

1 **Aquatic macroinvertebrate biodiversity associated with artificial agricultural drainage ditches**

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30 **Abstract**

31 Agricultural drainage channels and ditches are ubiquitous features in the lowland agricultural
32 landscapes, built primarily to facilitate land drainage, irrigate agricultural crops and alleviate flood
33 risk. Most drainage ditches are considered artificial waterbodies and are not typically included in
34 routine monitoring programmes, and as a result the faunal and floral communities they support are
35 poorly quantified. This paper characterizes the aquatic macroinvertebrate diversity (alpha, beta and
36 gamma) of agricultural drainage ditches managed by an internal drainage board in Lincolnshire, UK.
37 The drainage ditches support very diverse macroinvertebrate communities at both the site (alpha
38 diversity) and landscape scale (gamma diversity) with the main arterial drainage ditches supporting
39 greater numbers of taxa when compared to smaller ditches. Examination of the between site
40 community heterogeneity (beta diversity) indicated that differences among ditches were high spatially
41 and temporally. The results illustrate that both main arterial and side ditches make a unique
42 contribution to aquatic biodiversity of the agricultural landscape. Given the need to maintain drainage
43 ditches to support agriculture and flood defence measures, we advocate the application of principles
44 from 'reconciliation ecology' to inform the future management and conservation of drainage ditches.

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46 **Key words:** drainage channel; invertebrates; wetland habitat; reconciliation ecology; conservation;
47 species richness.

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50 **Introduction**

51 Land drainage improvements across Europe have historically been followed by the large-scale
52 conversion of lowland wetlands to intensive arable production. This has resulted in a wide range of
53 documented changes and adverse effects upon biological communities across terrestrial, riparian and
54 aquatic landscapes (Buisson et al., 2008; Holden et al., 2004; van Eerden et al., 2010; Watson &
55 Ormerod, 2004). Contemporary European wetlands exist as isolated fragments of their former extent,
56 with those that remain largely surrounded by agricultural land (Verdonschot et al., 2011). Wetland
57 habitat loss across Europe is most likely to continue as agricultural intensification, land conversion
58 and water abstraction continue to exert pressure (Maltby & Acreman, 2011). Frequently, the only
59 remaining aquatic habitat/refuges that exist in agricultural landscapes are ponds (e.g., Sayer et al.,
60 2012) and drainage ditch networks. However, the potential importance of drainage ditch habitats in
61 supporting aquatic biodiversity, the persistence of wetland floral or faunal communities, or species of
62 conservation interest, has been poorly quantified to date, internationally (Katano et al., 2003; Leslie et
63 al., 2012; Maltchik et al., 2011; Vaikre et al., 2015).

64 Ditches are defined as man-made channels created principally for agricultural purposes, which often
65 follow linear field boundaries, turning at right angles and frequently display little relationship with
66 natural landscape topography (Davies et al., 2008a). Drainage ditches created in lowland agricultural
67 regions often occur in dense networks, characterised by larger main ditches (arterial drainage channels
68 – where flow is preferentially conveyed by gravity or by pumping) and smaller side ditches (smaller
69 channels within which water levels can be controlled by the use of weirs and can be isolated from the
70 main arterial channel; Clarke, 2015). Extensive linear networks of drainage ditches extend over an
71 estimated 128,000 km in the UK (Clare & Edwards, 1983). The primary anthropogenic function of
72 drainage ditches is to convey water to agricultural land, to support crop irrigation during the growing
73 season/dry periods and to divert water away from agriculture and urban infrastructure within towns
74 and villages (flood alleviation) during wetter periods. Agricultural drainage ditches are frequently
75 subject to a range of routine management activities including dredging/ in-channel vegetation

76 management and bank vegetation cutting to maintain efficient conveyance of water and reduce flood
77 risks (Clarke, 2015).

78 For EU Water Framework Directive (WFD) purposes, most drainage ditches are classified as either
79 Artificial Water Bodies (AWB), or as Heavily Modified Water Bodies (HMWB) if they follow the
80 course of a pre-existing watercourse (EU, 2000); although the number of designations of AWB and
81 HMWB vary widely between EU nations (Lieverink et al., 2011). Given their importance in
82 supporting the irrigation of crops and flood defence, they are managed primarily as agricultural and
83 flooding alleviation infrastructure. As a result, unlike other lentic and lotic surface waterbodies, their
84 ecology may not be required to be monitored on a regular basis, and there is no obligation for them to
85 achieve the WFD requirement of Good Ecological Status (GES). Instead, the alternative target of
86 Good Ecological Potential (GEP) is applied to AWB and HMWB. This designation reflects the
87 anthropogenic requirements placed upon them, the social and economic benefits of the services they
88 provide, and that it may not be practically or economically possible to modify or change the existing
89 configuration (EU, 2000; Environment Agency, 2009).

90 Agricultural drainage ditches have typically been reported to support lower taxonomic richness
91 compared with other waterbodies (streams, rivers, lakes and ponds), which has been attributed to their
92 close proximity to intensive agricultural activities and the runoff of herbicides, pesticides and
93 fertilisers into them, the latter reducing floral richness with knock-on effects on the fauna (e.g., Davies
94 et al., 2008b; Williams et al., 2003). However, a number of case studies have demonstrated the
95 importance of drainage ditches as reservoirs for aquatic fauna and flora populations (Goulder, 2008;
96 Foster et al., 1990; Painter, 1999; Verdonschot et al., 2011; Whatley et al., 2015). A number of studies
97 have also illustrated that drainage ditches can have significant conservation value, supporting high
98 biodiversity and communities of conservation value, even in intensively cultivated and managed
99 agricultural landscapes (e.g., Armitage et al., 2003; Davies et al., 2008b; Foster et al., 1990; Goulder,
100 2008; Watson & Ormerod, 2004; Williams et al., 2003). Ditches supporting high taxonomic richness
101 typically occur in areas where historic lowland fen occurred and often have continuity with ancient
102 wetlands (Davies et al., 2008b).

103 This paper aims to highlight the aquatic macroinvertebrate biodiversity and conservation value
104 associated with lowland agricultural drainage ditches (Artificial Water Bodies) and how recognition
105 of this value can be used to reconcile their anthropogenic function and appearance. We sought to
106 examine the following assumptions: i) main (arterial) drainage ditches will have a lower aquatic
107 macroinvertebrate biodiversity and conservation value than side ditches and: ii) there will be
108 significant spatial (between sites) and temporal (seasonal) heterogeneity in macroinvertebrate
109 communities among agricultural drainage ditches. The differences recorded should reflect local ditch
110 management regimes and the life history of the organisms inhabiting individual ditches.

111 **Materials and Methods**

112 *Study Sites*

113 Deeping Fen (TF 17643 17347) is an area of low-lying, intensively cultivated agricultural land
114 encircled by the River Glen and River Welland, Lincolnshire, UK. Historically, Deeping Fen was part
115 of 100,000 ha of wild fenland, but as a result of extensive draining for intensive arable agriculture
116 over several centuries, less than 55 ha of natural fenland remain, representing a loss of 99% (Boyes &
117 Russell, 1977; Wet Fens Partnership, 2015). An extensive network of drainage ditches, river
118 embankments and water pumping systems operate within the Welland and Deepings Internal
119 Drainage Board area. The drainage ditches are surrounded by intensive arable farming and subject to
120 water level management with water pumped from the ditches during periods of high rainfall into the
121 tidal River Welland to reduce flood risk. During the growing season and periods of low precipitation
122 water levels within the ditches are raised through a reduction in pumping, the management of weir
123 boards in side channels to reduce the drainage of water and through a series of valves on the R.
124 Welland and Greatford Cut that allow water into the system. In effect, the drainage ditches are kept
125 artificially low during the winter and raised during the summer to support agricultural irrigation and
126 provide environmental benefits to support the Cross Drain SSSI (Natural England, 2015). This results
127 in highly regulated water levels that are in complete contrast to the pattern displayed in the proximal
128 River Welland.

129 A total of 12 sites were surveyed in Deeping Fen on three occasions during 2014, corresponding to
130 spring, summer and winter. Two types of drainage ditch sites were selected: (i) 7 sites on two of the
131 longest main arterial drainage ditches - wider (> 5 m wide) and longer ditches which are connected to
132 a large number of side ditches. The main arterial drainage ditches (North Drove and South Drove
133 Drains) are maintained on an annual basis, with the vegetation on alternate banks cut / mown every
134 year and (ii) 5 side ditches – smaller (< 3 m wide) and shorter ditches connected at either end to a
135 main arterial drainage ditch, but both banks experience maintenance and vegetation management on
136 both banks on an annual basis. In addition, a long-term records collected by the Environment Agency
137 of England and Wales for 3 sites (1989 – 2014) in the drainage network were available. These data
138 provide a long term historical perspective of macroinvertebrate biodiversity within the agricultural
139 drainage ditches.

140 *Macroinvertebrate sampling*

141 Aquatic macroinvertebrate taxa were sampled using a kick / sweep-sample technique with a standard
142 pond net (mesh size 1 mm) over a three minute period (Armitage et al., 2003; Murray-Bligh, 1999).
143 Aquatic macroinvertebrate samples were collected during each survey (spring - April, summer - June
144 and winter - December) from each site. The samples were preserved in the field in 4% formaldehyde
145 solution and processed into 70% industrial methylated spirits in the laboratory. The majority of faunal
146 groups were identified to species level, although Sphaeriidae were identified to genus, Cladocera,
147 Ostracoda, Oligochaeta, Hydracarina, Collembola and Diptera were recorded as such.

148 *Statistical analysis*

149 Three measures of ditch aquatic macroinvertebrate diversity were calculated: alpha, beta and gamma
150 diversity. Alpha diversity represents the faunal diversity within an individual sample site, beta-
151 diversity characterises the spatial/temporal distribution and heterogeneity in community composition
152 between individual sites within a given area, and gamma diversity represents the overall biodiversity
153 across the entire study region (Anderson et al., 2011; Arellano & Halffter, 2003; Poggio et al., 2010).
154 Taxon richness and abundance was calculated for each ditch site (alpha) using the Species Diversity

155 and Richness IV software (Pisces Conservation, 2008). To achieve this, species-abundance data from
156 individual ditches for each season were combined in the final analysis. In addition, macroinvertebrate
157 biodiversity between seasons was also examined. Total aquatic macroinvertebrate diversity (gamma)
158 was calculated by combining species-abundance data from each ditch site. Jaccard's Coefficient of
159 Similarity (C_j) was calculated in the Community Analysis Package 3.0 program (Pisces Conservation,
160 2004) to quantify beta-diversity. The data was examined to ensure that the data complied with the
161 underlying assumptions of parametric statistical tests (e.g., normal distribution and homogeneity of
162 variances). Where these assumptions were not met, abundance data were \log_{10} transformed.
163 Differences in faunal diversity among ditches (main and side) were examined using one-way analysis
164 of variance (ANOVA) in SPSS (version 21, IBM Corporation, New York). Seasonal differences
165 (nested within ditch type) in macroinvertebrate richness and abundance among the ditch types were
166 examined using a nested analysis of variance (nested ANOVA) with the Sidak *post-hoc* test used to
167 determine where significant differences between seasons occurred (van de Meutter et al. 2005).

168 One-way analysis of variance was used to statistically assess the differences in Jaccard's Coefficient
169 of Similarity C_j among main and side ditches. In addition, heterogeneity of macroinvertebrate
170 communities between main and side ditch sites, and season (spring, summer and winter) samples was
171 assessed using Analysis of Similarity (ANOSIM) and summarized using Non-metric
172 Multidimensional Scaling (NMDS) ordination plots (using Bray-Curtis dissimilarity metric) in
173 PRIMER v6 (Clarke & Gorley, 2006). SIMPER analysis was undertaken to determine which taxa
174 contributed most to the seasonal (spring, summer and winter) differences in macroinvertebrate
175 community composition and between site (main/side) differences in taxonomic composition. Faunal
176 abundance data was square root transformed prior to ANOSIM, NMDS and SIMPER analysis.

177 The conservation value of the aquatic macroinvertebrates within each ditch site was determined using
178 the Community Conservation Index (CCI). This incorporates both rarity of macroinvertebrate species
179 at a national scale in the UK and the community richness (see Chadd & Extence, 2004 for further
180 methodological details). CCI can provide the basis for the development of conservation strategies

181 when used in conjunction with knowledge of the habitat requirements of target organisms and
182 communities (Chadd & Extence, 2004; Armitage et al. 2012).

183 **Results**

184 *Macroinvertebrate biodiversity*

185 A total of 167 taxa was recorded from the main (total: 150 taxa, mean: 85.9) and side (total: 133 taxa,
186 mean: 71.2) ditch sites during the three surveys in 2014 (Table 1). The largest numbers of taxa were
187 recorded from the orders Coleoptera (53), Gastropoda (27), Trichoptera (19), Hemiptera (17) and
188 Odonata (13). Two non-native taxa, *Crangonyx pseudogracilis* (Amphipoda) and *Potamopyrgus*
189 *antipodarum* (Gastropoda), were both recorded from all 12 study sites. Both species were abundant;
190 *C. pseudogracilis* accounted for up to 13% of the sample abundance and *P. antipodarum* accounted
191 for up to 12% of sample abundance.

192 Aquatic macroinvertebrate taxonomic richness was significantly greater within the main arterial
193 ditches when compared with the side ditches (ANOVA $F_{1, 12} = 6.182$; $p < 0.05$). The greatest number
194 of taxa (96 taxa) was recorded from a main ditch site whilst the lowest diversity (64 taxa) was
195 recorded from two side ditches. Higher taxonomic richness in the main ditches was driven by a
196 greater richness of Hemiptera, Coleoptera and Trichoptera taxa when compared with the side ditches
197 (Figure 1). No significant difference in aquatic macroinvertebrate abundance among main and side
198 ditches was recorded ($p > 0.05$).

199 When individual seasons (spring, summer and autumn) were considered, a significant difference in
200 the number of taxa (nested ANOVA $F_{4, 29} = 8.513$; $p < 0.001$) was observed among main and side
201 drainage ditches (Figure 2a). *Post hoc* analysis indicated that macroinvertebrate faunal richness was
202 significantly lower during the winter season than the spring or summer season (Figure 2a). Aquatic
203 Coleoptera (spring = 38 taxa, summer = 40 taxa, winter = 17 taxa), Hemiptera (spring = 13 taxa,
204 summer = 14 taxa, winter = 9 taxa) and Ditpera (spring = 8 taxa, summer = 9 taxa, winter = 4 taxa)
205 taxa displayed a significantly lower richness during the winter season. Aquatic macroinvertebrate

206 abundance did not differ among the three seasons ($P>0.05$) (Figure 2b) or when all seasons were
207 considered (average abundance: 3640 individuals all site; 3604 individuals - main ditches; 3690
208 individuals – side ditches; Table 1).

209 *Community heterogeneity*

210 A significant difference in community composition was recorded between main and side ditch
211 macroinvertebrate communities for the spring, summer and winter seasons, and when all sampling
212 dates were considered together (ANOSIM $p<0.01$). This difference resulted in a consistent separation
213 of main and side ditch samples within the NMDS ordination plots. The main ditch sites formed
214 relatively distinct clusters within the NMDS site plots for each of the seasonal surveys (Figures 3a - c)
215 and when all samples from three seasons were combined (Figure 3d). The side ditch sites were more
216 widely dispersed, indicating greater community heterogeneity, although there was some overlap with
217 the main ditch sites during spring (Figure 3a). SIMPER analysis indicated significant community
218 heterogeneity and that differences between main and side ditches was driven by greater abundances of
219 2 gastropods (*Radix balthica* and *Physa fontinalis*) in the side ditches and greater abundances of an
220 Ephemeroptera larva (*Cloeon dipterum*) and an amphipod shrimp (*Gammarus pulex*) in the main ditch
221 sites (Table 2a). Side ditches had significantly lower Jaccard's Coefficient of Similarity value during
222 the spring (main $C_j = 0.45$ side $C_j = 0.32$), summer (main $C_j = 0.48$ side $C_j = 0.39$) and when all
223 sample sites were combined (main $C_j = 0.57$ side $C_j = 0.47$) than main channel ditch sites (ANOVA
224 $p<0.001$) (Table 3). No significant difference in Jaccard's Coefficient of similarity was recorded
225 between main and side ditches during winter.

226 When seasonal differences in macroinvertebrate community composition within the drainage ditches
227 over three seasons (spring, summer and winter) were examined using NMDS, clear clusters of
228 samples were identified for samples collected during the spring, summer and winter respectively
229 (Figure 4). In addition, ANOSIM indicated that there were significant differences between spring,
230 summer and winter macroinvertebrate community composition (ANOSIM $P<0.01$). Seasonal
231 macroinvertebrate heterogeneity was driven by greater abundances of *C. dipterum* and a freshwater

232 shrimp (*C. pseudogracilis*) during the winter, greater abundances of *G. pulex* during spring and
233 significantly greater abundances of *R. balthica* and non-biting midge larvae (Chironomidae) during
234 the summer (Table 2b).

235 *Conservation Value*

236 Three nationally scarce or nationally notable Coleoptera were identified within the ditch sites; *Agabus*
237 *uliginosus* (Dytiscidae) was recorded from a single side ditch, *Oulimnius major* (Elmidae) was
238 recorded within both main ditches, *Scarodytes halensis* (Dytiscidae) was recorded from one main and
239 side ditch site and *Agabus undulatus* (Dytiscidae), listed as Lower Risk - Near Threatened on the
240 IUCN red data list 2001, was recorded from a single side ditch. Based on the CCI scores derived, the
241 macroinvertebrate communities within two ditch sites were of *fairly high conservation value* (1 main
242 and 1 side ditch), one side ditch was of a *high conservation value* and a single main drainage ditch
243 was of a *very high conservation value* (Table 4). No ditches were recorded to have a low conservation
244 value. There was no significant differences in CCI scores between main and side ditches for any
245 season or for the combined dataset ($P>0.05$). In addition, no significant difference in conservation
246 value between the seasons was recorded ($P>0.05$).

247

248 **Discussion**

249 *Macroinvertebrate biodiversity and community heterogeneity*

250 This study sought to characterise the aquatic macroinvertebrate biodiversity and conservation value of
251 lowland agricultural drainage ditches. The results of the study illustrate that the drainage ditches
252 examined support very high biodiversity at both the individual site (alpha diversity) and landscape
253 scale (gamma diversity), and that there was significant between site heterogeneity (beta diversity).
254 The number of aquatic macroinvertebrate taxa recorded in this study (167 taxa) was markedly higher
255 than that recorded on other studies of drainage ditches in the UK (Davies et al., 2008b) and
256 comparable to other wetland habitats (Williams et al., 2003). When the long-term historical data

257 (1989-2014) available for the sites were added to the taxa list from this study, the number of taxa
258 recorded almost doubled to 338 taxa (including 131 Coleoptera, 51 Gastropoda/Bivalvia, 35
259 Hemiptera and 26 Trichoptera). This figure is markedly higher than any other study reported in the
260 UK and second highest among drainage ditch studies of macroinvertebrate biodiversity reported
261 internationally (Table 5). This probably reflects the high connectivity within the drainage network
262 (River Welland and the River Glen) and proximity to remnant fen wetlands (Baston Fen SSSI and
263 Thurlby Fen Nature reserve) and fen restoration projects (Willow Tree Fen nature reserve).
264 Traditional wetland fens in the UK typically support exceptionally high aquatic macroinvertebrate
265 diversity (Eyre et al., 1990; Foster et al., 1990 Painter, 1999; Rouquette & Thompson, 2005). The
266 drainage ditches may effectively function as aquatic corridors through the agricultural landscape,
267 linking natural, semi-natural and artificial habitats (Buisson et al., 2008; Mazerolle, 2004).

268 We assumed that due to more frequent management operations (water level change, dredging, bank
269 cutting), main arterial drainage ditches would support lower macroinvertebrate biodiversity and
270 conservation value than the less frequently managed side ditches. No evidence was found to support
271 this assumption since the side ditches supported significantly lower aquatic macroinvertebrate taxon
272 richness (alpha) than main drainage ditch sites. The management practices, primarily designed to
273 maintain the hydrological functioning (conveyance of water) may actually inadvertently promote and
274 enhance aquatic macroinvertebrate diversity. Ditch cleaning and dredging has been shown to
275 positively influence Trichoptera presence in ditches (Twisk et al., 2000), and dredging can remove
276 nutrient rich sediment (Whatley et al., 2014a) and reset ditch habitats to an earlier successional stages
277 (Clarke, 2015). The rotational management of sites over time means a variety of vegetation
278 successional stages will be present across the sites and collectively these provide a wide range of
279 habitats suitable for macroinvertebrates (Clarke, 2015; Painter et al., 1999). Aquatic macrophytes
280 have been shown to be an important driver of aquatic macroinvertebrate communities (Whatley et al.,
281 2014a; Whatley et al., 2014b) and the riparian banks and channel of the main arterial ditches are cut
282 on alternate years. As a result, aquatic macrophytes (submerged and emergent) were present at all
283 sites and able to provide refuge, oxygenation, oviposition and feeding sites for macroinvertebrate taxa

284 (Bazzanti et al., 2010; de Szalay & Resh, 2000; Warfe & Barmuta, 2004). The reduced biodiversity
285 recorded in side ditches may reflect the more extensive management strategy (a greater proportion of
286 vegetation cutting and dredging on both banks), despite being managed less frequently.

287 Significant aquatic macroinvertebrate community heterogeneity was recorded between the main and
288 side drainage ditches, and across the three seasons. This supports the second assumption of the study,
289 that there would be significant heterogeneity in macroinvertebrate communities among the main and
290 side drainage ditches. The primary differences in the communities reflects the presence of taxa
291 associated with slow flow and lotic conditions such as the Crustacea *G. pulex* in the main arterial
292 drains compared to the side ditches which supported much higher abundances of gastropods such as
293 *P. fontinalis* and *R. balthica*. The ponding of water in side ditches during the winter and abundance of
294 structurally complex macrophyte communities within them provide ideal habitats and conditions for
295 gastropods (Bronmark, 1985; Hinojosa-Garro et al., 2010). However, invertebrate communities
296 among side ditches were more heterogeneous than the main drainage ditches; Jaccard's Similarity was
297 lower for side ditches than main arterial drainage ditches. This reflects the wider range of successional
298 stages present across side ditches (from freshly managed to largely vegetation) when compared to the
299 main arterial ditches where one bank was always vegetated. The high seasonal heterogeneity recorded
300 reflects the life-cycle characteristics and natural seasonal variability of aquatic macroinvertebrate
301 communities and reflects the pattern recorded in other freshwater systems.

302 *Conservation value and management of the resource*

303 Biodiversity conservation in many regions currently relies on designated protected areas (e.g., nature
304 reserves) (Mainstone, 2008; McDonald et al., 2008; Twisk et al., 2000). Protected area legislation, at a
305 national and European scale largely concentrates on the identification and selection of the best
306 examples of natural or semi-natural habitats. Within these protected areas adverse anthropogenic
307 stressors are minimised and the deterioration of 'target' habitat conditions can be avoided (Mainstone,
308 2008). However, agricultural activities and urban expansion are projected to threaten the flora and
309 fauna within many of these protected areas (Gunalp & Seto, 2013). As a result, habitat, biodiversity

310 of species conservation strategies should not depend exclusively on protected areas (Chester &
311 Robson, 2013) and opportunities to enhance them should be taken wherever possible.

312 It is increasingly recognised that the long-term conservation of habitats and species requires new /
313 novel approaches. The use of management strategies to increase the physical diversity of
314 anthropogenic habitats has begun to be used in some aquatic systems as a means to support native
315 flora and fauna (therefore promoting and enhancing biodiversity) whilst not reducing the effectiveness
316 of their primary anthropogenic function (Moyle, 2014). The management and conservation of
317 agricultural drainage ditches represent a prime example of a location where the principles of
318 '*reconciliation ecology*' (sensu Rosenzweig, 2003) could be applied for the mutual benefit of societal
319 requirements and conservation of natural resources. Reconciliation ecology

320 '...discovers how to modify and diversify anthropogenic habitats so that they harbour a wide
321 variety of wild species. In essence, it seeks to give many species back their geographical
322 ranges without taking away ours' (Rosenzweig, 2003, p.37).

323 Reconciliation ecology acknowledges that humans increasingly dominate many ecosystems,
324 especially agricultural landscapes (Rosenzweig, 2003), and that society has a responsibility to
325 determine what it wants these systems to look like aesthetically, how they function and what target
326 species we want them to support. If more widely accepted and adopted, reconciliation ecology could
327 provide a framework for supporting future conservation of biota within habitats that are increasingly
328 anthropogenically modified or dominated (Chester & Robson, 2013; Dudgeon et al. 2006;
329 Rosenzweig, 2003).

330 It has been widely acknowledged that many agricultural practices and land use patterns, especially
331 those of traditional agriculture, are already compatible with supporting biodiversity and agricultural
332 production (Benayas & Bullock, 2012), even if this has occurred consequentially rather than by
333 design. Therefore, there is a strong case to suggest that the principles of reconciliation ecology are
334 already in operation at the drainage ditch sites examined in this study since they support diverse
335 macroinvertebrate communities (alpha and gamma diversity) and support a number of aquatic
336 macroinvertebrate taxa with conservation designations. The CCI indicated that 2 drainage ditches

337 were of high or very high conservation value. These findings support some previous research on
338 drainage ditches which have illustrated their importance for biodiversity conservation in agricultural
339 areas (Armitage et al., 2003; Clarke, 2015; Davies et al., 2008b; Foster et al., 1990; Watson &
340 Ormerod, 2004; Williams et al., 2003). In many areas there have been calls and incentives for de-
341 intensification of agricultural land to reverse the decline in biodiversity through the use of voluntary
342 agri-environment schemes (Davies et al., 2008a). Agri-environment schemes in the UK aim to reduce
343 the widespread pollution of aquatic systems in agricultural landscapes typically through the
344 development of buffer strips. These are effectively narrow bands of land (buffers) surrounding aquatic
345 habitats left free from agricultural production and to absorb nutrients and chemical run-off (Davies et
346 al., 2009). However, while this may be an option in low productivity and on land of marginal
347 agricultural value, in highly productive and agricultural intensive landscapes this is not a realistic or
348 economically viable option. In addition, it may be more difficult to legitimise and implement when
349 the waterbodies in question are designated as artificial or heavily modified waterbodies (AWB or
350 HMWB) under the EU Water Framework Directive and little pre-existing information regarding their
351 ecological value is available.

352 Reconciliation ecology may provide an alternative practical approach to maintain, protect and
353 enhance aquatic biodiversity in agricultural areas. Ditches are well suited to reconciliation ecology
354 and many already support significant taxonomic richness (Armitage et al., 2003; Verdonschot et al.,
355 2011). Only small modifications to management (e.g., cut bank sides on alternate years) can
356 significantly enhance aquatic alpha and gamma diversity and conservation value in agricultural
357 landscapes (Twisk et al., 2000) whilst not reducing the anthropogenic utility of ditches. Given there
358 will be no loss of agricultural land or change to the primary function of the ditches (irrigation and
359 flood risk management), only very minor changes to existing management strategies and no/very low
360 financial costs, land managers and farmers may be more willing to implement reconciliation ecology
361 approaches to protect or enhance biodiversity than agri-environment schemes. However, given that
362 linear agricultural drainage ditch habitats are often the only remaining freshwater habitat in many
363 agricultural landscapes a greater appreciation and understanding of the wildlife resource (biodiversity)

364 associated with them is required to provide evidence to underpin future management strategies to
365 maximise the dual utility/benefits of drainage ditches for anthropogenic purposes and aquatic
366 biodiversity.

367 In the absence of formal legislative protection (the Water Framework Directive and Habitats Directive
368 overlook ditches) the ecology of large networks of agricultural drainage ditches are currently
369 unknown, ignored and potentially under threat. In some intensively farmed landscapes, drainage
370 ditches are being increasingly replaced by sub-surface drainage pipes to increase crop yield (Herzon
371 & Helenius, 2008). Land managers, farmers, environmental regulators and policy makers need to
372 recognise the conservation value and biological importance of drainage ditches as one of the last
373 remaining aquatic habitats and refuges available in agricultural areas and, where appropriate, provide
374 protection for most valuable sites.

375

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382

383 **References**

384 Anderson, M.J., T.O. Crist, J.M. Chase, M. Vellend, B.D. Inouye, A.L. Freestone, N.J. Sanders, H.V.
385 Cornell, L.S. Comita, K.F. Davies, S.P. Harrison, N.J.R. Kraft, J.C. Stegen & N.G. Swenson,
386 2011. Navigating the multiple meanings of β diversity: a road map for the practicing ecologist.
387 Ecology Letters 14: 19-28.

388 Armitage, P.D., A. Hawczak & J.H. Blackburn, 2012. Tyre track pools and puddles – Anthropogenic
389 contributors to aquatic biodiversity. *Limnologica* 42: 254-263.

390 Armitage, P.D., K. Szoszkiewicz, J.H. Blackburn & I. Nesbitt, 2003. Ditch communities: a major
391 contributor to floodplain diversity. *Aquatic Conservation: Marine and Freshwater Ecosystems*
392 13: 165-185.

393 Arellano, L. & G. Halffter, 2003. Gamma diversity: derived from and a determinant of alpha diversity
394 and beta diversity. An analysis of three tropical landscapes. *Acta Zoologica Mexicana* 90: 27-
395 76.

396 Bazzanti, M., C. Coccia, Giuseppina & M. Dowgiallo, 2010. Microdistribution of macroinvertebrates
397 in a temporary pond of Central Italy: taxonomic and functional analyses. *Limnologica* 40: 291-
398 299.

399 Benayas, J.M.R. & J.M. Bullock, 2012. Restoration of biodiversity and ecosystem services on
400 agricultural land. *Ecosystems* 15: 883-899

401 Boyes, J. & R. Russell, 1977. *The canals of Eastern England*. David and Charles, UK.

402 Brönmark, C. 1985. Freshwater snail diversity: effects of pond area, habitat heterogeneity and
403 isolation. *Oecologia* 67: 127-131.

404 Buisson, R.S.K., P.M. Wade, R.L. Cathcart, S.M. Hemmings, C.J. Manning & L. Mayer, 2008. *The*
405 *Drainage Channel Biodiversity Manual: Integrating Wildlife and Flood Risk Management*.
406 Association of Drainage Authorities and Natural England: Peterborough.

407 Chadd, R. & C. Extence, 2004. The conservation of freshwater macroinvertebrate populations: a
408 community based classification scheme. *Aquatic Conservation: Marine and Freshwater*
409 *Ecosystems* 14: 597-624.

410 Chester, E.T. & B.J. Robson, 2013. Anthropogenic refuges for freshwater biodiversity: Their
411 ecological characteristics and management. *Biological Conservation* 166: 64-75.

412 Clare, P. & R.W. Edwards, 1983. The macroinvertebrate fauna of the drainage channels of the Gwent
413 Levels, South Wales. *Freshwater Biology* 13: 205-225.

414 Clarke, K.R. & R.N. Gorley, 2006. PRIMER v6: User Manual/Tutorial. PRIMER E-Ltd: Plymouth,
415 UK.

416 Clarke, S.J. 2015. Conserving freshwater biodiversity: The value, status and management of high
417 quality ditch systems. *Journal for Nature Conservation* 24: 93-100.

418 Davies, B.R., J. Biggs, P.J. Williams, T.J. Lee & S. Thompson, 2008a. A comparison of the
419 catchment sizes of rivers, streams, ponds, ditches and lakes: implications for protecting aquatic
420 biodiversity in an agricultural landscape. *Hydrobiologia* 597: 7-17.

421 Davies, B., J. Biggs, P. Williams & S. Thompson, S. 2009. Making agricultural landscapes more
422 sustainable for freshwater biodiversity: a case study from southern England. *Aquatic
423 Conservation: Marine and Freshwater Ecosystems* 19: 439-447.

424 Davies, B.R., J. Biggs, P. Williams, M. Whitfield, P. Nicolet, D. Sear, S. Bray & S. Maund, 2008b.
425 Comparative biodiversity of aquatic habitats in the European agricultural landscape.
426 *Agriculture. Ecosystems and Environment* 125: 1-8.

427 de Szalay, F.A & V.H. Resh, 2000. Factors influencing macroinvertebrate colonization of seasonal
428 wetlands: responses to emergent plant cover. *Freshwater Biology* 45: 295-308.

429 Dudgeon, D., A.H. Athington, M.O. Gessner, Z. Kawabata, D.J. Knowler, C. Leveque, R.J. Naiman,
430 A. Prieur-Richard, D. Soto, M.L.J. Stiassy & C.A. Sullivan, 2006. Freshwater biodiversity:
431 importance, threats, status and conservation challenges. *Biological Reviews* 81: 163-182.

432 EU, 2000. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000
433 establishing a framework for Community action in the field of water policy, 22/12/2000.
434 *Official Journal L 327*: 1-73.

435 Environment Agency, 2009. River Basin Management Plan, Anglian River Basin District Annex I:
436 Designating artificial & heavily modified water bodies. Environment Agency, UK.

437 Eyre, M.D., G.N. Foster & A.P. Foster, 1990. Factors affecting the distribution of water beetle species
438 assemblages in drains of eastern England. *Journal of Applied Ecology* 109: 217-225.

439 Foster, G.N., A.P. Foster, M.D. Eyre & D.T. Bilton, 1990. Classification of water beetle assemblages
440 in arable fenland and ranking of sites in relation to conservation value. *Freshwater Biology* 22:
441 343-354.

442 Goulder, R., 2008. Conservation of aquatic plants in artificial watercourses: are drains a substitute for
443 vulnerable navigation canals? *Aquatic Conservation: Marine and Freshwater Ecosystems* 18:
444 163-174.

445 Guneralp, B., & K.C. Seto, 2013. Futures of global urban expansion: uncertainties and implications of
446 biodiversity conservation. *Environmental Research Letters* 8: 1-10.

447 Herzon, I., & J. Helenius, 2008. Agricultural drainage ditches, their biological importance and
448 functioning. *Biological Conservation* 141: 1171-1183.

449 Hinojosa-Garro, D., C.F. Mason & G.J.C. Underwood, 2010. Influence of macrophyte spatial
450 architecture on periphyton and macroinvertebrate community structure in shallow waterbodies
451 under contrasting land management. *Fundamental and Applied Limnology* 177: 19-37.

452 Holden, J., P.J. Chapman & J.C. Labadz, 2004. Artificial drainage of peatlands: hydrological and
453 hydrochemical processes and wetland restoration. *Progress in Physical Geography* 28: 95-123.

454 Katano, O., K. Hosoya, K. Iguchi, M. Yamaguchi, Y. Aonuma & S. Kitano, 2003. Species diversity
455 and abundance of freshwater fishes in irrigation ditches around rice fields. *Environmental*
456 *Biology of Fishes* 66: 107-121.

457 Langheinrich, U., S. Tischew, R.M. Gersberg & V. Lüderitz, 2004. Ditches and canals in management
458 of fens: opportunity or risk? A case study in the Drömling Natural Park, Germany. *Wetland*
459 *Ecology and Management* 12: 429-445.

460 Leslie, A.W., R.F. Smith, D.E. Ruppert, K. Bejleri, J.M. McGrath, B.A. Needelman & W.O. Lamp,
461 2012. Environmental factors structuring benthic macroinvertebrate communities of agricultural
462 ditches in Maryland. *Environmental Entomology*: 41: 802-812.

463 Liefferink, D., M. Wiering & Y. Uitenboogaart, 2011. The EU Water Framework Directive: A multi-
464 dimensional analysis of implementation and domestic impact. *Land Use Policy* 28: 712-722.

465 Mainstone, C P., 2008. The role of specially designated wildlife sites in freshwater conservation – an
466 English perspective. *Freshwater Reviews* 1: 89-98.

467 Maltby, E. & M.C. Acreman, 2001. Ecosystem services of wetlands: pathfinder for a new paradigm.
468 *Hydrological Sciences Journal* 56: 1341-1359.

469 Maltchik, L., A. Silvia Rolon, C. Stenert, I. Farina Machado & O. Rocha, 2011. Can rice field
470 channels contribute to biodiversity conservation in Southern Brazilian wetlands? *Revista de*
471 *Biología Tropical* 59: 1895-1914.

472 Mazerolle, M.J., 2004. Drainage ditches facilitate frog movements in a hostile landscape. *Landscape*
473 *Ecology* 20: 579-590.

474 McDonald R.I., P. Kareiva & R.T.T. Forman, 2008. The implications of current and future
475 urbanization for global protected areas and biodiversity conservation. *Biological Conservation*
476 141: 1695-1703.

477 Moyle, P.B., 2014. Novel aquatic ecosystems: the new reality for streams in California and other
478 Mediterranean climate regions. *River Research and Applications* 30: 1335-1344.

479 Murray-Bligh J., 1999. Procedures for collecting and analysing macroinvertebrate samples—BT001 .
480 The Environment Agency: Bristol.

481 Natural England. 2015. SSSI citation for Cross Drain.
482 http://www.sssi.naturalengland.org.uk/citation/citation_photo/1003003.pdf (Accessed
483 31/07/15).

484 Painter, D., 1999. Macroinvertebrate distributions and the conservation value of aquatic Coleoptera,
485 Mollusca and Odonata in the ditches of traditionally managed and grazing fen at Wicken Fen,
486 UK. *Journal of Applied Ecology* 36: 33-48.

487 Pisces Conservation, 2004. Community Analysis Package Version 3.0. Pisces Conservation Ltd,
488 Lymington, UK.

489 Pisces Conservation, 2008. Species Diversity and Richness IV. Pisces Conservation Ltd, Lymington,
490 UK.

491 Poggio, S.L., E.J. Chaneton & C.M. Ghersa, 2010. Landscape complexity differentially affects alpha,
492 beta and gamma diversities of plants occurring in fence rows and crop fields. *Biological*
493 *Conservation* 143: 2477-2486.

494 Rosenzweig, M.L., 2003. Reconciliation ecology and the future of species diversity. *Oryx* 37: 194-
495 205.

496 Rouquette, J.R. & D.J. Thompson, 2005. Habitat associations of the endangered damselfly,
497 *Coenagrion mercurial*, in a water meadow ditch system in southern England. *Biological*
498 *Conservation* 123: 225-235.

499 Sayer, C., K. Andrews, E. Shiland, N. Edmonds, R. Edmonds-Brown, I. Patmore, D. Emson & J.
500 Axmacher, 2012. The role of pond management for biodiversity conservation in an agricultural
501 landscape. *Aquatic Conservation: Marine and Freshwater Ecosystems* 22: 626-638.

502 Twisk, W., M.A.W. Noordervliet & W.J. ter Keurs, 2000. Effects of ditch management on caddisfly,
503 dragonfly and amphibian larvae in intensively farmed peat areas. *Aquatic Ecology* 34: 397-411.

504 van de Meutter, F., R. Stoks & L. De Meester, 2005. The effect of turbidity state and microhabitat on
505 macroinvertebrate assemblages: a pilot study of six shallow lakes. *Hydrobiologia* 542: 379-390.

506 van Eerden, M., G. Lenselink & M. Zijlstra, 2010. Long term changes in wetland area and
507 composition in the Netherlands affecting the carrying capacity for wintering waterbirds. *Ardea*
508 98: 265-282.

509 Verdonschot, P.F.M. & L.W.G. Higler, 1989, Macroinvertebrates in Dutch ditches: a typological
510 characterization and the status of the Demmerik ditches. *Hydrobiological Bulletin* 21: 135-142

511 Verdonschot, R.C.M., H.E. Keizer-Vlek & P.F.M. Verdonschot, 2011. Biodiversity value of
512 agricultural drainage ditches: a comparative analysis of the aquatic invertebrate fauna of ditch
513 and small lakes. *Aquatic Conservation: Marine and Freshwater Ecosystems* 21: 715-727.

514 Vaikre, M., L. Remm & E. Rannap, 2015, Macroinvertebrates in woodland pools and ditches and
515 their response to artificial drainage in Estonia. *Hydrobiologia* 762: 157-168.

516 Warfe, D.M. & L.A. Barmuta, 2004. Habitat structural complexity mediates the foraging success of
517 multiple predator species. *Oecologia* 141: 171-178.

518 Watson, A.M. & S.J. Ormerod, 2004. The distribution of three uncommon freshwater gastropods in
519 the drainage ditches of British grazing marshes. *Biological Conservation* 118: 455-466.

520 Wet Fens Partnership. 2015. The restoration of fenland for people and wildlife. Environment Agency,
521 Peterborough.

522 Whatley, M.H., E.E. van Loon, H. van Dam, J.A. Vonk, H.G. van der Geest & W. Admiraal, 2014a.
523 Macrophyte loss drives decadal change in benthic invertebrates in peatland drainage ditches.
524 *Freshwater Biology* 59: 114-126.

525 Whatley, M.H., E.E. van Loon, J.A. Vonk, H.G. van der Geest & W. Admiraal, 2014b. The role of
526 emergent vegetation in structuring aquatic insect communities in peatland drainage ditches.
527 *Aquatic Ecology* 48: 267-283.

528 Whatley, M.H., J.A. Vonk, H.G. van der Geest & W. Admiraal, (2015). Temporal abiotic variability
529 structures invertebrate communities in agricultural drainage ditches. *Limnologica* 52: 20-29.
530 DOI: <http://dx.doi.org/10.1016/j.limno.2015.03.001>.

531 Williams, P., M. Whitfield, J. Biggs, S. Bray, G. Fox, P. Nicolet & D Sear, 2003. Comparative
532 biodiversity of rivers, streams, ditches and ponds in an agricultural landscape in Southern
533 England. *Biological Conservation* 115: 329-341.

534

Tables

Table 1 - Aquatic macroinvertebrate abundance (a), taxon richness and (b) in main arterial drains and side ditches for each season and combined over the year. Spr = spring, summ = summer, wint = winter. Standard error is presented in parenthesis.

		Abundance				Taxon richness			
		Spr	Summ	Wint	Comb	Spr	Summ	Wint	Comb
	a)								
Main	M1	1740	2359	1332	5431	55	60	39	92
	M2	1275	1246	1074	3595	51	68	35	86
	M3	1258	965	1115	3338	47	69	46	93
	M4	1456	1604	493	3553	49	73	27	92
	M5	1052	1855	640	3547	37	57	32	77
	M6	428	593	1058	2079	40	42	17	65
	M7	687	652	2344	3683	64	52	37	96
	Mean	1128.0 (±169.8)	1324.9 (±246.5)	1150.9 (±227.1)	3603.7 (±369.9)	49.0 (±3.4)	60.1 (±4.1)	33.2 (±3.5)	85.9 (±4.2)
Side	S1	1642	2515	1295	5452	43	40	31	69
	S2	321	969	634	1924	38	46	33	64
	S3	2082	3158	2060	7300	56	62	46	83
	S4	292	502	332	1126	33	44	23	64
	S5	1023	1303	321	2647	32	54	31	76
	Mean	1072 (±355.0)	1689.4 (±495.8)	928.4 (±333.6)	3689.8 (±1160.1)	40.4 (±4.3)	49.2 (±3.9)	32.8 (±3.7)	71.2 (±3.7)
	Total	13256	17721	12698	43675	130	132	95	167
	Mean	1104.7 (±168.3)	1476.8 (±244)	1058.2 (±185.4)	3639.6 (±497.6)	45.4 (±2.9)	55.6 (±3.2)	33.1 (±2.5)	79.8 (±3.5)

Table 2 - The top 4 aquatic macroinvertebrate taxa contributing most to community dissimilarity identified by SIMPER between: a) main and side ditches for all sampling dates and; b) spring, summer and autumn communities. Note - figure in parenthesis indicates the percentage contribution to community dissimilarity.

a)

	Side
Main	<i>Cloeon dipterum</i> (6.26)
	<i>Gammarus pulex</i> (4.77)
	<i>Radix balthica</i> (4.22)
	<i>Physa fontinalis</i> (3.58)

b)

	Spring	Summer	Winter
Spring			
Summer	<i>Gammarus pulex</i> (4.29)		
	<i>Radix balthica</i> (3.84)		
	<i>Cloeon dipterum</i> (3.79)		
	<i>Chironomidae</i> (3.52)		
Winter	<i>Cloeon dipterum</i> (7.41)	<i>Cloeon dipterum</i> (5.82)	
	<i>Gammarus pulex</i> (5.49)	<i>Gammarus pulex</i> (4.0)	
	<i>Crangonyx pseudogracilis</i> (4.47)	<i>Crangonyx pseudogracilis</i> (3.74)	
	<i>Chironomidae</i> (3.82)	<i>Radix balthica</i> (3.47)	

Table 3 - Jaccard's Coefficient of Similarity for macroinvertebrate communities for individual seasons and combined seasons from the main and side ditches

	Spring	Summer	Winter	Combined
Main	0.45	0.48	0.4	0.57
Side	0.32	0.39	0.42	0.47
All ditch samples	0.38	0.42	0.38	0.51

Table 4 - Macroinvertebrate Community Conservation Index (CCI) scores from the 12 sample sites for individual seasons and all seasons (Total), (0-5 low conservation value; >5-10 moderate conservation value; >10-15 fairly high conservation value; >15-20 high conservation value and >20 very high conservation value). Fairly high, high and very high conservation value scores are presented in bold.

	Spring	Summer	Winter	Total
Main				
M1	9	9	9	10
M2	12	13	8	14
M3	19	15	9	22
M4	9	9	8	10
M5	8	9	9	9
M6	9	8	8	9
M7	9	8	9	9
Side				
S1	13	9	7	14
S2	8	8	7	9
S3	8	8	8	8
S4	9	10	10	10
S5	22	9	9	20

Table 5 – The number of macroinvertebrate taxa recorded in other published studies which have examined the biodiversity or wider conservation value of artificial drainage channels and ditches. For each source the geographical location, number of ditches and sites examined, the number of macroinvertebrate taxa and duration of the study is included to provide comparison with the results of the current study and historic sampling on Deeping Fen.

Source	Location	Number sites	Number of taxa	Study date and duration
Armitage et al., (2003)	River Frome floodplain, Dorset, UK	1 ditch, 16 sites	145	1-year (1998)
Clare and Edwards, (1983)	Gwent Levels, River Severn Estuary, Wales, UK	60 sites	58 ¹	1-year, 6 surveys (1976)
Davies et al., (2008a)	Gloucestershire, Oxfordshire & Wiltshire, UK	20 sites / ditches	120	3 years (2000, 2002 and 2003)
Davies et al., (2008b)	River Cole, Coleshill, Oxfordshire, UK	11 sites	120	2-years (2000-2001)
	Whitchurch, Cheshire, UK	13 sites	75	2-years (1997-1998)
Hill et al., (This Study)	Deeping Fen, between River Glen and River Welland, Lincolnshire, UK	12 sites / 9 ditches	167 ²	1 Year, 3 surveys (2014)
Historic data		3 Sites	331 ²	1989-2014
Langheinrich et al., (2004)	Drömling, Saxony, Germany	11 sites / channels	227	3 years, 5 surveys (1996, 1998 and 2000)
Leslie et al., (2012)	Chesapeake Bay, Maryland, USA	29 sites / ditches	85	2 months (February-March 2008)
Painter, (1999)	Wicken Fen, Cambridgeshire, UK	17 sites / channels	109 ³	1 month (June 1994)
Verdonschot et al., (2011)	Central Netherlands	9 sites / drainage ditches	226	2-months (June-July 2005)
Verdonschot & Higler (1989)	Overijssel province, Drenthe provinde and Demmerik polder, Netherlands	150 sites	360 ⁴	Composite study of research in 1970's & 1980's
Whatley et al., (2014a)	Hoogheemraadschap, North Holland, Netherlands	29 sites	71	1985-2007
Whatley et al., (2014b)	Wormer, Jisperveld and Naardermeer, North Holland, Netherlands	6 sites / channels	70 ⁵	2 months (August-September 2011)
Whatley et al., (2015)	North Holland, Netherlands	84 sites / channels	159	4-years (2008-2011)
Williams et al., (2003)	River Cole, Coleshill, Oxfordshire, UK	20 sites / channels	90	1 year -2000

Notes: ¹Clare and Edwards (1983) report 58 taxa in a reduced dataset; ² Diptera larvae resolved to family level only; ³ Painter (1999) Only Coleoptera, Mollusca and Odonata reported; ⁴ Verdonschot and Higler (1989) the figure indicated comprises those selected for inclusion in analysis; ⁵ Whatley et al., (2014b) only insect taxa reported.

Figure captions

Figure 1 - Total number of taxa within the main macroinvertebrate groups recorded from the 12 sample sites.

Figure 2 - Error bar graphs indicating (a) Mean taxon richness (± 1 SE) and (b) mean community abundance (± 1 SE) recorded in the main and side drainage ditches during the spring, summer and winter sampling seasons.

Figure 3 - Two dimensional NMDS plot of dissimilarity (Bray-Curtis) of invertebrate communities within the main and side drainage ditches for: (a) spring (b) summer (c) winter and (d) all seasons combined.

Figure 4 - Two dimensional NMDS plot of dissimilarity (Bray-Curtis) of seasonal (spring, summer and winter) invertebrate communities within the agricultural drainage ditches.

Figure 1

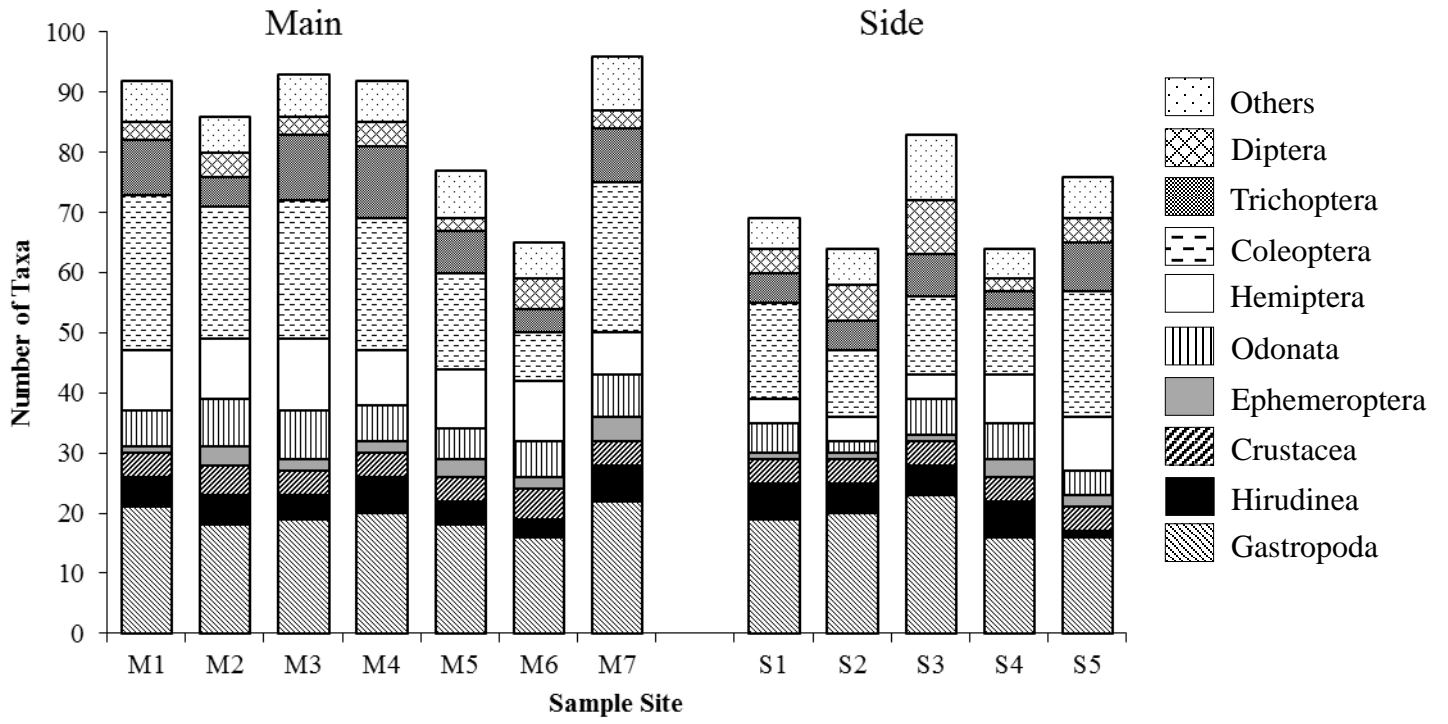


Figure 2

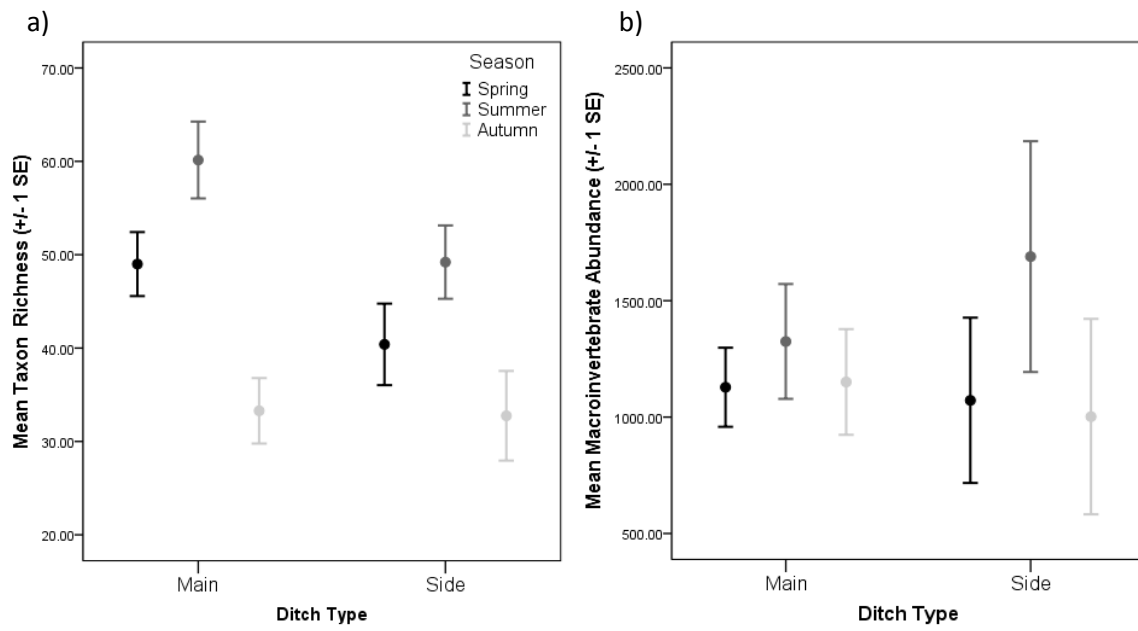


Figure 3

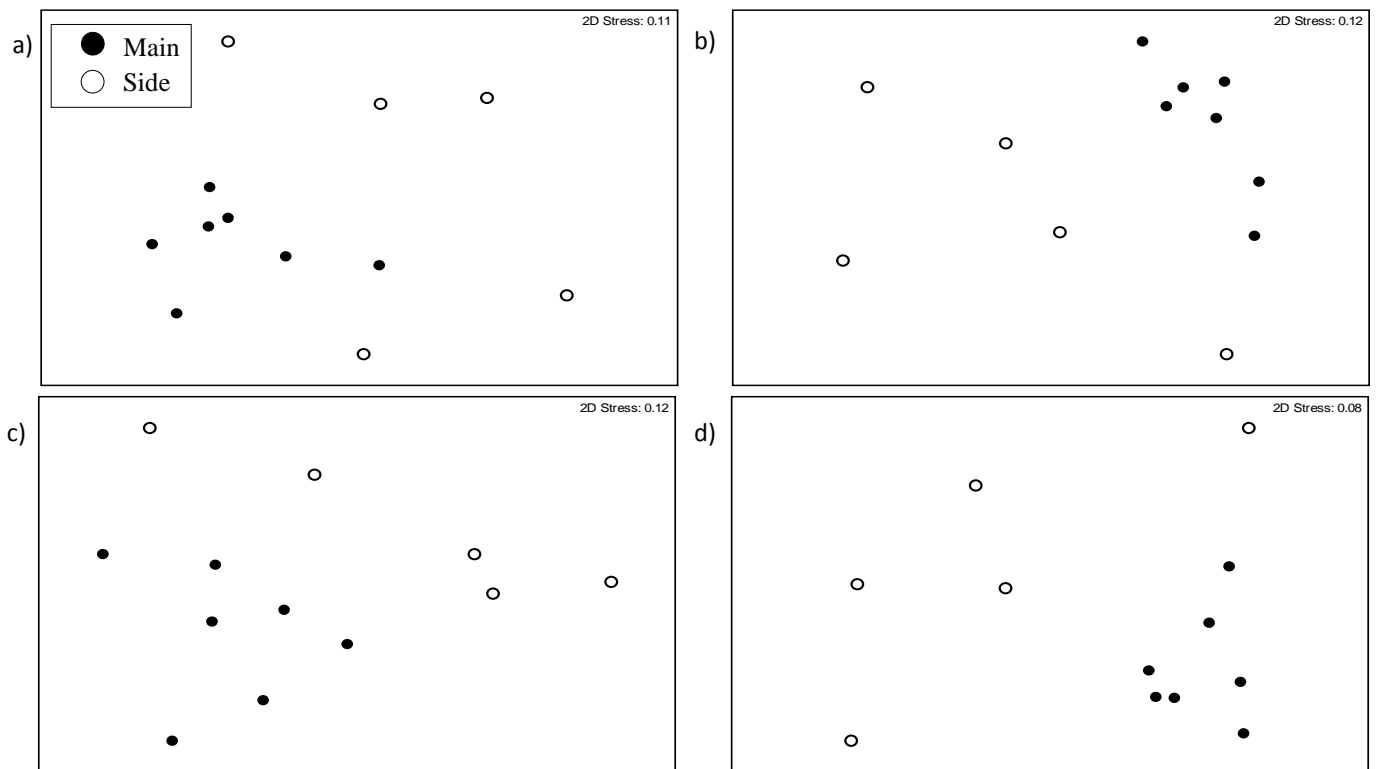


Figure 4

