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Milner, V.S., Hill, M.J., Gething, K.J. and Cunningham, S.B. (2025) 'What a load of rubbish: The impact of anthropogenic litter on urban freshwater diversity' *Environmental Pollution*, 373, article 126097.

Environmental Pollution 373 (2025) 126097

Contents lists available at ScienceDirect

Environmental Pollution

journal homepage: www.elsevier.com/locate/envpol

What a load of rubbish: The impact of anthropogenic litter on urban freshwater diversity☆

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ARTICLE INFO

Keywords:
Litter sources
Plastics
Biodiversity conservation
Urban rivers
Canals
Rubbish

ABSTRACT

The abundance of anthropogenic litter (AL) in aquatic environments is an increasing global problem. Most research on the effects of AL has focussed on marine environments, with research examining the ecological effects of AL in freshwater ecosystems largely missing. Our study examines the impacts of AL on macroinvertebrate diversity in two urban freshwater systems in West Yorkshire, in the United Kingdom. Two urban river and two canal reaches were surveyed for macroinvertebrates from the bed sediments of riffles (in a river), open water and macrophyte habitats (in a canal), and AL items (from both freshwaters). We found higher local taxonomic richness and differences in community composition in 1) the bed sediments of riffles compared to AL items within urban rivers, and 2) open water and emergent vegetation than AL items within our canal reaches. Taxonomic richness was higher on metals and plastics in urban rivers than canal reaches, and macroinvertebrate community composition was distinct between AL types (e.g., fabrics and metals, plastics and polystyrenes), possibly due to differences in structure, shape and texture. AL items in both urban freshwaters supported unique taxa, indicating that AL items may provide a habitat for macroinvertebrates where physical habitat complexity is poor. The restoration of natural physical habitat and removal of AL should be a key priority for freshwater conservation. However, rinsing AL items prior to removal during litter clean-ups is essential to minimise any potential macroinvertebrate loss. In urban freshwaters, physical habitat could be increased by gravel augmentation, installing large wood or boulders.

1. Introduction

Anthropogenic litter (AL) is increasing in both freshwater and marine ecosystems globally (Pace et al., 2024) and has negative impacts on ecological organization and biodiversity (Kühn et al., 2015; Conchubhair et al., 2019; Palmas et al., 2022). AL is defined as solid, manufactured, and persistent waste (McCormick and Hoellein, 2016), including plastics, rubber, glass and metals. The main impacts of AL on aquatic organisms includes ingestion, entanglement, entrapment, and inhalation (Poeta et al., 2017; Rech et al., 2018; Battisti et al., 2019; Roman et al., 2022). These effects may cause mortality, strangulation, drowning and abrasion, a reduced ability to escape predation, decreased food consumption and lower reproductive performance for aquatic organisms (Derraik, 2002; Gregory, 2009; Gall and Thompson, 2015). AL can also affect aquatic biota through exposure to harmful chemicals through pollutant leaching (Rochman, 2015). Several hazardous substances, including heavy metals, organic based colourants, flame retardants, fluorinated compounds, persistent organic pollutants and phthalates are found in plastic products to improve their performance and functionality (Stenmarck et al., 2017; Hahladakis et al., 2018), but may be released into aquatic environments, degrade into smaller fragments and be ingested by macroinvertebrates and fish (Teuten et al., 2009). Most previous research has focussed on the accumulation and effects of AL in marine environments (Wendt-Potthoff et al., 2020; Honorato-Zimmer et al., 2021; Palmas et al., 2022; Ferreira, 2024), with the impacts of AL in many freshwater systems, such as rivers and canals receiving little research attention. However, these freshwater systems are important stores of AL and facilitate their movement between terrestrial and

☆ This paper has been recommended for acceptance by Eddy Y. Zeng.

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https://doi.org/10.1016/j.envpol.2025.126097
Received 6 January 2025; Received in revised form 21 February 2025; Accepted 19 March 2025
Available online 19 March 2025
0269-7491/

marine ecosystems (European Commission - Joint Research Centre, 2016; Lebreton et al., 2017; Meijer et al., 2021).

Many urban freshwaters, such as rivers and canals are characterised by low habitat diversity due to dredging, vegetation control and bed/bank stabilisation works (Willby et al., 2001; Grizzetti et al., 2017; Ekka et al., 2020; Wilson et al., 2021). Urban rivers commonly have steep, unvegetated and modified channel banks, a lack of in-channel woody debris, uniform bed sediments, and increased channel widths and depths (Booth et al., 2016; Vietz et al., 2016). In comparison, canals are typically straight with uniform planforms and cross sections, fine bed sediments, and unvaried and slow flows (Bennett, 2009; Randima et al., 2017; Walker and Hassall, 2021; Tölgyesi et al., 2022). These morphological and hydraulic changes have reduced habitat complexity and often cause low macroinvertebrate diversity and modified assemblage composition (known as the urban stream syndrome; Walsh et al., 2005; Bohus et al., 2023).

Urban freshwaters receive disproportionately large inputs of AL, which although undesirable, may increase habitat diversity in degraded freshwater systems by providing complex shapes and textures (Wilson et al., 2021). The range of AL items, includes plastics, metals, rubber, polystyrenes, masonry, fabrics, wood and concrete. AL items and materials may replace natural habitats, cause their fragmentation and lead to disappearances of specific taxa and declines of local biodiversity and changes to community composition (Beavan et al., 2000; Airoldi et al., 2005; Brauns et al., 2007). Alternatively, AL items may generate localised changes in flow patterns, create new habitats, and increase the surface area of habitats available for invertebrate colonisation (Brauns et al., 2007; Francis and Hoggart, 2008). Objects with high surface structural complexity are recognised to support diverse macroinvertebrate communities due to increased shelter from hydraulic stress (Brooks et al., 2005) and predation (Everett and Ruiz, 1993). Therefore, AL items possessing more complex physical structures (e.g., presence of cracks or crevices) and/or interiors (e.g., bottles and cans) may provide areas of low hydraulic stress and minimise the risk of accidental entrainment or predation. Previous research has illustrated the colonisation of artificial structures, surfaces, and substrates by macroinvertebrates, providing these materials are non-toxic (Schmude et al., 1998; Hoggart et al., 2012; Wilson et al., 2021). For instance, macroplastics have been found to be a food source where grazers have feed on the established biofilm (Michler-Kozma et al., 2022). Other studies have identified macroplastics to be a habitat or a refuge (Artru and Lecerf, 2019; Wilson et al., 2021). Gallitelli et al. (2023) found macroinvertebrates colonised the inside and outside of plastic bottles in the Tiber River, in central Rome, Italy, whereas Kennedy and El-Sabaawi (2018) discovered macroinvertebrates inhabiting styrofoam and plastic bags (high-density polyethylene) in four urban streams in Victoria, British Columbia, Canada. However, studies on the effects of AL on taxonomic richness and community composition in freshwaters and marine ecosystems have generated mixed findings. Comparable epifaunal communities have been found on artificial substrates and natural habitats with similar physical properties (shape and structure; Jacobi and Langevin, 1996; Chapman and Clynick, 2006). Whilst other studies have demonstrated distinct faunal compositions between artificial and natural substrate (e.g., Hoffer and Richardson, 2007; Czarnecka et al., 2009; Wilson et al., 2021). Wilson et al. (2021) also found that material type can affect macroinvertebrate diversity, with the highest taxonomic richness recorded on fabrics and plastics, and the lowest richness recorded from glass.

We aimed to examine the effect of AL on aquatic macroinvertebrate taxonomic richness and community composition in urban rivers and canals. Specifically, this study identifies whether macroinvertebrate taxonomic richness differs between natural habitats (i.e., bed sediments in urban rivers and open water/vegetated mesohabitats within canals) and AL items (plastics, fabrics, metals, polystyrenes, rubber and other AL items) within and between urban rivers and canals. We hypothesised that.

1. Macroinvertebrate communities in natural habitats will have a higher local taxonomic richness and gamma diversity compared to macroinvertebrate communities inhabiting AL within urban rivers and canals due to differences in habitat heterogeneity and local environmental conditions.
2. Macroinvertebrate community composition in natural habitats will differ to the composition recorded in AL items within urban river and canal reaches.
3. Plastics and fabrics will support higher macroinvertebrate taxonomic richness and a different community composition to metals, polystyrenes and other AL items in urban rivers and canals.

2. Material and methods

2.1. Study area

Fieldwork was carried out on the River Holme, the River Colne and Huddersfield Narrow Canal, in West Yorkshire, UK (Fig. 1). Fieldwork was conducted between the 18th of October and the November 5, 2021 on four reaches (one urban river reach on the R. Holme, one urban reach on the R. Colne and two canal reaches on Huddersfield Narrow Canal; Fig. 1). The R. Holme and R. Colne have catchment areas of 97.4 km^2 and 245 km^2 respectively (National River Flow Archive [NRFA], 2021). The R. Holme catchment comprises grassland (50.4 %), woodland (19.8 %), mountain/heath/bog (13.0 %), urban extent (10.40 %) and arable/horticultural land (4.2 %; NRFA, 2021), and the R. Colne catchment has similar land uses and proportions. Mean annual discharge was 2.18 m^3/s for the R. Holme at Huddersfield Queens Mill gauging station (based on gauged daily flow data between 1979 and 2020; NRFA, 2021) and 4.44 m^3/s for the R. Colne (1964–2020) at the Colne Bridge gauging station (NRFA, 2021). Both catchments receive similar mean annual precipitation totals of 1237 mm (for the R. Holme; 1961–1990, NRFA, 2021) and 1144 mm (for the R. Colne; 1961–1990, NRFA, 2021), and an average minimum and maximum temperature of 5.49 and 12.05 °C respectively (1991–2020, UK Meteorological Office, 2023).

Both urban river reaches are characterised by pool-riffle topography with a meandering planform and are at an elevation of 65–80 m above sea level. In 2019, the Environment Agency classified the R. Holme as possessing a moderate ecological status and a heavily modified hydromorphology (Environment Agency, 2019). Both the urban reaches have a high proportion of reinforced and resectioned banks. The two reaches on Huddersfield Narrow Canal are situated at an elevation of 80–90 m above sea level. Both canal reaches are characterised by reinforced banks constructed of concrete and bricks. Both the river (mean water width 11.9 m; mean water depth 0.23 m) and the canal reaches (mean water width 9.0 m; mean water depth 0.63 m) are comparable in their width and depth.

2.2. Field procedure

Two urban reaches on the R. Holme, the R. Colne and two reaches on Huddersfield Narrow Canal were sampled for aquatic macroinvertebrates. In the UK, the standard national monitoring of sampling macroinvertebrate communities takes place in spring (March–May) and autumn (Environment Agency, 2014; McKenzie et al., 2022). Macroinvertebrate samples collected in autumn surveys typically exhibit greater macroinvertebrate diversity and richness compared to other seasons (Environment Agency, 2014). Hence, our macroinvertebrate and hydraulic sampling occurred in mid-autumn.

In this study, reach length was defined as 10 times the mean width of the active channel. On the R. Holme and the R. Colne, the active channel was determined as the area of the channel with inundated or exposed bed sediments and the absence of abrupt changes in slope (Corti and Datry, 2016). At each river reach, macroinvertebrates were sampled for 1 min equidistantly at three locations across a transect of a riffle. Macroinvertebrates were collected using a Surber sampler (size: 0.3 × 0.3 m,

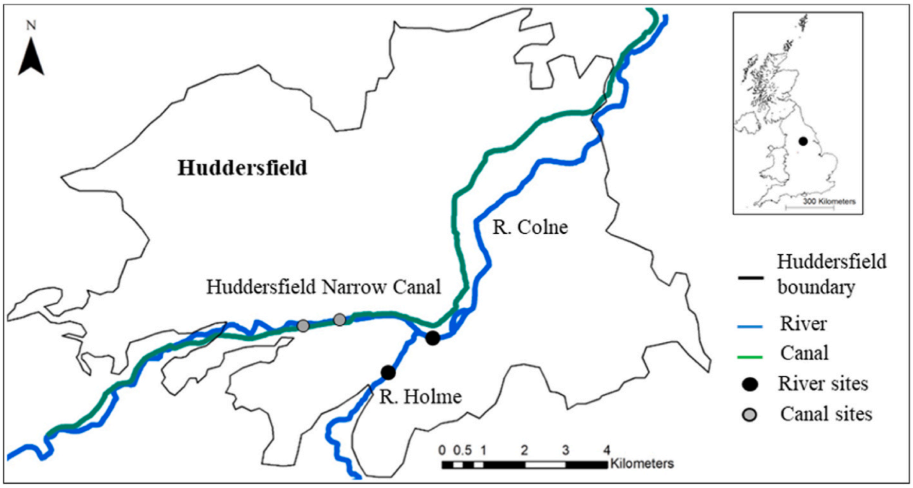

Fig. 1. Location of the sampling sites on the R. Holme, the R. Colne and on the Huddersfield Narrow Canal.

sampling area: 0.1 m^2, mesh size: 1 mm) and were collected from 3 riffles per river reach. In total, 18 macroinvertebrate samples were collected from the urban rivers (3 macroinvertebrate samples per riffle x 3 riffles per reach x 2 reaches = 18 macroinvertebrate samples).

In the two canal reaches, aquatic macroinvertebrates were collected for a total of 3 min using a hand-net (1 mm mesh) and sweep sampling technique from the edge of the bank. For each canal reach, the presence and spatial coverage (percentage of surface area) of mesohabitats were classified into (i) open water, (ii) submerged vegetation, (iii) emergent vegetation and (iv) floating vegetation (Hill et al., 2015). At each canal reach, only emergent vegetation and open water mesohabitats (which dominated the reaches) were present. Reflecting the dominance of the open water mesohabitat, macroinvertebrates were sampled from two open water (2 × 1 min sampling time) and one emergent vegetation mesohabitat (1 min sampling time). Throughout each canal reach, macroinvertebrate sampling (for 3 min) took place at the upstream, mid and downstream section (3 locations). Our sampling strategy consisted of collecting 3 macroinvertebrates samples (2 x open water and 1 emergent vegetation mesohabitat) x 3 different locations (upstream, mid and downstream section) within 2 reaches (a total of 18 macroinvertebrate samples).

For both the urban river and the canal reaches, AL was collected by retrieving items at the upstream end of the reach and progressing downstream to avoid cross contamination. Every piece of AL encountered within each of the four reaches were collected. Large AL items, such as construction materials (e.g., hardboard, piping, guttering) and large pieces of fabric, were sampled for macroinvertebrates *insitu* by placing a hand-net immediately downstream and scrubbing the item for 1 min, allowing any macroinvertebrates to flow into the net. The dimensions of all large AL items were measured in the field. Small AL items (i.e., <40 × 30 cm) were collected, placed into resealable plastic bags, and filled with industrial methylated spirits (IMS) at a concentration of 70 %, and cold stored at ~ 5 °C within a laboratory. A total of 214 items of AL were sampled for macroinvertebrates in the studied reaches. Most AL items were of a small surface area (mean of 0.28 m^2), and thus, comparable to the sampled riffle habitat (Surber sampler: 0.3 × 0.3 m).

Bankfull height, wetted width and bankfull width of the channel were measured at each riffle and mesohabitat (i.e., emergent vegetation and open water for canal reaches). The grain size distribution of bed sediments within riffles was determined by measuring the b axis of 50 random bed particles. At each macroinvertebrate and AL sampling location, water depth and velocity (m s^{-1}) were measured using a Valeport Model 801 EM Current Meter for 20 s at 0.6 depth. Water quality, including temperature (°C), pressure (mmHg), dissolved oxygen (%), conductivity (SPC) and pH were also recorded at each macroinvertebrate and AL sampling location using a YSi probe (YSi v.556).

2.3. Laboratory procedure

In a laboratory, all AL items were rinsed with distilled water through a series of sieves of decreasing mesh sizes (smallest mesh size of 500 μm) to remove debris and sediment and collect macroinvertebrates. The rinsed AL items were stored in resealable plastic bags and labelled for further analysis and surface area calculation. Macroinvertebrates were identified to the lowest taxonomic resolution possible, which was typically to genus or species levels. The exceptions were Chironomidae, Lumbricidae, Physidae, Neritidae, Hydrobiidae, Acroloxidae, Sphaeriidae, Lymnaeidae, Erpobdellidae, Sciomyzidae, Psychodidae, and Zonitidae (which were all identified to family level), Oligochaeta and Hirudinea (identified to subclass), Trichoptera (Instar 1–2; identified to order), Cladocera (super order) and Zooplankton.

The surface area of simple shaped AL items, such as cylinders, spheres and polygons were calculated using formulas for geometric shapes (Bergey and Getty, 2006). For large AL items, the dimensions were recorded in the field and the surface area calculated. The surface area of complex shapes was determined by covering the surface of an AL item with aluminium foil and subsequently weighing the foil pieces (1 g: 0.0214 m^2; Dudley et al., 2001; Wilson et al., 2021). Prior to wrapping, each AL item was rinsed and air dried to ensure that water, biofilm, or other debris would not impact the weight of the foil.

2.4. Statistical analyses

2.4.1. Examination of diversity within each freshwater type

Prior to any statistical analyses, the AL items collected in the urban river reaches were grouped into AL types: plastics, fabrics, metals, polystyrenes, rubber and other AL items (Table 1). Here, *local taxonomic richness* is defined as the taxonomic richness recorded from each sample

Table 1
Abundance of AL items in the urban study systems.

AL types	River reaches	Canal reaches
Plastics	42	75
Fabrics	11	–
Metals	8	23
Polystyrenes	–	27
Rubber	7	–
Other AL items	13	8
Total	**81**	**133**

and *gamma diversity* is defined at the total taxonomic richness recorded from all natural habitats and AL items. Local taxonomic richness was calculated for 1) the bed sediments of riffles (i.e., the natural habitat within the urban river reaches), and 2) the AL items within the urban river reaches. Mann-Whitney U-tests were used to examine differences in taxonomic richness between 1) the natural habitats (i.e., the bed sediments of riffles) and AL items, and 2) between AL types within the urban river reaches. Permutational Analysis of Variance (PERMANOVA) models were carried out to identify any statistical differences in macroinvertebrate community composition between 1) the bed sediments of riffles and river AL items, and 2) the AL types within the urban river reaches (using the *adonis* function in the vegan package; Oksanen et al., 2022).

In the canal reaches, AL items were classified into plastics, polystyrenes, metals and other AL items (Table 1). Like the analysis of the urban reaches, local taxonomic richness was determined for 1) the mesohabitats (open water and emergent vegetation), and 2) the AL types within the canal reaches. Subsequently, Mann-Whitney U-tests determined any significant variations in taxonomic richness between 1) the mesohabitats and the AL items, and 2) between the AL types within the canal reaches. In addition, PERMANOVA models were subsequently undertaken to determine differences in macroinvertebrate community composition between the mesohabitats and AL items, and between AL types within the canal reaches.

2.4.2. *Examination of diversity between freshwater type*

Estimated gamma diversity for the natural habitats and AL types within the urban rivers and canals were calculated using the Chao estimator (*specpool* function) in the vegan package. Differences in estimated gamma diversity were considered significant if the 95 % confidence intervals (CI) for natural habitats and AL types within urban rivers and canals did not overlap. To assess whether AL in the urban rivers or canal reaches contributed more significantly to landscape-scale biodiversity (i.e., diversity across the entire dataset), the number of taxa exclusive to AL in each freshwater system was compared.

PERMANOVA was undertaken to determine statistical differences in macroinvertebrate community composition between all AL types (from both the urban rivers and the canal). Principal Co-ordinate Analysis (PCoA) plots were used to visually compare macroinvertebrate communities between 1) natural habitats and AL items, and 2) different AL items within the urban river and canal reaches. Samples with zero macroinvertebrates were removed, which included 12 samples from the canal reaches in the plastics category, 2 canal samples in the metals category, and 4 canal samples in the other category. All PCoA plots were generated using the *metaMDS* function in the vegan package. Pairwise matrices of the contribution of species turnover and nestedness to total beta diversity (defined here as the difference in macroinvertebrate community heterogeneity among samples) between AL items and natural habitats among river and canal systems was calculated using the function *beta.div.comp* (based on the Baselga-Jaccard dissimilarity) from the adespatial package (Dray et al., 2019).

The association between the 13 environmental variables (e.g., temperature, surface area, conductivity, velocity, dissolved oxygen, pH, depth etc), natural habitats, AL items (e.g., rubber, fabrics, plastics, metals, polystyrenes), and macroinvertebrate communities was assessed using redundancy analysis (RDA). Prior to RDA, the environmental variables were examined for collinearity using the Pearsons correlation (*cor* function). No environmental variables were recorded to have a correlation above 0.6, and as a result all 13 environmental variable were included in subsequent analyses. A forward selection procedure was employed using the function *ordiR2step* in the vegan package to identify any significant ($p < 0.05$) variables influencing the aquatic macroinvertebrate communities All statistical analyses were undertaken in the R environment (R version 4.0.2; R Core Team, 2020).

3. Results

3.1. *Macroinvertebrate taxonomic richness in natural habitats and on AL items*

In total, 94 taxa were recorded across all sites from the study. 71 taxa were recorded from the urban river reaches (39 from the bed sediments and 63 from AL items) and 61 taxa (30 from the mesohabitats and 47 from AL items) were recorded from the canal reaches. The most dominant taxa in the urban river reaches were *Hydropsyche siltalai* (constituting 30.2 % of the community composition), *Polycentropus flavomaculatus* (8.1 %), Orthocladiinae (8.0 %), *Gammarus pulex* (6.0 %) and *Ancylus fluviatilis* (5.4 %). The most prevalent taxa in the canal reaches were *G. pulex* (comprising 51.0 % of the community composition), *Asellus aquaticus* (13.7 %), *Halesus* sp. (10.3 %), zooplankton (5.4 %), and Orthocladiinae (1.1 %).

Among AL items, estimated gamma diversity was not significantly different between the urban river (92 taxa, 95 % CI: 59.9–124.2) and the canal samples (83 taxa, 95 % CI: 39.1–127.8; Fig. 2). When the urban river and the canal sites were considered together, local taxonomic richness was significantly higher ($W = 2264.5$, $p < 0.001$) among all natural habitats (i.e., bed sediments and mesohabitats; median richness: 7.5 taxa) than AL items (median richness: 3 taxa; Fig. 3A; Table 2). When each freshwater system was assessed separately, local taxonomic richness was significantly higher in 1) the bed sediments of the urban river reaches (median richness: 12.5 taxa), and in 2) the mesohabitats in the canals (median richness: 3 taxa) than AL items (median richness – river: 7.1 taxa, canal: 2.25 taxa) in the urban river reaches ($W = 324$, $p < 0.001$; Fig. 3B) and the canal reaches ($W = 851.5$, $p = 0.044$; Fig. 3C).

AL items in the urban river reaches supported significantly greater local macroinvertebrate taxonomic richness ($W = 1662.5$, $p < 0.001$) than AL items in the canal reaches (Fig. 3D). When all AL items were grouped together (total of 214 items), only metals and plastics were found within both the urban river and canal reaches (Table 1). Local macroinvertebrate taxonomic richness was significantly higher on metals ($W = 24.5$, $p = 0.002$) and plastics ($W = 528$, $p < 0.001$; Fig. 4) in the urban river reaches (median richness; metal: 6.5 taxa, plastic: 5 taxa) than the canal reaches (median richness; metal: 2 taxa, plastic: 2 taxa). However, when canals and rivers were examined separately, no significant difference in local taxonomic richness was recorded between the

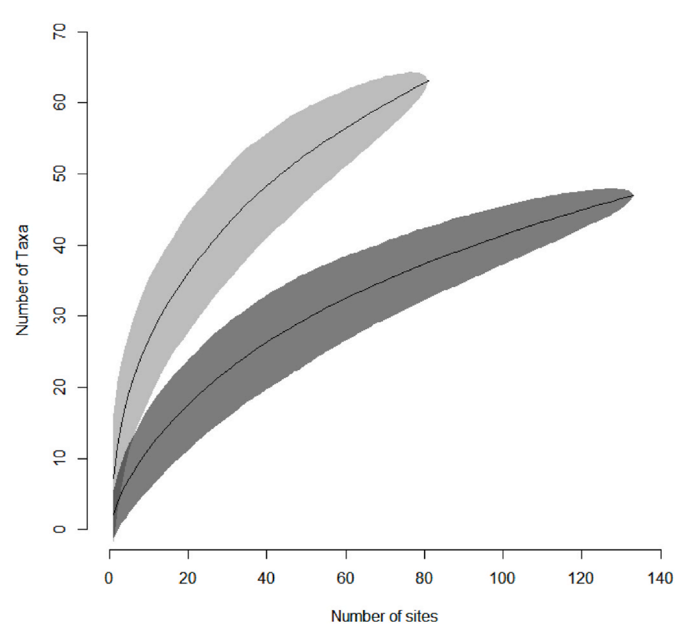

Fig. 2. Species accumulation curves of taxonomic richness among anthropogenic litter in urban rivers (light grey) and canal sites (dark grey).

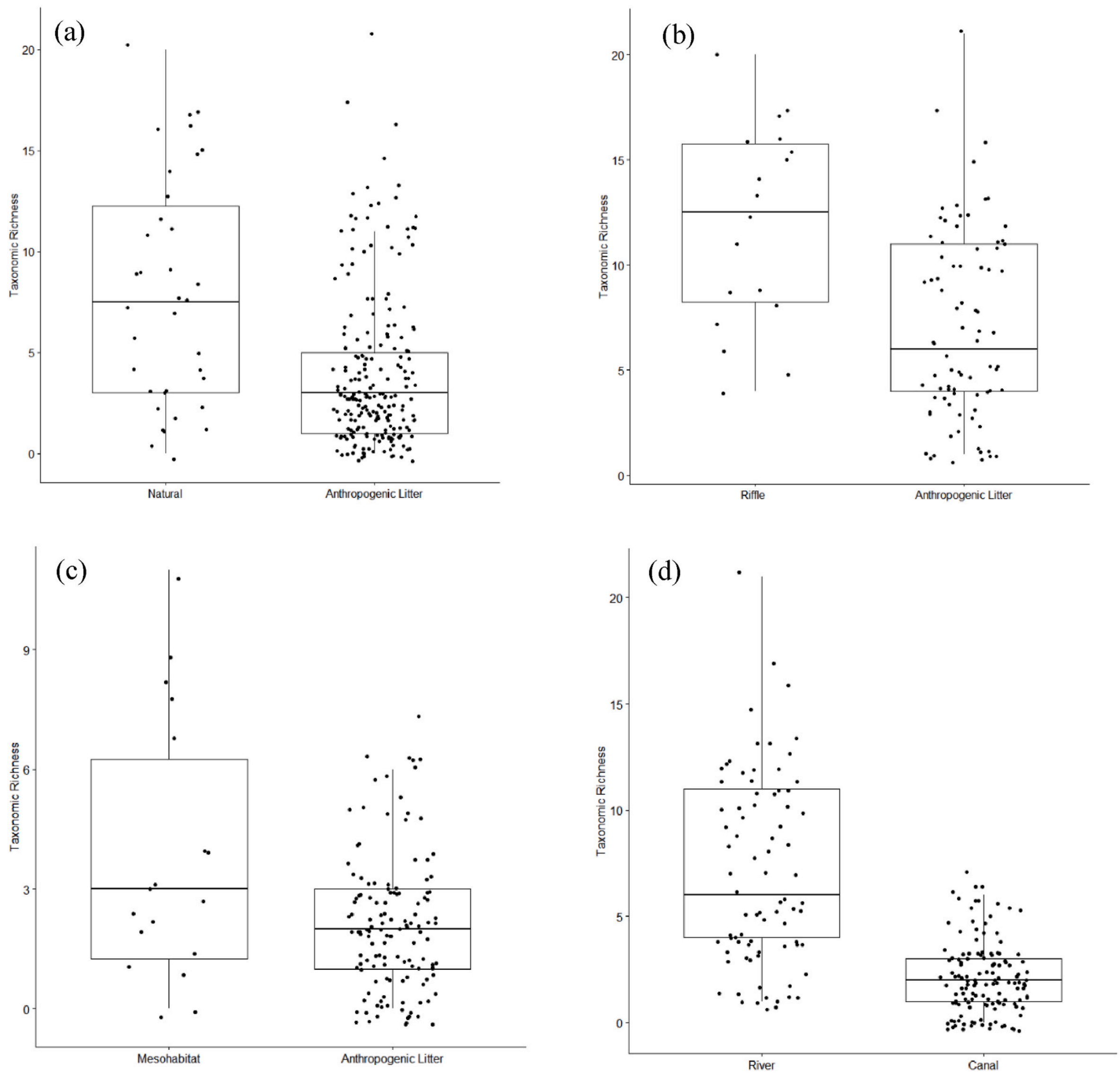

Fig. 3. Boxplots of taxonomic richness in natural habitats and AL (A) when the entire dataset was considered together, (B) among rivers sites (C) among canal sites, and (D) between canal and river sites. Boxes show 25th, 50th and 75th percentiles, upper whiskers shows 75th percentile +1.5 * IQR, and the lower whisker shows the 25th percentile −1.5 * IQR.

Table 2
Summary table of taxonomic richness of natural habitats and AL from urban river and canal sites.

Freshwater system				River						Canal				
		River	Canal	Natural habitats	Plastics	Metals	Fabrics	Rubber	Other	Natural habitats	Plastics	Metals	Polystyrenes	Other
Taxonomic Richness	Mean	7.9	2.8	11.9	6.6	6	10.2	8.7	5	4.3	2.5	1.9	3.3	2.1
	Median	7	2	12.5	5	6.5	11	9	4	3	2	2	3	2
	Std. Deviation	4.9	1.9	4.7	4.4	3.5	3.7	6.7	2.9	3.2	1.5	0.9	1.7	0.8
	Min	1	1	4	1	1	4	1	1	1	1	1	1	1
	Max	21	11	20	17	11	15	21	12	11	6	4	7	3

different types of AL within either the urban river (W = 10.91, p = 0.053) or canal (W = 6.073, p = 0.1) reaches (Table 1, Fig. S1).

3.2. *Macroinvertebrate community composition on AL items in urban rivers and canals*

A significant difference in macroinvertebrate community composition was recorded between natural habitats and AL items when both freshwater systems were considered together (PERMANOVA: R^2 = 0.019, $p < 0.01$). This finding was visually evident in the separation of natural habitats and AL sites in the PCoA biplot (Fig. 5A). When each freshwater system was considered separately, macroinvertebrate communities on AL items were significantly different to natural habitats in both the urban river (PERMANOVA: $R^2 = 0.127$, $p < 0.01$) and the canal

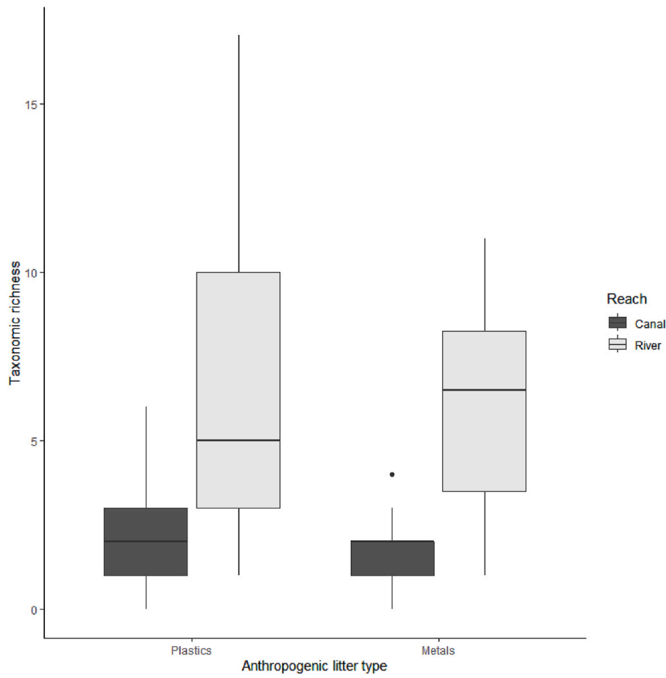

Fig. 4. Boxplots of taxonomic richness in metal and plastic AL among urban river and canal sites. Boxes show 25th, 50th and 75th percentiles, upper whiskers shows 75th percentile +1.5 * IQR, and the lower whisker shows the 25th percentile −1.5 * IQR.

reaches (PERMANOVA: $R^2 = 0.05$, $p < 0.01$), which is clearly demonstrated in the PCoA biplot (Fig. 5B).

Significant differences in macroinvertebrate community composition were recorded between AL types when all urban river and canal reaches were grouped together (PERMANOVA: $R^2 = 0.06$, $p < 0.01$). Pairwise PERMANOVA indicated that macroinvertebrate communities were significantly different between fabrics and plastics (pairwise PERMANOVA: $R^2 = 0.02$, $p < 0.01$), plastics and polystyrenes (pairwise PERMANOVA: $R^2 = 0.03$, $p < 0.01$), fabrics and metals (pairwise PERMANOVA: $R^2 = 0.08$, $p < 0.01$), fabrics and other AL items (pairwise PERMANOVA: $R^2 = 0.07$, $p < 0.01$), fabrics and polystyrenes (pairwise PERMANOVA: $R^2 = 0.13$, $p < 0.01$), metals and rubber (pairwise PERMANOVA: $R^2 = 0.05$, $p = 0.03$), metals and polystyrenes (pairwise PERMANOVA: $R^2 = 0.06$, $p < 0.01$), polystyrenes and other AL items (pairwise PERMANOVA: $R^2 = 0.08$, $p < 0.01$) and, rubber and polystyrenes (pairwise PERMANOVA: $R^2 = 0.12$, $p < 0.01$).

Among AL types in the urban river reaches, significant differences in macroinvertebrate community composition were recorded between plastics and other AL items (pairwise PERMANOVA: $R^2 = 0.05$, $p < 0.01$) and fabrics and other AL items (pairwise PERMANOVA: $R^2 = 0.17$, $p < 0.01$). No significant difference in macroinvertebrate community composition was recorded between any AL types in the canal reaches (PERMANOVA: $R^2 = 0.038$, $p = 0.09$). Most variation in macroinvertebrate community composition between the urban and canal sites (for all sites; total pairwise dissimilarity: 0.94) was explained by turnover (83 %) rather than nestedness (17 %). Similarly, when only AL samples were considered, most variation in macroinvertebrate community composition between the urban river and canal reaches (pairwise dissimilarity: 0.94) was explained by turnover (85 %) rather than nestedness (15 %). Differences in macroinvertebrate community composition between natural habitats and AL items when both freshwater systems were considered together (pairwise dissimilarity: 0.93), and within the urban river (pairwise dissimilarity: 0.78) and the canal reaches (total pairwise dissimilarity: 0.85) was driven by turnover (all sites: 80 %, river: 69 %, canal: 77 %) rather than nestedness (all sites: 20 %, river: 31 %, canal: 23 %).

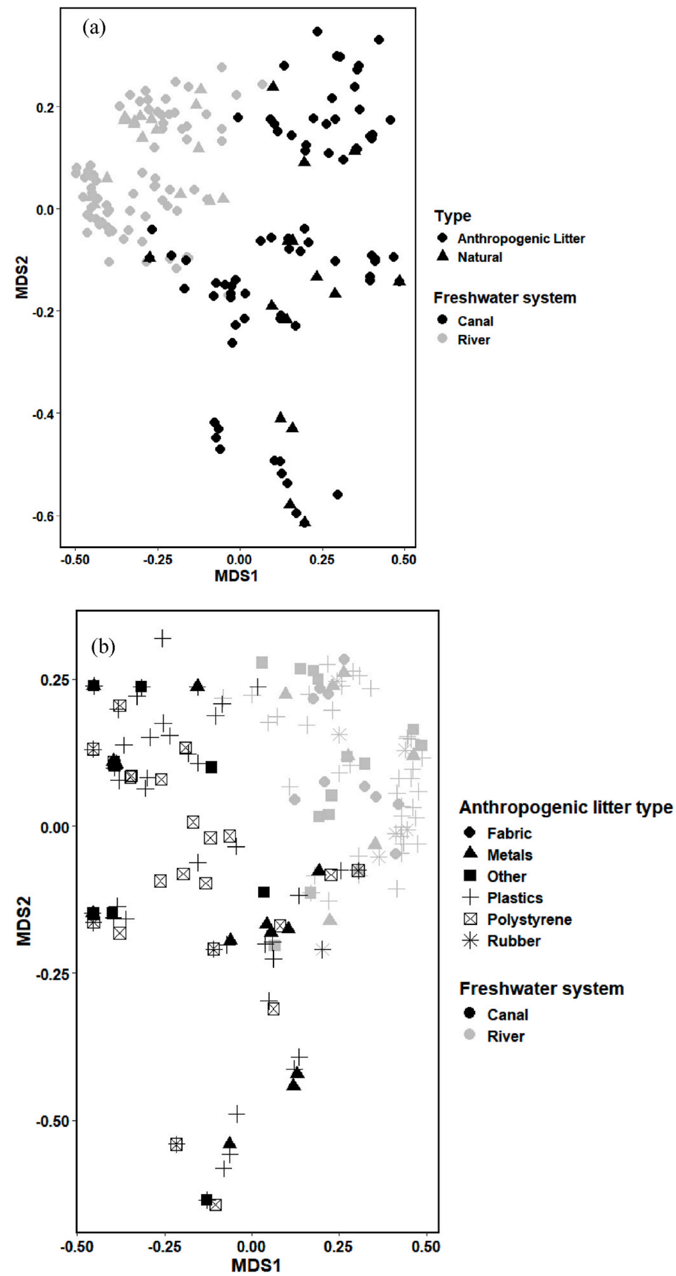

Fig. 5. PCoA plots of dissimilarity in macroinvertebrate communities (Bray Curtis dissimilarity) between (A) natural habitats and AL sites in the urban river and urban canal and (B) AL litter types within the urban river and canal.

A total of 32 macroinvertebrate taxa were unique to urban river reaches and 23 macroinvertebrate taxa were unique to the canal reaches (Table 3). Among AL types within the urban rivers, 7 taxa were unique to bed sediments (i.e., the natural habitats), 10 taxa were unique to plastics, 2 taxa were unique to metals, 3 taxa were unique to rubber and 1 taxa was unique to other AL items. However, no taxa were recorded only from fabrics (Table 3). Within the canal reaches, 13 taxa were only recorded from open water and emergent vegetation, 10 from plastics, 4 taxa from the polystyrenes, 4 from metals and 1 from other AL items (Table 3).

3.3. *Species environment relationships*

Redundancy analysis (RDA) demonstrated that macroinvertebrate communities from the urban river and canal reaches were separated on

Table 3
Macroinvertebrate taxa recorded only from a) river and canal sites (a), or from different AL types within b) the urban river or c) the canal sites.

a) Freshwater system		b) River					c) Canal				
River	Canal	Natural	Plastics	Metals	Rubber	Other	Natural	Plastics	Metals	Polystyrenes	Other
Ecdyonurus sp.	Glyphotaelius pellucidus	Rhithrogena sp.	Sphaeriidae	Lymnaea sp.	Seratella ignita	Rhyacophilidae	Glyphotaelius pellucidus	Oecetis sp.	Gyraulus albus	Eloeophila sp.	Physidae
Rhithrogena sp.	Corixinae	Sericostoma personatum	Eloeophila sp.	Caenis rivulorum	Antocha sp.		Corixinae	Simulium sp.	Velia saulii	Rhantus sp.	
Rhyacophila dorsalis	Physidae	Limnius volckmari	Leuctra sp.		Leptocerus sp.		Acroloxidae	Lymnaeidae	Dasyhelea	Tipula sp.	
Hydropsyche instabilis	Lymnaeidae	Limnophora sp.	Microvelia sp.				Sphaeriidae	Oulimnius sp.	Eubria palustris	Peltodytes sp.	
Dicranota	Hydroptila sp.	Odontocerum sp.	Copelatus sp.				Lymnaea sp.	Bithynia tentaculata			
Beris sp.	Limnephilus sp.	Hydraena sp.	Glossiphonia sp.				Limnephilus sp.	Lymnea stagnalis			
Lumbricidae	Collembola	Hydropsychidae	Physa fontinalis				Succinea putris	Dixa sp.			
Hydrobiidae	Hydrometra stagnorum		Agabus sp.				Sialis lutaria	Radix peregra			
Seratella ignita	Rhantus sp.		Psychodidae				Myxas sp.	Physa fontinalis			
Hemerodroma sp.	Succinea putris		Glossiphoniidae				Notonecta maculata				
Caenis rivulorum	Sialis lutaria						Agabus sp.				
Oreodytes septentrionalis	Bithynia tentaculata						Piscicola geometra				
Leuctra sp.	Lymnea stagnalis						Zonitidea				
Microvelia sp.	Velia saulii										
Copelatus sp.	Myxas sp.										
Limnius volckmari	Notonecta maculata										
Limnophora sp.	Dasyhelea										
Hydrophilidae	Sciomyzidae										
Hirudinea	Limnephilidae										
Elmis aenea	Peltodytes sp.										
Rhyacophilidae	Eubria palustris										
Sericostoma personatum	Piscicola geometra										
Glossiphonia sp.	Zonitidea										
Antocha sp.											
Mystacides longicornis											
Clinocerinae											
Odontocerum sp.											
Psychodidae											
Glossiphoniidae											
Dytiscus sp.											
Esolus sp.											
Hydraena sp.											

the first and second axes along gradients associated with pH, conductivity, velocity, temperature, surface area and dissolved oxygen (all $p < 0.01$; Fig. 6). AL type had no significant influence on the variation in macroinvertebrate community composition, and there was considerable overlap between AL types and the significant environmental variables. The RDA model was significant ($F = 3.77$, $p < 0.001$) and explained 15 % of the variation in macroinvertebrate community composition on all constrained axes, based on the adjusted R^2 value (adj. $R^2 = 0.15$). Canal reaches were associated with lower pH, conductivity, velocity, temperature, surface area and dissolved oxygen, while the urban river reaches were associated with faster velocities and shallower water depths (Fig. 6).

4. Discussion

4.1. Differences in taxonomic richness between natural habitats and AL items between and within urban rivers and canals

In our study, we examined the effects of AL on macroinvertebrate diversity within two urban freshwaters. Our first hypothesis that macroinvertebrate local taxonomic richness and gamma diversity would be higher in natural habitats (i.e., the bed sediments of riffles within urban rivers and open water/vegetation mesohabitats within canals) compared to AL items was partly supported. The bed sediments of riffles, open water and emergent vegetation possessed a higher local taxonomic richness than AL items, which supported the initial part of our first hypothesis. The higher local taxonomic richness in natural habitats may be due to the greater diversity and stability of bed sediments within our studied urban river reaches. A high proportion of AL items retrieved from the canal reaches were mobile, buoyant, and found floating on the water surface. Therefore, it is likely that emergent vegetation in the canal reaches and the bed sediments in the urban river reaches provided a more stable microhabitat for macroinvertebrates than many AL items. Stable items in watercourses typically support high abundances of macroinvertebrates (Death and Winterbourn, 1995; Entrekin et al., 2009), whilst unstable, mobile substrates generally support lower macroinvertebrate taxonomic richness (Barquín and Death, 2006), as movement minimises the opportunity for colonisation (Czarnecka et al., 2009). Macrophytes and bed sediments may also support greater macroinvertebrate diversity than AL items by accumulating food resources, such as detritus, seston and organic matter, and by helping macroinvertebrates avoid dislodgement (Jowett, 2003).

Our finding of bed sediments possessing a higher local taxonomic richness than AL items contrasts with previous research on AL in three urban gravel-bed rivers in Leicestershire and Nottinghamshire, UK (e.g., Wilson et al., 2021), a Polish reservoir (Czarnecka et al., 2009) and on beaches of the Baltic Sea (García-Vazquez et al., 2018). All three studies proposed that the higher macroinvertebrate diversity on AL items was due to a lack of habitat heterogeneity of bed sediments (of the urban rivers) or the instability of the natural substrate (i.e., sand) in the studied freshwater or marine systems. We sampled gravels, cobbles, and pebbles from riffles within our urban reaches, whereas Wilson et al. (2021) sampled rocks (lower substrate diversity), which may account for the differing findings. In this study, the urban river reaches possessed diverse bed sediments and did not lack hard surfaces for colonisation. Where waterbodies possess stable and diverse bed substrates, they provide suitable bed sediments and will likely support higher macroinvertebrate diversity than AL items.

In our study, gamma diversity was higher among AL items than natural substrates in the urban river and canal systems, and as a result,

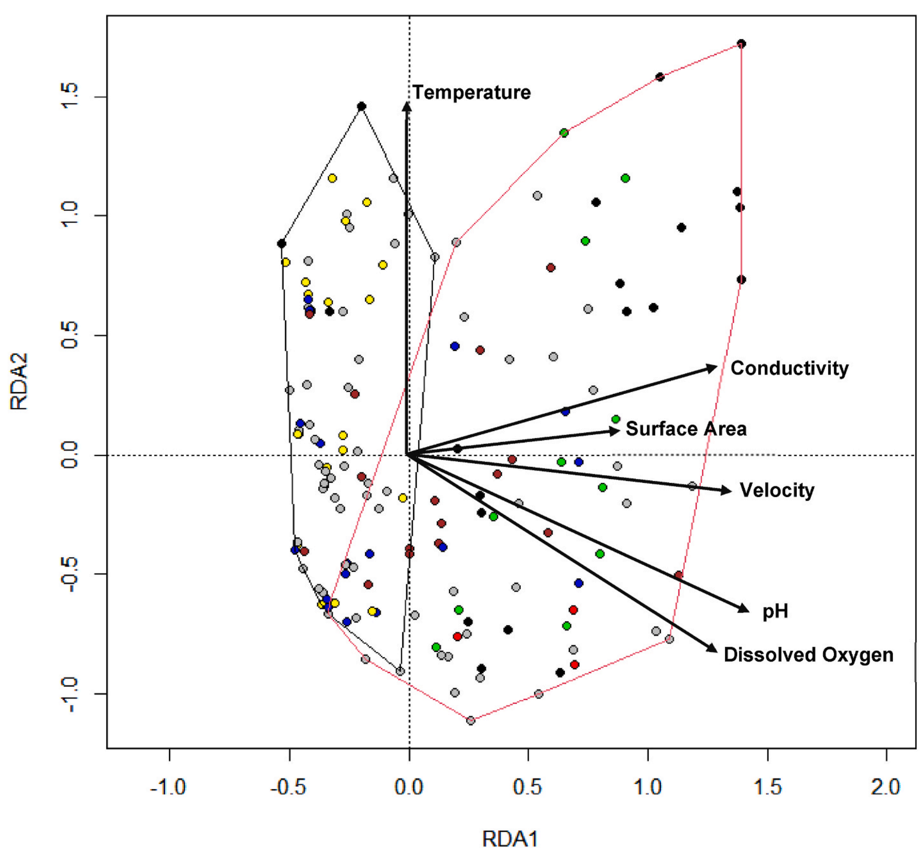

Fig. 6. RDA site plots for natural habitats and AL macroinvertebrate communities from the urban river and canal. Significant environmental parameters are presented. Red hull = river sites, grey hull = canal sites. natural habitat = black circles, plastic AL = grey circles, metal AL = blue circles, fabric AL = green circles, rubber AL = red circles, polystyrene AL = yellow circles, other AL = brown circles. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

we reject the second part of our first hypothesis. While AL items supported a lower local taxonomic richness than natural habitats; the higher gamma diversity most likely reflects the aggregated habitat complexity. A single piece of AL often has a consistent shape, size and texture, but when AL items are considered collectively, they have differing sizes, shapes, textures and structures. Cumulatively, AL items may have high habitat heterogeneity that increases the habitat available for colonisation by aquatic macroinvertebrates, and thus, increases overall richness at larger scales (Wilson et al., 2021).

4.2. Macroinvertebrate community composition differed between natural habitats and AL items

Our second hypothesis that macroinvertebrate community composition in natural habitats will be significantly different to the composition found on AL items was supported. Differences in macroinvertebrate community composition are likely due to the greater structural complexity and relative stability of bed sediments/emergent vegetation compared to AL items. Unique taxa in the bed sediments of riffles included *Rhithrogena* sp., *Limnius volckmari, Hydraena* sp. and Hydropsychidae. *Rhithrogena* sp. frequently inhabit well oxygenated, fast-flowing lotic environments with stable substrate (Vuataz et al., 2011), typical of riffles. The family Elmidae, including *L. volckmari* feed on fine detritus with associated microorganisms that are scraped from coarse bed sediment (a morphological characteristic of riffles; Giller and Malmqvist, 2003). Plastics contained unique leeches and Gastropoda, including Glossiphoniidae, *Physa fontinalis*, *Lymnea stagnalis* and *Radix peregra*. The suckers on leeches work most effectively on smooth surfaces (e.g., plastics) and Glossiphoniidae generally prefer slower flowing habitat to reduce accidental drift (typical of canals; Moss, 2010). *P. fontalis* is commonly found on vegetation, whilst *L. stagnalis* and *R. peregra* are generalists and do not have a strong habitat affinity (Giller and Malmqvist, 2003). Similarly, *Lymnaea* sp. was a unique species on metals. Freshwater snails adhere to surfaces through excreting a thin layer of mucus, which encourages attachment (Shirtcliffe et al., 2012). We noticed biofilms present on many metals and plastics, which serve as a primary food source for macroinvertebrates by providing algae, bacteria and organic matter (Guo et al., 2021). In our study, plastics and metals offered suitable smooth surfaces for attachment and feeding, which was favoured by leeches and snails.

Among vegetated mesohabitats in canals, *G. pulex* and *A. aquaticus* were dominant. *G. pulex* is generally found in faster flowing water but is often recorded in slower flowing and vegetated sites where food resources are abundant, and *A. aquaticus* is typically present in detritus rich or organically polluted water (Dobson et al., 2012), characteristic of the hydraulic and physical conditions in canal vegetated mesohabitats. Previous research has demonstrated a positive relationship between increased morphological complexity of macrophytes and macroinvertebrate diversity in different freshwater systems (e.g., Wolters et al., 2018; Law et al., 2019; do Nascimento Filho et al., 2021). More complex meso-habitats provide a greater availability of interstitial space and a greater diversity of ecological niches, which allows for the co-existence of different species with varying habitat requirements (do Nascimento Filho et al., 2021). Macrophytes also provide refuges for macroinvertebrates by providing a shelter from hydraulic stress (Buffagni et al., 2000) and predation (Diehl and Kornijów, 1998; Sagrario et al., 2009).

4.3. Variations in macroinvertebrate taxonomic richness and community composition on AL items

We found significantly greater local taxonomic richness on plastics and metals within the urban river than the canal reaches, which led us to accept the first part of our third hypothesis. This result was expected as canals generally support lower macroinvertebrate diversity compared to rivers due to reduced habitat heterogeneity, refuge availability, water quality, higher boat traffic disturbance, and decreased connectivity and drift dispersal pathways associated with the action of canal locks (Walker and Hassall, 2021). Thus, a more diverse community is often present in rivers than canal systems to colonise AL items. Our study also found distinct differences in macroinvertebrate community composition between AL types (when the urban river and canal reaches were grouped together). Communities on fabrics were clearly distinct from plastics, polystyrenes, metals and other AL items, and metals had dissimilar communities than rubber and polystyrenes (Table 3), leading us to accept the latter part of our third hypothesis. This finding implies these different AL categories function as distinct habitat types, which supports previous research into AL types possessing distinct macroinvertebrate communities and providing suitable habitat (e.g., Gething et al., 2020; Wilson et al., 2021). The flexibility and structure of AL types could also influence community composition. Fabrics have a flexible structure, possess different shapes, and are easily trapped between coarse clasts, which may increase stability and allow macroinvertebrate colonisation. The structure of fabrics may also promote the deposition of fine sediment and organic detritus, further encouraging macroinvertebrate colonisation, particularly by generalist taxa. No unique taxa were found on fabrics, but the community composition was dominated by *Cerratopogoninae bezzia* (26.1 %), *H. siltalai* (12.1 %), and *G. pulex* (8.3 %). The latter three taxa are characteristically abundant in low dissolved oxygen concentrations and organic enriched environments (Pennak, 1978; Armitage et al., 1995), which are typical of canals.

Several taxa were only recorded from plastics in our urban rivers, including Sphaeriidae, Glossiphoniidae and *P. fontinalis.* In canals, *Simulium* sp. and several snail species were only recorded from plastics. *P. fontinalis* are typically found on leaves or stems of macrophytes (Nature Spot, 2023) and *Simulium* sp. lay their eggs on rocks and submerged macrophytes (Freedman, 2006). In our study, we observed many plastic bottles were temporarily stored within emergent macrophyte patches within our canal reaches or trapped within coarser clasts of our urban river reaches. *P. fontinalis*, *Simulium* sp. and many other invertebrates could have actively or passively colonised plastic items from nearby macrophyte patches.

Differences in macroinvertebrate community composition between 1) natural habitats and AL items, and 2) AL items between urban river and canals were dominated by turnover rather than nestedness. This finding indicates AL is inhabited by different macroinvertebrates than natural habitats in our study reaches. Natural habitats, especially the bed sediments of riffles were dominated by Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa. Most taxa colonising AL were generally of lower conservation value that can tolerate poorer habitat quality (e.g., *G. pulex*, Oligochaeta and Tanypodinae), and thus, numerically dominate communities (Wilson et al., 2021). However, very low numbers of EPT (1–2 taxa) were present and unique on metals (*Caenis rivulorum*), rubber (*Seratella ignita*) and plastics (*Oecetis* sp.). Their occurrence may be due to accidental or deliberate colonisation from nearby natural habitats. The turnover of macroinvertebrate taxa between freshwater system may be due to differing hydraulic and sedimentary conditions. Rivers are dynamic systems, exhibiting variation in flow (Poff, 1997), channel geomorphic units (Wyrick and Pasternack, 2014; Burgazzi et al., 2021), depth, velocity (Lamouroux et al., 2004), substrate composition (Beisel et al., 2000; Dolédec et al., 2007) and in-channel features (e.g., boulders and large woody debris; Laasonen et al., 1998; Negishi and Richardson, 2003; Lepori et al., 2005; Yarnell et al., 2006), whilst canals are characterised by homogenous and slower flows (Bennett, 2009), uniform cross sections (Randima et al., 2017) and steep, reinforced banks (Chester and Robson, 2013). As habitat structure strongly influences invertebrate communities, these different environmental conditions may account for the turnover patterns between urban rivers and canals.

4.4. Environmental drivers of macroinvertebrate communities

Macroinvertebrate communities were largely separated by freshwater system with river sites characterised by faster velocities and a higher pH, conductivity, dissolved oxygen and larger AL items (Fig. 6). As expected, canals possessed higher temperatures, lower dissolved oxygen, conductivity and pH (Fig. 6). AL items were also generally smaller within the canal than river sites. Within each freshwater system, there was little association between macroinvertebrate communities found on different AL items (i.e., plastics, metals, polystyrenes etc) and the measured environmental conditions (Fig. 6). This finding may reflect 1) the high mobility of AL items across local environmental gradients (e.g., plastic bottles, cans and other AL items floating on the surface of the water), 2) consistent environmental conditions within each freshwater type, and 3) the influence of other unmeasured environmental factors on macroinvertebrate communities, such as water chemistry, which has been widely shown to be an important factor in shaping freshwater communities (Hu et al., 2022).

4.5. Conservation, management and policy implications

Our study has indicated that AL items can provide a habitat for macroinvertebrates in both urban rivers and canals, and that macroinvertebrate communities on AL items are distinct from macroinvertebrates inhabiting bed sediments in rivers and mesohabitats in canals. The removal of AL from freshwater systems is, and should remain, a key component of freshwater conservation and management actions, particularly as most marine litter originates from inland waterways (Rech et al., 2014), and the negative physical (e.g., entanglement, ingestion, abrasion) and chemical (e.g., plastic leachates) effects of AL on aquatic biota (Roman et al., 2022). However, the provision of physical habitat by AL for macroinvertebrates in urban freshwaters should be considered during clean-up strategies. Prior to removal, we recommend AL items are rinsed within the water column to dislodge macroinvertebrates and allow taxa to colonise new habitat. Following the removal of AL, the potential physical habitat loss should be offset by targeted restoration actions aimed at increasing physical habitat heterogeneity. Although this research has demonstrated that AL provides a novel habitat for macroinvertebrates, we have highlighted that bed sediments/mesohabitats provide a more favourable habitat by supporting higher local taxonomic richness and community heterogeneity. In urban rivers, habitat heterogeneity can be increased by diversifying particle size through gravel augmentation. Installing large wood or boulders within channels can provide alternative habitat and hydrological variability to help minimise any potential loss of habitat heterogeneity associated with AL removal. In canal reaches, greater habitat heterogeneity can be created by constructing shallow zones or backwater areas to improve hydrological variability and encourage macrophyte establishment.

Further research examining the ecological effects of AL on aquatic organisms is needed for mitigation interventions and to influence policy, pollution by-laws, levies, and initiatives on waste management by governments and non-governmental organisations (NGOs). Numerous international policies have been developed to protect freshwater and marine environments, such as the International Convention for the Prevention of Pollution from Ships (MARPPOL 73/78) of 1973, and the European Union Single-Use Plastics Directive of 2019. The latter legislation enforces a ban on single-use plastic items (e.g., straws), requires specific products to be labelled to inform consumers of appropriate disposal and an obligation for EU countries to reduce and monitor consumption (European Parliament and the Council of the European Union, 2019). NGOs and other organisations also play a key role through educating communities and citizens to act collectively to reduce AL (Kumar et al., 2021). For instance, the Canal and Rivers Trust encourages volunteer groups and individuals to remove litter from rivers and canals across the UK, and to reduce, reuse, and recycle plastics.

5. Conclusions

We acknowledge that our study was undertaken on two urban river and two canal reaches in one season. Future research should conduct surveys at a greater temporal scale (i.e., across multiple seasons) to assess whether macroinvertebrate colonisation of AL varies with seasonal variations in flow, water chemistry, light, and temperature (Šporka et al., 2006). In addition, similar studies should be carried out at larger spatial scales to assess whether the findings of our study are applicable to catchments with differing physical characteristics, and to examine whether AL colonisation by macroinvertebrates occurs in river and canal systems with varying anthropogenic modifications and different physical habitat conditions. Research assessing the ecological impacts of AL in freshwater systems is limited compared to marine systems (Hoellein et al., 2015; Rech et al., 2015; Roman et al., 2022), and studies investigating the potential use of AL as a novel habitat for macroinvertebrates in urban freshwater systems are also scarce (but see Czarnecka et al., 2009; Hessen, 2013; Wilson et al., 2021). We recommend that future studies are needed to improve understanding of the ecological impacts of AL in freshwater systems to help inform future conservation and management strategies.

CRediT authorship contribution statement

Victoria S. Milner: Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Software, Resources, Project administration, Methodology, Investigation, Funding acquisition, Data curation, Conceptualization. **Matthew J. Hill:** Writing – review & editing, Visualization, Validation, Supervision, Project administration, Methodology, Formal analysis, Data curation, Conceptualization. **Kieran J. Gething:** Writing – review & editing, Investigation. **Summer B. Cunningham:** Writing – review & editing, Visualization, Validation, Software, Resources, Methodology, Investigation, Formal analysis, Data curation, Conceptualization.

Declaration of competing interest

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

This study was partly funded by the University of Huddersfield Vice Chancellor's scholarship scheme through a tuition fee waiver for a MRes degree that was awarded to Summer Cunningham. Many thanks to Dr Mélanie Milin for providing technical support in macroinvertebrate identification. Thanks also to Samuel Townsend and Louise Britton for helping with fieldwork. Lastly, thank-you to two anonymous reviewers for comments that helped us improve the clarity of this paper.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envpol.2025.126097.

Data availability

Data will be made available on request.

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