

1 **Can aggregate quarry silt lagoons provide resources for wading birds?**

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16

17 **Abstract**

18 Wading birds have declined across Europe as the intensification of lowland agriculture has

19 resulted in the loss and degradation of wetland areas. Lowland aggregate extraction sites that

20 incorporate areas of fine, waste sediments deposited in silt lagoons have the potential to be

21 restored for wader conservation. We set out to determine the potential value of silt lagoons to

22 wading birds by comparing the water quality, sediment profiles, aquatic invertebrate

23 abundance and diversity (prey availability) and wader site use at five sites representing

24 various stages of active aggregate extraction and restoration for conservation purposes.

25 Wader counts were conducted monthly over a twelve month period using replicated scan

26 samples, and the invertebrate communities studied during the breeding and autumn migration  
27 season (June-September). Water quality variables were similar between sites, but sediments  
28 from active quarries were dominated by moderately sorted fine sands in comparison to the  
29 coarser sediment profiles of restored areas. June and September there was no significant  
30 difference in invertebrate diversity between sites, however richness was significantly lower  
31 and total abundance a factor of ten higher at restored sites than on silt lagoons, with the  
32 dominant taxa similar across all sites. Waders used all sites; albeit at lower abundance and  
33 richness on silt lagoons and two species were recorded breeding on active silting sites. We  
34 conclude that the fine, uniform sediments of modern silt lagoons limited invertebrate  
35 diversity and abundance, diminishing the value of silt lagoons to waders. Simple low-cost  
36 intervention measures increasing substrate heterogeneity and creating temporary ponds could  
37 increase invertebrate richness and abundance, and enhance the conservation potential of these  
38 sites.

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#### 40 **Keywords**

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

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#### 44 **1. Introduction**

45 Wetlands, a globally threatened habitat, are internationally important for water bird and  
46 wader communities (Grygoruk and Ignar, 2015; Kloskowski et al., 2009). Large-scale land  
47 use changes within the last century have led to significant declines in lowland perennial and  
48 seasonal wetlands within Europe (Gumiero et al., 2013; Verhoeven, 2014). Agricultural  
49 intensification leading to the abandonment of wet areas (Joyce, 2014), land reallocation or  
50 agricultural encroachment, over grazing, changes in water management or wetland drainage

51 have been held largely accountable for the loss of 50% of European wetlands within the last  
52 century (Silva et al., 2007; Henle et al., 2008). The internationally important European  
53 wetlands, salt marshes and mud flats support populations of over-wintering and migrating  
54 waterbirds and waders along the East Atlantic Flyway (EAF) (Rehfishch et al., 2003; Stroud et  
55 al., 2006; Holt et al., 2015). Recent estimates indicate that 37% of wader populations along  
56 the EAF have undergone a decline in recent decades (Delaney et al., 2013).

57         On an international scale, 44% of known wader populations are contracting (Wetlands  
58 International, 2010). According to Eaton et al., (2015) the number of UK wader species now  
59 classified as amber or red listed has increased over the last 30 years. Eight species are reliant  
60 on lowland wet grassland for breeding and all are recognized as being at varying levels of  
61 conservation concern (Wilson et al., 2005; Eaton et al., 2015). Black-Tailed Godwit *Limosa*  
62 *limosa* and Ruff *Philomachus pugnax* show very restricted breeding ranges within UK  
63 lowlands, often constrained to coastal grasslands, another declining habitat (Wilson et al.,  
64 2004). Other species such as Northern Lapwing (*Vanellus vanellus*), Eurasian Curlew  
65 (*Numenius arquata*) and Common Redshank (*Tringa totanus*) have demonstrated marked  
66 breeding population declines both within UK and Europe (Sheldon et al., 2004; O'Brien and  
67 Wilson, 2011; Eaton et al., 2015). Changes to seasonal tilling and sward height management  
68 along with grazing intensification and land drainage are thought to have been associated with  
69 breeding habitat loss (Sheldon et al., 2004; Wilson et al., 2004). Despite several attempts  
70 through numerous Agri-Environment Schemes (AES) to enhance lowland areas for breeding  
71 waders (e.g. Verhulst et al., 2007), declines continue (Wilson et al., 2004; Eglington et al.,  
72 2007; O'Brien and Wilson, 2011).

73         Aggregate extraction sites are typically located in flat, lowland valleys; areas that  
74 would have supported seasonal or permanent water bodies and wetlands (Andrews and  
75 Kinsman, 1990; Nicolet et al., 2004; Poschlod et al., 2005). Once sites have reached the end

76 of their extraction life span, they have the potential to be restored with the end result  
77 supporting a comparatively elevated biodiversity to that of the active extraction site (Milne,  
78 1974; Bell et al., 1997; Bradshaw, 1997; Santoul et al., 2004; Whitehouse, 2008). Post-  
79 extraction restoration guidelines tend to focus solely on the creation of lakes and the rapid  
80 establishment of reed beds (Ailstock et al., 2001; Jarvis and Walton, 2010). For example,  
81 open water areas in restored quarries benefit wintering and breeding waterfowl whilst reed  
82 beds provide breeding areas for species of conservation concern such as Bitterns (*Botaurus*  
83 *stellaris*) (Blaen et al., 2015), Bearded tit (*Panurus biarmicus*), Reed bunting (*Emberiza*  
84 *schoeniclus*) and Reed warbler (*Acrocephalus scirpaceus*) (Andrews and Kinsman, 1990;  
85 Peach et al., 1999; Poulin et al., 2002). However, reed beds provide little foraging or nesting  
86 opportunities for waders, who prefer open areas for foraging and shorter sward open  
87 grassland habitats for breeding (Cramp and Simmons, 1983; Milsom et al., 1998).

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

101 used these areas. We hoped to use this information to provide recommendations on the  
102 management of silt lagoons to improve their potential as sites for wader conservation.

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## 104 **2. Methods**

### 105 **2.1 Study Sites and Location**

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

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### 119 **2.2 Environmental parameters and invertebrate diversity**

120 Major labile physico-chemical water parameters (pH, electrical conductivity (COND),  
121 oxygen reducing potential (ORP), dissolved oxygen (DO) and temperature (TEMP)) were  
122 obtained across all sites. Surface waters were sampled with a Myron Ultrameter (for pH,  
123 COND, ORP, TEMP) and an YSI550 Dissolved Oxygen meter for measuring DO. Sample  
124 alkalinity (ALK) was assessed in the field via titration against 1.6 N H<sub>2</sub>SO<sub>4</sub> with bromocresol

125 green-methyl red indicators (to pH 4.6) using a Hach Digital Titrator. On each sampling  
126 occasion, this was repeated three times across each site.

127         During autumn 2015, three 250 cm<sup>3</sup> core sediment samples were collected from the  
128 bottom of the lake at a depth of 20 - 25cm at each site to characterize the sediment profile.  
129 Particle size distribution was obtained by oven drying samples at 105°C, and then fractions  
130 separated through a standard nest of sieves (2, 1 mm; 500, 250, 125, 90, 63 and 38 µm) and  
131 the percentage of each fraction calculated (Gee and Or, 2002). Sediment fractions  
132 incorporated into the analysis included gravels (G; < 2mm), very coarse sand (VCS; 1 –  
133 2mm), coarse sand (CS; 500µm – 1mm), medium sands (MS; 250 - 500µm), fine sands (FS;  
134 125 - 200µm), very fine sands (63 - 125µm) and coarse silts (38 - 65µm). The median  
135 particle size (D50) was used to summarise sediment size, and sediment profiles determined  
136 using Gradistat software (version 4.0) (Rice and Haschenburger, 2004). Organic content  
137 (LOI) of the substrate was obtained through loss on ignition at 550°C until constant weight  
138 was achieved (Generowicz and Olek, 2010).

139         To describe the food available to foraging waders (Warrington et al., 2014),  
140 freshwater invertebrates were collected on a monthly basis between June and September  
141 2015. The same worker sampled all sites using a kick sampling method, walking backwards  
142 at a constant steady pace for 30 seconds to dislodge invertebrates from the substrate into a D-  
143 frame pond net (0.25 mm mesh, 350 m x 180 mm frame) at a depth of 15 - 20 cm. Three  
144 replicate samples were collected on each sampling occasion and water depth was restricted to  
145 < 20 cm (García-Criado and Trigo, 2005) ensuring only invertebrates accessible to waders  
146 were collected. Samples were returned to the laboratory and invertebrates identified to Order  
147 / Class (Pawley et al., 2011) and the number of individuals in each taxon recorded for each  
148 replicate.

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150 **2.3 Wader site use**

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

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160 **2.4 Statistical analysis**

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

173 A Bray Curtis similarity matrix was generated from square root transformed  
174 invertebrate abundance data. The non-parametric two-way ANOSIM (Analysis of

175 Similarities) random permutation test was applied to the similarity matrix to test the null  
176 hypothesis that there was no significant difference in invertebrate community similarity  
177 between sites (averaged across months) or between months (averages across sites) (Clarke,  
178 1993). The SIMPER routine in PRIMER was then used to determine which taxa contributed  
179 the most to the average similarity within and between the five sites or months.

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

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



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### 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

225 means see Table 2b) but no significant difference between month not any interaction between  
226 site and month (ANOVA,  $P > 0.05$ ). Mean total invertebrate abundance was a factor of ten  
227 higher in the samples from North Cave and Filey Dams than that observed from the silt  
228 lagoon samples (Table 2b). There was no significant difference in Shannon Wiener  $H'$   
229 between the sites, months nor any significant interaction (ANOVA,  $P > 0.05$  in all cases;  
230 Table 2b). The monthly mean values for each site are provided in supplementary information  
231 2, with just the site means presented in Table 2 for brevity.

232 Analysis of Similarities (ANOSIM) showed there was a significant difference in  
233 invertebrate community similarity between sites (averaged across months) (ANOSIM, Global  
234  $R = 0.657$ ,  $P = 0.1\%$ ), with the two restored sites significantly different from the silt lagoons  
235 (Pairwise comparisons,  $P < 0.2\%$  in all cases), but the three silt lagoon sites were not  
236 significantly different from each other (Pairwise comparisons,  $P > 5\%$  in all cases). There  
237 was also a significant difference in invertebrate similarity between months (averaged across  
238 all sites) (ANOSIM, Global  $R = 0.387$ ,  $P < 0.1\%$ ) with the samples from June and July being  
239 significantly different to those from August and September (Pairwise comparisons,  $P < 0.3\%$   
240 in all cases). There was no significant difference between June and July samples nor between  
241 the August and September samples (Pairwise comparisons,  $P > 5\%$  in all cases). The  
242 SIMPER routine in the PRIMER software package was used to determine the key  
243 invertebrate taxa defining the community similarity at each site. Whilst there were minor  
244 differences in the occurrence of taxa between sites (e.g. the lack of Cladocera from silt  
245 lagoons; Table 3) overall the relevant abundance of key taxa was key to determining the  
246 similarities between the sites rather than major changes in community composition (Table 3).  
247 Overall, the relative abundance of Chironomidae, Corixidae and Oligochaeta was key in  
248 determining much of the community similarity between sites, however these were far more  
249 abundant at North Cave and Filey Dams than the silt lagoons. The relative abundance of

250 these taxa and Cladocera was the major factor contributing to differences between the early  
251 summer (June and July) compared to late summer samples (August and September) (full  
252 details in supplementary information S3).

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

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### 268 **3.2 Wader site use**

269 Waders were recorded at all sites, however the patterns of use were highly variable  
270 (Table 4). Over the entire year, there was a significant difference in the number of different  
271 wader species observed at each site (Chi-Squared test,  $X^2 = 13.6$ ,  $df = 4$ ,  $P = 0.008$ ) with  
272 more wader species than expected at Filey Dams and North Cave and less than expected on  
273 the active silt lagoons (Table 4). The total wader abundance at each site was also highly  
274 variable over the months throughout the year (see Table 4), but overall, Ripon supported

275 significantly higher wader abundance (mean = 32 (standard deviation (sd) = 32)) than all  
276 other sites (ANOVA,  $F_{4,33} = 4.1$ ,  $P = 0.008$ ; Tukey = 0.05). No significant difference in  
277 abundance occurred between the other sites (Filey Dams mean = 11.8 (sd = 16.3); Wykeham  
278 mean = 3.4 (sd = 5.3); Little Catwick mean = 1 (sd = 2.7); North Cave mean = 13.3 (sd =  
279 9.3)). The higher abundance at Ripon was largely due to large flocks of Lapwings (*Vanellus*  
280 *vanellus*) roosting on the site during late summer and autumn (Table 4). Re-analysis of the  
281 data after the removal of Lapwings from the data set revealed no significant difference in  
282 abundance between sites (ANOVA,  $P > 0.05$ ) indicating they were the major influential  
283 variable on abundance. Site water level management (See Table 1) was important, with low  
284 water levels at Filey Dams in late summer/autumn attracting migrants, but high levels during  
285 winter and spring restricting availability of feeding areas. Low water levels at North Cave  
286 during spring produced breeding habitat for Avocets (*Recurvirostra avosetta*) but as water  
287 levels increased in autumn few waders used the site (Table 4). The only other site where  
288 breeding was observed was Wykeham, where both Little Ringed Plover (*Charadrius dubius*)  
289 and Oystercatcher (*Haematopus ostralegus*) bred in the area (Table 4).

290 Overall, there was also a significant difference in the median percentage of waders at  
291 a site exhibiting feeding behaviour (Kruskal Wallis,  $H$  (adjusted for ties) = 9.0,  $df = 3$ ,  $P =$   
292 0.029) with a higher percentage of waders feeding at Filey Dams (median = 20% (range 0 -  
293 100%)) than all other sites (North Cave median = 12.5%, Range 0 - 100%; Ripon and  
294 Wykeham medians = 0% (Range 0-100)). However, there was a significantly higher median  
295 percentage of waders exhibiting roosting behavior (Kruskal Wallis,  $H$  (adjusted for ties) =  
296 9.4,  $df = 3$ ,  $P = 0.024$ ) at Ripon and Wykeham (medians = 100%, range 0 - 100% in both  
297 cases) than seen at Filey Dams (median = 66% (range 0 - 100%) or at North Cave (median =  
298 87% (range 0 - 100%)).

299

#### 300 **4. Discussion**

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

314           Silt lagoons are typically associated with low gradient banks of fine water-logged  
315 sediments that typically remain in a semi-liquid state (Jarvis and Walton, 2010). The median  
316 particle grain size at Wykeham and Ripon was < 158  $\mu\text{m}$  reflecting the fine screening used to  
317 remove the commercially-viable products (Jarvis and Walton, 2010; Fig. 2; Table 2a),  
318 whereas the Little Catwick sediments contained coarser material, possibly as a result of  
319 slippage from the steep banks surrounding the lagoon. On all silt lagoons, field observations  
320 showed a sub-surface anoxic hard-pan layer formed in shallow water at the lake edges,  
321 typically within < 20mm of the sediment surface. The compacted sediment may restrict  
322 burrowing by lentic freshwater invertebrates (Jones *et al.*, 2012) or restrict oxygen diffusion  
323 into deeper layers with the constant fine sediment deposition blocking interstitial spaces. In  
324 lotic systems, compact fine muds and sands are not readily colonized by invertebrates due to

325 the instability of the substrate (Xuehua *et al.*, 2009) however in these lentic systems the  
326 constant deposition of fine sediments may reduce invertebrate taxon diversity and abundance  
327 due to burial, abrasion and clogging of respiratory surfaces (Jones *et al.*, 2012).

328         The silt lagoon sediments contained very little organic material (Table 2a). Organic  
329 detritus is a key component driving benthic invertebrate diversity (Rehfishch, 1994; Chamier,  
330 1997; Zilli *et al.*, 2008). Such material originates from both allochthonous and  
331 autochthonous sources (Zimmer *et al.*, 2000; Batzer and Sharitz, 2014; Hill *et al.*, 2015),  
332 however silt lagoons had no aquatic plants and little organic input from the edge of the  
333 lagoons. At both Filey Dams and North Cave, there was obvious organic input from the  
334 emergent vegetation in the vicinity and algae were also present (Table 1) and it is well-known  
335 that submerged vegetation increases the abundance and diversity of grazing invertebrates and  
336 detritivores (Zimmer *et al.*, 2000; Batzer and Sharitz, 2014; Hill *et al.*, 2015). Grazers and  
337 detritivores (e.g. Lymnaea, Physidae, Asellidae) were found at Filey Dams and North Cave  
338 but absent from silt lagoon samples. Aiding vegetation colonization of open silting areas  
339 may help increase invertebrate abundance and provide organic nutrient pools for future  
340 succession (Mayes *et al.*, 2005).

341         Both invertebrate species richness and abundance were significantly higher in restored  
342 sites than silt lagoons (Table 2b). Increased invertebrate richness and abundance is typically  
343 associated with diverse sediment particle size and spatially complex substrates (Flecker and  
344 Alan, 1984; Jowett *et al.*, 1991), both within lentic (Sanders and Maloney, 1994; Sanders,  
345 2000) and lotic systems (Flecker and Allan 1984). Whilst there was no significant difference  
346 in diversity between sites (Table 2b), some taxa occurred on restored sites but were absent  
347 from silt lagoons, e.g. the filter feeders Cladocera and Sphaeriidae (Table 2c). Total  
348 abundance was a factor of ten higher in restored site samples, despite little variation in  
349 composition of the dominant taxa between all sites (Table 2c). The coarser sediments

350 coupled with higher organic content may have contributed to the markedly higher  
351 invertebrate abundance. In pond restoration trials, addition of coarser sediments into ponds  
352 with a silt-dominated substrates increased invertebrate biomass (Sanders and Maloney, 1994).  
353 Larger sediment grain size provides shelter, increases microhabitat diversity and amount of  
354 organic entrapment compared to more uniform sands and silts (Flecker and Allan 1984).

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

371 A different management strategy was applied at North Cave; water levels were  
372 lowered in spring to create breeding habitat, primarily for Avocets observed on site until late  
373 summer (Table 4). Avocets are non-selective feeders, sweeping the bill through the water or  
374 along the sediment surface (Dias *et al.*, 2009). The high abundance of invertebrates at this

375 site, especially Cladocera (Table 2c), provided food sources for breeding pairs. Little Ringed  
376 Plovers bred on one of the active silting sites, and have been previously recorded nesting in  
377 open gravel areas on quarry sites (Wiersma *et al.*, 2016). They forage primarily on terrestrial  
378 insects rather than aquatic invertebrates, therefore low aquatic invertebrate abundance would  
379 not affect breeding success (Cramp and Simmons, 1983). Oystercatchers also bred within the  
380 vicinity, but predominantly foraged in open grassland areas for soil invertebrates (Hulscher,  
381 1996; Furnell and Hull, 2014). Whilst probing attempts were made in the silt lagoon  
382 sediments, no successful prey capture was observed (Table 4).

383 To avoid predation, most waders maintain a constant level of vigilance (Fuller, 2012;  
384 Beauchamp, 2015) irrespective of foraging techniques (Barbosa, 1995). Site topography and  
385 vegetation height restrict the effectiveness of vigilance behaviour (Metcalf, 1984). Any  
386 obstructions to the birds field of view usually results in increased vigilance and reduced  
387 feeding rates (Metcalf, 1984; Beauchamp, 2015). Silt lagoon areas typically possess shallow  
388 sloping banks with little significant debris or dense vegetation, and two out of three active  
389 sites surveyed here had a wide open aspect. These open sites were commonly used as  
390 roosting areas for geese and less frequently Lapwings outside the breeding season (Table 4).  
391 Lapwings in particular, were only seen occasionally and probably roosted across a range of  
392 sites within the landscape rather than favouring particular roosting areas. Whilst silt lagoons  
393 possessed landscape features associated with roosting and foraging sites, there was probably  
394 insufficient invertebrates to support significant numbers of foraging waders (Saffran *et al.*,  
395 1997) and, whilst our study considers a limited number of active sites, the same pattern of  
396 paucity in the invertebrate assemblages was evident.

397

## 398 **5. Conclusions and management considerations**



399           Whilst the number of sites studied here was limited, it was evident that silt lagoons  
400 supported lower wading bird richness and abundance than restored sites in the same  
401 physiographic setting. The open aspect, shallow silt lagoon edges could provide important  
402 additional lowland habitat for wading birds (Catchpole and Tydeman, 1975; Murray et al.,  
403 2013) if managed for that purpose and water levels could be regulated. Open areas could be  
404 incorporated into future aggregate site restoration plans and the natural succession  
405 encouraged (Prach et al., 2001; Řehouňková and Prach, 2008; Joyce, 2014) but managed to  
406 ensure they retained an open aspect. Simple interventions, such as depositing over-burden  
407 (cobbles and gravels) at the lake edge would increase substrate heterogeneity, trap organics  
408 and provide substrate for aquatic macrophytes (Sanders and Maloney, 1994; Sanders, 2000)  
409 in turn, increasing overall aquatic invertebrate abundance (Sanders, 2000; Lods-Crozet and  
410 Castella, 2009). With many active lowland quarries lying within reach of internationally-  
411 important migratory pathways across Europe, such low-cost restoration efforts may play an  
412 important role in future landscape-scale wader conservation efforts and replace some of the  
413 lost lowland wet grasslands.

414

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745

746 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	<i>Phragmites</i> and <i>Juncus</i> 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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748 Table 2. Summary of a) median (range) of environmental parameters, b) mean (sd) invertebrate richness (S), diversity (Shannon Wiener H') and median  
 749 (range) total abundance (N) at each site during the summer months. In addition, c) the key invertebrate taxa from SIMPER analysis contributing to the  
 750 differences between the sites over the summer months. Numbers denote the mean (sd) abundance of taxa / those in **bold** denote % contribution of that taxon  
 751 to the average similarity within a site.

	Filey Dams (FD)	North Cave (NC)	Wykeham (WK)	Ripon (RP)	Little Catwick (LC)
<i>a) Environmental parameters</i>					
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 ( $\mu\text{S}/\text{cm}^2$ )	711 (526 - 884)	804 (754 - 856)	860 (856 - 900)	610 (604 - 616)	1121 (1028 - 1214)
ALK (mg/L as $\text{CaCO}_3$ )	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 ( $^{\circ}\text{C}$ )	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)
<i>b) Invertebrate diversity</i>					
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)
<i>c) Results from SIMPER</i>					
Average similarity within site (%)	21.4	48.3	19.2	20.4	27.8
Coroxidae	45.4 (62.3) / <b>14%</b>	91.1 (83) / <b>10%</b>	3.1 (3.3) / <b>54%</b>	1.7 (1.2) / <b>15%</b>	0.3 (0.8) / -
Oligochaeta	32.4 (25.7) / <b>26%</b>	178.4 (99.6) / <b>31%</b>	4.3 (9.8) / <b>22%</b>	3.0 (4.2) / <b>28%</b>	0.7 (0.8) / <b>26%</b>
Chironomidae	83.8 (100.0) / <b>34%</b>	55.1 (46.9) / <b>8%</b>	4.8 (10.3) / <b>13%</b>	7.0 (8.8) / <b>48%</b>	1.0 (2.0) / <b>35%</b>
Cladocera	36 (36) / <b>12%</b>	393 (266) / <b>44%</b>	0 / -	0 / -	0 / -

	Ceratopogonidae	7.5 (15.8) / -	21.2 (27.9) / -	0.6 (1.2) / -	1.8 (3.3) / -	2.7 (2.7) / <b>63%</b>
752	Sphaeridae	21.4 (24.3) / <b>8%</b>	2.2 (3.9) / -	0.1 (0.3) / -	0 / -	0 / -
753						
754						

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

757

	CCA Axis 1	CCA Axis 2	F	P value
Eigenvalues	0.422	0.353	-	-
% Variation explained by constrained variables	28.6%	23.9%	-	-
<i>Constrained variable scores</i>				
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*

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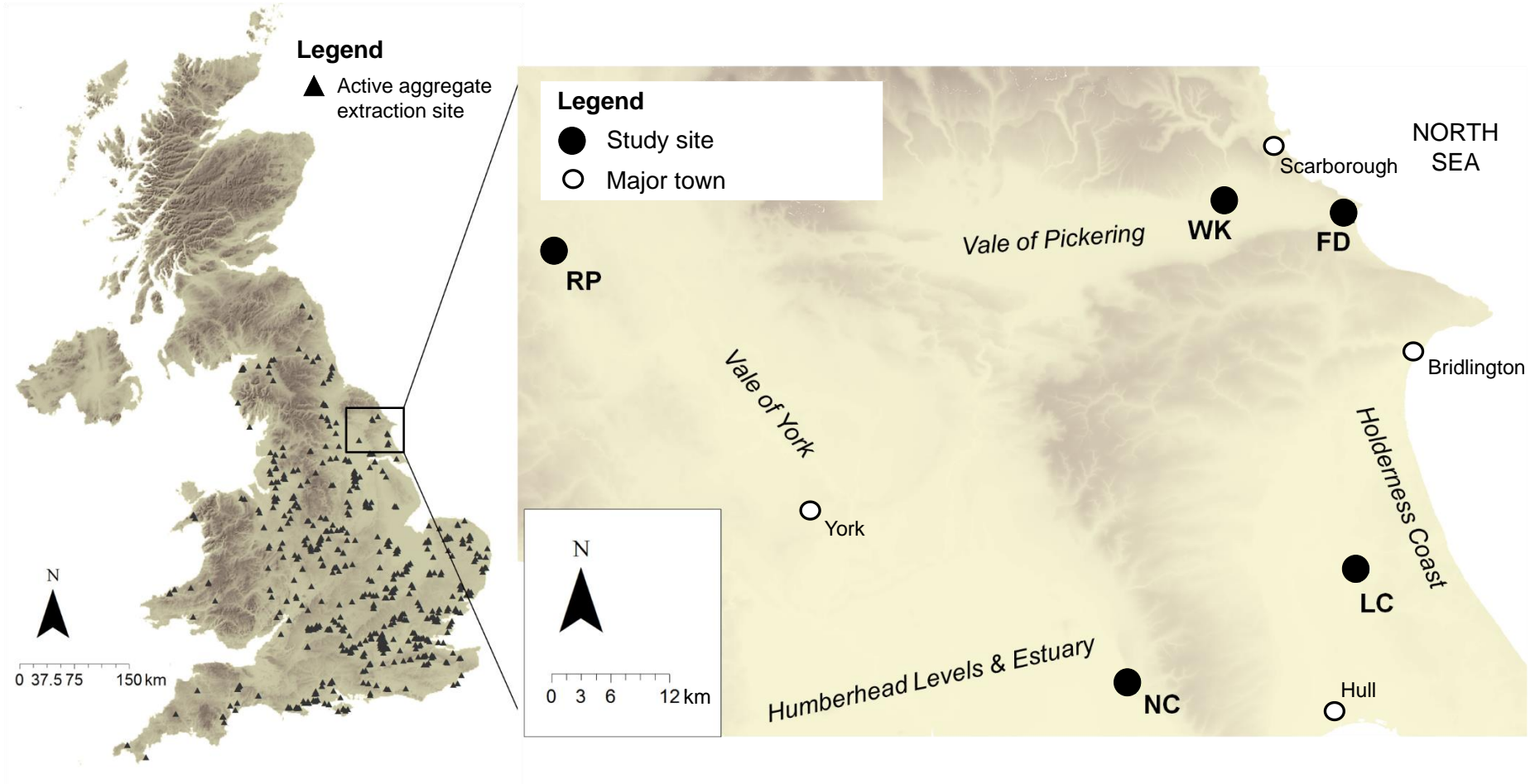
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762 Table 4. Maximum abundance of each wader species observed per hour at each site (winter months excluded due to frozen water bodies).

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

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

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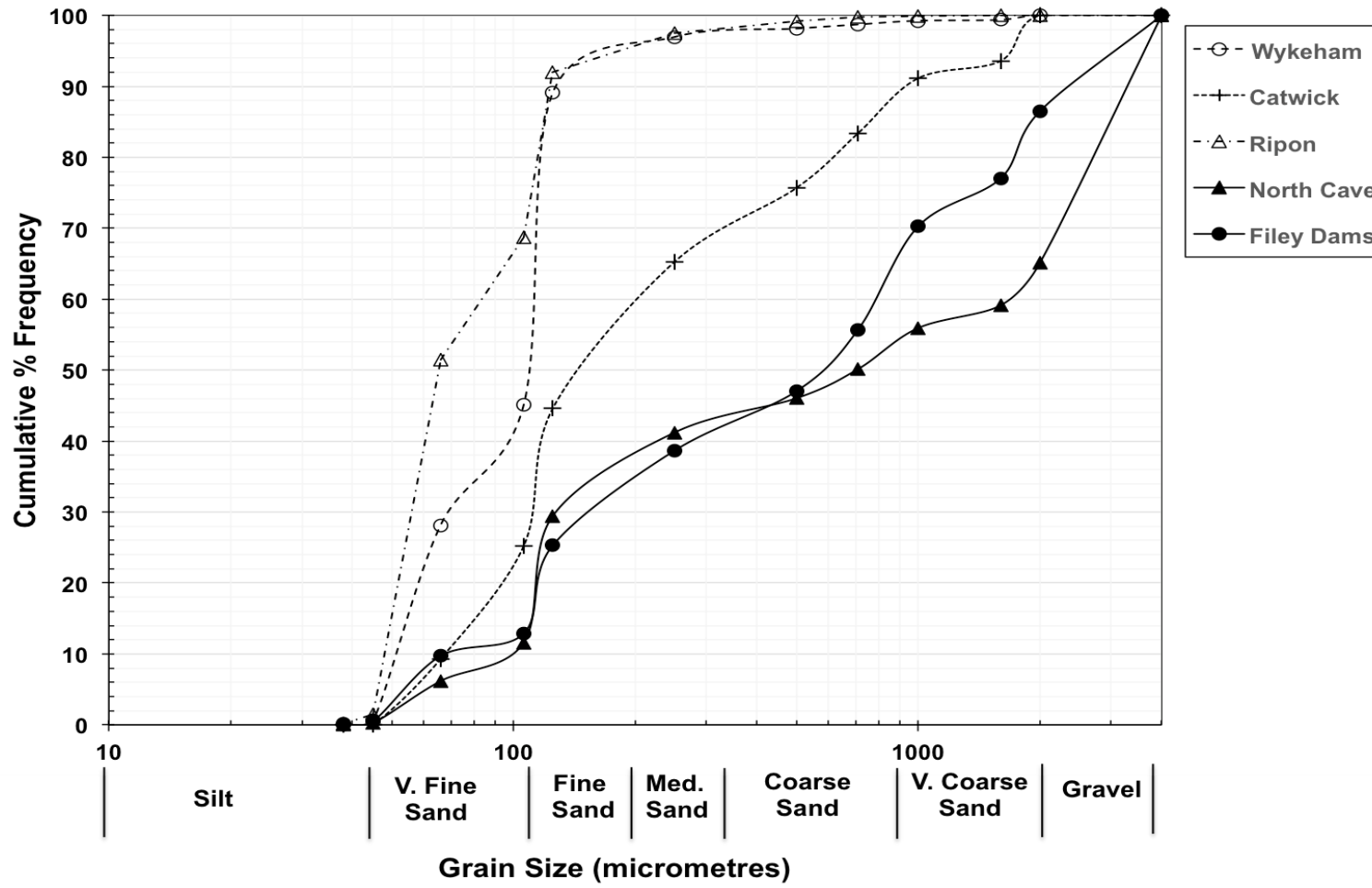


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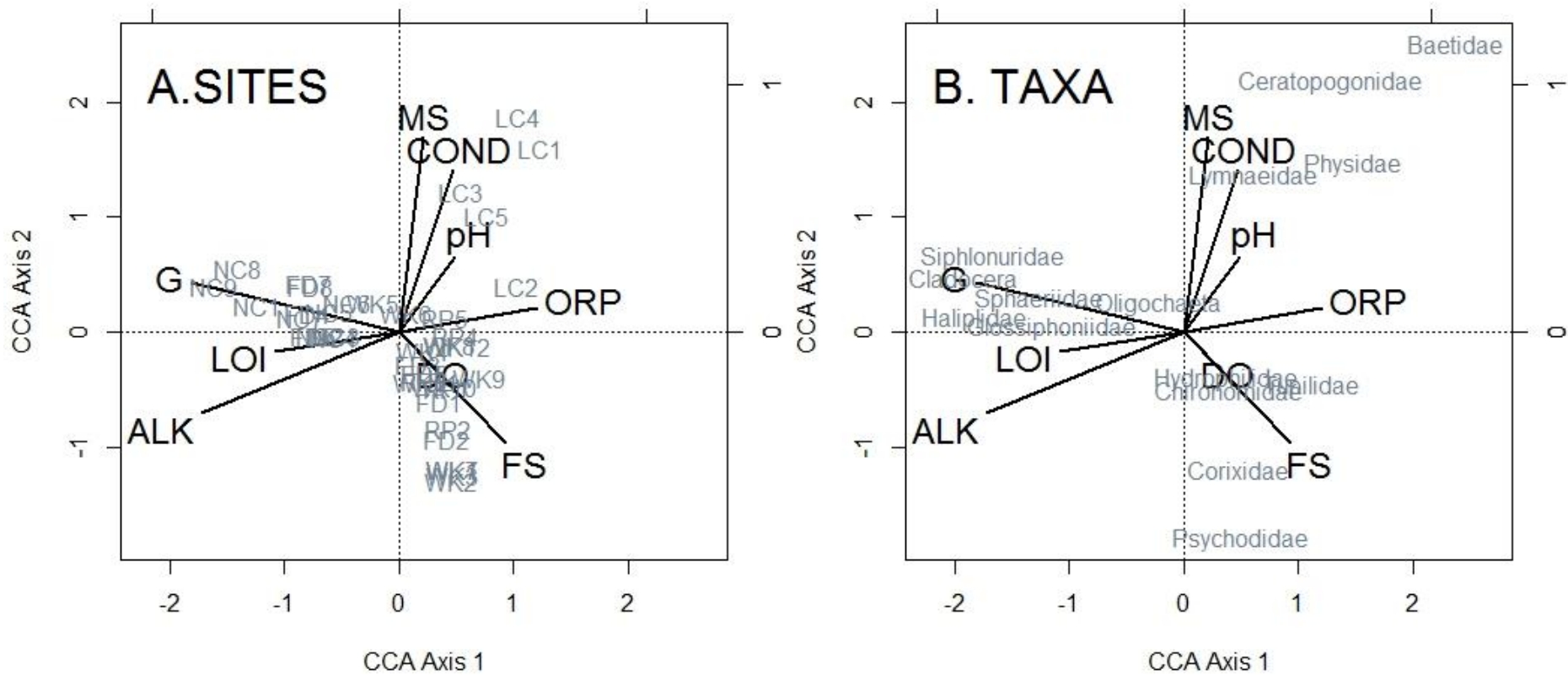
769 Figure 2. Cumulative frequency particle size distribution (%) for all sites. All active silt lagoons are highlighted with dashed lines, whilst solid  
770 black lines represent restored sites.

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



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