



Impact of dredged-material disposal on soft-bottom communities in a recurrent marine dumping area near to Guadalquivir estuary, Spain

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ARTICLE INFO

Keywords:

Disposal
Macrobenthic community
Stable isotopes
Food web
Gulf of Cadiz

ABSTRACT

This study assesses the effects of dredged material disposal in a recurrent marine dump near the Guadalquivir Estuary (south-western Spain). We compared the changes observed with two reference areas combining a classical ecological approach with new stable isotope techniques to analyse trophic structure. We detected permanent changes in the macrofaunal community structure as well as in the diversity and biotic indices applied, which showed higher values in the disposal area. The community in the marine dump had lost the natural temporal variations observed in the reference areas. These effects could be due to the last disposal event carried out in the summer of 2015 or to the recurrent disposals since 2010. Despite the structural changes shown by the benthic community, these impacts were not reflected in the food web structure of the marine dump. Our results confirm the high variability of disposal disturbances. Hence, we recommend performing studies in every disposal event, merging different functional and structural approaches.

1. Introduction

Both dredging and the dumping of dredged material are common practices around the world and are one of the most serious environmental concerns for coastal management (Marmin et al., 2016; Moog et al., 2015; Van Dolah et al., 1984). These practices are particularly developed in estuaries where sedimentation patterns are high and may be accelerated by human activities (Cesar et al., 2014). However, dredging is essential to maintain security in navigation and to support trade and economic sustainability (OSPAR, 2008; Bates et al., 2015). Although relocation of dredged material is one of the most important concerns in those activities, much of the material dredged is still disposed at sea for economic reasons (Harvey et al., 1998; Katsiaras et al., 2015; Tornero and Hanke, 2016). Dredging and disposal of dredge material may cause environmental problems in coastal and marine areas, both physically and through contaminants (e.g. Bolam et al., 2006; Bolam and Rees, 2003; Cesar et al., 2014; Fredette and French, 2004; Guerra et al., 2009, 2007; OSPAR, 2008). Within the implementation of the European Water Framework Directive, the improvement of our knowledge of human impacts is critical for marine management and conservation (Marmin et al., 2016).

Effects of dumping on coastal ecosystems depend on several factors, such as the disposal method, the amount of sediment disposed, the

physical and chemical characteristics of the dredged and receiving sediment, depth, the oceanographic and sedimentary conditions of the receiving habitat, season and the adaptations and composition of the inhabitant community (Katsiaras et al., 2015; Simonini et al., 2005b, and references therein). Dredged material disposal may affect marine organisms through physical alterations such as the smothering of the sea bottom, in addition to increased turbidity and enrichment of pollutant, organic matter and nutrient concentrations in the sediment, among other impacts (Cesar et al., 2014, and references therein). Furthermore, these impacts may lead to important habitat changes through long term effects (Powilleit et al., 2009). Benthic macrofauna are the organisms that are most sensitive to these impacts because of their relative immobility (Simonini et al., 2005b; Taupp and Wetzell, 2013). However, effects on primary production and changes in species composition can have an impact on the whole food web structure (Pezy et al., 2017). Nutrient inputs often affect the basal resources, sometimes converting a complex food web to a more homogeneous system (Rooney et al., 2008). Analysing community trophic structure is one way to assess the nature and magnitude of human impacts (Hussey et al., 2014).

When a potential anthropogenic disturbance is repeated over time, the recurrence of this event is also an important factor too (Glasby and Underwood, 1996; Villnäs et al., 2013). On the first hand, sediment

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disposal after dredging operations can be considered a short-term or “pulse” disturbance, in the sense outlined by Glasby and Underwood (1996). On the other hand, the sediment disposed, if it differs from the native sediment, may constitute permanent or a long-term change in the affected habitat. Recurrent sediment disposal operations and the accumulation of dredged material could generate a series of short-term responses in the biological community, followed by a recovery (to some extent) and convergence to a previous state, a permanent change or a combination of these two processes (Underwood, 1994).

There are several studies addressing the effects of disposal in offshore environments (e.g. Smith and Rule, 2001; Stronkhorst et al., 2003; Zimmerman et al., 2003; Simonini et al., 2005b; Ware et al., 2010; Bolam et al., 2011; Bolam, 2012; Taupp and Wetzel, 2013; Cesar et al., 2014; Katsiaras et al., 2015; Marmin et al., 2016). However, several authors have highlighted the divergent results obtained in disposal impact studies and concluded that the potential environmental effects must be evaluated case by case (Bolam et al., 2006; Bolam and Rees, 2003; Harvey et al., 1998; Katsiaras et al., 2015; OSPAR, 2008; Simonini et al., 2005b). Furthermore, to our knowledge, the effects of disposal of a recurrent and active marine dump on the benthic community have not been widely studied (Bolam et al., 2011). In addition, the impact of sediment disposal on the food web has been studied even less (e.g. Pezy et al., 2017).

In this study we evaluated the impact of dredge sediment deposition on a recurrent marine disposal area in south-western Spain. The Guadalquivir estuary provides access to Seville harbour, the only inland harbour in the Iberian Peninsula, through a 90-km long navigation channel. The strongly modified course and sedimentary dynamics made some dredging work necessary every few years to maintain navigability (Diez-Minguito, 2012). Part of the sediment dredged along the channel is brought ashore for storage, while the other part goes to beach nourishment and some is disposed in an authorised marine area established in 2010. We compared the changes observed in the disposal area with two nearby control areas. Specifically, this study is focused on the following objectives: (i) to assess the response of the benthic community to recurrent sediment discharges (ii) to evaluate the effect of the most recent disposal event (iii) to determine the effects on the food web structure and (iv) to describe the temporal variation of a biological community in a recurrently stressed area.

2. Material and methods

2.1. Study area

The marine disposal area is situated five miles offshore in an open water area in front of the river mouth of the Guadalquivir estuary, south-western Spain (Fig. 1). The bottom is 20 m in depth. Sediments are mainly composed of fine sand and mud and are affected by terrigenous inputs of the Guadalquivir River and the hydrodynamic conditions of the Gulf of Cádiz, with predominant eastward currents (Sainz and Ruiz, 2006). These currents are derived from the North Atlantic surface waters and have the greatest effect on the sediment distribution (Sainz and Ruiz, 2006). The marine dump has a rectangular shape with a surface of 66.26 ha (Fig. 1). The sea bottom near this area has similar sedimentary characteristics (Gonzalez and Dias, 2004) and biological communities (Fa et al., 2003). This dump has been receiving dredge material from harbours and the Guadalquivir estuary in four events since 2010 (2010: 338.652 m³, 2011: 353.488 m³, 2013: 354.795 m³ and 2015: present study), (data from Autoridad Portuaria de Sevilla (APS)).

From mid-August to the first week of September 2015, dredging operations for maintenance of the navigation channel of the Guadalquivir estuary and Bonanza port were performed and 314.275 m³ of the extracted sediment was discharged in the disposal area.

2.2. Sampling design

Three sampling areas were established in the same environment: one within the marine disposal area (DA) and two controls located approximately three miles from the affected area (C₁ and C₂) (Fig. 1). Controls were located at the same depth in a very similar area with equal hydrological and geomorphic characteristics (Sainz and Ruiz, 2006). Distances between controls and the disposal area were enough to avoid any possible impacts. In total, five sampling surveys were carried out: twice before the last dumping event (to test the previous affections of the recurrent disposals) and three times after the last disposal event of summer 2015 (to analyse both effects of this last perturbation and the temporal variation of the system). Before: June and July 2015. After: October 2015 (one month after the last disposal event), March and August 2016, (6 and 12 months after the event, respectively).

At each control and disposal area, three stations were randomly placed. In each station, three samples were taken for macrofaunal analysis with a van Veen grab (0.15 m² total sampling area per station and date). For posterior analysis, all stations were considered replicates of each area. Macrofaunal samples were sieved through a 0.5 mm size mesh. Infauna was preserved in ethanol (70%), and stained with Rose Bengal for subsequent identification and quantification to species level where possible. A sediment sample was taken for granulometric analysis. Grain size distribution was measured as percentages of 100 gr of dry sediment sieved through a sieve net (5 mm, 2 mm, 1 mm, 0.5 mm, 0.250 mm, 0.125 mm, 0.063 mm).

For the isotope analysis of carbon and nitrogen, fauna were collected in three of the five surveys (June 2015, October 2015 and August 2016) with a van Veen grab (0.05 m²) and an epibenthic sledge (46 × 25 cm, 2.5 mm net mesh size). The number of samples was sufficient to obtain a representative number of species of the benthic community. All samples were sieved by 0.5 mm mesh sieve and organisms were sorted by species, transferred to the laboratory in refrigerated containers and kept alive for 24 h to evacuate their gut content. In order to determine the isotopic composition of sediment organic matter (SOM), sediment was taken from the upper 2 cm of a van Veen grab sample. In the lab, species identification was confirmed and fauna were rinsed in distilled water. Muscle tissue samples of fish and molluscs were dissected. A pool of several organisms was used when the individuals had low biomass values (< 0.3 mg dry weight). Organisms were dried at 60 °C for at least 48 h and ground to a powder. Sediment samples were acidified with 0.1 M HCl to remove carbonates, and were then oven dried at 60 °C. Subsamples of powdered materials were weighed to the nearest 0.3 mg and placed into tin capsules for $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ determinations. All samples were combusted at 1020 °C using a continuous-flow isotope-ratio mass spectrometry system by means of Flash HT Plus elemental analyser coupled to a Delta-V Advantage isotope ratio mass spectrometer via a CONFLO IV interface (Thermo Fisher Scientific, Bremen, Germany). From the obtained results we performed a descriptive approach to the food web structure in the three areas by means of isospace plots based on carbon and nitrogen isotopic signatures (Fry, 2006). Species selected were those that appeared both in the disposal and control areas and in the three surveys.

2.3. Data analysis

At disposal and control areas, univariate measures such as species richness (as number of taxa, S), Shannon's diversity index (H' based on $\log e$) and total abundance (as ind./m², N), were calculated. Ecological quality status (EQS) was also assessed through the biotic indices AMBI (Borja et al., 2000), BENTIX (Simboura and Argyrou, 2010; Simboura and Zenetos, 2002) and BENFES (Sánchez-Moyano et al., 2017). The biotic indices are based on the assignation of ecological groups according to the sensitivity or tolerance toward anthropogenic disturbances (Ponti et al., 2009). All indices were calculated for each station. AMBI and BENTIX are based on the relative abundance of

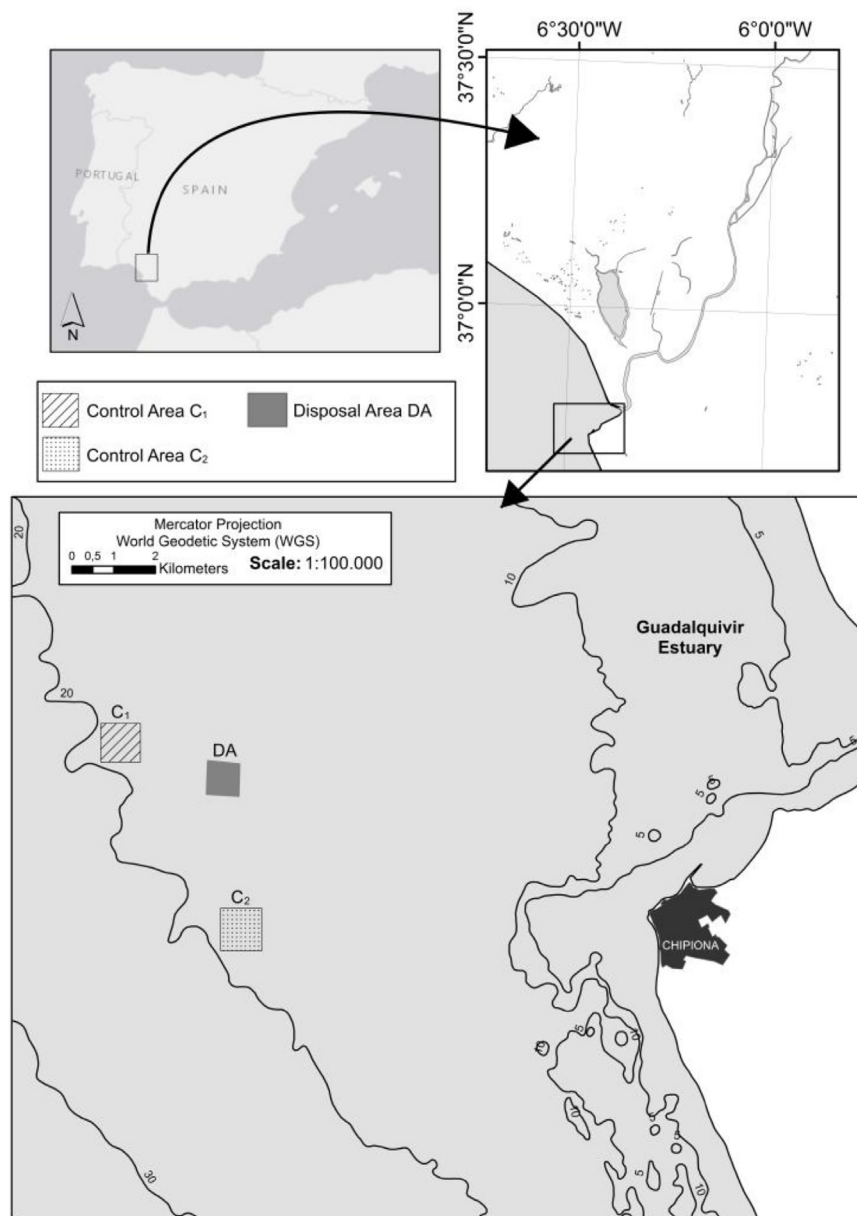


Fig. 1. Location of the disposal and control areas.

species, and BENFES is based on presence/absence and identification at family level.

To examine the dumping effects in the study area, a Permutational Multivariate Analysis of Variance (PERMANOVA) (Anderson, 2001) based on Euclidian similarity matrices of S, N and H' indices using 9999 permutations was performed. When the number of total possible permutations to obtain the p-values were low, we used the estimate obtained by Monte Carlo sampling (Anderson and Robinson, 2003). The experimental design included two crossed fixed factors: "Impact vs Control" with two levels (Impact and Control) and "Time" with two levels (Before and After the last dumping event). The design also included two random factors: "Area" nested within "Impact vs Control" with three levels (DA, C1, C2) and the sampling dates "Dates" nested within "Time" with five levels (Jun 15, Jul 15, Oct 15, Mar 16 and Aug 16).

The results of the multivariate communities' assemblages were also investigated using a PERMANOVA analysis, with the same experimental design as above, based on the Bray-Curtis similarity matrix from the square root transformed abundance sets. When the saturated model

(with all explained terms included in the model) generated negative estimates of some components of variation (this indicates that the model in turn is a poor one, according to Anderson et al., 2008), a model selection was performed by excluding the affected terms, one by one, beginning with those of higher order (interactions), and provided that their p values were higher than 0.25 (Anderson et al., 2008). Following this process with all the analyses performed here (both the multivariate and the univariate PERMANOVA), the factor "Area" and its interactions with "Dates" and with "Time" were excluded from all four models. It is important to notice that "Area" is nested within "Impact vs Control" and, given the asymmetric design of this study (there are no replicated areas within "Impact" level), this term only estimates the variance between the two control areas (Anderson et al., 2008), which was negligible in this study. The analysis presented here does not include these terms. Significant interactions, if detected, were further explored in separate analyses, within the levels of the interacting factors; i.e. the significant interactions between "Impact vs Control" and "Dates" were further analysed separately by disposal area and the control area.

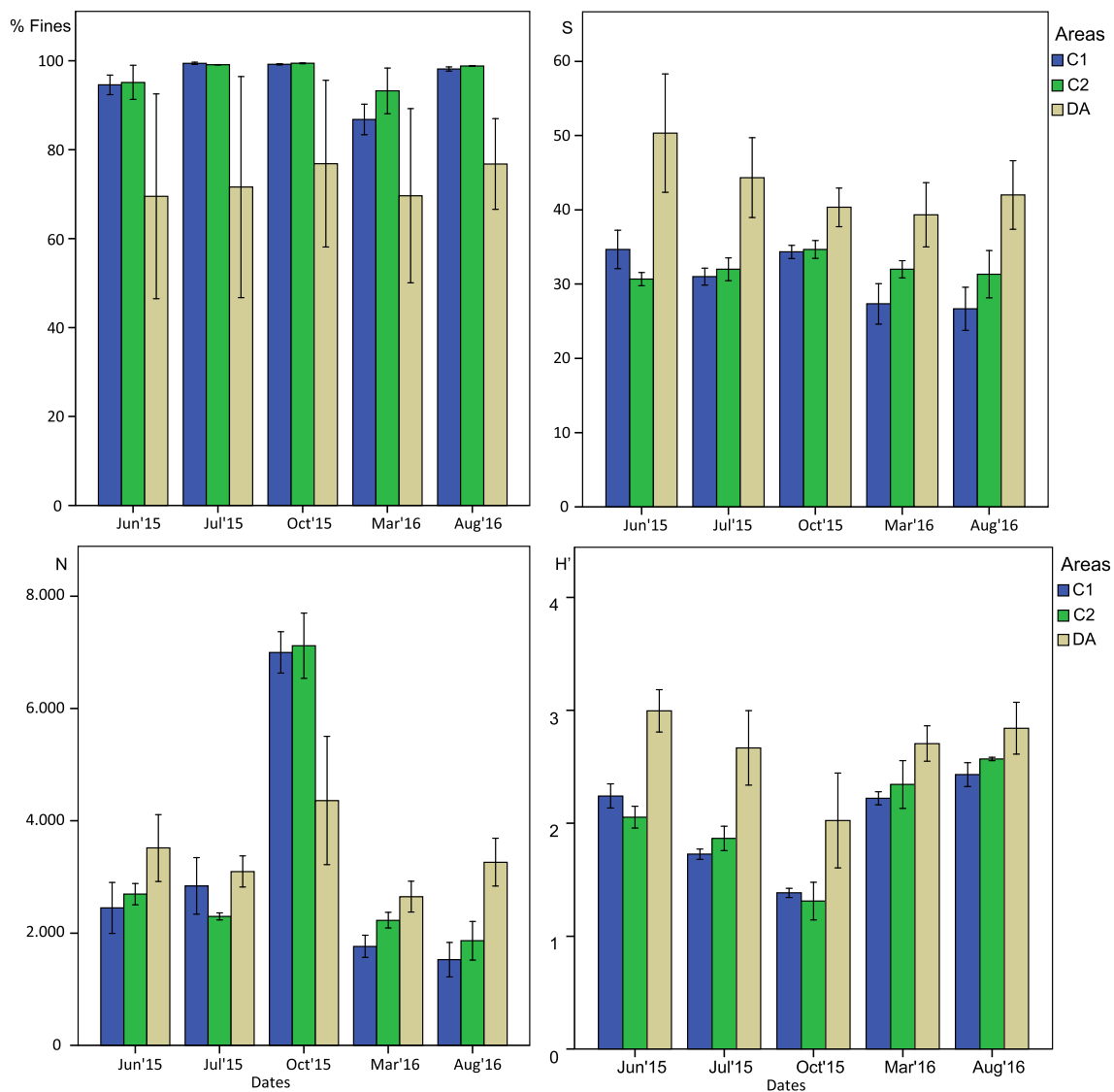


Fig. 2. Patterns of variation of the %Fines and univariate indices (S, N, H') in the three areas across the sampling dates. Vertical bars represent \pm Standard error.

Macrobenthic communities were also examined by a non-metric multidimensional scaling ordination (nMDS) based on the Bray-Curtis similarity index. SIMPER analysis was used to identify the species contributing most to any observed spatial or temporal pattern in the communities (Clarke, 1993).

All analysis and data visualisations were performed using the statistical software package PRIMER version 6 and SPSS version 24.

3. Results

Sedimentary analysis showed that the disposal area had lower % fines (fraction lower than 0.063 mm) than the control areas (Fig. 2). This difference remained across all sampling dates. On the other hand, biological analysis showed a total of 21,899 individuals in 128 species belonging to the phyla Annelida, Arthropoda, Mollusca, Echinodermata, Chordata, Platyhelminthes and Nemertea across the sampling dates. We did not find any predominant group except for some increments of particular species in all stations. For example, in October 2015 an increment of Arthropoda abundance was observed. The species with the maximum-recorded abundance per station were the polychaetes *Magelona papillicornis* (813 ind/m²) and *Sternaspis scutata* (307 ind/m²), the tanaidacean *Apseudopsis latreillii* (6313 ind/m²) and the decapod *Upogebia tipica* (467 ind/m²), and molluscs *Turritella turbona* (60 ind/

m²) and *Corbula gibba* (800 ind/m²).

Percentage of fines, Shannon's diversity index, Richness and Total Abundance (ind/m²) differences are shown in Fig. 2 and Table 1. As mentioned earlier, PERMANOVA analysis of univariate indices showed that there were no differences in the three indices between the two control areas despite the distance between them. PERMANOVA results of the univariate indices also showed that the disposal area had significantly ("Impact vs Control", $p < 0.0001$) higher values of richness and Shannon's diversity than controls. Differences between areas remained over all the sampling surveys. Total abundance did not show significant differences between controls and disposal area, however we detected a different pattern of variation over the surveys in the disposal area than in controls (Impact vs Control \times Dates, $p < 0.0001$). PERMANOVA analysis of separated samples of the disposal area did not show significant differences between dates ($p = 0.4485$). On the contrary, there was a significant difference ($p < 0.0001$) in the controls between sampling dates. In October 2016, a high number of individuals of the tanaid *Apseudopsis latreillii* occurred in the control areas (Fig. 3). More than 5000 ind/m² of this species were found in the control areas while this increase was lower in the disposal area (< 3000 ind/m²).

The mean of biotic indices per area are shown in Table 2. The disposal area showed higher values (lower for AMBI) than control areas. However, limits of the ecological statuses for the AMBI and BENFES did

Table 1

Univariate PERMANOVA results in the study areas based on the Euclidian similarity matrix of the richness data (S), Shannon's diversity (H') and total abundance (ind/m²) (N). *p estimation obtained by Monte Carlo sampling.

	df	MS	Pseudo-F	p	Unique perms
S					
Impact vs Control	1	1470.20	58.96	0.0074	9555
Time	1	146.22	5.07	0.1081*	10
Dates (Time)	3	28.84	0.86	0.47	9951
Impact vs Control x Time	1	79.35	3.18	0.1721	9549
Impact vs Control x Dates (Time)	3	24.94	0.74	0.5343	9935
Res	35	33.44			
Total	44				
N					
Impact vs Control	1	8.00E+05	0.12	0.7558	9545
Time	1	3.07E+06	0.72	0.7241*	10
Dates (Time)	3	1.95E+07	33.44	0.0001	9960
Impact vs Control x Time	1	1.93E+06	0.28	0.6651	9568
Impact vs Control x Dates (Time)	3	6.78E+06	11.59	0.0001	9952
Res	35	5.85E+05			
Total	44				
H'					
Impact vs Control	1	4.309	104.9	0.0051	9812
Time	1	0.133	8.21E-02	0.79*	10
Dates (Time)	3	1.63	17.42	0.0001	9953
Impact vs Control x Time	1	0.343	8.34	0.0697	9854
Impact vs Control x Dates (Time)	3	4.11E-02	0.44	0.7343	
Res	35	9.33E-02			
Total	44				

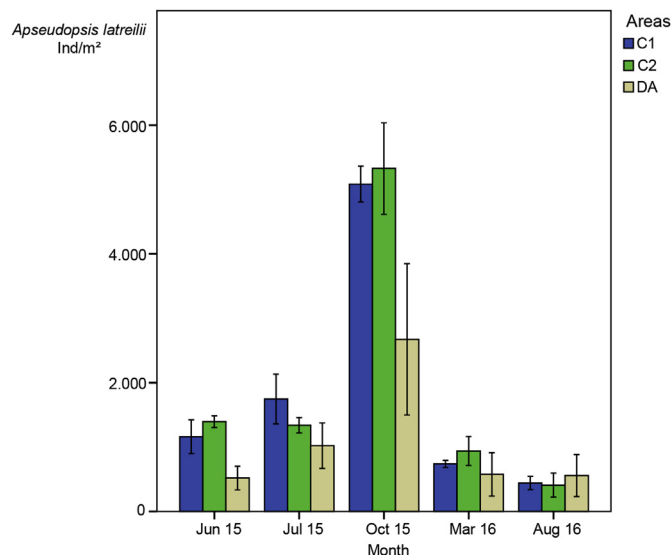


Fig. 3. Abundance of the tanaid *Apseudopsis latreillii* across the sampling surveys. Vertical bars represent \pm Standard error.

Table 2

Results of the biotic indices AMBI, BENFES and BENTIX. Ecological Quality Status (EQS): Blue = “High”, Green = “Good”, Yellow = “Moderate”.

	Jun 15			Jul 15			Oct 15			Mar 16			Aug 16		
	DA	C1	C2	DA	C1	C2	DA	C1	C2	DA	C1	C2	DA	C1	C2
AMBI	1.9	2.4	2.6	2.1	2.4	2.6	2.4	2.6	2.7	1.7	2.6	2.4	2.2	2.7	2.5
BENFES	174.0	124.7	108.3	158.0	113.3	117.3	148.3	117.0	124.7	146.3	100.7	113.6	152.7	98.3	117.7
BENTIX	4.0	3.3	3.0	3.8	3.1	3.0	3.2	2.7	2.6	4.4	3.4	3.5	3.9	3.5	4.0

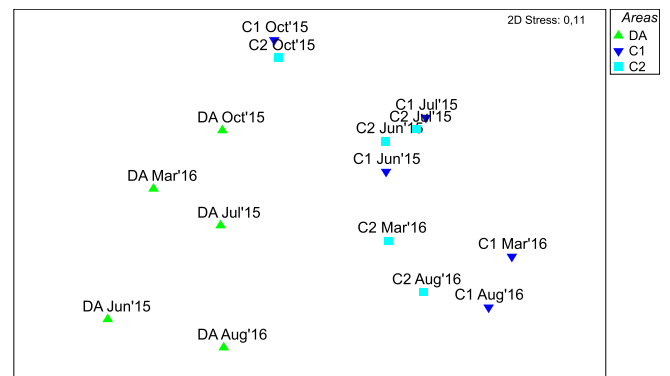


Fig. 4. MDS of the distance among centroids (resemblance) for the factor Area x Dates, with representation based on the abundance of the different species.

not allow for the differentiation between areas or dates. AMBI showed a “Good” ecological status in all the areas across the sampling dates. BENFES showed a “High” status in all areas except in C1 in August 2016. Only BENTIX showed a “Good” ecological status in the disposal area in all the surveys and “Moderate” in October 2015. Control areas showed a “Moderate” status of this index in June, July and October 2015.

The MDS plot showed different tendencies between the areas (Fig. 4). There were similar communities in the two control areas, with clear temporal differences. However, the community in DA was clearly separated from the controls, even before the most recent disposal event, and showed a fuzzy temporal variability.

PERMANOVA results for the community analysis again showed no differences between the two control areas. Results also showed significant differences between the community of the disposal area and controls ($p = 0.0205$) (Table 3). It must be noted that the pseudo-F for this term (Impact vs Control) is constructed using the mean squared of the interaction term “Impact vs Control x Dates” and provides a test for its significance over and above the potential variability in the effect among sampling dates; it is interpretable as significant even if the interaction between factors “Dates” and “Impact vs Control” were also significant ($p = 0.0024$). Instead, the results did not show a significant interaction between “Time” and “Impact vs Control” ($p = 0.4388$). These results indicate a different pattern of change in the control and the disposal area across the sampling dates, but not globally before and after the sediment disposal on October 2015. PERMANOVA results of the separated analysis of the disposal area showed no significant differences ($p = 0.1027$) across the sampling dates. Conversely, there were different significant community structures in the controls across the sampling dates ($p < 0.0001$).

SIMPER analysis (Table S1) showed dissimilarities between controls and the disposal area for each sampling date. Before the last dumping event, differences were mostly due to the higher abundance of the tanaid *Apseudopsis latreillii* and the polychaete *Sternaspis scutata* in the controls, and the polychaetes *Magelona papillicornis*, *Spiophanes kroyeri*, *Nephtys hombergii*, *Sigambra parva*, two species of the genus *Mediocorophium* and the cumacean *Eudorella truncatula* in the disposal area.

SIMPER analysis comparing dates within controls (Table S2) showed that differences between June and July 2015 were, firstly, due to several species, such as *A. latreillii* and the mollusc *Hyala vitrea*,

Table 3

PERMANOVA results of the Bray-Curtis similarity matrix based of the square root transformed data. *p estimation obtained by Monte Carlo sampling.

	df	MS	Pseudo-F	p	Unique perms
Impact vs Control	1	5635	6.3079	0.0205	9626
Time	1	2113	1.0245	0.4368*	10
Dates (Time)	3	2062	4.4828	0.0001	9881
Impact vs Controlx Time	1	949.63	1.063	0.4388	9648
Impact vs Controlx Dates (Time)	3	893.37	1.9422	0.0024	9873
Res	35	459.99			
Total	44				

which began to increase their abundance in July 2015. Secondly, new species such as *U. tipica* and the bivalve *Hemilepton nitidum* occurred in this last month. In October, the abundance of several species experienced a high increase (*A. latreillii*, *Aonides oxycephala*, *Capitella capitata*, *U. tipica*, *H. nitidum*, *H. vitrea*). In March 2016 the abundances of these species and others decreased to June 2015 levels. The polychaete *Terbellides stroemi* occurred in this month. In August 2016, abundances of some species increased while others decreased: e.g. *H. vitrea* and *C. gibba* began to increase while others, such as *A. latreillii*, *E. truncatula* and *T. stroemi* decreased.

On the other hand, in the disposal area (Table S3), SIMPER results showed that the most abundant species such as *A. latreillii* and *U. tipica* followed the same patterns as in the controls areas, but with a lower abundance increase. On the contrary, as with control areas, in October 2015, several species, such as *Mediocorophium* sp., *Spiophanes kroyeri* and *Lumbrineris latreillei*, suffered a decrease in their abundance. In March 2016 *A. latreillii* decreased while *U. tipica* remained at the same value until August 2016, where it practically disappeared from this area in addition to *H. nitidum*. A general increase in the abundance of other species was observed.

Isotope graphs showed practically no differences in food webs between the three areas (Fig. 5). The carbon isotope signal of SOM in the disposal area showed different signals over time, whilst these signals were similar over time in the control areas. The disposal area showed a more enriched carbon value in June 2015, which became depleted over time. Graphs also suggest that trophic interactions were similar in the three areas (Fig. 5). However, the species *Apseudopsis latreillii* presented a clear difference in the disposal area; the nitrogen signal experienced an increase in the most recent survey, while it remained at the same levels in the control areas.

4. Discussion

The community structure in the disposal area displayed marked differences in relation to the nearby environment. Shannon's diversity, richness and the biotic indices showed that the disposal area had developed a more diverse and rich community with higher EQS values than control areas established a few kilometres away. On the other hand, the most recent disposal event, carried out in the summer of 2015, seemed not to permanently affect to any of the indices measured, nor the community structure. Furthermore, other findings in this disposal area include the fact that the natural shifts in abundance and the community structure of the control areas were absent in the disposal area. However, it is difficult to establish whether these effects could be due to the most recent disposal event, or to the recurrent disposals since 2010. On the other hand, these effects on the community have not been reflected clearly in the trophic food web. Although isotopic sediment signals in the disposal area could be influenced by the historical disposals, the isospace occupied by all the organisms was similar in the three areas. This suggests the same use of basal resources, as well as the same trophic interactions over time. Only a tanaid species showed an

important increase in their nitrogen value a year after the disposal.

Our results showed that univariate indices such as diversity and richness reflected the changes driven by the historical disposal operations over time better than the biotic indices AMBI and BENFES. Only, BENTIX was able to differentiate the EQS of the disposal area from the controls, however, it did not show any pattern across the sampling dates. This agrees with Simboura et al. (2007), who found that BENTIX was useful for assessing the long term trends of community health in a metalliferous waste dumping. Total abundance and the Shannon's diversity indices also assessed the temporal changes over the sampling dates more effectively. This agrees with Katsiaras et al. (2015) and Ware et al. (2009), who pointed out that most sensitive ecological indicators of dumping effects were richness and abundance rather than AMBI or BENTIX. On the other hand, Taupp and Wetzel (2013) found that species richness, Shannon's diversity and AMBI among others were able to reflect the effects of dumping in estuaries.

Most of the species found in the control areas are characteristic of muddy bottoms, while a mixture of species characterised by both sandy and muddy bottoms were observed in the disposal area. The latter could be explained by the translocation of species from the original dredged material and by the immigration of species from the surrounding muddy community. In a soft bottom, habitat variability and structure are considered as key factors in biodiversity patterns (Hewitt et al., 2008; Reise, 2002; Zajac, 2008), and, generally, it is widely accepted that environments that show a great range of structural components can support a larger number of species and greater abundance (Carvalho et al., 2017). The sea bottom in the control areas was characterised by a very stable sedimentary and hydrological environment, typical of areas close to the mouth of great rivers, and host a biocoenosis typical of coastal terrigenous muds (Bellan-Santini et al., 1994) or *Amphiura* assemblages according to other authors, such as Thorson (1957). Consequently, these areas show a well-defined animal assemblage, so that the higher diversity and species richness in the disposal area could be due to the increase in structural complexity by the regular addition of more complex sediments characterised by lower % of fines. Johnson and Frid (1995), found similar results and suggested that it appears to be consistent with predictions of the Intermediate Disturbance Hypothesis (IDH) proposed by Connell (1978).

Differences in the temporal variation of the abundance in the disposal area compared to the controls could be explained by two factors: (1) Changes suffered by this area over the years has modified the natural seasonality of the species or (2) The most recent disposal event could lead to physical and chemical disturbances, which affect the macrofaunal community. Disposal could affect seasonal recruitment, affecting the species abundance (Manning et al., 2014). In fact, SIMPER analysis showed a decrease in the abundance of some species in the disposal area after the disposal of 2015. Furthermore, significant interaction between "Impact vs Control" and "Dates" in the abundance analysis, as well as the community structure, indicated a possible effect on the impacted area. Analyses revealed no changes in the disposal area over the dates, while control areas showed natural differences.

According to a Before After Control Impact (Underwood, 1991), if the putatively impacted area had changed over time from before to after the most recent disposal event, with a different pattern when compared to the controls, it would mean a permanent impact from the most recent disposal event. However, the absence of any interaction between the factors "Impact vs Control" and "Time" indicated that there was not a permanent effect from the most recent disposal. The relatively high diversity, richness and abundance found after the disposal could be attributed to a gradual and homogenous deposition of dredge sediment over a relatively large area. The thin layer created would permit the survival of a high proportion of different species, as in the study carried out by Simonini et al. (2005b). Organisms have the capacity to burrow into the sediment in order to avoid burial, making them capable of tolerating these perturbations (Powilleit et al., 2009). This would be in accordance with Marmin et al. (2016), who proposed

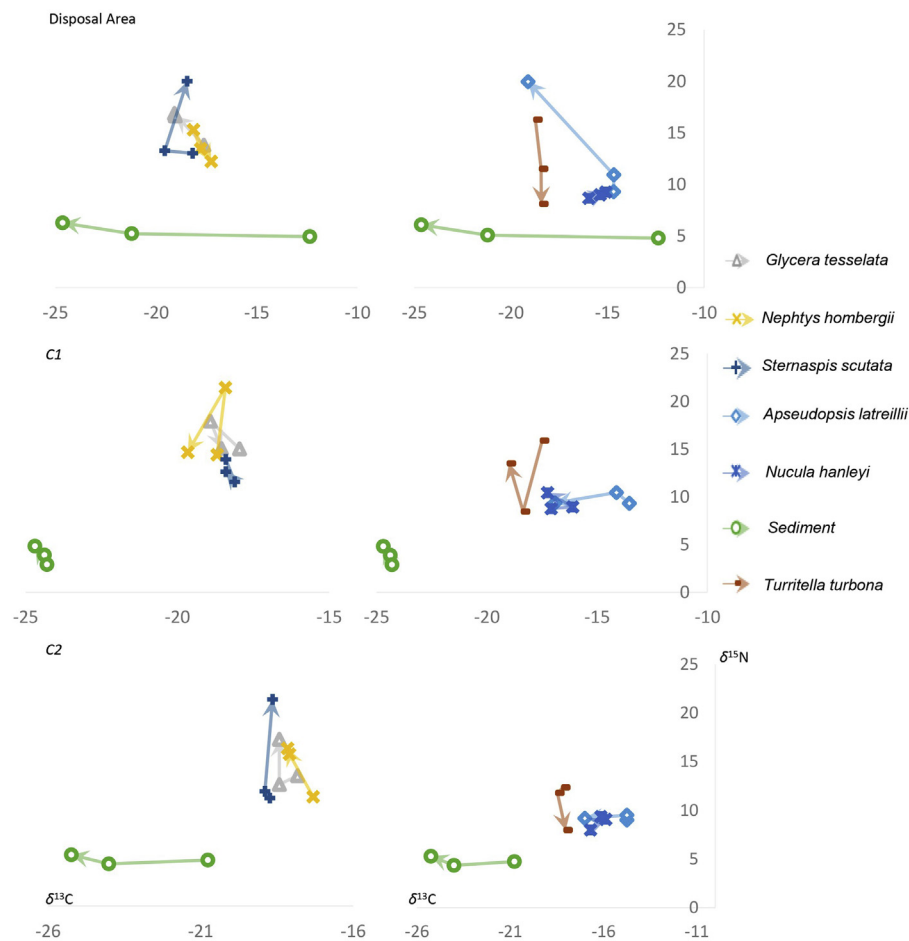


Fig. 5. $\delta^{13}\text{C}$ (x-axis) and $\delta^{15}\text{N}$ (y-axis) means of the organisms collected in the Disposal area (upper), C1 (middle) and C2 (bottom). Arrows represents isotopic variation over the three sampling dates. Polychaetes have been separated for better visualisation (left column). The x-axis is not always on the same scale.

that more dispersive disposals over a wider area were recommended, rather than depositions at a single point. Furthermore, organisms situated in a perturbed zone present more resilience to perturbations (Bolam and Rees, 2003). Furthermore, a rapid recovery after a disposal has been attributed to the great potential of juvenile settlement and adult colonisation from the edges (Bolam and Rees, 2003; Fredette and French, 2004; Munari and Mistri, 2014).

Several studies have found recovery in disposal areas in terms of abundance, richness and diversity, but not in terms of the community structure (OSPAR, 2008). Changes in macrofaunal community structure have been reported in many disposal effect studies: (e.g. Harvey et al., 1998; Zimmerman et al., 2003; Witt et al., 2004; Bolam et al., 2006; Powilleit et al., 2006; Ware et al., 2010; Taupp and Wetzel, 2013; Munari and Mistri, 2014; Katsiaras et al., 2015). However, other studies, such as Roberts and Forrest (1999), Smith and Rule (2001), Stronkhorst et al. (2003), Simonini et al. (2005b) and Bolam et al. (2011), found scarce indication of impact in their respective disposal areas. Our results pointed to an influence of the dredge material disposal in terms of community structure and a lack of temporal variation in the disposal area. Conversely, the control areas showed a very similar community trend with significant temporal variations. This could again be related to the changes established by the periodic sediment input from maintenance dredging for channel navigation in the Guadalquivir River, or at least to the most recent disposal event in the summer of 2015.

Recovery patterns after a disturbance, such as dredging and disposal operations, seem to take different pathways in every study, and depend on several factors (Newell et al., 1998). Bolam and Rees (2003) pointed

out that communities of naturally stressed environments could recover better than more stable zones. Shallower zones with physical disturbances tend to take less than one year to recover in many cases, while deeper, more stable zones take more time to converge with the initial community structure. Despite the fact that an assessment of recovery in an on-going disposal area is difficult, our results are consistent with OSPAR (2008), where it was pointed out that community structure often fails to converge with natural reference areas. SIMPER analysis showed that one month after the last disposal event was the date with the highest similarity between the control and the disposal area. This could be explained by the decrease of some species in the disposal area that were contributing to these differences before. In this month, the abundance of some abundant species also increased in both control and disposal areas. Six months and a year after the summer 2015 disposal, the community structure of the disposal area showed more dissimilarity with the control areas. This was due to the different patterns of change in terms of species common to both the controls areas and the disposal area, and the appearance of species in the disposal area that were not present in the control areas.

The tanaid *Apseudopsis latreillii* was the main species contributing to the dissimilarity found between the disposal area and the control areas after the disposal of 2015. This organism did not experience such a pronounced increase in its abundance in the disposal area as it did in the two control areas. This species usually spawns during late summer and early autumn, reaching high densities in the soft bottoms of this geographical area (Sánchez-Moyano et al., 2007). de-la-Ossa-Carretero et al. (2010) indicated that *A. latreillii* responds to sewage disposal with a decrease in population density and, furthermore, smaller juvenile

individuals appear to be particularly sensitive, since these individuals normally live near the surface and are more exposed to disturbance. Consequently, the coincidence of dumping with the reproductive peak may have adversely affected the populations of this species. However, another species that was very abundant among all studied areas was the decapod *Upogebia tipica* (and also, in smaller numbers, *U. deltaura*), which was not affected by the dumping. Upogebiidae usually need sandy and muddy substrates to construct long burrows (deeper than 20–30 cm) where they obtain shelter, protection from predators and appropriate conditions for feeding and reproduction (Coelho et al., 2000). *Upogebia* spp. occur in the coast of the Gulf of Cádiz between depths of 20 and 40 m and generally show a reproductive event during the spring-summer period with a planktonic larval phase of around three weeks (Pires et al., 2013). Their reproductive cycle prior to sediment disposal, and their deep burrows, could be the reasons why there were no differences between the control and the disposal areas. Similar results have been obtained with other abundant burrowing species, such as the polychaetes *Lumbrineris latreilli*, *Nephtys hombergii* and *Sternaspis scutata*.

Carbon isotopic signals of sediment in the disposal area could suggest a perturbation on the impacted area before the disposal of summer 2015 due to the periodical inputs of more carbon-enriched sediments from the estuary. However, this point was not reflected in the trophic food web, since analysed species showed similar carbon signal in the disposal area and in the two control areas. Since all species showed similar isotope signals over time, it suggests that the origin of organic matter was similar in the three areas. Furthermore, after the disposal in the summer of 2015, the isotopic signal of the sediment in the disposal area was similar to the control areas. Nitrogen isotopic values also suggested similar trophic niches for most organisms in the three areas across all sampling dates. However, *Apseudopsis latreilli* showed an important increase in their nitrogen values in the marine dump with respect to the control areas in the August 2016 survey. Variation in the tanaid isotopic signal could be attributed to the most recent disposal event, although more specific studies would be necessary to demonstrate this question. Change in diet composition is not immediately reflected by tissue isotopic values, which integrate trophic information over a certain period of time (Sampaio et al., 2010; Sweeting et al., 2005). This could explain the lack of differences in the October 2015 survey. Since there were no changes in nitrogen signal of the basal resources, increase in the nitrogen signal of that species could be explained by a change in the degree of trophic omnivory (Post and Takimoto, 2007). These results contrast with other studies where changes in the trophic structure of disposal impacted areas have been found (Bolam, 2012; Munari and Mistri, 2014; Pezy et al., 2017; Simonini et al., 2005a). However, these studies were focused on functional traits and secondary production rather than a stable isotopic approach.

5. Conclusion

The recurrent disposals constitute pulse disturbances, but repeated

periodically, which could produce: (1) recurrent pulse or short-term responses in the biological community followed by a convergence towards the reference communities, (2) a permanent change or (3) a combination of these two effects (Underwood, 1994). The results of this study suggest that a permanent change exists. However, we cannot discard some short-term effects of most recent disposal event such as the affection of the recruitment of some species.

The impact of depositing dredge material in marine environments depends on various factors, such as the amount of sediment, dumping season, water depth, currents and similarity of dredge material and the native sediment (Powilleit et al., 2006; Marmin et al., 2014). For this reason, the impacts of dredge-material disposal are mostly site-specific (OSPAR, 2008). There are few studies assessing the effects of dumping in a historic marine dump with well-established macrofaunal communities such as Ware et al. (2009, 2010) for example. Our data suggest that in the disposal area, where a disposal operation occurs periodically, there have been changes that have affected biological characteristics permanently. Contrary to other studies, the continuous disposal of dredge material from the Guadalquivir estuary has increased richness and diversity indices, although these historical impacts, as well as the most recent disposal event, could have led to the natural loss of seasonality in the marine dump. Conversely, the food web has not been affected. Furthermore, disposal timing is an important factor to consider. To minimise the impact on soft-bottom communities, dredging and disposal operations should avoid the main reproduction and recruitment periods and be developed, where technical aspects make it possible, in winter months (Sánchez-Moyano et al., 2004; Ceia et al., 2013). Our results confirm the high variability of impacts due to disposal operations and confirm again the 'site-specific' character of these perturbations. We suggest that impact studies are needed in every disposal operation work across temporal and spatial scales. Classic approaches and new techniques are also required to assess the effects of these anthropogenic impacts at different levels in macrofaunal soft-bottom communities such as functional and trophic traits. Using stable isotope techniques to assess effects of these perturbations appears to be an interesting tool that has never been used in dredging and dumping studies.

Acknowledgments

We thank Autoridad Portuaria de Sevilla (APS) and Acuario de Sevilla for financial and logistical support. This work was partially supported by the V Plan Propio Universidad de Sevilla via two pre-doctoral grants (I. Donázar-Aramendía, J.M. Miro). Isotopic analyses were carried out at the Laboratorio de Isótopos Estables of the Estación Biológica de Doñana (LIE-EBD, Spain; www.ebd.csic.es/lie/index.html). We thank Dr. Craig Layman for his invaluable help with stable isotope interpretation.

Supplementary data

Table S1

Average abundance (Av. Abund.) of the most relevant species of the Impact vs Control areas over all sampling dates. Species are listed in decreasing order according to their contribution to the average of the dissimilarity (Av. Diss.) between areas until 50% of the accumulated total similarity.

Impact vs Controls June 2015	I	C	Average dissimilarity: 44.52		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Apseudopsis latreilli</i>	21.96	35.48	2.43	2.38	5.46
<i>Magelona papillicornis</i>	16.02	7.66	1.78	1.57	4

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Table S1 (continued)

Impact vs Controls June 2015	I	C	Average dissimilarity: 44.52		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Hyalia vitrea</i>	7.07	7.9	1.56	1.54	3.5
<i>Sternaspis scutata</i>	7.33	13.02	1.55	2.8	3.49
<i>Spiophanes kroyeri</i>	11.39	2.47	1.52	1.62	3.41
<i>Eudorella truncatula</i>	17.11	9.2	1.5	2.31	3.37
<i>Sigambra parva</i>	11.94	3.84	1.44	1.86	3.23
<i>Medicorophium runcicorne</i>	7.58	5.55	1.05	3.26	2.35
<i>Capitella capitata</i>	7.49	9.88	1.04	1.26	2.33
<i>Medicorophium aculeatum</i>	4.3	2.15	1.01	0.77	2.28
<i>Nephtys hombergii</i>	11.21	6.73	0.8	0.97	1.8
<i>Iphinoe tenella</i>	4	0	0.79	2.21	1.76
<i>Bodotria scorioides</i>	4.55	0	0.75	1.17	1.69
<i>Glycera tessellata</i>	5.76	1.65	0.75	1.8	1.69
<i>Nassarius incrassatus</i>	3.84	0	0.72	3.86	1.61
<i>Microspio mecznikowianus</i>	7.95	4.47	0.7	1.47	1.57
<i>Upogebia tipica</i>	3.8	0	0.69	5.3	1.56
<i>Nemertea</i>	13.34	9.67	0.69	1.64	1.56
<i>Aonides oxycephala</i>	8.19	4.79	0.67	1.65	1.49
<i>Oestergrenia digitata</i>	4.32	4.56	0.65	1.67	1.47
<i>Cheirocratus sundeavalli</i>	1.72	3.29	0.59	1.26	1.34

Impact vs Controls July 2015	I	C	Average dissimilarity: 36.66		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Apseudopsis latreillii</i>	30.66	38.92	2.02	1.23	5.5
<i>Magelona papillicornis</i>	13.67	7.77	1.38	1.61	3.77
<i>Spiophanes kroyeri</i>	7.95	1.47	1.34	2.72	3.66
<i>Hyalia vitrea</i>	8.25	11.48	1.3	1.57	3.53
<i>Nephtys hombergii</i>	11.37	5.71	1.24	1.44	3.39
<i>Lumbrineris latreillii</i>	15.08	12.2	1.13	1.46	3.09
<i>Eudorella truncatula</i>	9.69	4.61	1.06	1.93	2.88
<i>Sigambra parva</i>	8.27	3.27	1.03	1.84	2.82
<i>Medicorophium runcicorne</i>	7.24	2.33	1.01	1.36	2.75
<i>Glycera tessellata</i>	5.49	1.29	0.89	2.67	2.42
<i>Spisula subtruncata</i>	4.55	0	0.86	0.69	2.36
<i>Corbula gibba</i>	11.19	7.11	0.86	1.49	2.34
<i>Mysia undata</i>	3.8	0	0.79	5.04	2.16
<i>Phoronida</i>	5.79	2.21	0.78	1.37	2.14
<i>Sternaspis scutata</i>	7.28	10.2	0.75	1.24	2.05
<i>Nucula hanleyi</i>	8.48	6.59	0.72	1.55	1.96
<i>Turritella turbona</i>	4.2	0.86	0.71	2.52	1.94
<i>Urothoe grimaldii</i>	3.33	0	0.69	1.32	1.88

Impact vs Controls October 2015	I	C	Average dissimilarity: 31.49		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Apseudopsis latreillii</i>	48.98	71.95	4.28	1.36	13.59
<i>Capitella capitata</i>	3.93	13.35	1.71	2.32	5.43
<i>Nephtys hombergii</i>	11.6	6.33	0.96	1.14	3.06
<i>Hemilepton nitidum</i>	6.05	10.69	0.95	1.36	3.03
<i>Upogebia tipica</i>	11.94	10.46	0.88	1.23	2.79
<i>Eudorella truncatula</i>	3.93	8.32	0.87	1.42	2.78
<i>Hyalia vitrea</i>	6.24	10.08	0.86	1.32	2.75
<i>Lumbrineris latreillii</i>	13.12	15.29	0.86	1.54	2.74
<i>Sternaspis scutata</i>	10.13	11.22	0.8	1.46	2.54
<i>Phoronida</i>	4.54	0.43	0.74	2.6	2.35
<i>Corbula gibba</i>	10.64	6.79	0.7	2.05	2.24

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Table S1 (continued)

Impact vs Controls June 2015	I	C	Average dissimilarity: 44.52		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Upogebia deltaura</i>	2.35	6.03	0.67	1.66	2.14
<i>Turritella turbona</i>	6.74	3.27	0.65	1.38	2.07
<i>Chamelea gallina</i>	3.44	0	0.63	2.64	1.99
<i>Neanthes fucata</i>	4.28	0.86	0.62	1.81	1.98

Impact vs Controls March 2016	I	C	Average dissimilarity: 44.00		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Hemilepton nitidum</i>	12.64	0.61	2.92	1.22	6.63
<i>Upogebia tipica</i>	12.84	1.67	2.88	1.34	6.53
<i>Apseudopsis latreillii</i>	21.73	28.68	2.59	1.78	5.88
<i>Magelona papillicornis</i>	10.99	7.1	1.98	2.06	4.49
<i>Capitella capitata</i>	6.44	13.48	1.82	1.53	4.14
<i>Terebellides stroemii</i>	0	6.71	1.55	5.04	3.51
<i>Medicorophium runcicorne</i>	7.61	1.9	1.33	2.12	3.01
<i>Corbula gibba</i>	8.16	3.43	1.1	1.91	2.51
<i>Hyalia vitrea</i>	6.9	9.75	1.01	1.26	2.29
<i>Amphiura chiajei</i>	6.31	2.04	0.99	2.4	2.25
<i>Urothoe grimaldii</i>	4.7	0.43	0.99	3.92	2.24
<i>Aonides oxycephala</i>	11.19	7.2	0.94	1.97	2.14
<i>Aricidea catherinae</i>	1.72	5.1	0.87	1.67	1.98
<i>Lagis koreni</i>	3.65	0	0.84	15.35	1.91
<i>Eudorella truncatula</i>	9.06	6.91	0.78	1.37	1.78

Impact vs Controls August 2016	I	C	Average dissimilarity: 42.01		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Apseudopsis latreillii</i>	21.5	19.9	2.07	1.24	4.92
<i>Corbula gibba</i>	17.95	8.56	2.06	1.45	4.9
<i>Magelona papillicornis</i>	13.67	5.92	1.66	1.63	3.95
<i>Aonides oxycephala</i>	12.16	5.2	1.42	1.73	3.37
<i>Nephtys hombergii</i>	13.82	8.19	1.15	1.32	2.73
<i>Eudorella truncatula</i>	8.47	3.4	1.14	1.75	2.72
<i>Spiophanes kroyeri</i>	5.83	0.43	1.08	1.53	2.57
<i>Hyalia vitrea</i>	10.01	11.45	1.05	1.46	2.5
<i>Photis longicaudata</i>	5.26	0.43	1.03	3.7	2.44
<i>Lumbrineris latreilli</i>	15.44	14.86	0.98	1.42	2.34
<i>Medicorophium aculeatum</i>	3.44	1.67	0.93	0.77	2.21
<i>Glycera tessellata</i>	8.07	3.47	0.91	1.4	2.16
<i>Urothoe grimaldii</i>	4.39	0.43	0.87	1.39	2.07
<i>Phoronida</i>	5.07	1.04	0.87	1.73	2.06
<i>Thracia phaseolina</i>	4.13	0	0.75	0.69	1.79
<i>Nucula hanleyi</i>	8.2	6.67	0.73	2.1	1.75
<i>Phaxas pellucidus</i>	3.65	0	0.73	1.36	1.75
<i>Tellina cf. compressa</i>	3.72	0	0.71	1.07	1.69
<i>Capitella capitata</i>	12.9	15.92	0.69	1.2	1.65
<i>Cossura soyeri</i>	2.94	0	0.68	1.33	1.62

Table S2

Average abundance (Av. Abund.) of the most relevant species over the sampling dates of samples in the control areas. Species are listed in decreasing order according to their contribution to the average of the dissimilarity (Av. Diss.) between areas until 50% of the accumulated total similarity.

June vs July 2015	Jun	Jul	Average dissimilarity: 30.25		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Apseudopsis latreillii</i>	35.48	38.92	1.55	1.23	5.12
<i>Hyalia vitrea</i>	7.9	11.48	1.43	1.18	4.71
<i>Upogebia tipica</i>	0	5.06	1.28	4.39	4.22
<i>Eudorella truncatula</i>	9.2	4.61	1.16	1.55	3.85
<i>Capitella capitata</i>	9.88	8.02	0.92	1.25	3.05
<i>Ampelisca diadema</i>	5.28	2.51	0.9	2.3	2.97
<i>Hemilepton nitidum</i>	0	3.53	0.89	1.72	2.94
<i>Sternaspis scutata</i>	13.02	10.2	0.86	1.49	2.84
<i>Medicorophium runcicorne</i>	5.55	2.33	0.8	1.96	2.66
<i>Kurtiella bidentata</i>	0	2.82	0.7	1.35	2.33
<i>Magelona papillicornis</i>	7.66	7.77	0.69	1.23	2.28
<i>Microspio mecznikowianus</i>	4.47	2.35	0.69	1.32	2.28
<i>Nucula hanleyi</i>	4.41	6.59	0.68	1.4	2.24
<i>Turritella turbona</i>	2.92	0.86	0.67	1.36	2.2
<i>Cheirocratus sundevalli</i>	3.29	2.64	0.65	1.36	2.16
<i>Nemertea</i>	9.67	7.44	0.64	1.49	2.11
<i>Aonides oxycephala</i>	4.79	2.51	0.62	1.26	2.06

June vs October 2015	Jun	Oct	Average dissimilarity: 36.15		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Apseudopsis latreillii</i>	35.48	71.95	7.46	4.53	20.64
<i>Hemilepton nitidum</i>	0	10.69	2.17	2.57	5.99
<i>Upogebia tipica</i>	0	10.46	2.11	1.81	5.83
<i>Aonides oxycephala</i>	4.79	11.28	1.33	1.99	3.68
<i>Hyalia vitrea</i>	7.9	10.08	1.1	1.33	3.03
<i>Nucula hanleyi</i>	4.41	9.66	1.07	2.07	2.95
<i>Leucothoe incisa</i>	1.47	6.4	1.01	2.21	2.8
<i>Capitella capitata</i>	9.88	13.35	0.91	1.61	2.52
<i>Ampelisca diadema</i>	5.28	1.18	0.9	1.84	2.49
<i>Sternaspis scutata</i>	13.02	11.22	0.63	1.61	1.75

July vs October 2015	Jul	Oct	Average dissimilarity: 33.27		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Apseudopsis latreillii</i>	38.92	71.95	7.05	3.84	21.18
<i>Aonides oxycephala</i>	2.51	11.28	1.86	3.05	5.6
<i>Hemilepton nitidum</i>	3.53	10.69	1.53	1.63	4.61
<i>Upogebia tipica</i>	5.06	10.46	1.32	1.29	3.98
<i>Capitella capitata</i>	8.02	13.35	1.26	2.02	3.8
<i>Leucothoe incisa</i>	1.9	6.4	0.95	2.16	2.86
<i>Hyalia vitrea</i>	11.48	10.08	0.9	1.19	2.71
<i>Eudorella truncatula</i>	4.61	8.32	0.84	1.57	2.52
<i>Upogebia deltaura</i>	2.26	6.03	0.79	1.65	2.38
<i>Nucula hanleyi</i>	6.59	9.66	0.74	1.44	2.23

October 2015 vs March 2016	Oct	Mar	Average dissimilarity: 37.58		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Apseudopsis latreillii</i>	71.95	28.68	9.29	5.73	24.71

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Table S2 (continued)

June vs July 2015	Jun	Jul	Average dissimilarity: 30.25		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Hemilepton nitidum</i>	10.69	0.61	2.14	2.34	5.69
<i>Upogebia tipica</i>	10.46	1.67	2.03	1.6	5.39
<i>Upogebia deltaura</i>	6.03	0	1.28	4.04	3.42
<i>Sternaspis scutata</i>	11.22	5.39	1.27	1.73	3.37
<i>Leucothoe incisa</i>	6.4	1.35	1.09	2.02	2.9
<i>Hyala vitrea</i>	10.08	9.75	1.01	1.39	2.68
<i>Terebellides stroemii</i>	2.21	6.71	0.97	2.04	2.59

March 2016 vs August 2016	Mar	Aug	Average dissimilarity: 31.96		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Apseudopsis latreillii</i>	28.68	19.9	2.67	1.41	8.34
<i>Hyala vitrea</i>	9.75	11.45	1.48	1.37	4.62
<i>Corbula gibba</i>	3.43	8.56	1.47	1.8	4.59
<i>Terebellides stroemii</i>	6.71	1.65	1.45	2.13	4.52
<i>Eudorella truncatula</i>	6.91	3.4	1.05	1.17	3.3
<i>Kurtiella bidentata</i>	0.61	4.15	1.05	1.31	3.27
<i>Spiophanes kroyeri</i>	3.82	0.43	0.97	1.4	3.02
<i>Lumbrineris latreilli</i>	12.96	14.86	0.93	1.44	2.9
<i>Capitella capitata</i>	13.48	15.92	0.92	1.22	2.88
<i>Ampelisca diadema</i>	2.76	5.71	0.82	1.4	2.58
<i>Magelona papillicornis</i>	7.1	5.92	0.81	1.14	2.54
<i>Venus casina</i>	0.43	3.08	0.77	1.7	2.41
<i>Spiochaetopterus costarum</i>	0.43	3.12	0.77	1.73	2.4
<i>Upogebia tipica</i>	1.67	1.92	0.76	0.8	2.39
<i>Turritella turbona</i>	2.82	0.43	0.75	1.34	2.33

Table S3

Average abundance (Av. Abund.) of the most relevant species over the sampling dates of samples in the disposal area. Species are listed in decreasing order according to their contribution to the average of the dissimilarity (Av. Diss.) between areas until 50% of the accumulated total similarity.

June vs July 2015	Jun	Jul	Average dissimilarity: 41.20		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Apseudopsis latreillii</i>	21.96	30.66	2.01	2.02	4.87
<i>Magelona papillicornis</i>	16.02	13.67	1.45	1.55	3.53
<i>Hyala vitrea</i>	7.07	8.25	1.45	1.63	3.52
<i>Eudorella truncatula</i>	17.11	9.69	1.25	2.3	3.04
<i>Sternaspis scutata</i>	7.33	7.28	1.2	1.76	2.92
<i>Nephtys hombergii</i>	11.21	11.37	1	1.55	2.44
<i>Phoronida</i>	0	5.79	0.96	2.49	2.32
<i>Medicorophium runcicorne</i>	7.58	7.24	0.95	1.51	2.3
<i>Sigambra parva</i>	11.94	8.27	0.83	1.76	2.02
<i>Spisula subtruncata</i>	2.72	4.55	0.83	0.89	2.02
<i>Lumbrineris latreilli</i>	14.2	15.08	0.8	1.58	1.94
<i>Medicorophium aculeatum</i>	4.3	0	0.8	0.66	1.93
<i>Capitella capitata</i>	7.49	7.88	0.79	1.57	1.92
<i>Spiophanes kroyeri</i>	11.39	7.95	0.79	0.98	1.91
<i>Hemilepton nitidum</i>	0	4.35	0.72	2.15	1.75
<i>Iphinoe tenella</i>	4	0	0.69	2.18	1.67
<i>Bodotria scorpioides</i>	4.55	0	0.67	1.12	1.63
<i>Aonides oxycephala</i>	8.19	4.07	0.64	2.05	1.56
<i>Oostergrenia digitata</i>	4.32	2.35	0.64	1.2	1.54
<i>Nemertea</i>	13.34	9.87	0.63	1.55	1.53
<i>Nassarius incrassatus</i>	3.84	2.43	0.62	3.32	1.5

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Table S3 (continued)

June vs July 2015	Jun	Jul	Average dissimilarity: 41.20		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Ampelisca diadema</i>	6.14	3.94	0.61	1.26	1.48
<i>Nucula hanleyi</i>	5.49	8.48	0.61	1.48	1.48
June vs October 2015	Jun	Oct	Average dissimilarity: 45.93		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Apseudopsis latreillii</i>	21.96	48.98	4.41	1.46	9.61
<i>Eudorella truncatula</i>	17.11	3.93	2.22	3.48	4.84
<i>Magelona papillicornis</i>	16.02	9.55	1.53	1.75	3.32
<i>Upogebia tipica</i>	3.8	11.94	1.39	3.4	3.03
<i>Sternaspis scutata</i>	7.33	10.13	1.27	1.54	2.76
<i>Hyala vitrea</i>	7.07	6.24	1.26	1.5	2.74
<i>Spiophanes kroyeri</i>	11.39	4.36	1.09	1.07	2.37
<i>Medicorophium runcicorne</i>	7.58	2.35	1.08	1.84	2.34
<i>Hemilepton nitidum</i>	0	6.05	1.02	2.92	2.21
<i>Capitella capitata</i>	7.49	3.93	1	1.65	2.18
<i>Sigambra parva</i>	11.94	6.35	0.95	1.65	2.07
<i>Nephtys hombergii</i>	11.21	11.6	0.92	1.2	2.01
<i>Nemertea</i>	13.34	8.01	0.9	2.19	1.95
<i>Medicorophium aculeatum</i>	4.3	0	0.81	0.67	1.76
<i>Lumbrineris latreilli</i>	14.2	13.12	0.79	1.67	1.71
<i>Nucula hanleyi</i>	5.49	9.99	0.78	2.89	1.69
<i>Phoronida</i>	0	4.54	0.76	3.19	1.66
<i>Neanthes fucata</i>	0	4.28	0.72	2.77	1.57
<i>Iphinoe tenella</i>	4	0	0.7	2.21	1.52
July vs October 2015	Jul	Oct	Average dissimilarity: 34.28		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Apseudopsis latreillii</i>	30.66	48.98	3.89	1.47	11.34
<i>Upogebia tipica</i>	4.07	11.94	1.45	2.09	4.24
<i>Lumbrineris latreilli</i>	15.08	13.12	1.15	1.38	3.34
<i>Hyala vitrea</i>	8.25	6.24	1.13	2.38	3.29
<i>Eudorella truncatula</i>	9.69	3.93	1.02	1.45	2.99
<i>Nephtys hombergii</i>	11.37	11.6	0.99	1.35	2.9
<i>Magelona papillicornis</i>	13.67	9.55	0.94	1.83	2.73
<i>Medicorophium runcicorne</i>	7.24	2.35	0.92	1.48	2.68
<i>Aonides oxycephala</i>	4.07	9.1	0.91	2.21	2.64
<i>Sternaspis scutata</i>	7.28	10.13	0.9	1.38	2.63
<i>Spisula subtruncata</i>	4.55	0.86	0.81	0.82	2.37
<i>Capitella capitata</i>	7.88	3.93	0.73	1.2	2.14
<i>Spiophanes kroyeri</i>	7.95	4.36	0.71	1.13	2.07
<i>Mysia undata</i>	3.8	0	0.68	4.66	1.98
<i>Abra nitida</i>	3.21	3.14	0.56	1.52	1.63
<i>Ampelisca diadema</i>	3.94	3.21	0.55	1.72	1.59
October 2015 vs March 2016	Oct	Mar	Average dissimilarity: 38.99		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Apseudopsis latreillii</i>	48.98	21.73	5.47	1.54	14.02
<i>Hemilepton nitidum</i>	6.05	12.64	2.07	1.58	5.31
<i>Upogebia tipica</i>	11.94	12.84	1.71	2.61	4.38
<i>Magelona papillicornis</i>	9.55	10.99	1.5	1.73	3.86
<i>Capitella capitata</i>	3.93	6.44	1.07	1.24	2.74
<i>Eudorella truncatula</i>	3.93	9.06	1.03	1.48	2.65

(continued on next page)

Table S3 (continued)

June vs July 2015	Jun	Jul	Average dissimilarity: 41.20		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Sternaspis scutata</i>	10.13	5.56	1.03	1.2	2.65
<i>Medicorophium runcicorne</i>	2.35	7.61	1	2.2	2.56
<i>Lumbrineris latreilli</i>	13.12	14.05	0.93	1.42	2.38
<i>Nephtys hombergii</i>	11.6	8.79	0.9	1.2	2.31
<i>Sigambra parva</i>	6.35	3.44	0.74	1.49	1.89
<i>Chamelea gallina</i>	3.44	0	0.66	2.56	1.7
<i>Neanthes fucata</i>	4.28	0.86	0.66	1.76	1.7
Nemertea	8.01	5.8	0.63	1.42	1.62
<i>Leucothoe incisa</i>	4.7	3.14	0.57	2.29	1.46

March 2016 vs August 2016	Mar	Aug	Average dissimilarity: 43.87		
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
<i>Upogebia tipica</i>	12.84	0.86	2.34	1.37	5.33
<i>Hemilepton nitidum</i>	12.64	3.65	2.17	1.12	4.96
<i>Apseudopsis latreilli</i>	21.73	21.5	2.02	1.14	4.61
<i>Corbula gibba</i>	8.16	17.95	1.77	1.34	4.03
<i>Magelona papillicornis</i>	10.99	13.67	1.56	1.56	3.56
<i>Capitella capitata</i>	6.44	12.9	1.39	1.39	3.17
<i>Nephtys hombergii</i>	8.79	13.82	0.97	1.18	2.21
<i>Photis longicaudata</i>	0	5.26	0.95	8.51	2.17
<i>Aricidea catherinae</i>	1.72	6.75	0.94	1.72	2.15
<i>Hyalia vitrea</i>	6.9	10.01	0.93	1.7	2.12
<i>Medicorophium runcicorne</i>	7.61	3.44	0.91	1.46	2.07
<i>Lumbrineris latreilli</i>	14.05	15.44	0.86	1.49	1.96
<i>Venus casina</i>	1.22	5.37	0.78	1.97	1.78
<i>Sigambra parva</i>	3.44	6.85	0.78	1.49	1.78
<i>Turritella turbona</i>	4.16	0	0.77	5.02	1.75
<i>Thracia phaseolina</i>	0.86	4.13	0.73	0.83	1.65
<i>Medicorophium aculeatum</i>	0	3.44	0.71	0.67	1.61
<i>Spiophanes kroyeri</i>	6.48	5.83	0.7	4.35	1.6
<i>Spisula subtruncata</i>	1.72	3.33	0.67	1.23	1.53

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