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Short term impact of artisanal dredges in a Patagonian mussel fishery: Comparisons with commercial diving and control sites

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A B S T R A C T

Mussels in the San Matías Gulf fishery are targeted using artisanal dredges and diving. The main objective of this study was to assess the direct impact of artisanal dredging on the biota and sediments, and to compare the composition of the catches and the individual damage induced by fishing between dredging and commercial diving. The experimental design included samplings from dredge catches, dredge tracks, control sites and commercial diving. According to their damage level, individuals were scored as undamaged, lightly damaged and severely damaged. Sediment characteristics were analyzed using coring samples and traps. Damage of mussels, mostly corresponding to the severely damaged category, was less than 5% both in samples from dredging and diving. Conversely, mean damage of the main bycatch species (sea urchins and ophiuroids) was 75 and 65% in samples from dredging and diving respectively, being most of the individuals lightly damaged. Considering also the catch sample composition of both fishing methods, dredging affected relatively more individuals than diving. Although sediment removal in dredged areas was three times higher than that in non-dredged ones, mean grain size and gravel percentage of sea floor sediments showed subtle differences between them.

1. Introduction

The impact caused by dredging or trawling operations has been the subject of intense research for many years (Caddy, 1973; Eleftheriou and Robertson, 1992; Thrush et al., 1995; Kaiser et al., 1998; Gaspar et al., 1999, 2002; Hall-Spencer et al., 1999; Currie and Parry, 1996; Bergman and van Santbrink, 2000; Dolmer and Frandsen, 2002; Løkkeberg, 2005; Gray et al., 2006). The effects described can be either direct, which include the scraping and furrowing of substrates, sediment re-suspension, destruction of the habitat structure and the spillover in the same process (Jones, 1992; Auster, 1998), or indirect, which include the capture of a range of unwanted species that are discarded into the sea, post-fishing mortality and long-term habitat changes, all induced by the fishing activity itself. These two kinds of impacts could occur combined in different ways and at different spatial–temporal scales (Jones, 1992; Jennings and Kaiser, 1998).

The structural engineering of the complex benthic habitats not only offers security and refuge to juvenile fishes and other benthic organisms, but also supplies food to epifauna, and constitutes an important food source for demersal fishes (Kaiser et al., 2005; Galván et al., 2009). Bottom sediments play important roles in the processes of transformation and exchange of organic matter and nutrients (Thrush and Dayton, 2002), whereas the sediment–water interface of marine sediments is also an important site of benthic primary production (Jones, 1992; Jones et al., 1997; Jennings et al., 2001; Løkkeberg, 2005; Gray et al., 2006).

In San Matías Gulf (SMG), Argentina (Fig. 1) dredging has been the main method used for fishing the bivalve molluscs *Aequipecten tehuelchus*, *Aulacomya atra* and *Mytilus edulis platensis*. The traditional dredge used to be of the industrial type, constructed completely of iron, with a width of 2.5 m, and a weight of over 300 kg. The supposed impact on species survival and the environment (Orensanz et al., 1991; Ciocco et al., 2006) generated controversy between scientists and fishermen about the viability of its use, and the consequent concern thus called for changes in the

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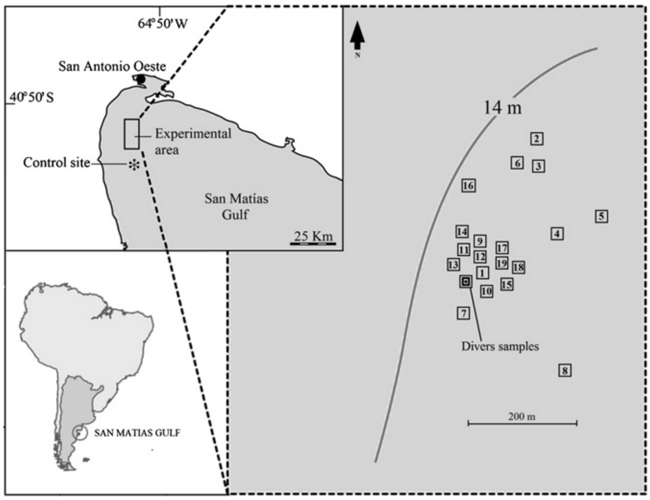

Fig. 1. Map showing the sampling area with details of the experimental fishing hauls and site of the commercial diving samples. The control site was located 7 km south of the fishing area.

fishing gear design. As from 2000, dredges have a maximum mouth width of 1.6 m, a maximum weight of 50 kg outside the water and a belly entirely made of netting. Under the new operational artisanal scheme, bivalve landings totaled 7451 tons in the 2000–2008 period. Nearly 50% of this amount was caught by boats (maximum 9.9 m length) equipped with the artisanal dredge. The efficiency of this dredge (89% for mussels, which represented nearly 80% of the dredge catches; Narvarte et al., 2011) was high compared to the industrial dredges previously used and those used worldwide (Meyer et al., 1991; McLoughlin et al., 1991; Gaspar et al., 2003; Pezzuto et al., 2010). In spite of the changes made on the fishery regulations and the expected reduction of the impact on the marine ecosystem, objective information about the actual impact of these fishing gears on the bottom structure and on the benthic communities is still missing. The main objective of this study was to assess the direct impacts (individual damage to the mussels) of the artisanal dredge (both on the organisms caught in the belly and those left on the seabed) and compare them with those caused by commercial diving and with control sites. A preliminary analysis of the dredge impact on the sea floor sediment structure in the fishing ground was also performed.

2. Material and methods

2.1. Study site and sampling

The study site was located at the northwest sector of SMG, on a mussel (*M. edulis platensis*) bed located between 14 and 20 m depth at El Sótano (40°55′S-40°57′S/65°05′W-65°07′W; Fig. 1). At this site, the bottom is mainly comprised of clastic igneous rock (basalt and jasper) and mollusc shells joined in a matrix of gravel–sand, sand–silt and silt–clay (Angulo et al., 1978). The sea floor consists of variable granulometric sediments, from very fine sand to gravel, which are mobilised by coastal tidal currents and winds (Aliotta et al., 2000). Benthic communities are established on this bottom, where scallops, mussels and infaunal bivalves become dominant in patches, depending on the bathymetry, bottom type, and population structure (Narvarte et al., 2007).

The experiment was performed on a monocohort coastal mussel bed, in an area where fishing activity had not been recorded since 1994 (Morsan, 2009). The artisanal boat “Nadia Belén”, equipped with a standard artisanal dredge, was used. Nineteen fishing tows, covering an area of 1250 m^2 within a total area of approximately 15 Ha, were performed in May 2007. In each survey, operations were similar to real commercial fishing activities. The dredge was deployed, on locations randomly selected, and two divers simultaneously recorded the starting point of each tow, elevating a buoy straight to the sea surface. At the end point of each tow, another buoy was deployed and the tow distance between both buoys was recorded using a differential precision GPS Trimble®. Each experimental tow lasted on average 5 min at a mean speed of 1.2 kts. The tow length ranged between 32 and 150 m. As dredging was being carried out, two scientific divers swam behind the dredge collecting all the megaepifauna, including mussels, left on the bottom along the dredge track. All the material was retained in a bag, numbered and brought to the lab. In common with during commercial operations, catches of each tow were overturned on the deck and kept in plastic boxes (64 × 42 × 20 cm; capacity 40 kg). The total number of boxes per tow was recorded and a sub-sample from at least two boxes per tow was separated for lab analysis of biota composition. Finally, two other scientific divers recorded video images and took digital photos of the dredge during the tows to evaluate the fishing gear performance.

Effects of the dredge activity on the bottom fauna were distinguished between the effect of the dredge on the direct capture (named “belly”) and the effect of the dredge on the biota/substrates left in the swept area (named “track”).

A control site covering an area of approximately 1 Ha, located 7 km south of the fishing area, was selected before the experiment, considering the absence of fishing activities, and the fact that the faunal composition and depth range were similar to those present in the fishing ground. The sampling of the control site was conducted simultaneously with the fishing experiment to avoid differences in the bottom structure and faunal composition which usually follow to storms in the study area. Samples of epibenthic organisms (including mussels) were obtained by the scientific divers by haphazardly throwing the quadrat on the seabed and bringing the contents of transects to the lab to be analyzed immediately. Five faunal samples were collected daily during the three-day experiment ($n = 15$ replicates) using quadrats (0.25 m^2) in the control site. We were forced to use quadrat sampling against other traditional methods (i.e.: line transects) due to logistical constraints imposed by SCUBA operations (i.e.: daily limits in diving time at the depth range considered). However, we assume that the number of quadrats taken was fairly representative of an unfished mussel bed.

To compare the damage in the samples from the dredge and diving fisheries, four commercial bags (40 kg each) from the mussel catch were randomly obtained from two artisanal diving boats, which were operating simultaneously in the same area as that of the experimental site (Fig. 1). Commercial divers fishing on Patagonian bivalve beds usually select beds with the highest densities and operate on them until the depletion of the densest patches (Orensanz et al., 1991; Ciocco et al., 2006). According to consultations to commercial divers, the covered area to obtain a catch of 40 kg was near 5 m^2.

Once in the lab, all the samples obtained from the belly, track, commercial diving and control site were processed and separated in substrates and biota. The faunal components were identified, counted and weighed. In the case of the dredge treatments (belly and track) data were extrapolated to the total biomass captured per tow.

Similarity of species composition of the bycatch (discarding mussel in the analysis) was determined by a non-metric multidimensional scaling (MDS), using the square root transformed density data from each treatment and the Bray–Curtis similarity

Table 1
Relative contribution of each species (mean % ±SD, of the total number of individuals, and number of samples with presence) in the samples from the dredge belly, dredge track, commercial diving and control site. Ref: *O. magellanica*: *Odontocymbiola magellanica*, *P. magallanicus*: *Pseudoechinus magallanicus*.

Phylum	Species	Belly	Track	Diving	Control
Mollusca	*Mytilus edulis platensis*	48.5 ± 18.6 (19)	44.9 ± 20.7 (19)	60.7 ± 3.9 (4)	71.0 ± 6.6 (15)
	Pododesmus leloiri	0.18 ± 0.1 (4)		0.05 – 1	
	Hiatella sp.	0.14 ± 0.1 (5)			
	Ostrea puelchana	0.32 ± 0.1 (13)	0.61 ± 0.2 (4)	0.06 ± 0.0 (2)	0.6 ± 0.1 (6)
	Eurhomalea exhalbida	0.14 ± 0.0 (7)	0.26 – (1)		
	Aulacomya atra	0.14 ± 0.0 (4)	1.7 ± 1.3 (2)		
	Hiatella árctica	0.12 – (1)			
	Protothaca antiqua	0.22 ± 0.0 (2)	0.89 – (1)		
	Aequipecten tehuelchus	0.19 ± 0.1 (4)		0.07 ± 0.0 (2)	1.6 ± 0.0 (2)
	Darina solenoides	0.1 – (1)			
	Atrina seminuda	0.09 ± 0.02 (2)	1.0 ± 0.7 (4)		
	Ostrea stentina	0.1 ± 0.0 (2)	0.26 – (1)	0.06 ± 0.0 (2)	
	Pitar rostratum		1.4 ± 0.23 (2)	0.07 – (1)	
	Semele proficua	0.33 ± 0.0 (2)	0.50 – (1)	0.11 – (1)	1.4 ± 0.81 (5)
	Glycimeris longior				1.1 ± 0.2 (6)
	Tegula patagonica	0.56 ± 0.5 (13)	1.3 ± 1.9 (9)	0.68 ± 0.0 (4)	0.56 ± 0.2 (9)
	O. magellanica	0.1 – (1)			
	Calliostoma coppingeri	0.12 ± 0.0 (2)		0.06 ± 0.0 (2)	0.79 – (1)
	Olivella tehuelcha		5.6 – (1)		
	Crepidula sp.	0.21 ± 0.1 (10)	0.68 ± 0.2 (3)	0.17 ± 0.1 (2)	0.48 ± 0.3 (4)
	Octopus tehuelchus	0.11 ± 0.0 (6)	1.0 ± 0.5 (5)		
	Semirossia tenera	0.12 ± 0.0 (2)			
	Chaetopleura isabellei	1.3 ± 0.8 (19)	1.2 ± 1.0 (12)	0.98 ± 0.4 (4)	5.3 ± 1.3 (5)
Annelida	*Glycera* sp.	0.49 ± 0.3 (2)	2.1 – (1)	0.92 ± 0.8 (2)	
	Aphrodite sp.		1.1 ± 0.7 (2)		
	Polichaeta sp. 1	0.31 ± 0.2 (11)	1.0 ± 0.6 (2)	0.42 – (1)	0.24 ± 0.2 (3)
	Polichaeta sp. 2	0.20 ± 0.1 (2)	0.26 – (1)	0.44 – (1)	0.94 – (1)
Nemertea	Nemertea sp. 1	0.54 ± 0.4 (2)	1.6 ± 0.5 (2)	0.53 ± 0.07 (3)	0.48 ± 0.3 (2)
Plathyelm.	Turbellaria sp.	0.09 – (1)		0.06 ± 0.0 (2)	
Arthropoda	*Coenophtalmus tridentatus*	0.09 ± 0.0 (4)	1.4 – (1)	0.06 – (1)	0.63 ± 0.5 (3)
	Pilumnoides hassleri	0.21 ± 0.17 (6)		0.24 ± 0.1 (4)	
	Pellia sp.	0.17 ± 0.1 (2)	1.05 – (1)		
	Platyxanthus patagonicus	0.11 ± 0.0 (3)			
	Leurocyclus tuberculosus	0.15 ± 0.1 (11)	1.22 – (1)		0.42 – (1)
	Rochinia gracilipes	0.28 ± 0.2 (11)	0.77 ± 0.2 (4)	0.13 ± 0.0 (4)	0.45 ± 0.0 (6)
	Tumidotheres maculatus	0.3 – (1)	0.70 – (1)		1 – (1)
	Peltarion spinosulum	0.43 ± 0.36 (6)		0.55 ± 0.11 (6)	
	Libinia spinosa	0.12 – (1)			
	Ovalipes trimaculatus	0.09 ± 0.0 (2)			
	Betaeus lilianae	0.20 ± 0.1 (9)	1.0 ± 0.3 (4)	0.15 ± 0.02 (4)	0.65 ± 0.2 (7)
	Alpheus puapeba	0.16 ± 0.1 (8)	1.8 – (1)	0.08 ± 0.0 (3)	0.97 ± 0.5 (2)
	Balanus sp.	0.2 – (1)	1.4 – (1)		
Echinod.	*Astropecten* sp.	6.2 ± 6.4 (16)	1.7 ± 1.5 (6)		
	Comasteria lurida	0.2 ± 0.2 (13)	0.38 ± 0.2 (4)		
	Allostichaster inaequalis				0.62 ± 0.2 (9)
	Ophioplocus januarii	35.7 ± 16.9 (19)	29.2 ± 17.3 (17)	31.8 ± 5.8 (4)	14.4 ± 7.2 (15)
	Arbacia dufresnei	4.6 ± 4.4 (19)	5.8 ± 7.1 (15)	1.9 ± 1.1 (4)	3.4 ± 1.5 (15)
	Cycethra pinguis	0.12 – (1)			
	P. magallanicus	0.22 ± 0.1 (8)	0.63 ± 0.3 (3)	0.23 ± 0.0 (3)	0.89 ± 0.0 (5)
Chordata	*Cnemidocarpa robinsoni*	0.20 – (1)			
	Paramolgula gregaria	0.24 ± 0.1 (4)	0.06 ± 0.0 (2)	0.09 ± 0.0 (4)	
	Ciona robusta	1.9 ± 1.8 (18)	1.3 ± 1.9 (9)	0.33 ± 0.1 (3)	2.1 ± 2.1 (11)

index (Clarke and Warwick, 2001). ANOSIM was performed to test for differences in composition of assemblages (in density) between treatments (belly, track, commercial diving and control site).

2.2. Individual damage

Individual damage in mussels was assessed by comparing percentages of damaged mussels and damage levels induced by the two fishing methods. In the last case, the relative proportions of damaged mussels in the belly and in the samples obtained from commercial diving were recorded and calculated by assigning them one of the three following categories: a) undamaged, b) lightly damaged (individuals with less than 20% of the shell surface broken), and c) severely damaged (individuals with more than 20% shell surface or hinge broken). Mean shell height and weight of undamaged and damaged mussels were recorded.

Sea urchins (*Arbacia dufresnei*) and ophiuroids (*Ophioploccus januari*) were the most abundant bycatch species (more than 77% in number and 58% in weight of the total bycatch) in samples obtained at the control site and from both the dredge and commercial diving (Narvarte et al., 2011). An *ad hoc* scale was adopted to assess damage in these bycatch species. Individual damage of sea urchins was assessed considering the percentage of lost spines, according to the following scale: a) undamaged, b) lightly damaged, with up to 50% of the spines lost, and c) severely damaged, more than 50% of the spines lost and/or of the body crushed. For ophiuroids, the damage scale was as follows: a) undamaged, b) lightly damaged, with up to 50% of the arms broken, and c) severely damaged, with >50% of the arms broken and or the disc crushed.

Mean shell height of undamaged and damaged mussels was compared using a *t*-test. To compare damage percentages and damage categories between treatments (belly, track, diving and

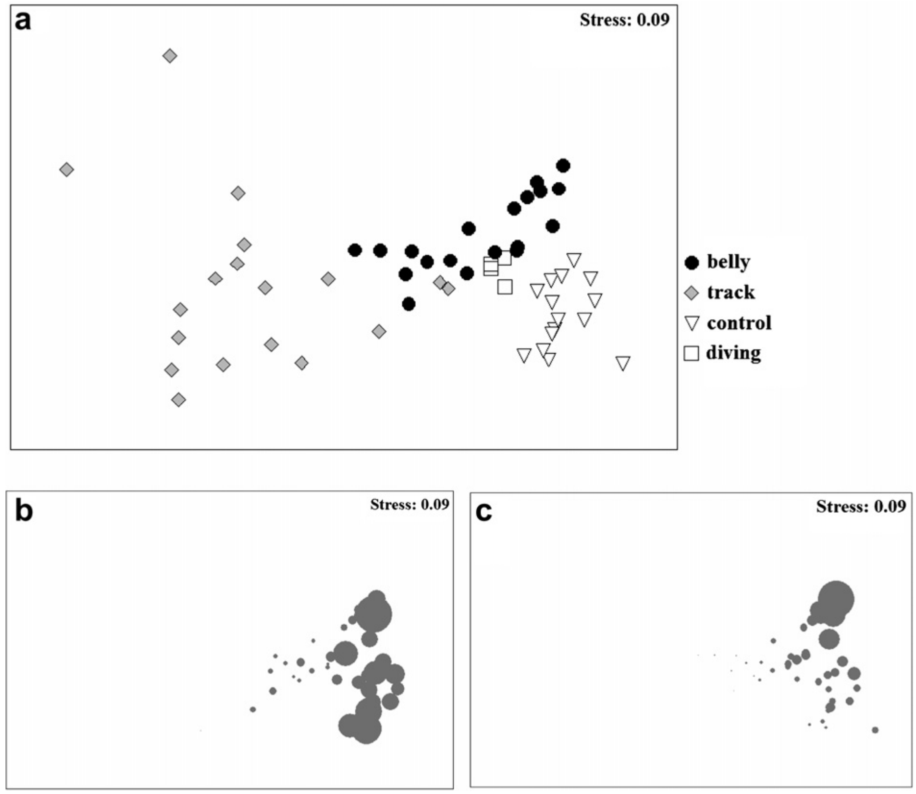

Fig. 2. Non-metric multidimensional scaling (MDS) plots for the density of the bycatch assemblage in each group, in (a); and corresponding bubble plots of the species *Arbacia dufresnei* (b) and *Ophioplocus januarii* (c) which primarily provided the discrimination between observed sample clusters. Positions of bubbles correspond to points on MDS ordination.

controls), ANOVA and Tukey test for *a posteriori* pair comparisons were used. When necessary, data were transformed (arcsin$\sqrt{p}$) to meet assumptions. Statistical analyses were conducted with a significance level of $\alpha = 0.05$ and the software package Infostat® (Universidad Nacional de Córdoba, Argentina).

2.3. Sediment analyses

For sediment characterization, two approaches were considered:

[1] Core sampling on the track, in the adjacent unfished area (local control), and in the control site, to assess the immediate impact. In two of the 19 tows (tows 3 and 6), plastic cores (5 cm in diameter and 25 cm in length) were used by a scientific diver swimming behind the dredge. For each tow, two core samples were taken, one within the track and the other 2 m aside the track in a site with mussels but unfished (local control). Cores were frozen and cut to obtain two portions (5 cm in height each): one from the surface level named "shallow", and another from the lower level, named "deep". Processing of sediment samples was performed according to Folk (1974) and data (weight and percentage of each fraction) were used for sediment classification and mean grain size estimates (Shepard, 1954; Vozza et al., 1974). Grain size was expressed as Phi (Φ) $= -\log_2 d$, where d is the grain diameter (in mm). Core samples were also taken in the control site. Similarity among samples was determined using the Bray–Curtis similarity coefficient. Cluster analysis was performed using group-average with the software PRIMER (Clarke and Warwick, 2001). Mean grain size, kurtosis, skewness, and gravel percentage were used as variables. ANOSIM (analysis of similarities) was used to test the significance of differences in composition of samples from dredged (track samples: T3 shallow, T3 deep, T6 shallow, T6 deep) and undredged sites (local control samples: LC3 shallow, LC3 deep, LC6 shallow, LC6 deep; and control site: CS, shallow and deep).

[2] Sediment trapping within the fished area and in the control site for one month. Four sediment traps constructed of concrete pieces (50 cm length × 20 cm width × 15 cm height) each carrying 5 five glass flasks (11.5 cm height and 5.5 cm mouth diameter) were deployed along the tidal current direction near the fishing area of the artisanal fleet, and left on the bottom for one month to catch sediments transported by currents. A fifth sediment trap was also deployed in the control site to catch natural sedimentation far from the fishing area. The top of each flask was covered to prevent sediment loss during diver ascent. Sediment accumulated in each flask was dried, weighed and qualitatively analyzed, since some samples were missed (see below).

3. Results

In a total of 57 samples (pooling all treatments together), there were 50 taxa (Table 1), of which only three species (*M. edulis platensis*, *A. dufresnei* and *O. januarii*) comprised more than 75% of all animals in each treatment. The MDS ordination for macrofauna density (Fig. 2) showed a considerable degree of similarity, and low stress values (0.09), indicating a good and useful 2D representation of the groups. The samples corresponding to track, belly and control site clustered more or less discretely, and were arranged in a clear linear sequence (left to right), whereas samples of commercial

Table 2
Summary ANOSIM statistics using multivariate analysis on bycatch species assemblages. High biological importance is illustrated by $R > 0.4$ and significant differences $p < 0.05$. Bold indicates meaningful differences. Right column indicates the cumulative percentage of the main bycatch species (*Arbacia dufresnei* and *Ophioplocus januarii*) in each case.

Treatment comparison	*R* statistic	Signif. level %	Cum. % main bycatch spp.
Global test	**0.542**	**0.1**	
Diving – control	**0.296**	**3.4**	**17.4**
Diving – track	**0.512**	**0.1**	**49.4**
Diving – belly	0.015	42.6	
Control – track	**0.825**	**0.1**	**34.0**
Control – belly	**0.591**	**0.1**	**25.6**
Track – belly	**0.631**	**0.1**	**46.7**

diving were clustered with belly. Consistently with the MDS ordinations, ANOSIM tests on square root transformed density indicated that with the exception of the group of belly + commercial diving, all treatments differed significantly from each other at the 5% level (Table 2). Bubble plots identified the species primarily providing the discrimination between observed clusters. *A. dufresnei* and *O. januarii* were the most important species contributing to the observed differences (17–49%; Table 2; Fig. 2b, c).

3.1. Damage of mussels and bycatch species

Damage of mussels induced by the two fishing methods in the experimental site was negligible (<4%, Fig. 3), whereas that of mussels from the control site was null. Percentages of damaged mussels were significantly different between belly, track and diving samples (1 way ANOVA: $F_{1,\ 41} = 26.69$, $p < 0.0001$). Mean percentages of damaged mussels in the belly and commercial diving catches were significantly higher than in the dredge track (Tukey *post hoc* test, $p < 0.05$). The percentage of severely damaged mussels (pooling data from belly and track) was significantly higher than that of lightly damaged ones (1 way ANOVA: $F_{1,\ 45} = 5.39$, $p = 0.0249$).

Mean height of undamaged individuals caught in the belly was 66.5 mm ($n = 1789$), whereas that of damaged individuals was 65.9 mm ($n = 129$). However, differences in mean size were not significant (*t*-test, $t = 0.45$, $p > 0.05$). Conversely, differences in mean weight were significant (undamaged: 32.4 g, $n = 1789$; damaged: 19.6 g, $n = 129$; *t*-test: $t = 15.1$, $p < 0.05$). For the commercial diving catches, mean shell height of undamaged mussels was 58.4 mm ($n = 1212$) and mean weight 23.1 g, whereas the values for damaged mussels were 62.4 mm and 15.7 g, respectively. In this case, significant differences in the weights between undamaged and damaged mussels were also found (*t*-test: $t = 5.34$, $p < 0.05$).

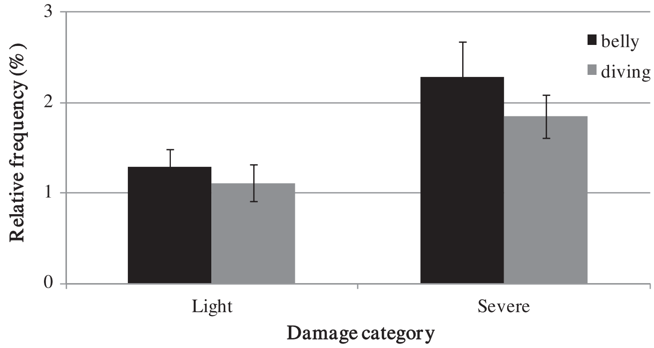

Fig. 3. Relative frequency (mean ±SD) of undamaged and damaged categories of mussels captured in the belly and by commercial diving.

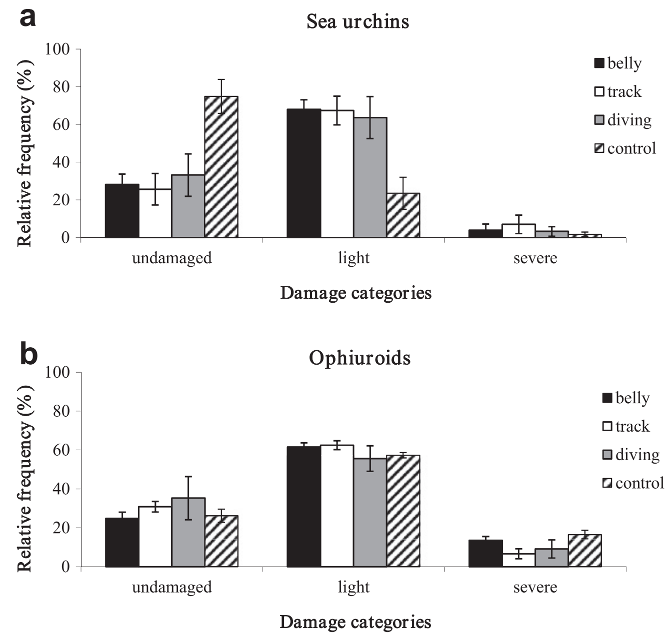

Fig. 4. Relative frequency (mean ±SE) of undamaged and damaged categories of the main bycatch species captured in the belly, track, commercial diving and control site for a) sea urchins and b) ophiuroids.

Significant differences were found in the mean percentages of damaged sea urchins between treatments (1 way ANOVA: $F_{3,\ 56} = 16.09$, $p < 0.0001$). Samples from the control site showed the lowest values (Tukey, $p < 0.05$) and no differences were found among damage percentages between samples from the belly, track and commercial diving (Tukey, $p > 0.05$). Damaged sea urchins constituted a mean (in numbers) of 75.9, 74.3 and 66.9% in the belly, track and commercial diving samples respectively (Fig. 4a). The proportion of damaged sea urchins in samples from the control site averaged 25.1 ± 14%. Considering all the samples with the similar damage level together (belly, track and commercial diving), percentages of lightly damaged sea urchins were significantly higher than those of severely damaged ones (1 way ANOVA: $F_{1,\ 83} = 292.57$, $p < 0.0001$). Interesting, combining the data from the Table 1 and those of the damage level (Fig. 4a), the mean number of damaged sea urchins per 1000 mussels caught in the belly and the diving samples were 71 and 21 respectively (without accounting for those that are left damaged in the seafloor, i.e., track).

Contrary to the results obtained for sea urchins, there were no significant differences in the mean percentages of damaged ophiuroids between treatments (1 way ANOVA: $F_{3,\ 56} = 0.97$, $p = 0.4161$, Fig. 3b). In this case, damaged ophiuroids averaged 75.1, 69.1, 64.7 and 73.8% in samples from the belly, track, diving and control site, respectively. Considering all the treatments together (belly, track, diving and control), percentages of lightly damaged ophiuroids were significantly higher than those of severely damaged ones (1 way ANOVA: $F_{1,\ 113} = 509.64$, $p < 0.0001$). Again, combining the data from the Table 1 and those of the damage level of Fig. 3b (without accounting for those of the track), the mean number of damaged ophiuroids per 1000 mussels caught in the belly and the diving samples were 552 and 339 respectively.

3.2. Sediment samples

Particle size distributions of the sediments collected were similar between the samples from the dredge track, local controls and control site, with a mean phi value (Φ) ranging from −0.38 to

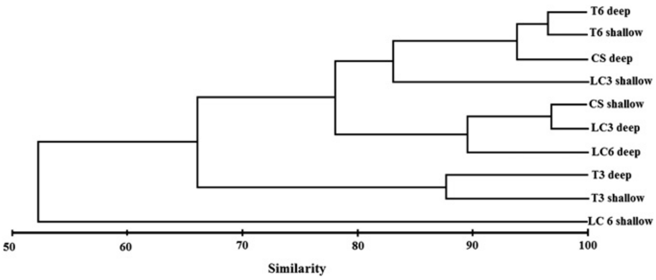

Fig. 5. MDS ordination of sediment samples based on granulometric analysis (mean grain size, kurtosis, skewness, gravel percentage) and divided into those fished and fished areas. [Ref.: CS, control site, the same as for biota sampling; T3, T6: samples taken from the dredge tracks on hauls 3 and 6, respectively; LC3, LC6: local control samples taken as immediate impact controls aside each track; "s" and "d" indicate the shallow and deep core sub-samples].

0.83. The gravel percentage between depth levels for all samples varied from 9 to 15% in the upper level and from 30 to 35% in the lower sections. The cluster analysis of samples based on granulometric analysis considering also kurtosis and skewness produced two groups with over 65% similarity (Fig. 5). While shallow and deep levels from track T3 were grouped, shallow and deep levels from local controls (LC3 and LC6) were separately grouped between tows. Shallow and deep samples from the control site were associated indistinctly with the other samples. No significant differences between treatments (fished and unfished) were identified using ANOSIM (Global R: 0.085, $p = 0.64$).

Only two of the five sediment traps deployed (one at the fishing ground and one at the control site) were recovered after one month. Three traps had almost certainly been trawled by the dredges during the fishing operations, and then we could not perform statistical inferences about the results found. Mean ($\pm$SD) weight per flask of the sediments collected in the fishing area (207.9 ± 12.2 g, $n = 5$) was three times higher than in sediments collected in the unfished control zone (65.9 ± 7.9 g, $n = 5$). Mean ($\pm$SD) grain size of sediments in traps of the fished area was $\Phi = 1.15 \pm 0.029$ and that of unfished area was $\Phi = 1.89 \pm 0.092$. Most of the sediment collected (>50%) in areas was composed of sand of intermediate sizes; however, sediment of the fished area had 30% of gravel and that of the control site had 30% of fine sand.

4. Discussion

4.1. Composition of catches

The greatest number of bycatch species on the mussel bed under exploitation could be a first indicator of impact through environmental changes by drag removal of organisms and/or sediments. The remaining species left on the bottom after removal can attract to closely related predators (e.g., crabs and sea stars). Studies carried out in other fisheries worldwide have shown that the primary impact of bottom fishing seems to be on mega-epibenthic organisms (Kaiser et al., 1998; Bergman and van Santbrink, 2000; Freese et al., 1999). Another effect is related to the removal of the fine sediments, which may expose infaunal organisms, not observed (or counted) in samples of epifauna collected by divers (both commercial, and scientific in the control site). For example, eight species of infaunal bivalves were found in samples from the belly and the track and only three were collected by divers. Previous studies have also shown that bottom trawling or dredging modifies the seafloor habitats by removing or damaging infauna and sessile organisms (Auster et al., 1996; Freese et al., 1999; Kefalas et al., 2003). Results of the community analyses of the mussel bed here studied showed remarkable differences between treatments (belly, track, diving and control site) mainly related with densities of some particular bycatch species accompanying sea urchins and ophiuroids. Interestingly, the exception was the comparison of sample composition between diving and belly, which did not show significant differences. However, our results should be carefully interpreted since an unequal spatial coverage and different sample sizes were considered for the different treatments, and results may reflect a species-area effect. For example, near 1200 m^2 were the effectively dredged area; in the control site, samples were taken with quadrats (3.75 m^2) within a greater mussel bed of approximately 1 Ha. In the case of commercial diving, it is expectable that fishermen select the target species, which may also explain part of the variation. Thus, although the sampling was representative for comparative purposes in the number of species, a more extended sampling is recommended for robust estimates of density or biomass per species. Also, it would be interesting to perform a detailed core sampling to evaluate infauna, as an additional important issue on the gear-impact assessment.

4.2. Damage levels on target and bycatch species

The damage caused by dredging on mussel catches was low (less than 5% of the total catch) and there were no differences in the mean size between damaged and undamaged individuals, indicating a similar vulnerability against the artisanal dredge, at least in the size range of the monocohort mussel bed analyzed. If recruits were present, due to their relative thinner shell, they probably would be more vulnerable than adults, as was observed for infaunal bivalves damaged by clam dredges (Gaspar et al., 2003). The variations found between weights of entire and damaged mussels were surprisingly high (near 40%), and would be explained by a sudden decrease in the water content of the individual once the shell is broken. The relative higher damage of mussels caught in the dredge and by commercial divers comparing to the damage rates of mussels left on the dredge track is indicative that the damage may occur during the on-board post-capture handling more than by the direct effect of the fishing methods. Considering that divers use only their hands to collect bivalves, shell damage was probably caused by the weight of the catch on hauling (both in the artisanal diver bags and in the belly) or even during sorting on the deck.

Individual damage of bycatch species also showed negligible differences between the different capture methods evaluated. High damage of ophiuroids was also found in control areas, indicating that these organisms are extremely fragile, resulting damaged with minimal stress inside the bags (including those of the scientific divers). Damage in bycatch species may be due to the abrasion between animals and/or between animals and debris (empty shells or, in this study, pebbles) inside the bag as also observed in other studies (Gaspar et al., 1998, 2002). Even when the damage level of the most important bycatch species appears to be significant (more than 75% in some cases), it corresponded mainly to lightly damaged individuals. Echinoderms are well known for their striking regenerative potential and their ability to rapidly and completely regenerate arms or spines (Hobson, 1930; Ebert, 1967; Heatfield and Travis, 1975; Candia Carnevali et al., 1995, 1998, 2006; Dolmatov, 1999; Thorndyke et al., 2001; Hotchkiss, 2009). Considering the extraordinary capacity of these species to regenerate body parts, the mortality rates induced by fishing should not be significant. However, survival of the individuals left on the sea bed should be experimentally tested, considering that despite their regenerative potential, they may become easy prey. Although differences in individual damage levels between the fishing methods were not significant, the overall damage level for both bycatch species, the combination of data from the catches composition and those of the damage level per 1000 mussels caught, showed greater differences between methods. For example, damage in the belly was shown to be 3.4 and 1.6 times higher than that of diving, for sea urchins and ophiuroids respectively. Moreover, damage of individuals left on the dredge track (also produced by the dredge) should also be considered in integral damage estimation. These differences in the number of individuals damaged during the fishing process (when the catch is somehow standardized) were not reflected by the percentage of individual damage (which differed just a 10% between treatments) illustrating that an all-encompassing analysis is more informative about the actual impacts at the ecosystem level.

In an assessment of the damage caused by a scallop dredge in North Irish Sea, it was found that the mean damage level to some common megafaunal species was similar between captured organisms landed on deck (bycatch) and those passed through the dredge but left on the seabed (Jenkins et al., 2001). For the Australian scallop “mud” dredge, the selectivity model made by McLoughlin et al. (1991) indicates that only 12–22% of the initial stock in Banks Strait was landed as catch, with the remaining of the stock wasted through direct and indirect mortality resulting from dredging. Undoubtedly, the artisanal gear used to fish mussels in SMG appears to be less destructive and more efficient than others reported in fisheries around the world. In relation with this, several authors (e.g. Caddy, 1973; Meyer et al., 1991; McLoughlin et al., 1991; Gaspar et al., 2003; Pezzuto et al., 2010) have found that there is a correlation between catch efficiency and damage: the higher the efficiency, the lower the damaged inflicted. The results presented in this paper may corroborate this statement.

Few attempts have been made to reduce the bycatch of benthic invertebrates, and increasing emphasis on fishing effects at the ecosystem level should incorporate non-target species in the future fishing management decisions. The development of gear designs and methods to reduce the bycatch of commercial species has been based on the assumption that animals that escape suffer negligible mortality (Jenkins et al., 2001). Whereas some studies (Broadhurst, 2000; Gaspar et al., 2001; Leitão et al., 2009) agree with this, other authors (Moschino et al., 2003; Jenkins et al., 2001; Chopin and Arimoto, 1995) have reported the opposite. Despite this, simple management measures directed to reducing the impact of dredging in mussel fisheries, i.e. discarding bycatch species on the same fishing grounds, should be enforced in the context of a precautionary management approach.

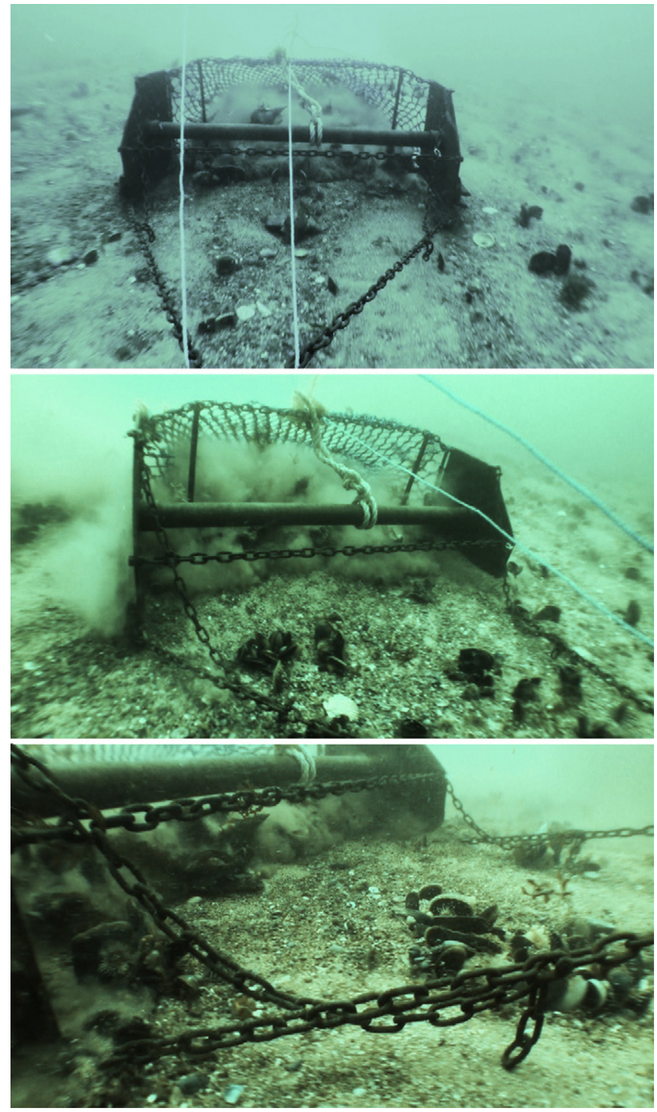

Fig. 6. Images of the artisanal dredge operating on the sea bottom of San Matías Gulf.

4.3. Physical disturbance

Most dredges around the world are rake-like devices equipped with a toothed lower bar in the gear mouth and use a net bag to collect the catch (FAO, 1987; Gaspar et al., 2002; Gabriel et al., 2005). Teeth are forbidden in several fisheries and are usually replaced by a sharp blade shearing downward (Schärfe, 1978; Gabriel et al., 2005). Dredges traditionally used in SMG fishery (both industrial and artisanal) present a different design, having a stretched footrope (made of chain or wire) to sweep the sea floor. In our study, the video images recorded by the divers showed that, during tows, the footrope of the artisanal dredge moves buried 1–2 cm under the seafloor, avoiding the escape of the epibenthic organisms and substrates under it. This effect, which contributes drastically to the high dredge efficiency, also causes the removal of finest sediment (Fig. 6).

We studied the immediate effect of this sediment removal by analyzing the granulometry of the sea floor after dredge passing and against unfished local controls and the distant control site. Although variations in the gravel percentages between fished and unfished areas were observed, we did not find relevant differences in terms of mean grain size or patterns induced by fishing. Moreover, sediment samples from the upper layer in the unfished area appeared similar to the samples from the lower level in the local controls, which demonstrates that sediment stratification is not uniform under natural conditions.

The most noticeable environmental impact of dredging on a mussel bed in the Irish Sea was the burial of organic material, which favored anaerobic microbial respiration (Meyer et al., 1991; Maguire et al., 2007). After a disturbance, the fine grains can be carried away by currents, and discover a harder substrate. Also, disturbance on pebble, sandy and muddy bottoms cause ground homogenization therefore reducing habitat complexity important for larvae recruitment (Maguire et al., 2007). Although the sedimentary analyses in the present study were preliminary (since only one sediment trap per area (fished/unfished) was obtained) and their repetition in a wider spatial and temporal scale would be convenient, we determined that the amount of sediment neighboring the fishing ground area which was removed was three times larger than that removed from the control site (non-dredged area). Despite the unnatural origin of this removal, sediment mobilization due to natural factors appears to be a usual phenomenon in the study site. Sediment composition in the northwestern area of SMG is similar along the coast line and on the seafloor over the 20 m isobaths (Aliotta et al., 2000). Significant volumes of sandy sediments are annually moved to this area by two main agents: the dominant Patagonian winds, which move fine sand from the coastal dunes and beaches, and the strong tidal currents, which transport suspended sediments from San Antonio Bay (Schnack et al., 1996). Tidal currents in the fishing area reach a mean speed of 0.21 m s^{-1} with maximum intensity of 0.54 m s^{-1} (PNUD ARG/02/018 GEF BIRF #28385-AR), which are considered strong enough to naturally lift and move fine sediments away (Schnack et al., 1996).

Results similar to those of our study were found comparing sediment composition between trawled and unfished areas. For example, for prawn trawling in the coast of Sinaloa–Sonora (Mexico), the bottom structure and composition have been described as able to absorb the trawling effects (Sánchez et al., 2009). Also, no differences in particle size distribution and sediment types were found between fished and unfished experimental sites in blue mussel beds (Dolmer et al., 2001). In our study, the dredging progressively eliminated the original sandy gravel, exposing the underlying shingles on the edge of the drag-head tracks, while fine sand derived from mobile sand-ripples and from overflow had filled the furrows.

The degree of the initial effects and the rate of recovery may depend on the habitat stability. In more stable biogenic habitats (gravel and mud) alterations are greater and recovery rates slower than in areas of high natural disturbance which present less consolidated coarse sediments and show fewer initial effects (Kaiser et al., 2000; NRC, 2002). Then, dredging in areas of relatively high natural disturbance may have relatively short-term biological significance because of the rapid recovery of the physical environment and the biological communities (Kaiser et al., 2000; Hiddink, 2003; Gaspar et al., 2009; Constantino et al., 2009). Indeed, communities of high-stress areas (e.g., shallow areas exposed to strong tidal currents and periodic storm disturbance) have a greater capacity to readjust to the impact of dredging operations than more stable communities and recovery may be largely completed within one year (Desprez, 2000). A similar result was found by Churchill (1989) and Kaiser et al. (2000) working on habitats of different stability levels. They found that scallop dredges and beam trawls used on more stable habitats appear to have greater impacts on the environment than lighter otter trawls used in shallower water with less stable sediments.

5. Conclusions

The primary impact found in this study was the removal of mega-epibenthic organisms and of fine sediments. The damage caused by the dredge on mussel catches, which was similar to that caused by divers, was negligible, and may occur during the on-board post-capture handling more than by the direct effect of the fishing methods. Damage of the most important bycatch species was more than 75% but it corresponded mainly to the lightly damaged individuals. Differences in the overall damage between fishing methods resulted to be higher if both sources of data, i.e. samples composition and damage level, are jointly used in the estimates. If the bycatch is rapidly discarded, mortality rates induced by fishing should be less important due to the rapid regeneration of arms or spines shown by these species. Undoubtedly, the artisanal gear used to fish mussels in San Matías Gulf appears to be less destructive and more efficient than others reported in fisheries around the world. However, simple management measures, e.g. discarding bycatch species on the same fishing grounds, should be enforced in the context of a precautionary management approach.

Finally, the amounts of sediment removed in the fishing areas are higher than in unfished areas. Despite the unnatural origin of this removal, which was monitored for one month, sediment mobilization due to natural factors is a usual phenomenon in the study site. Probably, in these shallow areas exposed to periodic storm disturbance, communities could readjust to the impact of dredging and recovery is expected to take place relatively soon.

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