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Effect of recreational diving on Patagonian rocky reefs

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A B S T R A C T

Tourism has grown considerably in the last decades, promoting activities such as recreational SCUBA diving that may affect marine benthic communities. In Puerto Madryn, Patagonia Argentina, sub-aquatic tourism areas (STA) receive about 7,000 divers per year. Diving is concentrated on a few small rocky reefs and 50% of the dives occur in summer. In this work, we evaluated the effect of recreational diving activities on benthic communities and determined whether diving causes a press (long-term) or a pulse (short-term) response. We quantified the percentage cover of benthic organisms and compared benthic assemblage structure and composition between two sites with contrasting usage by divers, 'highly disturbed' and 'moderately disturbed' sites, and two 'control' sites with similar physical characteristics but no diving activity, twice before and after the diving peak in summer. We found differences in benthic assemblage structure (identity and relative abundance of taxa) and composition (identity only) among diving sites and controls. These differences were consistent before and after the peak of diving in summer, suggesting that recreational diving may produce a press impact on overall benthic assemblage structure and composition in these STA. At the moderately disturbed site, however, covers of specific taxa, such as some key habitat-forming or highly abundant species, usually differed from those in controls only immediately after summer, after which they begun to resemble controls, suggesting a pulse impact. Thus, STA in Golfo Nuevo seem to respond differently to disturbances of diving depending on the usage of the sites. This information is necessary to develop sound management strategies in order to preserve local biodiversity.

1. Introduction

Human disturbances on formerly remote habitats are diverse and include activities such as hunting, fishing and the collection of organisms for recreational or commercial purposes (Vitousek et al., 1997; Castilla, 1999; Cloern, 2001; Crutzen, 2006). The increased use of natural habitats may lead to a decline in the resources and/or services that attracted people. Among the various humans disturbances that can lead to changes in natural systems are tourism and the activities derived from it (Miller, 1993; Hall, 2001; Davenport and Davenport, 2006). Tourism has grown significantly in the last decades and, as a consequence, marine benthic communities have been affected by boating, trampling, snorkelling and SCUBA diving (Addesi, 1994; Garrabou et al., 1998; Eckrich and Holmquist, 2000; Minchinton and Fels, 2013). In particular, SCUBA diving is amongst the fastest growing pastimes in the world, with *ca* 1 million new recreational divers being trained each year (Van Treeck and Schumacher, 1998; Zakai and Chadwick-Furman, 2002; Hasler and Ott, 2008). Participants are drawn to the most attractive diving sites, some of which are located in marine protected areas (Milazzo et al., 2004; Davenport and Davenport, 2006; Smith et al., 2008). Divers, however, can affect natural habitats through direct physical damage and can also interfere with ecological processes, including reproduction and growth of species, as well as trophic interactions (Zakai et al., 2000; Chabanet et al., 2005; Luna et al., 2009). Moreover, most studies on potential impacts of diving are done in tropical systems, while temperate systems have been largely

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overlooked.

In Patagonia Argentina, one of the main tourist attractions is Península Valdés (42°30′ S, 64°00′ W), a natural reserve created in 1983 and recognized as World Natural Heritage by the UNESCO in 1999. Puerto Madryn is the closest city to Península Valdés; therefore, tourism is one of the major activities of the city, which receives *ca* 250,000 tourists per year (Secretaría de Turismo 2014). Puerto Madryn has been among the top diving destinations in Patagonia since the late 50s, attracting several thousands of divers per year from all over the world (Sanabra, 2002; Torrejón et al., 2012). There are 14 sites within Golfo Nuevo bay, where Puerto Madryn is located, that are utilized as 'sub-aquatic tourism areas' ('STA'). STA are areas of ecological interest for sustainable tourism development and conservation. These areas are chosen because of desirable characteristics such as clear waters, shallow rocky reefs (<25 m depth), proximity to the city, impressive underwater scenery and high biodiversity. The local government is currently developing policies to formally create and regulate STA; however, these policies have not been finalized nor implemented yet.

STA usually have a predominantly flat topography with soft bottoms from which rocks outcrop forming rocky reefs (Ciocco, 1988; Galván, 2008; Irigoyen, 2010). These reefs extend from a few to several hundred square-meters and their cracks and ledges are used by a wide variety of fish species (Ciocco, 1988; Galván, 2008; Irigoyen, 2010). As in other temperate reefs, surfaces are mainly colonized by algae and sessile invertebrates such as mussels, anemones and ascidians (Osman, 1977; James and Underwood, 1994; Baynes, 1999; Piriz et al., 2003; Irigoyen et al., 2011). Water temperature in Golfo Nuevo ranges between 9 and 16 °C (Dellatorre et al., 2012); hence, divers typically use 7 mm neoprene wetsuits and large amounts of weight to control buoyancy. As a result, inexperienced divers often touch or even hit the organisms present on the rocky reefs, unintentionally damaging fragile benthic flora and fauna and creating patches of bare rock. Disturbance of the seabed and the creation of bare rock patches can, in turn, facilitate invaders, such as the invasive kelp *Undaria pinnatifida*, which has a negative effect on native algae, dramatically reducing its richness and diversity (Casas et al., 2004; Raffo and Irigoyen, 2011). In addition, local dive instructors frequently touch animals, feed fishes with benthic organisms collected on-site (*e.g.* mussels) and guide divers into tight spaces, all with the intention of enhancing the overall diving experience (Bravo, 2013). This diver–reef interaction, together with the great amount of divers over small areas (Bravo, 2013) could result in the deterioration of benthic communities in the STA of Puerto Madryn.

Bender et al. (1984) defined two types of human disturbances: 'pulse' and 'press'. Pulse disturbances are short-term, intense episodes of disturbances that are then removed. Press disturbances, instead, are chronic and persistent. For a conservation perspective, rehabilitation or reduction of the effects of pulse disturbances may be relatively easy compared with press disturbances, because the cause of the potential impact ends relatively fast (Bender et al., 1984; Underwood, 1992; 1994). Pulse disturbances can, however, elicit either a pulse or a press response on organisms (and *vice versa*; Glasby and Underwood, 1996). Most diving activities in Puerto Madryn occur during summer, with 50% of the dives occurring in January and February, while the rest is spread equally year round (Bravo, 2013). Diving may therefore be considered a pulse disturbance as most diving occurs in a particular time of the year, although at a larger temporal scale it may be considered a press disturbance as it has been repeatedly done every summer for decades. However, in terms of the type of response this disturbance may elicit on benthic assemblages, potential effects of diving may be short-term, with benthic community structure changing in response to diving during and after summer, but eventually becoming similar to non-disturbed sites before the next diving season starts. Such short-term or acute disturbances have been classified as 'pulse' disturbances, with regards to the way affected organisms respond to them (Bender et al., 1984; Underwood, 1989). Alternatively, the effects of diving on benthic communities may be long-term, with effects on community structure lasting longer than the period between diving seasons. These long-term, chronic disturbances have been referred to as 'press' disturbances as they elicit chronic changes in the ecology of the organisms affected (Bender et al., 1984; Underwood, 1989). Understanding the type of response that recreational diving can exert on benthic communities in STA of Puerto Madryn will be useful to develop successful management strategies (Underwood, 1994; Glasby and Underwood, 1996).

There are no quantitative data available concerning the potential impacts of diving on local benthic communities. Our aim was to evaluate the effect of recreational diving activities on rocky reef benthic communities and to determine whether disturbance by diving causes a press or a pulse impact on the community as a whole and/or on specific taxa. Specifically, we predicted differences in the structure and composition of benthic assemblages between disturbed and control sites. If diving results in a press impact, we predicted a consistent difference in assemblage structure between disturbed and control sites before and after the diving season, given that disturbed sites have been used intensively for the past two decades (Bravo, 2013). If, however, diving results in a pulse impact, we predicted differences between disturbed and control sites only after the diving season. We compared the percentage cover of benthic organisms between two 'disturbed' sites, which are regularly used by divers, and two 'control' sites with similar physical characteristics but where almost no diving is done, twice before and after the diving peak in summer, and used multivariate analyses to determine potential effects at the assemblage level and univariate analyses for specific taxa.

2. Materials and methods

The study was performed at two disturbed sites, a 'highly disturbed' one that receives *ca* 1000 divers per summer season (42.779° S, 64.992° W), and a 'moderately disturbed' site (42.778° S, 64.988° W) that receives *ca* 500 divers per season, as well as at two 'control' sites (42.779° S, 64.959° W; 42.776° S, 64.958° W) visited by few divers only occasionally (Bravo, 2013). Sites were chosen based on the information obtained from records of the Puerto Madryn Tourism Office (Secretaría de Turismo 2014) and interviews with dive instructors and dive-centre owners (Bravo, 2013). All sites have similar reef depth (6 m), orientation, slope and substrate type. The disturbed sites are, however, closer to the coast (300 m) and to Puerto Madryn city (3 km; hence the greater usage by diving companies) than the control sites (1 km from the coast and 5 km from the city), but all sites are within 2 km from each other. Two sampling periods before (t1: November; t2: December 2011) and two after (t3: April; t4: May 2012) the season of highest number of divers (January–February; see Introduction) were chosen. A non-destructive sampling methodology was used to avoid removing living organisms and degrade natural sites. Ten 25 × 25 cm quadrats were photographed at each site and time using an underwater digital camera equipped with a frame suitable to take all images at the same distance from the substrate. Quadrats were at least 2 m apart. Photographs were analysed using the free software Coral Point Count with Excel extensions (CPCe v4.1; Kohler and Gill, 2006). One hundred equidistant points were placed over each image and the percentage cover of benthic organisms was quantified using a point-intercept method. All organisms were identified to the lowest taxonomic level possible (42% of taxa to species, 33%

to genus, and the remainder were either classified at a broader taxonomic level or grouped into green, red or brown filamentous algae).

Benthic assemblage data were analysed using permutational analysis of variance (PERMANOVA) with the PERMANOVA+ extension in Primer v6.1.7 (Anderson et al., 2007; Clarke and Gorley, 2006). Similarity matrices based on Bray–Curtis measure of square-root transformed percentage covers ('community structure', which takes into consideration the identity of taxa and their relative abundances) or on the Jaccard measure, which essentially transforms cover data into presence/absence ('community composition', taking into consideration only the identity of taxa), were generated for the analyses, which used 9,999 permutations of residuals under a reduced model (Anderson et al., 2007). Analyses had four factors: (i) 'Disturbance' (D, fixed, 2 levels: Disturbed [D], Control [C]), (ii) Before *vs* After (BA, fixed, 2 levels: before and after the high diving season in summer), (iii) Site (Si(D), random, nested in D with 2 levels), (iv) Time (t(BA), random, nested in BA with 2 levels before and after). Additionally, differences in multivariate dispersion among levels of Disturbance were tested using PERMDISP before and after the summer season (Anderson et al., 2007). PERMDISP analyses used 9,999 permutations and were done in PRIMER v6.1.7. To visualise multivariate patterns in benthic assemblages, non-metric multi-dimensional scaling (nMDS) was used as an ordination method using PRIMER v6.1.7.

Similarity percentage analyses (SIMPER; Clarke, 1993) were used to determine the taxa contributing most to differences between levels in significant terms involving fixed factors in the PERMANOVA model. Univariate analyses of variance were done on covers of benthic taxa identified as contributing to the top 30% of the dissimilarities among levels of Disturbance. Univariate analyses were also done on bare rock covers as one of the main potential impacts of divers is the disturbance of the benthos, which could create bare rock patches, as well as for the invasive kelp *Undaria pinnatifida*, given its importance as habitat-forming species (Casas et al., 2004; Irigoyen et al., 2011; Raffo and Irigoyen, 2011). Analyses were done using a 4-factor PERMANOVA (factors as above). Prior to analyses, homogeneity of variances was tested using Cochran's *C* test and data were transformed as appropriate (details in Tables). Euclidean distance matrices were calculated for analyses, which used 9,999 permutations of residuals under a reduced model (Anderson et al., 2007). All these analyses were done in PRIMER v6.1.7.

3. Results

Overall, 18 sessile taxa were identified, which included the invasive kelp *U. pinnatifida*, foliose (e.g. *Dictyota dichotoma*), filamentous and coralline (e.g. calcareous red algal crusts) algae, bivalves (e.g. *Aulacomya atra*), anemones (e.g. *Anthothoe chilensis*), ascidians and bryozoans.

Assemblage structure, which takes into consideration the identity of taxa and their relative abundances, differed between the disturbed and control rocky reefs, despite significant variability among sites, particularly for disturbed reefs (Fig. 1a, Table 1a). This difference between disturbed and control reefs was consistent before and after the peak in diving activity (Table 1a). In addition, assemblage structure differed between before and after the diving peak (Table 1a). Multivariate dispersion of assemblage structure was greater at the disturbed reefs than at controls (mean dispersion ± SE, D: 38.5 ± 1.4, C: 25.6 ± 0.9; PERMDISP $F_{1,158} = 58$, $p < 0.0001$), as well as after the diving season (B: 31.8 ± 1.1, A: 42.9 ± 1.3; PERMDISP $F_{1,158} = 44$, $p < 0.0001$; Fig. 1a).

Similarly, assemblage composition (presence/absence only) differed between disturbed and control reefs (Fig. 1b) and this was consistent before and after the diving season (Table 1b). There also was significant variability between sites (Fig. 1b). However, no differences in composition occurred between before and after the peak in diving (Table 1b). In addition, multivariate dispersion of assemblage composition did not differ between disturbed and control reefs (D: 41.9 ± 1.3, C: 39.4 ± 1.3; PERMDISP $F_{1,158} = 2$, $p = 0.22$), but dispersion was generally greater after the diving season (B: 42.9 ± 0.9, A: 47.7 ± 1.1; PERMDISP $F_{1,158} = 12$, $p < 0.01$; Fig. 1b).

The algae *D. dichotoma* and the calcareous red algal crusts were the taxa that contributed most to the observed differences in assemblage structure between disturbed and control reefs (SIMPER, top 30%). *D. dichotoma* did not differ between disturbed and control reefs; rather, there was significant variability between sites and times and a general decrease in covers after the diving season (Table 2a). However, covers of *D. dichotoma* were *ca* 50% smaller in the high disturbance site than in controls, whereas covers in the moderate disturbance site were only *ca* 10–20% smaller than in controls after the diving season, matching those in the high disturbance site immediately after the season only (Fig. 2a). No differences in covers of encrusting red algae were found between disturbed and control reefs, nor between before and after the peak in diving, but there was significant variability between sites (Table 2b). Covers were *ca* 15–25% greater at the high disturbance site than at controls, whereas covers at the moderate disturbance site were only *ca* 5–15% greater than in controls after the diving season, matching those in the high disturbance site in the last sampling time (Fig. 2b).

Finally, there was a significant interaction between disturbance and before/after the peak in diving for covers of the kelp *U. pinnatifida* (Table 3a). *U. pinnatifida* on disturbed reefs increased significantly after the diving season and were 10–50% greater than on control reefs, while no changes in *U. pinnatifida* were observed on control reefs (Fig. 2c, Table 3a). Bare rock was significantly greater on disturbed than on control reefs both before and after the diving season, although this difference was of only *ca* 3–5% (Table 3b). Bare rock coverage was, however, consistently greater in the high disturbance than in moderate disturbance or control sites, with the exception of the first sampling time when covers in the high disturbance site were similar to those in the moderate disturbance site (Fig. 2d).

4. Discussion

We found differences in the benthic assemblage structure and composition among diving sites (STA) visited by contrasting numbers of recreational divers and these differences were consistent before and after the peak of the diving season. Recreational diving appears, therefore, to elicit a press impact on benthic assemblages on these Patagonian rocky reefs. In addition, there appeared to be differences in the response of organisms to the disturbance between the highly disturbed and the moderately disturbed site. Although formal comparisons between these sites would be confounded due to the lack of spatial replication of each disturbance intensity, effects of diving appeared to be greater in the highly disturbed site, which is visited by more than double the amount of divers than the moderately disturbed site. Covers of taxa contributing most to observed differences in assemblage structure, as well as those deemed ecologically important, such as the invasive kelp *U. pinnatifida*, were consistently different in the highly disturbed than in control reefs, reinforcing the model that diving results in a press impact in this system. Interestingly, however, the effect of diving on these benthic taxa at the moderately disturbed site appeared to elicit a pulse impact, *i.e.* covers at this site typically differed from those in controls only immediately after the diving

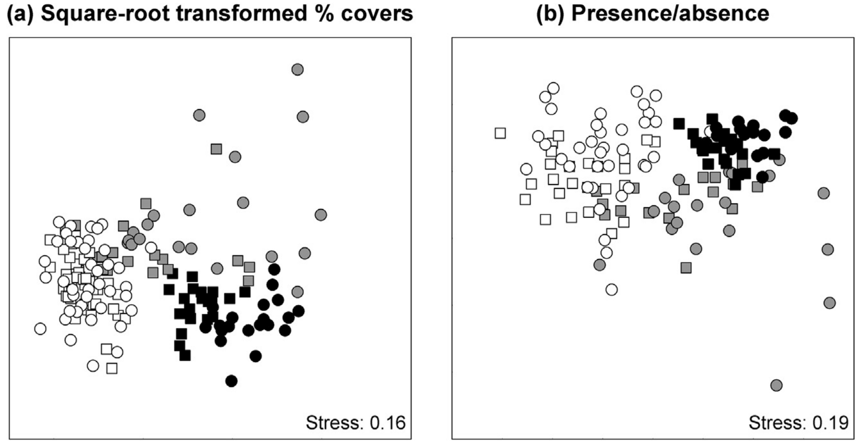

Fig. 1. nMDS based on (a) Bray–Curtis similarities of square-root transformed % covers or (b) Jaccard similarities (presence/absence) of benthic assemblages at highly disturbed (black symbols), moderately disturbed (grey symbols) or control sites (white symbols), two times before (circles) and after (squares) the peak in diving in summer.

Table 1
PERMANOVAs based on (a) Bray–Curtis measure of square-root transformed % covers or (b) Jaccard measure (presence/absence) of benthic assemblages at disturbed (D) or control (C) sites, before (B) and after (A) the diving peak in summer.

Source	*df*	(a) Square-root transformed %covers			(b) Presence/absence		
		MS	Pseudo-*F*	*p*(perm)	MS	Pseudo-*F*	*p*(perm)
BA	1	18472	3.46	0.04	18517	2.67	0.07
D	1	75249	3.72	0.03	73299	3.50	0.03
Time(BA)	2	4495	2.00	0.17	5611	1.96	0.17
Site(D)	2	17175	7.65	<0.01	17810	6.23	0.01
BA × D	1	4745	1.35	0.32	7625	1.66	0.21
BA × Si(D)	2	1498	0.67	0.64	2398	0.84	0.55
D × Ti(BA)	2	3680	1.64	0.24	3924	1.37	0.30
Ti(BA) × Si(D)	4	2245	3.14	<0.01	2859	2.17	<0.01
Residual	144	716			1317		

Table 2
Analyses of % cover of taxa contributing most to the dissimilarity in assemblage structure between disturbed (D) and control (C) sites before (B) and after (A) the peak in diving in summer.

Source	*df*	(a) *Dictyota dichotoma* (arcsin(%) transformed)			(b) Encrusting red algae		
		MS	Pseudo-*F*	*p*(perm)	MS	Pseudo-*F*	*p*(perm)
BA	1	1.68	22.44	0.01	379	4.50	0.08
D	1	7.33	4.67	0.09	6083	3.86	0.10
Time(BA)	2	0.05	0.28	0.76	106	0.77	0.57
Site(D)	2	1.04	6.41	0.06	1515	10.99	0.02
BA × D	1	1.75	3.20	0.15	385	4.90	0.09
BA × Si(D)	2	0.04	0.22	0.81	9	0.06	0.88
D × Ti(BA)	2	0.56	3.46	0.14	98	0.71	0.60
Ti(BA) × Si(D)	4	0.16	3.01	0.02	138	2.37	0.05
Residual	144	0.05			58		
Cochran's *C*		0.16; $p > 0.05$			0.34; $p < 0.01$		

season, after which they seemed to start becoming similar to those in controls. As diving sites seem to respond in different ways to diving depending on the usage, this information is key for developing sound management strategies when implementing STA policies and regulations. Nevertheless, spatially replicated designs are necessary to test this model as observed trends between the two disturbed sites could simply be due to spatial differences rather than differences in diving intensity.

Disturbed and control reefs not only differed in terms of the intensity of diving activities, but also in their proximity to the city. Urbanized coastlines often experience high levels of contaminants and habitats are often modified or replaced by the addition of artificial structures such as marinas or seawalls, which can affect benthic assemblages (Airoldi and Beck, 2007; Bulleri et al., 2005). Disturbed reefs are typically closer to the city and this is arguably the main reason why these reefs are intensively used for diving. Even if the rocky reefs studied may be experiencing some other human disturbances as oil pollution or pollution from the city which may confound the results, it should be noted that the places sampled are far from the high maritime or industry activity. Considering that the disturbed sites have been intensively used for diving activities for at least for the last two decades (Bravo, 2013) and that disturbed and control sites were within *ca* 2 km from each other, it is therefore likely that observed differences in their assemblages are the result of a press impact by recreational diving, and not due to differences in other environmental variables potentially related to the distance from the city.

Interestingly, although diving is not an activity restricted to the tropics, most of the literature on the ecological effects of diving refers to work in tropical sites, being our paper the first done in a temperate environment in Patagonia Argentina. In tropical areas, Hawkins and Roberts (1997) found that diving impact seems to increase when the number of divers exceeds 5000 diver/year/STA. Our highly impacted site typically receives *ca* 3000 divers/year (Bravo, 2013). However, STA in tropical areas are usually larger than those in Patagonia. In fact, the area of the highly disturbed site in our work is about 500 m^2, thus diver load may be higher than in the STA studied by Hawkins and Roberts (1997) in tropical environments.

The different usage by divers in the two STA studied was closely associated with differences in the amount of coverage of the algal species. Disturbed sites were characterized by a greater coverage of calcareous red algal crusts and the invasive kelp *U. pinnatifida*. In controls, instead, greater covers of *D. dichotoma* were observed. There are several studies that showed how encrusting species can establish and colonize environments that have been adversely affected by friction or trampling (Zedler, 1976; Beauchamp and Gowing, 1982; Povey and Keough, 1991). Moreover, the high abundance of *U. pinnatifida* in disturbed habitats suggests that the disturbance of the seabed by divers could play an important role in the establishment and spread of this invasive species (Valentine and Johnson, 2003). In Spain (Ugarte et al., 2006) and Tasmania (Valentine and Johnson, 2003), this species is able to rapidly

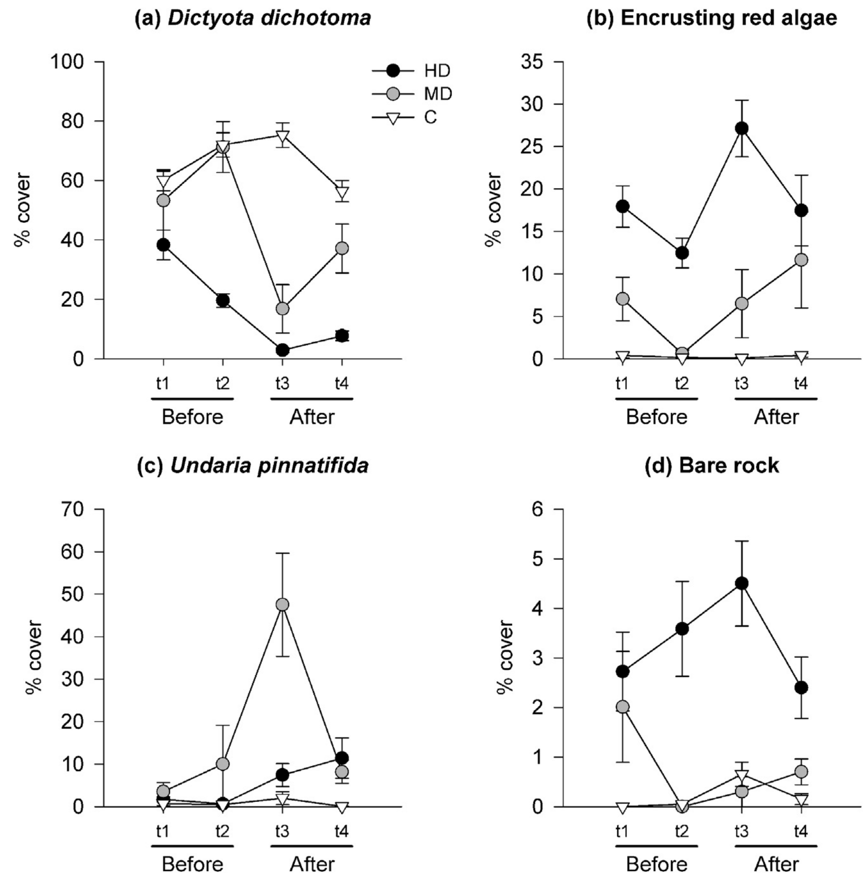

Fig. 2. Mean (±SE) % cover of (a) *Dictyota dichotoma*, (b) calcareous red algal crusts, (c) *Undaria pinnatifida* and (d) bare rock at highly disturbed (black symbols; $n = 10$), moderately disturbed (grey symbols; $n = 10$) or control sites (white symbols; $n = 20$), two times before (t1, t2) and after (t3, t4) the peak in diving in summer.

colonize recently disturbed substrate, while in undisturbed communities its competitive ability decreases markedly. In Patagonia, *U. pinnatifida* has a negative effect on native algae, dramatically reducing its richness and diversity (Casas et al., 2004; Raffo and Irigoyen, 2011). Conversely, its presence positively affects the benthic macroinvertebrate community (Irigoyen et al., 2011), possibly due to the provision of food and shelter, although no native species consumes this alga in high rates (Teso et al., 2009). Due to the important ecological role of the species, the effects of diving on *U. pinnatifida* populations should be explored in more detail.

Table 3
Analyses of % cover of (a) the kelp *Undaria pinnatifida* and (b) bare rock between disturbed (D) and control (C) sites, before (B) and after (A) the peak in diving in summer.

Source	*df*	(a) *Undaria pinnatifida* (ln(X+1) transformed)			(b) Bare rock		
		MS	Pseudo-*F*	*p*(perm)	MS	Pseudo-*F*	*p*(perm)
BA	1	22.34	5.30	0.07	198	1.01	0.49
D	1	52.26	14.02	0.02	10029	8.65	0.03
Time(BA)	2	3.40	1.32	0.36	355	1.13	0.41
Site(D)	2	3.46	1.34	0.35	839	2.67	0.18
BA × D	1	21.85	13.90	0.01	38	0.69	0.63
BA × Si(D)	2	1.30	0.51	0.64	155	0.49	0.64
D × Ti(BA)	2	0.45	0.18	0.84	357	1.13	0.41
Ti(BA) × Si(D)	4	2.58	2.98	0.02	314	4.03	<0.01
Residual	144	0.86			78		
Cochran's *C*		0.16; $p > 0.05$			0.21; $p < 0.01$		
Pairwise tests		B: D = C; A: D > C					
		D: B < A; C: B = A					

Finally, the presence of patches of bare substrate possibly are a result of the friction of the dive gear, which may reduce the area susceptible to colonization by benthic organisms and have negative effects on invasive algal survival (Airoldi and Cinelli, 1997; Irving and Connell, 2002).

Our results show that the intensive use of STA by divers may lead to differences in the distribution and abundance of species in the benthic community. The changes in the community could be mediated by direct mechanic disturbance, but also by complex cascade effects among the species affected. Most importantly, although diving activities are mainly concentrated within two months of the year, they can have long-term effects, particularly in areas where numbers of divers are high. Management guidelines should therefore include a limit in the total number of divers per STA and also enhance the environmental education of divers (e.g. during briefings). These kinds of recommendations have already been proposed for diving areas in Patagonian rocky reefs (Venerus, 2006). However, implementation of recommendations is essential in order to conserve local biodiversity and avoid environmental degradation. Understanding per capita effects in diving areas of Golfo Nuevo is therefore a necessary next step towards providing sound information for management and conservation of Patagonian rocky reefs.

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