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35

36 **Abstract**

37 Defining reference conditions is a crucial element in quantifying the extent of anthropogenic
38 modification and for identifying restoration targets in riverine ecosystems. Despite palaeoecological
39 approaches being widely applied in lake ecosystems to establish reference conditions, their use in lotic
40 ecosystems remains limited. In this study we examine contemporary, historical (1930 and 1972) and
41 palaeoecological macroinvertebrate biodiversity and biomonitoring scores in Eastburn Beck, a
42 headwater tributary of the River Hull (UK) to determine if palaeoecological approaches can be used to
43 characterise lotic system reference conditions. Palaeoecological samples comprised a greater gamma
44 diversity (18 taxa) than contemporary samples (8 taxa), and samples taken in 1972 (11 taxa) and 1930
45 (8 taxa). Palaeoecological samples supported taxonomically different Gastropoda, Trichoptera and
46 Coleoptera (GTC) communities compared to contemporary and historical samples (1930 and 1972).
47 Results from biomonitoring indices using the GTC community indicated that the palaeochannel had
48 (1) similar water quality, (2) less energetic flow regime and (3) increased fine sediment deposits
49 compared to the contemporary channel. The results clearly illustrate that palaeoecological data can
50 provide a suitable method to characterise reference conditions for lotic habitats. However, it is
51 important to recognise that faunal data from palaeochannel deposits provide a short-term ‘snapshot’ of
52 the conditions within the river immediately prior to its hydrological isolation. Riverine restoration
53 activities should therefore draw on multiple lines of evidence, including palaeoecological information

54 where possible, to characterise a range of ‘reference conditions’ to reflect the highly dynamic nature
55 of lotic ecosystems.

56 **Keywords:** biomonitoring, chalk stream, conservation, lotic ecosystems, palaeochannel, restoration

57 **1. Introduction**

58 Globally, riverine ecosystems have been subjected to a wide range of historical and contemporary
59 anthropogenic pressures. These factors include (but are not limited to); habitat modification,
60 degradation and fragmentation (e.g., resulting from channelization and construction of reservoirs /
61 dams), flow regime modifications (primarily in the form of abstraction and flow regulation),
62 pollution, over-exploitation of resources, the introduction of non-native taxa and climate change (Reid
63 et al., 2019; Strayer & Dudgeon, 2010). The intensity of anthropogenic stresses ranges from complete
64 ecosystem change in high intensity agricultural and urban landscapes (Foley et al., 2005) to low level
65 diffuse pollution in geographically remote landscapes (Lavers & Bond, 2017). It is estimated that as
66 little as 0.16% of the Earth's surface has been subjected to 'minimal anthropogenic disturbance' with
67 few rivers considered to be in pristine or natural condition (Reid et al., 2018; Vorosmarty et al., 2010).
68 Reductions in river ecosystem quality have led to changes in ecological community structure and
69 functioning associated primarily with large losses in freshwater biodiversity (Sanchez-Bayo &
70 Wyckhuys, 2019; Arthington, Dulvy, Gladstone, & Winfield, 2016; Steffen, Becker, Herr, &
71 Leuschner, 2013; Weijters, Janse, Alkemade, & Verhoeven, 2009).

72

73 Efforts to rehabilitate and restore riverine environments have increased significantly in recent decades
74 (Palmer, Zedler, & Falk, 2016; Kail, Brabec, Poppe, & Januschke, 2015). Such interventions have
75 predominantly focussed on management practices that promote the structural and functional diversity
76 of biotic communities/aquatic ecosystems. River restoration is defined here as 'assisting the
77 establishment of improved hydrologic, geomorphic, and ecological processes in a degraded watershed
78 system and replacing lost, damaged, or compromised elements of the natural system' (Wohl, 2005: 2).
79 To enable the extent of river degradation (long-term morphological, hydrological and ecological
80 change) to be quantified in order to determine restoration requirements, baseline or reference
81 conditions should be characterised where possible (Stoddard, Larsen, Hawkins, Johnson, & Norris,
82 2006; Palmer et al., 2005). In many regions, changes to channel morphology can be readily quantified
83 using historic maps (e.g., Grabowski & Gurnell, 2016; Sear & Arnell, 2006) and by reconstructing the

84 hydrological regime using long-term time series and modelling techniques (Galat & Lipkin, 2000).
85 However, the availability of ecological data over long time-periods (particularly prior to human
86 disturbance), represents a major limitation in characterising ecological baseline conditions in many
87 regions (Bennion, Battarbee, Sayer, Simpson, & Davidson, 2011).

88

89 Defining reference conditions is a crucial element in characterising the ecological status of European
90 rivers under the EU Water Framework Directive (WFD: EU 2003). Ecological quality is measured by
91 examining the deviation of contemporary conditions (chemical and biological) from reference
92 conditions (conditions expected prior to significant anthropogenic influence: Bennion et al., 2011). A
93 range of approaches have been proposed to establish reference conditions including spatial surveys
94 (space for time substitution), modelling and palaeoenvironmental approaches (EU 2003). Spatial
95 surveys are based on the principle that sampling of ‘pristine rivers’ provides a baseline (reference
96 point) for other locations within a given catchment. However, finding suitable pristine sampling sites
97 has proved difficult (Bennion et al., 2011a) given that the majority of European freshwater bodies
98 have been modified by anthropogenic activities for several centuries (Grizzetti et al., 2016). In
99 addition, some systems widely perceived to be semi-natural in the contemporary landscape, may not
100 accurately reflect pre-disturbance conditions (environmental and biological) of degraded rivers, as
101 land-use changes and drainage activities have occurred extensively over the last two centuries
102 (Seddon, Wood, Mainstone, Greenwood, & Howard, 2012).

103

104 For some lotic ecosystems, where appropriate minimally impacted sites do not exist, or that available
105 sites are unsuitable analogues, palaeoecological and palaeoenvironmental approaches may provide the
106 most suitable approach for establishing reference conditions (Seddon et al., 2012; Reid, Chilcott &
107 Thoms, 2018). Palaeoecological approaches have been widely applied within lake ecosystems to
108 characterise and quantify long term ecological change and establish reference conditions for several
109 decades (e.g., Finlayson, Clarke, Davidson, & Gell, 2016; Bennion et al., 2018; 2011; 2011a; Smol,
110 2008; Sayer & Roberts, 2001; Birks, 1996) as outlined in the EU WFD (EU, 2003). While

111 palaeohydrological and sedimentological research has been widely undertaken on rivers (Xue, Li,
112 Dang, Meyers, & Huang, 2017; Petts, Moller & Roux, 1989), palaeoecological research has
113 historically been limited, due to the relatively short temporal resolutions available, associated with
114 dynamic erosional and depositional processes within lotic ecosystems (Brown, 2002).

115

116 Palaeochannels are remnant channels that have been isolated as the river has migrated over its
117 floodplain and evolved, subsequently being infilled by younger sediment to the point that they may
118 become fully terrestrialised (Clarke, 2009); typically through the process of hydrosereal succession.
119 Palaeochannels represent key locations in the landscape to examine historic river channel and
120 community change, and can be frequently located using contemporary maps, field surveys and
121 increasingly via the use of topographic models based on aerial surveys using unmanned aerial
122 vehicles (UAVs) and LIDAR (Challis, Carey, Kincey & Howard, 2011). This information can be used
123 to inform the reconstruction of historic river courses and their environmental history (Howard et al.,
124 2008). The former channel deposits remain preserved under the land surface and contain the partially
125 fossilised remains of the floral and faunal communities which inhabited them (Greenwood, Wood, &
126 Monk, 2006). However, despite the relatively high abundance of sub-fossil invertebrate remains, such
127 as snails (Gastropods), caddisfly larvae (Trichoptera) and beetles (Coleoptera) within palaeochannel
128 sediments, their wider use has seldom been explored compared to lake ecosystems (but see Howard,
129 Wood, Greenwood, & Rendell, 2009; Greenwood et al., 2006).

130

131 A number of studies have employed palaeoecological data from riverine deposits to describe changes
132 in environmental conditions via differences in contemporary biomonitoring indices (e.g., The Lotic
133 Invertebrate Index for Flow Evaluation (LIFE) – Extence, Balbi, & Chadd, 1999; Proportion of
134 Sediment Sensitive Taxa: (PSI) - Extence et al., 2013) associated with faunal preferences for river
135 flow/hydraulic and sedimentological conditions (Seddon et al., 2012; Howard et al., 2009; Greenwood
136 et al., 2006). However, due to the episodic erosion of sediments and frequent channel migration,
137 palaeochannels typically represent relatively limited time-periods (~100 years or less). Despite these

138 limitations, examining the palaeoecological record of former river channels may provide important
139 information regarding historic instream conditions, thereby acting as a set of potential reference
140 conditions, as well as providing evidence regarding changes to the aquatic environment (habitat
141 characteristics) and the communities inhabiting them. In some instances, they may be able to form the
142 basis of river restoration targets, especially where the deposits can be directly dated radiometrically or
143 indirectly with reference to historic documents such as maps or written records (Brown, 2002; Brown,
144 Toms, Carey, Howard, & Challis, 2013).

145

146 Given the paucity of information regarding the potential application of palaeoecological datasets to
147 characterise reference conditions and hydroecological restoration targets in rivers, this study examines
148 the contemporary, historic and palaeoecological macroinvertebrate biodiversity and ecological health
149 of a headwater tributary of the River Hull, Eastburn Beck (Yorkshire, UK). Examination of a range of
150 macroinvertebrate biomonitoring indices was undertaken to determine if palaeoecological approaches
151 can establish baseline / reference conditions for lotic ecosystems.

152

153 **2. Study Area**

154 The River Hull is the most northern chalk (CaCO_3 – a fine powdered limestone) stream in the UK and
155 is designated a Site of Special Scientific Interest (SSSi: Natural England 2009). The river rises north
156 of Kingston-Upon-Hull close to the town of Driffield. The headwater comprises several small
157 tributaries including Eastburn Beck, which formed the primary study site for this research. The land
158 surrounding Eastburn Beck has been predominately mixed arable and pastoral agricultural land since
159 the 1850s. The headwaters are considered typical of chalk rivers in the UK and are characterised by
160 high nutrients and oxygen concentrations, and seasonally variable clear flowing water (high flows
161 during winter and base-flow occurring during the autumn: Table 1). Prior to 1850 the stream
162 displayed a sinuous planform which is clearly evident on the Ordnance survey map from 1849 (see
163 Fig. 1). The headwaters were heavily modified and managed with channelization, straightening and

164 deepening activities in the 1850's as part of land improvement measures (straightened channel shown
165 in Fig. 1) and subsequently, multiple weirs and a trout farm were constructed. Four sites on Eastburn
166 Beck were selected for detailed investigation. These encompassed the primary site where multiple
167 palaeochannels occurred on the adjacent floodplain (Fig. 1) and three additional sites. These sites also
168 allowed the contemporary data to be directly related to two historical studies of instream
169 macroinvertebrate communities undertaken on Eastburn Beck from 1972 (Pearson & Jones, 1984) and
170 1930 (Whitehead, 1935).

171

172 **3. Methods**

173 *3.1 Faunal community data collection and analysis*

174 Contemporary macroinvertebrates were sampled during spring, summer and autumn 2011 from the
175 four sites (total of 12 samples; a summary of corresponding environmental conditions is provided in
176 Table 1). In each season, a three-minute kick-sample was undertaken using a standard pond-net (1 mm
177 mesh), with sampling effort being divided equally amongst the instream habitats present, together
178 with an additional one-minute hand search of larger substrates and material (Murray-Bligh, 1999;
179 ISO, 2012). Samples were preserved in the field with 4% formaldehyde solution and subsequently
180 identified to species level where possible, with the exception of Chironomidae, Sphaeriidae and
181 Planariidae, which were resolved to family level and Oligochaeta, Tricladida (non-Planariidae) and
182 Collembola which were recorded as such.

183

184 Historic published data, in the form of macroinvertebrate faunal species lists, were available from two
185 published sources (Whitehead, 1935; Pearson & Jones, 1984) for two sites in the vicinity of Eastburn
186 Beck; resulting in a total of 21 macroinvertebrate community samples; 12 contemporary samples ,
187 seven samples from Whitehead (1935) and two samples from Pearson and Jones (1984). Historical
188 macroinvertebrates were collected using aerial quadrat-based samplers.

189

190 Palaeoecological samples of subfossil invertebrate remains were obtained via the collection of four
191 sediment cores from palaeochannels identified on Ordnance survey maps from 1849 still present on
192 the adjacent floodplain of the stream. Cores were retrieved using a petrol driven Cobra Corer in 1 m
193 long sections (50 mm internal diameter) and returned to the laboratory in sealed plastic containers to
194 maintain ‘field-damp’ conditions prior to processing and extraction of sub-fossil remains. Cores were
195 examined in 10 cm sections and only the fluvial sediments from the base of the palaeochannels were
196 processed for the analyses presented herein (11 samples in total). Specimens were extracted using the
197 paraffin flotation method developed by Coope (1986) with the addition of 125 µm and 90 µm sieves
198 to retain smaller Trichoptera fragments (see Greenwood, Agnew, & Wood, 2003 for further details).
199 Preserved Trichoptera and Coleoptera fragments were mounted on slides in water-soluble Hoyer’s
200 medium (to allow recovery of specimens if required). Identification of sub-fossil Trichoptera
201 fragments and larvae were made using the keys prepared by Wallace, Wallace, and Philipson (2003),
202 and Edington and Hildrew (1985) and photographic reference collections compiled by the authors.
203 Identification of Coleoptera (fragments and complete specimens) was completed using Friday (1988)
204 and Duff (2008) and through comparison with reference collections at the Leicestershire Museum at
205 Barrow upon Soar. Gastropoda (sub-fossil and contemporary) were identified using Macan (1977).

206

207 To allow direct comparison between the contemporary, historical and palaeoecological data, only
208 Gastropoda, Trichoptera and Coleoptera (GTC) were examined, as these were the primary sub-fossil
209 remains preserved within palaeochannel deposits (Greenwood et al., 2003). However, to ensure trends
210 observed in the GTC communities were representative of those displayed in the wider community,
211 both the entire macroinvertebrate community and GTC representatives were compared when
212 examining contemporary biomonitoring indices (see below). Macroinvertebrate and GTC datasets
213 were converted to presence-absence matrixes prior to statistical analysis to ensure contemporary,
214 historical and palaeoecological data were comparable and that any sampling bias was removed.

215

216 *3.2 Statistical analyses*

217 3.2.1 *Gamma and alpha macroinvertebrate diversity*

218 All statistical analyses were undertaken in the R environment (R development Core Team, 2013). For
219 each of the 32 samples (12 contemporary samples recorded across three seasons, 9 historical samples,
220 and 11 palaeoecological), gamma and alpha diversity of the macroinvertebrate community (GTC)
221 were determined. Gamma diversity is defined here as the total diversity within a predefined region
222 based on available samples, and alpha diversity as that within an individual sample (Hill, Chadd,
223 Morris, Swaine, & Wood, 2016). Statistical differences in GTC alpha diversity among contemporary,
224 historical (1972 – Pearson & Jones, 1984 and 1930 – Whitehead, 1935) and palaeoecological samples
225 were examined via a Kruskal-Wallis test using the function *Kruskal.test*. Pairwise comparisons were
226 undertaken using Nemenyi post hoc tests in the PMCMR package (Pohlert, 2016) to identify where
227 statistical differences occurred.

228

229 3.3.2 *Community heterogeneity - beta diversity*

230 For each of the 32 GTC community samples, beta-diversity, defined here as the spatial variation of
231 ecological communities among sample sites within a selected area (Socolar, Gilroy, Kunin, &
232 Edwards, 2016) was examined. This was achieved visually via Non- Metric Multidimensional Scaling
233 (NMDS; based on the Sorensen Dissimilarity Index) ordination plots using the *metaMDS* function and
234 tested statistically via permutational multivariate analysis of variance (PERMANOVA). Pairwise
235 comparisons were undertaken to identify where statistical differences occurred among contemporary,
236 historical and palaeoecological samples (Pedro Martinez Arbizu pers comm).

237

238 To examine the processes driving community differences between contemporary and palaeoecological
239 samples, beta-diversity was partitioned into nestedness and species turnover components using the
240 function *beta.temp* from the package betapart (Baselga, Orme, Villeger, De Bortoli, & Leprieur,
241 2017). *Beta.temp* computes pairwise dissimilarity values between two sampling periods.

242 Contemporary (seasons and sites) and palaeoecological samples (sites) were pooled separately to

243 provide contemporary and palaeoecological (gamma) diversity values, prior to beta-diversity
244 partitioning.

245

246 *3.3.3 Biomonitoring indices*

247 Four biomonitoring indices were derived for the entire species list (for contemporary samples only)
248 and for GTC species (across all sampling periods) as a measure of environmental quality and habitat
249 characteristics for contemporary, historical and palaeoecological conditions. The Biological
250 Monitoring Working Party (BMWP) score (Furse, Wright, Armitage, & Moss, 1981) and Average
251 Score Per-Taxon (ASPT – Armitage, Moss, Wright, & Furse, 1983) describe water quality; the LIFE
252 score (Extence et al., 1999) characterises faunal preferences to flow characteristics; and the PSI score
253 (Extence et al., 2013) examines instream fine sediment pressures. The water quality indices (BMWP
254 and ASPT) have been historically used to characterize the ecological status of rivers in the UK for EU
255 Water Framework reporting, whilst LIFE and PSI scores are routinely used by the Environment
256 Agency to characterise flow or fine sediment pressures (particularly if water quality is not a limiting
257 factor). Statistical differences in contemporary, historical (1972 and 1930) and palaeoecological
258 indices (based on the GTC taxa) were examined using the Kruskal–Wallis test with pairwise
259 comparisons to determine where significant differences occurred. The entire contemporary
260 macroinvertebrate community and GTC data was compared to examine how representative GTC
261 communities were of the wider macroinvertebrate community. Statistical differences between
262 seasonal contemporary samples were examined using a one-way ANOVA to determine where
263 significant differences occurred.

264

265 **4. Results**

266 *4.1 Gamma and alpha macroinvertebrate diversity*

267 Across all sampling periods (32 GTC samples) a total of 31 Gastropoda (5 taxa), Trichoptera (7 taxa)
268 and Coleoptera species (19 taxa) were recorded. The palaeoecological samples comprised greater

269 gamma diversity (18 GTC species) than contemporary samples (8 GTC species) and samples taken in
270 1972 (11 GTC species) and 1930 (8 species: see Table 2). At an alpha scale, significant differences
271 (Kruskal–Wallis test $df = 3$, $\chi^2 = 14.462$, $p < 0.005$) were recorded among the contemporary, 1972,
272 1930 and palaeoecological sampling periods (Fig. 2). Nemenyi post hoc tests indicated that: (1)
273 samples from 1930 (mean taxa: 4 median taxa: 4) had significantly lower alpha diversity ($p < 0.05$) than
274 the palaeoecological samples (mean: 8 median: 8), (2) invertebrate samples from 1972 (mean: 9
275 median: 9) had significantly ($p < 0.05$) greater alpha diversity than samples from 1930 and; (3) no
276 significant differences in alpha diversity were recorded between the contemporary samples (mean: 6
277 median: 6) and the historical (1972 or 1930 samples) or palaeoecological samples (Fig. 2).

278

279 *4.2 Community heterogeneity – beta diversity*

280 A clear distinction among GTC assemblages from contemporary, palaeoecological and historical
281 samples collected in 1972 and 1930 were observed in the NMDS biplot (Fig. 3). Contemporary and
282 historic samples cluster to the left, while palaeoecological samples cluster to the right. PERMANOVA
283 pairwise tests found GTC communities to be significantly ($p < 0.05$) different between: (1)
284 contemporary samples and all other sampling periods and; (2) palaeoecological samples and samples
285 taken in 1930 (Table 3). A total of 13 species were unique to the palaeoecological samples, 5 species
286 to 1972, 2 species to contemporary samples and 1 to 1930 (Table 2). Examination of the
287 contemporary and palaeoecological macroinvertebrate assemblages indicated high beta diversity
288 (heterogeneity - Sorensen dissimilarity: 0.77), with most of the variation in GTC composition
289 occurring between contemporary and palaeoecological samples which was explained by species
290 turnover (81.3%) rather than spatial patterns of nestedness (18.7%).

291

292 *4.3 Biomonitoring indices*

293 No significant differences in BMWP scores among the contemporary, historical (1930 and 1972) and
294 palaeoecological samples (GTC data) were recorded (Figure 4a). In contrast, the ASPT scores were

295 significantly ($p < 0.05$) higher during the contemporary sampling period than for the historical or
296 palaeoecological samples (Figure 4b: Table 4). Significant ($p < 0.05$) differences were recorded in
297 LIFE scores among contemporary, historical and palaeoecological samples in all but one instance
298 (1930 and 1972 comparison; Table 4). Contemporary samples supported assemblages associated with
299 faster flow velocities (taxa from LIFE flow groups 1 or 2: Extence et al., 1999) compared to historical
300 and palaeoecological samples (greater number of taxa from LIFE flow groups 3-6; Figure 4c).
301 Significant differences ($p < 0.01$) were observed in PSI scores between the contemporary, 1972
302 samples, and palaeoecological samples (Figure 4d) but samples from 1930 did not differ from
303 contemporary samples. PSI scores were highest in the contemporary and 1930 sampling periods (81-
304 100 PSI scores) suggesting the river contained little deposited fine sediment (Extence et al., 2013). In
305 marked contrast, the palaeoecological and 1972 PSI sample scores ranged between 20-40 and 41- 60
306 respectively, indicating a greater proportion of deposited fine sediment.

307

308 Examination of biotic indices derived for the entire contemporary community and those from only the
309 contemporary GTC communities, indicated that BMWP scores were typically higher when the entire
310 community was considered, while ASPT scores were lower (Figure 4a and Figure 4b). LIFE and PSI
311 scores were broadly comparable (Figure 4c and Figure 4d). There was a significant increase in ASPT
312 scores, with spring having the lowest score and autumn the highest ($p < 0.05$: Figure 4). There were no
313 significant differences between contemporary seasonal samples for the BMWP, LIFE and PSI scores
314 ($p > 0.05$).

315

316 **5. Discussion**

317 *5.1 Contemporary and palaeoecological macroinvertebrate diversity and community composition*

318 The results of this study highlight clear differences in macroinvertebrate community diversity and
319 composition between contemporary, historical (1930 and 1972) and palaeoecological samples. At the
320 gamma (landscape) scale, macroinvertebrate diversity was significantly higher within

321 palaeoecological samples compared to contemporary samples. Difference in gamma diversity were
322 potentially driven by significant habitat modifications to the River Hull headwaters during the 19th
323 Century that resulted from the straightening of the river channel and therefore resulted in the isolation
324 of the palaeochannel. The previously meandering headwaters were embanked and channelized during
325 the 1850s, creating a largely uniform, straight and over-deepened gravel bed channel (2m below
326 floodplain) for most of its course (Environment Agency, 2003). Habitat modification in the
327 headwaters of rivers across the UK has resulted in more homogeneous flow conditions (fast flow),
328 disconnection of the main channel from the floodplain, loss of most backwater habitats and a
329 reduction in habitat complexity (Collins et al., 2011).

330

331 As a result of habitat simplification, the contemporary macroinvertebrate community was almost
332 exclusively comprised of rheophilic taxa, adapted to high flow velocities, whilst the palaeoecological
333 community contained a mixture of low energy (limnophilic) and high flow velocity (rheophilic)
334 adapted taxa. Furthermore, examination from the 1849 topographical map indicated that riparian
335 wetlands which existed historically, have been lost as hydrological connectivity between river and
336 floodplain was reduced as a direct result of channel management and subsequent land drainage.
337 Significant habitat modifications, similar to that experienced on the headwater tributaries of the River
338 Hull, have occurred widely throughout northern Europe and have been shown to significantly reduce
339 alpha and gamma diversity (Käiro, Haldna, Timm, & Virro, 2017; Kennedy & Turner, 2011; Pliuraite
340 & Kesminas, 2011; Horsak, Bojkova, Zahradkova, Omesova, & Helesic, 2009).

341

342 Clear differences in community composition were recorded between palaeoecological and
343 contemporary samples in this study. A total of 15 GTC species were unique to the palaeoecological
344 samples, the majority of which were indicative of lentic to slow flowing conditions (except the net
345 spinning caddisfly larvae *Hydropsyche pellucidula*). For example, *Limnephilus marmoratus* and
346 *Anabolia nervosa*, the most abundant caddisflies recorded in the fossil record, are typically recorded
347 in still and slow flowing waters, often among vegetation (Wallace et al., 2003). Other common

348 caddisfly larvae recorded in the palaeoecological samples included *Athripsodes aterrimus*, which are
349 typically recorded within largely stagnant silty habitats with macrophytes (Wallace et al., 2003), as
350 well as *Potamophylax cingulatus* and *Halesus radiatus* (both Limnephilidae), common in headwater
351 streams with gravel substrates (Wallace et al., 2003). Only three species were common to all sampling
352 periods: *Elmis aenea* (Coleoptera), *Rhyacophila dorsalis* (Trichoptera) and *Sericostoma personatum*
353 (Trichoptera). *Elmis aenea* is a riffle beetle typically associated with unpolluted, moderately fast
354 flowing water over fine gravel and sandy riverbeds (Nilsson & Holmen, 1995). The caseless caddisfly
355 larvae, *Rhyacophila dorsalis* displays a preference towards medium to fast flowing rivers with gravel
356 substrates (Edington & Hildrew, 1995) and *Sericostoma personatum* a burrowing caddisfly, is often
357 associated with sandy deposits within gravel riverbeds (Wallace et al., 2003).

358

359 Although the physico-chemical conditions associated with the palaeochannels are unknown, the
360 biomonitoring indices derived from all samples provides a relatively accurate characterisation of
361 instream conditions (Howard et al., 2009; Monk et al., 2006). The BMWP and ASPT scores suggest
362 that water quality was broadly comparable over the study periods, with the exception of 1972. BMWP
363 scores for the 1972 study period were slightly higher and ASPT scores were lower, due to the
364 presence of a number of low scoring organic pollution tolerant taxa. This suggests that the river was
365 subject to mild organic pollution, probably associated with intensive agricultural activity within the
366 catchment at that time (Pearson & Jones, 1984).

367

368 Examination of LIFE scores provides further evidence that the contemporary community is dominated
369 by a relatively small number of relatively high flow velocity adapted rheophilic taxa (Extence et al.,
370 1999) compared to historical samples (palaeoecological, 1972 and 1930). This suggests that the
371 palaeochannel community was associated with a less energetic flow regime relative to the
372 contemporary system; although an average LIFE score of 7.1 indicates relatively fast flow velocities
373 must have occurred (Extence et al., 1999), and that there was significantly greater flow heterogeneity
374 in the palaeochannel compared to the contemporary channel. However, the LIFE score methodology

375 was not designed to be a quantitative measure of flow velocity since the individual groups encompass
376 relatively wide flow velocity classes. Therefore, it should only be used as a tool to characterise the
377 nature of the flow regime and the range of flow velocities present at the time of sample collection
378 (Howard et al., 2009). PSI scores suggest the presence of a larger proportion of fine sediment deposits
379 within the channel during the palaeoecological and 1972 sample periods. While individual
380 biomonitoring indices can provide specific information relating to an environmental parameter,
381 employing a multi-index approach to examine lotic ecosystems (as undertaken in this study), may
382 have greater bio-diagnostic capability and provide a more complete identification of temporal changes
383 in environmental and ecological conditions or specific management requirements (Clews & Ormrod,
384 2009).

385

386 When evidence from all sources is combined (maps and sub-fossil macroinvertebrates), the results
387 suggest that prior to 1850 when channel straightening and land-drainage occurred, the river was
388 located in a landscape with a greater number of riparian wetlands / marshlands habitats. The channel
389 and surrounding landscape of Eastburn Beck was more heterogeneous and instream conditions were
390 characterised by the presence of finer sediment deposits and more variable patterns of flow velocity
391 (comprising areas of slow and fast flow velocities). NMDS ordinations indicate the palaeochannel
392 community was markedly different and more biodiverse than the community recorded during all other
393 time-periods. However, despite the palaeochannel community being comprised of different species,
394 the biomonitoring scores derived for the BMWP, ASPT and PSI indices were comparable to those
395 recorded historically and those within the contemporary sampling period; with the exception of the
396 LIFE score.

397

398 *5.2 Implications for river management*

399 The results of this research suggest that communities recorded from the sediment cores (the
400 palaeocommunity) provide representative and realistic reference condition(s). Historically, the

401 characterisation and defining of reference conditions for lotic ecosystems has proved to be
402 problematic when compared to lentic ecosystems, particularly where little or no historical data is
403 available. However, the findings of this research indicate that palaeoecological approaches can
404 provide the baseline conditions for comparison with contemporary communities. In addition,
405 comparison of contemporary and historical/palaeoecological biomonitoring scores (based on the GTC
406 taxa) may also provide opportunities to examine how changes in management practices have shaped
407 river habitats and biotic communities over time. It is important to recognise that samples from
408 palaeochannels provide a short-term ‘snapshot’ (typically < 50 years) of the conditions within the
409 river / palaeochannel immediately prior to its hydrological isolation and subsequent infilling stage as
410 riverine sedimentary sequences are typically punctuated by episodic erosion. These coarse-grained
411 fluvial sediments at the base of palaeochannel deposits contain the sub-fossilised invertebrate remains
412 that can be used to characterise reference conditions for both biomonitoring purposes and/or
413 restoration targets (Seddon et al., 2012). Where possible, multiple records (cores from palaeochannels
414 and historic data) should be utilised in recognition that a range of reference conditions may exist,
415 rather than a single static target. This approach is now widely accepted in characterising reference
416 conditions and determining restoration targets for lake ecosystems (Bennion et al., 2011) but has not
417 been widely applied to lotic ecosystems.

418

419 The orders of Gastropoda, Trichoptera and Coleoptera identified in the palaeoecological record have a
420 number of advantages within riverine ecosystems: (1) all three groups are species rich compared with
421 other aquatic groups; (2) subfossil remains are abundant, durable and can be identified relatively
422 easily to a high taxonomic resolution; (3) ecological preferences of Gastropoda, Trichoptera and
423 Coleoptera are well known and; (4) all three groups comprise fauna that colonise a wide range of
424 ecological niches (Howard et al., 2009). These features provide greater confidence in characterising
425 reference conditions for future river restoration activities (Howard et al., 2009).

426

427 It not possible or realistic to turn back the clock and reinstate riparian marshland and conditions
428 present in Eastburn Beck from the mid-1800s. Currently the ‘Kirkburn, Eastburn Beck and Driffield
429 Trout Stream’ Site of Special Scientific Interest (SSSI) is in an unfavourable condition based on the
430 most recent assessment undertaken by Natural England (the government advising body on
431 environmental issues in England Natural England, 2019). This condition assessment recognised the
432 need for more sympathetic riparian land management. If this management strategy were to include the
433 provision of space to facilitate geomorphological heterogeneity by allowing the channel to migrate, it
434 may help reinstate and restore lost instream habitat heterogeneity and structural complexity. It is
435 widely acknowledged that high habitat heterogeneity is associated with diverse biological
436 assemblages (Franssen, Gilbert, & Propst, 2015; Loke, Ladle, Bouma, & Todd, 2015; Muotka &
437 Syrjanen, 2007), although this is not always the case (Nilsson et al., 2015; Palmer, Menninger, &
438 Bernhardt, 2010). The contemporary macroinvertebrate community composition and biomonitoring
439 indices are characterised by reduced biodiversity and similar water quality indices (BMWP and
440 APST), but LIFE and PSI scores suggest the flow and fine sediment regime (habitat heterogeneity)
441 have been homogenised. Consequently, there may be opportunities to restore abiotic heterogeneity in
442 this stream in order to enhance the diversity of invertebrate communities. Ensuring that the river is
443 comprised of a diversity of biotopes (e.g., variable hydrological conditions and depths, different
444 substrate types, exposed tree root systems, marginal and submerged vegetation, shaded and un-shaded
445 areas) may help mitigate some of the most extreme anthropogenic effects and ultimately support
446 greater biodiversity (White, Hill, Bickerton, & Wood, 2017; Miller, Budy, & Schmidt, 2010;
447 Ormerod, 2004). These conservation management goals directly reflect the current policy aspirations
448 outlined for freshwater habitats in England (Mainstone, Hall, & Diack, 2016). However, there may be
449 a number of ecological barriers to restoration including; dispersal limitation, abiotic resistance,
450 resource limitation and biotic resistance that may reduce the success and effectiveness of increasing
451 habitat complexity (Frame, Jones, Ormerod, Sadler, & Ledger, 2016; Palmer et al., 2010).

452

453 This research has demonstrated that palaeoecological data (used in combination with other evidence:
454 maps and historic data) may provide a robust method to characterise ‘reference conditions’ for lotic
455 systems. For many lowland rivers, with a long history of management and anthropogenic channel
456 modifications, palaeoecological approaches may be the only viable method available for the
457 identification of past in-stream biological and environmental conditions. However, given the highly
458 dynamic nature of lotic systems, any proposed restoration activities should be underpinned by
459 multiple lines of evidence and where possible a range of acceptable ‘reference conditions’ should be
460 utilised (Stoddard et al., 2006) rather than being confined to a single static target which may
461 ultimately be unachievable.

462

463 **Data Availability Statement** – the data underpinning this research are available from the authors
464 upon request.

465

466 **Conflict of Interest Statement** – The authors declare no conflicts of interest.

467

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669 **Tables**

670 Table 1 – Environmental conditions recorded across the three contemporary sampling seasons in the

671 River Hull headwaters (UK) .

		Season		
		Spring 2011	Summer 2011	Autumn 2011
Depth (cm)	Mean	36.45	33.4	21.25
	Min	29	26	6
	Max	49	40	81
Flow Velocity (m/s)	Mean	0.63	0.58	0.21
	Min	0.51	0.32	0.09
	Max	0.79	0.79	0.37
pH	Mean	8.3	7.6	8.4
	Min	8	7.4	8.3
	Max	8.5	7.7	8.5
Conductivity (µm)	Mean	562	421	557
	Min	560	420	552
	Max	564	422	564
Temperature (°C)	Mean	11.45	10.68	14.43
	Min	11.2	10.2	14.3
	Max	11.6	11.6	14.6

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682 Table 2 - Summary table of macroinvertebrate diversity (Gastropoda, Trichoptera and Coleoptera)
 683 recorded from contemporary, historical (1930 and 1972) and palaeoecological communities from the
 684 River Hull (UK)

	Sample period			
	Contemporary	1972	1930	Palaeoecological
Total richness	8	11	8	18
Mean	6	9	4	8
SE	0.25	1	0.59	1
Median	6	9	4	8
Min	4	8	2	0
Max	7	10	7	12
Unique species	Elmis aenea larvae <i>Drusus annulatus</i>	<i>Ancylus fluviatilis</i> Pisidium spp. Haliphus spp. Hydroporus spp. Sericostomatidae	Limnephilidae	<i>Nebrioporus elegans</i> Helophorus Cercyon <i>Hydropsyche pellucidula</i> <i>Hydropsyche angustipennis</i> <i>Limnephilus marmoratus</i> Limneph.3..in.det. Limneph.4..in.det. <i>Anabolia nervosa</i> <i>Potamophylax cingulatus</i> <i>Halesus radiatus</i> <i>Athripsodes aterrimus</i> Athripsodes spp.

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692 Table 3 - PERMANOVA pairwise tests of Gastropoda, Trichoptera and Coleoptera assemblages
 693 (based on the Sorenson dissimilarity) from contemporary, historical (1972 and 1930) and
 694 palaeoecological samples from the River Hull (UK). Significant values ($p < 0.05$) are presented in bold.

	F. value	R ²	adjusted p. value
Contemporary v 1972	38.61	0.76	0.042
Contemporary v 1930	41.81	0.71	0.006
Contemporary v Palaeoecological	71.44	0.78	0.006
1972 v 1930	9.63	0.58	0.114
1972 v Palaeoecological	12.36	0.55	0.108
1930 v Palaeoecological	30.81	0.67	0.006

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711 Table 4 - Kruskal-Wallis pairwise comparison of ecological indices between contemporary (spring,
 712 summer and autumn), historical (1972 and 1930) and palaeoecological Gastropoda, Trichoptera and
 713 Coleoptera assemblages in the River Hull (UK). *P < 0.05, ** P < 0.01, *** P < 0.001.

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		Spring	Summer	Autumn	Palaeo	1972
BMWP	Summer	NS	-	-	-	-
	Autumn	NS	NS	-	-	-
	Palaeo	NS	NS	NS	-	-
	1972	NS	NS	NS	NS	-
	1930	NS	NS	NS	NS	NS
ASPT	Summer	NS	-	-	-	-
	Autumn	NS	*	-	-	-
	Palaeo	*	**	**	-	-
	1972	**	*	**	NS	-
	1930	*	**	*	NS	NS
LIFE	Summer	NS	-	-	-	-
	Autumn	NS	*	-	-	-
	Palaeo	***	**	**	-	-
	1972	***	**	**	*	-
	1930	***	***	**	*	NS
PSI	Summer	NS	-	-	-	-
	Autumn	NS	NS	-	-	-
	Palaeo	**	**	**	-	-
	1972	**	**	*	NS	-
	1930	NS	***	***	**	*

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725 **Figure captions**

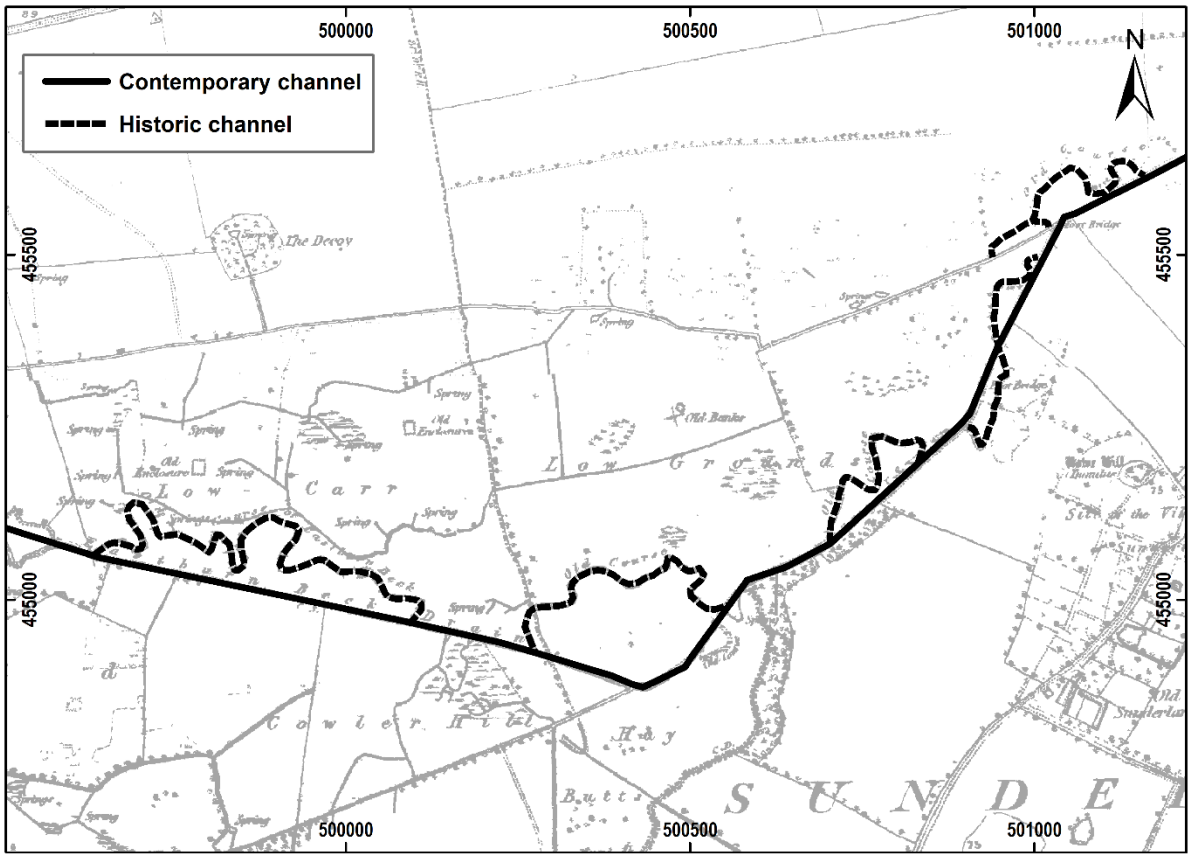
726 Figure 1 - Location of study reach and sampling sites of the River Hull headwaters from the
727 contemporary and palaeochannel based on Ordnance Survey maps prepared in 1849 and 1899,
728 and its location in relation to the UK.

729 Figure 2 - Median macroinvertebrate richness (Gastropoda, Trichoptera and Coleoptera) recorded
730 from contemporary, historical (1972 and 1930) and palaeoecological samples from the River
731 Hull. Boxes show 25th, 50th, and 75th percentiles and whiskers show 5th and 95th percentiles.

732 Figure 3 - Non-metric multidimensional scaling plots of dissimilarity (Sørensen dissimilarity) in
733 aquatic macroinvertebrate communities (Gastropoda, Trichoptera and Coleoptera) from
734 contemporary, historical (1972 and 1930) and palaeoecological samples from the River Hull
735 (UK).

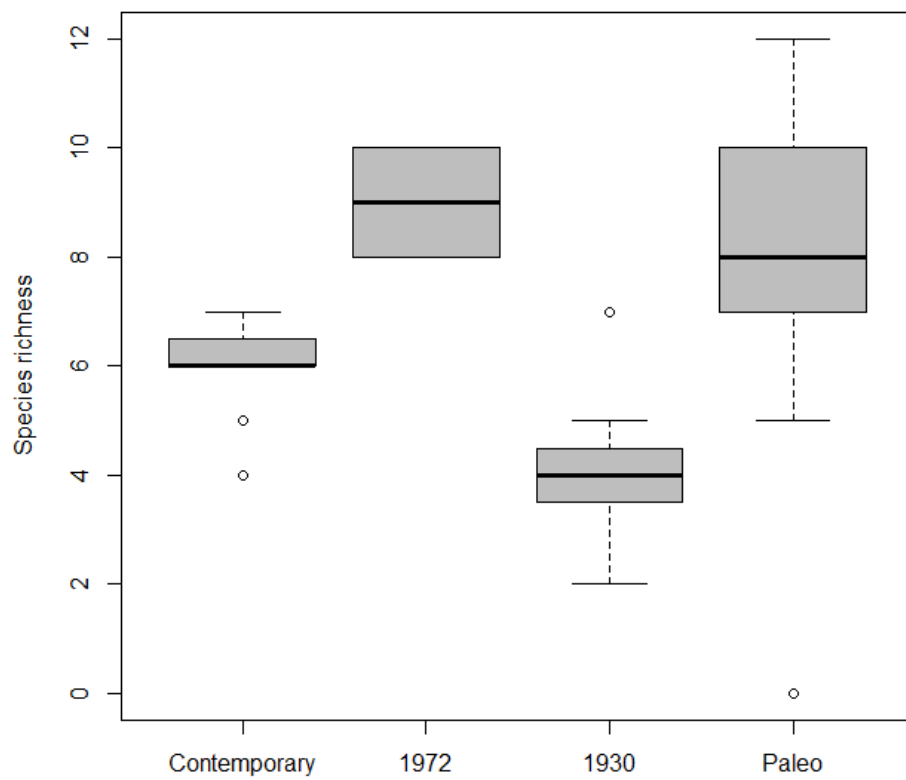
736 Figure 4 - Mean (a) BMWP, (b) ASPT, (c) LIFE and, (d) PSI scores (+/- one standard error) for the
737 contemporary (spring, summer and autumn samples), historical (1972 and 1930) and
738 palaeoecological samples derived from Gastropoda, Trichoptera and Coleoptera communities
739 (black) and from the full contemporary species list (grey). Note only contemporary samples are
740 presented as seasons (spring, summer and autumn).

741 Fig. 1



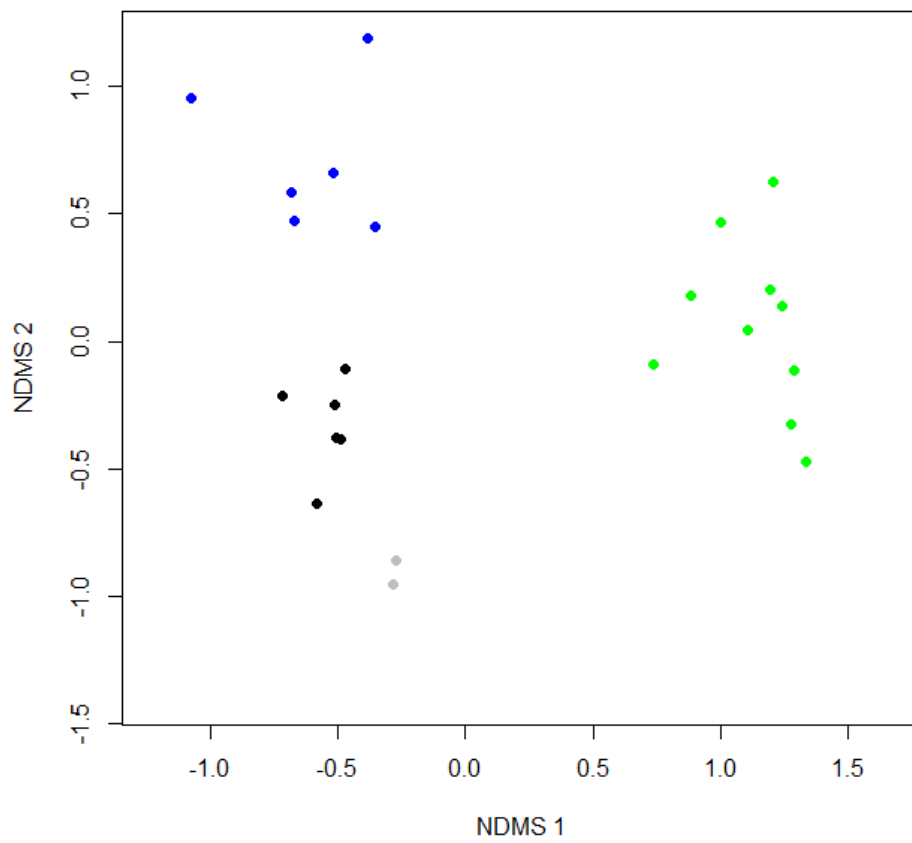
742 Created using ESRI ArcMap 10.6. Projected in British National Grid.
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743 Fig. 2



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745 Fig. 3



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