Aquatic macroinvertebrate biodiversity associated with artificial agricultural drainage ditches

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Abstract

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

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

Introduction

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

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

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

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

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

Materials and Methods

Study Sites

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

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

Macroinvertebrate sampling

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

Statistical analysis

Three measures of ditch aquatic macroinvertebrate diversity were calculated: alpha, beta and gamma diversity. Alpha diversity represents the faunal diversity within an individual sample site, beta-diversity characterises the spatial/temporal distribution and heterogeneity in community composition between individual sites within a given area, and gamma diversity represents the overall biodiversity across the entire study region (Anderson et al., 2011; Arellano & Halffter, 2003; Poggio et al., 2010). Taxon richness and abundance was calculated for each ditch site (alpha) using the Species Diversity and Richness IV software (Pisces Conservation, 2008). To achieve this, species-abundance data from individual ditches for each season were combined in the final analysis. In addition, macroinvertebrate biodiversity between seasons was also examined. Total aquatic macroinvertebrate diversity (gamma) was calculated by combining species-abundance data from each ditch site. Jaccard's Coefficient of Similarity (Cj) was calculated in the Community Analysis Package 3.0 program (Pisces Conservation, 2004) to quantify beta-diversity. The data was examined to ensure that the data complied with the underlying assumptions of parametric statistical tests (e.g., normal distribution and homogeneity of variances). Where these assumptions were not met, abundance data were $\log_{10}$ transformed. Differences in faunal diversity among ditches (main and side) were examined using one-way analysis of variance (ANOVA) in SPSS (version 21, IBM Corporation, New York). Seasonal differences (nested within ditch type) in macroinvertebrate richness and abundance among the ditch types were examined using a nested analysis of variance (nested ANOVA) with the Sidak *post-hoc* test used to determine where significant differences between seasons occurred (van de Meutter et al. 2005).

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

The conservation value of the aquatic macroinvertebrates within each ditch site was determined using the Community Conservation Index (CCI). This incorporates both rarity of macroinvertebrate species at a national scale in the UK and the community richness (see Chadd & Extence, 2004 for further methodological details). CCI can provide the basis for the development of conservation strategies when used in conjunction with knowledge of the habitat requirements of target organisms and communities (Chadd & Extence, 2004; Armitage et al. 2012).

Results

Macroinvertebrate biodiversity

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

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

When individual seasons (spring, summer and autumn) were considered, a significant difference in the number of taxa (nested ANOVA $F_{4, 29} = 8.513$; $p<0.001$) was observed among main and side drainage ditches (Figure 2a). *Post hoc* analysis indicated that macroinvertebrate faunal richness was significantly lower during the winter season than the spring or summer season (Figure 2a). Aquatic Coleoptera (spring = 38 taxa, summer = 40 taxa, winter = 17 taxa), Hemiptera (spring = 13 taxa, summer = 14 taxa, winter = 9 taxa) and Ditpera (spring = 8 taxa, summer = 9 taxa, winter = 4 taxa) taxa displayed a significantly lower richness during the winter season. Aquatic macroinvertebrate abundance did not differ among the three seasons (P>0.05) (Figure 2b) or when all seasons were considered (average abundance: 3640 individuals all site; 3604 individuals - main ditches; 3690 individuals – side ditches; Table 1).

Community heterogeneity

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

When seasonal differences in macroinvertebrate community composition within the drainage ditches over three seasons (spring, summer and winter) were examined using NMDS, clear clusters of samples were identified for samples collected during the spring, summer and winter respectively (Figure 4). In addition, ANOSIM indicated that there were significant differences between spring, summer and winter macroinvertebrate community composition (ANOSIM P<0.01). Seasonal macroinvertebrate heterogeneity was driven by greater abundances of *C. dipterum* and a freshwater shrimp (*C. pseudogracilis*) during the winter, greater abundances of *G. pulex* during spring and significantly greater abundances of *R. balthica* and non-biting midge larvae (Chironomidae) during the summer (Table 2b).

Conservation Value

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

Discussion

Macroinvertebrate biodiversity and community heterogeneity

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

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

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

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

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

Conservation value and management of the resource

Biodiversity conservation in many regions currently relies on designated protected areas (e.g., nature reserves) (Mainstone, 2008; McDonald et al., 2008; Twisk et al., 2000). Protected area legislation, at a national and European scale largely concentrates on the identification and selection of the best examples of natural or semi-natural habitats. Within these protected areas adverse anthropogenic stressors are minimised and the deterioration of 'target' habitat conditions can be avoided (Mainstone, 2008). However, agricultural activities and urban expansion are projected to threaten the flora and fauna within many of these protected areas (Guneralp & Seto, 2013). As a result, habitat, biodiversity of species conservation strategies should not depend exclusively on protected areas (Chester & Robson, 2013) and opportunities to enhance them should be taken wherever possible.

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

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

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

It has been widely acknowledged that many agricultural practices and land use patterns, especially those of traditional agriculture, are already compatible with supporting biodiversity and agricultural production (Benayas & Bullock, 2012), even if this has occurred consequentially rather than by design. Therefore, there is a strong case to suggest that the principles of reconciliation ecology are already in operation at the drainage ditch sites examined in this study since they support diverse macroinvertebrate communities (alpha and gamma diversity) and support a number of aquatic macroinvertebrate taxa with conservation designations. The CCI indicated that 2 drainage ditches were of high or very high conservation value. These findings support some previous research on drainage ditches which have illustrated their importance for biodiversity conservation in agricultural areas (Armitage et al., 2003; Clarke, 2015; Davies et al., 2008b; Foster et al., 1990; Watson & Ormerod, 2004; Williams et al., 2003). In many areas there have been calls and incentives for de-intensification of agricultural land to reverse the decline in biodiversity through the use of voluntary agri-environment schemes (Davies et al., 2008a). Agri-environment schemes in the UK aim to reduce the widespread pollution of aquatic systems in agricultural landscapes typically through the development of buffer strips. These are effectively narrow bands of land (buffers) surrounding aquatic habitats left free from agricultural production and to absorb nutrients and chemical run-off (Davies et al., 2009). However, while this may be an option in low productivity and on land of marginal agricultural value, in highly productive and agricultural intensive landscapes this is not a realistic or economically viable option. In addition, it may be more difficult to legitimise and implement when the waterbodies in question are designated as artificial or heavily modified waterbodies (AWB or HMWB) under the EU Water Framework Directive and little pre-existing information regarding their ecological value is available.

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

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

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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)				b)			
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	*Cloeon dipterum* (6.26) *Gammarus pulex* (4.77) *Radix balthica* (4.22) *Physa fontinalis* (3.58)

b)

	Spring	Summer	Winter
Spring			
Summer	*Gammarus pulex* (4.29) *Radix balthica* (3.84) *Cloeon dipterum* (3.79) *Chironomidae* (3.52)		
Winter	*Cloeon dipterum* (7.41) *Gammarus pulex* (5.49) *Crangonyx pseudogracilis* (4.47) *Chironomidae* (3.82)	*Cloeon dipterum* (5.82) *Gammarus pulex* (4.0) *Crangonyx pseudogracilis* (3.74) *Radix balthica* (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: [1]Clare 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

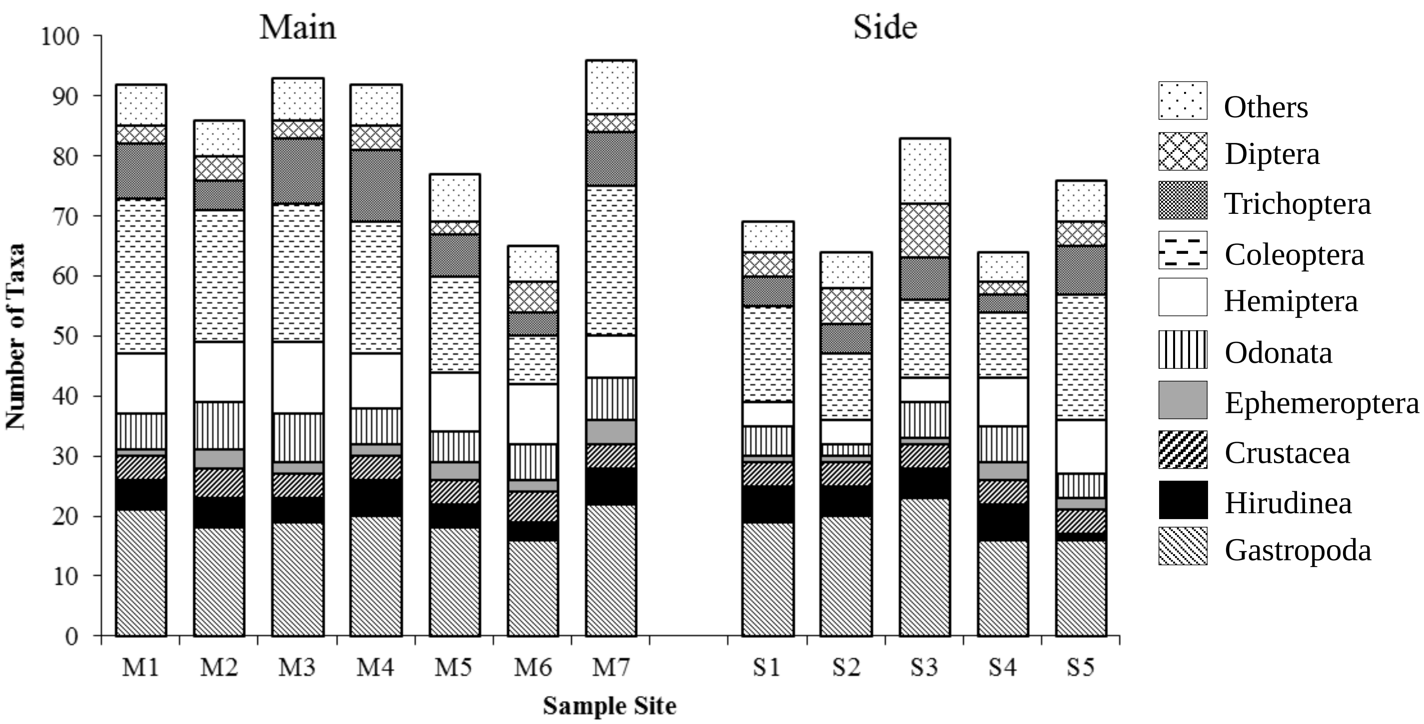
Figure 1 - Total number of taxa within the main macroinvertebrate groups recorded from the 12 sample sites.

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



Main
Side
Number of Taxa
0
10
20
30
40
50
60
70
80
90
100
M1
M2
M3
M4
M5
M6
M7
S1
S2
S3
S4
S5
Sample Site
Others
Diptera
Trichoptera
Coleoptera
Hemiptera
Odonata
Ephemeroptera
Crustacea
Hirudinea
Gastropoda

Figure 2

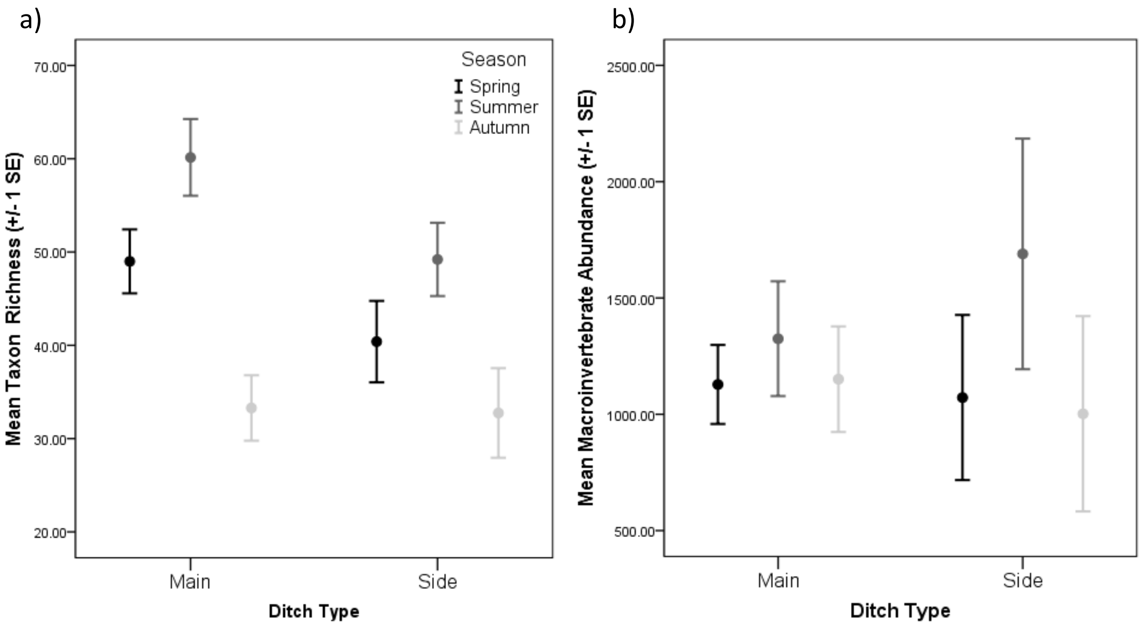

a)
Season
Spring
Summer
Autumn
Mean Taxon Richness (+/- 1 SE)
70.00
60.00
50.00
40.00
30.00
20.00
Main
Side
Ditch Type
b)
Mean Macroinvertebrate Abundance (+/- 1 SE)
2500.00
2000.00
1500.00
1000.00
500.00
Main
Side
Ditch Type

Figure 3

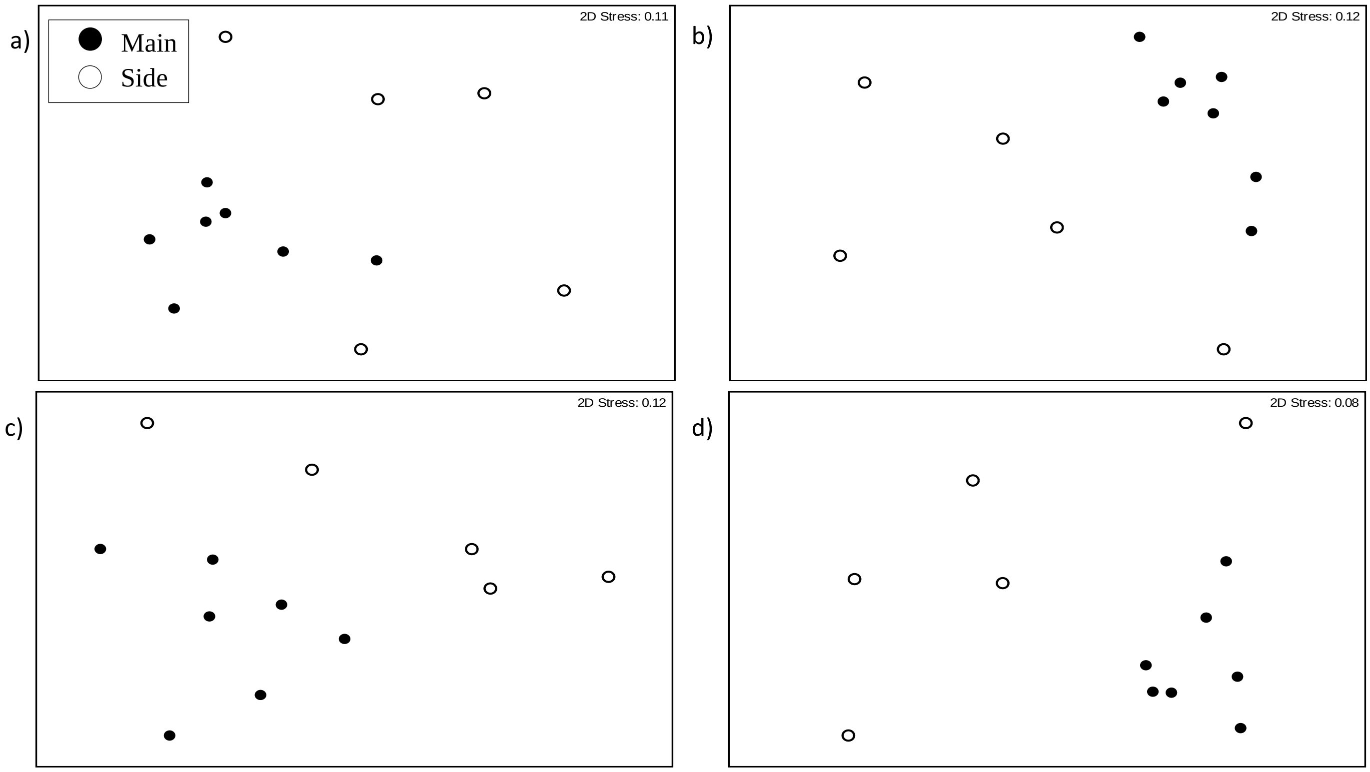

a)
Main
Side
2D Stress: 0.11
b)
2D Stress: 0.12
c)
2D Stress: 0.12
d)
2D Stress: 0.08

Figure 4