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

9

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25

26 **Abstract**

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

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51 **Keywords**

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

54

55 **1. Introduction**

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

68 On an international scale, 44% of known wader populations are contracting (Wetlands
69 International, 2010). According to Eaton et al., (2015) the number of UK wader species now
70 classified as amber or red listed has increased over the last 30 years. Eight species are reliant
71 on lowland wet grassland for breeding and all are recognized as being at varying levels of
72 conservation concern (Wilson et al., 2005; Eaton et al., 2015). Black-Tailed Godwit *Limosa*
73 *limosa* and Ruff *Philomachus pugnax* show very restricted breeding ranges within UK
74 lowlands, often constrained to coastal grasslands, another declining habitat (Wilson et al.,
75 2004). Other species such as Northern Lapwing (*Vanellus vanellus*), Eurasian Curlew
76 (*Numenius arquata*) and Common Redshank (*Tringa totanus*) have demonstrated marked

77 breeding population declines both within UK and Europe (*Sheldon et al., 2004; O'Brien and*
78 *Wilson, 2011; Eaton et al., 2015*). Changes to seasonal tilling and sward height management
79 along with grazing intensification and land drainage are thought to have been associated with
80 breeding habitat loss (*Sheldon et al., 2004; Wilson et al., 2004*). Despite several attempts
81 through numerous Agri-Environment Schemes (AES) to enhance lowland areas for breeding
82 waders (e.g. *Verhulst et al., 2007*), declines continue (*Wilson et al., 2004; Eglington et al.,*
83 *2007; O'Brien and Wilson, 2011*).

84 Aggregate extraction sites are typically located in flat, lowland valleys; areas that
85 would have supported seasonal or permanent water bodies and wetlands (*Andrews and*
86 *Kinsman, 1990; Nicolet et al., 2004; Poschlod et al., 2005*). Once sites have reached the end
87 of their extraction life span, they have the potential to be restored with the end result
88 supporting a comparatively elevated biodiversity to that of the active extraction site (*Milne,*
89 *1974; Bell et al., 1997; Bradshaw, 1997; Santoul et al., 2004; Whitehouse, 2008*). Post-
90 extraction restoration guidelines tend to focus solely on the creation of lakes and the rapid
91 establishment of reed beds (*Ailstock et al., 2001; Jarvis and Walton, 2010*). For example,
92 open water areas in restored quarries benefit wintering and breeding waterfowl whilst reed
93 beds provide breeding areas for species of conservation concern such as Bitterns (*Botaurus*
94 *stellaris*) (*Blaen et al., 2015*), Bearded tit (*Panurus biarmicus*), Reed bunting (*Emberiza*
95 *schoeniclus*) and Reed warbler (*Acrocephalus scirpaceus*) (*Andrews and Kinsman, 1990;*
96 *Peach et al., 1999; Poulin et al., 2002*). However, reed beds provide little foraging or nesting
97 opportunities for waders, who prefer open areas for foraging and shorter sward open
98 grassland habitats for breeding (*Cramp and Simmons, 1983; Milsom et al., 1998*).

99 There is limited evidence that active aggregate sites can provide opportunities for
100 waders, with some species nesting in gravel scrapes (e.g. Little Ringed Plover, *Charadrius*
101 *dubius*) (*Catchpole and Tydeman, 1975; Parrinder, 1989*). Given the ubiquity of such

102 settlement areas in quarry operations, there is the potential for them to contribute to regional,
103 national and international wader conservation goals by replacing lost lowland wet areas both
104 during operation and after post-extraction restoration. Little is known about wader use of
105 active silting areas and management strategies for such areas aimed at wader conservation are
106 not well-developed (Andrews and Kinsman, 1990). We wanted to assess the potential value
107 of active silt lagoons in lowland areas for wader conservation. By integrating environmental
108 and biological data from three active silt lagoon sites of different ages and two restored sites
109 we aimed to (1) characterize the physico-chemical nature of these areas, (2) determine the
110 important environmental factors influencing aquatic invertebrate diversity and abundance, (3)
111 assess how wader richness and abundance varied between sites and 4) how waders actively
112 used these areas. We hoped to use this information to provide recommendations on the
113 management of silt lagoons to improve their potential as sites for wader conservation.

114

115 **2. Methods**

116 **2.1 Study Sites and Location**

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

127 mineral substrate (ranging from extremely coarse to fine sediment), an open aspect, natural
128 terrestrial vegetation and shallow and deep lentic waters.

129

130 **2.2 Environmental parameters and invertebrate diversity**

131 Major labile physico-chemical water parameters (pH, electrical conductivity (COND),
132 oxygen reducing potential (ORP), dissolved oxygen (DO) and temperature (TEMP)) were
133 obtained across all sites. Surface waters were sampled with a Myron Ultrameter (for pH,
134 COND, ORP, TEMP) and an YSI550 Dissolved Oxygen meter for measuring DO. Sample
135 alkalinity (ALK) was assessed in the field via titration against 1.6 N H₂SO₄ with bromocresol
136 green-methyl red indicators (to pH 4.6) using a Hach Digital Titrator. On each sampling
137 occasion, this was repeated three times across each site.

138 During autumn 2015, three 250 cm³ core sediment samples were collected from the
139 bottom of the lake at a depth of 20 - 25cm at each site to characterize the sediment profile.
140 Particle size distribution was obtained by oven drying samples at 105°C, and then fractions
141 separated through a standard nest of sieves (2, 1 mm; 500, 250, 125, 90, 63 and 38 µm) and
142 the percentage of each fraction calculated (Gee and Or, 2002). Sediment fractions
143 incorporated into the analysis included gravels (G; < 2mm), very coarse sand (VCS; 1 –
144 2mm), coarse sand (CS; 500µm – 1mm), medium sands (MS; 250 - 500µm), fine sands (FS;
145 125 - 200µm), very fine sands (63 - 125µm) and coarse silts (38 - 65µm). The median
146 particle size (D50) was used to summarise sediment size, and sediment profiles determined
147 using Gradistat software (version 4.0) (Rice and Haschenburger, 2004). Organic content
148 (LOI) of the substrate was obtained through loss on ignition at 550°C until constant weight
149 was achieved (Generowicz and Olek, 2010).

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

152 2015. The same worker sampled all sites using a kick sampling method, walking backwards
153 at a constant steady pace for 30 seconds to dislodge invertebrates from the substrate into a D-
154 frame pond net (0.25 mm mesh, 350 m x 180 mm frame) at a depth of 15 - 20 cm. Three
155 replicate samples were collected on each sampling occasion and water depth was restricted to
156 < 20 cm (García-Criado and Trigal, 2005) ensuring only invertebrates accessible to waders
157 were collected. Samples were returned to the laboratory and invertebrates identified to Order
158 / Class (Pawley *et al.*, 2011) and the number of individuals in each taxon recorded for each
159 replicate.

160

161 **2.3 Wader site use**

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

170

171 **2.4 Statistical analysis**

172 For all replicate sample from each site collected between June and September,
173 invertebrate family richness (S), total invertebrate abundance (N) and the Shannon Wiener
174 diversity index (H') were calculated using the PRIMER software package (Clarke and
175 Warwick, 2001). The N, S and H' data conformed to a normal distribution (Kolmogorov
176 Smirnov test, P> 0.05 in all cases) and variances could be considered equal (Levene's test,

177 $P > 0.05$ in all cases), therefore a general linear model ANOVA was used to determine if there
178 was any significant difference in S, N and H' between the factors Site (5 levels corresponding
179 to survey sites) and Month (4 levels June – September inclusive). Post-hoc Tukey tests were
180 used to determine the significance of any pairwise differences between the factors Site and
181 Month (Sokal and Rohlf, 1995). A non-parametric Kruskal Wallis test was used to determine
182 if there was a significant difference in the median values of each of the physico-chemical
183 variables during the summer months between sites (Sokal and Rohlf, 1995).

184 A Bray Curtis similarity matrix was generated from square root transformed
185 invertebrate abundance data. The non-parametric two-way ANOSIM (Analysis of
186 Similarities) random permutation test was applied to the similarity matrix to test the null
187 hypothesis that there was no significant difference in invertebrate community similarity
188 between sites (averaged across months) or between months (averages across sites) (Clarke,
189 1993). The SIMPER routine in PRIMER was then used to determine which taxa contributed
190 the most to the average similarity within and between the five sites or months.

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

202 analysis was undertaken in R using the Vegan package (Oksanen *et al.*, 2016) and base
203 package (R Core team, 2016).

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

211

212 **3. Results**

213 **3.1 Environmental variables and invertebrate diversity**

214 All sites surveyed had circum-neutral pH, with TEMP, DO and ORP showing little
215 variation between sites over the summer period (Kruskal Wallis, $P > 0.05$ in all cases; Table
216 2a). Little Catwick had a markedly higher median COND and lower ALK than samples from
217 other sites (Kruskal Wallis, $P < 0.05$ in both cases; Table 2a). This pattern was consistent
218 across the sites throughout the year and the median (range) values for the year are provided in
219 supplementary information S1.

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

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

229 Overall, there was a significant difference in invertebrate family richness between the
230 sites (ANOVA, $F_{4, 43} = 21.7$, $P < 0.001$ Table 2b), but not between months nor any interaction
231 between site and month (ANOVA, $P > 0.05$). North Cave had significantly higher family
232 richness than all other sites (Table 2b), and Filey Dams was significantly higher than the silt
233 lagoon sites (Tukey $P = 0.05$; Table 2b for means). However, there was no significant
234 difference in richness between the three silt lagoon sites (Table 2b). There was also a
235 significant difference in total abundance between sites (ANOVA, $F_{3, 43} = 11.7$, $P < 0.001$ for
236 means see Table 2b) but no significant difference between month not any interaction between
237 site and month (ANOVA, $P > 0.05$). Mean total invertebrate abundance was a factor of ten
238 higher in the samples from North Cave and Filey Dams than that observed from the silt
239 lagoon samples (Table 2b). There was no significant difference in Shannon Wiener H'
240 between the sites, months nor any significant interaction (ANOVA, $P > 0.05$ in all cases;
241 Table 2b). The monthly mean values for each site are provided in supplementary information
242 2, with just the site means presented in Table 2 for brevity.

243 Analysis of Similarities (ANOSIM) showed there was a significant difference in
244 invertebrate community similarity between sites (averaged across months) (ANOSIM, Global
245 $R = 0.657$, $P = 0.1\%$), with the two restored sites significantly different from the silt lagoons
246 (Pairwise comparisons, $P < 0.2\%$ in all cases), but the three silt lagoon sites were not
247 significantly different from each other (Pairwise comparisons, $P > 5\%$ in all cases). There
248 was also a significant difference in invertebrate similarity between months (averaged across
249 all sites) (ANOSIM, Global $R = 0.387$, $P < 0.1\%$) with the samples from June and July being
250 significantly different to those from August and September (Pairwise comparisons, $P < 0.3\%$
251 in all cases). There was no significant difference between June and July samples nor between

252 the August and September samples (Pairwise comparisons, $P > 5\%$ in all cases). The
253 SIMPER routine in the PRIMER software package was used to determine the key
254 invertebrate taxa defining the community similarity at each site. Whilst there were minor
255 differences in the occurrence of taxa between sites (e.g. the lack of Cladocera from silt
256 lagoons; Table 3) overall the relevant abundance of key taxa was key to determining the
257 similarities between the sites rather than major changes in community composition (Table 3).
258 Overall, the relative abundance of Chironomidae, Corixidae and Oligochaeta was key in
259 determining much of the community similarity between sites, however these were far more
260 abundant at North Cave and Filey Dams than the silt lagoons. The relative abundance of
261 these taxa and Cladocera was the major factor contributing to differences between the early
262 summer (June and July) compared to late summer samples (August and September) (full
263 details in supplementary information S3).

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

276 samples on the right hand side of the plot, and Cladocera, Siphonuridae, Haliplidae and
277 Sphaeriidae and with the restored sites on the left hand side (Figure 3b).

278

279 **3.2 Wader site use**

280 Waders were recorded at all sites, however the patterns of use were highly variable
281 (Table 4). Over the entire year, there was a significant difference in the number of different
282 wader species observed at each site (Chi-Squared test, $X^2 = 13.6$, $df = 4$, $P = 0.008$) with
283 more wader species than expected at Filey Dams and North Cave and less than expected on
284 the active silt lagoons (Table 4). The total wader abundance at each site was also highly
285 variable over the months throughout the year (see Table 4), but overall, Ripon supported
286 significantly higher wader abundance (mean = 32 (standard deviation (sd) = 32)) than all
287 other sites (ANOVA, $F_{4, 33} = 4.1$, $P = 0.008$; Tukey = 0.05). No significant difference in
288 abundance occurred between the other sites (Filey Dams mean = 11.8 (sd = 16.3); Wykeham
289 mean = 3.4 (sd = 5.3); Little Catwick mean = 1 (sd = 2.7); North Cave mean = 13.3 (sd =
290 9.3)). The higher abundance at Ripon was largely due to large flocks of Lapwings (*Vanellus*
291 *vanellus*) roosting on the site during late summer and autumn (Table 4). Re-analysis of the
292 data after the removal of Lapwings from the data set revealed no significant difference in
293 abundance between sites (ANOVA, $P > 0.05$) indicating they were the major influential
294 variable on abundance. Site water level management (See Table 1) was important, with low
295 water levels at Filey Dams in late summer/autumn attracting migrants, but high levels during
296 winter and spring restricting availability of feeding areas. Low water levels at North Cave
297 during spring produced breeding habitat for Avocets (*Recurvirostra avosetta*) but as water
298 levels increased in autumn few waders used the site (Table 4). The only other site where
299 breeding was observed was Wykeham, where both Little Ringed Plover (*Charadrius dubius*)
300 and Oystercatcher (*Haematopus ostralegus*) bred in the area (Table 4).

301 Overall, there was also a significant difference in the median percentage of waders at
302 a site exhibiting feeding behaviour (Kruskal Wallis, H (adjusted for ties) = 9.0, $df = 3$, $P =$
303 0.029) with a higher percentage of waders feeding at Filey Dams (median = 20% (range 0 -
304 100%)) than all other sites (North Cave median = 12.5%, Range 0 - 100%; Ripon and
305 Wykeham medians = 0% (Range 0-100)). However, there was a significantly higher median
306 percentage of waders exhibiting roosting behavior (Kruskal Wallis, H (adjusted for ties) =
307 9.4, $df = 3$, $P = 0.024$) at Ripon and Wykeham (medians = 100%, range 0 - 100% in both
308 cases) than seen at Filey Dams (median = 66% (range 0 - 100%) or at North Cave (median =
309 87% (range 0 - 100%).

310

311 **4. Discussion**

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

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

339 The silt lagoon sediments contained very little organic material (Table 2a). Organic
340 detritus is a key component driving benthic invertebrate diversity (Rehfishch, 1994; Chamier,
341 1997; Zilli *et al.*, 2008). Such material originates from both allochthonous and
342 autochthonous sources (Zimmer *et al.*, 2000; Batzer and Sharitz, 2014; Hill *et al.*, 2015),
343 however silt lagoons had no aquatic plants and little organic input from the edge of the
344 lagoons. At both Filey Dams and North Cave, there was obvious organic input from the
345 emergent vegetation in the vicinity and algae were also present (Table 1) and it is well-known
346 that submerged vegetation increases the abundance and diversity of grazing invertebrates and
347 detritivores (Zimmer *et al.*, 2000; Batzer and Sharitz, 2014; Hill *et al.*, 2015). Grazers and
348 detritivores (e.g. Lymnaea, Physidae, Asellidae) were found at Filey Dams and North Cave
349 but absent from silt lagoon samples. Aiding vegetation colonization of open silting areas

350 may help increase invertebrate abundance and provide organic nutrient pools for future
351 succession (Mayes *et al.*, 2005).

352 Both invertebrate species richness and abundance were significantly higher in restored
353 sites than silt lagoons (Table 2b). Increased invertebrate richness and abundance is typically
354 associated with diverse sediment particle size and spatially complex substrates (Flecker and
355 Alan, 1984; Jowett *et al.*, 1991), both within lentic (Sanders and Maloney, 1994; Sanders,
356 2000) and lotic systems (Flecker and Allan 1984). Whilst there was no significant difference
357 in diversity between sites (Table 2b), some taxa occurred on restored sites but were absent
358 from silt lagoons, e.g. the filter feeders Cladocera and Sphaeriidae (Table 2c). Total
359 abundance was a factor of ten higher in restored site samples, despite little variation in
360 composition of the dominant taxa between all sites (Table 2c). The coarser sediments
361 coupled with higher organic content may have contributed to the markedly higher
362 invertebrate abundance. In pond restoration trials, addition of coarser sediments into ponds
363 with a silt-dominated substrates increased invertebrate biomass (Sanders and Maloney, 1994).
364 Larger sediment grain size provides shelter, increases microhabitat diversity and amount of
365 organic entrapment compared to more uniform sands and silts (Flecker and Allan 1984).

366 Silt lagoons evidently can and do support wading birds, but both richness and
367 abundance were lower here than on restored nature reserve sites (Table 4). However, patterns
368 of wader abundance and richness also varied between the restored sites, primarily as the
369 result of different water management practices. The highest wader richness was observed at
370 Filey Dams during late summer (Table 4) when water levels were lowered to produce
371 shallow, muddy areas for foraging migrants in autumn. Some wader species are known to
372 exhibit high site fidelity, returning to the same areas to feed year upon year during migration
373 (Catry *et al.*, 2004) so stop-over sites need to hold predictable and accessible food supplies
374 (Warnock, 2010). Filey Dams has an open aspect, high invertebrate abundance and easy

375 access to prey; three factors closely linked to stop-over site selection in many species of
376 wading birds (Finn *et al.*, 2008; Estrella and Masero, 2010). In contrast, whilst the shallow-
377 water open areas of active silt lagoons were readily accessible, prey availability was
378 markedly lower. In addition, high levels of disturbance also limit wader abundance and
379 diversity (Milsom *et al.*, 1998). Whilst quarrying activity continued on all silt lagoon sites,
380 periodic disturbance due to land re-profiling occurred close to the silt lagoon at Little
381 Catwick which may have resulted in the lowest wader abundance (Table 4).

382 A different management strategy was applied at North Cave; water levels were
383 lowered in spring to create breeding habitat, primarily for Avocets observed on site until late
384 summer (Table 4). Avocets are non-selective feeders, sweeping the bill through the water or
385 along the sediment surface (Dias *et al.*, 2009). The high abundance of invertebrates at this
386 site, especially Cladocera (Table 2c), provided food sources for breeding pairs. Little Ringed
387 Plovers bred on one of the active silting sites, and have been previously recorded nesting in
388 open gravel areas on quarry sites (Wiersma *et al.*, 2016). They forage primarily on terrestrial
389 insects rather than aquatic invertebrates, therefore low aquatic invertebrate abundance would
390 not affect breeding success (Cramp and Simmons, 1983). Oystercatchers also bred within the
391 vicinity, but predominantly foraged in open grassland areas for soil invertebrates (Hulscher,
392 1996; Furnell and Hull, 2014). Whilst probing attempts were made in the silt lagoon
393 sediments, no successful prey capture was observed (Table 4).

394 To avoid predation, most waders maintain a constant level of vigilance (Fuller, 2012;
395 Beauchamp, 2015) irrespective of foraging techniques (Barbosa, 1995). Site topography and
396 vegetation height restrict the effectiveness of vigilance behaviour (Metcalf, 1984). Any
397 obstructions to the birds field of view usually results in increased vigilance and reduced
398 feeding rates (Metcalf, 1984; Beauchamp, 2015). Silt lagoon areas typically possess shallow
399 sloping banks with little significant debris or dense vegetation, and two out of three active

400 sites surveyed here had a wide open aspect. These open sites were commonly used as
401 roosting areas for geese and less frequently Lapwings outside the breeding season (Table 4).
402 Lapwings in particular, were only seen occasionally and probably roosted across a range of
403 sites within the landscape rather than favouring particular roosting areas. Whilst silt lagoons
404 possessed landscape features associated with roosting and foraging sites, there was probably
405 insufficient invertebrates to support significant numbers of foraging waders (Saffran *et al.*,
406 1997) and, whilst our study considers a limited number of active sites, the same pattern of
407 paucity in the invertebrate assemblages was evident.

408

409 **5. Conclusions and management considerations**

410 Whilst the number of sites studied here was limited, it was evident that silt lagoons
411 supported lower wading bird richness and abundance than restored sites in the same
412 physiographic setting. The open aspect, shallow silt lagoon edges could provide important
413 additional lowland habitat for wading birds (Catchpole and Tydeman, 1975; Murray *et al.*,
414 2013) if managed for that purpose and water levels could be regulated. Open areas could be
415 incorporated into future aggregate site restoration plans and the natural succession
416 encouraged ([Prach *et al.*, 2001](#); [Řehouňková and Prach, 2008](#); Joyce, 2014) but managed to
417 ensure they retained an open aspect. Simple interventions, such as depositing over-burden
418 (cobbles and gravels) at the lake edge would increase substrate heterogeneity, trap organics
419 and provide substrate for aquatic macrophytes (Sanders and Maloney, 1994; Sanders, 2000)
420 in turn, increasing overall aquatic invertebrate abundance (Sanders, 2000; Lods-Crozet and
421 Castella, 2009). With many active lowland quarries lying within reach of internationally-
422 important migratory pathways across Europe, such low-cost restoration efforts may play an
423 important role in future landscape-scale wader conservation efforts and replace some of the
424 lost lowland wet grasslands.

425

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756

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	<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.

758

759 Table 2. Summary of a) median (range) of environmental parameters, b) mean (sd) invertebrate richness (S), diversity (Shannon Wiener H') and median
 760 (range) total abundance (N) at each site during the summer months. In addition, c) the key invertebrate taxa from SIMPER analysis contributing to the
 761 differences between the sites over the summer months. Numbers denote the mean (sd) abundance of taxa / those in **bold** denote % contribution of that taxon
 762 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 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) / 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%
763	Sphaeridae	21.4 (24.3) / 8%	2.2 (3.9) / -	0.1 (0.3) / -	0 / -	0 / -
764						
765						

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

768

	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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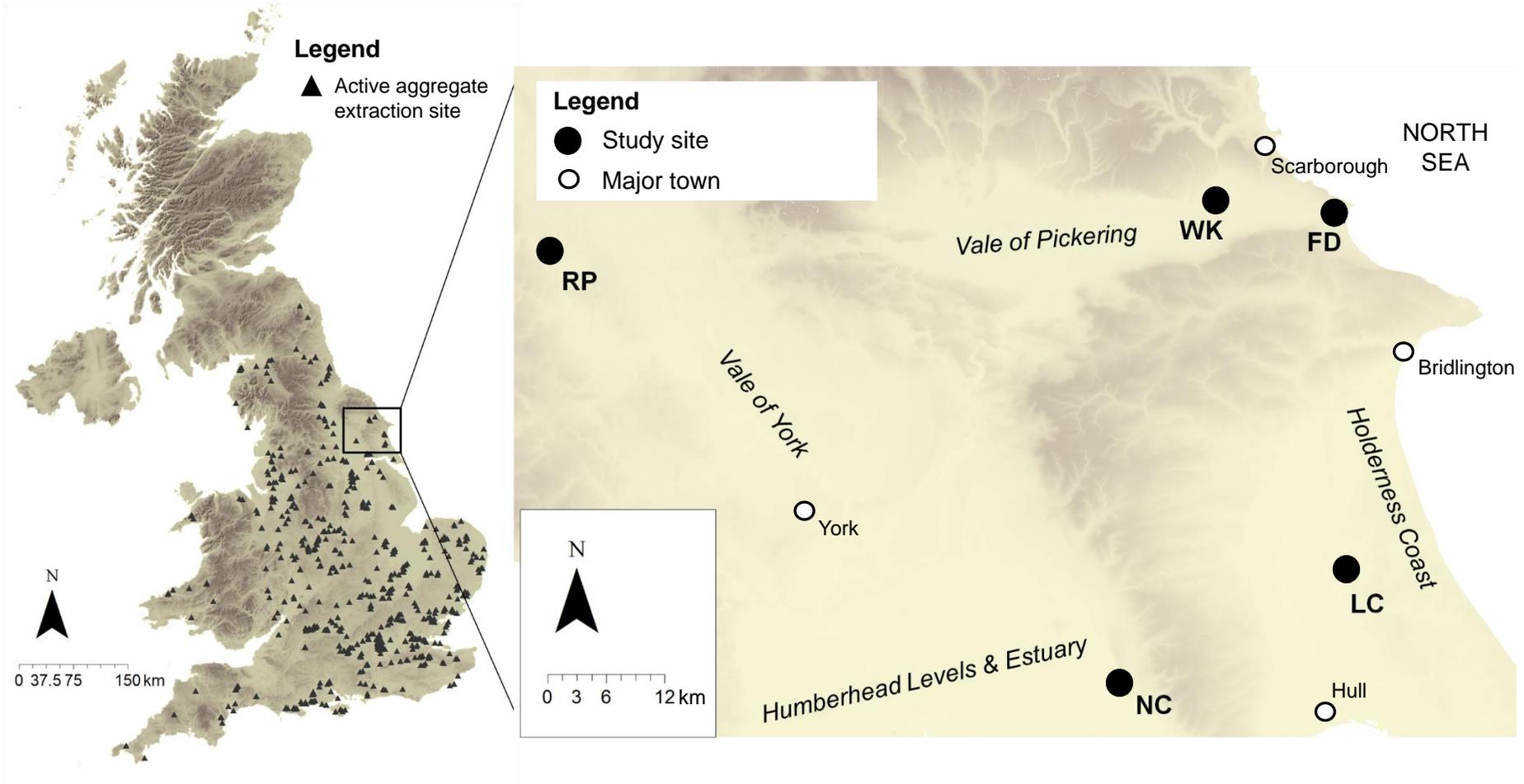
772

773 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							

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

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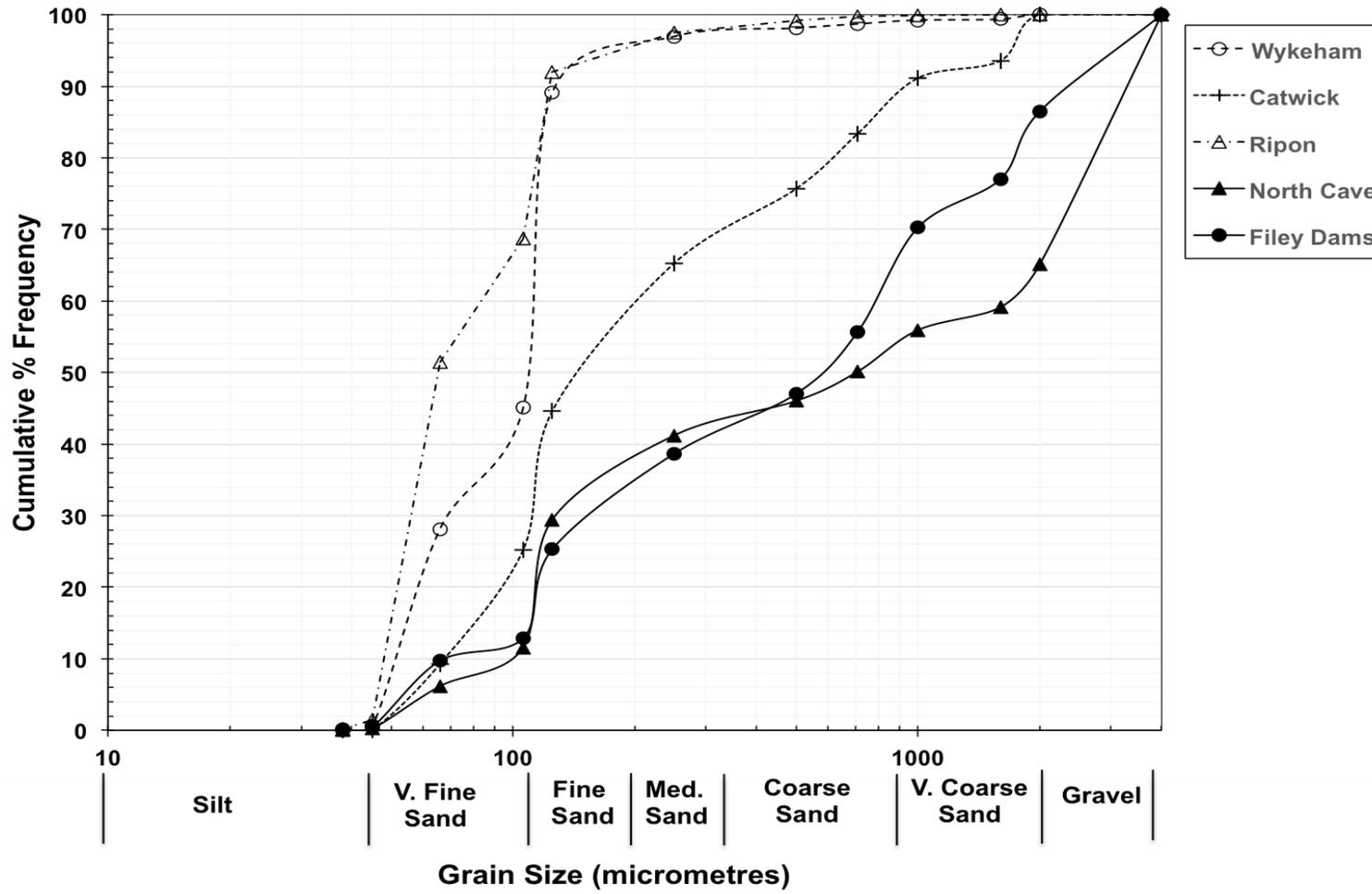


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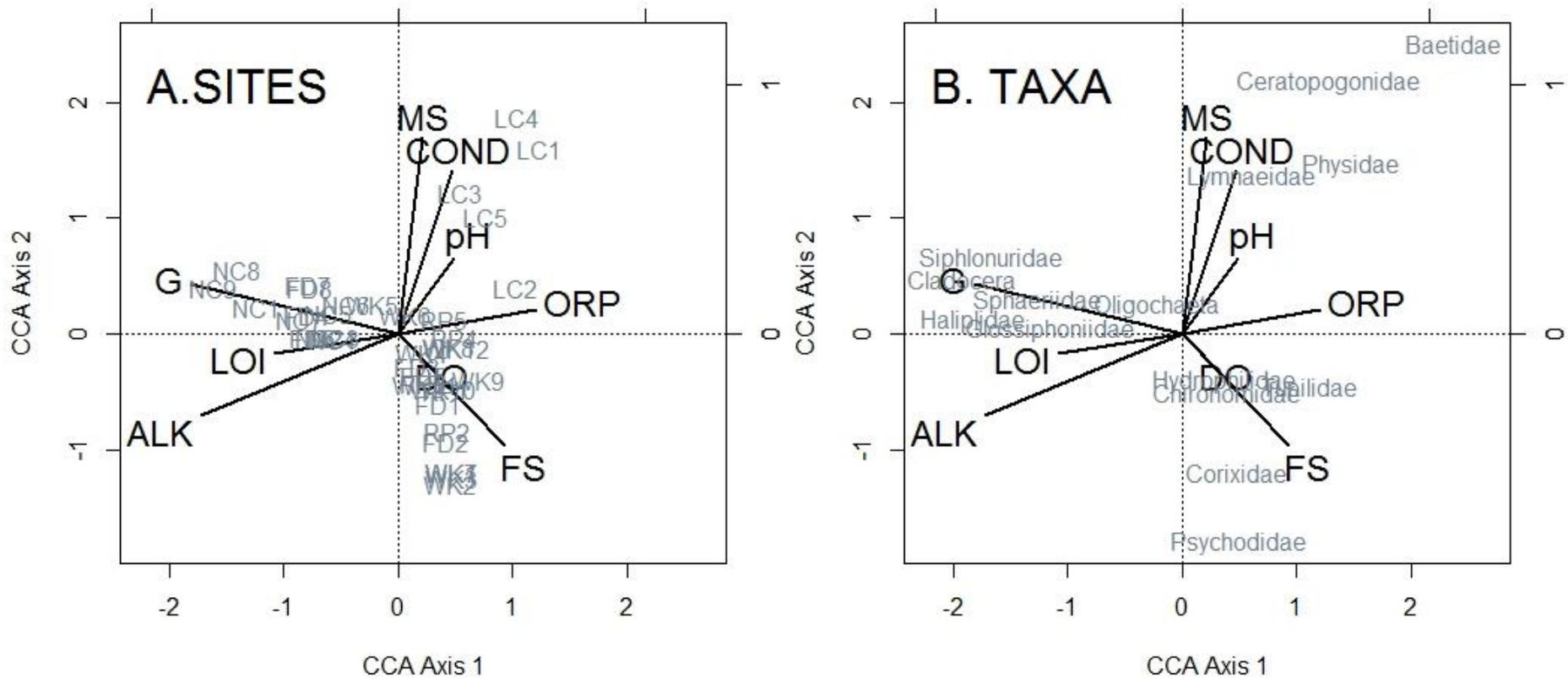
780 Figure 2. Cumulative frequency particle size distribution (%) for all sites. All active silt lagoons are highlighted with dashed lines, whilst solid
781 black lines represent restored sites.

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783

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



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