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Short-Term Benthic Recolonization after Dredging in the Harbour of Ceuta, North Africa

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With 6 figures and 2 tables

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Abstract. The benthic recovery after dredging (area: 2625 m^2) was studied in a polluted and enclosed area of the harbour of Ceuta, in which the recolonization through the water column (larvae and adult bedload transport) could be limited by the lack of renewal. The benthos was sampled at two sites (control and dredged) using a van Veen grab and adopting a BACI (Before, After, Control, Impacted) approach. Five samplings were conducted after dredging (3, 15, 30, 90, 180 days). The proportion of gravel in the sediment of the dredged site increased after dredging, while the organic matter decreased. The impact on the community was estimated at species level, using both univariate and multivariate analyses. The maximum negative effect on benthic macrofauna was a reduction by 65% for species richness (15 days after dredging) and by 75% for abundance (3 days after dredging). Between 15 and 30 days after dredging, the abundance of some species such as the molluscs *Parvicardium exiguum* and *Retusa obtusa* and the polychaete *Pseudomalacoceros tridentata* increased considerably in the dredged site, while typical 'opportunistic' species such as *Capitella capitata* were disfavoured by the disturbance. For this small-scale dredging, about 6 months are required for the disturbed area to re-establish a sediment structure and a macrobenthic community similar to the undisturbed area. Small-patch dredging operations are proposed in harbour management whenever possible, since they allow a quick re-adjustment of the initial sediment structure and benthic communities.

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Problem

Dredging operations in harbours are long-established human-induced disturbances in the marine environment. These anthropic perturbations usually drastically reduce the benthic population and temporarily change the environmental abiotic features (Pranovi *et al.*, 1998). Although the effect of dredging and the dredged material disposal have been relatively well-documented (López-Jamar & Mejuto, 1988; Kenny & Rees, 1996; Giovanardi *et al.*, 1998; Zajac *et al.*, 1998; Lewis *et al.*, 2001), the subsequent recolonization is a site-specific process, with both time and spatial scale involved and dependent upon local hydrodynamic and sedimentary conditions (Zajac *et al.*, 1998; De Grave & Whitaker, 1999).

As was pointed out by Kenny & Rees (1996), even though the literature about dredging effects is abundant, in most of the cases the exact location, duration and magnitude of the dredging is not well known. Furthermore, the existence of control sites with similar sediment characteristics, depth and benthic community to the dredge sites is not always easy to find, and this restricts the accuracy of many studies.

In this paper, we examine the impact of a dredging event of small scale (area: 2625 m^2) on the benthic community and sediment characteristics, adopting a BACI (Before, After, Control, Impacted) approach. We intend to evaluate the effectiveness of recovery in the most enclosed area of the harbour, where water movement is limited.

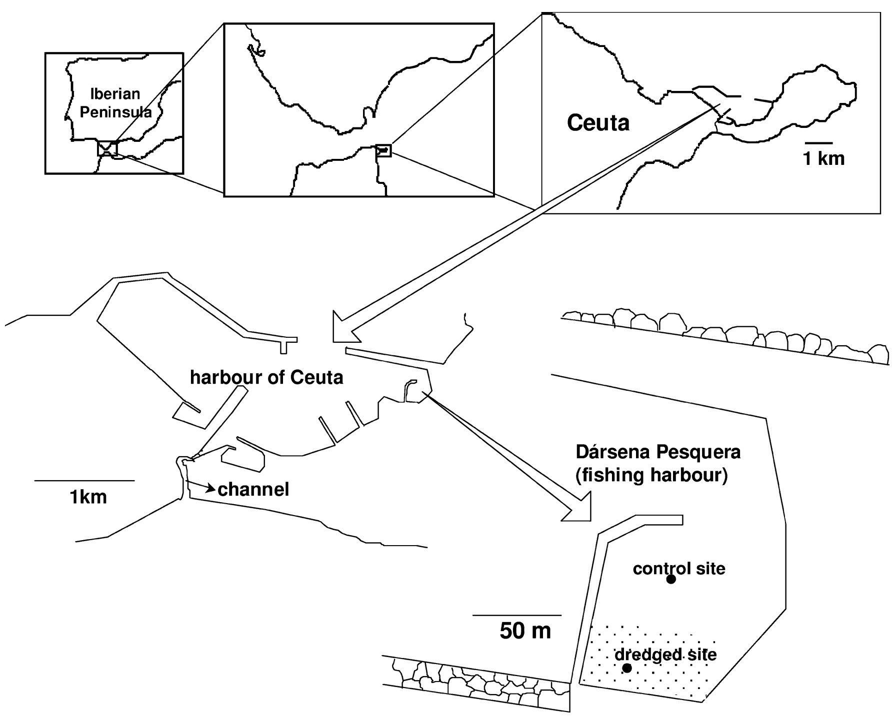

Fig. 1. Map of the study area showing the location of the Dársena Pesquera in the harbour of Ceuta, North Africa.

Material and Methods

The studied area, Dársena Pesquera, is an enclosed fishing bay located in the harbour of Ceuta, a Spanish enclave in North Africa, Gibraltar area (35°53′N; 5°18′W) (Fig. 1). The harbour of Ceuta is one of the most important harbours in the Strait of Gibraltar, being the entrance from Europe to Africa. It is characterized by intense shipping traffic, and frequent loading and dumping is involved in shipping operations. The depth of the harbour ranges between 3 and 16 m, and in the Dársena Pesquera the depth is between 3 and 3.5 m. The harbour is located between two bays connected by a channel (Fig. 1) which increases the water movement across the middle of the harbour (Guerra-García, 2001). However, the Dársena Pesquera is the most enclosed area and the water renewal is very reduced here because it is not affected by the currents north-south across the harbour (Guerra-García, 2001).

During 2 weeks in June 1999, a small area of 75 × 35 m was dredged using a trailing suction hopper dredge to maintain a navigable depth for small boats and ships. Two stations were selected, the dredged site located in the centre of the dredged area, and the control site outside the perturbed area about 70 m away from the dredged site (Fig. 1). Both stations, about 3 m deep, were sampled 1 week before the dredging operations to check if the sediment characteristics and benthic community were similar and a BACI approach could be adequately used. Five more samplings were conducted in both stations 3, 15, 30, 90 and 180 days after dredging.

A van Veen grab of 0.05 m^2 was used for sampling; sediment penetration was never less than 20 cm deep (see López-Jamar & Mejuto, 1988). On each sampling, four van Veen grab samples were obtained at each station. Three samples were allocated to the study of the macrofauna (0.15 m^2 of total area) and the fourth was used for the sediment analysis (granulometry and organic matter percentage). The macrofaunal samples were processed through a sieve with a mesh size of 0.5 mm and the retained fraction was fixed in 4% neutral formalin stained with Rose Bengal. Organisms were sorted out of the residue by eye (see de Grave & Whitaker, 1999), identified to species level, and counted. The organic content was analysed by ashing to 500 °C samples of sediment previously dried at 100 °C over 24 h. Granulometry was determined by Buchanan and Kain's method (Buchanan & Kain, 1984).

Univariate analyses provided the total number of species, total abundance, Shannon–Wiener diversity and Pielou's evenness indices (Shannon & Weaver, 1963; Pielou, 1966), and the Margalef index (Margalef, 1958). Using the values of the replicates (0.05 m^2), the possible variations of these community descriptors were tested with one-way ANOVA, after verifying normality using the Kolmogorov–Smirnov test and Bartlett test for homogeneity of variances. Homogeneous groups were separated by the Tukey test. The similarity among samples was calculated using the Bray–Curtis similarity index (Bray & Curtis, 1957), and an MDS analysis (non-metric multidimensional scaling) was used with the matrix of species abundance. The goodness of the ranking was tested using the stress coefficient of Kruskal (Kruskal & Wish, 1978), and the abundance data were double square root transformed so that the ensuing ordination was not determined solely by the most dominant species (Clarke & Green, 1988; Guerra-García & García-Gómez, 2001). To test possible differences in the species abundances between the dredged and the control sites at different times, the two-way crossed ANOSIM was selected. The Plymouth Routines In Multivariate Ecological Research (PRIMER) computer package v. 5 (Clarke & Gorley, 2001) and the program PC-ORD v. 3.05 (McCune & Mefford, 1997) were used for multivariate analyses. For univariate analyses the BMDP was used (Dixon, 1983).

Results

1. Sediment structure

Both stations – the dredged and control sites – showed similar granulometry before dredging, being dominated by the fine fraction (silt, clay and fine–very fine sand) (Fig. 2). While the sediment structure remained constant in the control site, the content of gravel and gross–very gross sand increased considerably in the dredged station just after dredging. Nevertheless, the gravel percentage progressively decreased during the

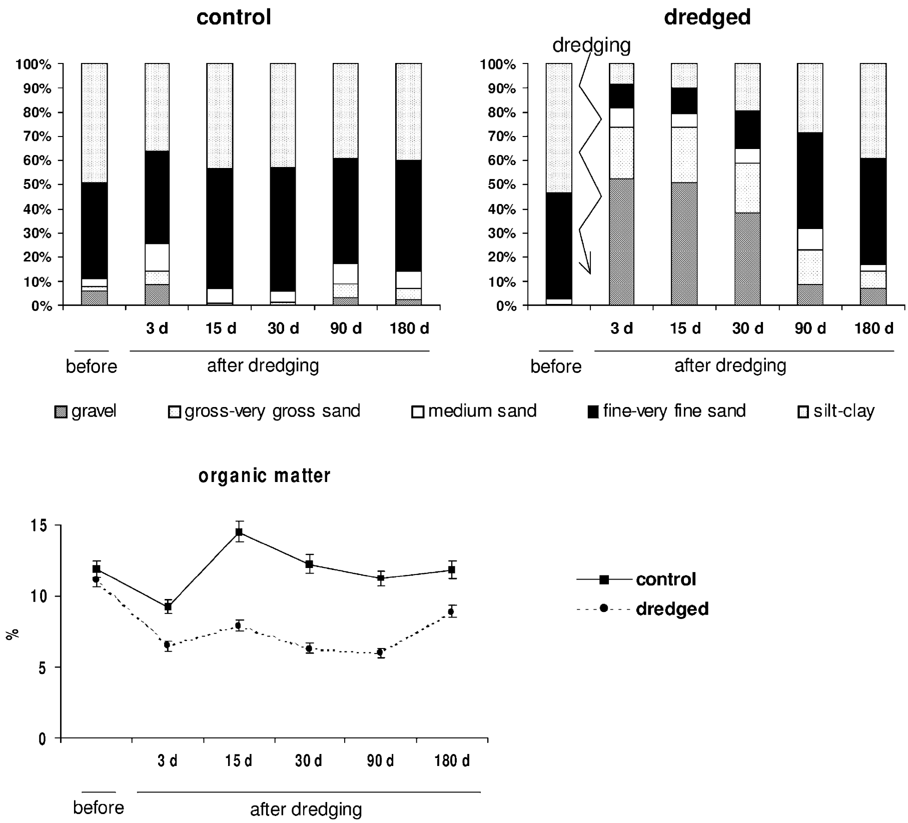

Fig. 2. Sediment granulometry and organic matter measured before the dredging and 3, 15, 30, 90, 180 days after the disturbance in the control and the dredged site. Gravel: Me (median grain diameter) > 2 mm; gross–very gross sand: 0.5 mm > Me > 2 mm; medium sand: 0.25 mm > Me > 0.5 mm; fine–very fine sand: 0.065 mm > Me > 0.25 mm; silt–clay: Me < 0.065 mm. For the organic matter, the mean values of three replicates and the standard errors are included.

post-dredging period, reaching similar values to the initial sampling. The organic content at the dredged site decreased considerably from 11.14% to 6.48% after dredging (Fig. 2).

2. Temporal variation of the macrobenthos

Table 1 indicates the abundance [indiv. $\cdot$ $(0.05\ m^2)^{-1}$] of the species in both stations (dredged and control) during the study period. The sampling provided a total of 37 species, of which 14 were crustaceans, 13 polychaetes and 10 molluscs. The number of species and abundance decreased after dredging (Fig. 3); the maximum impact on benthic macrofauna was a reduction of the abundance by 75% (3 days after dredging) and of species richness by 65% (15 days after dredging). Fifteen days after dredging, the richness, abundance and the Margalef index began to recover. In contrast, the diversity and the evenness indices started to re-adjust 30 days after dredging

Table 1. Species recorded during the study and their abundance [indiv. $\cdot$ $(0.05\ m^2)^{-1}$] in the control site (C) and the dredged site (D) before dredging and 3, 15, 30, 90 and 180 days after the disturbance.

	before dredging		after dredging									
			3 d		15 d		30 d		90 d		180 d	
	C	D	C	D	C	D	C	D	C	D	C	D
Polychaeta												
Anaitides madeirensis (Langerhans, 1880)	–	–	–	–	–	–	–	–	0.3 (0.3)	–	–	–
Capitella capitata (Fabricius, 1780)	20.3 (11.8)	49.3 (21.5)	4.7 (3.8)	21.7 (6.4)	9.0 (3.1)	11.3 (4.8)	6.3 (2.2)	2.0 (0.0)	–	0.3 (0.3)	–	53.0 (33.1)
Cirriformia tentaculata (Montagu, 1808)	0.3 (0.3)	–	7.7 (5.1)	–	0.3 (0.3)	–	0.3 (0.3)	–	5.3 (2.3)	–	8.7 (4.8)	–
Ehlersia ferruginea (Langerhans, 1881)	–	–	1.7 (0.3)	–	–	–	–	–	–	–	0.3 (0.3)	–
Lumbrinereis latreilli (Audouin & Milne-Edwards, 1834)	–	–	3.3 (1.1)	–	0.3 (0.3)	–	0.3 (0.3)	–	0.3 (0.3)	–	3.3 (1.9)	–
Nephthys cirrosa (Ehlers, 1868)	–	–	–	–	–	–	0.3 (0.3)	–	–	0.7 (0.3)	–	0.3 (0.3)
Nereis falsa (Quatrefages, 1865)	2.0 (0.9)	0.7 (0.3)	4.0 (1.4)	1.3 (0.7)	8.0 (2.1)	0.7 (0.3)	2.3 (1.2)	0.3 (0.3)	3.3 (0.7)	5.3 (3.5)	6.4 (1.4)	2.3 (0.3)
Platynereis dumerilii (Audouin & Milne-Edwards, 1833)	–	7.0 (4.9)	9.7 (5.9)	–	0.3 (0.3)	–	–	–	0.3 (0.3)	1.0 (0.8)	10.3 (4.5)	0.3 (0.3)
Potamilla reniformis (Linnaeus, 1788)	–	–	5.7 (1.9)	–	0.3 (0.3)	–	0.3 (0.3)	–	2.0 (0.5)	–	1.7 (0.7)	–
Pseudomalacoceros tridentata (Southern, 1914)	68.3 (16.7)	56.0 (27.3)	99.7 (41.6)	2.0 (2.5)	192.3 (29.8)	122.0 (19.2)	107.7 (41.9)	1200.0 (282.8)	71.7 (5.9)	1033.3 (422.5)	84.7 (46.6)	76.7 (18.2)
Pseudopolydora antennata (Claparede, 1868)	1.0 (0.5)	0.3 (0.3)	0.3 (0.3)	6.3 (2.7)	1.0 (0.5)	–	3.0 (2.0)	–	–	0.7 (0.3)	–	0.3 (0.3)
Scolelepis fuliginosa (Claparede, 1870)	–	–	–	–	–	–	–	0.3 (0.3)	–	–	–	–
Sphaerosyllis pirifera (Claparede, 1868)	1.0 (0.5)	–	0.3 (0.3)	–	0.3 (0.3)	–	–	–	0.7 (0.3)	–	–	–

Table 1. Continued.

	before dredging		after dredging									
			3 d		15 d		30 d		90 d		180 d	
	C	D	C	D	C	D	C	D	C	D	C	D
Crustacea												
Anthura gracilis (Montagu, 1808)	0.3 (0.3)	–	4.0 (1.7)	–	0.3 (0.3)	–	3.3 (0.7)	–	4.0 (0.8)	–	1.7 (1.4)	–
Aora spinicornis (Afonso, 1976)	12.6 (5.1)	1.0 (0.8)	9.7 (1.5)	0.3 (0.3)	–	–	–	2.3 (0.7)	–	3.3 (0.9)	7.0 (2.6)	2.7 (0.5)
Apseudes latreilli (Milne-Edwards, 1828)	37.0 (25.5)	3.3 (1.4)	104.6 (40.6)	2.3 (1.2)	54.3 (40.7)	0.3 (0.3)	37.3 (8.1)	1.3 (0.3)	92.7 (16.0)	0.7 (0.5)	54.0 (16.7)	5.0 (2.3)
Corophium runcicorne (Della Valle, 1893)	1.0 (0.0)	1.3 (0.1)	2.3 (1.9)	–	0.7 (0.3)	3.0 (0.5)	1.3 (0.7)	0.3 (0.3)	1.3 (0.5)	–	4.3 (2.4)	0.3 (0.7)
Corophium sextonae (Crawford, 1937)	0.3 (0.3)	0.3 (0.3)	1.7 (1.4)	0.3 (0.3)	2.0 (0.8)	–	3.0 (1.4)	1.0 (0.5)	0.7 (0.3)	3.3 (0.3)	1.0 (0.5)	1.0 (0.5)
Dexamine spinosa (Montagu, 1813)	3.7 (0.6)	0.3 (0.3)	7.3 (0.9)	–	0.3 (0.3)	–	0.3 (0.3)	1.3 (0.3)	–	1.3 (0.3)	1.0 (0.8)	4.0 (1.2)
Gnathia sp.	–	–	–	–	–	–	–	–	–	0.3 (0.3)	–	0.3 (0.3)
Leucothoe lilljeborgi (Boeck, 1861)	0.3 (0.3)	1.0 (0.5)	3.7 (2.2)	–	0.3 (0.3)	–	1.0 (0.8)	–	1.7 (0.7)	–	–	–
Nebalia bipes (Fabricius, 1780)	0.3 (0.3)	–	–	0.3 (0.3)	–	–	–	–	–	0.7 (0.5)	0.7 (0.5)	–
Pariambus typicus (Kröyer, 1844)	0.7 (0.5)	6.0 (1.6)	39.7 (3.7)	2.3 (0.7)	2.3 (1.5)	2.7 (0.9)	2.7 (1.1)	17.3 (3.8)	31.7 (12.9)	4.3 (1.7)	19.0 (9.2)	2.7 (0.9)
Perioculodes longimanus (Bate & Westwood, 1868)	–	–	1.3 (0.7)	–	–	–	0.3 (0.3)	–	–	–	0.7 (0.5)	–
Phtisica marina (Slabber, 1769)	9.0 (2.3)	1.3 (0.3)	12.3 (6.0)	–	0.3 (0.3)	–	0.3 (0.3)	9.0 (4.6)	0.7 (0.3)	2.3 (0.7)	1.0 (0.8)	1.7 (0.3)
Stenothoe sp.	–	–	–	–	–	–	–	1.0 (0.4)	–	1.0 (0.8)	–	–
Zeuzo normani (Richardson, 1905)	–	–	–	–	–	–	–	–	1.3 (0.7)	0.3 (0.3)	3.3 (2.7)	0.3 (0.3)

Table 1. Continued.

	before dredging		after dredging									
			3 d		15 d		30 d		90 d		180 d	
	C	D	C	D	C	D	C	D	C	D	C	D
Mollusca												
Abra alba (Wood, 1802)	–	–	0.7 (0.5)	–	–	–	0.3 (0.3)	–	0.7 (0.3)	–	0.3 (0.3)	–
Calyptraea chinensis (Linnaeus, 1758)	–	–	–	–	–	–	–	–	0.3 (0.3)	–	0.3 (0.3)	–
Chamelea striatula (Da Costa, 1778)	–	–	0.3 (0.3)	–	–	–	–	–	0.7 (0.3)	0.3 (0.3)	–	–
Corbula gibba (Olivi, 1792)	0.3 (0.3)	–	–	–	–	–	–	–	–	–	–	–
Cyclope neritea (Linnaeus, 1758)	0.3 (0.3)	1.0 (0.4)	–	–	–	–	–	–	–	–	–	–
Nassarius incrassatus (Stroem, 1768)	–	0.3 (0.3)	–	–	–	–	–	–	–	0.7 (0.3)	–	–
Parvicardium exiguum (Gmelin, 1791)	0.7 (0.3)	3.0 (0.0)	1.3 (0.3)	–	1.7 (0.7)	–	0.7 (0.3)	7.7 (3.2)	3.7 (2.2)	6.7 (0.7)	1.3 (0.5)	3.0 (0.8)
Retusa obtusa (Montagu, 1803)	4.7 (1.5)	15.7 (4.4)	11.7 (4.9)	0.3 (0.3)	2.7 (1.8)	–	2.0 (0.9)	24.7 (3.0)	3.0 (0.9)	17 (2.2)	2.7 (0.7)	12.3 (1.9)
Rissoa similis (Scacchi, 1836)	–	2.0 (1.6)	–	–	–	0.3 (0.3)	–	–	–	1.3 (1.1)	–	1.3 (0.3)
Tellina nitida (Poli, 1791)	0.3 (0.3)	0.3 (0.3)	–	–	–	–	–	–	0.3 (0.3)	–	–	–

Data are averages of three replicates. Standard errors are included in parentheses.

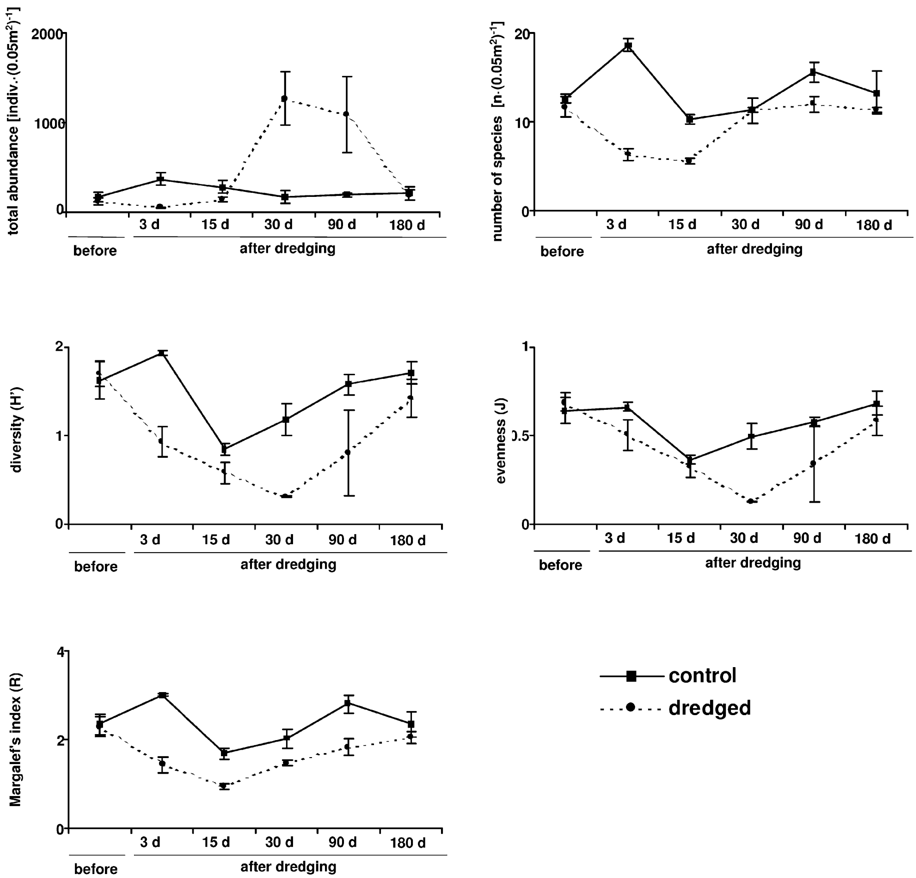

Fig. 3. Trend of the total abundance, species number, Shannon–Wiener diversity (H′), Pielou evenness (J) and the Margalef index (R) before and after dredging (3, 15, 30, 90, 180 days) in the control and dredged sites. Mean values and standard errors are included.

(Fig. 3). Between 15 and 30 days after the disturbance, the abundance of some species, such as the molluscs *Parvicardium exiguum, Retusa obtusa*, and specially the polychaete *Pseudomalacoceros tridentata*, strongly increased in the dredged site, while the typical 'opportunistic' species such as the polychaete *Capitella capitata* was not favoured by the dredging (Fig. 4).

The results of the one-way ANOVA among samples (Table 2) showed significant differences for all variables, but grouping according to the Tukey tests varied depending on the variable considered. Thus, for diversity and evenness indices, the dredged site differed from the control site at 15, 30 and 60 days after dredging, whereas no differences were found just after the dredging (3 days) and 6 months later. For the species richness, the differences were found 3 and 15 days after dredging. Regardless of the parameter considered, the differences between the dredged and control sites 6 months after dredging were not significant.

When the ANOSIM was used to compare the data of the control and dredged sites at each sampling date, significant differences ($r > 0.9$, $P < 0.05$) were found in all

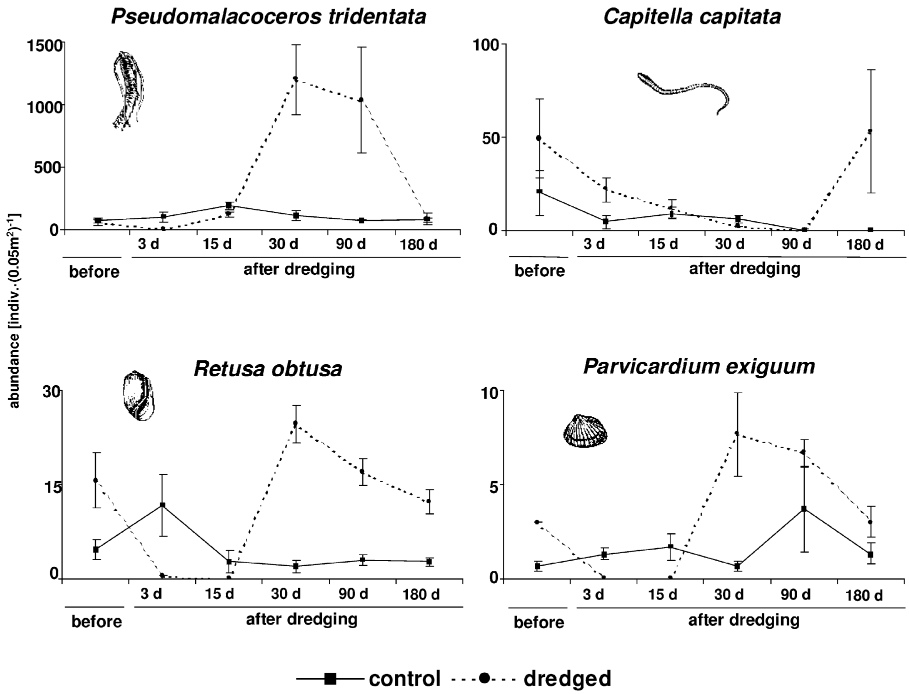

Fig. 4. Trend of the abundance [individuals · (0.05 m^2)$^{-1}$] of *Pseudomalaceros tridentata, Capitella capitata, Retusa obtusa* and *Parvicardium exiguum* before and after dredging (3, 15, 30, 90, 180 days) in the control and dredged sites. Mean values and standard errors are included.

Table 2. Results of the one-way ANOVA for the data of abundance [indiv. · (0.05 m^2)$^{-1}$], number of species, diversity (H′), evenness (J) and Margalef's index (R).

	df	F	homogeneous groups	
abundance	11, 35	4.35**	B(C) B(D) 3(C) 3(D) 15(C) 15(D) 30(C) 90(C) 180(C) 180(D)	30(D) 90(D)
number of species	11, 35	7.55***	B(C) B(D) 3(C) 15(C) 30(C) 30(D) 90(C) 90(D) 180(C) 180(D)	3(D) 15(D)
diversity (H′)	11, 35	4.52**	B(C) B(D) 3(C) 3(D) 15(C) 30(C) 90(C) 180(C) 180(D)	15(D) 30(D) 90(D)
evenness (J)	11, 35	2.91*	B(C) B(D) 3(C) 3(D) 15(C) 30(C) 90(C) 180(C) 180(D)	15(D) 30(D) 90(D)
Margalef's index (R)	11, 35	7.33***	B(C) B(D) 3(C) 15(C) 30(C) 90(C) 90(D) 180(C) 180(D)	3(D) 15(D) 30(D)

df: degrees of freedom; F: statistic; *$P < 0.05$, **$P < 0.01$, ***$P < 0.001$. The homogeneous groups according to the Tukey test ($P < 0.05$) are separated by a vertical line. B: before dredging; 3, 15, 30, 90, 180: days after dredging; (C): control site; (D): dredged site.

samplings (3, 15, 30 and 90 days after dredging), except for 180 days ($r < 0.5$, $P > 0.05$). The same pattern was obtained by the MDS plot of Fig. 5, where the control site samples are grouped with the dredged site samples taken before the dredging and 180 days after the disturbance. This indicates that both sites were biologically similar before dredging and that after 6 months the benthic community is again similar to the pre-dredged community.

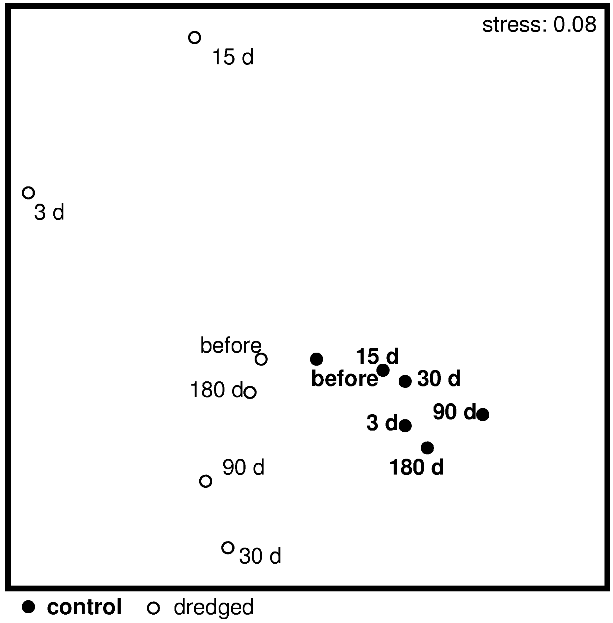

Fig. 5. MDS ordination of the samples according to the average values of species abundances [indiv. · (0.05 m^2 $)^{-1}$] before dredging and 3, 15, 30, 90 and 180 days (d) after the disturbance.

Discussion

The main physical effect of the dredging operations carried out in the Dársena Pesquera of the harbour of Ceuta was the change of the sediment structure, registered as an increase in the percentage of gross–very gross sand and gravel, and a decrease in the organic content. A similar development after dredging has been reported by López-Jamar & Mejuto (1988) during a study carried out in La Coruña Bay. In connection with the macrobenthic community, the present study has shown that the main immediate effect of dredging is the elimination of many of the macrofaunal organisms. According to the review of Newell *et al.* (1998), dredging can be expected to reduce species diversity by 30–70% and the number of individuals by 40–95%. In the present study, natural recolonization processes proceeded rapidly after dredging, and only about 6 months were required for the disturbed areas to re-establish a sediment composition and a macrobenthic community structure similar to those in the control (undisturbed) area.

The recovery time of macrobenthic communities after this type of disturbance depends on the spatial scale, the hydrodynamic conditions, the bottom grain size and the structure of the community affected by the disturbance (Kaiser & Spencer, 1996; Pranovi *et al.*, 1998). Zajac *et al.* (1998) pointed out that the relative combination of factors controlling recolonization and successional process (environmental conditions, life history and population processes, and biotic interactions) may be very different depending on the spatial scales. Despite the difficulty to draw a generalized pattern, the recovery time was considered proportional to the spatial scale of the dredging area (Zajac *et al.*, 1998). Figure 6 shows recovery times after disturbance of various scales. For small-scale dredging of less than 1 m^2, the recovery is very fast (days/weeks)

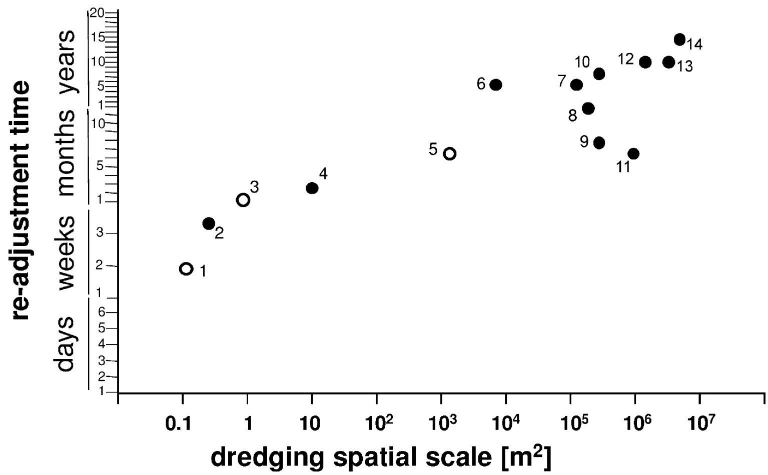

Fig. 6. Representation of the re-adjustment or recovery time over different spatial scales. [1,3] Guerra-García, unpublished data; [2] Cruz-Mota & Bone, unpubl. data; [4] Dernie *et al.*, 2003; [5] present study; [6] Harvey *et al.*, 1998; [7] Kenny & Rees, 1996; [8] López-Jamar & Mejuto, 1988; [9] Sánchez-Moyano, unpublished data; [10,13,14] van Dalfsen *et al.*, 2000; [11] Pranovi & Giovanardi, 1994; [12] Desprez, 2000. (open circles: data from authors of this study)

through two sources of colonists: larvae and/or adults transported from the water column, and post-settlement movement of juvenile and/or adult life-stages across the sediment. When the disturbed area increases, the larger patches possess a smaller edge/surface area ratio; in the central sites of the dredged area the recolonization is only influenced by the water column, and the recovery time increases. Considering that the study area is located in the most enclosed zone of the harbour, the colonization through the water column (larvae and/or adults) is probably limited by lack of water movement. Here the post-settlement movement of species across the sediment could be the key recolonization process. However, the species which increased most in the dredged area were not found in such large abundances in the control area, neither before nor after the dredging; therefore, recolonization could also be due to larval settlement, especially given the time of the year that the research was conducted. Although a substantial literature exists on the effects of dredging (see Newell *et al.*, 1998; Desprez, 2000), only few investigations have followed a BACI approach and estimated the recovery time based on a spatio-temporal study in control and dredged sites. Although more studies on the recovery process after dredging are necessary, especially at small spatial scales, there is a general pattern of increasing recovery times with the spatial scale of the dredging (Fig. 6). Newell *et al.* (1998) used literature data to suggest that recovery times also depend directly on the type of sediment. They estimated a time of 6–8 months for mud habitats and 2–3 years for communities of sand and gravels, but their review includes no data on spatial scale size.

In the present study, although the number of species start recovering 15 days after dredging, the diversity and evenness indices continue declining until 30 days after dredging. This is due to the strong increase in the abundance of some species such as the opportunistic spionid polychaete *P. tridentata*. When areas are depopulated through dredging operations, some opportunistic species usually have a good chance of building up large populations in such 'open spaces'. This short-term change reflected by the abundance increase of opportunistic species has been reported in other studies (Grassle & Sanders, 1973; López-Jamar & Mejuto, 1988; van Dalfsen *et al*., 2000). The establishment of a highly dense population at the initial post-dredging stages is normally followed by a high mortality resulting from severe intra-specific competition (Grassle & Grassle, 1974). This was exactly the behaviour of *P. tridentata* in the present study; after a marked peak of abundance 30 days after the disturbance, the population decreased from 90 days after dredging. Some other polychaetes of the family Spionidae have also been reported to be good initial colonizers after sediment disturbances (Lu & Wu, 1998). The polychaete *C. capitata* is also considered to be one of the global opportunistic species in disturbed marine sediment (Grassle & Grassle, 1974; McCall, 1977; Pearson & Rosenberg, 1978; Estacio *et al*., 1997; Newell *et al*., 1998). However, although this species was abundant before dredging in this study, it was not favoured by the disturbance. This unexpected result has also been recorded by Lu & Wu (1998). Several reasons could be suggested: (1) the dominance in the initial phase of recolonization should be determined by the availability of benthic larvae at the time the habitat was made available, and the amount of larvae could be greater for *P. tridentata* than for *C. capitata*; (2) the change in sediment characteristics (increase in the grain size and decrease in the organic matter) might not be favourable to the capitellid *C. capitata*.

Conclusions

The quick benthic recovery found in the present study suggests that small-scale dredging operations are unlikely to produce a long-term, negative impact on marine benthic communities of harbours, even in the most enclosed areas in which the water movement is more limited. Accessibility of coastal ports is vital to the economic growth of coastal regions and navigable depths must be maintained by frequent dredging. We propose the use of small-scale dredging operations in harbour management whenever possible. This approach allows a quick re-adjustment of the initial sediment structure and benthic communities.

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