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Maintenance dredging impacts on a highly stressed estuary (Guadalquivir estuary): A BACI approach through oligohaline and polyhaline habitats

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ABSTRACT

Understanding the effects of dredging in estuaries is a hard task due to the difficulty of implementing an adequate environmental diagnosis, as a consequence of the salinity gradient and anthropogenic disturbances. To assess the effects of maintenance dredging work on the Guadalquivir estuary (southwestern Spain), we used a Before-After-Control-Impact (BACI) approach to determine both direct and indirect effects in two salinity ranges. No effects were found on water and sediment physicochemical characteristics. The small impacts on dredged areas were followed by a rapid recovery of opportunistic species. The poor status of the benthos does not permit the detection of significant effects on macrofaunal community structure. The use of stable isotopes analysis to determine impacts on food web structure showed that changes over time seem to be explained by natural temporal variation rather than the dredging works. This paper emphasises the need to define proper management and conservation plans to improve the status of the benthic communities of the Guadalquivir estuary.

1. Introduction

Although estuaries are one of the most productive marine coastal environments in terms of biomass (Wolf, 1983; Wetzel et al., 2013), they often face perturbations (Dauvin et al., 2006; Sánchez-Moyano and García-Asencio, 2010). With more than 60% of Earth's population living in the coastal realm, estuarine ecosystems have been extensively altered by human activities (Ray, 2006). Furthermore, estuaries are dynamic and complex systems where high variability of the physical-chemical gradients makes them one of the most stressful aquatic environments (González-Ortegón et al., 2006; Dauvin, 2008). In this changeable scenario, characteristics of estuarine communities are strongly and directly related to parameters, such as turbidity, temperature and, particularly, salinity (Baldó and Cuesta, 2005; Dauvin, 2008). As a consequence, benthic community diversity is limited, but it is often associated with a high tolerance to variable environmental conditions (Dauvin, 2007). Interpreting disturbance effects in estuaries often is complex, because the dynamic geological, physical and chemical characteristics that rule those systems might be confused with anthropogenic impacts (Morrisey et al., 2003; Dauvin et al., 2006; Dauvin, 2008). An accurate evaluation of the anthropogenic impacts in estuaries is vital for the proper management of resources and maintaining good environmental health as well as reaching a "good environmental status" in the context of the requirements of the European Water Framework Directive (Taupp and Wetzel, 2013; Rehitha et al., 2017).

The Guadalquivir estuary (southwestern Spain) is a good example of this kind of stressed scenario. In this system, mixed natural perturbations, such as a horizontal salinity gradient, govern the composition and spatial distribution of the aquatic communities, while human activities have deeply modified the ecosystem (González-Ortegón et al., 2006; Castañeda and Drake, 2008; Llope, 2017). They vary from desiccation of tidal marshes and isolation of the estuary course from the original tidal marshes, reduction of freshwater inputs and eutrophication from urban and agricultural waters to maintenance dredging work (Taglialatela et al., 2014; Llope, 2017). The Guadalquivir estuary is the only navigable river in Spain and gives access to Seville harbour. To maintain navigability, the Autoridad Portuaria de Sevilla (APS) has performed maintenance dredging work every one or two years since 1985 (Gallego and García Novo, 2006). Dredging operations represent a potential risk to the estuarine environment; effects basically depend on the method used, duration and extension, amount of dredge material and sediment characteristics. These activities may cause changes in the seabed and natural fluctuations in water conditions, population dynamics and sedimentary composition of the system and the surrounding

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areas (Sánchez-Moyano et al., 2004; Barrio Froján et al., 2011; Ceia et al., 2013; Rehitha et al., 2017). Dredging often has more repercussions on benthic communities due to the relative immobility of organisms (Simonini et al., 2005). Macrofaunal communities play a crucial role in the structure and functioning of ecosystems, such as sediment stability, nutrient processing and contaminant sequestering (Thrush and Dayton, 2002; Ceia et al., 2013). In estuaries, macrofauna are also an important link between organic matter and predators (Kon et al., 2015) acting as a food source for the next trophic level, generally secondary consumers such as fish and shellfish (Bolam et al., 2011).

Studies assessing dredging effects on macrofaunal assemblages are widely available (Klapan et al., 1975; Newell et al., 1998; Sánchez-Moyano et al., 2004; Bemvenuti et al., 2005; Ponti et al., 2009; Rehitha et al., 2017). However, more focused studies on dredging effects in different salinity ranges in estuaries are rare, despite the fact that salinity is the major environmental factor influencing the distribution of organisms in estuaries (Attrill, 2002). Most monitoring programs in estuaries have been developed in higher salinity ranges, while low salinity areas have been scarcely studied (Vinagre et al., 2015). Moreover, studies analysing dredging impacts on food web structure are few. Stable isotopes analysis is a useful tool to determine anthropogenic impacts on food web structure in aquatic ecosystems (Ke et al., 2016). Nitrogen and carbon isotopic ratios can be used for tracing the natural or anthropogenic sources of nutrients in estuaries (Castro et al., 2007; Kon et al., 2012; Van De Merwe et al., 2016). Also, the different rates of nutrient assimilation by different organisms can reflect estuarine status over temporal scales (Van De Merwe et al., 2016). For this reason, isotope analysis could be a useful tool to assess dredging impacts and the potential following recovery.

In this context, we analysed the effects of dredging work carried out in the Guadalquivir estuary in two different salinity gradient ranges with a Before-After Control-Impact (BACI) analysis (Underwood, 1991). We combined a classical approach assessing the dredging impact on the physicochemical and biological characteristics of the system, and we incorporated a new approach based on the analysis of stable isotope values of carbon and nitrogen. This study specifically aims to assess (i) effects of dredging on sediment and water characteristics and on macrofaunal communities and (ii) indirect effects on the surrounding shallower habitat and on the whole food web structure.

2. Materials and methods

2.1. Study area

The Guadalquivir estuary is located in southwestern Spain. It extends from the mouth in Sanlúcar de Barrameda (Atlantic Ocean) to the Alcalá del Río dam, 110 km upstream. This estuary plays a critical role in the ecological and economic sustainability of very sensitive and protected areas of southwestern Spain (e.g., National Park of Doñana) (Tornero et al., 2014)· The Guadalquivir estuary is a well-mixed and tidally dominated system (3.5 m tidal range at the mouth in spring tides) (Díez-Minguito, 2012), which presents a longitudinal salinity gradient with temporal displacement by tides, discharges and seasonal variations (González-Ortegón et al., 2014). In order to guarantee a minimum navigation depth of 6.5 m, the channel is dredged every one or two years (Ruiz et al., 2015). In summer 2015, a maintenance dredging operation was carried out in several estuarine sections. The dredging work was performed by trailer suction dredge. Our study was focused on two dredging sections, one in the polyhaline range (18–30 PSU) and the other in the oligohaline range (< 5 PSU), locally known as Salinas and La Gola, respectively (Fig. 1). Approximately 74,000 and 22,000 m^3 of dredged material were extracted in each range, respectively.

2.2. Sampling design

Our sampling was designed according to a BACI approach (Underwood, 1994). In total, four sampling surveys were carried out: two pre-dredging (June and July 2015) and two post-dredging (October 2015 and August 2016) surveys. In both salinity ranges, two areas were established: one within the dredged section and the other (as a control) far away from the influence of these operations but always at the same salinity range intervals. Establishing more control areas in the same salinity ranges were not possible due to the areas not affected by the dredging being spatially limited (ca. 2 km). In each area, three stations were randomly located inside of the navigation channel and the other three in the shallower left margin in order to assess the direct and indirect effects of dredging in those habitats, respectively. Three samples were taken for macrofaunal analysis with a Van Veen grab (0.15 m^2 total sampling area per station and date). For posterior analysis, all stations were pooled together and were considered replicates of each area. Macrofaunal samples were sieved through a 0.5-mm mesh sieve, and infauna was preserved in ethanol (70%) and stained with rose bengal for subsequent identification and quantification at the lowest possible taxonomic level.

To relate the effects of dredging on sediment characteristics, one additional sample was taken for grain size distribution, particulate organic matter (POM) content and redox potential. Grain size distribution was measured as percentages of 100 g of dry sediment passed through a series of sieves (5 mm, 2 mm, 1 mm, 0.5 mm, 0.250 mm, 0.125 mm and 0.063 mm). Also, the median grain size (Q_{50}) and sorting coefficient (S_0) (Trask, 1950) were calculated. Granulometric typology was established according to the Wentworth geometric scale (Buchanan, 1984). The POM content was determined by calculating the weight difference between the dried sediment samples of three replicates (at 60 °C until dried weight stabilisation) and after combustion (500 °C for 4 h). Apparent redox potential was measured with a pH meter (WTW pH 1970i with SenTix ORP electrode).

For the heavy metals and trace element concentrations analyses, sediments were taken from the uppermost 2 cm. In the laboratory, sediment samples were air-dried, crushed and sieved though a 2-mm sieve and then ground to < 60 μm. These samples were digested with aqua regia (1:3 conc HNO_3: HCl) in a microwave digester. Quantification of elements in the extracts was achieved using a VARIAN ICP 720-ES (simultaneous ICP-OES with axially viewed plasma). The accuracy of the analytical methods was assessed via a reference soil sample from the Wageningen Evaluating Programs for Analytical Laboratories (WEPAL) for soils, International Soil-Analytical Exchange (ISE). The index of geoaccumulation (I_{geo}) has been used as a relative measure of metal pollution in sediments for Cr, Cu and Zn according to the regional background established by Ruiz (2001) for unpolluted sandy and silty-clayey sediments and is given by: $I_{geo} = \log_2$ (Cn/1.5 Bn), where Cn is the value of the element n and Bn is the background data of that element. Following Ruiz (2001), the index values were divided into five groups: unpolluted ($I_{geo} < 1$); very lowly polluted ($1 < I_{geo} < 2$); lowly polluted ($2 < I_{geo} < 3$); moderately polluted ($3 < I_{geo} < 4$); highly polluted ($4 < I_{geo} < 5$) and very highly polluted ($I_{geo} > 5$). Comparisons between metal concentrations and sediment quality values (SQVs) proposed by Long et al. (1995) and Delvalls and Chapman (1998) have also been performed. Heavy metals in water and sediment were only measured in the channel area in July and October 2015 and August 2016.

Water parameters were analysed from the bottom layer with a multiparametric probe Eureka Manta 2 with pH, dissolved oxygen, salinity and turbidity sensors. A 5-l water sample from 1 m above the bottom was collected with a Niskin bottle and then filtered through a GF/C Whatman glass fibre filter with an air vacuum pump; then, suspended organic matter (SUOM) and total suspended solids (TSS) were calculated. SUOM was determined with the same procedure as POM.

We investigated the possible impact of the dredging work on the

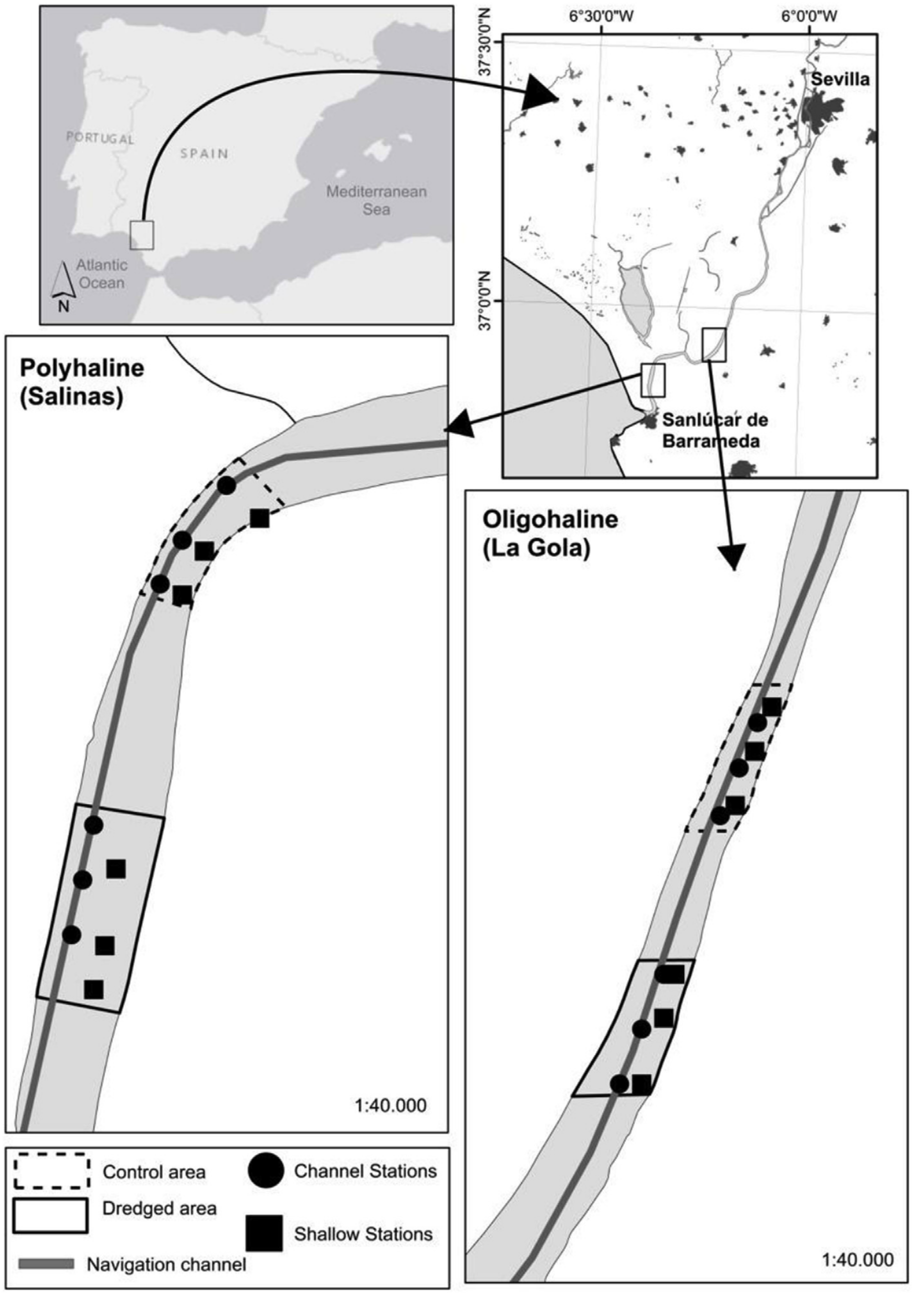

Fig. 1. Location of the study area and sampling stations in both ranges of the salinity gradient.

nekton-benthonic food web of the two salinity ranges. Although sampling was carried out at the same time as the macrofaunal surveys, we did not differentiate control and dredging areas because the daily tide movements did not allow the establishment of control areas. We selected this community because it was more diverse and accessible to sample than strictly benthic fauna. Samples of the planktonic community were collected before dredging (July 2015) and twice after it (October 2015 and August 2016). Organisms were collected with a 1000-μm mesh zooplankton net with a 1 m mouth diameter. Oblique tows were performed from surface to bottom during flood tide in the main channel. All organisms were sorted by species, transferred to the laboratory in refrigerated containers and kept alive for 24 h to evacuate their gut contents. Sediment was taken from the upper 2 cm of a Van Veen grab sample for sediment organic matter (SOM) analysis. We rinsed samples with distilled water. Muscle tissue samples of fish larvae and shrimp abdomen were extracted. Pools of several organisms were used when individuals had low biomass values. Samples were dried at 60 °C and ground to a powder. Sediment samples were acidified with 0.1M HCl to remove carbonates and oven-dried. Subsamples of powdered materials were weighed to the nearest 0.3 μg and placed into tin capsules for δ^{13}C and δ^{15}N determinations. 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). 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).

2.3. Data analysis

Direct and indirect dredging effects were independently examined in both salinity ranges. Channel and shallower left margin habitats were also separately compared with their respective controls in both salinity ranges. Water and sediment variables differences were tested on Euclidean distances using a permutational univariate analysis of the variance (PERMANOVA) (Anderson, 2001a). PERMANOVA was chosen for univariate analyses because resulting sums of squares and F-ratios are exactly the same as Fisher's univariate F-statistic in traditional ANOVA and does not assume a normal distribution of errors (Anderson, 2005, 2001b; Scyphers et al., 2011). The experimental design included two crossed fixed factors: "Impact vs. Control" with two levels (Impact and Control areas), "Time" with two levels (Before and After the dredging work) and a random factor, the sampling dates "Dates" nested within "Time" with four levels (Jun 15, Jul 15, Oct 15 and Aug 16).

According to a BACI design, if the disposal had a permanent impact, the putatively impacted areas will change over time from the samplings before the dredging work to the samplings after with a different pattern when compared to control areas. This difference can be detected as a significant "Time" x "Impact vs. Control" interaction (Guerra et al., 2009).

Univariate measures, such as species richness (number of taxa, S), Shannon's diversity index (H′, based on log *2*), total abundance (N, ind/m^2) and Pielou's evenness (J′) were calculated. Significant interactions were tested using a permutational univariate analysis of the variance (PERMANOVA) with the same experimental design as above. The p-values were provided using unrestricted (9999) permutation of the abundance data based on the Euclidean distance matrix. When the number of total possible permutations to obtain the p-values were low (< 100), we used the estimate obtained by Monte Carlo sampling (Anderson and Robinson, 2003). Significant interactions, if detected, were further explored in separate analyses, within the levels of the interacting factors; in other words, the significant interactions between "Impact vs. Control" and "Dates" were further analysed separately by impact area and the control area.

The effects on the multivariate structure of the communities were investigated using a PERMANOVA analysis based on the Bray-Curtis similarity index of square-root transformed abundance data with the same design as above. Macrobenthic communities were also investigated by a non-metric multidimensional scaling ordination (nMDS). SIMPER analysis was used to identify the species contributing most to any observed spatial or temporal pattern in the communities (Clarke, 1993).

Previously standardised sediment and water variables were examined using principal components analysis (PCA). Spearman correlations were done with the heavy metal concentrations and univariate community indices.

From the obtained results of stable isotopes analysis, we created graphical plots of the carbon and nitrogen signals (Fry, 2006). Only species found in all surveys in both ranges were used. Differences between stable isotopes signals of carbon and nitrogen were tested with non-parametric Kruskal-Wallis tests. All analyses were carried out in IBM SPSS for Windows and PRIMER v 6.0 software (Clarke and Gorley, 2006).

3. Results

3.1. Environmental variables

Water and sediment parameters are shown in Table S1. Granulometry of the sediments in the channel habitat of the oligohaline range, La Gola, oscillated between very fine sand and fine sand with a reduced bottom according to the redox potential. In the channel habitat of the polyhaline range, Salinas, sediments ranged from very fine sand to fine sand, while in the shallow habitat of both ranges, there was always very fine sand (Fig. 2). In the oligohaline range, the turbidity was higher than in the polyhaline range (Fig. 3).

The PERMANOVA results for both ranges showed significant temporal differences ($p < 0.01$) in both areas and both habitats (control and dredged and shores and channel, respectively) for most of the water parameters, while sediment parameters remained constant. There were not significant interactions between the factors "Time" and "Impact vs. Control" for any variables. Heavy metal concentrations are shown in Table S2 and mean concentration variation of selected metals in Fig. 4. Results showed generally higher concentrations after the operations in the dredging area of the polyhaline range. The PERMANOVA results of heavy metal concentrations for the two ranges did not show significant differences ($p > 0.01$) between the control and the dredging areas for all the heavy metals analysed. It also did not show interactions between the factors "Time" and "Impact vs. Control" in both salinity ranges. The I_{geo} index for Cr, Cu, Zn and Pb showed that the sediment was uncontaminated in the majority of the areas and sampling periods though some samples had higher levels of Pb. We found moderately contaminated values only in Salinas in August 2016. All heavy metal concentrations were below quality values of the sediment (SQV: Cd: 98 mg/kg, Cu: 270 mg/kg, Ni: 51.6 mg/kg, Pb: 84.6 mg/kg and Zn: 225 mg/kg (Delvalls and Chapman, 1998; Long et al., 1995; Tornero et al., 2014).

The PCA did not show any relationship between the dredging operation and the physicochemical variables (Fig. 5). In the oligohaline range, sample points were grouped, following the period, independently of the control or dredging area and channel or shallow habitat. The situation at the Salinas site was similar with major homogeneity between stations and/or periods.

3.2. Macrofaunal analysis

In total, 17 species were found in the oligohaline range and 38 in the polyhaline range. The most abundant group in all the samples was the annelids, especially the polychaetes *Alkmaria romijni* and *Streblospio shrubsolii*. Crustaceans also showed some importance in contributing to the diversity of the polyhaline range. There was practically no presence of molluscs in the oligohaline range, except some young specimens of the invasive species *Corbicula fluminea* (Table S3).

Univariate community indices are shown in Fig. 6. Species richness showed differences in the oligohaline range in the channel area a month after the dredging operations in relation to previous sampling dates. In October 2015, there were no species present in the dredging area, while the control area did have species. In the shallow habitat, the number of species did not change over the sampling periods. In August 2016, a year after the dredging, the richness in the dredging area was similar to that of the pre-dredging period. A PERMANOVA analysis showed no significant interaction ($p > 0.05$) between "Time" and "Impact vs. Control" in both channel and shallow habitats (Table 1). On the other hand, in the polyhaline range, we always found some species in every survey, but, in October 2015, a reduction in the number of species was found in both habitats (Fig. 6).

The Shannon's diversity index showed low values in the oligohaline range in all the sampling periods (Fig. 6). The greatest values were found in the polyhaline range. The PERMANOVA results did not show a significant interaction between factors (Table 1). In the polyhaline range, the index values were higher (Fig. 6). In the dredging area of the channel, there was temporal variation over the sampling dates, while in the control, it was more stable. In the shallow habitat of the control area, we found low values due to the high abundance of the polychaete *S. shrubsolii*. The PERMANOVA results showed a significant interaction between "Impact vs. Control" and "Dates" in the channel habitat ($p = 0.0013$) (Table 1). Separate analysis of the "Impact" level showed significant differences over the sampling dates ($p = 0.0039$), whilst the "Control" level showed no differences.

Abundance (ind/m^2) values found in the oligohaline range were low in all areas and dates (< 60 ind/m^2) (Fig. 6). The PERMANOVA analysis showed an interaction near significance between "Impact vs. Control" and "Dates" in the shallow habitat ($p = 0.051$) (Table 1).

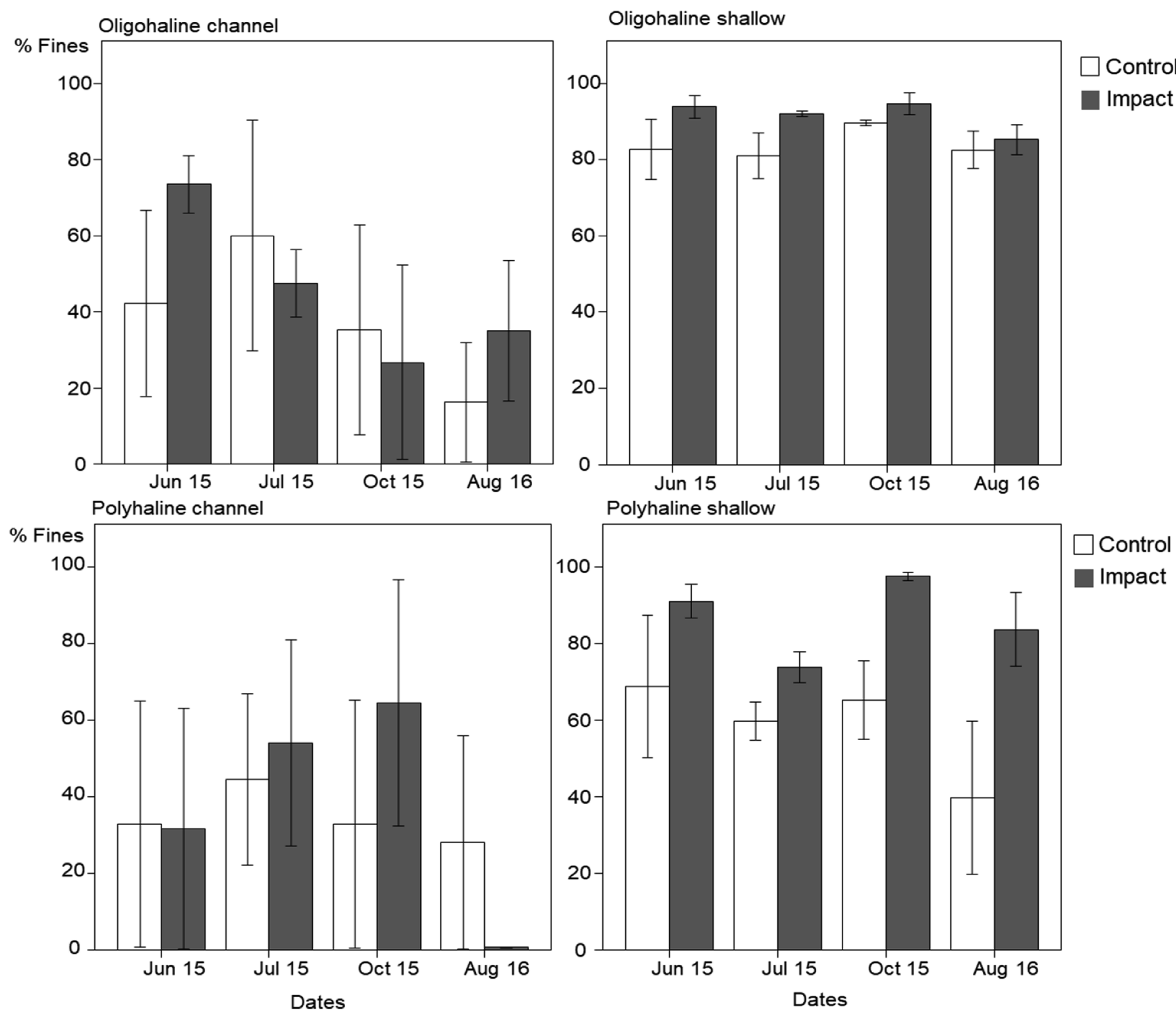

Fig. 2. Mean (± standard error) of fine percentage (< 0.063 mm) of both channel and shallow habitats of the two salinity ranges.

Separate analysis did not show significant differences over the sampling dates in both control or impact areas. The abundance values in the polyhaline range were higher than in the oligohaline range due to high numbers of the polychaete *S. shrubsolii* (Fig. 6). A PERMANOVA test showed a significant interaction between "Impact vs. Control" and "Dates" in both channel and shallow habitats ($p = 0.022$ and $p = 0.0296$, respectively) (Table 1). In the two separate analyses of control and impact stations of the two habitats, the impacted area showed significant differences over the sampling periods and controls did not.

Evenness showed greater values in the oligohaline range and followed similar trends as with Shannon's diversity and richness indices. A significant interaction was detected between the factors "Impact vs. Control" and "Dates" in the channel habitat of the polyhaline range ($p = 0.04$) (Table 1). Separate analysis also showed significant differences between sampling dates in the impacted area whilst controls did not.

The nMDS analysis in the oligohaline range showed the most of the stations with a similar macrofaunal community and no spatial or temporal patterns (Fig. 7). In the polyhaline range, there was more segregation between channel and shallow stations. The community of the shallow habitat, in both dredging and control areas, was similar in all sampling dates, while the channel habitat had more temporal variations. The community structure results did not show significant interactions in the oligohaline range. On the other hand, results showed a significant interaction between "Impact vs. Control" and "Dates" in both habitats of the polyhaline range ($p = 0.0197$ and $p = 0.0061$, respectively) (Table 2). Both separate analyses of the control and impact stations in the two habitats did not show significant differences in the control over the "Dates" while the "Impact" showed it.

SIMPER analysis of the polyhaline range (Table S4) showed no temporal trends in the channel habitat of the control area over the sampling dates. In the impacted area, results showed a general decrease in October 2015 of the abundances of species, such as the amphipod *Bathyporeia pilosa*, the isopod *Lekanesphaera levii* and *S. shrubsolii*, with respect to pre-dredging surveys, which were characterised by the dominance of these species. In August 2016, more species with no dominance patterns were observed. In the shallow habitat, differences found were mostly due to August 2016 where, in the impacted area, high abundances of the polychaetes *A. romijni* and *S. shrubsolii* and the isopod *Cyathura carinata* occurred.

The Spearman correlation between univariate community indices showed significant negative relationships between richness species and Co (−0.361), and abundance and As (−0.332), Co (−0.426) and Ni (−0.386). Concentrations of As, Co and Ni showed an increase in October in the dredging area of both salinity ranges, except Ni which

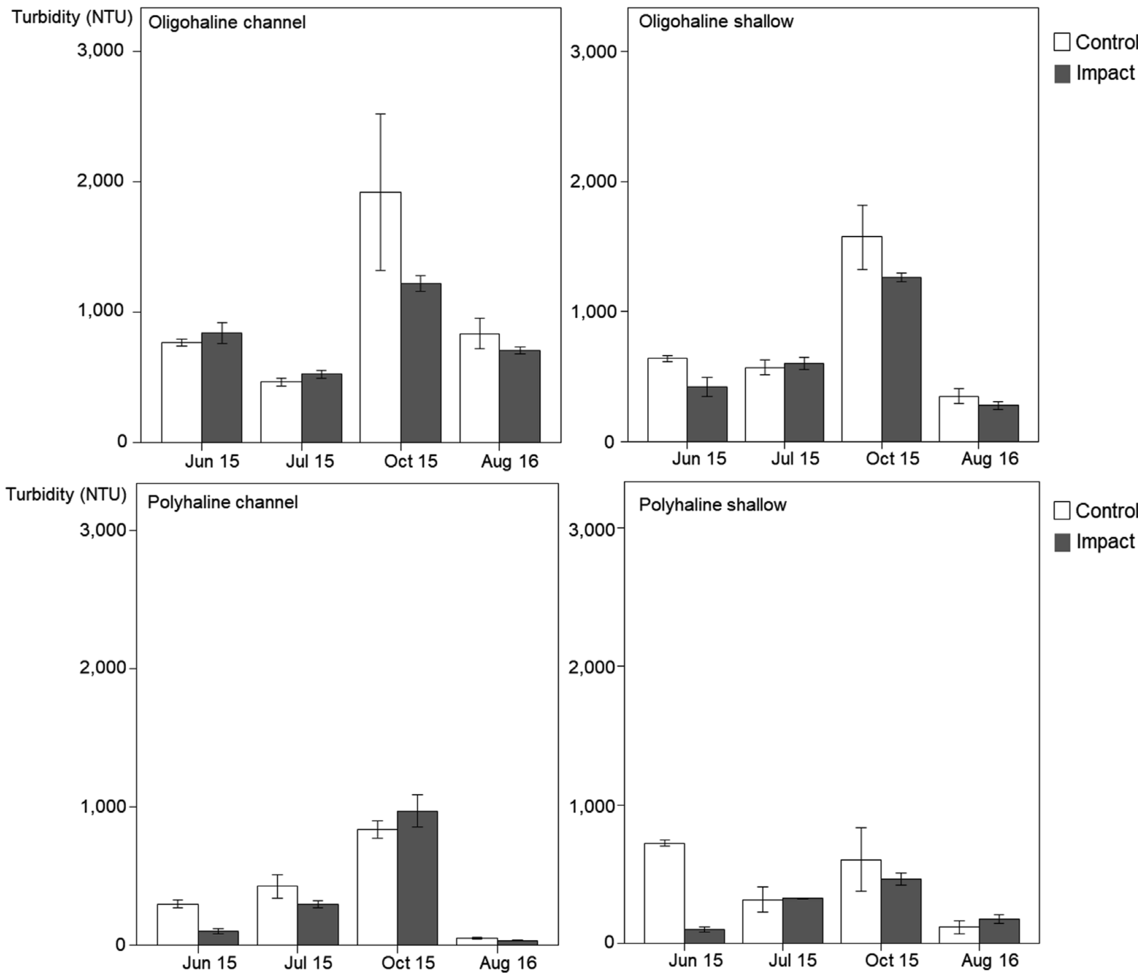

Fig. 3. Mean turbidity values (± standard error) of both channel and shallower habitats of the two salinity ranges.

showed a decrease in the oligohaline range. In the control areas, concentrations of these metals showed a decrease or remained at the same levels than the pre-operational measures. In August 2016, an increase in the concentrations was detected, except for Co and As in the dredging area of the oligohaline range (Fig. 4).

Stable isotope plots suggested a more widespread food web structure in the oligohaline range than in the polyhaline range (Fig. 8). This could suggest that organisms in La Gola occupied different trophic niches. The Salinas samples showed similar carbon and nitrogen isotope signals, suggesting similar trophic interactions. Plots also suggested a different organic matter origin in the polyhaline range than in the oligohaline due to the different carbon enrichment values observed in the food web of the polyhaline range.

The two salinity ranges did not show the same pattern over time. In the oligohaline range, some organisms suffered changes in the surveys after the dredging operations. There was a significant ($H = 7.64$; $p = 0.02$) decrease in nitrogen values of the mysid *Neomysis integer* in October, one month after the dredging. One year later, the mysid nitrogen signal was at the same level as before. Although plots also showed an increase after dredging on the vegetal matter nitrogen values, there were no significant differences. This value remained at the same level one year after dredging. Carbon values only showed enrichment in the anchovy (*Engraulis encrasicolus)* in October. One year later, the carbon signal was at pre-dredging levels. The other species did not show any changes. Conversely, in the polyhaline range, isotope signals of the organisms were similar across all sampling months. Only the mysid *Mesopodopsis slaberii* suffered a slight depletion in their carbon signal one year after the dredging work.

4. Discussion

Increasing anthropogenic pressures on the benthic environment in estuaries has not always been an issue of concern (Rehitha et al., 2017). Our study assessed the grade of impact of dredging operations carried out in summer 2015 on benthic communities in two salinity ranges in the Guadalquivir estuary. The most noticeable feature observed in the dredging ranges, both in the channel and shallow habitats, is the absence of an evident effect in sediment and water parameters and the low impact in the biological communities independent of the salinity range. Also, the food web structures in both salinity ranges were not clearly affected by the dredging. Changes in the isotopic composition of the anchovies and the mysids could be explained by the natural variation of the system. The impossibility of the establishment of more replicated controls per dredging area makes it necessary to interpret these results with caution. When there are not replicated control areas under study, we do not have a measure of the natural random variability among any two different areas. In case the analysis leads to the identification of differences between the evolution of the control and the potentially impacted areas, these differences cannot be unconfoundedly assigned to an impact. When the differences are found to be not significant, this weakness is less critical given that the inclusion of more replicates of control areas will not change that the observed variation in dredging areas was overlapping with the natural variation.

Although we have no data about the very early effects immediately after dredging, the deepening of channels may significantly increase suspended matter concentrations in the long-term by the stirring up of bottom substratum or erosion from locations that were not sensitive to

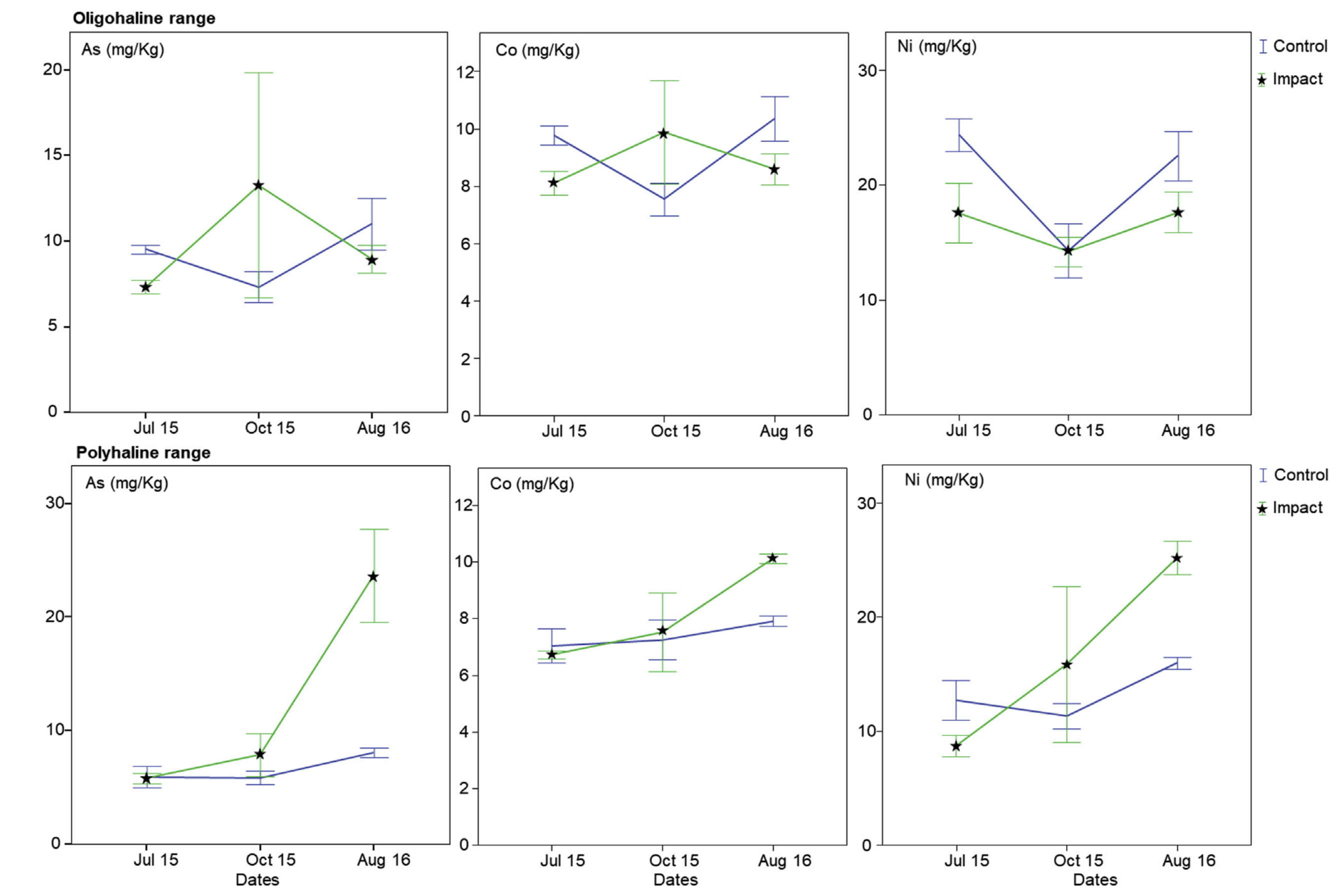

Fig. 4. Mean concentration variation (± standard error) over the sampling dates of As, Co and Ni in both salinity ranges.

erosion before (de Jonge et al., 2014; Rehitha et al., 2017). In both ranges, water parameters (pH, oxygen and salinity) showed the expected values for a temperate estuary during the sampling period. Turbidity was notably higher in the oligohaline range than in the polyhaline range, because this range is in the maximum turbidity zone of the estuary (Vilas et al., 2008). The same pattern was also observed in the sediment characteristics. In both ranges, the granulometry remained stable in both the dredging and control areas. Our data suggest

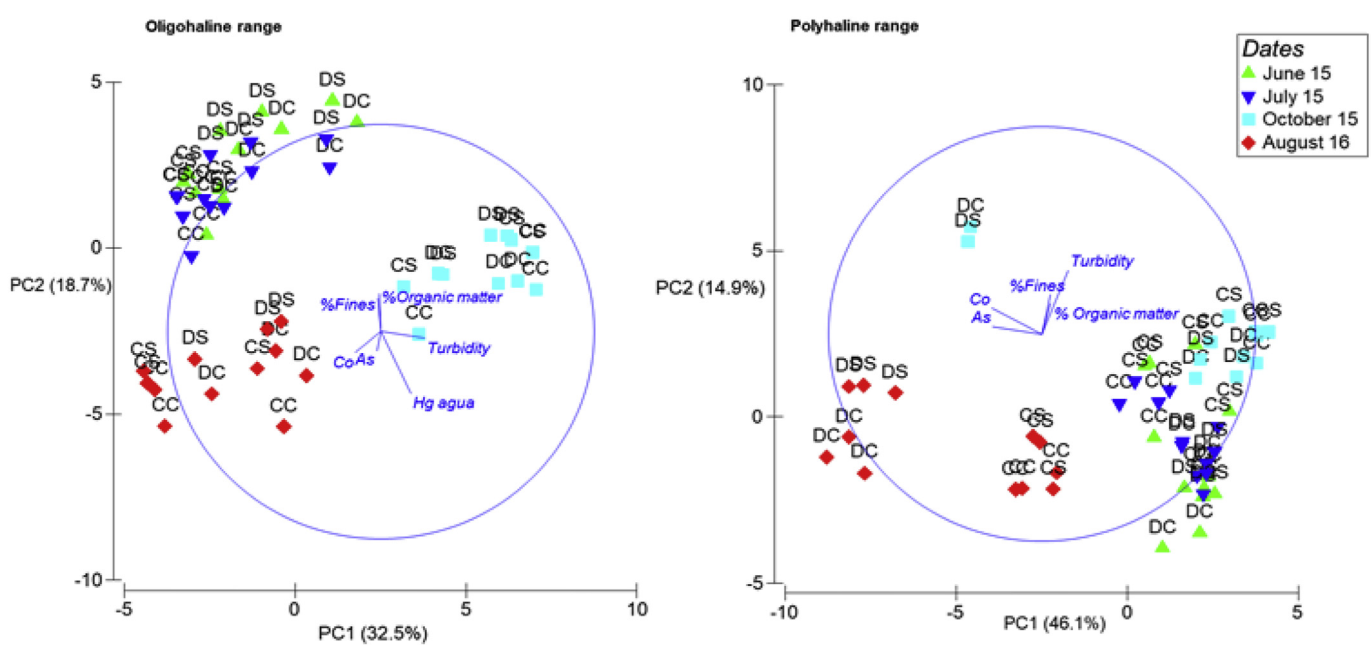

Fig. 5. Principal components analysis (PCA) results for sediment and water parameters at all the stations in dredging and control areas and channel and shallow habitats over the sampling period. The percentage of variability explained by the two principal axes and vectors of a selection of parameters are given. (CC: Control channel, CS: Control shallow, DC: Dredging channel, DS: Dredging shallow).

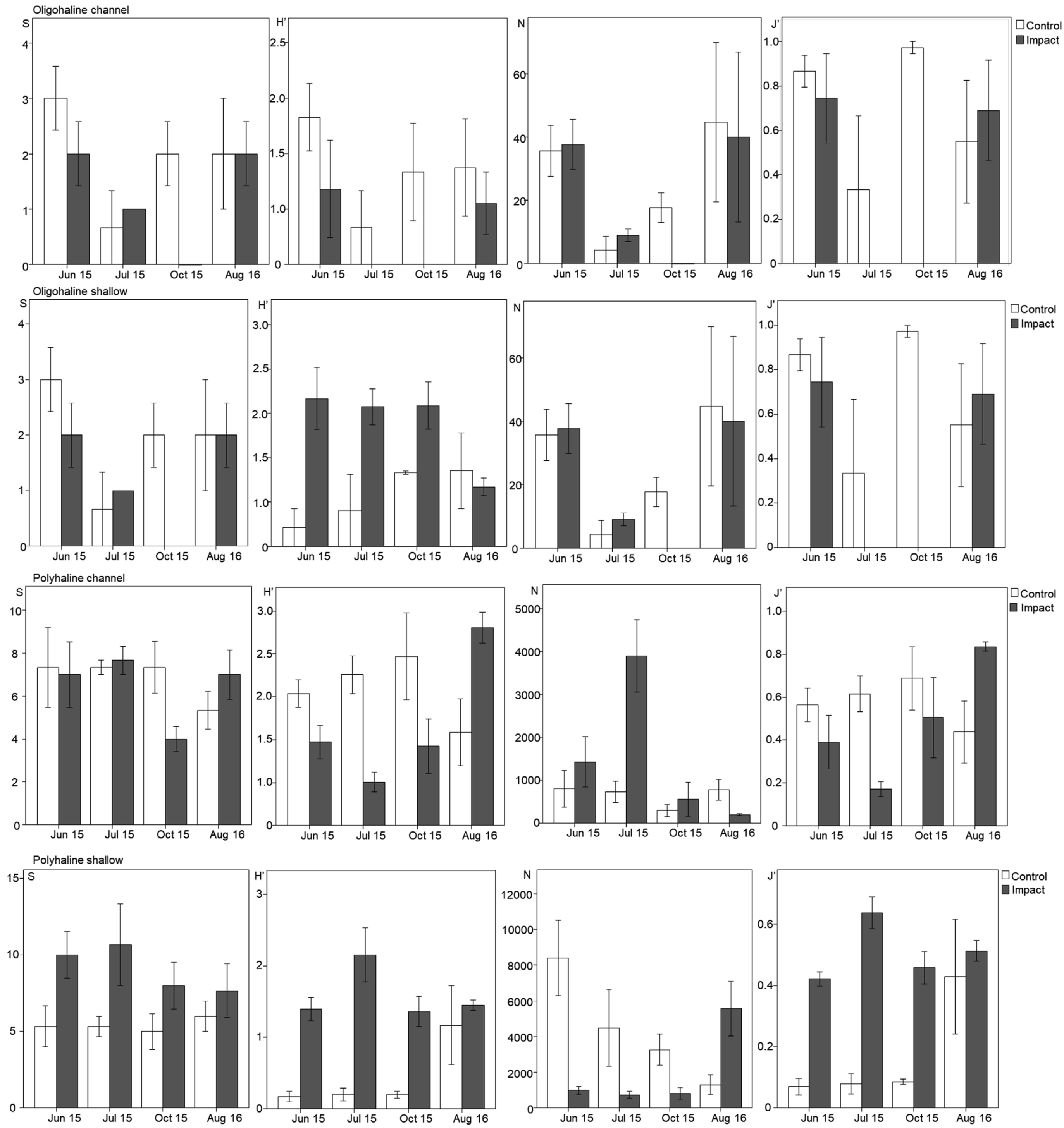

Fig. 6. Mean (± standard error) values of the univariate community indices (S, N, H′ and J′) in the two salinity ranges over the sampling periods.

that sediment characteristics changed similarly in the control and dredging areas following natural variations. Therefore, dredging operations seem not to affect the water and granulometry, possibly due to the extraction method used, which minimized sediment overflow. However, the high flows originated by tides and the high chronic turbidity in the Guadalquivir estuary (Losada et al., 2017) could overshadow these effects.

Dredging operations may also release contaminants that were trapped in bottom sediments to the water column (Wasserman et al., 2016). However, all heavy metal concentrations were below SQV values. Furthermore, PCA analysis indicated that sampled points were grouped according to the period of survey instead of being grouped according to area (dredging vs. control). Our results agree with those reported by Guerra et al. (2009), who found that sediment deposited after dredging had the same contamination levels as before dredging operations. The higher levels of some heavy metal concentrations found after the operations in the dredging area of the polyhaline range were mostly due to the increase in August 2016, one year after dredging. Deepening of channels could lead to a greater dominance of fine fractions of sediment in dredging areas for a few hundred meters due to the dredge plume and lower current velocities, which favoured the deposition of fine sediment with higher levels of heavy metals (Klapan

Table 1
Univariate PERMANOVA results in both salinity ranges based on the Euclidean distance matrix of the richness data (S), Shannon's diversity (H′), total abundance (ind/m^2) (N) and Pielou's evenness (J′). *p estimation obtained by Monte Carlo sampling.

Oligohaline Channel	df	MS	Pseudo-F	p	Unique perms
S					
Time	1	0.16667	2.94E-02	0.874*	3
Impact vs Control	1	2.6667	1.2308	0.3623	204
Dates (Time)	2	5.6667	5.44	0.0212	7150
Time x Impact vs Control	1	0.66667	0.30769	0.6148	242
Dates(Time) x Impact vs Control	2	2.1667	2.08	0.1518	9369
Res	16	1.0417			
Total	23				
N					
Time	1	9.20E+01	3.03E-02	0.8796*	3
Impact vs Control	1	92.042	1.3937	0.3473	141
Dates (Time)	2	3033.4	5.3209	0.0176	9957
Time x Impact vs Control	1	315.38	4.7754	0.151	801
Dates(Time) x Impact vs Control	2	66.042	0.11585	0.8947	9951
Res	16	570.08			
Total	23				
H'					
Time	1	2.78E-03	2.34E-03	0.9664*	3
Impact vs Control	1	1.7103	12.66	0.082	800
Dates (Time)	2	1.1844	3.6732	0.0528	9951
Time x Impact vs Control	1	1.14E-02	8.45E-02	0.7155	799
Dates(Time) x Impact vs Control	2	0.1351	0.41899	0.6591	9970
Res	16	0.32244			
Total	23				
J'					
Time	1	5.26E-04	1.17E-03	0.974*	3
Impact vs Control	1	0.78073	6.6382	0.1209	799
Dates (Time)	2	0.44765	2.6904	0.0961	9948
Time x Impact vs Control	1	4.52E-04	3.85E-03	0.8174	793
Dates(Time) x Impact vs Control	2	0.11761	0.70686	0.5099	9950
Res	16	0.16639			
Total	23				

Polyhaline Channel	df	MS	Pseudo-F	p	Unique perms
S					
Time	1	1.20E+01	22.231	0.0437*	3
Impact vs Control	1	1.04E+00	0.10917	0.7357	776
Dates (Time)	2	5.42E-01	0.1413	0.8711	9950
Time x Impact vs Control	1	1.0417	0.10917	0.737	776
Dates(Time) x Impact vs Control	2	9.5417	2.4891	0.1126	9953
Res	16	3.8333			
Total	23				
N					
Time	1	9.49E+06	4.3854	0.1682*	3
Impact vs Control	1	4.56E+06	1.6906	0.3259	794
Dates (Time)	2	2.16E+06	3.7833	0.0411	9947
Time x Impact vs Control	1	6.34E+06	2.3478	0.2483	798
Dates(Time) x Impact vs Control	2	2.70E+06	4.72	0.022	9950
Res	16	5.72E+05			
Total	23				
H′					
Time	1	0.85436	7.4997	0.1113*	3
Impact vs Control	1	1.0158	0.48277	0.5492	794
Dates (Time)	2	0.11392	0.45563	0.6363	9951
Time x Impact vs Control	1	1.4878	0.70709	0.4999	800
Dates(Time) x Impact vs Control	2	2.1042	8.4158	0.0013	9956
Res	16	0.25003			
Total	23				
J′					
Time	1	0.19654	15.167	0.061*	3
Impact vs Control	1	6.09E-02	0.3959	0.588	800
Dates (Time)	2	1.30E-02	0.32005	0.7317	9959
Time x Impact vs Control	1	0.25978	1.6892	0.3216	794
Dates(Time) x Impact vs Control	2	0.15378	3.7981	0.0404	9957
Res	16	4.05E-02			
Total	23				

Oligohaline Shallow	df	MS	Pseudo-F	p	Unique perms
S					
Time	1	1.0417	5	0.1548*	3
Impact vs Control	1	35.042	168.2	0.0738	748
Dates (Time)	2	0.20833	7.81E-02	0.9257	9950
Time x Impact vs Control	1	0.375	1.8	0.3104	108
Dates(Time) x Impact vs Control	2	0.20833	7.81E-02	0.9264	9953
Res	16	2.6667			
Total	23				
N					
Time	1	5.01E+05	0.97118	0.4217*	3
Impact vs Control	1	5.85E+05	1.542	0.3375	798
Dates (Time)	2	5.16E+05	4.2402	0.018	9954
Time x Impact vs Control	1	3.42E+05	0.90149	0.4698	812
Dates(Time) x Impact vs Control	2	3.79E+05	3.1158	0.0506	9954
Res	16	1.22E+05			
Total	23				
H'					
Time	1	2.46E-03	8.08E-03	0.9381*	3
Impact vs Control	1	3.8231	10.61	0.0913	800
Dates (Time)	2	0.30438	1.2763	0.3041	9956
Time x Impact vs Control	1	1.5611	4.3323	0.1662	798
Dates(Time) x Impact vs Control	2	0.36033	1.5109	0.2498	9965
Res	16	0.23848			
Total	23				
J'					

Polyhaline Shallow	df	MS	Pseudo-F	p	Unique perms
S					
Time	1	8.1667	24.5	0.0387*	3
Impact vs Control	1	80.667	96.8	0.076	739
Dates (Time)	2	0.33333	4.57E-02	0.9547	9675
Time x Impact vs Control	1	10.667	12.8	0.0785	530
Dates(Time) x Impact vs Control	2	0.83333	0.11429	0.8926	9928
Res	16	7.2917			
Total	23				
N					
Time	1	5.02E+06	0.53537	0.5388*	3
Impact vs Control	1	3.27E+07	1.4896	0.3404	798
Dates (Time)	2	9.38E+06	1.9784	0.1719	9936
Time x Impact vs Control	1	6.31E+07	2.8717	0.2385	800
Dates(Time) x Impact vs Control	2	2.20E+07	4.6293	0.0296	9941
Res	16	4.74E+06			
Total	23				
H′					
Time	1	2.40E-02	3.69E-02	0.867*	3
Impact vs Control	1	8.00E+00	16.279	0.082	794
Dates (Time)	2	0.65004	3.183	0.0655	9940
Time x Impact vs Control	1	1.128	2.2956	0.2415	796
Dates(Time) x Impact vs Control	2	0.49137	2.4061	0.1272	9947
Res	16	2.04E-01			
Total	23				
J′					

(continued on next page)

Table 1 (*continued*)

Oligohaline Channel						Polyhaline Channel					
	df	MS	Pseudo-F	p	Unique perms		df	MS	Pseudo-F	p	Unique perms
Time	1	4.02E-02	0.22745	0.6885*	3	Time	1	2.89E-02	0.36995	0.6083*	3
Impact vs Control	1	0.4861	9.8016	0.0983	798	Impact vs Control	1	0.70139	14.877	0.0787	801
Dates (Time)	2	0.17683	2.5882	0.102	9948	Dates (Time)	2	7.82E-02	4.7287	0.0142	9952
Time x Impact vs Control	1	0.7592	15.308	0.0796	800	Time x Impact vs Control	1	7.70E-02	1.6334	0.3284	795
Dates(Time) x Impact vs Control	2	4.96E-02	0.72589	0.4991	9958	Dates(Time) x Impact vs Control	2	4.71E-02	2.8504	0.0752	9949
Res	16	6.83E-02				Res	16	1.65E-02			
Total	23					Total	23				

et al., 1975; Newell et al., 1998; Ponti et al., 2009; Crowe et al., 2016). However, our results did not show an increase of the percentage of fine sediments in this area. Moreover, the increase of concentration of pollutant one year after could indicate a possible input of contaminants from different sources. This fact has been pointed out by Tornero et al. (2014), who suggest that other sources, such as mining activities upstream, could explain As and Pb concentrations in clams in the Guadalquivir estuary. Areas affected by dredging work could experience drastic reductions in richness species, abundance and biomass or become completely defaunated (Klapan et al., 1975; Newell et al., 1998; Fraser et al., 2006; Gutperlet et al., 2017). In other studies, with similar volumes dredged, impacts were detected on community structure and other univariate community indices (Ceia et al., 2013; Ponti et al., 2009; Van Dolah et al., 1984). In October 2015, one month after the dredging work, there were no species in the dredging channel area of the oligohaline range. Direct removal of the species seems to be the explanation. Salinas had a more structured and rich community; as a consequence, the reduction suffered in October 2015 in the channel of the dredging area was more pronounced, but did not reach the azoic level, probably due to lateral and vertical migration of surrounding bottom communities (Hall, 1994). On the other hand, it seems that there was no effect in the shallow habitats of the dredging areas. This agrees with the results of Ponti et al. (2009), who found direct effects on dredging channels and no effects on nearby areas. Richness and abundance have been proven to be more effective to indicate the first impacts of a perturbation than the Shannon's diversity index (Katsiaras et al., 2015), and our results showed that richness was the most useful index to describe shifts in the macrofaunal community in both salinity ranges.

The absence of any significant interaction between the factors “Time” and “Impact vs. Control” indicated there was not a permanent effect in the univariate indices or the community structure from the dredging operations (Underwood, 1994). Despite the absence of permanent effects, significant interactions were found between the factor “Dates” and “Impacts vs. Control” in some univariate community indices as well as in the community structure in the polyhaline range. This indicated different trends in the control and dredging areas over the random sampling dates. Separate analysis of the impact and controls always showed a significant variation in the impacted areas whilst the controls did not. SIMPER analysis showed a decrease of abundance of predominant species in the channel habitat of the impacted area one month after the dredging, which could indicate a possible impact. In spite of this, most of the differences were due to changes in August 2016 in the impacted areas of both channel and shallow habitats where a higher number species were found.

Dredging effects on macrofaunal communities and posterior recolonisation rates are site specific (Thrush and Dayton, 2002; Bemvenuti et al., 2005; Fraser et al., 2006; Gutperlet et al., 2015). Estuaries characterised by a muddy bottom and high dynamic areas often have more rapid recoveries than those with stable sand and gravel areas (Gutperlet et al., 2015; Rehitha et al., 2017). For example, rates reviewed by Newell et al. (1998) suggest a recovery time of 6–8 months for muddy estuaries, while communities with sand and gravel may take 2–3 years to re-establish. Our results were in concordance with these studies. One year after the dredging work, abundances in the oligohaline range as well as richness and Shannon's diversity in the polyhaline range reached pre-dredging values.

Dredged habitats are often first colonised by opportunistic species (Sánchez-Moyano et al., 2004). According to Newell et al. (1998), a large population of small sedentary deposit feeders, like polychaetes, would be the first colonisers after cessation of the disturbance and then would progress towards the same levels as before the disturbance. In

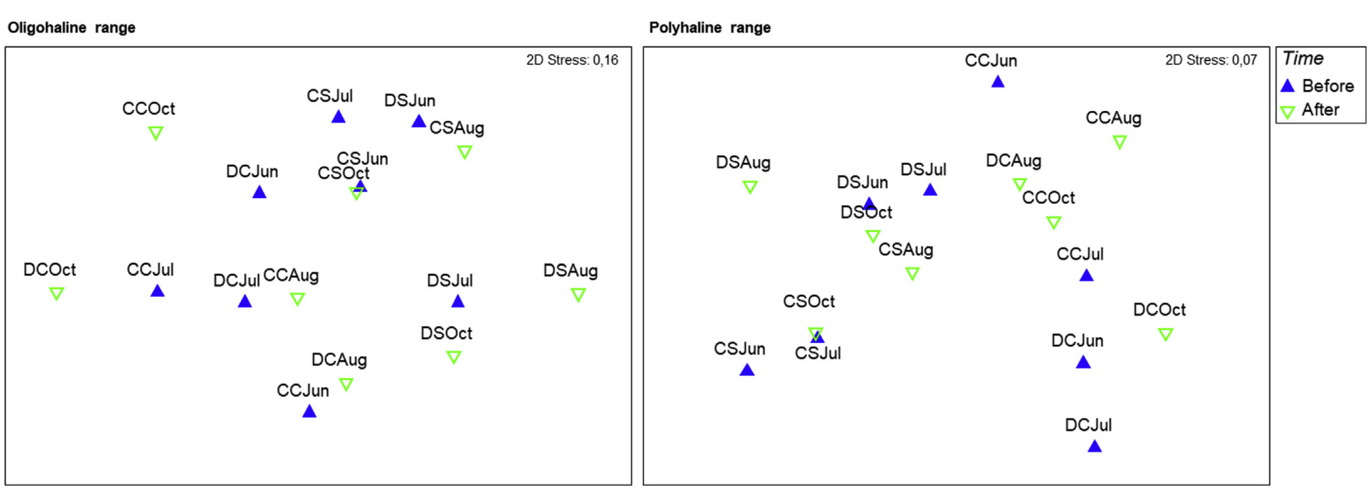

Fig. 7. nMDS of the distance among centroids resemblance for the combined factor between the “Dates” and the two dredging and control areas before and after the dredging of both habitats based on abundance of the different species in both salinity ranges. (CC: Control channel, CS: Control shallow, DC: Dredging channel, DS: Dredging shallow).

Table 2
PERMANOVA results of the Bray-Curtis similarity matrix based on square-root transformed data in both channel and shallow habitats of both salinity ranges. *p estimation obtained by Monte Carlo sampling.

Oligohaline Channel	df	MS	Pseudo-F	p	Unique perms	Polyhaline Channel	df	MS	Pseudo-F	p	Unique perms
Time	1	1095.5	0.14673	0.9809*	3	Time	1	3743.7	1.2603	0.3536*	3
Impact vs Control	1	2307.7	0.79748	0.5166	801	Impact vs Control	1	7689.8	2.3883	0.1507	799
Dates(Time)	2	7466	3.5928	0.0003	9933	Dates(Time)	2	2970.6	2.2071	0.0367	9929
TimexImpact vs Control	1	3075.3	1.0627	0.4183	800	TimexImpact vs Control	1	3148	0.97769	0.4498	800
Dates(Time)xImpact vs Control	2	2893.8	1.3926	0.1778	9922	Dates(Time)xImpact vs Control	2	3219.8	2.3922	0.0197	9926
Res	16	2078				Res	16	1345.9			
Total	23					Total	23				
Oligohaline Shallow	df	MS	Pseudo-F	p	Unique perms	**Polyhaline Shallow**	df	MS	Pseudo-F	p	Unique perms
Time	1	4082.3	1.0272	0.444*	3	Time	1	1684.4	1.4344	0.2797*	3
Impact vs Control	1	8354.1	4.8568	0.1106	801	Impact vs Control	1	7760.4	3.9154	0.0814	801
Dates(Time)	2	3974.1	1.8304	0.1073	9945	Dates(Time)	2	1174.2	1.435	0.1399	9919
TimexImpact vs Control	1	2368.2	1.3768	0.3253	798	TimexImpact vs Control	1	3131.5	1.5799	0.2822	800
Dates(Time)xImpact vs Control	2	1720.1	0.79227	0.5742	9941	Dates(Time)xImpact vs Control	2	1982	2.4222	0.0061	9933
Res	16	2171.1				Res	16	818.27			
Total	23					Total	23				

other studies, community recovery demonstrated that univariate community indices, such as abundances and richness, after a dredging impact could reach pre-operational levels after a certain period of time, but the ecological function could be not the same (Ceia et al., 2013). These shifts seem to be related to changes in sediment characteristics. Conversely, Sánchez-Moyano et al. (2004) observed a recovery in one month, reaching the same community structure and not only opportunistic species. In estuaries, Rehitha et al. (2017) detected changes in the granulometry toward more fine sediment in dredged areas as well as a reduction in species richness and diversity followed by a rapid colonisation of opportunistic species compared to non-dredging areas. They also reported that complex communities in the dredging areas could not be reached due to continuous dredging activities. In the Guadalquivir estuary, the benthic community, principally in the oligohaline range, was characterised by high abundances of the polychaetes *A. romijnii* and *S. shrubsolii*, even in the control areas. Therefore, these r-strategist species rapidly colonised the dredged areas, showing a rapid and complete recovery. This is consistent with Bemvenuti et al. (2005), who assessed that in areas that annually experience high changes in salinity (e.g., estuaries), fauna were reduced and community structure was altered by dredging activities, but there was also a rapid recovery due to the high resilience of the system. This is consistent with other studies (Fraser et al., 2006 and references therein).

Despite the low number of species present in all surveys is a major constraint for the isotopic analysis in this study, some conclusions can be made. Differences in the food web structures of the two salinity ranges could indicate the use of more carbon-depleted sources of organic matter with a possible terrestrial origin in the oligohaline range. Conversely, in Salinas, the marine inputs coming with the high tides can introduce more enriched carbon sources of organic matter into the food web (Selleslagh et al., 2015). The variation in isotope values in the oligohaline range suggests more complex feeding pathways than in the polyhaline range. In Salinas, the primary consumers could be feeding on the same organic matter sources, because similar isotope signals of secondary consumers could indicate similar diet composition. Conversely, in the oligohaline range, different nitrogen signals of the

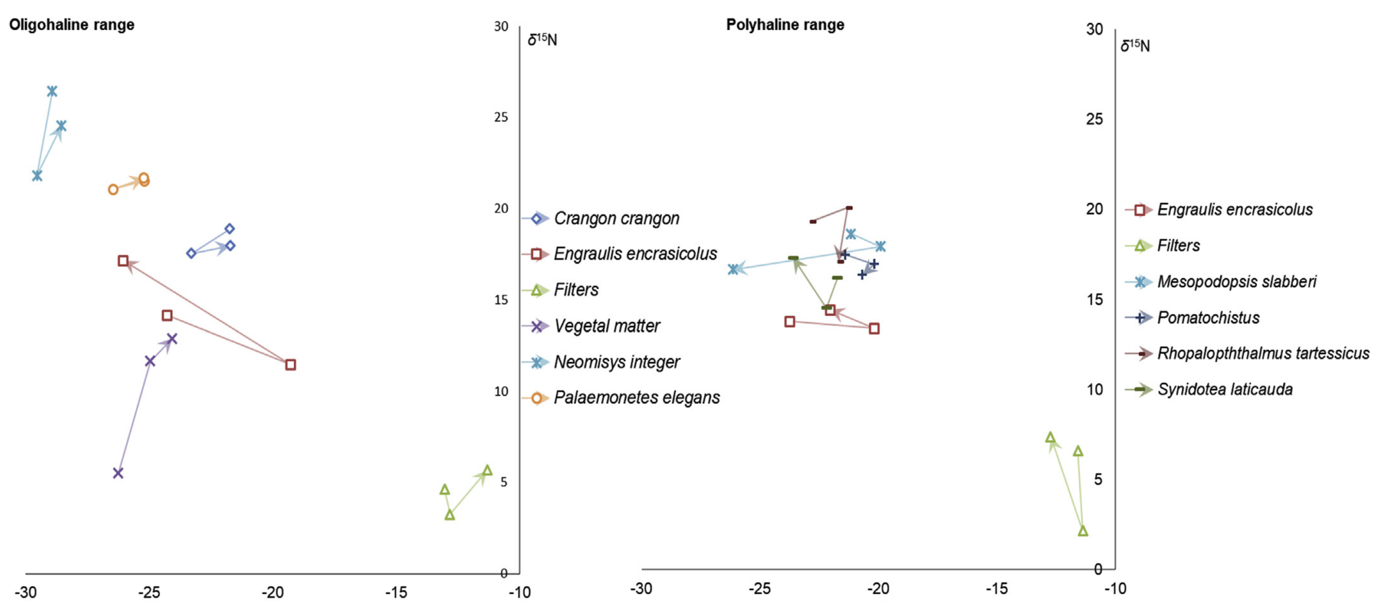

Fig. 8. Means for δ13C (x-axis) and δ15N (y-axis) of the organisms collected in oligohaline (left) and polyhaline (right) sampling areas. Arrows represent isotopic variation over the three sampling periods (July 2015, October 2015 and August 2016).

secondary consumers could suggest that they feed on a different suite of prey.

Changes in the isotopic signals of some organisms in the oligohaline range over the sampling periods could suggest an effect of dredging. A decrease in nitrogen levels of *N. integer* could suggest a change in trophic niche. Differences in trophic position of this species in October could be caused by the elimination of an intermediate consumer or a change in the degree of trophic omnivory (Post and Takimoto, 2007). The increase in the nitrogen signal of the vegetal matter could suggest the presence of more enriched nutrients with an anthropogenic origin. Dredging can resuspend fine sediments, nutrients and pollutants that had been trapped over the years (Ponti et al., 2009; Wasserman et al., 2016), making them available to the food web. In that sense, the variation in the carbon signal of *E. encrasicolus* would indicate that they use sources of organic matter with different origin over time (Dias et al., 2017). The multispecies approach of selecting organisms with different turnover rates would assess the dredging impact over a temporal scale (Modéran et al., 2012; Selleslagh et al., 2015). Changes in the diets of organisms are not immediately reflected by stable isotopes signals; higher trophic level organisms can show an integrated time response to nutrients better than primary producers (Van De Merwe et al., 2016). One year after the dredging work, mysid and vegetal matter isotope signals were still at the same levels as one month after; however, anchovies showed the same levels as pre-dredging measures. Despite changes in the oligohaline range in some isotope values one month post-dredging, variability in the patterns of isotope signals for carbon and nitrogen do not allow us to confirm an impact of dredging on food web structure. Thus, changes seem to be more related to natural variations rather than a dredging impact.

5. Conclusion

Maintenance dredging work is common activity that is necessary to maintain navigability and support trade. However, these human impacts may lead to several direct or indirect threats for estuarine ecosystems. The site-specific component of these impacts necessitates the study of these effects in every system (Fraser et al., 2006). In a highly variable scenario with anthropogenic and natural frequent perturbations, such as in the Guadalquivir estuary, macrofaunal communities often are characterised by low diversity and large populations of species well adapted to rapid recolonisation (Newell et al., 1998). The poor benthic community status in both salinity ranges in the Guadalquivir estuary explains the absence of a detectable effect on the community structure, diversity and richness and the quick recovery of the punctual affections by recolonisation of organisms of nearby areas. Moreover, the dredging work did not evidently affect the food web structure either. This poor status has been reported by other authors (Baldó and Drake, 2001; Sánchez-Moyano et al., 2017) and even in drastic impacts, such as acid mining spills released to the estuary, an impact on the benthic community was not detected (Baldó and Drake, 2001). In the management of estuaries, Ceia et al. (2013) reported that higher dredging frequency and extension means a longer recovery period for macrofaunal assemblages due to sediment structure destabilisation. However, the actual pressures on the Guadalquivir estuary, beyond the maintenance dredging work (e.g., unnatural freshwater inputs in summer for rice agriculture, permanent turbidity and high regulation of the natural flow by upstream dams) does not permit the establishment of more complex communities. Therefore, in poor diversity systems, like the Guadalquivir estuary, and from economic and management efficiency perspectives, research efforts should focus on the most diverse areas, such as the polyhaline range. This study marks the need for a proper management plan that involves all the administrations for the improvement of the biological benthic communities of the Guadalquivir estuary.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.marenvres.2018.07.012.

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