Development of macrofaunal communities on dredged material used for mudflat enhancement: a comparison of three beneficial use schemes after one year

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Abstract

In recent years, dredged material has become regarded as a potential resource and used to create and/or improve intertidal habitats (‘beneficial use’ schemes). This paper presents the results of a sampling programme to investigate the short-term macrofaunal recovery of three beneficial use schemes in south-east England in terms of species and functional diversity. Environmental parameters (sediment redox potential, and water, organic carbon and silt/clay contents) and univariate community attributes (total individuals and species, diversity, evenness and biomass) at the recharge sites had attained reference levels at two schemes while assemblages differed significantly in terms of species composition at all three schemes. While trophic group proportionality had re-established at one scheme, an increased grazer dominance was apparent at another while the proportion of sub-surface deposit feeders decreased at the third. Total individuals and species number of the developing communities were negatively correlated with sediment redox potential at 4 cm and % silt/clay, respectively. The implications of these results for monitoring the recovery of future fine-grained beneficial use schemes are discussed.

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Keywords: South-east England; Macrofaunal recovery; Dredged material; Beneficial use; Mudflat

1. Introduction

The disposal of maintenance dredged material constitutes one of the most important problems in coastal zone management (Van Dolah et al., 1984; OSPAR, 1998). Furthermore, since ocean disposal of industrial waste and sewage sludge has been phased out, there is greater focus on behalf of concerned citizens, the media and legislative bodies on dredged material disposal (Vogt and Walls, 1991). This has resulted in a greater emphasis on the relocation of fine-grained maintenance dredged material in such a way as to derive environmental benefits (Murray, 1994; Bolam et al., 2003a). As a result, a number of ‘beneficial use’ options have developed whereby the material is regarded as a potential resource and used to recharge or recreate intertidal habitats. In the US, dredged material has been shown to successfully create new mudflats (Ray, 2000) and saltmarshes (Laselle et al., 1991; Posey et al., 1997; Streever, 2000) which ultimately function like natural systems. In the UK, concerns over the eventual fate of the material and the ecological consequences of placing fine-grained material onto intertidal habitats have limited this practice to small-scale field trials. Currently, less than 1% of the 40–50 million m³ produced in the UK is used beneficially (Bolam et al., 2003b).

Over recent years, the large number of studies investigating macrofaunal recovery following a number intertidal disturbances has resulted in a good understanding...
of the potential macrofaunal recovery rates and mechanisms (Evans et al., 1998; Beukema et al., 1999; Bolam and Fernandes, 2002; Bolam et al., 2002, 2004; Lewis et al., 2003). In general, recolonisation of some species can be rapid, although this depends on the spatial scale and timing of the disturbance, together with the life history characteristics of the recolonising fauna. However, apart from a small number of monitoring studies of variable quality, there have been very few studies investigating the macrofaunal recovery of fine-grained beneficial use schemes in the UK (Atkinson et al., 2001; Bolam and Whomersley, 2004). This, together with concerns over the eventual fate of the material, presents a major barrier to the large-scale use of dredged material for habitat creation/improvement for the foreseeable future. As macrofaunal recolonisation of beneficial use schemes initially depends upon the re-establishment of the physical (bulk density, particle size) and physico-chemical (redox potential) properties of the dredged material (Bolam, 2003; Bolam et al., 2004), recovery rates may conceivably vary from those of other disturbances.

When dredged material is placed onto an intertidal mudflat the resident invertebrates are smothered and recovery occurs via a combination of adult/juvenile settlement and lateral and/or vertical migration (Bolam et al., 2003a). Which mechanism predominates in any instance depends upon the timing, rate, depth and spatial scale of the recharge, together with changes in the properties of the sediment itself. This study documents the invertebrate communities at a number of recharge and reference stations within three comparable beneficial use schemes in south-east England one year after recharge. This is the first UK study designed to allow within- and between-scheme comparisons using a standardised sampling approach. The main objectives of the study were to compare (i) univariate community attributes, (ii) species composition, and (iii) trophic composition within and between schemes, and to propose the factors responsible for any differences. In this study it is shown that although univariate community attributes may recover within one year, longer-term differences in community structure may have implications for the functioning of beneficial use schemes.

2. Methods

2.1. Study sites and sampling

The three beneficial use schemes investigated during the present study were at Westwick Marina (WW) (51° 38.692’ N; 00° 39.61’ E), Titchmarsh Marina (TM) (51° 51.763’ N, 001° 15.133’ E) and North Shotley (NS) (51° 57.973’ N, 001° 16.469’ E) (see Fig. 1). Each scheme consisted of either a mudflat (NS) or muddy channels within a saltmarsh system (WW and TM) recharged with 60–80 cm (vertical overburden) of fine-grained, maintenance dredged material. The sediments recharged were considered uncontaminated in terms of metals and TBT (CEFAS, unpubl.). The resulting tidal height of each scheme was below the limits of saltmarsh plant colonisation (i.e., 2.1 mOD) and, consequently, high-level mudflats were the most appropriate references to assess faunal recovery. Reference sites were located as near as possible to the recharge area without being impacted by the recharge process itself.

For each scheme, three stations were positioned within the recharge (hereafter described as recharge stations 1–3) and reference sites (reference stations 1–3). At each station, replicate macrofaunal samples (n = 3) were taken using a 0.01 m² perspex corer to a depth of 15 cm. Samples were preserved in 10% buffered formalin with 0.01% Rose Bengal stain. These were later washed over a 500 μm mesh sieve in the laboratory, the invertebrates were then sorted under a dissecting microscope, identified to the lowest possible taxonomic resolution and counted. Replicate 5L samples (n = 3) of dredged material which were collected during the recharge process were sieved then treated as described above.

The top 3 cm of the sediment at each station was sampled for sediment analyses. These sediments were then frozen prior to the determination water content (weight loss on drying at 80°C for seven days), organic content (Leeman CE440 element analyser) and particle size distribution (Coulter LS-130 laser diffraction) analyses. Replicate (n = 3) redox potential profiles (1, 2 and 4 cm) were measured at each station using a calibrated Russell RL100 Redox Metre with a calomel probe following the methods outlined by Pearson and Stanley (1979).

![Fig. 1. Map showing relative locations of the three beneficial use schemes and the positions of the recharge (circles) and reference (triangles) stations. While WW is located at the head of a small estuary, TM is situated within an embayment and NS towards the confluence of two larger estuaries.](image-url)
2.2. Data analysis

The sediment and macrofaunal data collected one-year post-recharge were analysed using both univariate and multivariate (macrofauna only) data analysis techniques. For univariate analyses, the data were checked for normality using the Anderson-Darling test and homogeneity of variance was assessed by the Bartlett test. Any data not conforming to either of these were transformed using an appropriate transformation (Zar, 1984). Two-sample t-tests were then conducted to test for differences in recharge and reference sites for each scheme. Relationships between total individuals and number of species with sediment variables of the recharge stations were investigated using Spearman’s rank correlation tests.

Multivariate analyses were carried out to assess (dis)similarities in community structure between recharge and reference stations and between schemes. All multivariate analyses were performed using the PRIMER package, version 5.2.3 (Clarke and Warwick, 1994; Clarke and Gorley, 2001). Non-metric multidimensional scaling (MDS) was carried out from the Bray-Curtis similarity matrices on root-transformed data to produce an ordination plot. In ordination plots, the relative distances apart reflect relative (dis)similarities in species composition. Since the MDS plot reduces a multidimensional ordination to two dimensions, each algorithm has an associated stress value, discussed by Clarke and Warwick (1994). Testing for significance between recharge and reference site within and between schemes was performed using a priori, one-way analysis of similarities (ANOSIM) tests in which the null hypothesis ($H_0$) in each case was that there were no significant community differences between recharge and reference sites and between schemes. The test statistic $R$ will always be between 0 and 1; if $R \approx 1$, all replicates within sites are more similar to each other than any other replicates from different sites, while $R \approx 0$, similarities between and within sites will be the same on average (Clarke, 1993). The SIMPER (similarity of percentages) programme was used to indicate the most discriminating taxa between recharge and reference stations. The BIO-ENV programme was used to investigate relationships between community structure and sediment variables.

3. Results

3.1. Sediment data

The water, organic carbon and silt/clay contents of the dredged sediments during recharge and after one year are given in Table 1. The material recharged at WW and TM had markedly greater water contents relative to the sediments recharged at NS. However, these sediments greatly dewatered (Table 1) to give significantly lower water contents than reference sediments at TM after one year. The organic carbon and silt/clay contents were similar in all three dredged materials. The silt/clay contents were similar between the schemes, both in the recharge and reference sites, however, organic carbon content of the recharged sediments were significantly lower than reference sediments after one year. There were no significant differences in the redox potential profiles between recharge and reference sites ($z = 0.05$; t-test), except a significantly increased redox potential at 1 cm sediment depth at TM recharge site (Fig. 2(ii)).

3.2. Invertebrate data

3.2.1. Univariate analysis

The univariate indices of the communities 12 months post-recharge are presented in Fig. 3(i)–(v). The five indices show various degrees of convergence to reference values; while there were no significant differences for diversity, evenness and biomass for any scheme, total individuals (TM) and number of species (WW and TM) were significantly lower in the recharge than reference sites.

<table>
<thead>
<tr>
<th>Scheme</th>
<th>Tidal height (m OD)*</th>
<th>Water content (%)</th>
<th>Carbon content (%)</th>
<th>Silt/clay (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>WW</td>
<td>Dredged material</td>
<td>–</td>
<td>91.2 ± 3.0</td>
<td>1.3 ± 0.1</td>
</tr>
<tr>
<td></td>
<td>Recharge site</td>
<td>1.2–1.3</td>
<td>66.5 ± 3.7</td>
<td>2.7 ± 0.26</td>
</tr>
<tr>
<td></td>
<td>Reference site</td>
<td>0.6–1.2</td>
<td>62.2 ± 2.2</td>
<td>2.0 ± 0.1</td>
</tr>
<tr>
<td>TM</td>
<td>Dredged material</td>
<td>–</td>
<td>91.7 ± 0.3</td>
<td>1.2 ± 0.1</td>
</tr>
<tr>
<td></td>
<td>Recharge site</td>
<td>1.5–1.9</td>
<td>60.6 ± 0.62</td>
<td>1.58 ± 0.06</td>
</tr>
<tr>
<td></td>
<td>Reference site</td>
<td>1.4–1.5</td>
<td>73.6 ± 0.53</td>
<td>2.78 ± 0.17</td>
</tr>
<tr>
<td>NS</td>
<td>Dredged material</td>
<td>–</td>
<td>60.3 ± 2.3</td>
<td>1.5 ± 0.1</td>
</tr>
<tr>
<td></td>
<td>Recharge site</td>
<td>1.4–1.5</td>
<td>55.2 ± 2.57</td>
<td>1.37 ± 0.10</td>
</tr>
<tr>
<td></td>
<td>Reference site</td>
<td>0.5–0.6</td>
<td>50.8 ± 10.41</td>
<td>1.27 ± 0.28</td>
</tr>
</tbody>
</table>

Recharge and reference values in bold are significantly different ($z = 0.05$; two-sample t-test).

* Refers to eventual tidal height after dredged material placement for recharge sites.
3.2.2. Multivariate analysis

The MDS plot of all recharge and reference samples is presented in Fig. 4. The plot reveals firstly, that the reference communities were very dissimilar between schemes, and secondly, that the reference communities exhibited large within-site variability, i.e., between stations. Furthermore, the plot reveals large differences between the recharge and reference communities for each scheme. These differences, which contrast with the results from univariate analyses, were formally tested using ANOSIM tests (Table 2) which revealed that the invertebrate communities of recharge sites were
significantly different from reference sites for each scheme (Table 2(i)). The results also indicate that there were significant differences between the recharge communities of each scheme from each of the other schemes (Table 2(ii)). The same holds true for the reference communities.

Differences between recharge and reference communities were mainly due to dissimilarities in the mean abundance of *Tubificoides benedii* (Udekem) and *Streblospio shrubsolii* (Buchanan), while *Corophium volutator* (Pallas), *Capitella capitata* (Fabricius), *Hydrobia ulvae* (Pennant) and *Tharyx* sp. were also important discriminating taxa (Table 3). *C. volutator* and *H. ulvae* were the only discriminating species to show increased abundance at the recharge sites, the other taxa being more abundant at the reference sites.

### 3.2.3. Trophic groups

In total, six trophic groups were sampled throughout this study, most of these being present in all recharge and reference communities (except the reference site at NS which was composed of three trophic groups only) (Fig. 5). The recovery of trophic groups varied between schemes. For example, while the relative proportions of trophic groups were similar between recharge and reference communities at WW, the sub-surface deposit feeder numerical dominance at TM was replaced by a grazer dominance at the recharge site. Although the high proportion of surface deposit feeders at NS reference site

![Fig. 4. MDS plot of all recharge and reference samples from the three schemes (based on a Bray-Curtis similarity matrix from root-transformed data). Shaded symbols represent recharge stations while clear symbols represent reference stations.](image)

![Fig. 5. Proportions of each trophic group sampled from recharge and reference stations for the three schemes one year post-recharge.](image)

<table>
<thead>
<tr>
<th>Table 2</th>
</tr>
</thead>
<tbody>
<tr>
<td>Results of one-way ANOSIM tests of the differences between (i) recharge and reference community structures within each scheme, and (ii) the recharge stations between schemes and reference station between schemes</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Scheme</th>
<th>R-statistic</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td>(i) Within schemes: difference between recharge and reference stations</td>
<td></td>
<td></td>
</tr>
<tr>
<td>WW</td>
<td>0.961</td>
<td>0.001</td>
</tr>
<tr>
<td>TM</td>
<td>0.732</td>
<td>0.001</td>
</tr>
<tr>
<td>NS</td>
<td>0.39</td>
<td>0.003</td>
</tr>
<tr>
<td>(ii) Between schemes: for both recharge stations and reference stations</td>
<td></td>
<td></td>
</tr>
<tr>
<td>WW &amp; TM</td>
<td>1.00</td>
<td>0.001</td>
</tr>
<tr>
<td>WW &amp; NS</td>
<td>0.79</td>
<td>0.001</td>
</tr>
<tr>
<td>TM &amp; NS</td>
<td>0.79</td>
<td>0.001</td>
</tr>
</tbody>
</table>

* p values in bold are significant (*α* = 0.05) after a Bonferroni correction.

**Table 3**

The three main taxa responsible for the dissimilarities between recharge and reference sites for each scheme

<table>
<thead>
<tr>
<th>Scheme</th>
<th>Taxa</th>
<th>Mean recharge abundance</th>
<th>Mean reference abundance</th>
<th>% contribution to recharge–reference dissimilarity</th>
</tr>
</thead>
<tbody>
<tr>
<td>WW</td>
<td><em>Corophium volutator</em></td>
<td>365.7</td>
<td>1.7</td>
<td>30.5</td>
</tr>
<tr>
<td></td>
<td><em>Tubificoides benedii</em></td>
<td>171.7</td>
<td>466.3</td>
<td>25.8</td>
</tr>
<tr>
<td></td>
<td><em>Streblospio shrubsolii</em></td>
<td>53.3</td>
<td>204.0</td>
<td>12.3</td>
</tr>
<tr>
<td>TM</td>
<td><em>Capitella capitata</em></td>
<td>0.0</td>
<td>1962.0</td>
<td>24.78</td>
</tr>
<tr>
<td></td>
<td><em>Hydrobia ulvae</em></td>
<td>252.0</td>
<td>12.5</td>
<td>15.5</td>
</tr>
<tr>
<td></td>
<td><em>Tubificoides benedii</em></td>
<td>3.7</td>
<td>722.1</td>
<td>9.1</td>
</tr>
<tr>
<td>NS</td>
<td><em>Tubificoides benedii</em></td>
<td>215.0</td>
<td>280.3</td>
<td>15.7</td>
</tr>
<tr>
<td></td>
<td><em>Tharyx sp.</em></td>
<td>0.7</td>
<td>41.0</td>
<td>11.0</td>
</tr>
<tr>
<td></td>
<td><em>Streblospio shrubsolii</em></td>
<td>20.0</td>
<td>56.7</td>
<td>9.8</td>
</tr>
</tbody>
</table>

Dissimilarities were 62.2%, 84.5% and 72.2% for WW, TM and NS, respectively. The mean abundance per core of each taxa are given together with the % contribution to the overall dissimilarity.
(primarily *C. capitata*; see Table 3) had not established at the recharge site, the recharge site exhibited a larger number of trophic groups.

### 3.2.4. Biotic relationships with environmental variables

The relationships between univariate indices (total individuals and number of species) and sediment variables (water, carbon and silt/clay contents, 1, 2 and 4 cm redox potential) were investigated using a correlation approach. Total individuals were negatively correlated with 4 cm redox potential and number of species negatively correlated with % silt/clay. These relationships are displayed in Fig. 6 (i) and (ii).

BIOENV analysis revealed poor relationships between sediment variables and multivariate community structure (Table 4). The single variable which best groups the sites, in a manner consistent with the faunal patterns, was sediment carbon content ($q = 0.169$). The best two-variable combination also involves sediment carbon content with 1 cm redox, but is a weaker correlation ($q = 0.125$) than with carbon content only. Generally, these $q$ values are very low and one can conclude that the measured environmental variables do not capture the variability of multivariate faunal dataset.

### 4. Discussion

Typically low-diversity assemblages dominated by opportunistic species (Levin, 1984; Moy and Levin, 1991; Trueblood et al., 1994) mudflat assemblages appear to be in a perpetual state of early succession. Large, deep-burrowing taxa characteristic of equilibrium assemblages (*sensu* Rhoads and Boyer, 1982) are rare or absent. In this respect, one might expect (assuming suitability of the recharged sediments) recovery rates following dredged material placement to be relatively quick compared to those of other habitats (Bolam and Rees, 2003). Bolam et al. (2004) found that recolonisation of 1 m$^2$ defaunated sediments resulted in recovery of univariate indices after only three months and community structure after 6–12 months on a mudflat in south-east England. Beukema et al. (1999) noted that number of species and individuals took 6 and 12 months to recover, respectively following the defaunation of larger areas (120 m$^2$) of mudflat in the Dutch Wadden Sea. Evidence from one of the few studies using dredged material for mudflat creation/enhancement supports the notion of relatively quick recovery on mudflats. For example, Bolam and Whomersley, 2004 found that the diversity, abundance and species richness of two mudflat communities created from dredged material near Jonesport, Maine, had re-established the levels found at reference mudflats after two years.

Experiences gained from saltmarsh creation studies suggest that failure of created communities to converge to those of reference sites can be partly explained by differences in sediment characteristics between created and natural habitats (Levin et al., 1996). Differences in sediment organic contents and particle size between recharge and references sites have been found to exist for many years at some schemes (Langis et al., 1991; Lee et al., 1998; Craft et al., 1999). Significant differences in sediment properties (water, carbon and silt/clay contents, 1 cm redox potential) between recharge and reference sites existed only at TM, the scheme which displayed the greatest deviation from the reference site for univariate indices (significant difference for total individuals and species) and multivariate community structure (84.5% dissimilarity from reference site community). This may indicate that sediment differences may have been responsible for the lack of convergence of recharge site communities with reference communities at TM and in previous studies. However, although 4 cm redox potential and % silt/clay were significantly correlated with total abundance and number of species, respectively, sediment variables exhibited poor

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**Table 4**

<table>
<thead>
<tr>
<th>Variables</th>
<th>$q$</th>
</tr>
</thead>
<tbody>
<tr>
<td>% carbon</td>
<td>0.169</td>
</tr>
<tr>
<td>% carbon and 1 cm redox</td>
<td>0.125</td>
</tr>
<tr>
<td>% carbon and 4 cm redox</td>
<td>0.123</td>
</tr>
</tbody>
</table>

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**Fig. 6.** Scatterplots of significant relationships between univariate indices and sediment variables.
correlation with multivariate community structure of the recharge sites. Clearly, how critical the degree of similarity of recharged sediments to those in reference areas is for invertebrate recolonisation requires further investigation.

Although most univariate indices recovered within 12 months, macrofaunal community structure of the recharge sites failed to converge to those of reference sites for all three schemes. These differences are unlikely to have resulted from sediment differences but may reflect some underlying differences in environmental conditions between recharge and reference sites. Bolam and Whomersley (2004) discussed that at WW, natural spatial heterogeneity of the main species, C. volutator, resulted in inherently different community structures between the recharge (prior to dredged material placement) and reference site. The communities recolonising the dredged sediments developed towards those collected at the same stations prior to dredged material disposal rather than those of the reference site. This suggests that natural spatial variability in response to underlying physical and/or biological processes may prevent total convergence between recharge and reference sites. Bolam and Whomersley (2004) also noted that tidal height differences may be partly responsible for the lack of apparent recovery of recharge communities. While the reference stations used in this study were initially chosen to have tidal heights as close as possible to the recharge sites following dredged material placement, the eventual tidal heights were between high mudflat and saltmarsh (Table 1). It was not, therefore, possible to have reference stations with identical tidal heights as those of recharge stations. Monitoring programmes need to be able to distinguish between macrofaunal community differences that reflect differences in tidal height or sediment properties with those differences that reflect incomplete establishment of a normally functioning community. Few of the published or unpublished studies have previously collected or stated sufficient detailed data in this respect and, therefore, we are unable to make accurate assessments of their importance. However, the results presented here provide important information to help address this problem.

A more functional-based approach may allow us to assess recovery regardless of whether community structure has been restored relative to a reference site. Bolam et al. (2002) experimentally manipulated the number of species on an intertidal mudflat and concluded that this had no effect on ecosystem functioning (nutrient flux, sediment stability, redox potentials and bulk properties). They concluded that ecosystem functioning on mudflats may be more dependent on the number of trophic groups rather than species per se. Although the relative proportions of trophic groups between recharge and reference communities varied after one year, the number of trophic groups had re-established in each case. If the community of a recharge site had recovered in some aspect of function yet had a significantly different community structure, can we describe it as 'recovered'? The answer must depend upon what function (or 'ecosystem service' sensu Ehrlich and Wilson, 1991) the habitat is expected to perform. As an important function of mudflat assemblages is to support higher trophic levels (e.g., fish and birds) a valued assessment of recovering community in this respect may be more ecologically important and more meaningful than community structure attributes. While inherent differences between recharge areas and reference sites exist (see above) we may run the risk of failing to accept recovery (i.e., based on sensitive community structure attributes) when indeed the assemblage may be sufficiently performing its most fundamental ecological role.

In conclusion, this study demonstrates that rich and diverse infaunal communities can be established on mudflats enhanced using dredged materials. The relatively rapid recolonisation observed here can be attributed, at least in part, to the high resilience of mudflat communities and to the similarity of the dredged sediments to reference sediments in terms of organic carbon and silt/clay contents. Natural spatial variability of macrofauna and tidal height differences indicate that a functional-based approach may prove to be more suitable than community structure attributes for recovery assessment.

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References


