



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,

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