Science of the Total Environment 806 (2022) 151304

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

# Effects of maintenance dredging on the macrofauna of the water column in a turbid estuary

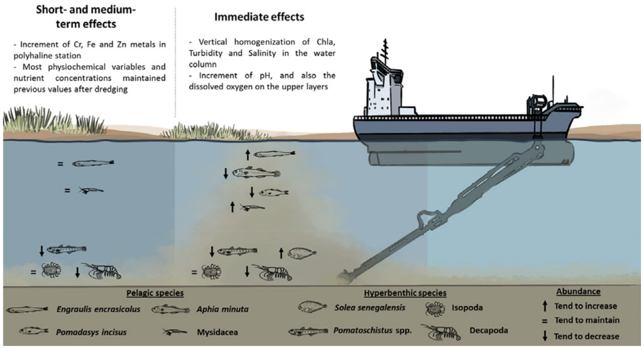
J.M. Miró [a,*], C. Megina [b], I. Donázar-Aramendía [a], J.C. García-Gómez [a]

[a] *Laboratorio de Biología Marina, Seville Aquarium R + D + I Biological Research Area, Department of Zoology, Faculty of Biology, University of Sevilla, Sevilla, Spain*
[b] *Biodiversidad y Ecología Acuática, Seville Aquarium R + D + I Biological Research Area, Department of Zoology, Faculty of Biology, University of Sevilla, Sevilla, Spain*

## HIGHLIGHTS

- Organisms can suffer entrainment by the trailer suction, especially epibenthic ones.
- Dredger action homogenized most physiochemical variables in the water column.
- Metal concentration of Cr, Fe y Zn increased in the water column after dredging.
- Effects of this dredging were similar or less than other natural ones (e.g. freshets).

## GRAPHICAL ABSTRACT



## ARTICLE INFO

*Article history:*
Received 5 July 2021
Received in revised form 7 October 2021
Accepted 24 October 2021
Available online 29 October 2021

Editor: Daniel Wunderlin

*Keywords:*
Dredging
Macrofauna
Plankton
Hyperbenthos
Water column
Metals

## ABSTRACT

Many human activities in or near aquatic habitats generate alterations in their environmental conditions, which could affect the organisms that inhabit them. Maintenance dredging of navigation channels in order to allow large ships access to inland ports is one such source of disturbance. In this study, by taking multiple approaches (immediate-, short- and medium term), we analysed the effects of a maintenance dredging operation on physiochemical variables and the early life stages of fish and other macrofauna groups present in two zones of the Guadalquivir estuary with different salinity ranges (poly- and mesohaline). Most physiochemical variables were homogenized in the water column immediately after the water mass passed by the dredger, including sediment resuspension. However, this process seemed to be transient as no significant increments in the depth-averaged levels of turbidity were observed in the short- and medium-terms. Instead, metal concentrations of Cr, Fe and Zn increased in the polyhaline station. Even so, these perturbations did not appear to be severe enough to influence the macrofauna. Still, organisms can suffer direct mechanical impacts of the trailer suction. Hyperbenthic species, like *Pomatoshcistus* spp. or decapods, tended to decrease slightly, while pelagic species such as *Engraulis encrasicolus* or mysids did not, indicating that benthic organisms are usually more susceptible to high entrainment. Nonetheless, the possible effects of this disturbance were of the same order or less than those of natural ones; therefore, organisms of the macrofauna could be well adapted to cope with them.

## 1. Introduction

Many human activities in or near aquatic habitats generate alterations in their environmental conditions, which could affect the organism that inhabit them (Halpern et al., 2008; Lotze et al., 2006). Capital

---

* Corresponding author.
*E-mail address:* jmmiro@us.es (J.M. Miró).

https://doi.org/10.1016/j.scitotenv.2021.151304

dredging of navigation channels in order to reclaim land and to allow ever larger ship access to inland waterways is one such source of disturbance. Adverse effects of dredging operations in coastal systems have generally included habitat degradation, increased turbidity and suspended sediment, tidal amplification, altered current dynamics, changes in salinity and water quality etc. (e.g. Torres et al., 2009; Wilber and Clarke, 2001; Winterwerp and Wang, 2013). On the other hand, periodic maintenance dredging operations are necessary to maintain the appropriate bottom depth of the navigable channels. These recurrent dredging activities may have serious repercussions on the coastal environment, although to a lesser extent than capital dredging, since they may also alter the bottom topography, resuspend sediments, release pollutants, modify the water column and lead to the removal of a stable substrate (Donázar-Aramendía et al., 2018; Jones et al., 2015; Rehitha et al., 2017). The container port industry has experienced phenomenal growth along the past decades since the era of containerization, where shipping currently moves over 80% of world's commodities (Tsolaki and Diamadopoulos, 2010; Yap and Lam, 2013). The growth of world trade will increase the number of ships and their capacities, which consequently require extensive dredging services in coastal areas to reach ports (Yap and Lam, 2013).

Coastal ecosystems are among the most ecologically and economically important worldwide (Barbier et al., 2011). In particular, estuaries are sites of important connectivity and intense gradients that make them highly productive ecosystems with an essential nursery function for many species (Elliott et al., 2019). At the same time, they are dynamic and complex systems where high variability of the physical-chemical gradients makes them one of the most stressful aquatic environments for aquatic fauna (González-Ortegón et al., 2010, 2015). The constant fluctuation of environmental characteristics such as temperature, turbidity, oxygen and salinity due to tidal dynamics and freshwater inputs results in singular communities inhabiting these ecosystems (Day et al., 2013). Therefore, alterations in these ecosystems that occur due to anthropic disturbances could be difficult to distinguish from natural changes (Elliott and Quintino, 2007). Achieving an accurate assessment of anthropic impacts is necessary to improve the management of coastal development while maintaining a balance with a ‘good ecological status’ of the coastal environment (Borja and Elliott, 2007).

Impacts on benthic communities as a consequence of dredging have been documented in numerous studies (e.g. Bemvenuti et al., 2005; Donázar-Aramendía et al., 2018; Ponti et al., 2009). However, organisms that inhabit the water column, such as plankton or fishes, remain largely unquantified. Although dredging often has more repercussions on benthic communities due to the relative immobility of organisms (Simonini et al., 2005), extensive literature has demonstrated that dredging can directly impact fishes (Kjelland et al., 2015; Wenger et al., 2017) and their associated habitats (Erftemeijer and Lewis, 2006; Jones et al., 2016). Many studies have analysed the different effects of increased suspended sediment on behaviour (Collin and Hart, 2015), predation (Ohata et al., 2011) and physiology (Au et al., 2004); other investigations observed the effect of released contaminants such as metals or hydrophobic organic pollutants (Haynes and Johnson, 2000); some of them researched the entrainment of fish in different life-history stages (Reine et al., 1998); and a few studies assessed the dredging sounds (Reine et al., 2014). Notwithstanding, most of these studies were carried out in the laboratory under controlled environments, whereas in situ investigations are very scarce. The methodology to assess the real impacts over the organisms that inhabit in the water column are usually difficult to apply in the field due to the continuous changes in multiple variables such as current dynamics, tidal conditions, the salinity gradient, mobility of the organisms etc.

Recent studies show that adult fish are more likely to undergo sublethal stress from dredging operations rather than lethality because of their ability to move away from or out of an area of higher impact to one of lower impact (Wenger et al., 2018). However, larvae and eggs are subject to lethal impacts more frequently due to their lower mobilities (Wenger et al., 2018), as are small individuals of macrozooplankton or hyperbenthos (Hoffmann and Dolmer, 2000). For this reason, early life stages of fishes or plankton could be more sensitive and may show more clearly the impacts of dredging in species that inhabit the water column.

Our in situ study aimed to determine different effects on small organisms that inhabit the water column, such as early life stages of fish, macrozooplankton and hyperbenthos species, during a maintenance dredging operation in different zones of an estuary with a horizontal salinity gradient, which is considered one of the most important coastal areas of the region for its nursery function (Miró et al., 2020). Two temporal aims were proposed: i) to analyse the immediate effects of a working trailer suction dredger on the physiochemical variables and macrofauna species present in the water column and ii) to analyse the accumulated short- and medium-term effects of a maintenance dredging operation on the physiochemical variables and main macrofauna species present in the water column of two zones with different salinity ranges.

## 2. Material and methods

### 2.1. Study area

The Guadalquivir estuary is located in the south-west of the Iberian Peninsula, a warm temperate region, and its waters flow into the Gulf of Cadiz (Atlantic Ocean). The estuary extends 110 km inland from its mouth. It is a well-mixed mesotidal system with a 3.5-m amplitude range (spring tides) in the river mouth (Díez-Minguito et al., 2012), which presents a longitudinal salinity gradient with temporal displacement by tides, discharges and seasonal variations (González-Ortegón et al., 2014). The morphology of the estuary is a single channel mostly isolated from surrounding natural areas, with a main navigable channel of 7.1 m average depth, which is dredged every one or two years to guarantee the navigation depth (Ruiz et al., 2015). In autumn 2017, a maintenance dredging operation was carried out in several zones of the estuary. The dredging work was performed by a trailer suction dredge. Our study was focused on two dredging zones (Fig. 1), one in the polyhaline water mass and the other in the mesohaline water mass. Approximately 19,600 and 20,500 $m^3$ of dredged material was extracted in each zone, respectively, and dredging was carried out for 15 days (18-11-2017 to 3-12-2017).

### 2.2. Field sampling

To analyse the immediate effect on the whole water column, biological samples were collected against the main water current, before (in front of the bow) and after (behind the stern) the water mass would have passed the dredging vessel while it was working, at three different moments (Fig. 2). Samples of physiochemical variables were also collected. To analyse the short- and medium-term cumulative effects, biological and physiochemical samples were collected in three cruises before, five cruises during and three cruises after the dredging with four samples in every zone. In order to analyse the intra-seasonal trends in abundances of the main organisms found in the estuary, comparisons of a monthly monitoring sampling were performed in the same zones using the same periods of the two years prior to 2017. As this study was carried out during a monthly monitoring program in the same zones from 2015 to 2018, samples from similar periods (autumn) were used for comparison of the biological trends. No dredging operations were carried out in 2015, which was used as the natural trend, while a similar dredging operation was performed in 2016, which was used for comparison with the effects of 2017.

Biological samples were collected with a plankton net of 1 m diameter and 1 mm mesh size equipped with a General Oceanics 2030R flow meter. Oblique tows of 10 min ($305 \pm 46$ $m^3$; mean ± SD) were

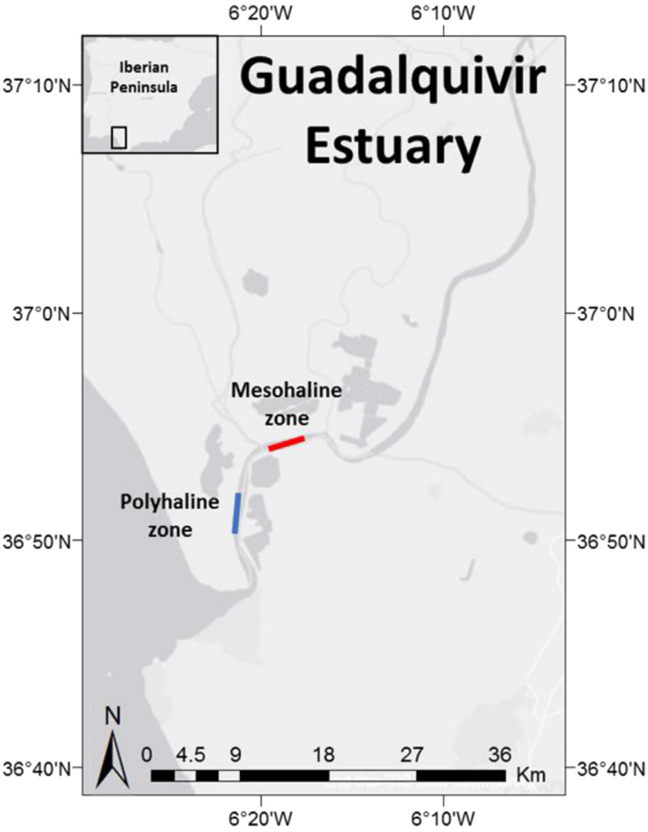


**Fig. 1.** Study area of Guadalquivir estuary with dredging zones.

performed with a boat against the water current at a speed of 2–2.5 knots. Samples were fixed in 70% ethanol. The early fish stages and the rest of the macrofauna groups were sorted. Fishes were counted and, whenever possible, identified to the species level. The rest of the macrofauna were quantified as biomass per group (mysids, decapods and isopods) being blotted dry and wet-weighted with a balance (0.01 g).

Physiochemical profiles of the whole water column were recorded before every plankton tow with a multiprobe (depth, temperature, salinity, turbidity, dissolved oxygen concentration [DO], pH and chlorophyll-a concentration [Chla]; Eureka™ Manta2).

Water samples were taken at mid-depth with a Niskin bottle to measure the concentrations of total suspended solids (TSS), inorganic nutrients ($NO_2$, $NO_3$, $NH_4$, $PO_4$, $SiO_4$) and metals (As, Cd, Co, Cr, Cu, Fe, Ni, Pb, Zn). To measure TSS, water was filtered through 0.7 μm pore pre-combusted (4 h, 500 °C) filters (Whatman GF/F); thereafter, filters were dried (24 h, 60 °C) and weighed. Suspended organic (SOM) and inorganic matter (SIM) were obtained as weight loss by ignition (500 °C, 4 h). Concentrations of $NO_2$, $NO_3$, $NH_4$, $PO_4$ and $SiO_4$ were determined in filtered (0.7 μm) water samples, in a segmented flow autoanalyzer (Skalar, San$^{Plus}$) based on classic spectrophotometric methods (Grasshoff et al., 2007). The accuracy was ±0.08 μM for nitrate, ±0.002 μM for nitrite, ±0.03 μM for ammonium, ±0.03 μM for silicate and ±0.03 μM for phosphate. Analysis of metal concentrations in water samples was determined by ICP-OES (Varian ICP 720-ES) equipped with ultrasonic nebulizer CETAC U5000AT+ after filtration through Nylon filters (pore size = 0.45 μm) and acidification with 2% $HNO_3$ (30%). Water samples acidified were stored one month before extraction. Calibration and Quality Control (QC) solutions were prepared from an ICP multi-element standard solution IV Certipur obtained from Merck and Spectrascan certified reference solution from LGC Standards GmbH (Wesel, Germany). The accuracy of the analytical methods was assessed through reference water sample (TR-434 Trace of metals in drinking water) from INTER 2000 Program (Trace Elements in EstuarineWater CRM 505 No. 048). The recoveries were 89.2–109.4% for all the metals. The differences in metal concentrations between analysed and certified values were generally <10%.

### 2.3. Data analysis

To investigate the effects of the dredging operation on the different variables measured, generalized linear mixed models (GLMMs) were applied using 'Lme4' (Bates et al., 2015). Different experimental designs were run for every approach.

1) In the immediate approach, models were applied to the most abundant (>2% of total) fish species and main macrofauna groups caught. The normal distribution was the best fit for biomass of macrofauna groups, and the Poisson distribution for count data of fish species with the log of filtered volume as an offset variable. If the model showed high overdispersion (>2), a negative binomial distribution was applied instead of the Poisson distribution. The experimental design included two factors: one fixed factor 'Moment' (with two levels, 'Before and After') and one random factor 'Cruise' (with three levels, '1, 2 and 3').
2) In the short- and medium-term approach, models were applied to the most abundant fish species and macrofauna groups, as well as all environmental variables measured for every zone (polyhaline and mesohaline) separately. The normal distribution was the most adequate for most of the response variables, except for counts of fish species abundances, for which we used a negative binomial

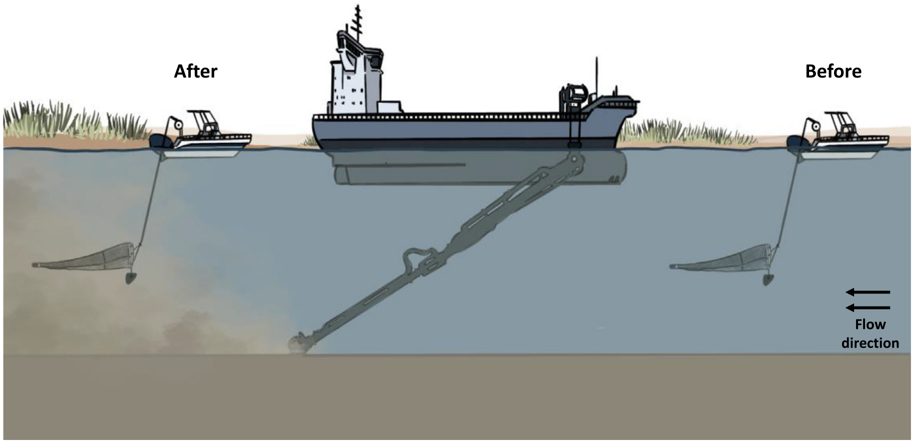


**Fig. 2.** Experimental design for analysis of the immediate effects in the water column before and after the water mass passed by the working dredger.

distribution and the log of the filtered volume as an offset variable. The experimental design included two factors: one fixed factor 'Period' (with three levels, 'Before, During and After') and one random factor nested within Period 'Cruise' (within three levels Before, five levels During and three levels After). If the Period factor was significant, a post hoc pairwise comparison between levels was performed using the package 'emmeans' (Lenth, 2018).

Generalized additive models (GAMs) were fitted for depth profiles of physiochemical variables recorded with the multiprobe. As GLMMs uses a single value of every predictor variable for every value of the response variable, predicted values of GAMs were depth averaged to obtain a single value representative of the complete water column. The results for all variables were plotted by zone using the package 'ggplot2'. Analyses were performed using the R 3.5.2 software (R Core Team, 2018).

Additionally, daily mean discharge from Alcalá del Río Dam during the whole study period was observed to analyse the influence of freshwater input (data provided by Confederación Hidrográfica del Guadalquivir, http://www.chguadalquivir.es/saih/DatosHistori-cos.aspx) on the environmental variables measured.

## 3. Results

### 3.1. Immediate effects

### 3.1.1. Environmental analysis

Profiles of the different environmental variables recorded with the multiprobe are plotted in Fig. 3. A general pattern was found after dredging for chlorophyll, turbidity and salinity variables, which consisted of a homogenization of the whole water column with similar values at the surface to those at the bottom. Only dissolved oxygen showed the inverse tendency, with higher values at the surface after dredging. The pH maintained homogeneous values at both moments, similar to those of temperature, except in cruise 1.

### 3.1.2. Biological analysis

The fish species found were *Engraulis encrasicolus* (45.6%), *Pomatoschistus* spp. (44.9%), *Aphia minuta* (2.9%), *Pomadasys incisus* (2.3%), *Solea senegalensis* (2.2%), *Sardina pilchardus* (1.1%), *Argyrosomus regius* (0.5%), *Anguilla anguilla* (0.3%) and *Gobius paganellus* (0.2%). Among the rest of the macrofauna groups, mysids were the most abundant (77.9%; e.g. *Rhopalophthalmus tartessicus*, *Mesopodopsis slabberi* and *Neomysis integer*), followed by decapods (17.4%; e.g. *Palaemon* spp. and *Crangon crangon*) and isopods (4.6%; e.g. *Synidotea laticauda* and *Lekanesphaera rugicauda*).

Species showed different responses after the water mass passed by the dredger, although the variations between moments hindered the discovery of clear patterns (Fig. S.1). Only the mysids and *Solea senegalensis* showed significant differences, increasing in all cruises (Table 1). Also, the anchovy *Engraulis encrasicolus* tended to increase. In contrast, decapods, *Pomatoschistus* spp., *Aphia minuta* and *Pomadasys incisus* tended to decrease. Isopods maintained stable densities in front of and behind the dredge.

### 3.2. Short- and medium-term effects

### 3.2.1. Environmental analysis

The duration of the study was 81 days between the first and the last cruise, and different temporal patterns were observed in the different physiochemical variables. Statistically significant differences in the fixed effect 'Period' from GLMM on all variables are summarized in the plots by letter codes and extended in Table S.1.1. Daily mean freshwater input into the estuary (Fig. 4A) increased (discharges higher than 50

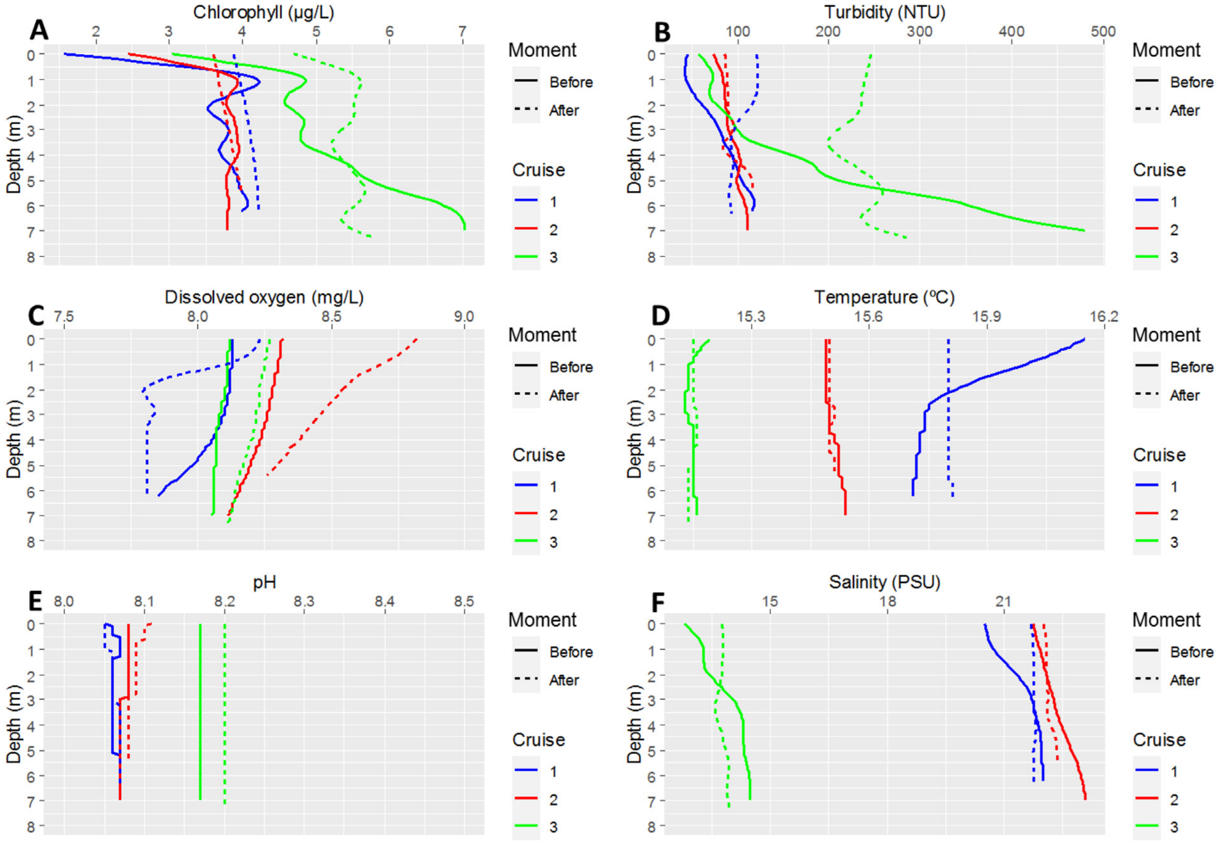


**Fig. 3.** Vertical profiles of the environmental variables (A: chlorophyll; B: turbidity; C: Dissolved oxygen; D: Temperature; E: pH; F: Salinity) before (solid line) and after (dashed line) the water mass passed by the working dredger during the three different cruises.

**Table 1**
Results of the fixed effect "Moment" for GLMM on the main early fish species (count data) and macrofauna groups (biomass data) of the immediate approach. Level "Before" was used as the intercept to calculate estimates.

| | Estimate | SE | z value | *p* value |
|---|---|---|---|---|
| *Engraulis encrasicolus* | 0.395 | 0.205 | 1.924 | 0.054 |
| *Pomatoschistus* spp. | −0.289 | 0.405 | −0.714 | 0.475 |
| *Aphia minuta* | −0.369 | 0.492 | −0.75 | 0.453 |
| *Pomadasys incisus* | −0.36 | 1.195 | −0.302 | 0.763 |
| *Solea senegalensis* | 1.778 | 0.763 | 2.329 | 0.019 |

| | Estimate | SE | t value | p value |
|---|---|---|---|---|
| Mysids | 41.85 | 10.2 | 4.103 | >0.001 |
| Decapods | −10.17 | 23.27 | −0.437 | 0.662 |
| Isopodos | 0.376 | 1.169 | 0.322 | 0.747 |

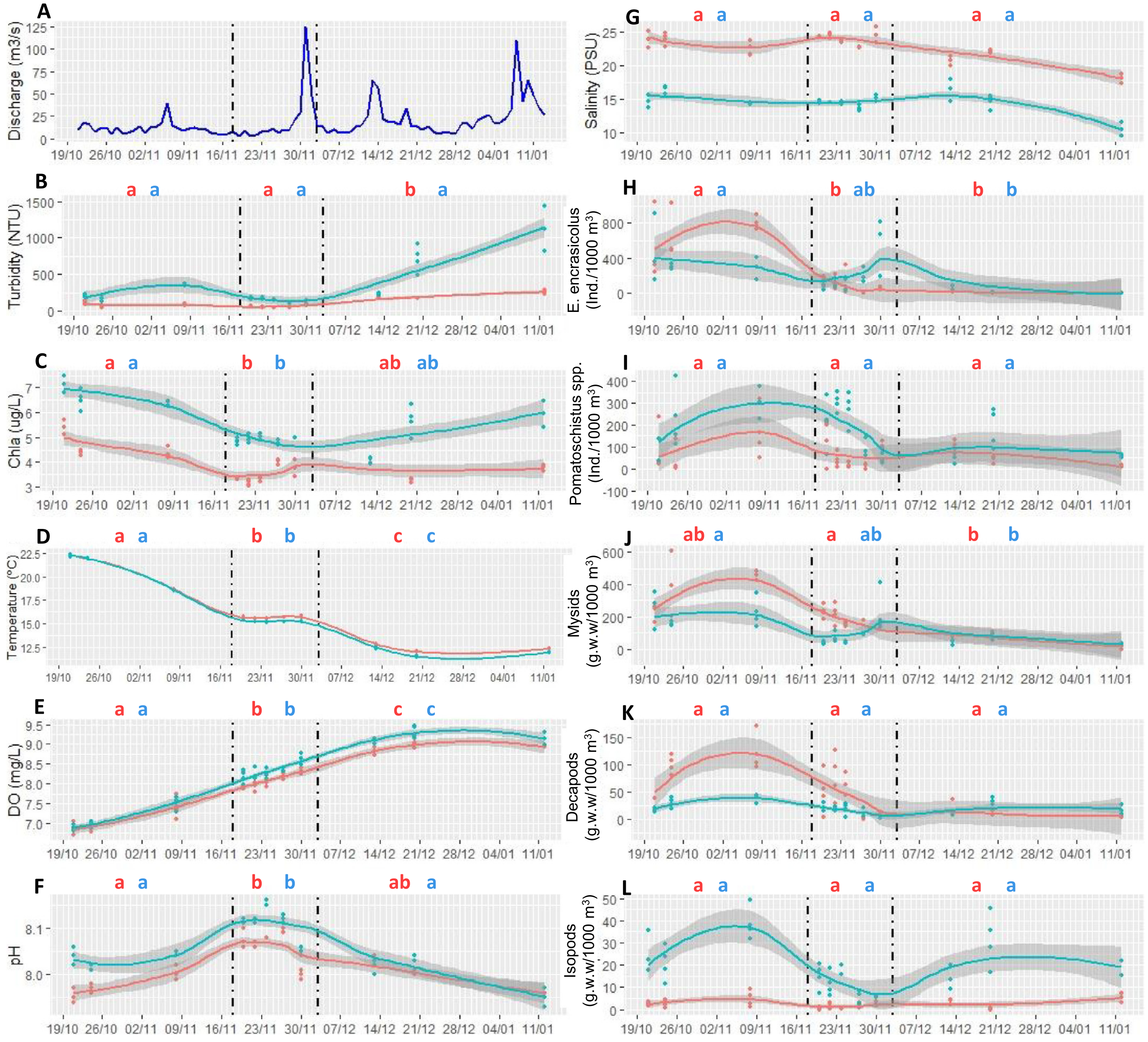


**Fig. 4.** Daily mean of freshwater inputs in the Guadalquivir estuary (A) and temporal values of physiochemical variables measured in the water column (turbidity [B], chlorophyll [C], temperature [D], dissolved oxygen [E], pH [F], salinity [G]) and biological (*Engraulis encrasicolus* [H], *Pomatoschistus spp.* [I], mysids [J], decapods [K], isopods [L]). Dashed lines point out the dredging period. Solid lines are smoother models with the loess method for polyhaline (Red) and mesohaline (Blue) zones. Grey shading indicates standard errors. Different letters indicate significant differences ($p < 0.05$) between levels of the Period factor (before, during and after) from GLMM in every zone (Red letters: polyhaline; Blue letters: mesohaline).

$m^3/s$) at the end of the dredging period and thereafter. Turbidity did not show any change during dredging, while its values increased during the post-dredging period, being significant in the polyhaline zone (Fig. 4B). Chlorophyll concentration showed a decrease during the dredging period and a soft recovery thereafter (Fig. 4C). Water temperature decreased gradually from 22.5 °C to 12.5 °C, typical of the autumn-winter transition of temperate regions in the northern hemisphere, although this decline ceased during dredging (Fig. 4D). Dissolved oxygen showed the inverse trend during the study (Fig. 4E), although their values increased more during the dredging period despite the fact that temperature was stable during these dates. The pH increased in both zones during dredging, decreasing to pre-dredging values afterwards (Fig. 4F). Salinity was stable in both zones during the whole study (Fig. 4G), although it tended to decrease after dredging associated with the increment of freshwater input in this period.

TSS, directly correlated with SIM and SOM (Fig. 5A–C), showed the same pattern as turbidity, being the post-dredging period when higher concentrations were found in every zone. Still, the wide range of turbidity levels found in the mesohaline zone did not make this zone significantly different (Table S.1.2). Silicate and nitrite concentrations did not show differences (Fig. 5D–E). Nitrates showed a gradual increment in both zones, with significant differences in the polyhaline zone between all periods (Fig. 5F). Ammonium showed a similar trend, but without significant differences (Fig. 5G). Phosphates did not show any difference in the mesohaline zone, while values measured in the polyhaline zone were higher during dredging, with a partial recovery thereafter (Fig. 5H).

Similar trends between zones were found in the concentration of most metals analysed, although the mesohaline zone did not show significant differences between periods for any of them (Fig. 6 and Tables S.2.1 and S.2.2). As, Cd, Co, Cu and Pb did not show any clear patterns due to variations between replicates (Fig. 6A–D, H). Ni, despite not being significantly different between periods, showed a cumulative trend during the dredging period (Fig. 6G). Cr and Fe started to increase gradually during the dredging and continued in the post-dredging period (Fig. 6D, F). Zn showed the highest relative increment during dredging, with different trends after dredging for the polyhaline zone, in which it decreased slightly, and the mesohaline zone, in which it remained high but with oscillations (Fig. 6I).

### 3.2.2. Biological analysis

A total of 16 fish species in early life stages were found with two dominant species that comprised more than 90% of the total abundance of the fish assemblage, the anchovy *Engraulis encrasicolus* (58.3%) and the goby *Pomatoschistus* spp. (32.8%). In relation to the rest of the macrozooplankton and hyperbenthos groups, similar species to those described in the immediate approach were found, with mysids showing the highest biomass (76.6%), followed by decapods (17.6%) and isopods (5.3%).

Temporal series of the densities of the most abundant fish species and the rest of the macrofauna are plotted by zone in Fig. 4. A general pattern was observed for most taxa, which showed a wider density variation before and during dredging cruises, as well as a temporal decrease along the whole study. Notwithstanding, distinct responses and significant differences were found between periods for every species (Table S.3). Anchovy and mysids showed similar patterns in both zones, with a marked decrease from the beginning of dredging, especially in polyhaline waters, that continued until the last cruise (Fig. 4H, J). Isopods, goby and decapods did not show differences between periods due to high intra-period variability, although different tendencies were observed (Fig. 4I, K, L). Isopod densities were too low in the polyhaline zone to detect any change along periods, while in the mesohaline zone, its biomass showed a notable descent during dredging in comparison with the last cruise of the period before; also, it showed a partial recovery in the second cruise after dredging, although it did not continue in the next one. Gobies and decapods showed a similar trend, but in opposite zones, with a decrease at the end of the dredging, which also continued in the period after dredging.

Interannual comparison (2015, 2016 and 2017) of *E. encrasicolus*, *Pomatoschistus* spp., and the rest of the macrofauna groups together (the main component was mysids) in every zone are plotted in Fig. 7. Anchovy (Fig. 7A) showed stable densities in the polyhaline zone during the whole period in 2015 and 2016, despite the dredging operation carried out in the latter year. Instead, 2017 presented higher anchovy abundances before dredging, but it decreased during the operation to similar levels as in previous years on the same dates. The densities of anchovies in the mesohaline zone showed a different trend, with a gradual decrease but with oscillations depending on the year. In fact, the natural inter-month variations found in 2015 where higher than those

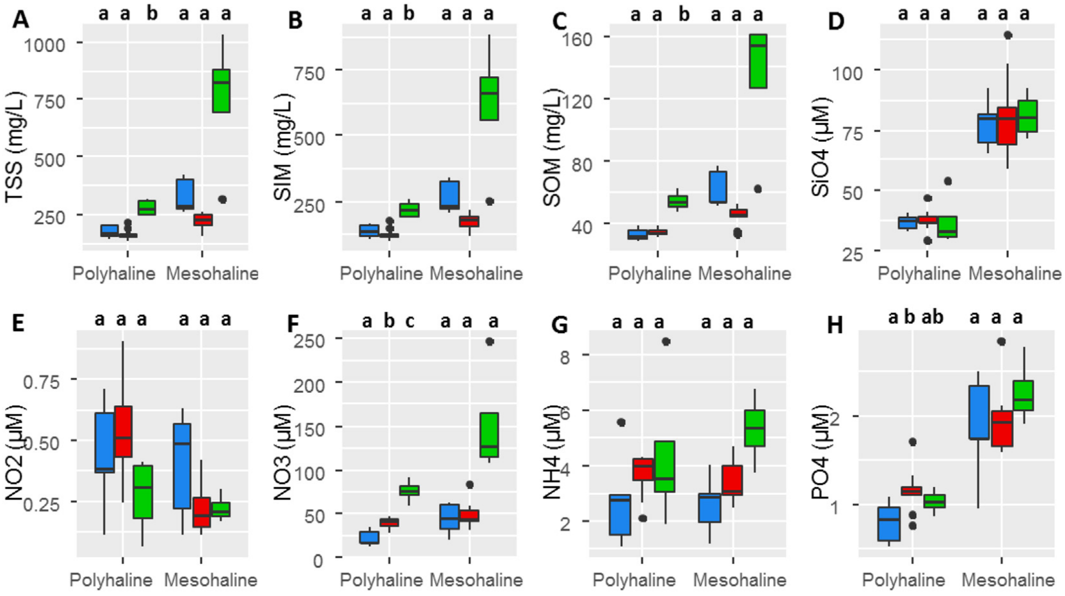


**Fig. 5.** Boxplot of total suspended solids (A), suspended inorganic matter (B), suspended organic matter (C), $SiO_4$ (D), $NO_2$ (E), $NO_3$ (F), $NH_4$ (G) and PO4 (H) in every zone. Blue: before dredging; red: during dredging; green: after dredging. Different letters indicate significant differences ($p < 0.05$) between levels of the Period factor from GLMM in every zone.

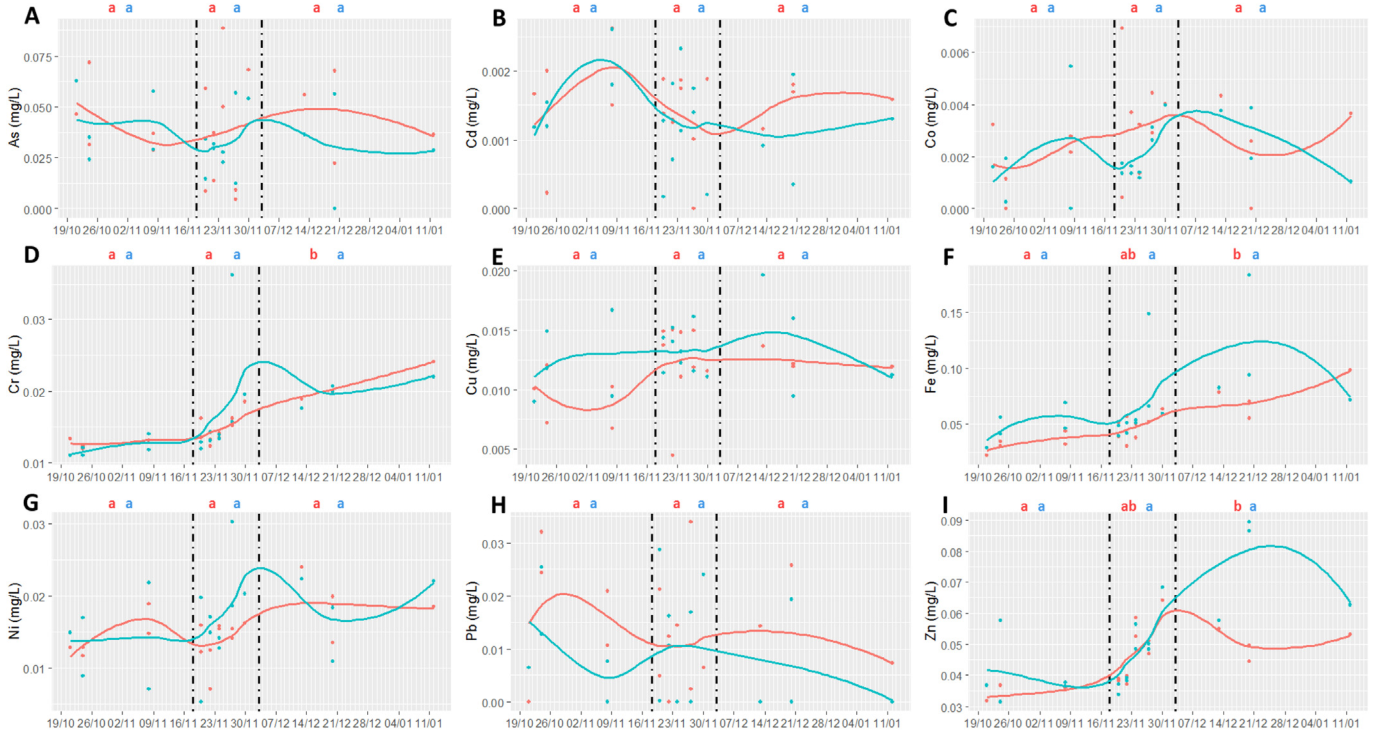


**Fig. 6.** Temporal series of metal concentrations (As, Cd, Co, Cr, Cu, Fe, Ni, Pb, Zn; A-I respectively) in the water column. Dashed lines point out the dredging period. Solid lines are smoother models with the loess method for polyhaline (Red) and mesohaline (Blue) zones. Different letters indicate significant differences ($p < 0.05$) between levels of the Period factor from GLMM in every zone (Red letters: polyhaline; Blue letters: mesohaline). Values lower than limit of detection were plotted as 0.

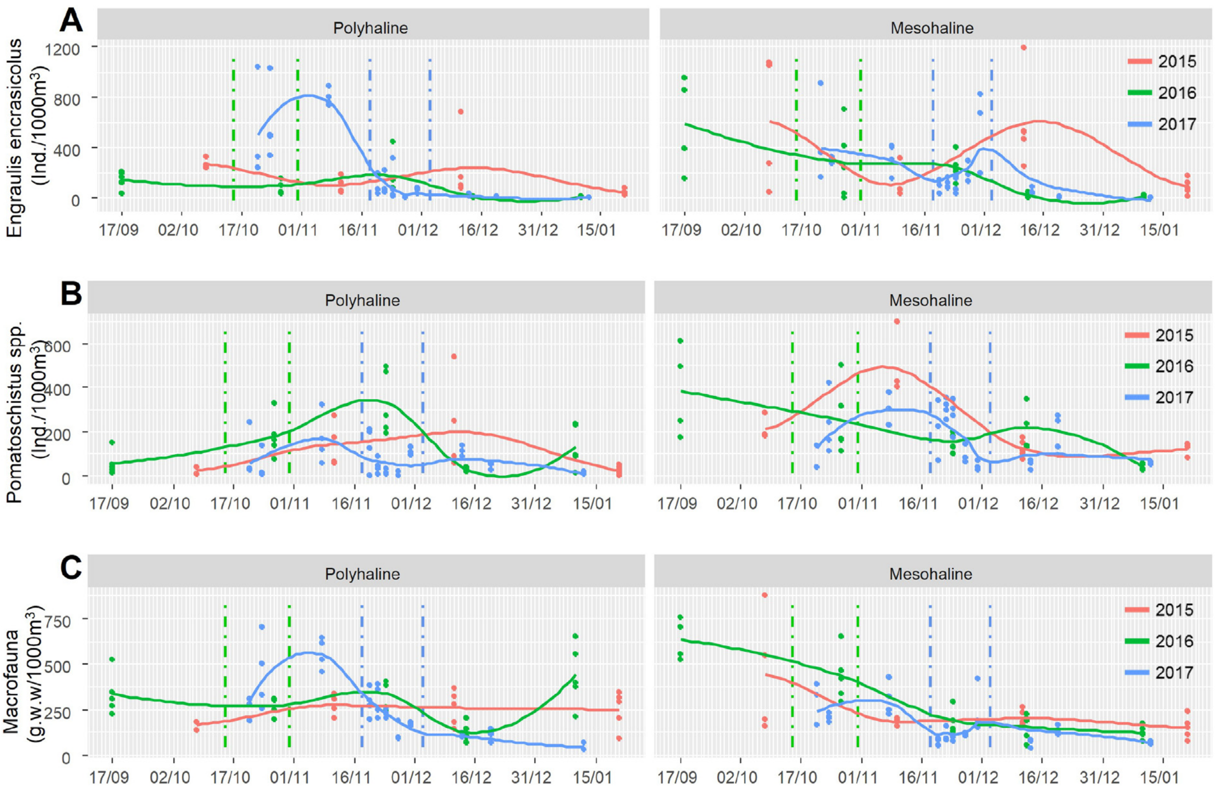


**Fig. 7.** Density of *Engraulis encrasicolus* (A) and *Pomatoschistus* spp. (B), and biomass (wet weight) of the rest of the macrofauna (C) during the dredging study (2017) and on similar dates in previous years (2015 and 2016) in every zone. Dashed lines point out the dredging periods: green color is for 2016 and blue for 2017. Solid lines are smoother models with loess method for every year.

observed in years with dredging. The trend of the rest of the macrofauna (Fig. 7C) was similar to that in anchovies in the polyhaline zone, and a gradual decrease was repeated for all years in the mesohaline zone. The goby (Fig. 7B) showed a distinct pattern in both zones in all years, except for the mesohaline zone in 2016, with a progressive increment at the beginning of the studied period that varied between dates, to finally decrease with different slopes.

## 4. Discussion

Estuaries are ecosystems with wide environmental variations that make it difficult to detect anthropic stress from natural changes, unless the human impact is severe, leading to the 'Estuarine Quality Paradox' (Elliott and Quintino, 2007). In addition, the impossibility of establishing control replicates per zone, due to the absence of polyhaline or mesohaline waters without dredging in the same estuary, makes it necessary to interpret these results with caution. Notwithstanding, the immediate-, short- and medium-term design applied in this study, in addition to interannual comparisons, helped to discriminate different effects of a dredging operation in biological and physiochemical variables of the water column in a highly fluctuating estuary such as Guadalquivir.

Among the physiochemical variables, only DO and pH showed a clear influence of dredging, increasing their levels. These unexpected observations contrast with the general assumption that sediment resuspension releases chemical substances which react with DO, temporally reducing its concentration and acidifying the water (Jones-Lee and Lee, 2005). In this case, observing the stratification of DO in the immediate approach, it is possible that mechanical perturbation, such as by the ship's propeller and/or cavitation, in addition to the action of the trailer arm, could mix the water column with atmospheric air (Bowie et al., 1985), balance the chemical demand of DO and even increase its values. Nonetheless, the DO concentration never reached levels lower than 6 mg/L during the whole study at any depth in both zones, and the pH increased by only 0.1, which did not seem to have a relevant negative effect in well oxygenated waters with low temperatures, as reported by Jabusch et al. (2008).

On the other hand, DO depletion is also associated with persistent high turbidity levels, which reduce light penetration in the water column and limit photosynthetic activity (Desmit et al., 2005). Still, Guadalquivir estuary is considered a turbid system where the primary production in the water column is scarce and constrained to the surface layer (Ruiz et al., 2015). Hence, oxygen production by phytoplankton would be influenced little by a temporal increase in turbidity. Turbidity increase, directly correlated with TSS, has been widely studied as a common effect of dredging operations in coastal areas, which may affect marine biodiversity (Magris and Ban, 2019; Wenger et al., 2017), being able to cause lethal and sublethal impacts in 10% and 20% of fish species, respectively (Wenger et al., 2018). An analysis of surface satellite images showed increments of total suspended solids (1000 mg/L approx.) in plumes during a dredging operation in the upper zones of the same estuary (Caballero et al., 2018). These observations coincided with the immediate profiles recorded after the water mass passed by the working dredger, when the homogenization of the water column increased the turbidity levels on the surface. But surprisingly, we did not detect significant changes in the depth-averaged values of turbidity. The dredging operation must have necessarily increased the suspended sediment and, consequently, the turbidity, but the effect seems to be spatially very local and its persistence temporally short, and we could neither detect significant changes in the short- nor in the medium-term approaches. Nonetheless, turbidity, TSS, SIM and SOM concentration increased after dredging, which seems to be rather associated with higher freshwater inputs in this period. González-Ortegón et al. (2010) reported similar observations during other freshets in the same estuary (up to 700 NTU approx.), describing adverse effects on the estuarine food web at different levels. Increasing exposure to suspended sediment may causes damage to gill tissue and structure, as it is easier to clog the gills and reduce their efficiency in smaller fish and larvae (Au et al., 2004). The Guadalquivir estuary has been considered the most productive and important nursery area in the region in spite of being the most turbid (Miró et al., 2020). The high flows originated by tides and the high chronic turbidity in the Guadalquivir estuary (Losada et al., 2017) could have overshadowed the sediment resuspension effect caused by dredging in our sampling zones; consequently, we could not observe a clear increase nor any direct effect on the species found.

The dynamics of chlorophyll concentration appear to follow a seasonal pattern related to temperature reduction, provided that dredging did not significantly modify turbidity and, consequently, light penetration. However, higher values were observed in the latter cruises while temperature continued decreasing and turbidity increased. These observations could be explained by imports from upstream waters with higher primary production caused by rising freshwater inputs (González-Ortegón and Drake, 2012) and/or a higher resuspension of microphytobenthos from riversides (Díez-Minguito and de Swart, 2020; Miró et al., 2020). Also, nutrients such as nitrate and ammonium, which presented similar tendencies to those of chlorophyll, could be imported due to sewage effluents and nitrification processes from upstream waters close to urban and agricultural areas (Mendiguchía et al., 2007). Different stages of ammonium could be found depending on pH levels, with a higher un-ionized (ammonia [$NH_3$]) proportion associated with high pH, which has been considered toxic to fishes (Brinkman et al., 2009). In our case, the changes of pH observed during the dredging could increase the $NH_3$ proportion by around 0.1–0.2% (poly- and mesohaline zones respectively), which translates to a total concentration of 0.1 μM $NH_3$ during that period. Therefore, un-ionized ammonia showed levels far below the toxic reference value of 1.16 μM $NH_3$ (Eddy, 2005). The concentration of phosphate, whose increment is usually attributed to fertilizers via river flow (Mainstone and Parr, 2002), showed changes during dredging in polyhaline waters, although the concentrations reached were lower than the levels found in the mesohaline zone. Globally, we could not clearly assign an increase in inorganic nutrients due to dredging operations.

In contrast, dredging has been primarily related to remobilized metals associated with sediment particles in the water column, which change its environmental conditions and promotes the shift of metals from the particulate in the dissolved state (Van Den Berg et al., 2001; Chen et al., 2020). This phenomenon was even more noticeable in the Guadalquivir estuary given that it received a toxic spill from the Aznalcóllar mine in 1998 (Riba et al., 2002); although 10 years afterwards, studies showed that a decline in the metal contamination in the area was evident (Tornero et al., 2011, 2014). It is known that oxidation of sulphides liberates different heavy metals because the precipitates are degraded (Caille et al., 2003). This phenomenon could be observed in our case for Cr, Fe, Ni and Zn, with a cumulative trend during dredging. However, the oxidation of Fe also causes precipitation of iron-(oxo)hydroxides (Dang et al., 2020), which could form a very strong adsorptive layer on the surface of the new dredged bottom and decrease the release of metals (Goossens and Zwolsman, 1996). On the other hand, a previous study in the Guadalquivir estuary of heavy metal concentrations in the sediment of the same zones observed an increment of As, Co and Ni after dredging operations, especially in the polyhaline site (Donázar-Aramendía et al., 2018). In our water samples, greater effects were found in the same zone, where slight differences could be due to the salinity of the water, the oxidation-reduction potential of the sediment and the pH of the sediment pore water and overlying water on site (Eggleton and Thomas, 2004; Roberts, 2012).

The concentration of dissolved metal provided could be overestimated here with regard to other studies that use filters with smaller pore size (for instance, 0.22 μm; González-Ortegón et al., 2019), due to the higher presence of colloids. Consequently, the precision to detect smaller variations in metal concentrations of the dissolved phase could be lower. Still, maximum values recorded in the field were lower than the minimal concentration used in controlled laboratory experiments to test for lethal effects of trace metals in the early life stages of fishes (Jezierska et al., 2009). Experiments in crustacea also show that the ranges measured usually do not cause significant effects on individuals (Fetters et al., 2016; Lavolpe et al., 2004; Martin and Holdich, 1986). Thus, the metal uptake for fish and the rest of the macrofauna present in both zones is expected to be low in this study. In addition to the effect of dredging, the daily tidal currents (Jonas and Millward, 2010), wind energies or storms (Birch and O'Hea, 2007) in estuarine systems can cause periodical remobilization of surface sediments, releasing metals naturally. The higher freshwater inputs observed after dredging, and the associated increment in TSS, could also help to maintain the increased values of some metals. On the other hand, some metals can be released and/or re-absorbed more readily than others (Maddock et al., 2007). Faster release and re-absorption could be occurring for Ni and Cr, which increased quickly only during dredging, and slower release and re-absorption for Zn and Fe, which reached higher levels after dredging. Also, fine sediments could remain longer in suspension and consequently liberate more metals after dredging (Maddock et al., 2007). The Guadalquivir estuary has shown high persistent turbidity events in wet years (González-Ortegón et al., 2010) and is considered one of the estuaries in the region with higher metal fluxes (González-Ortegón et al., 2019) due to urban and agriculture supports (Mendiguchía et al., 2007). As organisms take up and accumulate trace metals during their whole life cycle, there is the potential for toxic effects over time (Rainbow, 2007), special attention should be paid to metal release after higher bottom disturbance situations such as longer dredging periods or torrential freshets. Still, this is more likely to be a concern with longer-living estuarine organisms, such as adult resident fishes, than with short-lived ones, as most species included in this study.

Physiochemical alterations caused by dredging operations appear to be minor in comparison with the natural changes observed; however, other possible effects have been observed in planktonic organisms. The limited swimming capacity of small individuals could make it impossible for them to avoid the water mass affected by dredging. Therefore, the main and direct cause that could decrease plankton densities after the water mass passes by the working dredger is hydraulic entrainment, which leads to their death from the mechanical action of the suction arm (Reine and Clarke, 1998).

Different tendencies were found for hyperbenthic and pelagic species in the immediate approach. Hyperbenthic species, like *Pomatoshcistus* spp. or decapods, tended to decrease, probably due to the direct impact of physical removal of bottom sediments inhabited by hyperbenthic organisms (Hoffmann and Dolmer, 2000). In fact, a previous study showed that *Pomatoschistus* spp. is prone to a high entrainment by a trailer suction with estimated rates between 0.0018 and 0.009 ind./$m^3$ (Drabble, 2012a). Also, Armstrong et al. (1982) reported that sand shrimp (*Crangon* sp.) showed the highest rates of entrainment by dredges in Pacific northwest estuaries, with a range between 0.08 and 4.44 ind./$m^3$, and estimated a population decrease during a dredging project of around 1.2%–6.5%. In contrast, pelagic species like *E. encrasicolus*, *S. pilchardus*, *P. incisus* or mysids tended to maintain stable densities, or even to increase in density in some cases. Their behaviour of inhabiting the water column could minimize the entrainment risk. Most studies report demersal organisms entrained (Barletta et al., 2016; Reine and Clarke, 1998); however, adult stages of pelagic species have been collected as well, including anchovy (0.001 ind./$m^3$), herring (0.01 ind./$m^3$) and smelt (0.01 ind./$m^3$) (Armstrong et al., 1982). In fish larvae, some authors estimated the entrainment of striped bass (*Morone saxatilis*), herring (*Alosa* spp.) and white perch (*Morone americana*), involving the simultaneous operation of four hydraulic dredges in the Delaware River, and concluded that less than 1% of the total larval population would be entrained by the dredges (Burton et al., 1992). In our case, the lack of a significant difference obtained for any species with this approach suggests that entrainment caused a low incident over hyperbenthic species and no effects on pelagic ones.

On the other hand, the pelagic species exhibited different trends between immediate and short-medium approaches. A high decrease in density was found just after the beginning of dredging in *E. encrasicolus* and mysids in the polyhaline zone, suggesting that this disturbance could affect these species. However, interannual comparisons found that their densities can fluctuate similarly without dredging (anchovy in the mesohaline zone of 2015) or long after the dredging activity (macrofauna in the polyhaline zone of 2016). Also, these organisms showed low and stable values (anchovy in the polyhaline zone of 2015–16) or decreased gradually (anchovy and macrofauna in the mesohaline zone of 2015–16) along this season, in years either with or without dredging operations. The goby showed different trends, notably, decreasing in density in the mesohaline zone of 2015, when there was no dredging operation, and even increasing in the polyhaline zone during and after dredging in 2016. These patterns make it difficult to elucidate whether these changes correspond to the natural variability and reduction typical of this period (Drake et al., 2002) or whether they are an impact of dredging. A multi-year-long monitoring study conducted in the Eastern English Channel (UK), which is dredged annually, observed a temporal and gradual reduction in several species, including *Pomatoschistus* spp. (Drabble, 2012b). In our case, no clear differences were found between the seasonal densities of previous years. Some of these species are present across the whole estuary section, with higher densities in the shallower banks of the Guadalquivir estuary (unpublished data). Further, most species found were marine migrants, such as *E. encrasicolus*, *S. pilchardus*, *S. senegalensis* etc., which locate their spawning zones offshore (Baldó et al., 2006). These behaviours could minimize the mechanical impact of dredging, which was carried out in the estuary and only in the middle channel, constraining the impact on nursery function.

In summary, the observed modifications of water physiochemical variables in comparison with natural changes such as freshets, the non-significant differences found in the immediate approach and the similar temporal fluctuations of density as in previous years with or without dredging suggest that this dredging operation did not cause a severe impact on the Guadalquivir estuary. Still, this does not mean that there is no effect. This ecosystem showed high natural fluctuations, which precluded a clear association of the observed variations with the dredging effects, leading to the 'Estuarine Quality Paradox' (Elliott and Quintino, 2007). Nonetheless, as the possible effects of this disturbance were of the same order or less than those of natural ones, planktonic organisms could be well adapted to cope with them. Still, polyhaline water mass showed more notorious effects than mesohaline waters. In addition, the dredging operation was undertaken during the natural decline period of recruitment, which could minimize the effects over nursery function. However, dredging activity during the main recruitment period and larval development (March to November in the case of Guadalquivir estuary [Drake et al., 2002, 2007]), could directly constrain the larval supply by contributing to higher mortality rates among larvae or lowering recruitment success (Wenger et al., 2017). The evidence found here and the changes detected make it advisable to implement systematic monitoring programs in any dredging project. The accumulated experience and the use of new approaches may allow the nature of the effect of these operations to be more clearly defined, allowing the design of specific control strategies to mitigate impacts, as well as a thorough evaluation of the effectiveness of these strategies (Wenger et al., 2018), thus promoting sustainable fishery management.

## CRediT authorship contribution statement

**J.M. Miró:** Conceptualization, Methodology, Software, Validation, Formal analysis, Investigation, Data curation, Writing – original draft.

**C. Megina:** Conceptualization, Methodology, Formal analysis, Investigation, Writing – review & editing, Project administration. **I. Donázar-Aramendía:** Methodology, Resources, Writing – review & editing. **J.C. García-Gómez:** Supervision, Funding acquisition, Writing – review & editing.

## Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

## Acknowledgements

We thank Autoridad Portuaria de Sevilla and Acuario de Sevilla for financial and logistical support. This work was partially supported by the Plan Propio Universidad de Sevilla via two pre-doctoral grants (I. Donázar-Aramendía, J.M. Miró). We thank all the members of the LBM who participate in field surveys and in samples processing.

## Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2021.151304.

## References

Armstrong, D.A., Stevens, B.G., Hoeman, J., 1982. Distribution and abundance of Dungeness crab and Crangon shrimp, and dredged-related mortality of invertebrates and fish in Grays Harbor, Washington. Tech. Report. Sch. Fish. Univ. Washington, Washingt. Dep. Fish. U.S. Army Eng. Dist. Seattle.

Au, D.W.T., Pollino, C.A., Wu, R.S.S., Shin, P.K.S., Lau, S.T.F., Tang, J.Y.M., 2004. Chronic effects of suspended solids on gill structure, osmoregulation, growth, and triiodothyronine in juvenile green grouper Epinephelus coioides. Mar. Ecol. Prog. Ser. 266, 255–264. https://doi.org/10.3354/meps266255.

Baldó, F., García-Isarch, E., Jiménez, M.P., Romero, Z., Sánchez-Lamadrid, A., Catalán, I.A., 2006. Spatial and temporal distribution of the early life stages of three commercial fish species in the northeastern shelf of the Gulf of Cádiz. Deep Sea Res. Part II Top. Stud. Oceanogr. 53, 1391–1401. https://doi.org/10.1016/j.dsr2.2006.04.004.

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C., Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. Ecol. Monogr. 81, 169–193. https://doi.org/10.1890/10-1510.1.

Barletta, M., Cysneiros, F.J.A., Lima, A.R.A., 2016. Effects of dredging operations on the demersal fish fauna of a south american tropical-subtropical transition estuary. J. Fish Biol. 1–31. https://doi.org/10.1111/jfb.12999.

Bates, D., Mächler, M., Bolker, B., Walker, S., 2015. Fitting linear mixed-effects models using lme4. J. Stat. Softw. 67, 1–48. https://doi.org/10.18637/jss.v067.i01.

Bemvenuti, C.E., Angonesi, L.G., Gandra, M.S., 2005. Effects of dredging operations on soft bottom macrofauna in a harbor in the Patos lagoon estuarine region of southern Brazil. Braz. J. Biol. 65, 573–581. https://doi.org/10.1590/s1519-69842005000400003.

Birch, G., O'Hea, L., 2007. The chemistry of suspended particulate material in a highly contaminated embayment of port Jackson (Australia) under quiescent, high-wind and heavy-rainfall conditions. Environ. Geol. 53, 501–516. https://doi.org/10.1007/s00254-007-0662-5.

Borja, A., Elliott, M., 2007. What does "good ecological potential" mean, within the european water framework Directive? Mar. Pollut. Bull. 54, 1559–1564. https://doi.org/10.1016/j.marpolbul.2007.09.002.

Bowie, G.L., Mills, W.B., Porcella, D.B., Campbell, C.L., Pagenkopf, J.R., Rupp, G.L., Jhonson, K.M., Chan, P.W., Gherini, S.A., 1985. Rates, Constants, and Kinetics Formulations in Surface Water Quality Modeling. EPA/600/3-85/040 June 455 pp.

Brinkman, S.F., Woodling, J.D., Vajda, A.M., Norris, D.O., 2009. Chronic toxicity of ammonia to early life stage rainbow trout. Trans. Am. Fish. Soc. 138, 433–440. https://doi.org/10.1577/t07-224.1.

Burton, W., Weisberg, S., Jacobson, P., 1992. Rep. Submitt. to Delaware Basin Fish Wildl. Manag. Coop. Trenton, NJ. Entrainment effects of maintenance hydraulic dredging in the Delaware River Estuary on Striped Bass Ichthyoplankton.

Caballero, I., Navarro, G., Ruiz, J., 2018. Multi-platform assessment of turbidity plumes during dredging operations in a major estuarine system. Int. J. Appl. Earth Obs. Geoinf. 68, 31–41. https://doi.org/10.1016/j.jag.2018.01.014.

Caille, N., Tiffreau, C., Leyval, C., Morel, J.L., 2003. Solubility of metals in an anoxic sediment during prolonged aeration. Sci. Total Environ. 301, 239–250. https://doi.org/10.1016/S0048-9697(02)00289-9.

Chen, C., Kong, M., Wang, Y.Y., Shen, Q.S., Zhong, J.C., Fan, C.X., 2020. Dredging method effects on sediment resuspension and nutrient release across the sediment-water interface in Lake taihuChina. 27, 25861–25869. https://doi.org/10.1007/s11356-019-06192-w.

Collin, S.P., Hart, N.S., 2015. Vision and photoentrainment in fishes: the effects of natural and anthropogenic perturbation. Integr. Zool. 10, 15–28. https://doi.org/10.1111/1749-4877.12093.

Dang, D.H., Layglon, N., Ferretto, N., Omanović, D., Mullot, J.U., Lenoble, V., Mounier, S., Garnier, C., 2020. Kinetic processes of copper and lead remobilization during sediment resuspension of marine polluted sediments. Sci. Total Environ. 698. https://doi.org/10.1016/j.scitotenv.2019.134120.

Day, J.W., Hall, C.A.S., Kemp, W.M., Yanez-Arancibia, A., 2013. Estuarine ecology. Estuar. Ecol. https://doi.org/10.2307/2937399.

Desmit, X., Vanderborght, J.P., Regnier, P., Wollast, R., 2005. Control of phytoplankton production by physical forcing in a strongly tidal, well-mixed estuary. Biogeosciences 2, 205–218. https://doi.org/10.5194/bg-2-205-2005.

Díez-Minguito, M., de Swart, H.E., 2020. Relationships between chlorophyll-a and suspended sediment concentration in a high-nutrient load estuary: an observational and idealized modeling approach. J. Geophys. Res. Ocean. 125, no. https://doi.org/10.1029/2019JC015188.

Díez-Minguito, M., Baquerizo, A., Ortega-Sánchez, M., Navarro, G., Losada, M.A., 2012. Tide transformation in the Guadalquivir estuary (SW Spain) and process-based zonation. J. Geophys. Res. Ocean. 117, 1–14. https://doi.org/10.1029/2011JC007344.

Donázar-Aramendía, I., Sánchez-Moyano, J.E., García-Asencio, I., Miró, J.M., Megina, C., García-Gómez, J.C., 2018. Maintenance dredging impacts on a highly stressed estuary (Guadalquivir estuary): a BACI approach through oligohaline and polyhaline habitats. Mar. Environ. Res. 140, 455–467. https://doi.org/10.1016/j.marenvres.2018.07.012.

Drabble, R., 2012a. Projected entrainment of fish resulting from aggregate dredging. Mar. Pollut. Bull. 64, 373–381. https://doi.org/10.1016/j.marpolbul.2011.10.033.

Drabble, R., 2012b. Monitoring of East Channel dredge areas benthic fish population and its implications. Mar. Pollut. Bull. 64, 363–372. https://doi.org/10.1016/j.marpolbul.2011.10.035.

Drake, P., Arias, A.M., Baldó, F., Cuesta, J.A., 2002. Spatial and temporal variation of the nekton and hyperbenthos from a temperate European estuary with regulated freshwater inflow. Estuaries 25, 451–468. https://doi.org/10.1007/BF02695987.

Drake, P., Borlán, A., González-Ortegón, E., Baldó, F., Vilas, C., Fernández-Delgado, C., 2007. Spatio-temporal distribution of early life stages of the european anchovy Engraulis encrasicolus L. Within a european temperate estuary with regulated freshwater inflow: effects of environmental variables. J. Fish Biol. 70, 1689–1709. https://doi.org/10.1111/j.1095-8649.2007.01433.x.

Eddy, F.B., 2005. Ammonia in estuaries and effects on fish. J. Fish Biol. 67, 1495–1513. https://doi.org/10.1111/j.1095-8649.2005.00930.x.

Eggleton, J., Thomas, K.V., 2004. A review of factors affecting the release and bioavailability of contaminants during sediment disturbance events. Environ. Int. 30, 973–980. https://doi.org/10.1016/j.envint.2004.03.001.

Elliott, M., Quintino, V., 2007. The estuarine quality paradox, environmental homeostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. Mar. Pollut. Bull. 54, 640–645. https://doi.org/10.1016/j.marpolbul.2007.02.003.

Elliott, M., Day, J.W., Ramachandran, R., Wolanski, E., 2019. A synthesis: what is the future for coasts, estuaries, deltas and other transitional habitats in 2050 and beyond? In: Wolanski, E., John, W., Michael, E., Ramesh, R. (Eds.), Coasts and Estuaries. The Future. Elsevier Inc., p. 729 https://doi.org/10.1016/b978-0-12-814003-1.00001-0

Erftemeijer, P.L.A., Lewis, R.R., 2006. Environmental impacts of dredging on seagrasses: a review. Mar. Pollut. Bull. 52, 1553–1572. https://doi.org/10.1016/j.marpolbul.2006.09.006.

Fetters, K.J., Costello, D.M., Hammerschmidt, C.R., Burton, G.A., 2016. Toxicological effects of short-term resuspension of metal-contaminated freshwater and marine sediments. Environ. Toxicol. Chem. 35, 676–686. https://doi.org/10.1002/etc.3225.

González-Ortegón, E., Drake, P., 2012. Effects of freshwater inputs on the lower trophic levels of a temperate estuary: physical, physiological or trophic forcing? Aquat. Sci. 74, 455–469. https://doi.org/10.1007/s00027-011-0240-5.

González-Ortegón, E., Subida, M.D., Cuesta, J.A., Arias, A.M., Fernández-Delgado, C., Drake, P., 2010. The impact of extreme turbidity events on the nursery function of a temperate european estuary with regulated freshwater inflow. Estuar. Coast. Shelf Sci. 87, 311–324. https://doi.org/10.1016/j.ecss.2010.01.013.

González-Ortegón, E., Baldó, F., Arias, A., Cuesta, J.A., Fernández-Delgado, C., Vilas, C., Drake, P., 2014. Freshwater scarcity effects on the aquatic macrofauna of a European Mediterranean-climate estuary. Sci. Total Environ. 9, 503–504. https://doi.org/10.1016/j.scitotenv.2014.06.020.

González-Ortegón, E., Baldó, F., Arias, A., Cuesta, J.A., Fernández-Delgado, C., Vilas, C., Drake, P., 2015. Freshwater scarcity effects on the aquatic macrofauna of a European Mediterranean-climate estuary. Sci. Total Environ. 9, 503–504. https://doi.org/10.1016/j.scitotenv.2014.06.020.

González-Ortegón, E., Laiz, I., Sánchez-Quiles, D., Cobelo-Garcia, A., Tovar-Sánchez, A., 2019. Trace metal characterization and fluxes from the guadiana, tinto-odiel and Guadalquivir estuaries to the Gulf of Cadiz. Sci. Total Environ. 650, 2454–2466. https://doi.org/10.1016/j.scitotenv.2018.09.290.

Goossens, H., Zwolsman, J.J.G., 1996. An evaluation of the behaviour of pollutants during dredging activities. Terra Aqua 62, 20–28.

Grasshoff, K., Kremling, K., Ehrhardt, M., 2007. Methods of Seawater Analysis. Third, Completely Revised and Extended edition. https://doi.org/10.1002/9783527613984.

Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa, C., Bruno, J.F., Casey, K.S., Ebert, C., Fox, H.E., Fujita, R., Heinemann, D., Lenihan, H.S., Madin, E.M.P., Perry, M.T., Selig, E.R., Spalding, M., Steneck, R., Watson, R., 2008. A global map of human impact on marine ecosystems. Science 319, 948–952. https://doi.org/10.1126/science.1149345 80.

Haynes, D., Johnson, J.E., 2000. Organochlorine, heavy metal and polyaromatic hydrocarbon pollutant concentrations in the great barrier reef (Australia) environment: a review. Mar. Pollut. Bull. 41, 267–278. https://doi.org/10.1016/S0025-326X(00)00134-X.

Hoffmann, E., Dolmer, P., 2000. Effect of closed areas on distribution of fish and epibenthos. ICES J. Mar. Sci. 57, 1310–1314. https://doi.org/10.1006/jmsc.2000.0921.

Jabusch, T., Melwani, A., Ridolfi, K., Connor, M., 2008. Effects of Short-term Water Quality Impacts Due to Dredging and Disposal on Sensitive Fish Species in San Francisco Bay. San Fr. Estuary Inst. US Army Corps Eng. San Fr. Dist. Thomas.

Jezierska, B., Ługowska, K., Witeska, M., 2009. The effects of heavy metals on embryonic development of fish (a review). Fish Physiol. Biochem. 35, 625–640. https://doi.org/10.1007/s10695-008-9284-4.

Jonas, P.J.C., Millward, G.E., 2010. Metals and nutrients in the Severn estuary and Bristol Channel: contemporary inputs and distributions. Mar. Pollut. Bull. 61, 52–67. https://doi.org/10.1016/j.marpolbul.2009.12.013.

Jones, R., Fisher, R., Stark, C., Ridd, P., 2015. Temporal patterns in seawater quality from dredging in tropical environments. PLoS One 10. https://doi.org/10.1371/journal.pone.0137112.

Jones, R., Bessell-Browne, P., Fisher, R., Klonowski, W., Slivkoff, M., 2016. Assessing the impacts of sediments from dredging on corals. Mar. Pollut. Bull. 102, 9–29. https://doi.org/10.1016/j.marpolbul.2015.10.049.

Jones-Lee, A., Lee, G.F., 2005. Water quality aspects of dredged sediment management. Water Encycl https://doi.org/10.1002/047147844x.wq1507.

Kjelland, M.E., Woodley, C.M., Swannack, T.M., Smith, D.L., 2015. A review of the potential effects of suspended sediment on fishes: potential dredging-related physiological, behavioral, and transgenerational implications. Environ. Syst. Dec. 35, 334–350. https://doi.org/10.1007/s10669-015-9557-2.

Lavolpe, M., López Greco, L., Kesselman, D., Rodríguez, E., 2004. Differential toxicity of copper, zinc, and lead during the embryonic development of chasmagnathus granulatus (Brachyura, varunidae). Environ. Toxicol. Chem. 23, 960–967. https://doi.org/10.1897/02-645.

Lenth, R., 2018. emmeans: Estimated Marginal Means, aka Least-Squares Means.

Losada, M.A., Díez-Minguito, M., Reyes-Merlo, M.A., 2017. Tidal-fluvial interaction in the Guadalquivir River estuary: spatial and frequency-dependent response of currents and water levels. J. Geophys. Res. Ocean. 122, 847–865. https://doi.org/10.1002/2016JC012264.Received.

Lotze, H.K., Lenihan, H.S., Bourque, B.J., Bradbury, R.H., Cooke, R.G., Kay, M.C., Kidwell, S.M., Kirby, M.X., Peterson, C.H., Jackson, J.B.C., 2006. Depletion degradation, and recovery potential of estuaries and coastal seas. 10.1126/science.1128035Science 80 (312), 1806–1809. https://doi.org/10.1126/science.1128035.

Maddock, J.E.L., Carvalho, M.F., Santelli, R.E., Machado, W., 2007. Contaminant metal behaviour during re-suspension of sulphidic estuarine sediments. Water Air Soil Pollut. 181, 193–200. https://doi.org/10.1007/s11270-006-9290-z.

Magris, R.A., Ban, N.C., 2019. A meta-analysis reveals global patterns of sediment effects on marine biodiversity. Glob. Ecol. Biogeogr. 28, 1879–1898. https://doi.org/10.1111/geb.12990.

Mainstone, C.P., Parr, W., 2002. Phosphorus in rivers - ecology and management. Sci. Total Environ. 282–283, 25–47. https://doi.org/10.1016/S0048-9697(01)00937-8.

Martin, T.R., Holdich, D.M., 1986. The acute lethal toxicity of heavy metals to peracarid crustaceans (with particular reference to fresh-water asellids and gammarids). Water Res. 20, 1137–1147. https://doi.org/10.1016/0043-1354(86)90060-6.

Mendiguchía, C., Moreno, C., García, M., 2007. Evaluation of natural and anthropogenic influences on the Guadalquivir River (Spain) by dissolved heavy metals and nutrients. Chemosphere 69, 1509–1517. https://doi.org/10.1016/j.chemosphere.2007.05.082.

Miró, J.M., Megina, C., Donázar-Aramendía, Í., Reyes-Martínez, M.J., Sánchez-Moyano, E., García-Gómez, J.C., 2020. Environmental factors affecting the nursery function for fish in the main estuaries of the Gulf of Cadiz (south-West Iberian Peninsula). Sci. Total Environ. 737, 139614. https://doi.org/10.1016/j.scitotenv.2020.139614.

Ohata, R., Masuda, R., Ueno, M., Fukunishi, Y., Yamashita, Y., 2011. Effects of turbidity on survival of larval ayu and red sea bream exposed to predation by jack mackerel and moon jellyfish. Fish. Sci. 77, 207–215. https://doi.org/10.1007/s12562-010-0320-9.

Ponti, M., Pasteris, A., Guerra, R., Abbiati, M., 2009. Impacts of maintenance channel dredging in a northern adriatic coastal lagoon. II: effects on macrobenthic assemblages in channels and ponds. Estuar. Coast. Shelf Sci. 85, 143–150. https://doi.org/10.1016/j.ecss.2009.06.027.

Rainbow, P.S., 2007. Trace metal bioaccumulation: models, metabolic availability and toxicity. Environ. Int. 33, 576–582. https://doi.org/10.1016/j.envint.2006.05.007.

Rehitha, T.V., Ullas, N., Vineetha, G., Benny, P.Y., Madhu, N.V., Revichandran, C., 2017. Impact of maintenance dredging on macrobenthic community structure of a tropical estuary. Ocean Coast. Manag. 144, 71–82. https://doi.org/10.1016/j.ocecoaman.2017.04.020.

Reine, K., Clarke, D., 1998. Entrainment by hydraulic dredges – a review of potential impacts. DOER Tech. Note Collect. 1–14.

Reine, K.J., Dickerson, D.D., Clarke, D.G., 1998. Environmental win- dows associated with dredging operations. U.S. Army Corps Eng. Eng. Res. Dev. Center, Vicksburg, MS, Tech. Note DOER-E1, pp. 1–14.

Reine, K.J., Clarke, D., Dickerson, C., 2014. Characterization of underwater sounds produced by hydraulic and mechanical dredging operations. J. Acoust. Soc. Am. 135, 3280–3294. https://doi.org/10.1121/1.4875712.

Riba, I., DelValls, T.A., Forja, J.M., Gómez-Parra, A., 2002. Influence of the Aznalcóllar mining spill on the vertical distribution of heavy metals in sediments from the Guadalquivir estuary (SW Spain). Mar. Pollut. Bull. 44, 39–47. https://doi.org/10.1016/S0025-326X(01)00171-0.

Roberts, D.A., 2012. Causes and ecological effects of resuspended contaminated sediments (RCS) in marine environments. Environ. Int. 40, 230–243. https://doi.org/10.1016/j.envint.2011.11.013.

Ruiz, J., Polo, M.J., Díez-Minguito, M., Navarro, G., Morris, E.P., Huertas, E., Caballero, I., Contreras, E., Losada, M.A., 2015. The Guadalquivir estuary: a hot spot for environmental and human conflicts. Environmental Management and Governance. Coastal Research Library, pp. 199–232 https://doi.org/10.1007/978-3-319-06305-8.

Simonini, R., Ansaloni, I., Cavallini, F., Graziosi, F., Iotti, M., Massamba N'Siala, G., Mauri, M., Montanari, G., Preti, M., Prevedelli, D., 2005. Effects of long-term dumping of harbor-dredged material on macrozoobenthos at four disposal sites along the Emilia-Romagna coast (Northern Adriatic Sea, Italy). Mar. Pollut. Bull. 50, 1595–1605. https://doi.org/10.1016/j.marpolbul.2005.06.031.

R Core Team, 2018. A Language and Environment for Statistical Computing. R Foundation for Statistical Computing. https://www.R-project.org/.

Tornero, V., Arias, A.M., Blasco, J., 2011. Following the Aznalcóllar toxic spill. Environ. Res. 111, 1033–1036. https://doi.org/10.1016/j.envres.2011.07.011.

Tornero, V., Arias, A.M., Blasco, J., 2014. Trace element contamination in the Guadalquivir River estuary ten years after the Aznalcóllar mine spill. Mar. Pollut. Bull. 86, 349–360. https://doi.org/10.1016/j.marpolbul.2014.06.044.

Torres, R.J., Abessa, D.M.S., Santos, F.C., Maranho, L.A., Davanso, M.B., do Nascimento, M.R.L., Mozeto, A.A., 2009. Effects of dredging operations on sediment quality: Contaminant mobilization in dredged sediments from the Port of Santos, SP, Brazil. J. Soils Sediments 9, 420–432. https://doi.org/10.1007/s11368-009-0121-x.

Tsolaki, E., Diamadopoulos, E., 2010. Technologies for ballast water treatment: a review. J. Chem. Technol. Biotechnol. 85, 19–32. https://doi.org/10.1002/jctb.2276.

Van Den Berg, G.A., Meijers, G.G.A., Van Der Heijdt, L.M., Zwolsman, J.J.G., 2001. Dredging-related mobilisation of trace metals: a case study in the Netherlands. Water Res. 35, 1979–1986. https://doi.org/10.1016/S0043-1354(00)00452-8.

Wenger, A.S., Harvey, E., Wilson, S., Rawson, C., Newman, S.J., Clarke, D., Saunders, B.J., Browne, N., Travers, M.J., Mcilwain, J.L., Erftemeijer, P.L.A., Hobbs, J.P.A., Mclean, D., Depczynski, M., Evans, R.D., 2017. A critical analysis of the direct effects of dredging on fish. Fish Fish. 1–19. https://doi.org/10.1111/faf.12218.

Wenger, A.S., Rawson, C.A., Wilson, S., Newman, S.J., Travers, M.J., Atkinson, S., Browne, N., Clarke, D., Depczynski, M., Erftemeijer, P.L.A., Evans, R.D., Hobbs, J.P.A., McIlwain, J.L., McLean, D.L., Saunders, B.J., Harvey, E., 2018. Management strategies to minimize the dredging impacts of coastal development on fish and fisheries. Conserv. Lett. 11, 1–10. https://doi.org/10.1111/conl.12572.

Wilber, D.H., Clarke, D.G., 2001. Biological effects of suspended sediments: a review of suspended sediment impacts on fish and shellfish with relation to dredging activities in estuaries. North Am. J. Fish. Manag. 21, 855–875. https://doi.org/10.1577/1548-8675(2001)021<0855:beossa>2.0.co;2.

Winterwerp, J.C., Wang, Z.B., 2013. Man-induced regime shifts in small estuaries - I: theory. Ocean Dyn. 63, 1279–1292. https://doi.org/10.1007/s10236-013-0662-9.

Yap, W.Y., Lam, J.S.L., 2013. 80 million-twenty-foot-equivalent-unit container port? Sustainability issues in port and coastal development. Ocean Coast. Manag. 71, 13–25. https://doi.org/10.1016/j.ocecoaman.2012.10.011.