Journal of Coastal Conservation 9: 159-169, 2003
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Invertebrate recolonization of fine-grained beneficial use schemes: An example from the southeast coast of England

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Abstract. The disposal of maintenance dredged material constitutes one of the most important problems in coastal zone management. To minimise ecological impacts, 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. This paper presents the results of a sampling programme to investigate the macrofaunal recovery rates, and the underlying mechanisms responsible for them, following a beneficial use scheme involving the placement of fine-grained dredged material on a salt marsh in southeast England.

Three stations in the recharge area and three reference stations, located within the same salt marsh system, were selected. These stations were sampled prior to recharge (recharge stations only) then 1 week, 3, 6, 12 and 18 months after the recharge. Sediment redox potentials (1, 2 and 4 cm sediment depths) were also measured on each sampling occasion. The results indicated a rapid recolonization of the fauna typical of the surrounding salt marsh channels. All univariate parameters had recovered after 3 months after the recharge. Active post-juvenile immigration is likely to have been the predominant recovery process. Multivariate data analysis revealed that the community structure of the recharge stations, however, did not progress towards those of the reference sites. Natural spatial variability in community structure at the scale of the recharge-reference station distance, and differences in eventual tidal elevations are factors responsible for these differences. The need to carefully assess reference site suitability in monitoring beneficial use schemes is discussed.

Keywords: Dredged material; Macrofauna; Monitoring; Recovery; Salt marsh.

Abbreviation: MDS = Non-metric Multidimensional Scaling.

Introduction

The disposal of material from maintenance dredging constitutes one of the most important problems in coastal zone management (Van Dolah et al. 1984; Anon. 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 & 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 or other benefits (Murray 1994). 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. Dredged material has been shown to successfully create new mud flats (Ray et al. 1994; Evans et al. 1998) and salt marshes (LaSalle et al. 1991; Posey et al. 1997; Streever 2000) which eventually become capable of functioning like natural systems.

At present, the beneficial placement of material from maintenance dredging within the UK is limited to small-scale trials. There are several reasons for this. Firstly, there are concerns over subsequent movement of the material under natural forces (wave and tidal current action) and hence the potential for interference with other uses/users of the area. Secondly, our lack of knowledge of the rate of invertebrate recovery, and how this is affected by other factors (timing, rate and depth of recharge, properties of the dredged material), limits our ability to predict the effects of sediment placement on bird and fish populations (Evans et al. 1998; Bolam et al. 2003). This is particularly important as the majority of beneficial use schemes are located in estuarine intertidal habitats, areas important for sustaining such populations.

When dredged material is placed onto an intertidal mud flat the resident invertebrates are smothered and recovery occurs via adult/juvenile settlement and/or lateral migration (Bolam et al. 2003). Clearly, therefore, a good understanding of the recovery processes following intertidal dredged material placement is needed if detrimental ecological consequences of this practice are to be minimised. This paper presents the findings of a sampling programme to investigate the recovery of macrofaunal invertebrates following a beneficial use scheme involving fine-grained dredged material at Westwick Marina, Crouch Estuary, England. Specifically, this paper focuses on the rate and type of recovery and discusses the findings with respect to implications for the future monitoring of such schemes.

Methods

Study site

The Crouch Estuary, Essex, is located north of the Thames Estuary, England (Fig. 1). The estuary could be described more appropriately as a sea inlet. The volume of freshwater input is low as a proportion of estuary volume, and for most of the year the system is dominated by tidal ebb and flow of high-salinity waters along much of its length (Waldock et al. 1999).

Westwick Marina is located on the north bank of the estuary. The marina is situated in a protected inlet of the main channel and requires maintenance dredging of silts on a regular basis. In 2000, a licence was granted by the Department for Environment, Food and Rural Affairs (DEFRA) for the marina to place maintenance dredged material directly onto an area of eroding salt marsh adjacent to the marina. The infaunal communities of the channels of this salt marsh were numerically rich, dominated by tubificid oligochaetes, the amphipod *Corophium volutator* and the polychaete *Hediste diversicolor*. The material was dredged using a suction dredger and pumped along a floating pipeline to the recharge area (a total distance of 50 m). A small area of marsh was selected for recharge, woven fences being used to retain as much of the dredged material as possible after recharge.

Sampling

Three stations were positioned within the area to be recharged (hereafter described as recharge stations 1-3), and three reference stations within the same salt marsh system, at equivalent tidal heights and away from the effects of the recharge (hereafter referred to as reference stations 1-3). These stations were located within the salt marsh creeks rather than on the marsh surface, and consequently, they are typified by large abundances of mud flat invertebrates. The macrofaunal

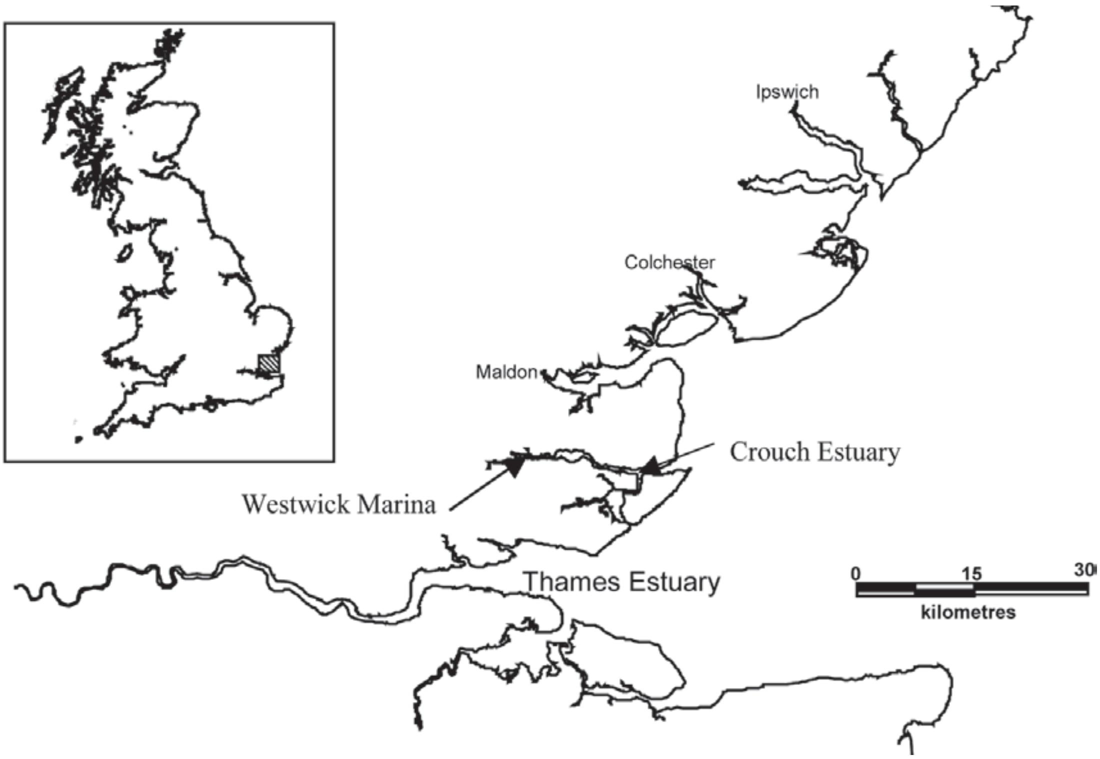

Fig. 1. Map showing the location of Westwick Marina on the Crouch Estuary, England.

communities of the three recharge stations were sampled 1 week before recharge using a 0.01-m^2 corer to a depth of 15 cm. Three replicates were sampled at each station. These samples were used to give an indication of the faunal communities prior to recharge. All stations were then sampled on 11 August, 2001 (one week after recharge) using the same methodology as above, then 3, 6, 12, and 18 months after recharge. Samples were preserved in 10% buffered formalin with 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 5-L samples of dredged material were also collected during the recharge process; these were sieved then treated as above. Replicate sediment redox profiles were also taken at each sampling station at 1, 2, and 4 cm sediment depths using the methodology of Pearson & Stanley (1979).

Data analysis

The invertebrate data were analysed using both univariate and multivariate data analysis techniques. For univariate analyses, the data were checked for normality using the Anderson-Darling test and homogeneity of variances were assessed by the Bartlett test. Any data not conforming to either of these were transformed using an appropriate transformation (Zar 1984). To test for differences between recharge and reference stations within each sampling occasion, ANOVA tests were conducted. Tukey multiple comparison tests were performed (Zar 1984, pp. 185-190) to investigate the differences ($p < 0.05$) observed in the ANOVA tests. The Tukey test is a global comparison test and allows for multiple testing to be conducted without increasing the risk of type I errors. All univariate analyses were conducted using Minitab v13.3.

Multivariate analyses were carried out to assess (dis)similarities between community assemblages between recharge and reference stations and between sampling times. All multivariate analyses were performed using the PRIMER package, version 5.2.3 (Warwick & Clarke 1994). 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 multi-dimensional ordination to two dimensions, each algorithm has an associated stress value, discussed by Warwick & Clarke (1994). The SIMPER (similarity of percentages) program was used to indicate which were the most discriminating taxa between recharge and reference stations.

Results

Sediments

The resulting depth of recharged material at each of the recharge stations is shown in Table 1. The depths were similar between all three stations, between 49 and 57 cm of sediment. Table 1 also gives the height above ordnance datum and tidal immersion times for each recharge (post-recharge) and reference station. The recharge stations following recharge were with one exception notably higher than the reference stations. This could not be prevented as the reference stations were the highest channels that could be found within the salt marsh.

The mean redox potential values at 1, 2, and 4 cm sediment depths are shown in Fig. 2a-e. These results, which give an indication of the physico-chemical conditions within the sediments (which may affect the invertebrates), suggest that the reduction-oxygenation gradients established within the dredged material a very short time after recharge. Within 1 week (Fig. 2a), there were no visible differences in the redox potentials between the three recharge stations and the reference stations (although one reference station was lower relative to the others). This was consistent with the values observed throughout the sampling period, similar temporal changes occurring in the recharge as the reference stations.

Invertebrates

A total of 36 136 individuals from 40 taxa were enumerated and identified over the sampling period. The most abundant taxa were (in order of decreasing abundance) the oligochaete *Tubificoides benedii* (Udekem), the polychaete *Streblospio shrubsolii* (Buchanan), the amphipod *Corophium volutator* (Pallas), nematodes and the polychaete *Hediste diversicolor* (Müller). Together, these five taxa comprised 79.8% of the individuals sampled. The mean abundances per core (with standard errors) of each of these taxa are displayed in Fig. 3a-b. The three replicate samples of dredged

Table 1. Characteristics of the recharge and reference stations.

Station	Depth of recharge (m)	Tidal height (m above OD)	Immersion time (%)
Recharge 1	0.49	3.7	29.2
Recharge 2	0.54	4.1	24.6
Recharge 3	0.57	4.1	23.6
Reference 1	-	3.6	32.6
Reference 2	-	3.7	29.2
Reference 3	-	3.7	30.6

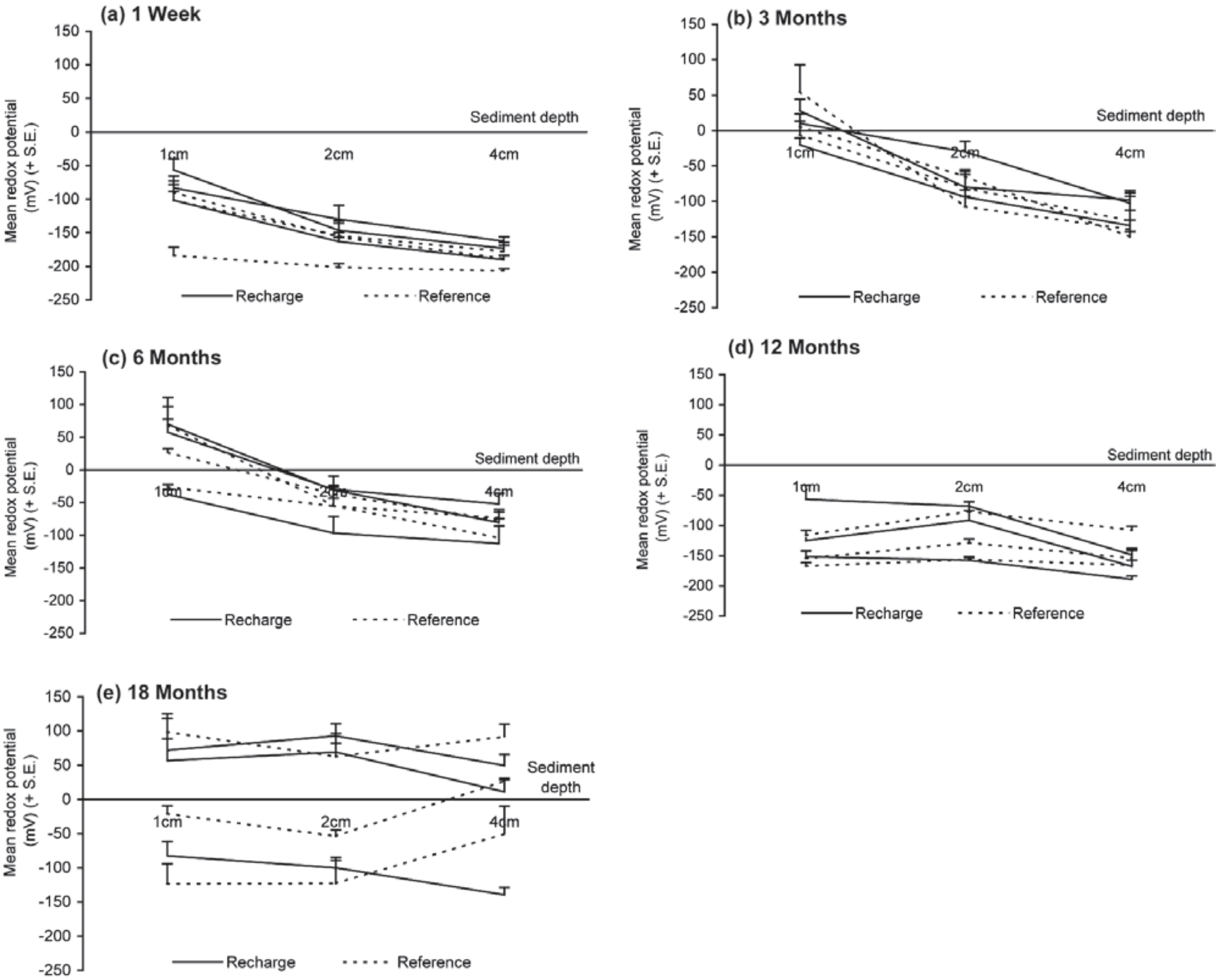

Fig. 2a-e. Mean redox potential values (+ SE) for all recharge and reference stations throughout the sampling period.

material taken during recharge did not contain any macrofauna.

Fig. 3a shows that *T. benedii*, the most abundant species overall, slowly recovered in the recharge stations and only after 18 months did densities reach those of the reference stations and those in the recharge stations prior to recharge. *S. shrubsolii* was much more abundant in the reference stations than at the recharge area prior to recharge. Within one week after recharge, densities found at the recharge stations were comparable to those prior to recharge, although densities always remained lower than those found at the reference stations until 18 months after recharge. In contrast, *C. volutator* was found at very high abundances in the recharge area prior to recharge, between 300-400 per core, yet was rarely present at the reference stations. Their densities only slowly recovered and remained low until 12 months post-recharge. The nematodes recovered relatively rapidly, although densities were very variable both temporally and spatially (between replicates and stations); recovery to reference levels occurred after one week. Finally, *H. diversicolor* slowly increased in abundance within the recharge stations until 12 months post-recharge where densities were noticeably higher than those found at the reference stations.

The mean values of the univariate parameters (with standard errors) for each station from each sampling occasion are displayed in Fig. 4a-e. The results of statistical testing of these results are given in Table 2. These results give an indication of ‘recovery’ of univariate parameters for the recharge stations relative to the reference stations. Total individuals for all recharge stations were not significantly different from any of the reference stations 3 months after recharge although, after 18 months, the numbers of

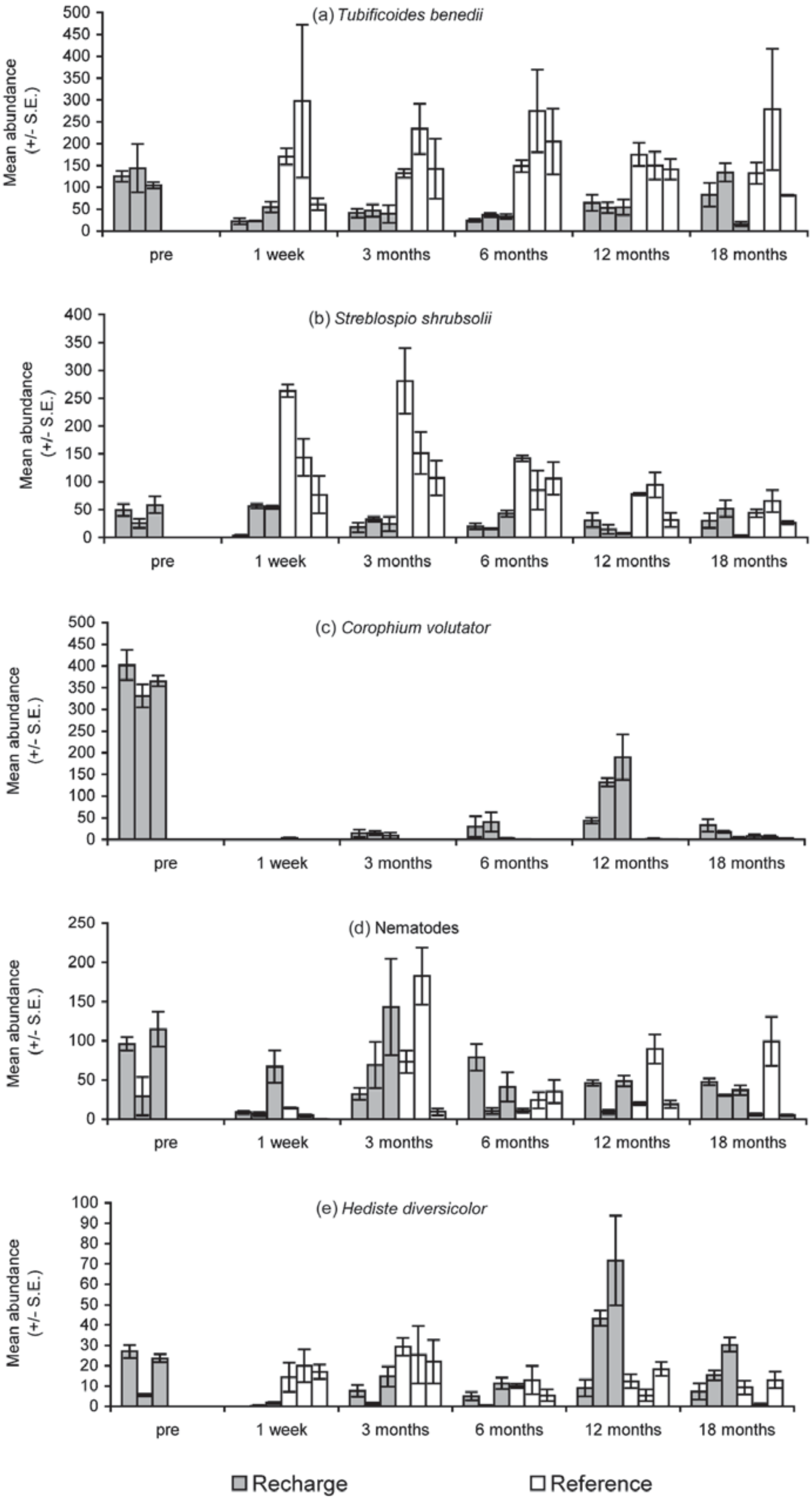

Fig. 3a-e. Mean densities (+/– SE; *n* = 3) per core of the five most abundant taxa.

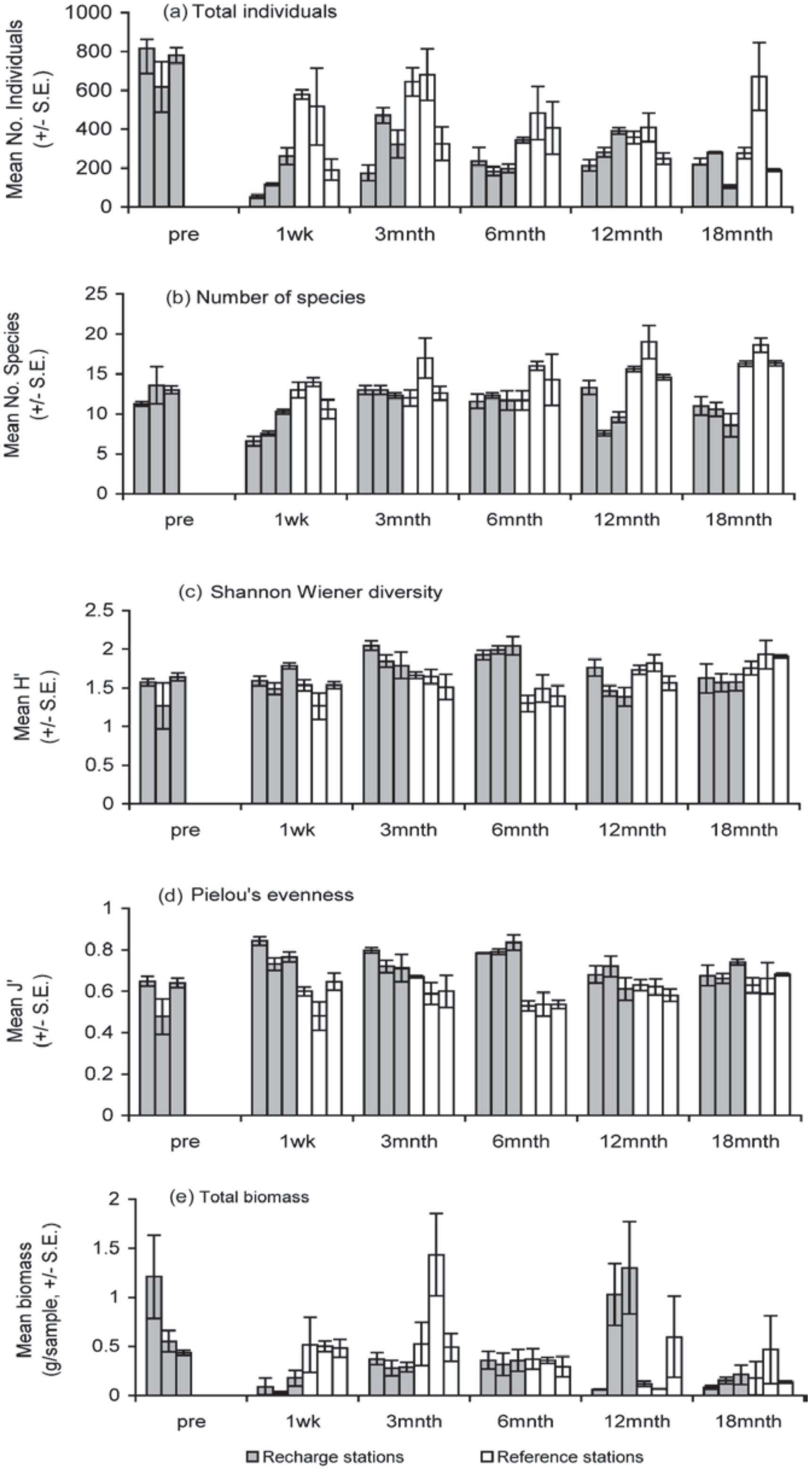

Fig. 4a-e. Mean values (+/– SE; $n = 3$) of the univariate parameters for each station at each sampling time.

Table 2. Results of ANOVA tests with Tukey multiple comparison tests of univariate parameters for each sampling period. Significant differences between each recharge station and reference station are indicated; - means non-significant at $\alpha = 0.05$.

Variable		1 week	3 months	6 months	12 months	18 months
No. individuals	Rech 1	Refs 1 + 2	-	-	Ref 2	Ref 2
	Rech 2	Ref 1	-	-	-	-
	Rech 3	-	-	-	-	Ref 2
No. species	Rech 1	Refs 1+ 2	-	-	-	Refs 1 – 3
	Rech 2	Ref 1	-	-	Refs 1 – 3	Refs 1 – 3
	Rech 3	Ref 2	-	-	-	Refs 1 – 3
Diversity	Rech 1	-	Refs 1 – 3	Refs 1 – 3	-	-
	Rech 2	-	Refs 1 + 2	Refs 1 – 3	-	-
	Rech 3	Ref 2	-	Refs 1 – 3	-	-
Evenness	Rech 1	-	-	-	-	-
	Rech 2	-	-	-	-	-
	Rech 3	-	-	-	-	-
Biomass	Rech 1	-	Ref 2	-	-	-
	Rech 2	-	Ref 2	-	-	-
	Rech 3	-	Ref 2	-	-	-

individuals in reference station 2 increased to give significant differences.

Similarly, the mean number of species were not significantly different after 3 months, although the numbers of species significantly lowered after 12 months in recharge station 2 and in all 3 recharge stations after 18 months. Diversity recovered very quickly: after one week, recharge station 3 had a significantly higher diversity than reference station 2 while, after 6 months, diversity in all the recharge stations were significantly higher than the three reference stations. There was never any significant differences between the evenness values of the recharge and reference stations. Total invertebrate biomass was a very variable parameter within each station resulting in large standard errors and a low power in the statistical tests. Values in the recharge stations were not significantly different from those in the reference stations (except after 3 months when reference station 2 had a noticeable increase in biomass).

Fig. 5a-e shows separate MDS plots for each sampling occasion. These plots indicate firstly that, as recovery proceeded within the recharge stations, they do not appear to have progressed towards a similar community structure to those found in the reference stations. The line drawn to delineate the recharge and reference station replicates emphasizes this point. Secondly, within-station variability is generally less than that between stations, especially for the recharge site. This indicates that the community structure is relatively similar at the replicate (within 1 m^2) scale but more variable between stations. Fig. 6 shows the MDS plot produced when the replicates are combined. This plot allows an assessment of the temporal changes in community structure at all stations. The temporal development of the recharge stations can again be seen to be different from the reference stations. Furthermore, it is evident that although the community structure of the recharge stations prior to recharge are on the same side of the MDS plot as those post recharge, recovery does not progress to a similar community structure. However, 12 months after recharge (i.e., same time of the year) the recharge stations were at their most similar to the pre-recharge sampling, and then less similar after 18 months. This indicates that the difference between the pre and post recharge stations may be primarily due to natural temporal variability.

Table 3 gives the average abundance per core of each taxon from the recharge and reference areas and the percentage contribution of that taxon to the total dissimilarity displayed in two dimensions in Fig. 6 (only those taxa contributing to 80% cumulative dissimilarity are shown). The most abundant taxa (see Fig. 3a-e) tend to be those contributing to the dissimilarity: *Corophium volutator* and nematodes contribute greatly because of their greater abundance in the recharge stations while, for *Tubificoides benedii* and *Streblospio shrubsolii*, their contribution arises from lower abundances at the recharge areas. However, a significant amount of dissimilarity was also due to species with lower abundances such as tellinid bivalves, *Paranais littoralis*, *Hydrobia ulvae* and *Tharyx killariensis*. Consequently, the recharge and reference stations were distinctly separated on the MDS plot partly due to differences in the abundances of a number of relatively rare species which tended to be moderately abundant in either the recharge stations or reference stations, but not both.

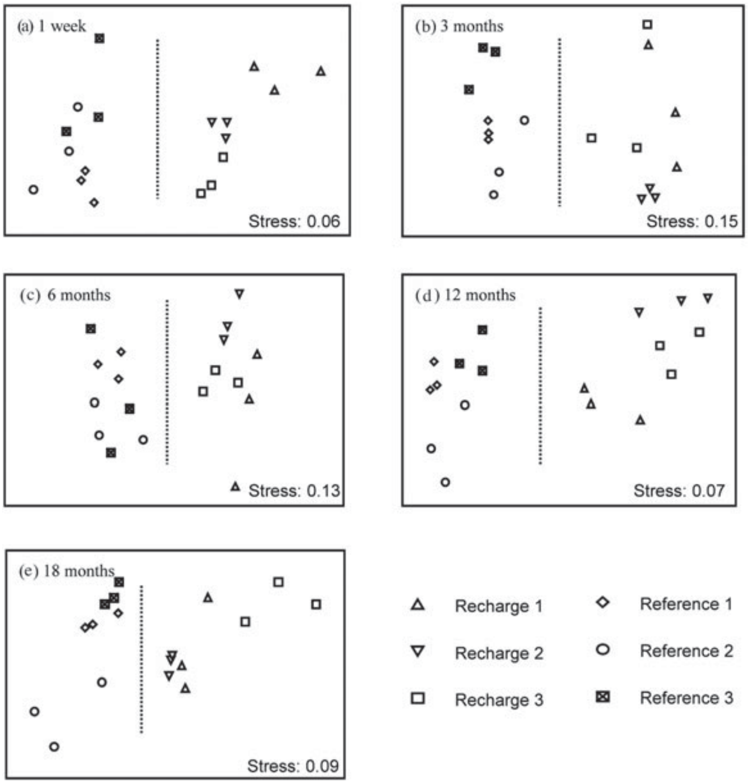

Fig. 5a-e. MDS plots of all replicates from each station at each sampling occasion. Dashed lines separate replicates from the recharge and reference stations. The stress value for each ordination is shown.

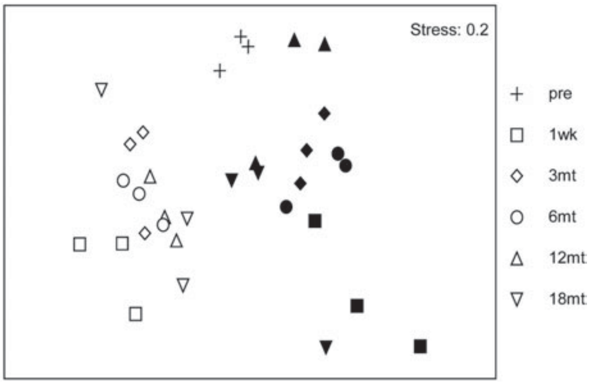

Fig. 6. Multidimensional scaling plot of all stations from all sampling times with replicates combined. Recharge stations are in black while reference stations are in white.

Table 3. Average abundance for recharge and reference stations for the species contributing to 80 % of the observed dissimilarities observed in the MDS plot of Fig. 6. The individual dissimilarity contribution of each species is also given.

	Average abundance		%
	Recharge	Reference	Contribution
Corophium volutator	271.8	4.3	11.6
Tubificoides benedii	182.9	524.7	11.4
Streblospio shrubsolii	90.1	338.9	10.2
Nematoda	152.7	118.6	7.3
Tellinidae	1.1	27.9	4.9
Paranais littoralis	47.8	17.1	4.8
Hydrobia ulvae	8.11	42.5	4.8
Tharyx killariensis	44.4	5.8	4.2
Hediste diversicolor	45.9	43.1	4.1
Manayunkia aestuarina	18.8	26.1	3.9
Tubificoides pseudogaster	43.1	36.0	3.6
Nereididae	1.5	15.4	3.0
Heterochaeta costata	18.1	3.5	2.6
Enchytraidae	10.5	2.6	2.5

Discussion

There are generally two main methods for evaluating recovery of a habitat creation scheme (Callaway et al. 2001). Firstly, recovery can be gauged by assessing total macrofaunal densities and some measure of diversity such as species richness or a diversity index (Levin et al. 1996). Secondly, the functional equivalence between natural and created marshes can be compared. Functional restoration, a requirement of many mitigation and restoration plans, however, has an additional imperative relative to macrofauna. Species composition must be similar, or, alternatively, functionally similar species must have replaced those originally present. In this study the first approach of assessing recovery using attributes of community structure was adopted. The results indicate that, in contrast to those reported in most other studies, invertebrate community recovery of fine-grained beneficial use schemes can occur quite rapidly, i.e. well within a year. Thus, univariate measures of community structure (total individuals, number of species, diversity and evenness) were not significantly different to those found in any of the reference stations within three months after the recharge. Evans et al. (1998) found that the main invertebrate species (*Corophium volutator*, *Hediste diversicolor*, *Hydrobia ulvae*) took over three years to recover in a created mud flat in the Tees Estuary, northeast England. Ray et al. (1994) determined that the abundances of *Nereis virens* and *Mya arenaria* had recovered three years after the placement of dredged material at a constructed mud flat on Sheep Island, Maine, USA. Although the precise rates of colonisation could not be estimated from their data, Ray et al. (1994) stated that a diverse infaunal assemblage was present within two years. Direct comparisons of infaunal recovery rates are difficult as most reported studies differ in respect of factors such as the timing and spatial scale of recharge, amount and type of sediment recharged, and in the chosen methods of recovery-assessment (e.g., the abundance of important species, uni- or multivariate data analysis, and so on). The relatively small scale of the scheme at Westwick Marina, allowing enhanced post-juvenile immigration (see below) and the rapid de-watering afforded by the wooden fences used to retain the dredged material, may have contributed to the rapid recolonization observed.

In this study, recovery of a parameter was assessed by comparison with a set of reference sites. The large temporal variability inherent in macrofaunal abundances makes comparison with reference sites sampled simultaneously to the disturbed area a prerequisite for effective interpretation of events (Green 1979). At Westwick Marina, macrofaunal species abundances and community structure greatly varied at the spatial scale between the recharge and reference stations, i.e., the recharge stations prior to recharge were separated from the reference stations on the MDS plot. This was in part due to the heterogenous distribution of *Corophium volutator* which was very abundant at the recharge stations (both prior to and after recharge) yet rarely sampled at the reference sites. Furthermore, the recharge event inherently raised the tidal elevation of the recharge stations; as the eventual tidal height of the surface of the recharge area was between the salt marsh surface and channels, it was not possible to locate reference stations with comparable tidal heights. Consequently, in addition to differences due to spatial variability, the recharge stations were unlikely to attain a comparable community structure to those of the reference stations by virtue of tidal height differences. Although invertebrate data existed for the area prior to recharge, these are of limited value with respect to recovery assessment as it follows that there can be no temporal continuity on account of smothering of the original habitat. We propose that in these situations where suitable reference stations cannot be found for a beneficial use scheme, a critical assessment of the quality of the habitat must be made, and the use of reference sites must be made with caution.

When maintenance dredged material is recharged onto an intertidal area during a beneficial use scheme, the invertebrate community may develop via four possible recovery processes (Fig. 7). Firstly, invertebrates can be transferred from the dredged area to the recharge area with the dredged material (direct transfer). For this to be a successful recovery mechanism the organisms must survive the dredging process, the high-pressure pumping through the pipeline and, finally, the eventual deposition. The results from the dredged material samples from this study indicated that this was an impossible mechanism as no invertebrates were found either as a result of the 'dilution' (the majority of the dredged material would have been below the zoic depth) or the

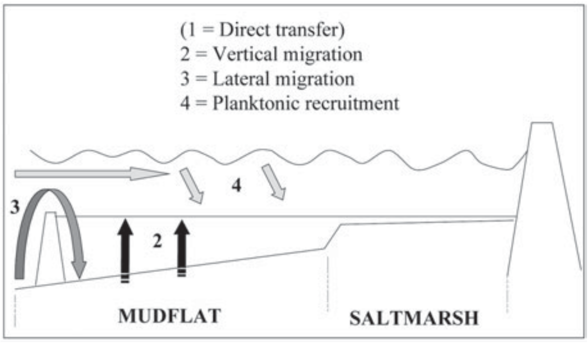

Fig. 7. Macrofaunal recovery mechanisms following intertidal placement of dredged material (from Bolam et al. 2003).

destruction of organisms through the dredging process. Secondly, *in situ* invertebrates may vertically migrate through the deposited material to regain their position in the upper sediment layer. A number of studies have been conducted (Shulenberger 1970; Maurer et al. 1981, 1982; Roberts et al. 1998; Essink 1999) on the ability of macrofauna to vertically migrate; however, many of them have focused on sandy-sediment species and are therefore unlikely to accurately reflect the capabilities of species within the present mud flat community. Of the few studies on muddy-habitat organisms, results indicate that such species are, unlike those of sandy habitats, unable to migrate through more than 10 cm of sediments (Saila et al. 1972, cited by Morton 1977; Chandrasekara & Frid 1998; Essink 1999; Bolam et al. 2003). This implies that vertical migration was not a possible recovery mechanism in the present study where approximately 50 cm of sediment was deposited during the recharge process. However, caution must be exercised when generalising from the results of such experimental, 'one-off' placements of dredged material to large-scale beneficial use schemes. For example, the recharge process at Westwick Marina continued for several weeks to achieve the final sediment depth on account of the high-water content of the dredged material and the behaviour of the invertebrates throughout this period is unknown.

The third possible recovery mechanism is the active migration and settlement of post-juvenile individuals following placement. Many estuarine invertebrate species are active migrators either via swimming in the water column (Levin 1984a; Armonies 1988, 1994; Cummings et al. 1995) or crawling across the sediment surface (Levin 1984b; Smith & Brumsickle 1989; Wilson 1992, 1994). Such active migration of post-juveniles has been shown to be an important colonisation mechanism at newly-deposited or recently disturbed sediments. The fourth, and potentially the most important, recovery mechanism is planktonic settlement of larvae. Most estuarine invertebrates reproduce via the liberation of planktonic larvae which develop in the water column and settle when ready to metamorphose. This can result in dramatic increases in abundance (Bolam 1999; Bolam & Fernandes 2002), especially if the sediments have low abundances of residents such as the situation found after dredged material disposal. However, this tends to be very seasonal, usually restricted to late spring or early summer in temperate regions. There may be an interaction between these last two recovery mechanisms. For example, Smith & Brumsickle (1989) and Shull (1997) have suggested that adult immigration to disturbed sediments might assume a greater significance during periods of decreased larval availability. Determining the relative importance of these two mechanisms for a particular scheme involves an assessment of the size distribution of the early recolonizing species: early post-larval recruits from the plankton will clearly result in a smaller size spectrum than that produced from active migration of more mature adults. Since size distributions of the major recolonizing species were not assessed in this study, the primary recovery mechanism at Westwick Marina cannot unequivocally be determined. However, there is evidence to suggest that post-juvenile immigration was the predominant recovery mechanism at the recharge stations. For example, the recharge was completed during August after the main planktonic recruitment phase and biomass recovered as rapidly as the total number of individuals. As there was no time lag in the biomass recovery it implies that these colonising individuals were adults. The continued increase in the numbers of individuals after three and six months (before the next planktonic recruitment phase) in the recharge stations supports this. Furthermore, tubificid oligochaetes, which lack a planktonic recruitment phase, colonised from the first week and densities had recovered within 18 months.

Acknowledgements. This work was funded by the Science Directorate of the Department for Environment, Food and Rural Affairs (project code AE0231). An earlier version of this manuscript was greatly improved following comments made by Dr. Hubert Rees. We would like to thank Nigel Proctor for help with sample analysis.

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Received 9 May 2003;
Accepted 16 December 2003.
Co-ordinating Editor: A.T. Williams.