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Version: Accepted Manuscript

Link(s) to article on publisher's website:
http://dx.doi.org/doi:10.1016/j.ecoleng.2017.04.055

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Citation: Day G., Mayes W.M., Wheeler P.M., Hull S.L. (2017) Can aggregate quarry silt lagoons provide resources for wading birds? *Ecological Engineering* **105:** 189-197.

Can aggregate quarry silt lagoons provide resources for wading birds?

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Abstract

Wading birds have declined across Europe as the intensification of lowland agriculture has resulted in the loss and degradation of wetland areas. Lowland aggregate extraction sites that incorporate areas of fine, waste sediments deposited in silt lagoons have the potential to be restored for wader conservation. We set out to determine the potential value of silt lagoons to wading birds by comparing the water quality, sediment profiles, aquatic invertebrate abundance and diversity (prey availability) and wader site use at five sites representing various stages of active aggregate extraction and restoration for conservation purposes. Wader counts were conducted monthly over a twelve month period using replicated scan samples, and the invertebrate communities studied during the breeding and autumn migration season (June-September). Water quality variables were similar between sites, but sediments from active quarries were dominated by moderately sorted fine sands in comparison to the coarser sediment profiles of restored areas. June and September there was no significant difference in invertebrate diversity between sites, however richness was significantly lower and total abundance a factor of ten higher at restored sites than on silt lagoons, with the dominant taxa similar across all sites. Waders used all sites; albeit at lower abundance and richness on silt lagoons and two species were recorded breeding on active silting sites. We conclude that the fine, uniform sediments of modern silt lagoons limited invertebrate diversity and abundance, diminishing the value of silt lagoons to waders. Simple low-cost intervention measures increasing substrate heterogeneity and creating temporary ponds could increase invertebrate richness and abundance, and enhance the conservation potential of these sites.

Keywords

Silt lagoons, wetland intervention, waders, aquatic invertebrate diversity and abundance, conservation, restoration.

1. Introduction

Wetlands, a globally threatened habitat, are internationally important for water bird and wader communities (Grygoruk and Ignar, 2015; Kloskowski et al., 2009). Large-scale land use changes within the last century have led to significant declines in lowland perennial and seasonal wetlands within Europe (Gumiero et al., 2013; Verhoeven, 2014). Agricultural intensification leading to the abandonment of wet areas (Joyce, 2014), land reallocation or agricultural encroachment, over grazing, changes in water management or wetland drainage have been held largely accountable for the loss of 50% of European wetlands within the last century (Silva et al., 2007; Henle et al., 2008). The internationally important European wetlands, salt marshes and mud flats support populations of over-wintering and migrating waterbirds and waders along the East Atlantic Flyway (EAF) (Rehfisch et al., 2003; Stroud et al., 2006; Holt et al., 2015). Recent estimates indicate that 37% of wader populations along the EAF have undergone a decline in recent decades (Delaney et al., 2013).

On an international scale, 44% of known wader populations are contracting (Wetlands International, 2010). According to Eaton et al., (2015) the number of UK wader species now classified as amber or red listed has increased over the last 30 years. Eight species are reliant on lowland wet grassland for breeding and all are recognized as being at varying levels of conservation concern (Wilson et al., 2005; Eaton et al., 2015). Black-Tailed Godwit *Limosa limosa* and Ruff *Philomachus pugnax* show very restricted breeding ranges within UK lowlands, often constrained to coastal grasslands, another declining habitat (Wilson et al., 2004). Other species such as Northern Lapwing (*Vanellus vanellus*), Eurasian Curlew (*Numenius arquata*) and Common Redshank (*Tringa totanus*) have demonstrated marked breeding population declines both within UK and Europe (*Sheldon et al., 2004; O'Brien and Wilson, 2011; Eaton et al., 2015). Changes to seasonal tilling* and sward height management along with grazing intensification and land drainage are thought to have been associated with breeding habitat loss (Sheldon *et al*., 2004; Wilson *et al*., 2004). Despite several attempts through numerous Agri-Environment Schemes (AES) to enhance lowland areas for breeding waders (e.g. Verhulst et al., 2007), declines continue (Wilson et al., 2004; Eglington et al., 2007; O'Brien and Wilson, 2011).

Aggregate extraction sites are typically located in flat, lowland valleys; areas that would have supported seasonal or permanent water bodies and wetlands (Andrews and Kinsman, 1990; Nicolet et al., 2004; Poschlod et al., 2005). Once sites have reached the end of their extraction life span, they have the potential to be restored with the end result supporting a comparatively elevated biodiversity to that of the active extraction site (Milne, 1974; Bell et al., 1997; Bradshaw, 1997; Santoul et al., 2004; Whitehouse, 2008). Post-extraction restoration guidelines tend to focus solely on the creation of lakes and the rapid establishment of reed beds (Ailstock et al., 2001; Jarvis and Walton, 2010). For example, open water areas in restored quarries benefit wintering and breeding waterfowl whilst reed beds provide breeding areas for species of conservation concern such as Bitterns (*Botaurus stellaris*) (Blaen *et al*., 2015), Bearded tit (*Panurus biarmicus*), Reed bunting (*Emberiza schoeniclus*) and Reed warbler (*Acrocephalus scirpaceus*) (Andrews and Kinsman, 1990; Peach et al., 1999; Poulin et al., 2002). However, reed beds provide little foraging or nesting opportunities for waders, who prefer open areas for foraging and shorter sward open grassland habitats for breeding (Cramp and Simmons, 1983; Milsom et al., 1998).

There is limited evidence that active aggregate sites can provide opportunities for waders, with some species nesting in gravel scrapes (e.g. Little Ringed Plover, *Charadrius dubius*) (Catchpole and Tydeman, 1975; Parrinder, 1989). Given the ubiquity of such settlement areas in quarry operations, there is the potential for them to contribute to regional, national and international wader conservation goals by replacing lost lowland wet areas both during operation and after post-extraction restoration. Little is known about wader use of active silting areas and management strategies for such areas aimed at wader conservation are not well-developed (Andrews and Kinsman, 1990). We wanted to assess the potential value of active silt lagoons in lowland areas for wader conservation. By integrating environmental and biological data from three active silt lagoon sites of different ages and two restored sites we aimed to (1) characterize the physico-chemical nature of these areas, (2) determine the important environmental factors influencing aquatic invertebrate diversity and abundance, (3) assess how wader richness and abundance varied between sites and 4) how waders actively used these areas. We hoped to use this information to provide recommendations on the management of silt lagoons to improve their potential as sites for wader conservation.

2. Methods

2.1 Study Sites and Location

Five sites were selected representing a range of conditions from highly disturbed and dynamic (ongoing deposition at active extraction sites) to minimal disturbance (well-established restored nature reserve sites). All sites were in lowland locations (< 60 m above sea level) within the same broad geographic region: North and East Yorkshire, UK (Figure 1). The active quarry sites were selected based upon safety considerations and access permissions and the general details of each site provided in Table 1. The two restored sites are man-made nature reserves managed for breeding waders and wildfowl by the Yorkshire Wildlife Trust; North Cave represented a restored aggregate extraction site and Filey Dams, a marshland near the coast, was included as an example of a mature, well-established site for comparison (Figure 1). All sites had a mixture of terrestrial and aquatic areas with exposed mineral substrate (ranging from extremely coarse to fine sediment), an open aspect, natural terrestrial vegetation and shallow and deep lentic waters.

2.2 Environmental parameters and invertebrate diversity

Major labile physico-chemical water parameters (pH, electrical conductivity (COND), oxygen reducing potential (ORP), dissolved oxygen (DO) and temperature (TEMP)) were obtained across all sites. Surface waters were sampled with a Myron Ultrameter (for pH, COND, ORP, TEMP) and an YSI550 Dissolved Oxygen meter for measuring DO. Sample alkalinity (ALK) was assessed in the field via titration against 1.6 N H_2SO_4 with bromocresol green-methyl red indicators (to pH 4.6) using a Hach Digital Titrator. On each sampling occasion, this was repeated three times across each site.

During autumn 2015, three 250 cm^3 core sediment samples were collected from the bottom of the lake at a depth of 20 - 25cm at each site to characterize the sediment profile. Particle size distribution was obtained by oven drying samples at 105^{o}C, and then fractions separated through a standard nest of sieves (2, 1 mm; 500, 250, 125, 90, 63 and 38 µm) and the percentage of each fraction calculated (Gee and Or, 2002). Sediment fractions incorporated into the analysis included gravels (G; < 2mm), very coarse sand (VCS; 1 – 2mm), coarse sand (CS; 500µm – 1mm), medium sands (MS; 250 - 500µm), fine sands (FS; 125 - 200µm), very fine sands (63 - 125µm) and coarse silts (38 - 65µm). The median particle size (D50) was used to summarise sediment size, and sediment profiles determined using Gradistat software (version 4.0) (Rice and Haschenburger, 2004). Organic content (LOI) of the substrate was obtained through loss on ignition at 550^{o}C until constant weight was achieved (Generowicz and Olek, 2010).

To describe the food available to foraging waders (Warrington et al., 2014), freshwater invertebrates were collected on a monthly basis between June and September

2015. The same worker sampled all sites using a kick sampling method, walking backwards at a constant steady pace for 30 seconds to dislodge invertebrates from the substrate into a D-frame pond net (0.25 mm mesh, 350 m x 180 mm frame) at a depth of 15 - 20 cm. Three replicate samples were collected on each sampling occasion and water depth was restricted to < 20 cm (García-Criado and Trigal, 2005) ensuring only invertebrates accessible to waders were collected. Samples were returned to the laboratory and invertebrates identified to Order / Class (Pawley *et al*., 2011) and the number of individuals in each taxon recorded for each replicate.

2.3 Wader site use

Bird surveys were conducted on a monthly basis at each site between (August 2014 and September 2015) to record changes in wader diversity and abundance over time. Scan sampling was undertaken from a fixed point at each site, with a sampling unit comprising of four replicate scan samples conducted every 15 minutes over a 1 hour period (Altmann, 1974; Cresswell, 1994). During each scan, the total number of individuals of each species was recorded along with their observed behaviour (e.g. foraging (feeding on the substrate), roosting (sleeping or preening) or other - used for all additional behaviours observed)) to reflect site use.

2.4 Statistical analysis

For all replicate sample from each site collected between June and September, invertebrate family richness (S), total invertebrate abundance (N) and the Shannon Wiener diversity index (H') were calculated using the PRIMER software package (Clarke and Warwick, 2001). The N, S and H' data conformed to a normal distribution (Kolmogorov Smirnov test, P> 0.05 in all cases) and variances could be considered equal (Levene's test, P>0.05 in all cases), therefore a general linear model ANOVA was used to determine if there was any significant difference in S, N and H’ between the factors Site (5 levels corresponding to survey sites) and Month (4 levels June – September inclusive). Post-hoc Tukey tests were used to determine the significance of any pairwise differences between the factors Site and Month (Sokal and Rohlf, 1995). A non-parametric Kruskal Wallis test was used to determine if there was a significant difference in the median values of each of the physico-chemical variables during the summer months between sites (Sokal and Rohlf, 1995).

A Bray Curtis similarity matrix was generated from square root transformed invertebrate abundance data. The non-parametric two-way ANOSIM (Analysis of Similarities) random permutation test was applied to the similarity matrix to test the null hypothesis that there was no significant difference in invertebrate community similarity between sites (averaged across months) or between months (averages across sites) (Clarke, 1993). The SIMPER routine in PRIMER was then used to determine which taxa contributed the most to the average similarity within and between the five sites or months.

The response of the invertebrate taxa to the physico-chemical variables was modelled using Canonical Correspondence Analysis (CCA). As the invertebrate dataset contained some double zeros, the data was subjected to a Chord-Hellinger transform prior to analysis (Zuur *et al.*, 2009). Cleveland dotplots were used to check for outliers, and multi-panel scatterplots used to determine if there was any collinearity amongst the physico-chemical variables (Zuur *et al.*, 2009). Due to high levels of collinearity VFS, VCS, CSI were removed from the analysis and a reduced set of physico-chemical variables used in the ordination including pH, COND, ORP, DO, ALK, LOI, FS, MS and G. A forward selection procedure accompanied by permutation tests (9999 permutations) was used to determine the significant variables contributing to the ordination (Zuur *et al.*, 2009). A table-wide sequential Bonferroni test was then applied to the results of the permutation tests in order to reduce Type I errors. All analysis was undertaken in R using the Vegan package (Oksanen *et al*., 2016) and base package (R Core team, 2016).

A Chi-squared test for homogeneity was used to determine if the overall wader species richness observed across the year varied significantly between the sites. For each replicate scan sample, the percentage of waders exhibiting roosting or foraging behaviours was calculated and subjected to an arcsin transform applied. In order to determine of all sites were used as feeding areas over the full observation period, a Kruskal-Wallis analysis was applied to the arcsin transformed proportional foraging or roosting behaviour data to determine if this was significantly between the sites over the year (Sokal and Rolf, 1995).

3. Results

3.1 Environmental variables and invertebrate diversity

All sites surveyed had circum-neutral pH, with TEMP, DO and ORP showing little variation between sites over the summer period (Kruskal Wallis, $P>0.05$ in all cases; Table 2a). Little Catwick had a markedly higher median COND and lower ALK than samples from other sites (Kruskal Wallis, $P<0.05$ in both cases; Table 2a). This pattern was consistent across the sites throughout the year and the median (range) values for the year are provided in supplementary information S1.

Silt lagoon sites had lower amounts of organic matter contained within the sediments (see Table 2a for LOI values) and also lower median particle size (D50) than the restored sites (Table 2a). Overall, silt lagoon sediments consisted of well-sorted very fine sands whereas those from the restored sites consisted of poorly sorted very coarse sands. Two of the silt lagoon sites, Wykeham (Fig 2a) and Ripon (Fig.2b)) were dominated by fine and very fine sands (0.25-0.062mm). Sediments from the third silt lagoon site, Little Catwick, contained coarser materials, but were still dominated by fine sands (Fig. 2c). North Cave

(Fig. 2d) and Filey Dams (Fig. 2e) exhibited a far more even grain size distribution, characterized by a greater proportion of coarse sands (0.5-2mm).

Overall, there was a significant difference in invertebrate family richness between the sites (ANOVA, $F_{4, 43} = 21.7$, $P < 0.001$ Table 2b), but not between months nor any interaction between site and month (ANOVA, P>0.05). North Cave had significantly higher family richness than all other sites (Table 2b), and Filey Dams was significantly higher than the silt lagoon sites (Tukey P = 0.05; Table 2b for means). However, there was no significant difference in richness between the three silt lagoon sites (Table 2b). There was also a significant difference in total abundance between sites (ANOVA, $F_{3, 43} = 11.7$, $P < 0.001$ for means see Table 2b) but no significant difference between month not any interaction between site and month (ANOVA, P >0.05). Mean total invertebrate abundance was a factor of ten higher in the samples from North Cave and Filey Dams than that observed from the silt lagoon samples (Table 2b). There was no significant difference in Shannon Wiener H' between the sites, months nor any significant interaction (ANOVA, P > 0.05 in all cases; Table 2b). The monthly mean values for each site are provided in supplementary information 2, with just the site means presented in Table 2 for brevity.

Analysis of Similarities (ANOSIM) showed there was a significant difference in invertebrate community similarity between sites (averaged across months) (ANOSIM, Global R = 0.657, P = 0.1%), with the two restored sites significantly different from the silt lagoons (Pairwise comparisons, P < 0.2% in all cases), but the three silt lagoon sites were not significantly different from each other (Pairwise comparisons, P > 5% in all cases). There was also a significant difference in invertebrate similarity between months (averaged across all sites) (ANOSIM, Global R = 0.387, P < 0.1%) with the samples from June and July being significantly different to those from August and September (Pairwise comparisons, P < 0.3% in all cases). There was no significant difference between June and July samples nor between the August and September samples (Pairwise comparisons, P > 5% in all cases). The SIMPER routine in the PRIMER software package was used to determine the key invertebrate taxa defining the community similarity at each site. Whilst there were minor differences in the occurrence of taxa between sites (e.g. the lack of Cladocera from silt lagoons; Table 3) overall the relevant abundance of key taxa was key to determining the similarities between the sites rather than major changes in community composition (Table 3). Overall, the relative abundance of Chironomidae, Corixdae and Oligochaeta was key in determining much of the community similarity between sites, however these were far more abundant at North Cave and Filey Dams than the silt lagoons. The relative abundance of these taxa and Cladocera was the major factor contributing to differences between the early summer (June and July) compared to late summer samples (August and September) (full details in supplementary information S3).

Figure 3 shows the site conditional CCA triplot split into, a) the sites plotted at the centroids of the family scores and b) families plotted close to the sites where they occur. The nine environmental variables in the model accounted for 52% of the total inertia (2.83), with the first two canonical axes explaining 53% of the variation (Table 3). Overall, all nine variables were included in the model, with the proportions of each sediment size class and COND all highly significant after forward selection and sequential Bonferroni correction (Table 3). Sediment fractions were important in separating the silt lagoons from the restored sites along CCA1, with samples containing a higher proportions of fine sands (FS) on the right of the plot and coarser gravels (G) on the left (Fig. 3a). In addition, COND was a major factor separating Little Catwick (LC) from the remainder of the silt lagoons along CCA2, along with the relative proportion of medium sands (MS) (Figure 3a and Table 3). Corixidae and Chironomidae were aligned with the Wykeham (WK) and Ripon (RP) silt lagoon samples on the right hand side of the plot, and Cladocera, Siphlonuridae, Haliplidae and Sphaeriidae and with the restored sites on the left hand side (Figure 3b).

3.2 Wader site use

Waders were recorded at all sites, however the patterns of use were highly variable (Table 4). Over the entire year, there was a significant difference in the number of different wader species observed at each site (Chi-Squared test, $X^2 = 13.6$, df = 4, P = 0.008) with more wader species than expected at Filey Dams and North Cave and less than expected on the active silt lagoons (Table 4). The total wader abundance at each site was also highly variable over the months throughout the year (see Table 4), but overall, Ripon supported significantly higher wader abundance (mean = 32 (standard deviation (sd) = 32)) than all other sites (ANOVA, $F_{4, 33} = 4.1$, P = 0.008; Tukey = 0.05). No significant difference in abundance occurred between the other sites (Filey Dams mean = 11.8 (sd = 16.3); Wykeham mean = 3.4 (sd = 5.3); Little Catwick mean = 1 (sd = 2.7); North Cave mean = 13.3 (sd = 9.3)). The higher abundance at Ripon was largely due to large flocks of Lapwings (*Vanellus vanellus*) roosting on the site during late summer and autumn (Table 4). Re-analysis of the data after the removal of Lapwings from the data set revealed no significant difference in abundance between sites (ANOVA, P>0.05) indicating they were the major influential variable on abundance. Site water level management (See Table 1) was important, with low water levels at Filey Dams in late summer/autumn attracting migrants, but high levels during winter and spring restricting availability of feeding areas. Low water levels at North Cave during spring produced breeding habitat for Avocets (*Recurvirostra avosetta*) but as water levels increased in autumn few waders used the site (Table 4). The only other site where breeding was observed was Wykeham, where both Little Ringed Plover (*Charadrius dubius*) and Oystercatcher (*Haematopus ostralegus*) bred in the area (Table 4).

Overall, there was also a significant difference in the median percentage of waders at a site exhibiting feeding behaviour (Kruskal Wallis, H (adjusted for ties) = 9.0, df = 3, P = 0.029) with a higher percentage of waders feeding at Filey Dams (median = 20% (range 0 - 100%)) than all other sites (North Cave median = 12.5%, Range 0 - 100%; Ripon and Wykeham medians = 0% (Range 0-100)). However, there was a significantly higher median percentage of waders exhibiting roosting behavior (Kruskal Wallis, H (adjusted for ties) = 9.4, df = 3, P = 0.024) at Ripon and Wykeham (medians = 100%, range 0 - 100% in both cases) than seen at Filey Dams (median = 66% (range 0 - 100%) or at North Cave (median = 87% (range 0 - 100%).

4. Discussion

There was very little difference in the water quality between sites, with all sites characterized by circum-neutral pH, mineral-rich and well-oxygenated waters typical of many UK lowland settings (Shand *et al.*, 2007). However, silt lagoons had well-sorted fine sands with little organic content (Fig.2), whereas sediment from restored sites contained medium-sorted coarse sands and higher organic content (LOI) (Table 2a). The homogeneity of silt lagoon particle size may have limited the abundance of aquatic invertebrates (Fig. 3 and Table 3), with abundance at restored sites a factor of ten higher than that found on the silt lagoons (Table 2b). Waders were recorded using all sites (Table 4), albeit in both lower richness on the silt lagoons than that observed on the restored sites. Both Little Ringed Plover and Oystercatcher bred on one of the silt lagoons. Patterns of wader abundance and richness varied between the restored sites, primarily as the result of different water management practices and the movement of large Lapwing flocks in and out of the area (Tables 3 and 4).

Silt lagoons are typically associated with low gradient banks of fine water-logged sediments that typically remain in a semi-liquid state (Jarvis and Walton, 2010). The median particle grain size at Wykeham and Ripon was < 158 µm reflecting the fine screening used to remove the commercially-viable products (Jarvis and Walton, 2010; Fig. 2; Table 2a), whereas the Little Catwick sediments contained coarser material, possibly as a result of slippage from the steep banks surrounding the lagoon. On all silt lagoons, field observations showed a sub-surface anoxic hard-pan layer formed in shallow water at the lake edges, typically within < 20mm of the sediment surface. The compacted sediment may restrict burrowing by lentic freshwater invertebrates (Jones *et al*., 2012) or restrict oxygen diffusion into deeper layers with the constant fine sediment deposition blocking interstitial spaces. In lotic systems, compact fine muds and sands are not readily colonized by invertebrates due to the instability of the substrate (Xuehua *et al*., 2009) however in these lentic systems the constant deposition of fine sediments may reduce invertebrate taxon diversity and abundance due to burial, abrasion and clogging of respiratory surfaces (Jones *et al.*, 2012).

The silt lagoon sediments contained very little organic material (Table 2a). Organic detritus is a key component driving benthic invertebrate diversity (Rehfisch, 1994; Chamier, 1997; Zilli *et al.,* 2008). Such material originates from both allochthonous and autochthonous sources (Zimmer *et al.,* 2000; Batzer and Sharitz, 2014; Hill *et al*., 2015), however silt lagoons had no aquatic plants and little organic input from the edge of the lagoons. At both Filey Dams and North Cave, there was obvious organic input from the emergent vegetation in the vicinity and algae were also present (Table 1) and it is well-known that submerged vegetation increases the abundance and diversity of grazing invertebrates and detritivores (Zimmer *et al.*, 2000; Batzer and Sharitz, 2014; Hill *et al.,* 2015). Grazers and detritivores (e.g. Lymnaea, Physidae, Asellidae) were found at Filey Dams and North Cave but absent from silt lagoon samples. Aiding vegetation colonization of open silting areas may help increase invertebrate abundance and provide organic nutrient pools for future succession (Mayes *et al.*, 2005).

Both invertebrate species richness and abundance were significantly higher in restored sites than silt lagoons (Table 2b). Increased invertebrate richness and abundance is typically associated with diverse sediment particle size and spatially complex substrates (Flecker and Alan, 1984; Jowett et al., 1991), both within lentic (Sanders and Maloney, 1994; Sanders, 2000) and lotic systems (Flecker and Allan 1984). Whilst there was no significant difference in diversity between sites (Table 2b), some taxa occurred on restored sites but were absent from silt lagoons, e.g. the filter feeders Cladocera and Sphaeridae (Table 2c). Total abundance was a factor of ten higher in restored site samples, despite little variation in composition of the dominant taxa between all sites (Table 2c). The coarser sediments coupled with higher organic content may have contributed to the markedly higher invertebrate abundance. In pond restoration trials, addition of coarser sediments into ponds with a silt-dominated substrates increased invertebrate biomass (Sanders and Maloney, 1994). Larger sediment grain size provides shelter, increases microhabitat diversity and amount of organic entrapment compared to more uniform sands and silts (Flecker and Allan 1984).

Silt lagoons evidently can and do support wading birds, but both richness and abundance were lower here than on restored nature reserve sites (Table 4). However, patterns of wader abundance and richness also varied between the restored sites, primarily as the result of different water management practices. The highest wader richness was observed at Filey Dams during late summer (Table 4) when water levels were lowered to produce shallow, muddy areas for foraging migrants in autumn. Some wader species are known to exhibit high site fidelity, returning to the same areas to feed year upon year during migration (Catry *et al.*, 2004) so stop-over sites need to hold predictable and accessible food supplies (Warnock, 2010). Filey Dams has an open aspect, high invertebrate abundance and easy access to prey; three factors closely linked to stop-over site selection in many species of wading birds (Finn *et al.*, 2008; Estrella and Masero, 2010). In contrast, whilst the shallow-water open areas of active silt lagoons were readily accessible, prey availability was markedly lower. In addition, high levels of disturbance also limit wader abundance and diversity (Milsom *et al*., 1998). Whilst quarrying activity continued on all silt lagoon sites, periodic disturbance due to land re-profiling occurred close to the silt lagoon at Little Catwick which may have resulted in the lowest wader abundance (Table 4).

A different management strategy was applied at North Cave; water levels were lowered in spring to create breeding habitat, primarily for Avocets observed on site until late summer (Table 4). Avocets are non-selective feeders, sweeping the bill through the water or along the sediment surface (Dias *et al.*, 2009). The high abundance of invertebrates at this site, especially Cladocera (Table 2c), provided food sources for breeding pairs. Little Ringed Plovers bred on one of the active silting sites, and have been previously recorded nesting in open gravel areas on quarry sites (Wiersma *et al*., 2016). They forage primarily on terrestrial insects rather than aquatic invertebrates, therefore low aquatic invertebrate abundance would not affect breeding success (Cramp and Simmons, 1983). Oystercatchers also bred within the vicinity, but predominantly foraged in open grassland areas for soil invertebrates (Hulscher, 1996; Furnell and Hull, 2014). Whilst probing attempts were made in the silt lagoon sediments, no successful prey capture was observed (Table 4).

To avoid predation, most waders maintain a constant level of vigilance (Fuller, 2012; Beauchamp, 2015) irrespective of foraging techniques (Barbosa, 1995). Site topography and vegetation height restrict the effectiveness of vigilance behaviour (Metcalfe, 1984). Any obstructions to the birds field of view usually results in increased vigilance and reduced feeding rates (Metcalfe, 1984; Beauchamp, 2015). Silt lagoon areas typically possess shallow sloping banks with little significant debris or dense vegetation, and two out of three active sites surveyed here had a wide open aspect. These open sites were commonly used as roosting areas for geese and less frequently Lapwings outside the breeding season (Table 4). Lapwings in particular, were only seen occasionally and probably roosted across a range of sites within the landscape rather than favouring particular roosting areas. Whilst silt lagoons possessed landscape features associated with roosting and foraging sites, there was probably insufficient invertebrates to support significant numbers of foraging waders (Saffran *et al.*, 1997) and, whilst our study considers a limited number of active sites, the same pattern of paucity in the invertebrate assemblages was evident.

5. Conclusions and management considerations

Whilst the number of sites studied here was limited, it was evident that silt lagoons supported lower wading bird richness and abundance than restored sites in the same physiographic setting. The open aspect, shallow silt lagoon edges could provide important additional lowland habitat for wading birds (Catchpole and Tydeman, 1975; Murray et al., 2013) if managed for that purpose and water levels could be regulated. Open areas could be incorporated into future aggregate site restoration plans and the natural succession encouraged (Prach et al., 2001; Řehounková and Prach, 2008; Joyce, 2014) but managed to ensure they retained an open aspect. Simple interventions, such as depositing over-burden (cobbles and gravels) at the lake edge would increase substrate heterogeneity, trap organics and provide substrate for aquatic macrophytes (Sanders and Maloney, 1994; Sanders, 2000) in turn, increasing overall aquatic invertebrate abundance (Sanders, 2000; Lods-Crozet and Castella, 2009). With many active lowland quarries lying within reach of internationally-important migratory pathways across Europe, such low-cost restoration efforts may play an important role in future landscape-scale wader conservation efforts and replace some of the lost lowland wet grasslands.

Acknowledgements.

The authors would like to thank both Birdlife International, Heidelberg Cement and Hanson UK for providing funding for the project. We are particularly grateful to Martin Crowe and Tim Harvey (Hanson Quarries) for facilitating site access and would like to thank the reviewers for their constructive comments on a previous version of this ms. Thanks for Bryonie Fox and Carl Thomas for assistance with fieldwork. The views represented in this paper are those of the authors and do not necessarily reflect those of the funding bodies.

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757 Table.1 Location and description of the five sampling sites used in the current study.

	Location (Lat/Long)	Site Age (yrs)	Area (ha)	Site perimeter (m)	Brief description
Filey Dams (Yorkshire Wildlife Trust)	North Yorkshire, (54° 12'39.56 N, 000°18'14.68" W)	35	2.53	1062	*Phragmites* and *Juncus* marsh with open water. Managed for breeding wildfowl and autumn migrants. Water levels dropped in late summer; kept high for breeding wildfowl during spring.
North Cave (Yorkshire Wildlife Trust)	East Yorkshire, (53°47'12.39 N, 000°39'51.43" W)	15 – 20	2.44	789.7	A restored aggregate lagoon in a mosaic of habitats, bordered by active aggregate quarry. Low water levels in spring for breeding waders, high water levels during late autumn/winter.
Wykeham	North Yorkshire, (54°13'27.33 N, 000°29'26.44" W)	0-5	1.35	538.8	Active quarry silt lagoon discharging sediment into an extraction pit. Large flat silt bank boarded by farmland. No active water management.
Ripon	North Yorkshire, (54°11'35.89 N, 001°32' 41.18" W)	5-10	2.41	629.7	Active steep sided silt lagoon, with a single open silt bank on southern fringe. Sediment input varies depending on extraction process; no active water management.
Little Catwick	East Yorkshire, (53°53' 01.42 N, 000°17'56.74" W)	5-10	0.22	185.9	Active small silt lagoon with steep sides and a single flat silt bank. Input of sediment varied with extraction process. Noticeable active quarry works adjacent to area. No active water management.

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Table 2. Summary of a) median (range) of environmental parameters, b) mean (sd) invertebrate richness (S), diversity (Shannon Wiener H') and median (range) total abundance (N) at each site during the summer months. In addition, c) the key invertebrate taxa from SIMPER analysis contributing to the differences between the sites over the summer months. Numbers denote the mean (sd) abundance of taxa / those in **bold** denote % contribution of that taxon to the average similarity within a site.

	Filey Dams (FD)	North Cave (NC)	Wykeham (WK)	Ripon (RP)	Little Catwick (LC)
a) Environmental parameters					
pH	8.13 (7.81 - 8.60)	7.84 (7.4 – 7.90)	7.59 (6.40 – 8.03)	8.16 (8.10 - 8.21)	8.06 (8.03 – 8.09)
ORP (mV)	191 (135 – 206)	155 (115 – 184)	221 (184 – 252)	232 (197 – 268)	226 (211 – 241)
COND (µS/cm^2)	711 (526 – 884)	804 (754 – 856)	860 (856 – 900)	610 (604 – 616)	1121 (1028 – 1214)
ALK (mg/L as $CaCO_3$)	251 (220 – 280)	222 (218 – 225)	174 (167 – 177)	158 (146 – 169)	97 (83 – 110)
DO (% saturation)	45 (32.7 – 69.1)	75.7 (56.9 – 79.1)	82.6 (76.2 – 91.4)	94.3 (90.1 – 98.5)	62.6 (27.5 – 97.8)
TEMP (^{o}C)	16.9 (12.3 – 21.2)	18.7 (17.5 – 19)	15.3 (13.4 – 18.7)	15.1 (12.5 – 17.8)	15.2 (14.1 – 16.3)
LOI (%)	11.3 (5.2 – 17.9)	4.4 (3.1 – 5.9)	1.9 (0.9 – 3.7)	2.1 (1.1 – 3.6)	1.8 (1.0 – 3.0)
D50	832 (607 – 907)	885 (683 – 1579)	132 (113 – 158)	109 (89 – 132)	228 (227 – 511)
b) Invertebrate diversity					
S	7.7 (3.4)	12.8 (2.0)	3.4 (2.0)	3.0 (2.3)	3.5 (2.6)
N	436 (494)	927 (555)	15 (17.8)	14 (18.3)	8 (5.6)
Shannon Wiener H'	1.13 (0.45)	1.34 (0.41)	0.85 (0.48)	0.81 (0.67)	1.00 (0.70)
c) Results from SIMPER					
Average similarity within site (%)	21.4	48.3	19.2	20.4	27.8
Coroxidae	45.4 (62.3) / **14%**	91.1 (83) / **10%**	3.1 (3.3) / **54%**	1.7 (1.2) / **15%**	0.3 (0.8) / -
Oligochaeta	32.4 (25.7) / **26%**	178.4 (99.6) / **31%**	4.3 (9.8) / **22%**	3.0 (4.2) / **28%**	0.7 (0.8) / **26%**
Chironomidae	83.8 (100.0) / **34%**	55.1 (46.9) / **8%**	4.8 (10.3) / **13%**	7.0 (8.8) / **48%**	1.0 (2.0) / **35%**
Cladocera	36 (36) / **12%**	393 (266) / **44%**	0 / -	0 / **-**	0 / **-**

Ceratopogonidae	7.5 (15.8) / **-**	21.2 (27.9) / -	0.6 (1.2) / -	1.8 (3.3) / -	2.7 (2.7) / **63%**
Sphaeridae	21.4 (24.3) / **8%**	2.2 (3.9) / -	0.1 (0.3) / -	0 / -	0 / -

Table 3. Summary of the results from the CCA analysis showing the contribution of the constrained environmental variables to the first two canonical coefficients and the F and P values from permutation tests. Use of *denotes significant P values after table-wide sequential Bonferroni correction.

	CCA Axis 1	CCA Axis 2	F	P value
Eigenvalues	0.422	0.353	-	-
% Variation explained by constrained variables	28.6%	23.9%	-	-
Constrained variable scores				
DO	-0.094	0.089	2.18	0.014*
COND	-0.215	-0.658	3.53	0.001*
ORP	-0.562	-0.096	2.74	0.002*
ALK	0.798	0.332	4.88	0.001*
G	0.846	-0.196	4.96	0.001*
MS	-0.094	-0.789	3.68	0.001*
FS	-0.432	0.449	2.87	0.001*
pH	-0.224	-0.302	2.33	0.015*
LOI	0.501	0.082	2.56	0.007*

Table 4. Maximum abundance of each wader species observed per hour at each site (winter months excluded due to frozen water bodies).

	August 2014	September 2014	October 2014	November 2014	April 2015	May 2015	June 2015	July 2015	August 2015
a) Filey Dams									
Lapwing (*Vanellus vanellus*)	2	43	6	2					21
Greenshank (*Tringa nebularia*)									3
Ruff (*Philomachus pugnax*)	4								2
Black-tailed Godwit (*Limosa limosa*)			1						1
Green Sandpiper (*Tringa ochropus*)	1							1	
Snipe (*Gallinago gallinago*)		1							
Ringed Plover (*Charadrius hiaticula*)	1	1							
Eurasian Curlew (*Numenius arquata*)		1	1						
Common Sandpiper (*Actitis hypoleucos*)	2		1						
Common Redshank (*Tringa tetanus*)	2		1						
Oystercatcher (*Haematopus ostralegus*)				3		1			
Dunlin (*Calidris alpina*)		2							
Wood Sandpiper (*Tringa glareola*)	1								
b) North Cave									
Lapwing (*Vanellus vanellus*)		13			3	3	3		15
Black-tailed Godwit (*Limosa limosa*)									2
Snipe (*Gallinago gallinago*)		1							
Eurasian Curlew (*Numenius arquata*)					1				
Oystercatcher (*Haematopus ostralegus*)						1			
Little Ringed Plover (*Charadrius dubius*)						1			
Avocet (*Recurvirostra avosetta*)						16	19		1
Wood Sandpiper (*Tringa glareola*)									1
c) Wykeham									
Little Ringed Plover (*Charadrius dubius*)	1	1			15	1	4	4	
Lapwing (*Vanellus vanellus*)									1
Oystercatcher (*Haematopus ostralegus*)					2				
Dunlin (*Calidris alpina*)									2
d) Ripon									
Lapwing (*Vanellus vanellus*)			49	34	3	1	1	86	46
Eurasian Curlew (*Numenius arquata*)								3	
Ringed Plover (*Charadrius hiaticula*)				1					
e) Little Catwick									
Common Sandpiper (*Actitis hypoleucos*)		1							
Ringed Plover (*Charadrius hiaticula*)		6							

Figure 1. The distribution of active aggregate extraction sites in the UK (left) and the study sites in northeast England (inset). Sample codes: WK=Wykeham, FD = Filey Dams, RP=Ripon, NC=North Cave and LC=Little Catwick. Elevation ranges from 0m above sea level (light gray) to 320m (dark gray) in inset.

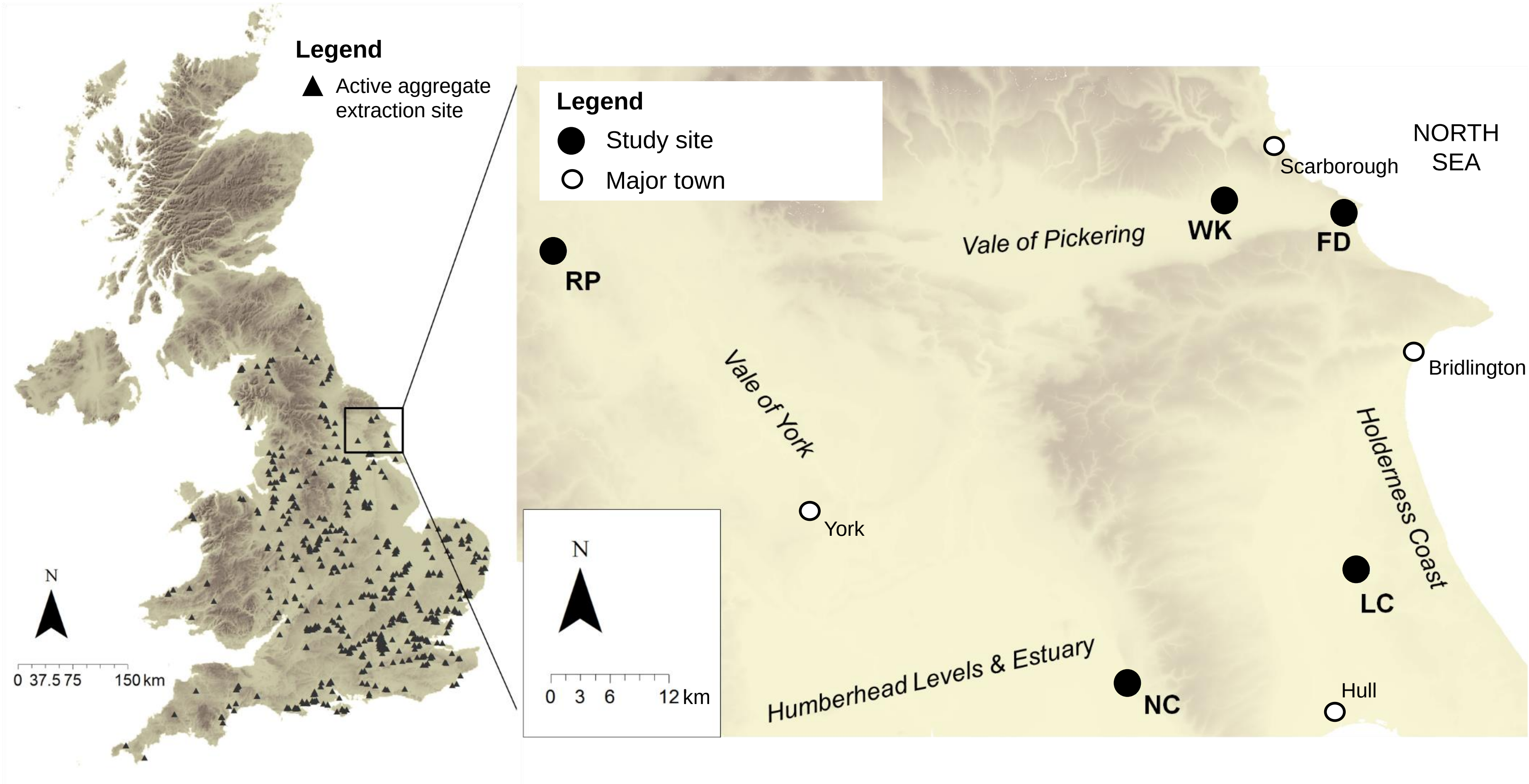


Figure 2. Cumulative frequency particle size distribution (%) for all sites. All active silt lagoons are highlighted with dashed lines, whilst solid black lines represent restored sites.

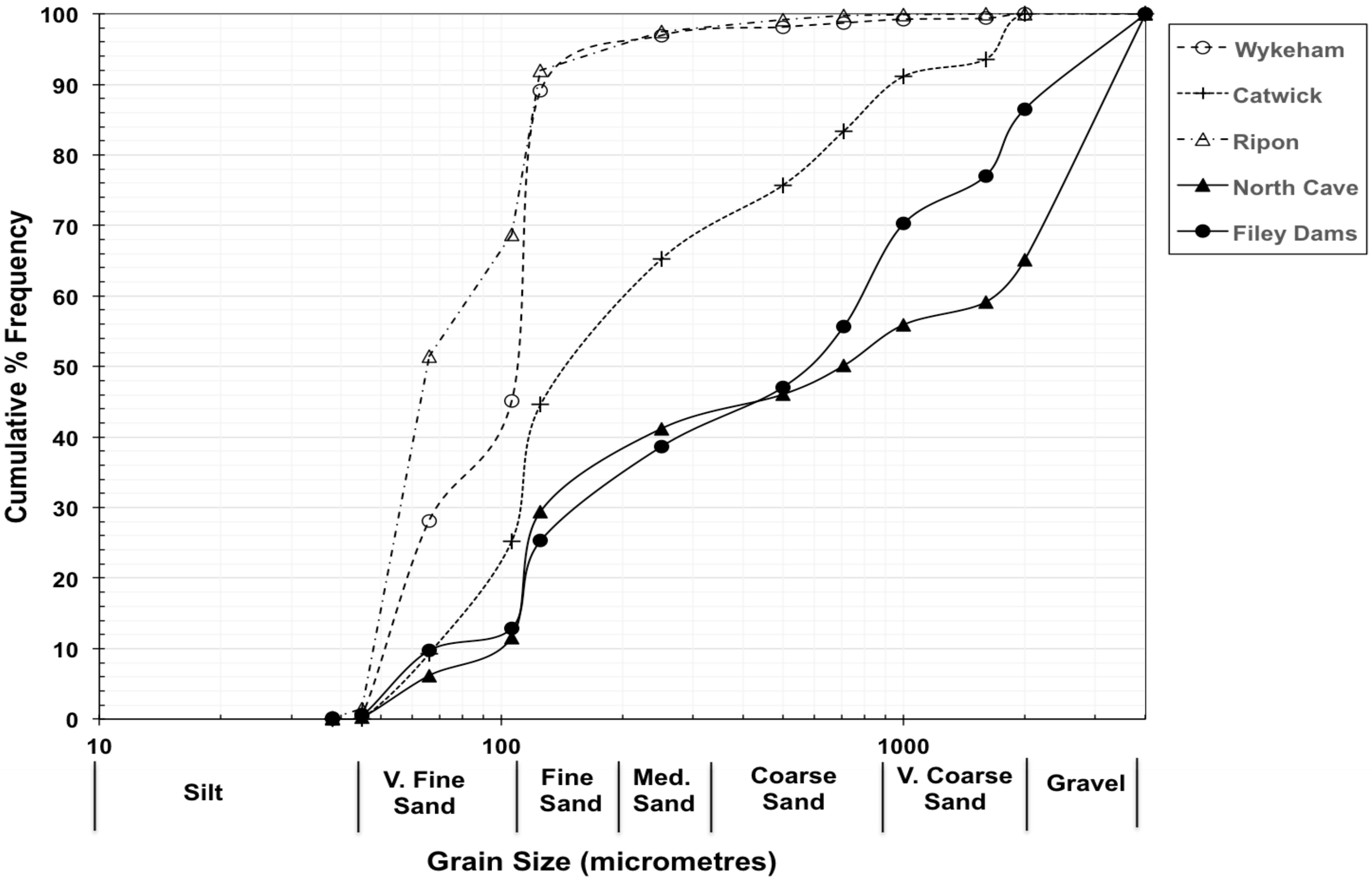
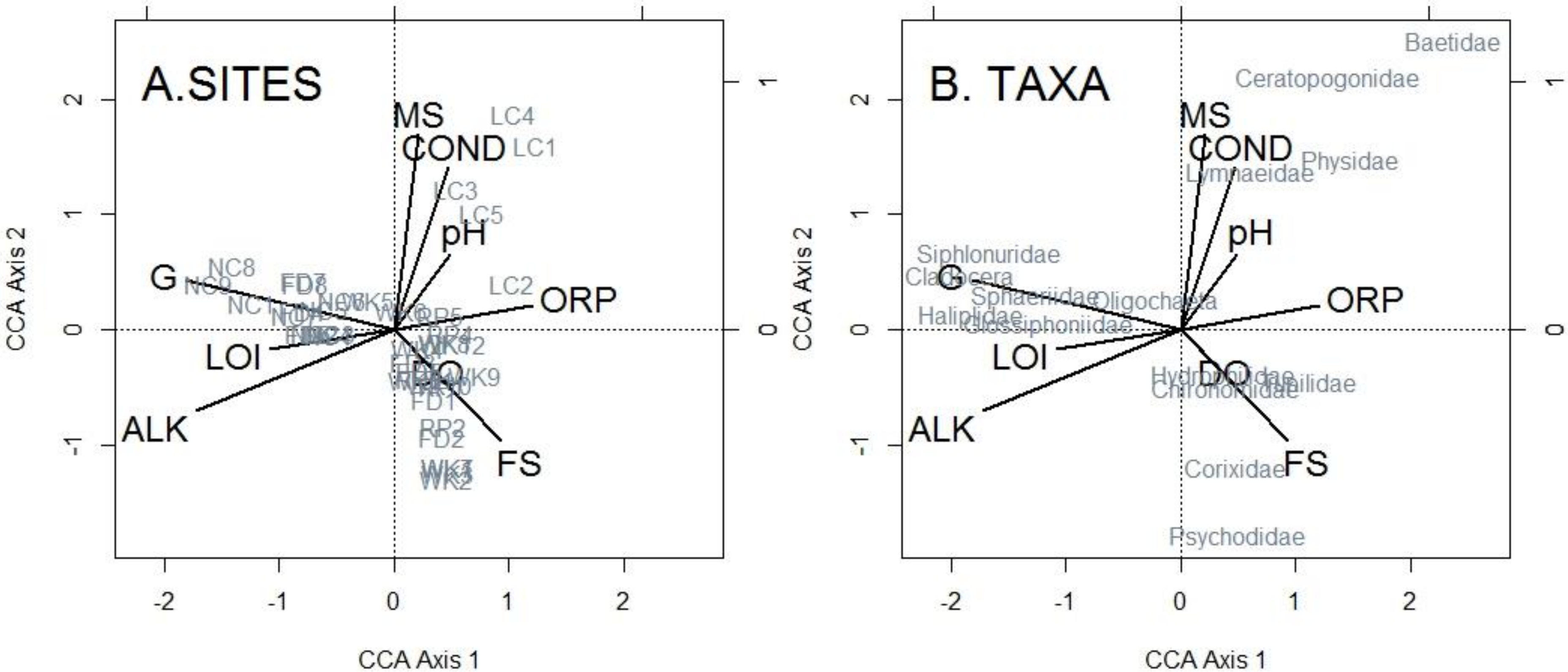


Figure 3. Site conditional CCA scaling plot for the samples from restored and active quarry sites, with a) Sites (FD = Filey Dams, NC = North Cave, WK=Wykeham, RP = Ripon, LC = Little Catwick) and b) invertebrate families plotted separately. Abbreviations in **black** on both plots correspond to the environmental variables defined in text.



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