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ORIGINAL ARTICLE

Relict sand dredging for beach nourishment in the central Tyrrhenian Sea (Italy): effects on benthic assemblages

Barbara La Porta, Monica Targusi, Loretta Lattanzi, Paola La Valle, Daniela Paganelli & Luisa Nicoletti

ISPRA formerly ICRAM, Central Institute for Marine Research, Rome, Italy

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Correspondence
B. La Porta, ISPRA formerly ICRAM, Central Institute for Marine Research, via di Casolotti 300, 00166 Rome, Italy.
E-mail: barbara.laporta@isprambiente.it

Conflicts of interest
All authors declare no conflicts of interest.

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Abstract

The aim of this study is to analyse the effects in space and time of relict sand-dredging activities on macrobenthic assemblages, in an area situated offshore Montalto di Castro (central Tyrrhenian Sea, Italy), and to analyse the recolonisation processes of macrobenthos in the dredged areas. The area in question is characterised by relict sand deposits (Holocenic paleo-beaches), used for beach nourishment along the Latium coast. The effects of sand extraction on benthic assemblages were investigated before, during and after three dredging operations. The sites analysed are located within the dredged areas (inside stations) and in neighbouring, not dredged, areas (outside stations). The results showed that the impact of sand extraction was confined to the dredged stations and to the areas in proximity to the dredged areas. During dredging activities, the structure of benthic assemblages within the impacted stations was characterised by low species richness and diversity. Both the direct removal of sediment and the re-suspension and consequent deposition of fine sediment affected benthic assemblages of the impacted stations. A few months after the dredgings, a recolonisation process was still observed at all the impacted stations. A gradual recolonisation process was observed at those stations affected by only one dredging, whereas a different recolonisation was observed at those stations affected by two dredgings over time. This study suggests that differences of re-colonisation processes of benthic assemblages are related to the intensity of dredging operations in terms of dredging frequency.

To combat coastal erosion along the Italian coasts, the local governments and the environmental protection agencies of several regions have planned nourishment operations exploiting relict sand deposits, within the framework of the European project INTERREG IIIC BEACHMED-e (http://www.beachmed.eu).

Relict sands are non-diagnosed sedimentary deposits situated along the continental shelf in a state of disequilibrium with the present sedimentary dynamics. The removal of such sediments, occurring offshore at high depths, does not affect the wave motion regime and, therefore, coastal dynamics. The relict sand extraction is performed through the use of suction trailers or anchor dredges. A common consequence of trailer dredging is the development of shallow furrows 1–3 m in width and sometimes up to 5 m in depth (Desprez 2000). Anchor dredging leads to the formation of deep, cup-shaped depressions, typically up to 8–10 m deep (Boyd & Rees 2003). Both dredging methods can result in significant environmental alterations, which may take place on both physical and biological levels. The main physical effects involve variations in morphological and bathymetric features, modifications of superficial sediment characteristics, and an increase in water turbidity caused by the re-suspension of fine sediment in the water column during dredging activities. Concerning the biological effects, both dredging methods cause severe disturbances in macrozoobenthos assemblages in terms of the direct effect on sediment removal and the indirect effect associated with the deposition of suspended sediment caused by sand extraction (Desprez 2000; Sardà *et al.* 2000; Boyd & Rees 2003; Szymelfenig *et al.* 2006; Simonini *et al.* 2007). Nevertheless, the type of dredge employed, as well as the nature of the receiving environment, can potentially influence the spatial scale of impact on the benthic fauna, in terms of both direct and indirect effects caused by sand extraction (Boyd & Rees 2003). Boyd & Rees (2003), Newell *et al.* (2004), Robinson *et al.* (2005) and, more recently, Cooper *et al.* (2007) have shown that the impact on benthic assemblages is also related to the process of repeated dredgings within the dredged site. Robinson *et al.* (2005) and Cooper *et al.* (2007) also highlighted that benthic recolonisation processes in repeatedly dredged areas are particularly difficult to predict, because of both the different benthic responses to the intensity of dredging operations in terms of dredging frequency and the variations in environmental characteristics.

Between July 2004 and September 2005, three relict sand-dredging activities were performed in an area offshore Montalto di Castro (Lazio, Italy) in the central Tyrrhenian Sea, with the final aim of nourishing various beaches along the Lazio coasts. This area was characterised by the presence of relict sand deposits that were covered by a muddy layer of recent deposition, with a thickness that varies between a few centimetres and a few metres (Chiocci & La Monica 1999). For these operations, ISPRA, formerly ICRAM (Central Institute for Marine Research), carried out an environmental impact study related to marine relict sand extraction for beach nourishment, funded by the Regione Lazio local authority. This monitoring program has provided an opportunity to collect useful information for the evaluation of the consequences of sand extraction over a relatively short time period in an offshore area that until now has been poorly investigated. In particular, in this study we analysed: (i) the effects of relict sand-dredging activities on the macrobenthos assemblages; (ii) the recolonisation processes of macrobenthos in the dredged areas; (iii) the effects over time of repeated dredging activities on macrobenthos assemblages.

Material and Methods

The study area was located 3.5 nautical miles offshore from Montalto di Castro (Lazio, Italy) in the central Tyrrhenian Sea, on the continental shelf at 50 m of water depth.

The relict sand-dredging activities in this area took place in three different periods, July 2004 (first dredging), June 2005 (second dredging), and September 2005 (third dredging). Over this period, three changes in the boundaries of the extraction areas were reported (Fig. 1). For the first dredging, an anchor dredge was used, whereas for the second and third dredging a trailer dredge was used. The monitoring surveys were carried out from May 2004 to October 2006, before, during and after the dredging activities, as indicated in Nicoletti *et al.* (2006) (Table 1). The sampling plan provided five stations (named stations 1, 2, 3, 4 and 5), one of which was located inside the dredged area in order to monitor the first dredging. The second and the third dredging activities were carried out in proximity (N–NE) to the first area dredged. Three stations (6, 7 and 8) were added to the sampling plan to monitor these dredgings, as shown in Fig. 1. Macrobenthos sampling was carried out using a Van Veen grab with a surface of 0.1 m^2. Two replicates were collected at each station. Samples were sieved through a 1-mm mesh and the retained material was preserved in 4% $CaCO_3$ buffered formalin in seawater. For each station, samples of superficial sediments were collected through a box-corer to determine grain size distribution. Superficial sediments were classified according to Shepard (1954). The collected organisms were counted and classified to the lowest possible taxonomic level. In

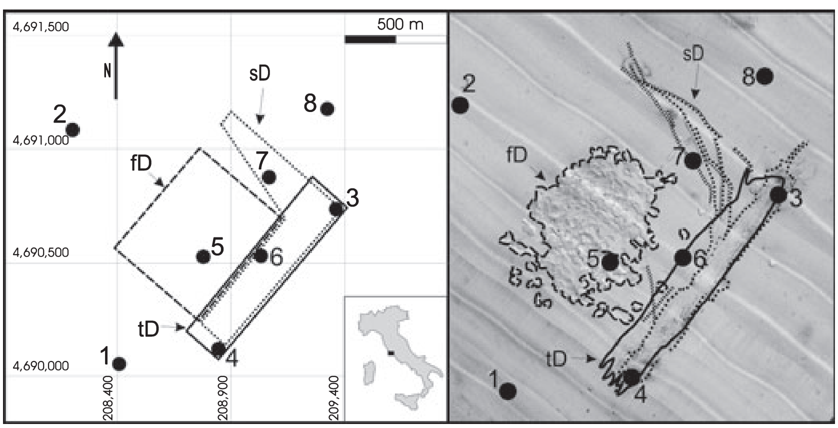

Fig. 1. On the left, the location of relict sand-extraction areas with a map of sampling stations (black point) is represented; on the right, side scan sonar reliefs of the dredged areas (fD = first dredged area; sD = second dredged area; tD = third dredged area) is reported.

Table 1. Sand-dredging characteristics and sampling plan of the three dredged areas.

	First dredged area	Second dredged area	Third dredged area
Volume sand extracted (m^3)	600,000	150,000	700,000
Water depth (m)	50	50	50
Type of dredge	Anchor dredge	Trailer dredge	Trailer dredge
Dredging period	July 2004	June 2005	September 2005
Sampling stations			
Inside the dredged area	5	6,7	3, 4, 6
Outside the dredged area	1, 2, 3, 4	1, 2, 3, 4, 5, 8	1, 2, 5, 7, 8
Surveys			
May 2004 – before dredging (B)	√	–	–
July 2004 – during first dredging (fD)	√	–	–
September 2004 – 2 months after dredging (A2)	√	–	–
April 2005 – 9 months after dredging (A9)	√	–	–
August 2005 – 14 months after dredging (A14)	√	√	√
September 2005 – during third dredging (tD)	√	√	√
May 2006 – 22 months after dredging (A22)	√	√	√
October 2006 – 27 months after dredging (A27)	√	√	√

particular, Polychaeta, Crustacea, Mollusca and Echinodermata were analysed. The main ecological indices (abundance, number of species, Margalef species richness and Shannon–Wiener diversity) were calculated. Multivariate analysis was performed with abundance data to analyse the benthic assemblage variation patterns in terms of species composition and numerically dominant species. The output from the non-metric multidimensional scaling (nMDS) ordination model of the Bray–Curtis similarity matrix was obtained for each station and sampling period. Univariate and multivariate analyses were performed using the software package PRIMER v. 6.1.5 (Clarke & Gorley 2001).

Results

During the study period, 4553 individuals belonging to 191 species were collected (Table 2). Polychaetes were the most abundant taxon (3371 individuals and 103 species), followed by crustaceans (626 individuals and 48 species), echinoderms (328 individuals and 10 species), and molluscs (228 individuals and 30 species). The most abundant species were the polychaetes *Paralacydonia paradoxa*, *Glycera unicornis*, *Paraprionospio pinnata*, *Metasychis gotoi*, the tanaid *Tuberapseudes echinatus,* and the ophiuroid *Amphiura chiajei*. In general, benthic assemblages were characteristic of muddy bottoms. The species composition did not show considerable variations over time. Only a few taxa showed variation over time; these were the opportunistic species *Corbula gibba* and *Terebellides stroemi,* and the sabulicolous polychaetes *Streblosoma bairdi*, *Nephtys hombergi* and *Diplocirrus glaucus*. These latter species were mainly found at the dredged stations.

The univariate analysis showed that the first dredging caused a drastic reduction of the ecological indices exclusively at station 5 located within the dredged area. Stations 1, 2, 3 and 4, located outside the first dredged area, seemed not to have been affected by dredging. Fourteen months after the end of the first dredging activity, impacted station 5 showed an increase in the ecological parameters. During the second dredging, no surveys for the macrobenthos monitorings were carried out. Nevertheless, the monitoring survey carried out 2 months after the second extraction, showed that only station 6 was characterised by extremely low indices values. During the third dredging, all stations except 1, 2 and 8, which were located outside the dredged area, showed a drastic decrease of the ecological indices (Fig. 2 and Table 3).

In general, data relating to the two monitoring surveys carried out after the end of the third dredging highlighted that all the impacted stations showed an increase in the ecological indices. Stations 5 and 6 were characterised by a particularly strong increase in these values (Fig. 2, Table 3), mainly due to the high abundance of a few opportunistic species (e.g. *C. gibba* and *T. stroemi*) and to the presence and abundance of previously absent species that colonised the impacted substrata (e.g. *S. bairdi, N. hombergi, D. glaucus*).

The nMDS ordination plot of data relating to each station and to each sampling period shows an overlapping of samples (Fig. 3). Station distribution confirms the homogeneity of the benthic assemblage observed over time. Station 5, which was affected during the first and the third dredging, segregated on the left side of the plot. Furthermore, on the right side we find stations 5 and 6 analysed during the last two monitoring surveys and

Table 2. Species collected during the study period.

Mollusca
Pseudotorinia architae (O.G. Costa, 1839)
Calliostoma (Ampullotrochus) gramtlatum (Von Born, 1 778)
Turritella communis Risso, 1826
Hyala vitrea (Montagu, 1803)
Calyptraea chinensis (Linnaeus, 1758)
Polinices macilenta (Philippi, 1844)
Polinices nitida (Donovan, 1804)
Eulima glabra (Da Costa, 1778)
Nassarius (Gussonea) cfr. *comiculus* (Olivi, 1792)
Nucula nucleus (Linnaeus, 1758)
Nucula sulcata (Bronn, 1831)
Saccella commutata (Philippi, 1844)
Thyasira biplicata (Philippi, 1836)
Glans aculeata (Poli, 1795)
Astarte sulcata (Da Costa, 1778)
Plagiocardium papillosum (Poli, 1795)
Lutraria sp.
Phaxas adriaticus (Coen, 1933)
Tellina donacina Linnaeus 1758
Tellina serrata Brocchi, 1814
Gari fervensis (Gmelin, 1791)
Abra alba (Wood,1802)
Abra prismatica (Montagu, 1808)
Abra renierii (Bronn, 1831)
Pitar rudis (Poli 1795)
Timoclea ovata (Pennant, 1777)
Corbula gibba (Olivi,1792)
Antalis inaequicostata (Dautzenberg, 1891)
Crustacea
Iphinoe rhodaniensis Ledoyer, 1965
Iphinoe serrata Norman, 1867
Apseudes acutifrons G O. Sars, 1882
Apseudes elisae Bacescu, 1961
Apseudes latreilli (Milne-Edwards, 1828)
Tuberapseudes echinatus (GO. Sars, 1882)
Leptochelia savignyi (Kroyer, 1842)
Arcturella dilatata (GO. Sars, 1883)
Gnathia sp.
Anthura gracilis (Montagu, 1808)
Cirolana borealis Lilljeborg, 1852
Cirolana sp.
Ampelisca diadema (A Costa, 1853)
Ampelisca spinifer Reid, 1951
Ampelisca spinipes Boeck, 1861
Ampelisca typica (Bate, 1856)
Haploops dellavallei Chevreux, 1900
Haploops nirae Kaim Malka, 1976
Leptocheirus guttatus (Grube, 1864)
Leptocheirus mariae G Karaman, 1973
Medicorophium rotundirostre (Stephensen, 1915)
Photis longicaudata (Bate & Westwood, 1862)
Leucothoe incisa Robertson, 1892
Leucothoe lilljeborgi Boeck, 1861
Leucothoe oboa G. Karaman, 1971
Liljeborgia dellavallei Stebbing, 1906
Hippomedon massiliensis Bellan-Santini, 1965
Maera grossimana (Montagu, 1808)
Othomaera schmidtii (Stephensen, 1915)
Westwoodilla rectirostris (Delia Valle, 1893)
Harpinia agna G Karaman, 1987
Harpinia ala G. Karaman, 1987
Harpinia antennaria Meinert, 1890
Harpinia karamani King, 2004
Harpinia sp.
Metaphoxus fultoni (Scott, 1890)
Phtisica marina Slabber, 1769
Alpheus glaber (Olivi, 1792)
Athanas nitescens (Leach, 1814)
Processa canaliculata Leach, 1815
Callianassa subterranea (Montagu, 1808)
Goutretia denticulata (Lutze, 1937)
Jaxea nocturna Nardo, 1847
Paguristes eremita (Linnaeus, 1767)
Anapagurus laevis (Bell, 1845)
Anapagurus serripes (Hope, 1851)
Pagurus cuanensis Bell, 1845
Medorippe lanata (Linnaeus, 1767)
Ebalia deshayesi Lucas, 1845
Liocarcinus maculatus (Risso, 1827)
Goneplax rhomboides (Linnaeus, 1758)
Polychaeta
Capitella capitata (Fabricius, 1870)
Heteromastus filiformis (Claparede, 1864)
Leiocapitella glabra Hartman, 1947
Notomastus aberans Day, 1957
Notomastus latericeus Sars, 1850
Notomastus lineatus Claparede, 1870
Pseudoleiocapitella fauveli Harmelin, 1964
Cossura soyeri Laubier, 1964
Clymenura clypeata (Saint-Joseph, 1894)
Praxillella affinis (M. Sars, 1872)
Praxillella gracilis (M. Sars, 1872)
Maldane glebifex Grube, 1860
Maldane sarsi Malmgren, 1865
Nematonereis unicornis (Schmarda, 1861)
Palola siciliensis (Grube, 1840)
Metasychis gotoi (Izuka, 1902)
Nicomache lumbricalis (Fabricius, 1780)
Maldanidae gen.sp
Polyophthalmus pictus (Dujardin, 1839)
Polyodontes maxillosus (Ranzani, 1817)
Harmothoe longisetis (Grube, 1863)
Lepidonotus clava (Montagu, 1808)
Lepidonotus squamatus (Linnaeus, 1767)
Malmgreniella lunulata (Delle Chiaje, 1830)
Sthenelais boa (Johnston, 1833)
Podarkeopsis arenicola (La Greca, 1947)
Pilargis verrucosa (Saint-Joseph, 1899)
Sigambra tentaculata (Treadwell, 1941)
Glycera alba (O.F. Muller, 1776)
Glycera tesselata Grube, 1863
Glycera unicornis Savigny, 1818
Glycinde nordmanni (Malmgren, 1866)
Goniada maculata Oersted, 1843
Nephtys hombergi Savigny, 1818

Table 2. (*Continued.*)

Nepthys hystricis Mclntosh, 1900
Paralacydonia paradoxa Fauvel, 1913
Phyllodoce lineata (Claparede, 1870)
Dorvillea (Schistomeringos) neglecta (Fauvel, 1923)
Dorvillea (Schistomeringos) rudolphii (Delle Chiaje, 1828)
Aglaophamus rubellus (Michaelsen, 1897)
Eunice pennata (O.F. Muller, 1776)
Eunice vittata (Delle Chiaje, 1828)
Lysibranchia paucibranchiata Cantone, 1983
Marphysa belli (Audouin & Milne-Edwards, 1833)
Marphysa kinbergi Mclntosh, 1910
Lumbrineriopsis paradoxa (Saint-Joseph, 1888)
Lumbrineris gracilis (Fillers, 1868)
Lumbrineris latreilli Audouin & Milne Edwars, 1834
Scoletoma emandibulata-mabiti (Ramos, 1976)
Scoletoma fragilis (O.F. Muller, 1776)
Scotetoma tetrawa (Schmarda, 1861)
Arabella tricolor (Montagu, 1804)
Drilonereis filum (Claparede, 1870)
Apomtphis bilineata (Baird, 1870)
Apomtphys brementi (Fauvel, 1916)
Apomtphis fauveli (Rioja, 1918)
Hyalinoecia tubicola (O.F. Muller, 1776)
Myriochele oculata Zachs, 1923
Owenia fusiformis Delle Chiaje, 1841
Aphelochaeta marioni (Saint-Joseph, 1894)
Caulleriella mitltibranchiis (Grube, 1863)
Chaetozone caputesocis (Saint-Joseph, 1894)
Chaetozone setosa Malmgren, 1867
Monticellina dorsobranchialis (Kirkegaard, 1959)
Brada villosa (Rathke, 1843)
Diplocirrus glaucus (Malmgren, 1867)
Flabelligera affinis M. Sars, 1829
Sternaspis scutata (Ranzani, 1817)
Amage adspersa (Grube, 1863)
Amage gallasii Marion, 1875
Ampharete acutifrons (Grube, 1860)
Amphicteis gunneri (M. Sars, 1835)
Anobothrus gracilis (Malmgren, 1866)
Eclysippe vanelli (Fauvel, 1936)
Lysippe labiata Malmgren, 1866
Sabellides octocirrata (M. Sars, 1835)
Melinna palmata Grube, 1870
Pectinaria auricoma (O. F. Muller, 1776)
Pectinaria koreni (Malmgren, 1866)
Pista brevibranchia Caullery, 1915
Pista cnstata (O. F. Muller, 1776)
Streblosoma bairdi (Malmgren, 1866)
Terebellides stroemi M. Sars, 1835
Magellona spl
Magelona sp2
Spiochaetopteus costarum (Claparede, 1868)
Aonides paucibranchiata Southern, 1914
Laonice *cirrata* (M. Sars, 1851)
Minuspio cirri/era Wiren, 1883
Paraprionospio pinnata (Fillers, 1901)
Prionospio caspersi Laubier, 1962
Prionospio ehlersi Fauvel, 1928
Prionospio fallax Soderstrom, 1920
Prionospio steenstrupi Malmgren, 1867
Scolelepis bonnieri (Mesnil, 1896)
Scolelepis foliosa (Audouin & Milne-Edwards, 1833)
Spio decoratus Bobretzky, 1870
Spio filicornis (O. F. Muller, 1776)
Spio multioculata (Rioja, 1918)
Spiophanes bombyx (Claparede, 1870)
Spiophanes kroyeri Grube, 1860
Spiophanes kroyeri reyssi Laubier, 1961
Poecilochaetus serpens Alien, 1904
Echinodermata
Pseudotrachytyone sp.
Trachythyone elongata (Duben Koren, 1844)
Trachythyone tergestina (M. Sars, 1857)
Thyone fusus (O.F. Muller, 1776)
Phyllophorus urna Grube, 1840
Labidoplax digitata (Montagu, 1815)
Amphiura chiajei Forbes, 1843
Amphiura filiformis (O.F. Muller, 1776)
Ophiopsila aranea Forbes, 1843
Ophiura albida Forbes, 1839
Schizaster canaliferus (Lamarck, 1816)

characterised by high species richness and diversity. Concerning the grain size distribution of the sediments, some grain size variations were observed after the dredgings, both inside and outside the dredged areas. In particular, a significant increase in the sandy fraction (from 28% to 94.3%) was observed after the first dredging in station 5 (inside the dredged area) and another (from 47% to 88.7% of sand) was recorded after the third dredging in station 6 (inside the dredged area). No relevant grain size variations were reported in the other stations.

Discussion

The results obtained from this study, as expected and in accordance with some authors (Blake *et al.* 1996; Newell *et al.* 1998; Sardà *et al.* 2000; Van Dalfsen *et al.* 2000; Boyd & Rees 2003; Simonini *et al.* 2005), highlighted that the direct effects of relict sand dredgings on macrobenthos assemblages were limited to the dredged areas. In particular, all the stations located inside the dredged areas during the first (station 5) and the third dredging (stations 3, 4 and 6) showed a strong decrease in ecological indices as a consequence of the complete removal of superficial sediments. Despite the lack of data, both before and during the second dredging, it is important to highlight the case of station 6, where both the low values of the ecological indices recorded a few months after the second extraction and its position (inside the second dredged area) allowed us to hypothesise that this station was dredged during the second extraction.

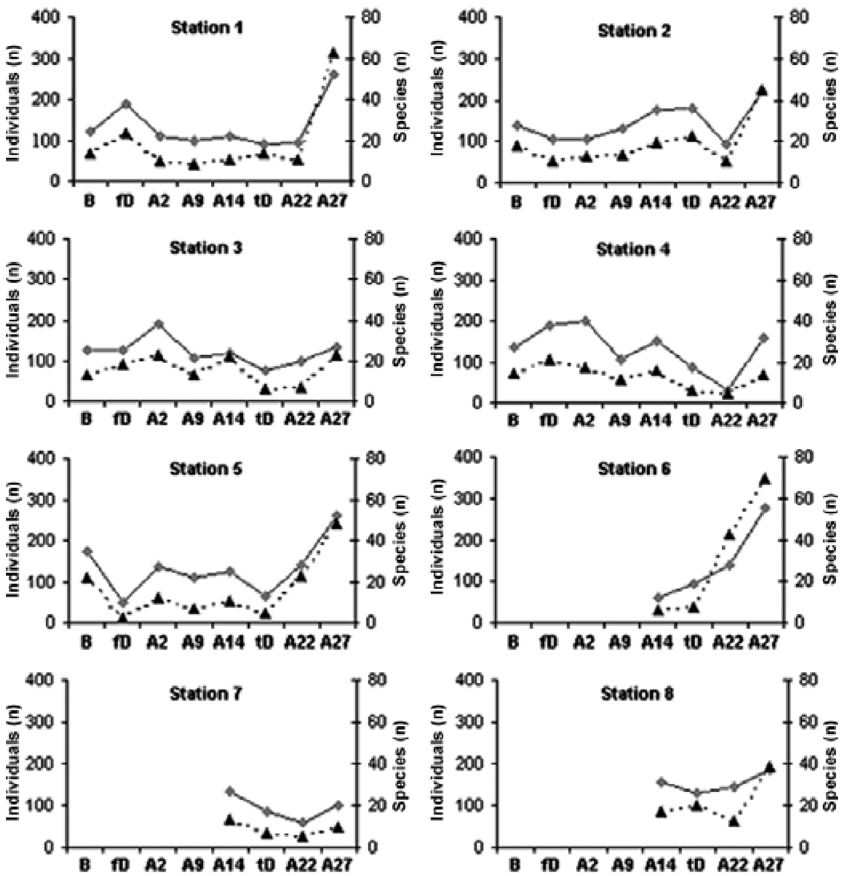

Fig. 2. Number of individuals (black line) and species (grey line) collected at each station over time.

Table 3. Species richness (d) and Shannon diversity (H′) values calculated for each station over time.

Stations		B	fD	A2	A9	A14	tD	A22	A27
1	d	5.74	8.03	5.62	5.34	5.86	4.37	5.06	9.37
	H′	4.21	4.86	4.10	3.97	4.14	3.30	3.95	4.67
2	d	6.40	5.35	5.29	6.53	7.92	7.90	5.15	8.62
	H′	4.05	4.02	3.99	4.50	4.77	4.87	3.77	4.96
3	d	5.96	5.94	8.05	5.32	5.23	4.25	5.64	5.93
	H′	4.24	4.36	4.71	3.97	3.88	3.51	4.11	4.07
4	d	6.18	8.68	8.93	5.35	7.34	5.03	1.85	7.77
	H′	3.95	4.93	5.00	4.11	4.64	3.91	2.15	4.64
5	d	7.61	3.75	6.87	6.07	6.51	3.96	6.40	9.99
	H′	4.54	3.28	4.50	4.25	4.47	3.36	4.38	5.11
6	d	–	–	–	–	3.97	5.19	5.31	9.60
	H′	–	–	–	–	3.45	4.09	3.34	4.07
7	d	–	–	–	–	6.75	4.75	3.74	5.30
	H′	–	–	–	–	4.47	3.71	3.22	4.03
8	d	–	–	–	–	7.19	5.85	7.09	7.31
	H′	–	–	–	–	4.63	4.28	4.53	4.35

This study highlighted that the impacts of relict sand dredgings on macrobenthos assemblages were observed in the zones in proximity to the dredged areas. These indirect impacts were due to the re-suspension and subsequent deposition of fine sediments caused by sand-extraction operations and was mainly evident at stations 5 and 7, which were located in proximity to the third dredged area. The increase in the fine fraction of superficial

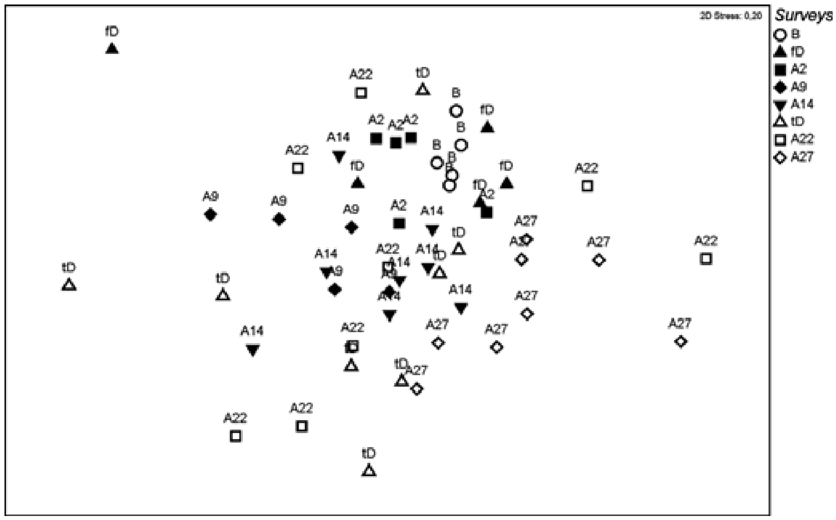

Fig. 3. 2D-nMDS ordination plot of abundance data of each station and each sampling period.

sediments observed in station 5 after dredging confirmed that fine sediment re-deposition had occurred.

These results also highlighted that a stronger sediment suspension was generated by the trailer dredge (used for the second and the third dredging), whose impact was greater than that of the anchor dredge (used for the first dredging).

Concerning the analysis of the recolonisation processes of macrobenthos assemblages, our results showed that a few months after the end of dredgings, the recolonisation processes could still be observed at all the impacted stations, in accordance with Green (2002), Boyd & Rees (2003), Simonini *et al.* (2005). In general, these processes are mainly due to the settlement of new recruits from the planktonic larvae and immigration of the adults from the neighbouring areas (Bonvicini Pagliai *et al.* 1985; Rees & Dare 1993; Newell *et al.* 1998; Van Dalfsen *et al.* 2000), but recolonisation processes are difficult to predict because they are strongly influenced by many different factors (*e.g.* biological cycles of different species, hydrodynamic regime, changes in sediment structure depth).

This study also revealed differences in the recolonisation processes of the impacted stations. The gradual recolonisation process was observed at stations 3, 4 and 7, whereas different processes (with an exponential trend) were observed at stations 5 and 6. These stations were initially characterised by the abundance of a few opportunistic species (*e.g. Corbula gibba* and *Terebellides stroemi*) and, subsequently (in the last monitoring), by an increase in abundance and in the number of sabulicolous species (e.g. *Streblosoma bairdi, Nephtys hombergi* and *Diplocirrus glaucus*) which had not been collected in the previous investigated periods. This phenomenon is normally observed in dredged substrata where the defaunation allows the opportunistic species to form dense populations in the first phase of the recolonisation process, followed by an increase in the number of species and individuals (Bonsdorff 1980; Kenny & Rees 1994, 1996; Newell *et al.* 1998; Sardà *et al.* 2000; Van Dalfsen *et al.* 2000; Nicoletti *et al.* 2004). The differences between two recolonisation processes at the impacted stations were probably related to the fact that the first group of stations (3, 4 and 7) was influenced exclusively by only one dredging (the third one), whereas the second group (5 and 6) was affected by two dredgings (respectively the first and the third one for station 5 and the second and the third one for station 6). Moreover, these differences could be related to the intensity of dredging operations in terms of dredging frequency, as also observed by Boyd & Rees (2003), Newell *et al.* (2004), Robinson *et al.* (2005) and Cooper *et al.* (2007).

This study has confirmed the observations of some authors (Robinson *et al.* 2005; Smith *et al.* 2006) concerning the difficulties in evaluating the effects over time of relict sand dredgings on benthic assemblages, due to the high number of factors involved. In our specific case, the analysis of the impact on the assemblages was further complicated by the use of two different types of dredge, and by the fact that dredging activities were repeated within a relatively short period of time, as well as in areas that are very close to one another. Further, medium-term monitoring surveys will provide a more detailed description of how the recolonisation process of macrobenthos assemblages affected by sand dredging will occur, as well as how long this will take.

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