roots', and 'in-stream woody material', but do not describe habitats covering the entire channel, despite using the term 'building blocks' (after Harper, Smith & Barham, 1992) and describing rivers as 'a patchwork of linked habitats called a "habitat mosaic"' (Addy et al., 2016). The assessment of all biotopes in a river — as illustrated in Demars et al. (2012) — has been shown here to clearly measure rehabilitation effectiveness. The full inclusion of biotopes, at reach level, would add a hierarchical layer that is at present missing in recommendations for conservation and restoration of rivers (Mainstone & Wheeldon, 2016). For example, the Wensum, in East Anglia (England), one of the country's most important chalk streams, was hailed as a case study of successful restoration brought about by many activities, such as re-meandering and gravel reinstatement (Mainstone & Wheeldon, 2016). However, no evidence has been produced for these claimed successes, anywhere along the length of the river; biotope level is the simplest. The citizen science programme Riverfly (Brooks et al., 2019) is now being enlarged to seek to fill this gap (D. M. Harper, personal observation).

Successful river conservation and restoration of degraded systems require the ability to quantify status and success. Landscape-scale methodologies have been developed, such as RHS in the UK (Environment Agency, 2003) and the IUCN National Committee UK river restoration strategy (Addy et al., 2016). All such large-scale approaches depend upon the smallest components being accurate, reliable, and replicable. Biotopes represent the smallest hydro-morphological scale at which macroinvertebrate community structure or function can be quantified, and so are crucial for the success of large-scale methods. This study has shown that the addition of one biotope, LWM, to a degraded stream stretch causes an increase in the overall biotope diversity. Visually, this is seen as the accumulation of leafy material behind the LWM and of silt patches in specific areas of low flow velocity rather than spread widely over the river bed. That diversity increase causes a significant improvement in invertebrate community matrices, thus clearly demonstrating that biotope diversity is a reliable and simple method for evaluating biological conservation value (and restoration success) in rivers.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of this article.

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