



Do recreational activities affect coastal biodiversity?



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ABSTRACT

Human activities are largely affecting coastal communities worldwide. Recreational perturbations have been overlooked in comparison to other perturbations, yet they are potential threats to marine biodiversity. They affect coastal communities in different ways, underpinning consistent shifts in fish and invertebrates assemblages. Several sites were sampled subjected to varying effects by recreational fishermen (low and high pressure) and scuba divers (low and high) in an overpopulated Atlantic island. Non-consistent differences in ecological, trophic and functional diversity were found in coastal communities, considering both factors (“diving” and “fishing”). Multivariate analyses only showed significant differences in benthic invertebrates between intensively-dived and non-dived sites. The lack of clear trends may be explained by the depletion of coastal resources in the study area, an extensively-affected island by overfishing.

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1. Introduction

The world population, approx. 7.2 billion in 2013, is steadily increasing, especially since the Industrial Revolution (*ca.* 18th century), and approx. four-fold in the last century (Bongaards, 2009); numbers are expected to rise by 50% in this century (Steck, 2014). This increase is not homogeneous across the globe, with a growing proportion of the world’s population in coastal regions which is expected to reach 75% by 2025 (Connelly, 2008). Hence, the magnitude of human pressure is becoming larger on the coasts worldwide (Mora et al., 2011). The most important threat to the global ecology and biodiversity is being driven by human-induced stressors such as overfishing, pollution, habitat loss and invasive species (Crain et al. 2008). These perturbations are triggering global shifts on the whole planet, extensively known as global climate change, resulting in sea warming, ocean acidification and anoxia (Mora and Zapata, 2013). However, other human activities have been neglected in recent decades, yet they consistently affect coastal ecosystems (Paudel et al., 2011). This phenomenon occurs to recreational activities despite environmental evidences

are considerable at many sites worldwide (Milazzo et al., 2002; Sala et al., 1996).

Recreational activities are widespread on temperate and tropical touristic regions, especially scuba diving and fishing throughout the last decades (Font and Lloret, 2014; Claudet et al., 2010; Luna et al., 2009; Di Franco et al., 2009; Milazzo et al., 2005). Scuba diving is an important and growing component of the international tourism market, and is heavily reliant upon natural marine areas (Davis and Tisdell, 1995), looking for areas with large fish biodiversity. Unfortunately, scuba diving may degrade coastal ecosystems (Hawkins et al., 1999), especially fragile sessile communities, *i.e.* corals or sponges that are easily damaged by physical contact (Uyarra and Côté, 2007). Scuba divers and snorkelers can even trigger changes in coastal communities from feeding activities (Milazzo et al., 2005), with shifts on fish behavior in coastal communities (Milazzo et al., 2006).

Recreational fishing mainly involves angling by hook and line, and spearfishing (McPhee et al., 2002), with similar ecological consequences on fish populations that are commercially fished (Lewin et al., 2006). They range from direct impacts on the commercial-interest species to perturbations on the whole coastal ecosystem (Lloret and Font, 2013). Recreational fishing as a rule is open-access, *i.e.* with no restrictions, and the effort is increasing

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throughout the world (Ihde et al., 2011). In several coastal areas, recreational fishing is becoming, and is even now, the dominant amount of fish catches (Font and Lloret, 2014). These effects are particularly accentuated in overfished areas that are subjected to a high human coastal pressure (Tuya et al., 2004), as it occurs in heavily populated places (e.g. the Canary archipelago).

The present study was conducted in Tenerife (Canary archipelago, NE Atlantic Ocean), an overcrowded island with massive touristic resorts in several coastal areas, receiving over 4,500,000 tourists during 2012 (ISTAC, 2013). A high pressure from a range of coastal recreational activities (angling, boat fishing and spear fishing) occurs along the coast, being especially important close to coastal settlements (Pascual et al., 2012). Similarly, scuba divers are spatially-condensed in a limited number of coastal places because of their coastal accessibility, state of conservation, wildlife variety or geological formations (Hanquet, 2014). Surprisingly, scarce information exists about the environmental consequences of recreational activities on coastal biodiversity in the Canary archipelago.

The present study analyzed differences in coastal marine biodiversity (e.g. fishes and benthic invertebrates) between zones with high pressure of human activities (fishing and/or scuba diving) and other areas with a low level of anthropogenic disturbance or sites considered *a priori* as pristine or not-altered, characterized by occasional perturbations from recreational users. In addition, these were addressed to determine if these differences are noticeable at species, trophic or functional traits level.

2. Material and methods

2.1. Study area

This study was carried out in Tenerife, Canary archipelago (28° NE Atlantic Ocean) (Fig. 1). Tenerife is an ideal location for our study, because of the high number of tourists interested in the marine realm, with some very intensively-dived sites (hereafter 'dived' sites) alongside low intensity and un-dived areas (hereafter 'non-dived' sites). This classification was based on a scuba guide from the Canary Islands (Hanquet, 2014). The division between both much frequented (hereafter 'fished' sites) and less frequented (hereafter 'non-fished' sites) was based on polls conducted by a social study (Pascual et al., 2012). During the study period (March–May 2013) a total of 20 sites were sampled on the SE and W coasts of the island, with similar numbers of low/high levels of diving sites and low/high levels of recreational fishing sites (Fig. 1, Table 1). At each site, four replicates (transects) at the same depth were carried out by divers.

2.2. Sampling methodology

Underwater census protocols followed RLS (Reef Life Survey) procedures, described in detail by Edgar and Stuart-Smith (2009) and Edgar et al. (2009). In brief, the method involves divers laying 50 m transect lines at each site. The number and size of all fishes sighted within 5 m of the line during a swim up each side of the transect line were recorded. Size classes of fish used were 25, 50, 75, 100, 125, 150, 200, 250, 300, 400, 500, 625 and >625 mm total length. Larger fish were individually estimated to the nearest 125 mm.

Mobile invertebrates and cryptic fishes were surveyed along 2 adjoining 50 m-long transects in a 1 m swath to one side of the transect line. Within each 50 m transect, a diver swam along the seabed carefully searching for all echinoderms, crustaceans, mollusks and other minor taxa (flatworms, etc.) larger than 1 cm within 1 m of the transect line without moving rocks or any structure (e.g. shells, residues) from the seabed.

Fishes were grouped into five trophic guilds: Benthic invertevore, Browsing herbivore, Higher carnivore, Planktivore and Scraping herbivore. Invertebrates were also grouped in the following trophic guilds: Herbivore, Benthic carnivore, Omnivore, Detritivore, Planktivore and Sedimentivore.

2.3. Functional traits

The functional strategy of each fish was described using seven categorical traits based on locomotion and feeding of species that are the main key to determining their role within marine assemblages (Bellwood et al., 2006; Villéger et al., 2011).

The *body shape* is a single factor at a lower level contributing to multiple traits at a higher such as swimming, searching for food, striking and capturing prey, evading predators, migration, courtship dances, defending territories, spawning or burrowing (Walker, 2010). Species were placed in 6 groups: fusiform, elongated, oblong, oval, symmetrical and asymmetrical flatfish (Farré et al., 2013). The *swimming* is the main way to avoid and survive of the attack of predators, as well as to obtain food (Plaut, 2001) and is associated to BCF (body and/or caudal fin) and MPF (median and/or paired fin) (Sfaskisotakis et al., 1999). According to our species, we differentiated nine categories: anguilliform, rajiform, subcarangiform, carangiform, ostraciform, balistiform, labriform and tetradontiform. The *mobility* determines energy needs separating mobile species from sedentary species (Norman and Jones, 1984). Three categories were defined (Wiedmann et al., 2014): sedentary or territorial, roving, and highly mobile or migratory.

The *diet* provides information about feeding links (Wiedmann et al., 2014; Mouillot et al., 2014). Each species was assigned to one of ten trophic groups proposed by Bellwood et al. (2004) and Ferreira et al. (2004): macrocarnivores, strict piscivores, mobile benthic invertivores, sand invertivores, colonial sessile invertivores, diurnal planktivores, nocturnal planktivores, scrapers, macroalgae browser and general omnivores. In our study, the most species were already classified by Halpern and Floeter (2008). The *fish size* determines energy needs per unit of body mass and constrains prey–predator relationships because mouth gape scales with body size (Scharf et al., 2000; Mouillot et al., 2014). It represents a key functional trait because size strongly influences the physiological, behavioral, and population ecology of organisms (Naisbit et al., 2011). We used five ordered categories (Farré et al., 2013): small (<10 cm), small-medium (>20 cm), medium (>30 cm), medium-large (>40 cm) and large (>80 cm).

The level in the water column occupied by fish or *habitat* (pelagic, benthopelagic and benthic) is critical for determining fish ecological niche as it influences the set of potential prey available and fish impacts on nutrient transfer between vertical strata (Mouillot et al., 2014; Wiedmann et al., 2014). And finally, the living strategy also influences the success of predator or prey, and the *burying ability* sows the ability to hide (Farré et al., 2013).

The functional diversity was estimated from a multidimensional approach using three functional diversity indices (Mason et al., 2003; Villéger et al., 2008): functional richness (FRic), functional evenness (FEve) and functional divergence (FDiv). The functional richness was assessed through the functional dispersion index (FDis) (Laliberté and Legendre, 2010), which corresponds to the mean distance of a 'species' to the centroid of the community in the community trait space. FDis accounts for not only the trait space filled by a community (convex hull volume), but also dispersion and species relative abundance (Laliberté and Legendre, 2010; Koehn et al. 2014). FEve may be seen as the degree to which the biomass of a community is distributed in niche space to allow effective utilisation of the entire range of resources available to it. Assuming resource availability is even throughout niche space,

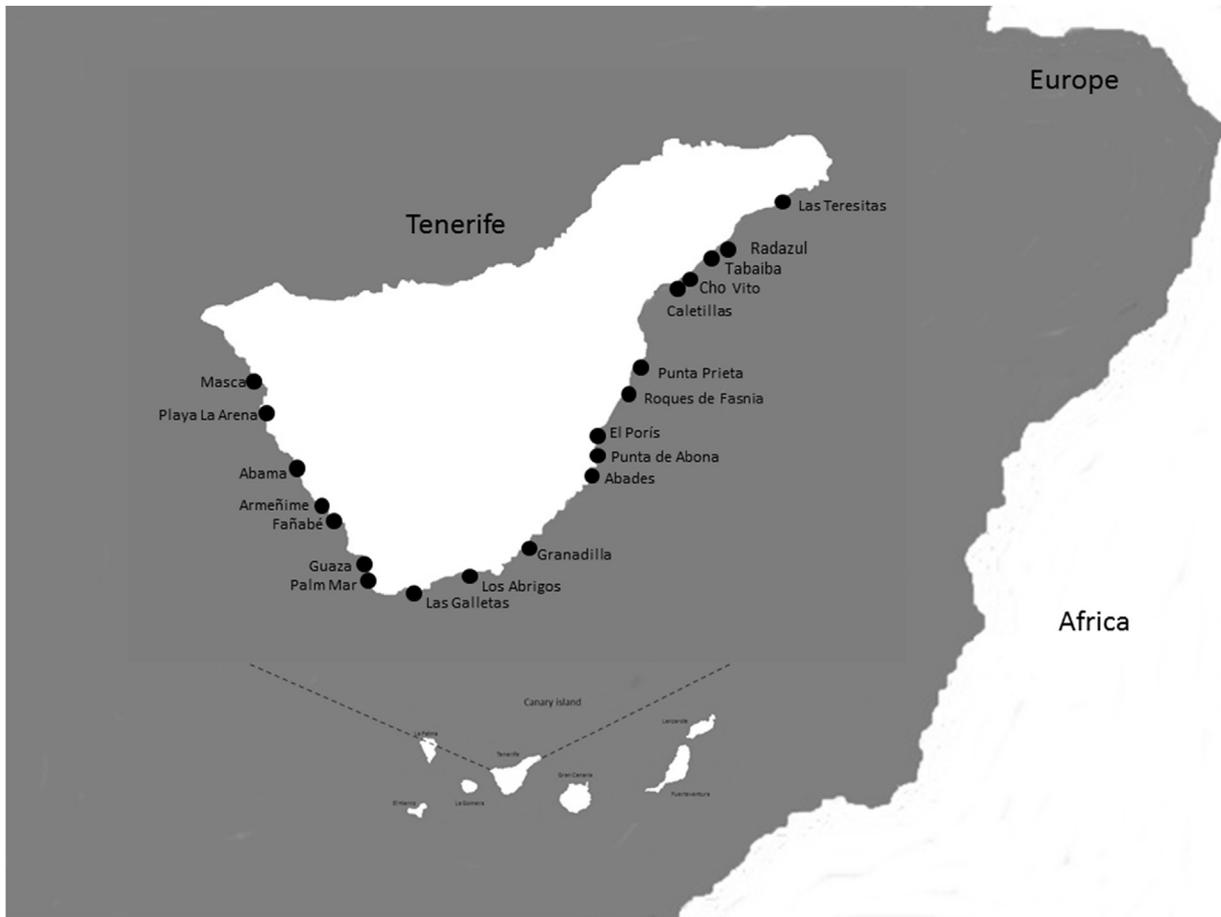


Fig. 1. Map of the study area showing sampling sites.

Table 1
Coordinates, depth, fishing and diving pressure of sampling stations.

Locality	Coordinates		Depth (m)	Fishing	Diving
	X	Y			
Playa La Arena	28,229	−16,842	4	Low	Low
Abama	28,170	−16,805	5	Low	Low
Fañabé	28,089	−16,743	5	Low	Low
Masca	28,284	−16,863	5	Low	Low
Roques de Fasnía	28,221	−16,412	3	Low	Low
Las Teresitas	28,508	−16,181	6	Low	High
Armeñime	28,508	−16,181	4	Low	High
Palm Mar	28,026	−16,705	5	Low	High
Radazul	28,402	−16,327	8	Low	High
Guaza	28,042	−16,711	5	High	Low
Los Abrigos	28,028	−16,595	4	High	Low
Punta de Abona	18,153	−16,430	5	High	Low
Granadilla	28,088	−16,489	5	High	Low
Cho Vito	28,381	−16,356	4	High	Low
Las Caletillas	28,384	−16,351	5	High	Low
Las Galletas	28,007	−16,661	6	High	High
Abades	28,142	−16,439	3	High	High
El Porís	28,164	−16,428	6	High	High
Punta Prieta	28,275	−16,383	7	High	High
Tabaiba	28,401	−16,331	7	High	High

lower functional evenness indicates that some parts of niche space, whilst occupied, are under-utilised (Mason et al. 2003). Finally, FDiv defines how far high species abundances are from the centre of the functional space.

2.4. Statistical methods

To examine patterns of fish and invertebrates assemblage structure among the sampled sites, n-MDS procedure was conducted based on Bray-Curtis similarity of $\log(x+1)$ transformed data; the MDS was used to visualize data spatially. Only centroids (i.e. means) for each coastal locality were plotted to facilitate visualization of multivariate patterns in the ordination space. Differences in community structure (fishes and invertebrates) were tested by means of a permutational MANOVA (PERMANOVA) (Anderson et al. 2006). The factors considered were 'Fishing' (fished and non-fished) and 'Diving' (dived and non-dived), both fixed factors. In a univariate context, a *t*-student test was used to test differences separately in each of the two considered factors: 'Diving' and 'Fishing'.

3. Results

3.1. Fishes

A total of 17,046 individuals were counted, belonging to 42 species. Only four species made up 84.5% of the overall abundance, being the bogue (*Boops boops*) the most abundant (6527 ind), followed by the canary damsel (*Similiparma lurida*) with 3178 ind. Most of the fish (33 species) were scarce (<1%). Community descriptors (i.e. species richness and individual abundances) not showed significant differences considering each factor separately ('Diving' and 'Fishing') and their interaction (Table 2).

Table 2
Results of ANOVA for differences in univariate parameters (richness and abundance) considering fishing (“Fishing”, fixed factor) and diving (“Diving”, fixed factor) pressure in coastal sites.

	Source of variation		df	SS	MS	F	p
Fishes	Richness	Diving	1	2.262	2.262	0.272	0.609
		Fishing	1	1.313	1.313	0.158	0.696
		Diving × Fishing	1	0.082	0.082	0.01	0.922
		Residual	16	132.687	8.304		
	Abundance	Diving	1	64.328	64.328	1.804	0.198
Fishing		1	9.482	9.482	0.266	0.613	
Diving × Fishing		1	11.328	11.328	0.318	0.581	
		Residual	16	570.467	35.654		
Invertebrates		Richness	Diving	1	2.482	2.482	0.325
	Fishing		1	0.328	0.328	0.043	0.838
		Diving × Fishing	1	0.328	0.328	0.043	0.838
		Residual	16	122.133	7.633		
	Abundance	Diving	1	0.552	0.552	0.069	0.797
Fishing		1	1.624	1.624	0.202	0.659	
Diving × Fishing		1	2.74	2.74	0.341	0.567	
		Residual	16	128.589	128.589		
Fished species		Richness	Diving	1	0.221	0.221	0.605
	Fishing		1	0.746	0.746	2.038	0.173
		Diving × Fishing	1	0.138	0.138	0.377	0.548
		Residual	16	5.857	5.857		
	Abundance	Diving	1	1.29	1.29	1.471	0.243
Fishing		1	0.022	0.022	0.026	0.875	
Diving × Fishing		1	1.666	1.666	1.899	0.187	
		Residual	16	14.034	14.034		

Fish composition did not differ consistently in both comparisons (fished vs non-fished sites; dived vs non-dived sites) (Fig. 2A), as it occurs considering only commercial-interested fish species (Table 3). Fished, i.e. commercial-interested, species showed similar species richness and individual abundances between dived and non-dived sites, as well as, fished and non-fished sites. The interaction of both factors (‘Diving’ and ‘Fishing’) did not show consistent differences (Table 2). Community composition showed no consistent differences considering both factors (Table 3).

3.2. Invertebrates and cryptic fishes

A total of 2162 individuals were counted, belonging to 21 species. Nine species overwhelmingly dominated the invertebrate and cryptic fish community, with 99% of the overall abundance. Non-consistent differences were found considering the species richness and individual abundances between factors (‘Diving’ and ‘Fishing’). No differences were neither observed in the interaction between them (Table 3).

Invertebrates and cryptic fish communities showed only significant differences between dived sites and non-dived sites (Table 3), but no consistent trends were observed considering both factors: ‘Diving’ and ‘Fishing’ (Fig. 2B). These differences were mainly explained by significant higher abundances of the holothurian *Holothuria sanctori* in dived sites (One-way ANOVA, $F = 5.007$, $p = 0.038$), as well as consistent higher abundances of the sea urchin *Diadema africanum* (One-way ANOVA, $F = 5.515$, $p = 0.03$) and the crab *Percnon gibbesi* (One-way ANOVA, $F = 5.042$, $p = 0.038$) in non-dived sites (Fig. 3).

3.3. Trophic diversity

Non-consistent trends between dived and non-dived sites, even between fished and not-fished sites, were found in the trophic fish structure in the sampling stations (Table 2). Similar results were obtained with the trophic structure of benthic invertebrates and cryptic fishes (Table 3). Thus, no significant differences in trophic structure were found considering ‘diving’ (dived vs non-dived sites)

and fishing (fished vs non-fished sites).

3.4. Functional diversity

None of the four FD indices (FEvenness, FDiversity, FDissimilarity and FRichness) showed significant differences between dived and non-dived sites (Student-*t* test, $p > 0.05$) (Table 4). Same results were obtained considering fished and non-fished sites (Student-*t* test, $p > 0.05$) (Table 4).

4. Discussion

We attempted to test if ecosystem differences are found between sites with different coastal pressure from recreational activities (fishing and scuba diving), using fish and invertebrates as surrogates of the whole coastal biodiversity. Non-consistent differences were found among all sites (fished vs non-fished sites, dived vs not-dived sites) considering the whole community, as well as descriptors such as individual abundance and species richness. However, if sampling methods (method 1: fishes; method 2: invertebrates; method 3: commercial-interested fish species) are analyzed separately some trends were observed, e.g. invertebrates were consistently more abundant in dived sites, mainly due to the lower abundances of the sea urchin *Diadema africanum* in dived areas, and to a lesser extent, by the crab *Percnon gibbesi*. However, the opposite pattern was observed in the holothurian *Holothuria sanctori*, with higher abundances in dived sites. The remaining biodiversity metrics used (trophic diversity and functional diversity) showed no consistent trends between sites with different coastal pressure, i.e. fished vs non-fished sites and dived vs non-dived sites.

A significant negative correlation has been found between diving activity and coastal biodiversity in coral reefs (Hawkins and Roberts, 1992; Hawkins et al., 1999; Tratalos and Austin, 2001). Evidence of cumulative effects of the disturbances by dives can produce significant localized destruction of sensitive benthic organisms (Barker and Roberts, 2004; Di Franco et al. 2010). Divers can easily damage marine organisms through contact with their

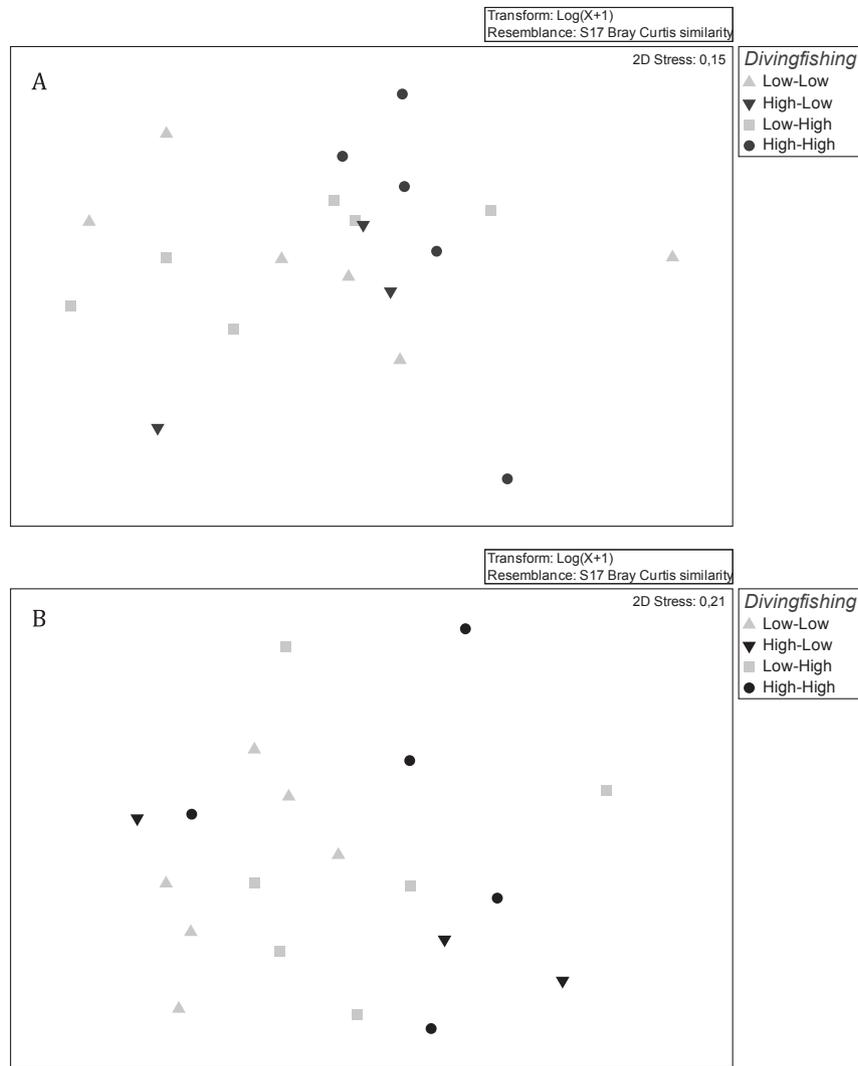


Fig. 2. MDS of sampling sites showing assemblage composition. (A) Fishes, (B) Benthic invertebrates and cryptic fishes.

hands, body or equipment (Uyarra and Côté, 2007). Even the pelagic diversity can be affected by feeding practices conducted by scuba divers (Milazzo et al., 2006). Feeding might alter the natural composition of the reef fish community, and indirectly it might affect coral and other benthic sessile communities (Hawkins et al., 1999). Even if feeding practices are not conducted, fish have been observed to learn patterns with the continuous presence of scuba divers, following them and wait for any contact of the diver with the substratum (Di Franco et al., 2013). In high-dived sites, a large range of individuals show interest for divers and approach them during the dive, looking for any opportunity to catch prey or food, especially abundant are fast-growing species, such as the ornate wrasse (*Thalassoma pavo*) (pers. obs.).

Recreational fishing has the potential to negatively affect fish and fisheries (Cooke and Cows, 2004). This activity is wide-spread worldwide and as a rule is open-access, with no restrictions on the number of anglers, catches, etc. (McPhee et al., 2002). Recreational fishing affects in several ways: (i) the pressure on vulnerable or threatened species, not only fishes; (ii) the reduction of stocks, with special emphasis in adult-size individuals with a large reproductive potential and (iii) the impact of lost fishing gear at fishing sites, among others (Font et al., 2012). In many places, recreational fishing is the dominant source of fishing mortality, as currently

occurs in the Canary archipelago, where catches exceed that of commercial fishing (Pascual et al., 2012). Thus, their effects should not be underestimated, since it has been accentuated in recent decades worldwide (Ihde et al., 2011).

Low densities of the sea urchin *Diadema africanum* in dived sites may be due to the control of their populations conducted by divers. The crab *Percnon gibbesi* was also observed in low densities at dived sites but apparently this trend may be explained by small-scale variability within transect. The opposite pattern, high densities at dived sites, was found with the holothurian *Holothuria sanctori*, and as it occurs to the crab *P. gibbesi*, seabed characteristics (e.g. rugosity, anfractuosity, among others) may be pivotal to the ecology of this species.

The present study clearly suggests that both recreational activities (diving and fishing) do not induce overall significant effects on coastal fish and invertebrates assemblages, as well as their ecological, trophic and functional diversities. The lack of consistent trends between high pressure (fishing and/or diving) and low-pressure sites might be explained by the impoverished state of coastal marine ecosystems in the Canary archipelago. The present results may be explained by the extensive overfishing that occurs in the Canary archipelago. This situation could mask other environmental affections such as, recreational coastal activities (e.g. diving

Table 3
Results of PERMANOVA for differences in community composition considering fishing (“Fishing”, fixed factor) and diving (“Diving”, fixed factor) pressure in coastal sites. Significant differences ($p < 0.05$) are highlighted in bold.

	Source of variation	df	SS	MS	Pseudo-F	p
Fish	Fishing	1	664.03	664.03	0.537	0.833
	Diving	1	1071.1	1071.1	0.867	0.546
	Fishing × Diving	1	12,226.5	1226.5	0.992	0.451
	Residual	16	19,679	1235.6		
	Total	19	22,783			
Invertebrates	Fishing	1	1895.7	1895.7	1.186	0.297
	Diving	1	3013.5	3013.5	1.886	0.045
	Fishing × Diving	1	1912.4	1912.4	1.197	0.286
	Residual	16	25,563	1597.7		
	Total	19	32,677			
Fished species	Fishing	1	1690.5	1690.5	1.264	0.245
	Diving	1	1114.7	1114.7	0.833	0.635
	Fishing × Diving	1	1462.6	1462.6	1.094	0.414
	Residual	16	21,399	1337.4		
	Total	19	25,771			
Trophic diversity fishes	Fishing	1	204.26	204.26	0.719	0.619
	Diving	1	103.52	103.52	0.364	0.841
	Fishing × Diving	1	201.23	201.23	0.709	0.626
	Residual	16	4543	283.94		
	Total	19	4994.1			
Trophic diversity invertebrates	Fishing	1	1567.1	1567.1	2.071	0.099
	Diving	1	773.15	773.15	1.022	0.41
	Fishing × Diving	1	459.31	459.31	0.067	0.66
	Residual	16	12,109	12,109		
	Total	19	14,992	14,992		

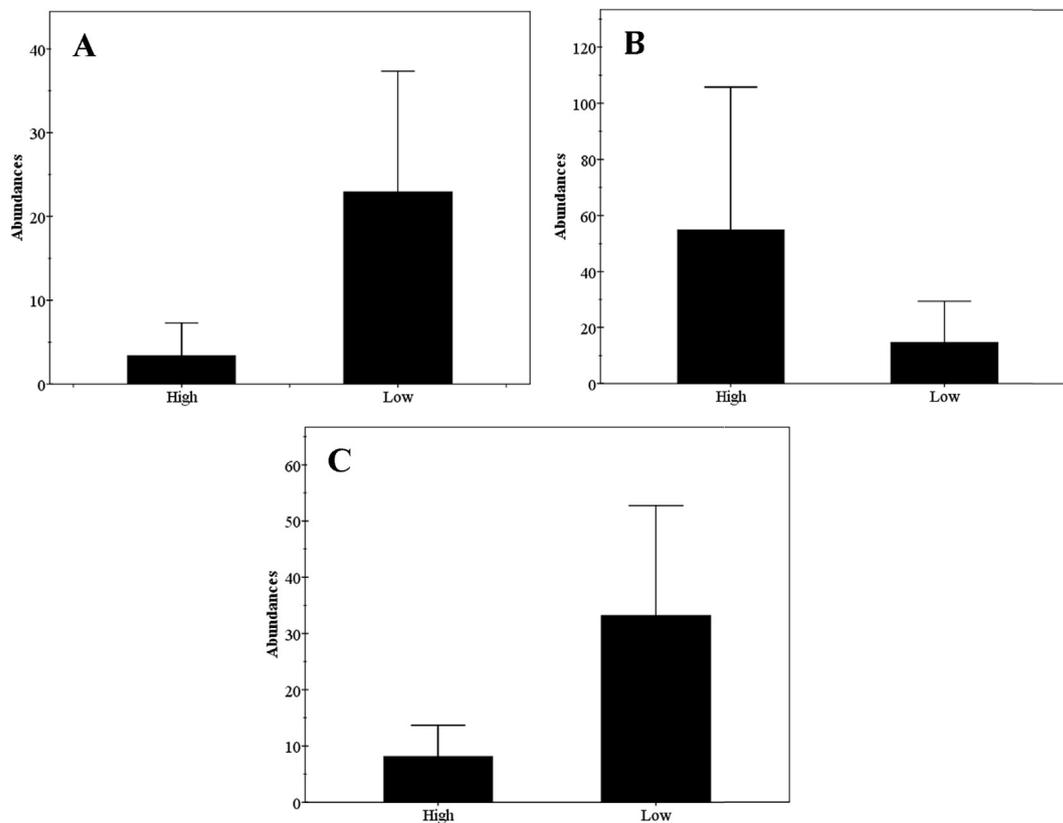


Fig. 3. Abundances of benthic invertebrates in high-dived and low-dived sites. (A) *Diadema africanum*; (B) *Holothuria sanctori*; (C) *Percnon gibbsii*.

and sport fishing) in intensively-used areas. Pristine or sites without any coastal pressure from human activities are rather scarce in the Canary archipelago and only in isolated and scarcely-populated islands, such as the minor islands of El Hierro (the most western island with 8000 inhabitants) and La Graciosa (the most

eastern island with a permanent population of 400 inhabitants). In the last decades, the overall situation of overfishing in the Canary Islands, accentuated in the capital islands (Tenerife and Gran Canaria), has underpinned a huge decrease of fish abundances and species richness in coastal rocky reefs (Falcón et al., 1996). The

Table 4

T-student test of functional parameters considering diving and fishing pressure in coastal sites.

		F	p
Diving	FEvenness	3.229	0.089
	FDiversity	1.94	0.181
	FDissimilarity	0.021	0.888
	FRichness	1.888	0.293
Fishing	FEvenness	0.218	0.647
	FDiversity	0.035	0.854
	FDissimilarity	0.427	0.522
	FRichness	0.082	0.777

opportunistic sea urchin *Diadema africanum* has proliferated, producing sea-barren bottoms characterized by scarce algae coverage and thus, a significant decrease of primary production in shallow coastal ecosystems (Tuya et al., 2004). Moreover, this key species has been observed to be involved in the existence of 'stable alternate state system' (Tuya et al., 2005) which damages the resilience of marine systems with the subsequent establishment of undesired conservation status. Thus, a high resistance from the sea-barrens to ecosystem restoration is expected (Guidetti and Sala, 2007).

The present study is the first attempt to assess the effects of recreational activities (fishing and diving) on several coastal sites on the island of Tenerife. No clear trends were found in fish community structure, as well as diversity measures (ecological, trophic and functional) yet some tendencies were found in certain key species of benthic invertebrates. A more detailed study is necessary in the remaining islands of the Canary archipelago, characterized by high (e.g. Gran Canaria), intermediate (Fuerteventura and Lanzarote) and low (e.g. La Palma, La Gomera and El Hierro) human pressure, in order to determine accurately the effects of these activities in Canarian coastal ecosystems.

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