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Río de la Plata native *Erodona mactroides* (Bivalvia, Erodonidae) and exotic bivalves respond differently to sediment supply

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Abstract. The Río de la Plata estuary has low diversity and abundance of zoobenthos due to its high turbidity and environmental variability. To determine the influence of sediment dredging on the composition, abundance and size of macroinvertebrates, we studied a beach in the north coast of the estuary, where a trench and a pipe were built underwater. The water variables and the sediment grain size did not differ between before and during dredging. *Erodona mactroides* was the dominant species, followed by the aliens *Corbicula fluminea* and *Limnoperna fortunei* and by the native *Nephtys fluviatilis*. The density of *E. mactroides* was higher and the height of the shell was lower before than during dredging. Invaders increased during dredging, when *N. fluviatilis* decreased substantially. Dredging can affect the benthos and the entire coastal ecosystem through the contribution of thicker sediments and *E. mactroides* can be a sentinel species of the coastal environment.

Key words: Estuary, dredging, exotic species *Corbicula fluminea, Limnoperna fortunei,* Macrozoobenthos.

Resumen: **La almeja nativa *Erodona mactroides* (Bivalvia, Erodonidae) y los bivalvos exóticos del estuario del Río de la Plata responden distinto ante el ingreso de sedimentos.** El Río de la Plata tiene una baja diversidad y abundancia de zoobentos debido a su alta turbidez y variabilidad ambiental. Para determinar la influencia del dragado de sedimentos en la composición, la abundancia y el tamaño de los macroinvertebrados, estudiamos una playa en la costa norte del estuario, donde se construyeron una zanja y una tubería subacuáticas. Las variables del agua y el tamaño de grano del sedimento no difirieron entre antes y durante el dragado. *Erodona mactroides* fue la especie dominante, seguida por las invasoras *Corbicula fluminea* y *Limnoperna fortunei* y por el poliqueto nativo *Nephtys fluviatilis*. La densidad de *E. mactroides* fue mayor y la altura de la concha fue menor antes que durante el dragado. Los invasores aumentaron durante el dragado, cuando *N. fluviatilis* casi desapareció. El dragado puede afectar el bentos y todo el ecosistema costero a través de la contribución de sedimentos más gruesos. *Erodona mactroides* puede ser una especie centinela del medio ambiente costero.

Palabras clave: Estuario, dragado, especies exóticas, *Corbicula fluminea, Limnoperna fortunei,* Macrozoobentos.

Introduction

The Río de la Plata (Southwestern Atlantic, 35º S, 57º W) estuary has low tidal amplitude and it is strongly dominated by wind, due to its great extension and shallow depth (Guerrero *et al*. 1997, Mianzan *et al*. 2001, Cortelezzi *et al*. 2007). The benthic fauna is less diverse and abundant there than on the oceanic coast. This can be due to the high variability of salinity and high anthropogenic disturbance urbanization, harbor, shipping and industrial activities as the main perturbation factors for the Río de la Plata (Brugnoli *et al* 2007)

This estuary presents important coastal artisanal fisheries (Cortelezzi *et al*. 2007, Acha *et al*. 2008), as well as several cities, ports and spas (Fig. 1). In consequence the north coast has several types of pollution (Moyano *et al*. 1993, Kurucz *et al*. 1998, Danulat *et al*. 2002, Muniz *et al*. 2002, 2004, Nagy *et al*. 2003, Simionato *et al.* 2004, Burone *et al*. 2006, Muniz *et al*. 2011, Muniz *et al.* 2019). The zoobenthos of the Río de la Plata is composed by few euryhaline species tolerant to suspended sediments (Masello & Menafra 1998). One of the most abundant bivalves of small estuaries flowing to the Río de la Plata is the subtidal clam *Erodona mactroides* Bosc, 1802 (Scarabino *et al*. 2006), also frequent in the mixohaline coasts of the southwestern Atlantic (Scarabino *et al*. 1975, Muniz & Venturini 2001, Geraldi 2002), where it is endemic (Bemvenuti & Rosa Filho 2000).

The main impact of dredging on the zoobenthos is direct destruction (Morton 1977), but other effects include the enrichment of the surrounding area by sedimentation of the resuspended material (Poiner & Kennedy 1984), and the structure of the benthic community change from one of coarse sands to one of fine sands (Van Dalfsen *et al*. 2000). *Erodona mactroides* plays an important ecological role as the main consumer of phytoplankton and food of decapods and fish (Bemvenuti & Netto, 1998).

This bivalve is normally found in sandy sediments with a large amount of organic matter (Masello & Menafra 1998). It is present along the Río de la Plata (Calliari et al 2003, Darrigran & Lagreca 2005; Scarabino et al 2006), where it can be indicative of anthropogenic alteration (Masello & Menafra 1998) so it could be a good sentinel organism.

The construction of underwater structures in the north coast of the intermediate Río de la Plata produces the sediments movement that may have negative effects on the benthic community and therefore on the entire ecosystem. Since these effects are not well known until now, it is important to develop studies that can help to understand the processes involved.

The objective of this work was to determine the influence of sediment dredging on the composition of benthic organisms, mainly on the abundance of the *E. mactroides*, a native bivalve very abundant in the study area as observed during pilot sampling.

Material and methods

Study area: A submerged bank divides the Río de la Plata into a shallow interior fluvial system (<5 m) and a brackish exterior one (Mianzan *et al.* 2001). The fluvial zone is divided into an internal freshwater zone and a brackish intermediate zone (Darrigran & Lagreca 2005) (Fig. 1). The intermediate zone has a variable salinity gradient due to changes in the flow of its main tributaries, the Paraná and Uruguay rivers, and the prevailing winds, and a turbidity front that separates freshwater from estuarine waters (Cortelezzi *et al*. 2007, Nagy *et al.* 2013).

On the north coast near the outer boundary of the intermediate zone, at the so called Punta del Tigre beach (34 ° 45' S; 56 ° 32'W), a breakwater was built and a trench was dug for the installation of an underwater pipeline to take water from the estuary for refrigeration of an industry and then, to return it to the estuary. The site is at the western end of a 12 km long beach, which stretches between the mouth of the Santa Lucia River and Punta del Tigre, west of Montevideo city. There the sediments are silty-clay and the tidal oscillation of only 40 cm (Nagy *et al.* 1998).

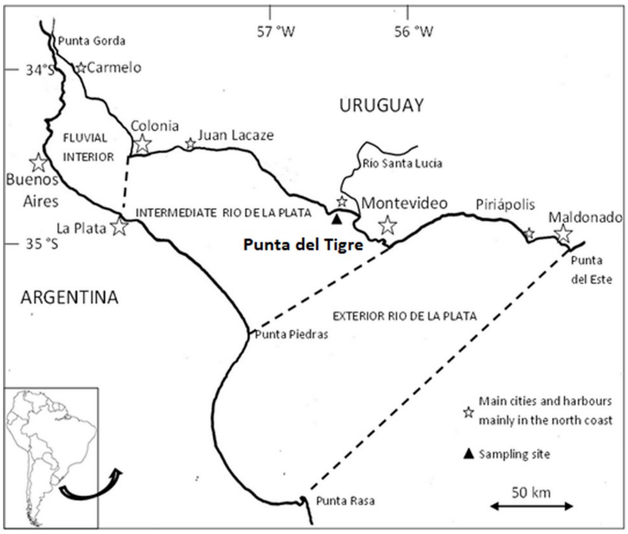

Figure 1. Río de la Plata with its interior, intermediate and exterior zones, main cities and ports on the North coast and sampling site at Punta del Tigre. Modified from Wells & Daborn 1997.

Sampling: The sampling sites were chosen considering the isobaths and the dominant currents, both parallel to the coast, and the location of the underwater outfall. The sampling sites were: B, C and E approximately 700 m from the coast and D at 1000 m from the coast (Fig.2). At each site, three sediment replicates were taken with a 625 cm^2 Ekman dredge to analyze the community of zoobenthos, approximately every month and a half between February 2015 and June 2017. Foreseeing the possible impact of dredging on the epibenthic fauna it was decided to take an additional sample for

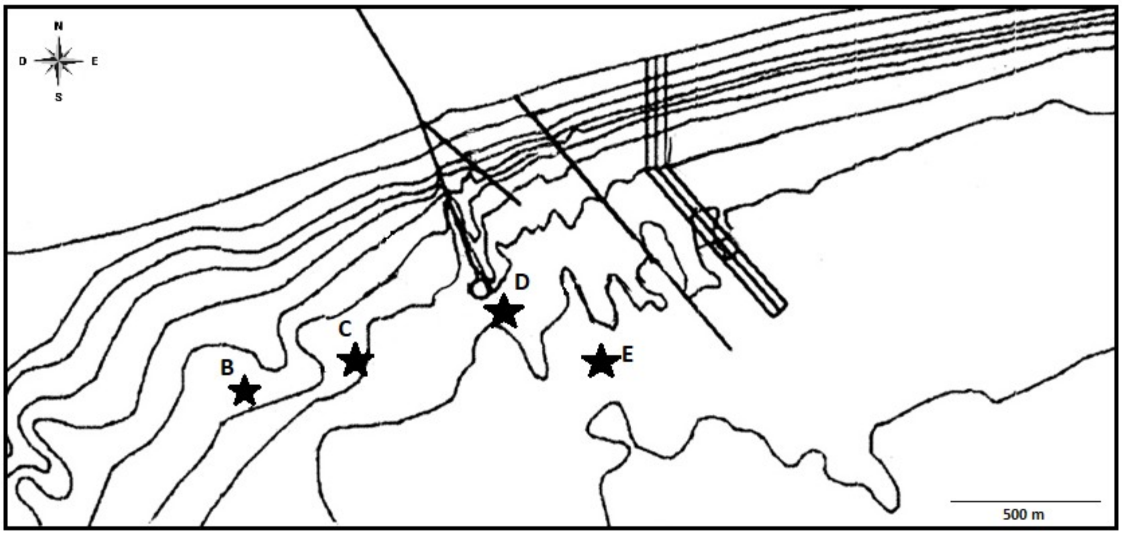

Figure 2. Study area on the north coast of the Río de la Plata in Punta Tigre, with isobaths every 0.5 m and sampling sites B, C, D and E (stars). The location of the pipeline where the sediment dredging and the temporary breakwater was dug are shown.

granulometry since October 2015 during the last eleven samples.

These samplings included a previous period and another one during the sediment dredging works, which took place since March 2016. The trench excavated to install the underwater pipe was *ca*. 1500 m long by 20 - 50 m wide and 2 - 12 m deep, removing between 100,000 and 600,000 m^3 of sediment in approximately one year.

In situ measurements of depth with Fish-Finder portable echo sounder in all sites, pH and water conductivity with Oakton® sensor, transparency with Secchi disk of 20 cm, dissolved oxygen and temperature with Horiba® D25 oxymeter were made at all sites. Irradiance profiles with Li Cor 1500 radiometer every 20 cm deep were made only at site C.

Laboratory: The sediment classification was made by sieving and gravimetry, according to the Krumbein scale and the Wentworth nomenclature (Arocena 1999). The fractions of gravel, sand (very coarse, coarse and fine, later summed) and silt were separated and weighed. For the macrofaunal analyses the three grab replicates of collected sediment were sieved *in situ* by a mesh of 0.5 mm opening. The retained organisms were separated under a stereoscopic microscope, identified with specific dichotomous keys and then counted. The length, width and height of dominant *Erodona mactroides* were measured with digital caliper (0.1 mm). Although these morphometric data did not have a normal distribution (Shapiro-Wilks, p <0.05), they were very close to it, so their allometric relationships were calculated.

The statistical analyzes were performed with the STATISTICA® 8.0 package. First, it was tested whether the biotic and abiotic data had a normal distribution by means of the Shapiro-Wilks test, and the homogeneity of variance by means of the Levene test. Density data of zoobenthos were transformed (log+1). The Kruskal-Wallis analysis was carried out to compare the biotic and abiotic variables between sites and months, except the depth that was compared by means of ANOVA and Bonferroni test. The Mann Whitney test was used to verify differences between before and after the dredging in the same variables and Spearman correlation analysis was made between them. The Simpson dominance and Shannon diversity indices were determined in the zoobenthos using the PAST software version 2.17c (Hammer *et al*. 2001).

Results

Abiotic environment: The water depth (2.7 - 4.8 m, Fig. 3) varied between dates and sampling sites (ANOVA p<0.001). Sites B and C showed depths similar to each other and significantly different than D and E (Bonferroni, p <0.001). Oxygen always recorded values > 7 mg / L and close to saturation, with no spatial differences in depth (Wilcoxon Matched Pairs Test p >0.05) and sites (Friedman ANOVA p = .40829). The conductivity and transparency of the Secchi disk were similar between sites (Friedman ANOVA and ANOVA, p>0.05) but different between the dates (p<0.001).

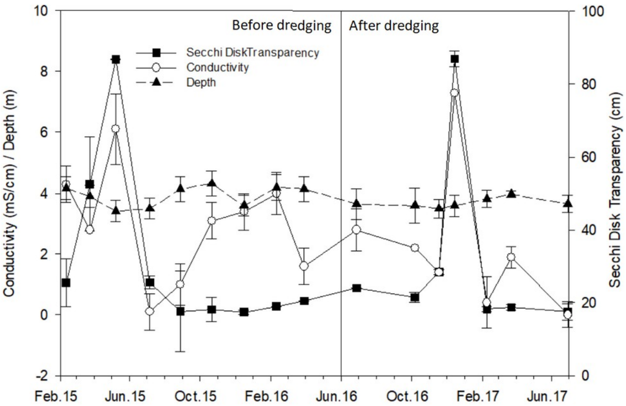

Figure 3. Variation of depth, conductivity and transparency of Secchi disk throughout the study period. Average of the four sampling sites and standard deviation. The vertical line separates the previous period from the after dredging one.

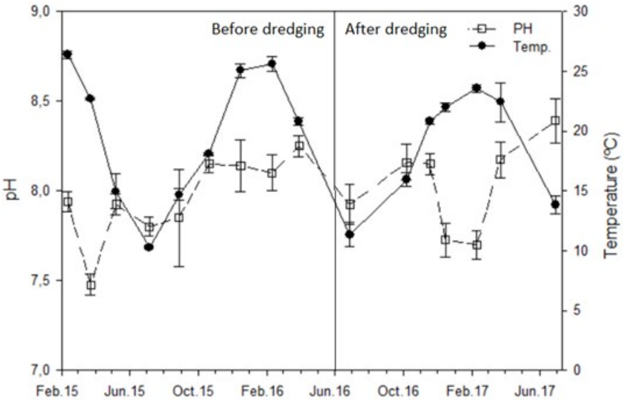

Figure 4. Variation of pH and temperature throughout the study period, average of the four sites and standard deviation. The vertical line separates the previous period from the after dredging one.

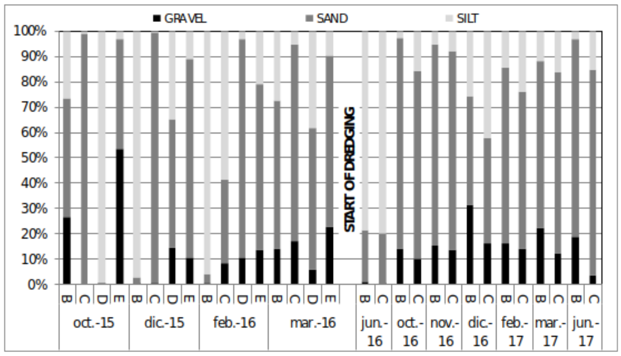

Figure 5. Granulometry in sites B and C during the entire study period, and in D and E from October 2015 to March 2016.

Conductivity was lower than 1.5 mS / cm except in May 2015 and December 2016, when it exceeded 8.0 mS / cm (Fig. 3). Secchi disk transparency showed two maxima on the same dates (67 and 77 cm, respectively) while for the rest of the period it did not exceeded 52 cm. The irradiance profiles showed an average light extinction coefficient of 4.18 ± 2.42 m^{-1} and the euphotic zone varied between 0.4 and 3.4 m depth.

The pH was slightly alkaline and the water temperature varied between 9.2 and 26.5 ºC, showing a clear seasonality (Fig. 4). The temperature in winter was always lower than 14 ºC and in summer higher than 22 ºC. The depth correlated positively with the temperature and this one with the transparency, which correlated positively with the conductivity (Spearman, p<0.05). These variables, except the depth (Student t, p <0.001), did not show significant differences between the stages before and during dredging (Mann Whitney, p> 0.1). The coefficient of extinction of the light and the depth of the euphotic zone in site C also showed no significant differences before and during dredging (Mann Whitney, p> 0.3).

Sediment granulometry presented high variability both between sites and over time (CV = 69-102%) (Fig. 5), varying between silt and sandy gravel, with a higher frequency of sand. The silt prevailed (> 75%) in October and December 2015, February and June 2016 in different places. However, none of the fractions differed significantly between sampling dates (Kruskal-Wallis, p> 0.1), nor between sites B, C, D and E from the beginning until March 2016 (Kruskal-Wallis, p> 0.1) or between B and C from June 2016 to the end (Mann-Whitney, p> 0.05). There were also no differences between before and after the start of dredging (Mann-Whitney p> 0.1) in any fraction in sites B and C, except in coarse sand (p = 0.0049), whose percentage increased in the second period. This difference was only in B (p = 0.0242) where the silt content also decreased.

Zoobenthos: A total of 2093 organisms were collected, being *Erodona mactroides* the dominant species (average density: 182 ± 126 ind m^{-2}), followed by the invasive bivalve *Corbicula fluminea* with 4 ± 6 ind m^{-2}, the native polychaete *Nephtys fluviatilis* (Monro, 1837) with 2 ± 4 ind m^{-2} and another alien bivalve, *Limnoperna fortunei* with 1 ± 2 ind m^{-2}. The maximum density of *E. mactroides* was recorded in site E (1488 ind m^{-2}, May 2015). The average monthly density of *E. mactroides* -by site or as a whole- did not correlate with any abiotic variable of the water column excepting negatively with pH (in site B). Lack of correlation was also observed for sediment excepting negatively with very coarse sand, but also only in site B (Spearman p

< 0.05). The highest densities occurred in the three austral summers of each year (February 2015, 2016 and 2017) and in the autumn 2015 (May-Jun) (Fig. 6).

Significant differences were found in the density of *E. mactroides* between the period before and the period after dredging for sampling sites B and C (Mann-Whitney U Test, $p = 0.0228$), being higher in the previous period (172 ± 246 vs. 98 ± 163 ind / m^2).

The average dimensions of *E. mactroides* were (length x width x height): 23.6 x 13.5 x 9.3 mm and the maximum 35.0 x 20.3 x 18.9. The allometric relationships of all individuals collected were:

Length = 1.989 + 1.605 Width (r = 0.8754)

Length = 4.534 + 2.043 Height (r = 0.8730)

Width = 2.647 + 1.159 Height (r = 0.8754)

In the period before dredging, the average height of the shell was 9.4 ± 1.4 mm, significantly lower than in the period after dredging, when it was 9.9 ± 1.7 mm (Mann Whithney U, $p < 0.001$). Significant differences of the three dimensions were obtained between the sites ($p < 0.001$), but only in E the organisms were smaller (Kruskal Wallis, $p < 0.001$) than in B and C - equal to each other except for the height that was lower in B than in C (Mann Whitney, $p = 0.002$).

The length of the shell of *E. mactroides* was not correlated with sediment fractions ($R^2 = 0.0652$) or with density of organisms ($R^2 = 0.0731$), although in the most abundant sample the average size was among the smallest (22.5 x 12.8 x 8.9 mm). No temporal or spatial trends were observed in the height or total abundance.

Prior of dredging there were just two taxa: *N. fluviatilis* and the highly dominant *E. mactroides* (Simpson 0.97; Equitability 0.06 and Shannon diversity 0.04). After dredging there was an increase to four taxa (Simpson 0.87, Equitability 0.28 and Shannon diversity 0.27). An analysis of similitude (ANOSIM) showed significant differences between both periods (p=0.017).

Discussion

In the present work we found that the density of *Erodona mactroides* in the north coast of the intermediate Río de la Plata decreased after the beginning of the dredging of sediment for the laying of a pipe 1500 x 30 x 10 m. There was also an increase in invasive bivalves (*L. fortunei* and *C. fluminea*) during dredging. This agrees with Van Dalfsen & Essink (2001) for whom the responses of

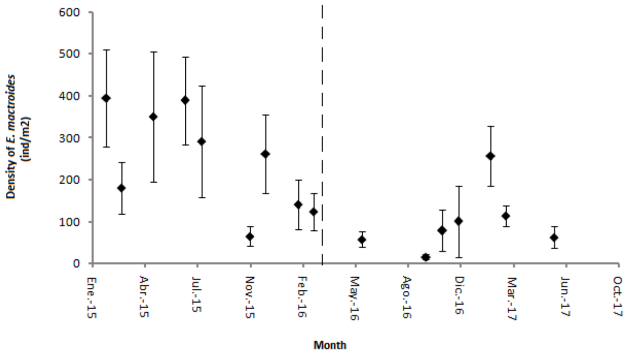

Figure 6. Average density and standard deviation of *Erodona mactroides* throughout the study period, in sites B, C, D and E. The vertical line indicates the start of dredging.

the benthic community to the extraction of sand consist of a rapid development of invasive species and a slow recovery of the structure of the original community. However, the re-colonization of the area is a process dependent on local hydrodynamic and sedimentary conditions (Morton 1977, Van der Veer *et al*. 1985). The density of *E. mactroides* did not correlate with the type of sediment, but there is a clear decrease in density after dredging, coinciding with the increase of fine sediment. The decrease of the native species *E. mactroides* and the increase of the invasive species *C. fluminea* may be indicating a substitution of species of the same functional group, as explained by Fonseca & Ganade (2001). It is unknown at what stage of species substitution we are and what their ecological effects are. Consequently, it is important to keep a watch on this process. These invasive species are filter feeders of phytoplankton like the native species, which can be displaced by them. Invasive species are also a natural food for croaker (*Micropogonias furnieri*) and other fish of commercial importance in the area (Mendoza-Carranza & Vieira 2008, Lercari & Bergamino 2011), so the displacement of *E. mactroides* can also negatively affect these predators.

The positive correlation between depth and temperature indicates a seasonal behavior of the depth, opposite to the fluvial flows, which are generally higher in winter. However, this scheme can be modified by inter-annual variations due mainly to disturbances such as El Niño, which occurred between October 2015 and June 2016, coinciding with greater depths. This event was followed by La Niña between July 2016 and January 2017, when lower depths were recorded than in the previous and subsequent periods. Correlations between depth, temperature, transparency and water conductivity show a certain seasonality of the hydrological regime, controlled by the interaction between seasonal fluvial flows, daily winds, and the more spaced and irregular interannual perturbations.

About thirteen species of marine and estuarine bivalves inhabit this area, of which only four are estuarines, and only 2 species are abundant in unconsolidated substrates (Scarabino *et al.* 2006). The only 4 species (3 bivalves, 1 polychaeta) found in the macrozoobenthos in this area and the almost absolute domain of *E. mactroides* agrees with the high variability of salinity and currents of the studied ecosystem, added to the continuous disturbance of estuarine coasts. However, other areas of the same estuary of the Río de la Plata show a greater benthic diversity (Wells & Daborn 1997). Therefore, it must be another reason related to the intermediate coastal zone like high hydrodinamics together with transitional conditions, and meeting of different water masses. There are few benthic organisms that have wide tolerance to survive and reproduce successfully in such conditions (Masello & Menafra 1998, Venturini *et al.* 2004). This is reflected in the low diversity found in the Rio de la Plata, where molluscs constitute close to 90% of the total biomass of the zoobenthos (Cortelezzi *et al*. 2007). *Erodona mactroides* has been recorded also in sandy and sandy-silt bottoms of brackish- and fresh-waters of the southern intermediate zone (César *et al* 2000), together with the invasive bivalves *Corbicula fluminea* and *Limnoperna fortunei*, present in the Plata basin since the beginning of the 70s (Ituarte 1981) and since 1991 (Brugnoli *et al*. 2011), respectively.

The specific composition found in this study, as well as the low richness and diversity, is concordant with that of similar ecosystems in the region (Scarabino *et al.* 1975, Darrigran 1994, Mianzan *et al.* 2001, Cortelezzi *et al.* 2007, Viana 2009). However, Shannon's diversity was lower than those determined by Ieno & Bastida (1998), Mianzan *et al*. (2001) and Cortelezzi *et al*. (2007) for the intermediate zone of the Río de la Plata, probably due to the greater variability of the study site.

Scarabino *et al*. (1975) distinguished between freshwater species such as *Corbicula* spp. and estuarine species such as *E. mactroides*. In this work we found both species together, although very few individuals of the first one. On the other hand, *N. fluviatilis* and *E. mactroides* are dominant species of the zoobenthos in mouths of rivers and in coastal lagoons of Uruguay (Muniz & Venturini 2001, Viana 2009). However, *N. fluviatilis* was present in very low densities in the study area, perhaps because it is typical of low energy environments. It was only found in Punta del Tigre in very low abundance in two summers and one autumn, coinciding with its reproductive periods (Muniz & Venturini 2001).

Although the average density of *E. mactroides* was much higher than that found by Cortelezzi *et al.* (2007), these authors report a higher number of unidentified juvenile bivalves. In the present work, the variation of the abundance of *E. mactroides* is not explained by the granulometry or by the abiotic variables of the water column. On the contrary, Colling *et al.* (2010) found negative relationships over time in Laguna de los Patos between the abundance of *E. mactroides* and fine sediments and the water level, and direct with salinity, which they attribute to a high rainfall caused by El Niño. The fine sediments thus contributed would be the true cause of the decrease of the bivalve, and not the salinity. As Jorcin (1996) maintains, the salinity would not interfere with the distribution of a species adapted to colonize mixohaline environments. Therefore, the alterations of bottom sediments due to human interventions such as dredging can have negative effects on this benthic species and indirectly on the entire coastal ecosystem through the feeding links of *E. mactroides* mainly with predatory fishes.

Although there was an increase in the abundance of the invasive species and a decrease in polychaetes and *E. mactroides* during the dredging, no other temporal variations were found. On the contrary, in Laguna de los Patos, *E. mactroides* showed a significant temporal variation over 2 years (Colling *et al.* 2010), correlated with temperature, salinity and sediments, as in other temperate regions from USA, Chile and Argentina (Rosa & Bemvenuti 2006). There are important spatial variations of *E. mactroides* density which make necessary a sampling design with many sampling sites to assess temporal changes.

The average shell length of *E. mactroides* (24 mm) was lower than those recorded in Laguna de Rocha (43 mm, Jorcin 1996), Laguna de los Patos (35 mm, Geraldi 2002) and Arroyo Pando (35 mm, Passadore *et al.* 2007). This may be due to the fact that the coast of the Río de la Plata is a more dynamic and unstable environment than the coastal lagoons and small estuaries that flow into it. In Punta del Tigre, individuals less than 20 mm were not found, sizes that are considered juveniles in other environments (Geraldi 2002). Apparently, the high hydrodynamics of the study site would prevent the recruitment of juveniles near the coast, those of which are in deeper waters. The larger size of *E. mactroides* individuals collected during dredging can be related to the increase in grain size (coarse sand). Giménez *et al.* (2014) already showed that the density of this bivalve in Uruguay increases in sites dominated by coarse sand, so it is possible that its size also increases in such conditions.

In the present work it was determined that the specific richness of macrozoobenthos in Punta del Tigre is even lower than that of other nearby subtidal bottoms, although the composition is similar (Muniz *et al.* 2002, 2011, Brugnoli *et al.* 2007). The density of the population and the body size of *E. mactroides* were also found to be lower than those registered in nearby places, presumably due to the high hydrodynamics of the area, caused by factors acting at different scales from hourly to interannual such as winds or El Niño events, respectively. Although no relationship was found between the temporal variations of these parameters and the environmental factors, it is likely that a negative effect of the contribution of fine sediments is hidden by the high variability of the system. This contribution can come from the motion of sediment during dredging, as well as from fluvial currents, dominant in autumn-winter or during El Niño events. Dredging or other anthropic impacts or even natural phenomena such as El Niño, which affect the granulometry of the sediment, can result in the loss of abundance of some native species. These can then be quickly replaced by invasive species that compete for the same food. Consequently, anthropic activities in coastal areas, where this species grows, must take these processes into account.

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