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Free-diving underwater fish photography contests: a complementary tool for assessing littoral fish communities

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Summary: Characterizing fish communities must be a priority to safeguard resources and determine critical changes. Here, species richness and the spatial and temporal evolution in the structure of fish assemblages were analysed based on photos taken in underwater free-diving contests. A total of 29 contests held from 2008 to 2015 at four different locations along the northeastern Spanish coast, including a marine protected area were analysed. Contests reward the number of species per participant and photographic quality. Species image frequency from each tournament were standardized to catch image rate. A total of 88 taxa were recorded, including 32 cryptobenthic species, the highest number recorded in the Mediterranean littoral system so far. Cluster analyses yielded four major groups. Catch image rates in the marine protected area were significantly higher for seven species of high commercial interest and for two big labrids of recreational interest, including an endangered species (*Labrus viridis*). Overall, the study showed that photographic free-diving contest data are a potential tool for determining species richness in littoral systems since contest rules promote competition between participants to obtain maximum fish diversity. We believe that this type of cost-effective data can be applied worldwide as a complementary way of monitoring littoral fish assemblage.

Keywords: littoral fish assemblages; species richness; diversity; photography contests; Mediterranean Sea.

Competiciones fotográficas de peces en buceo libre: una herramienta complementaria para la valoración de las comunidades de peces litorales

Resumen: La caracterización de las comunidades de peces debe ser una prioridad para salvaguardar los recursos y para la detección de cambios críticos. La información de las competiciones fotográficas a buceo libre se utilizó para analizar la riqueza específica y las variaciones espaciales y temporales de la comunidad íctica. Se analizaron un total de 29 concursos celebrados entre el 2008 y el 2015 en cuatro puntos de la costa Noroeste de España, incluida un área marina protegida. Estas competiciones premian a los participantes por el número de especies fotografiadas y su calidad. En cada concurso se estandarizó la frecuencia de imágenes por especie, convirtiéndola a la tasa de imágenes capturadas. Se registraron 88 taxones, incluidas 32 especies criptobénticas, el mayor número registrado hasta la fecha en el litoral Mediterráneo. El análisis de conglomerados identificó cuatro grupos principales. La tasa de imágenes capturadas de siete especies comerciales fue superior en el AMP y también superior para dos especies de lábridos de interés recreativo, una de ellas considerada como especie amenazada (*Labrus viridis*). En suma, el estudio mostró que los datos de los concursos fotográficos a buceo libre son una herramienta potencial para determinar la riqueza específica en los sistemas litorales debido a que las competiciones fomentan la competencia entre los participantes para obtener la máxima diversidad de especies. Consideramos que estos datos, eficaces en los costes, se pueden utilizar a nivel mundial para complementar los sistemas de seguimiento de las comunidades de peces litorales.

Palabras clave: comunidades de especies litorales; riqueza específica; diversidad; concursos fotográficos, mar Mediterráneo.

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INTRODUCTION

The marine coastal environment is threatened both locally and globally in different ways. In Europe the coastal habitats have been progressively degraded over the last few decades (e.g. Benedetti-Cecchi et al. 2001, Lotze et al. 2006, Claudet and Fraschetti 2010). Future projections are not encouraging: The Mediterranean Action Plan predicts that the urban population of the coastal Mediterranean could reach 176 million by 2015 plus 350 million tourists yearly. Furthermore, an increase in the number of introduced species (Boudouresque et al. 2005) and climate-induced stressors are contributing to changes in the Mediterranean biodiversity (Fraschetti et al. 2011), emphasizing the need to foster the surveillance of coastal systems.

Maintaining diversity in coastal systems is essential for the sustainability of communities, ecosystem functioning and services (Pickaver 2009, Costanza et al. 2014). Changes in coastal ecosystems can be one-off events and have dramatic effects (i.e. artificial modifications of the coast or severe storms), or rather progressive and relatively slow (at the scale of human life), and are difficult to notice through snapshot studies. Moreover, the negative impact on them may be difficult to document (Støttrup 2009). One of the aims of the European Marine Directive is to ensure that biodiversity is maintained, that is, kept in line with the natural state appropriate to the area in question. However, baseline shifts can affect reference sites as much as impact sites (Støttrup 2009), masking variations and distorting perceptions.

In the Mediterranean, littoral fish communities are threatened by numerous stressors, including habitat loss, global warming, changes in the continental water discharges, artificialization of the coastlines, introduction of alien species and fishing pressure (e.g. Guidetti et al. 2002, Claudet et al. 2006). Fish communities are a key component of the aquatic ecosystem (Holmlund and Hammer 1999) as fish provide fundamental services for ecosystem functioning and resilience (Schindler et al. 1997, Vanni 2002, Myers et al. 2007). Moreover, numerous studies support the use of fish assemblages as biological indicators for marine coastal waters (e.g. Sano 2000, Seytre and Francour 2008, Azzurro et al. 2011). Nevertheless, studying and monitoring fish assemblages is as complex as the diversity of groups represented. Littoral fish assemblages include groups of different niches, so different complementary sampling methods are required to capture all the components (Elliott et al. 2002). It is widely recognized that no single assessment technique can provide unbiased, qualitative or quantitative estimates of fish assemblages (Sale and Douglas 1981, Ackerman and Bellwood 2000). This problem was highlighted decades ago (Harmelin-Vivien et al. 1985) but it is still an open issue and it is recommended to apply different sampling methods simultaneously to the same study site (Connell et al. 1998, Willis et al. 2000, Cappo et al. 2004).

Studies of fish populations in shallow littoral waters usually rely on underwater visual censuses (e.g. Harmelin-Vivien et al. 2008, Bussotti et al. 2015, Prato et al. 2017). Video techniques have been increasingly used (Mallet and Pelletier 2014) and are recognized mainly as an additional technique to underwater visual censuses, with stationary (Francour 1999), diver-operated (Tessier et al. 2013) or roving cameras (Tessier and Chabanet 2006), or stationary baited underwater video (Gledhill et al. 1996, Willis and Babcock 2000, Stobart et al. 2007). Fishing techniques are also used as a complementary method in littoral fish research surveys (Franco et al. 2012). The appropriateness of a methodological approach depends on its capacity to fulfil the purposes of the study. However, accurate standardized measurements of fish species richness and community structure are essential for monitoring the progress towards biodiversity targets (Hutchings and Baum 2005), as well as for conservation actions (MacNeil et al. 2008). Underwater visual census methods cannot assume equal detectability across all species, and it is recognized that this methodology under-represents a large number of cryptobenthic fish species (e.g. Smith 1988, Ackerman and Bellwood 2000, Kovačić et al. 2012) because more than 90% of them may go undetected by underwater visual censuses (Willis 2001). In fact, cryptic fish are particularly difficult to accurately survey. They are under-sampled components of fish communities and their ecological roles have generally been ignored (Smith-Vaniz et al. 2006). The use of destructive techniques such as ichthyocides and anaesthetics for cryptic species is controversial but also justified for obtaining reasonably complete inventories of reef fish (Smith-Vaniz et al. 2006, Glavičić et al. 2016, Thiriet et al. 2016).

In addition to research surveys, fisheries-dependent data are also used for fish assemblage monitoring. For all their flaws and lack of scientific methodology, fisheries-dependent data have some advantages: they generally provide large data sets with wide temporal and spatial coverage and they are cost-effective. Furthermore, other cost-effective sources of information are providing good findings in different fields (http://www.observadoresdelmar.es, Fairclough et al. 2014) jointly with citizen science (e.g. www.seawatchers.org, redmap.org.au), which is currently expanding worldwide. Valuable data gathered by public agencies or private entities also have informative potential that is worthy of being analysed. This is the case of recreational fishing tournaments in certain regions, which have extensive spatial coverage and/or temporal continuity, and have been shown to be valuable for characterizing and monitoring littoral fish assemblages (Lincoln Smith 1989, Coll et al. 2004, Gordoa 2009).

Another potential source of information is underwater photo contests. The technological developments in underwater photography have led to the expansion of a new sport activity: free-diving underwater photography. A recent event in underwater fish photography contests rewards the number of species photographed. Participants are given scores according to the number of species photographed, and they therefore search in every possible habitat to obtain the largest possible number of species. Consequently, the species richness

resulting from these tournaments is expected to be higher than when traditional visual survey techniques are used. In Spain, and particularly in Catalonia, these tournaments began in the 1980s and have now become regular. The objective of this study was to assess species richness and classify fish communities based on the underwater free-diving fish photography contests held in Catalonia at four different locations from 2008 to 2015, including a no-take marine protected area (Medes Islands marine reserve).

MATERIALS AND METHODS

Data description

This study analysed the information from the free-diving photography contests held in Catalonia, which was provided by the Catalan Underwater Activities Federation (FECDAS). The objective of these contests is to photograph only of live fish in the natural system while free-diving. The participants aim to photograph the highest number of fish species possible. Each participant can only present one photo per species so the number of photos per participant is equivalent to the number of species per participant. In Catalonia, the first official championship was held in 2005, and since then FECDAS has gathered the data from each contest, which includes the following information: date, location, number of participants, number of species and total number of photographs per species. In this study we only considered the championships held after 2007 when analogue cameras were completely replaced by digital ones.

A total of 29 tournament reports were analysed. Each tournament had on average 18 participants and lasted 5 hours. FECDAS provided us with a copy of the photographic database, which allowed us to validate the records of dubious species appearing in the contest reports.

Site features

The contests were held at four locations along the Catalan coast in northeast Spain (Fig. 1). Annual contests are held at the same locations and provide informative data for studying the spatial and annual variability of littoral fish communities. In addition to their latitudinal differences, the four locations also have different habitat features, and these differences should also be reflected in their fish assemblages.

The Medes Islands, located in the northern area, are a group of islands ~1 km from the mainland coast. This archipelago is one of the oldest marine reserves in the western Mediterranean. It has been protected since 1983 and has become a popular scuba diving destination. The contests were located on the southern side of the main island, at depths ranging from 0 to 20 m and characterized by diverse habitats: rocks, sand, *Posidonia oceanica* meadow and coralligenous. The second location, also in the northern area and near to the Medes Marine Reserve, was a coastal zone of Palamós, Margarida Cove, which has a diverse habitat: rocks, breakwater blocks, sand and *P. oceanica* meadow, at depths ranging from 0 to 15 m. The third location was Mataró, where the contest site was located ~500 m offshore from the coast at depths ranging from 6 to 12 m. The location is restricted to a natural rock barrier that is over 150 m long, has many cracks and cavities, and is surrounded by sandy beds with some *P. oceanica* patches at the eastern extreme. Finally, the fourth location was L'Ametlla, a coastal area at Port d'Estany, which has brackish water of depths between 0 and 10 m with a similar bottom structure: rocks, sand and a large *P. oceanica* meadow.

Data analysis

Species relative abundance data from the tournament information were estimated from the catch image

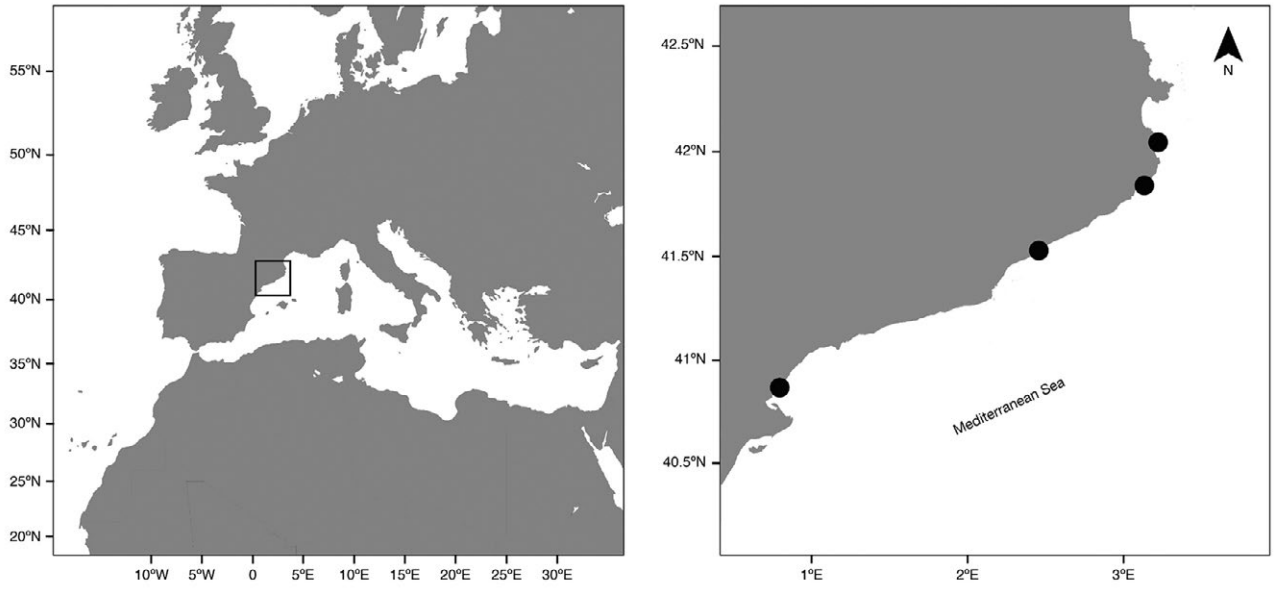

Fig. 1. – Study area and locations of free-diving underwater fish photography. A, Medes Islands; B, Palamós; C, Mataró; and D, Ametlla de Mar.

Table 1. – Descriptors of the free-diving fish photograph contests studied.

Location	Date	No. photos	No. species	No. participants	No. photos/participants
Ametlla de Mar	30/05/2009	574	61	15	38
Ametlla de Mar	16/06/2012	603	62	20	30
Ametlla de Mar	26/05/2013	355	52	13	27
Ametlla de Mar	08/06/2014	478	68	13	37
Ametlla de Mar	31/05/2015	315	67	7	45
Mataró	17/05/2009	452	46	14	32
Mataró	23/05/2010	635	47	24	26
Mataró	22/05/2011	537	47	22	24
Mataró	20/05/2012	337	42	13	26
Mataró	09/06/2013	277	48	9	31
Mataró	18/05/2014	449	53	15	30
Mataró	17/05/2015	289	47	9	32
Medes	18/03/2007	251	49	9	28
Medes	05/04/2008	560	58	20	28
Medes	04/04/2009	650	62	21	31
Medes	17/04/2010	1001	62	34	29
Medes	09/04/2011	931	63	31	30
Medes	22/04/2012	536	57	20	27
Medes	22/06/2013	917	70	22	42
Medes	21/06/2014	1147	78	24	48
Medes	20/06/2015	389	66	9	43
Palamós	27/05/2007	368	52	13	28
Palamós	18/05/2008	627	62	21	30
Palamós	22/03/2009	469	60	16	29
Palamós	07/03/2010	692	54	28	25
Palamós	03/06/2012	669	56	20	33
Palamós	12/05/2013	641	59	18	36
Palamós	04/05/2014	686	62	18	38
Palamós	03/05/2015	472	65	11	43

rate for each location, which was estimated as the total number of reported photos per species divided by the total number of participants. Taxonomic resolution was not fully attained and some species were classified at the level of genera (*Atherina* and *Sphyraena*) or family (Mugilidae).

Absolute number of species richness (S) and Margalef index, $\alpha=(S-1)/\ln(N)$ (N total number of images) were estimated as measures of species richness and diversity for each contest. One-way ANOVAs using STATISTICA 10 (Stat-Soft Inc. 2010) were performed to assess potential differences in species richness and diversity (α) among locations. As the number of samples varies among localities the weighted mean was used in the analysis. The spatial structure and fish community was analysed using the PRIMER software package (Clarke KR and tutorial. PRIMER-E) and agglomerative hierarchical clustering was applied. A community similarity matrix based on the Pearson correlation coefficient was calculated for hierarchical group average linking. Catch rates were not transformed because participants could only present a single photo per species, regardless of the number of photos or encounters they had with the most frequent species. Consequently, the most frequent species were already down-weighted by the contest rules, but the information of species relative frequency was retained because it is equivalent to the probability of being photographed.

A silhouette analysis (Rousseeuw 1987) was performed to evaluate the precision of the hierarchical clustering using the R statistical software (R Development Core Team, 2013). Silhouette analyses were performed to evaluate consistency within the identified clusters. Differences were examined with an analysis of the similarity percentages (SIMPER). This procedure used to identify which species contributed to the resulting clusters. The analysis was complemented by multidimensional scaling (MDS).

One-way ANOVA was used to assess catch image rate differences between locations in the species of commercial interest. Unequal N HSD post hoc tests (multiple comparison test), a modification of the Tukey test for unequal sample sizes (Winer et al. 1991), were used to determine the significant differences to investigate the efficacy of free-diving photographic data to detect the already known effect of this marine protected area (MPA).

RESULTS

In total, 16307 photographs corresponding to 29 different contests were analysed. The mean number of participants per contest was 17 and the average number of photographs per participant was 33. The number of photographs per participant has increased over recent years at two locations, Palamós and the Medes Islands, with no effect on the total number of species recorded (Table 1).

A total of 88 taxa belonging to 26 families were photographed (Table 2). The actual species richness was higher, as the genera *Atherina, Sphyraena, Trachurus* and *Trachinus* and the family Mugilidae were not identified to the species level. The families with the highest number of species were Sparidae with 17 species and Labridae with 15 species, followed by Blenniidae and Gobiidae with 14 and 10 species, respectively. According to the International Union for Conservation of Nature (IUCN) Red List categories and criteria, 2 of these species are not yet evaluated, 1 is data-deficient to assess its status, 80 are on the least-concern list, 3 are considered vulnerable, 1 is near threatened endangered and 1 is already endangered. It is worth mentioning that

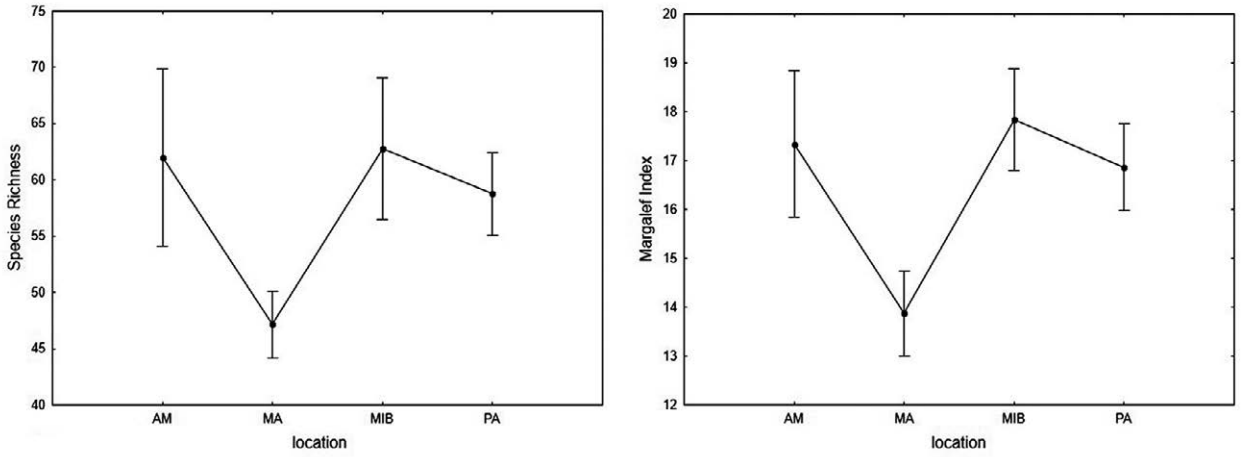

Fig. 2. – Weighted mean and 0.95 confidence intervals of species richness and Margalef diversity index per location. AM, Ametlla de Mar (N=5); MA, Mataró (N=7); MIB, Medes Islands (N=9); and PA, Palamós (N=8).

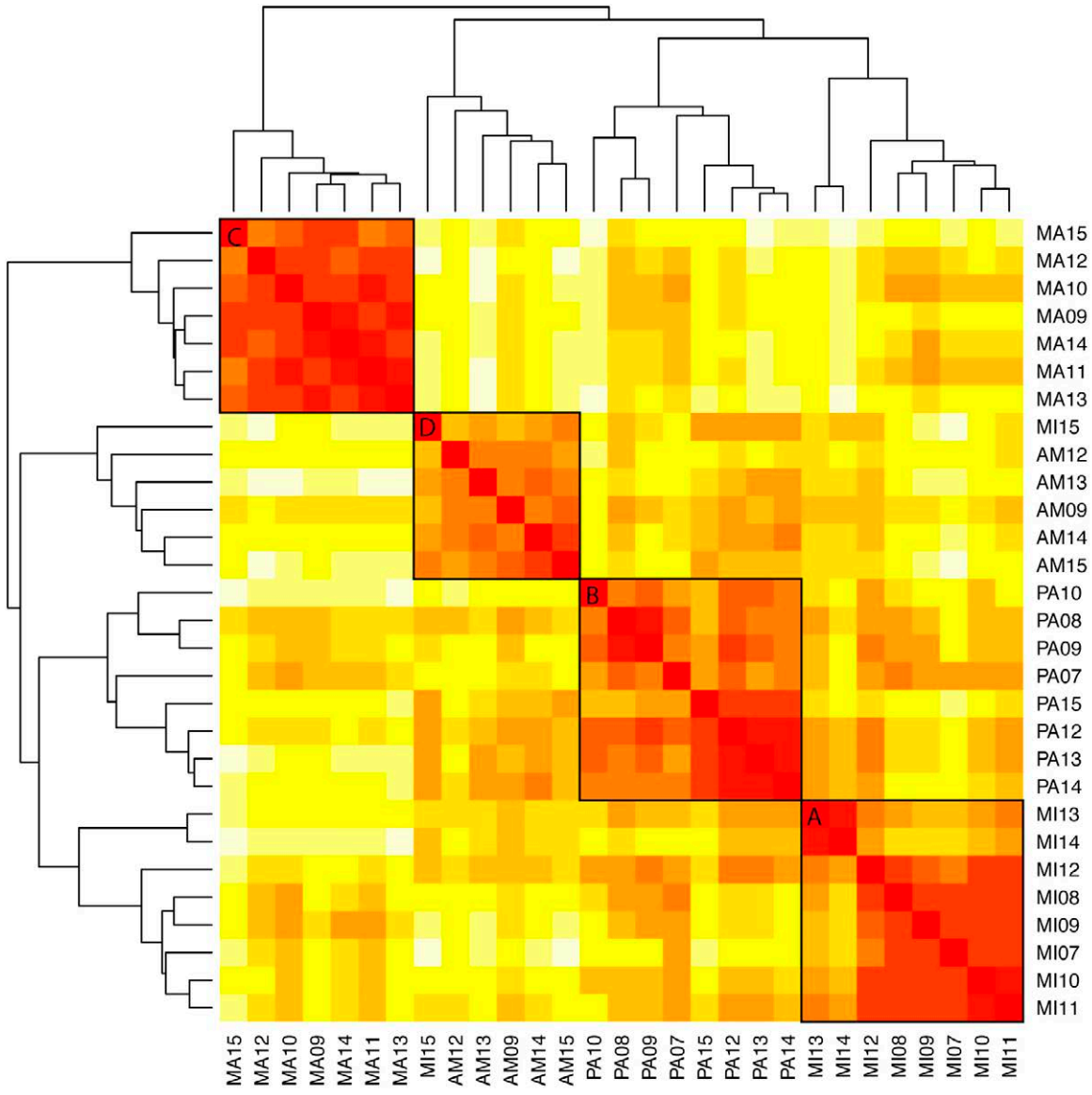

Fig. 3. – Heatmap result of cluster analysis of locations on the basis of species images catch rates. A, B, C and D represent the four main groups identified in the analysis.

Table 2. – Fish species average catch image rate and standard deviation by location. The letters within brackets next to the species correspond to those with a catch image rate significantly higher in the Medes Marine Reserve; each letter corresponds to the location where the significant differences were observed. Asterisks indicate a species described for the first time in the Reserve.

Species	Feeding (trophic levels)	Feeding behaviour	Commercial value	IUCN status	Ametlla		Mataró		Medes Islands		Palamós	
					Mean	Std	Mean	Std	Mean	Std	Mean	Std
Ammodytidae												
*Gymnammomytes cicerelus**	≥2.8	Filtering plankton	Unknown	LC	0.06	0.08	0.20	0.36	0.00	0.01	0.01	0.02
Apogonidae												
Apogon imberbis	≥2.8	Hunting macrofauna	NONE	LC	0.47	0.34	0.54	0.28	0.41	0.38	0.68	0.17
Atherinidae												
Atherina sp.	≥2.8	Hunting macrofauna	Low	LC	0.54	0.25			0.41	0.26	0.68	0.30
Blenniidae					0.47		0.46		0.33		0.37	
Aidablennius sphynx (M)	2.2-2.79	Grazing	NONE	LC	0.31	0.27			0.48	0.33	0.64	0.31
Coryphoblennius galerita	2-2.19	Grazing	None	LC	0.03	0.06			0.05	0.06	0.11	0.15
Lipophrys trigloides	2.2-2.79	Variable	NONE	LC	0.46	0.17			0.26	0.30	0.29	0.27
Microlipophrys canevae	2-2.19	Variable	NONE	LC	0.76	0.17			0.53	0.32	0.32	0.30
*Microlipophrys dalmatinus**	2.2-2.79	Variable	NONE	LC	0.25	0.23			0.12	0.22	0.05	0.06
Microlipophrys nigriceps	2.2-2.79	Variable	NONE	LC	0.29	0.17	0.15	0.18	0.19	0.22	0.02	0.06
Parablennius gattorugine	2.2-2.79	Grazing	Unknown	LC	0.8	0.1	0.45	0.17	0.47	0.25	0.79	0.14
Parablennius incognitus	2.2-2.79	Variable	NONE	LC	0.98	0.21			0.45	0.36	0.49	0.23
Parablennius pilicornis	≥2.8	NA	NONE	LC	0.89	0.13	0.77	0.18	0.80	0.21	0.88	0.19
Parablennius rouxi	2.2-2.79	Variable	NONE	LC	0.04	0.06	0.49	0.3	0.17	0.11	0.09	0.09
Parablennius sanguinolentus	2-2.19	Grazing	Unknown	LC	0.54	0.24			0.32	0.36	0.65	0.30
Parablennius tentacularis	≥2.8	Browsing	NONE	LC					0.02	0.04	0.05	0.09
Parablennius zvonimiri	2.2-2.79	Variable	NONE	LC	0.62	0.13	0.44	0.21	0.44	0.24	0.46	0.28
Scartella cristata	2-2.19	Grazing	NONE	LC	0.11	0.14						
Bothidae												
Bothus podas	≥2.8	Predator	High	LC	0.38	0.26	0.08	0.1	0.04	0.09	0.02	0.06
Callionymidae												
*Callionymus pusillus**	≥2.8	Predator	NONE	LC	0.01	0.02			0.05	0.07		
Carangidae												
Trachurus sp.	≥2.8	Predator	Medium	VU	0.02	0.03	0.08	0.13	0.04	0.07	0.02	0.06
Centracanthidae												
Spicara smaris	≥2.8	Predator	Low	LC					0.02	0.05	0.02	0.04
Clinidae												
Clinitrachus argentatus	≥2.8	Predator	NONE	LC	0.09	0.11			0.02	0.07	0.06	0.13
Congridae												
Conger conger (P)	≥2.8	Predator	Medium	LC	0.19	0.18	0.33	0.18	0.43	0.24	0.14	0.14
Gadidae												
Phycis phycis (A,M,P)	≥2.8	Predator	High	LC	0.03	0.06	0.09	0.22	0.42	0.14	0.04	0.08
Gobiesocidae												
Lepadogaster candollei	2.2-2.79	NA	NONE	N.E.			0.20	0.4	0.26	0.22	0.34	0.27
Lepadogaster lepadogaster	NA	NA	NONE	LC	0.18	0.18			0.15	0.14	0.33	0.32
Gobiidae					0.40		0.23		0.20		0.39	
Gobius bucchichi	2.2-2.79	Variable	NONE	LC	0.97	0.11	0.65	0.31	0.78	0.17	0.94	0.19
Gobius cobitis	2.2-2.79	Variable	NONE	N.E.	0.43	0.35			0.23	0.24	0.52	0.21
Gobius cruentatus	≥2.8	Browsing	NONE	LC	0.41	0.3	0.05	0.12	0.06	0.18	0.35	0.26
Gobius geniporus	≥2.8	NA	NONE	LC	0.24	0.18	0.80	0.16	0.29	0.14	0.49	0.24
Gobius niger	≥2.8	plankton feeding	NONE	LC			0.01	0.03	0.01	0.02	0.09	0.11
Gobius paganellus	≥2.8	Predator	NONE	LC	0.6	0.05	0.03	0.05	0.34	0.31	0.65	0.31
Gobius vittatus	2.2-2.79	Variable	NONE	LC			0.01	0.03	0.03	0.06		
Pomatoschistus sp*	3.2	Hunting macrofauna	NONE	LC	0.55	0.34	0.16	0.24	0.11	0.19	0.13	0.14
Gobius xanthocephalus	NA	NA	NONE	LC	0.12	0.16	0.03	0.05	0.09	0.11	0.03	0.07
*Zebrus zebrus**	NA	NA	NONE	LC	0.06	0.06			0.01	0.03	0.03	0.10
Haemulidae												
*Pomadasys incisus**	≥2.8	Predator	Medium	LC	0.13	0.19			0.01	0.02	0.03	0.07
Labridae					0.55		0.57		0.57		0.56	
Coris julis	≥2.8	Predator	Low	LC	0.99	0.03	0.99	0.03	0.98	0.04	0.99	0.04
Ctenolabrus rupestris	≥2.8	Predator	Low	LC	0.12	0.19	0.76	0.14	0.25	0.19	0.66	0.11
Labrus merula (A,M,P)	≥2.8	Predator	Low	LC	0.68	0.26	0.57	0.17	0.99	0.08	0.74	0.14
Labrus mixtus	≥2.8	Predator	Low	LC					0.06	0.09	0.02	0.06
Labrus viridis (A,M,P)	≥2.8	Predator	NONE	VU	0.11	0.11	0.03	0.06	0.71	0.20	0.37	0.16
Symphodus cinereus	≥2.8	Predator	Low	LC	0.82	0.22	0.49	0.3	0.38	0.31	0.18	0.16
Symphodus doderleini	≥2.8	Predator	NONE	LC	0.06	0.11	0.03	0.06	0.09	0.07	0.19	0.08
Symphodus mediterraneus	≥2.8	Predator	Low	LC	0.3	0.13	0.81	0.09	0.72	0.19	0.77	0.17
Symphodus melanocercus	≥2.8	Predator	NONE	LC	0.04	0.06	0.33	0.19	0.62	0.27	0.59	0.21
Symphodus melops	≥2.8	Predator	Low	LC	0.32	0.19	0.01	0.02	0.10	0.09	0.10	0.16
Symphodus ocellatus	≥2.8	Variable	NONE	LC	0.98	0.06	0.69	0.2	0.62	0.26	0.84	0.15
Symphodus roissali	≥2.8	Predator	Low	LC	0.96	0.04	0.97	0.1	0.97	0.15	0.99	0.12
Symphodus rostratus	≥2.8	Predator	NONE	LC	0.66	0.14	0.29	0.11	0.58	0.20	0.37	0.16
Symphodus tinca	≥2.8	Predator	Low	LC	0.94	0.07	0.97	0.06	0.99	0.07	0.93	0.08
Thalassoma pavo	≥2.8	Predator	Low	LC	0.72	0.42	0.97	0.05	0.50	0.29	0.73	0.34
Moronidae												
Dicentrarchus labrax (M,P)	≥2.8	Predator	High	LC	0.47	0.22			0.44	0.27	0.08	0.16
*Dicentrarchus punctatus**	≥2.8	Predator	Low	LC	0.94	0.1	0.75	0.16	0.65	0.29	0.42	0.42
Mugilidae												
	2-2.19	Variable	Low	LC	0.79	0.14	0.02	0.04	0.85	0.14	0.81	0.21

Table 2 (cont.). – Fish species average catch image rate and standard deviation by location. The letters within brackets next to the species correspond to those with a catch image rate significantly higher in the Medes Marine Reserve; each letter corresponds to the location where the significant differences were observed. Asterisks indicate a species described for the first time in the Reserve.

Species	Feeding (trophic levels)	Feeding behaviour	Commercial value	IUCN status	Ametlla		Mataró		Medes Islands		Palamós	
					Mean	Std	Mean	Std	Mean	Std	Mean	Std
Mullidae												
Mullus surmuletus	≥2.8	Predator	High	LC	0.65	0.22	0.96	0.06	0.89	0.11	0.76	0.27
Muraenidae												
Muraena helena	≥2.8	Predator	Low	LC	0.3	0.77	0.28	0.17	0.36	0.17	0.20	0.09
Pomacentridae												
Chromis chromis	≥2.8	Predator	Unknown	LC	0.95	0.07	0.96	0.06	0.94	0.03	0.98	0.07
Sciaenidae												
Sciaena umbra (M,P)	≥2.8	Predator	High	NT	0.3	0.28	0.03	0.04	0.29	0.22		
Scorpaenidae												
Scorpaena maderensis	≥2.8	Predator	Low	LC	0.09	0.13	0.41	0.19	0.03	0.04	0.09	0.11
Scorpaena notata	≥2.8	Predator	Low	LC	0.47	0.22	0.90	0.09	0.21	0.19	0.51	0.21
Scorpaena porcus	≥2.8	Predator	Low	LC	0.82	0.12	0.44	0.22	0.24	0.28	0.58	0.19
Scorpaena scrofa (P)	≥2.8	Predator	High	LC	0.3	0.32	0.63	0.26	0.61	0.33	0.03	0.04
Serranidae												
Epinephelus marginatus (A,M,P)	≥2.8	Predator	High	EN	0.21	0.17	0.15	0.17	0.49	0.16	0.08	0.07
Serranus cabrilla	≥2.8	Predator	Medium	LC	0.82	0.37	0.98	0.03	0.97	0.05	0.96	0.05
Serranus scriba	≥2.8	Predator	Medium	LC	0.91	0.17	0.97	0.05	0.94	0.05	0.76	0.26
Sparidae					0.43		0.46		0.45		0.31	
Boops boops	≥2.8	plankton feeding	Low	LC	0.31	0.22	0.77	0.22	0.22	0.26	0.32	0.33
Dentex dentex	≥2.8	Predator	High	VU	0.08	0.07	0.06	0.06	0.16	0.24	0.04	0.11
Diplodus annularis	≥2.8	Predator	Low	LC	0.58	0.26	0.75	0.14	0.30	0.39	0.15	0.16
Diplodus cervinus (A)	≥2.8	Predator	Low	LC	0.06	0.09	0.21	0.25	0.42	0.20	0.14	0.21
Diplodus puntazzo	≥2.8	Predator	Low	LC	0.94	0.1	0.75	0.16	0.65	0.29	0.42	0.42
Diplodus sargus (P)	≥2.8	Predator	High	LC	0.96	0.04	0.85	0.11	0.98	0.04	0.77	0.14
Diplodus vulgaris	≥2.8	Predator	Low	LC	1.00	0	0.98	0.04	0.95	0.05	0.90	0.12
Lithognathus mormyrus	≥2.8	Predator	Medium	LC	0.13	0.11			0.17	0.28	0.03	0.05
Oblada melanura	≥2.8	Predator	Low	LC.	0.96	0.04	0.83	0.1	0.82	0.14	0.66	0.22
Pagellus acarne	≥2.8	Predator	Medium	LC	0.1	0.17	0.01	0.03	0.10	0.12	0.06	0.11
Pagellus erythrinus	≥2.8	Predator	Medium	LC	0.01	0.02	0.01	0.03	0.05	0.11	0.02	0.04
Pagrus pagrus	≥2.8	Predator	High	LC	0.11	0.2	0.01	0.03			0.08	0.10
Sarpa salpa	2-2.19	Grazing	Low	LC	0.98	0.13	0.70	0.18	0.96	0.05	0.96	0.08
Spicara smaris	≥2.8	Predator	Low	LC					0.02	0.05	0.02	0.04
Spicara maena	≥2.8	plankton feeding	Low	LC	0.03	0.06	0.18	0.14	0.29	0.25	0.08	0.08
Spondyliosoma cantharus	≥2.8	Variable	High	LC	0.25	0.22	0.53	0.26	0.24	0.22	0.03	0.04
Sphyraenidae												
Sphyraena sp.	≥2.8	Predator	Medium	LC	0.01	0.02			0.01	0.02	0.04	0.10
Torpedinidae												
Torpedo marmorata	≥2.8	Predator	NONE	DD	0.06	0.13	0.05	0.07	0.06	0.11		
Trachinidae												
Trachinus sp.	≥2.8	Predator	Low	LC	0.02	0.03	0.08	0.14	0.04	0.07	0.02	0.06
Tripterygiidae												
Tripterygion delaisi	≥2.8	Predator	NONE	LC	0.5	0.31	1.00	0.15	0.81	0.16	0.85	0.08
Tripterygion melanurus	≥2.8	Predator	NONE	LC	0.14	0.32	0.14	0.38	0.10	0.30	0.11	0.32
Tripterygion tripteronotus	≥2.8	Predator	NONE	LC	0.46	0.37	0.11	0.06	0.66	0.23	0.95	0.12

the species identified until now as *Gobius bucchichi* in this region, should be considered to be the recently described *G. incognitus* (Kovačić and Šanda 2016). The lists of species observed at the Medes Islands are 7 species that do not appear in the previous available fish inventory in this MPA (Dufour et al. 2007). Species richness and Margalef diversity indices showed significant differences among locations (Fig. 2). The multiple comparison Tukey test showed that only Mataró has a significantly lower species richness (Table 1), mainly due to the lower number of species of blenniids and gobids (Table 2).

Cluster analysis based on the species catch rates by location and year yielded four distinctive clusters (Fig. 3) corresponding to each specific location. Regardless of the year, each location was grouped in the same cluster, with one exception, the Medes Islands in 2015. The silhouette analysis (mean silhouette width=0.46) revealed that all contests had positive silhouette widths, which is indicative of correct classification within groups. The two-dimensional ordination of the 29 contests (Fig. 4) yielded a moderate level of ordination (stress=0.14), reflecting cluster separation through a gradual continuum of change. A gradient from north to south was observed among locations near the shoreline (Medes, Palamós and L'Ametlla). Consistently, the results of the cluster analysis showed that the most dissimilar group was represented by the contests held in Mataró.

The first cluster, grouping all the samples from Mataró, had a low number of gobiid and bleniid species, scarcity of mugilids and the absence of Atherinidae and European seabass (*Dicentrarchus labrax*), which mainly marked the difference of this group. An additional difference was the high values of species such as *Scorpaena notata*, *Thalassoma pavo*, *Boops boops* and *Gobius geniporus*.

The second cluster grouped all the contests of L'Ametlla de Mar, the southernmost location (the last contest held at the Medes Islands was also part of this group). The main differences compared with other groups were the high abundance of the genera *Pagel-*

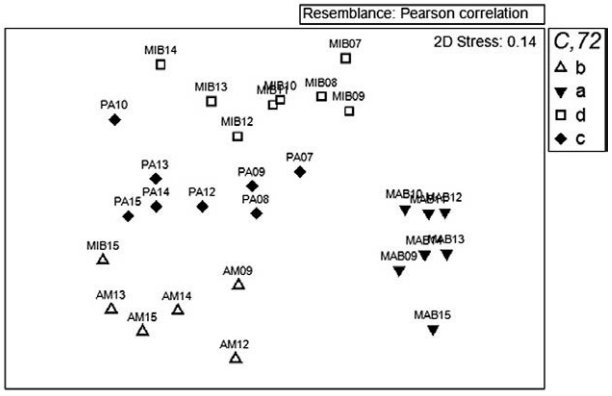

Fig. 4. – 2-d nMDS ordination plot for the 29 contests. Superimposed symbols for the different clusters.

lus, *Symphodus cinereus* and *Parablennius incognitus* together with the scarcity of *Ctenolabrus rupestris*, *Microlipophrys canevae*, *Symphodus melanocercus* and *Diplodus cervinus*. In this group the relative abundance of European seabass was also high, and only comparable with the values observed at the Medes Islands.

The third and fourth cluster comprised the contests held at the northernmost location corresponding to a non-protected and protected area respectively with the exception of the most recent year (see above) for the latter group. The main features of this group were the low relative abundance of gobiids, the maximum observed abundance of some species of labrids (*Symphodus melanocercus*, *Symphodus viridis* and *Labrus merula*) and the scarcity of *S. ocellatus*. This group was also characterized by high abundance of commercial species (e.g. *Epinephelus marginatus*, *Conger conger* and *Phycis phycis*), including the red scorpion fish (*S. scrofa*), but it also had the lowest abundance of the other three species of the same genus (*S. notata*, *S. porcus* and *S. maderensis*).

Although the clusters did not show any temporal variability within locations, except for the last contest held in Medes Islands, some shifts can be depicted. Some temporal changes were observed in the contest distances within the Medes Islands cluster, in which the 2013 and 2014 contests were clearly separated from the earlier ones. Similarly, differences were observed within the Palamós (cluster C) contests: those from 2012 onwards were distant from the previous ones, while in L'Ametlla the temporal pattern was more gradual.

The post hoc test showed significantly higher catch rates at the Medes Islands MPA with respect to the other locations (Table 2). These higher rates were mostly observed in species of commercial interest such as *P. phycis* and *E. marginatus*, but the list can be extended to *Conger conger*, *D. labrax*, S. *umbra*, *S. scrofa*, *D. cervinus* and *D. sargus*. In addition, the MPA also showed significantly higher catch rates for two species of labrids that are of negligible commercial interest but are highly targeted by spear fishers (*L. merula* and *L. viridis*). Furthermore, the relative abundance of *S. cinereus* decreased from southern to northern locations, while *S. melanocercus* displayed the opposite pattern.

DISCUSSION

Overall, the obtained outcomes show that free-diving fish photography contests could be a very useful tool for analysing fish assemblages and species richness in the littoral system. In general, fish classification results grouped each location separately, as expected from their differences in latitude, depth range and habitat. In addition, the already known effect of the Medes Islands MPA (García-Rubies and Zabala 1990, García-Rubies et al. 2013) was evidenced by the higher abundance of species of high commercial values.

The high number of species recorded in this study exceeds by far those reported in any published study on littoral communities carried out in the Mediterranean (e.g. Consoli et al. 2016), and is only comparable to but still higher than the number of species reported in the marine reserve of Ustica Island (La Mesa and Vacchi 1999). It is worth mentioning that the high number of species reported in the study of Ustica Island could be due to the combined sampling methodology used (strip transects by SCUBA, SCUBA diving tracts and snorkelling), which minimized the constraints inherent in survey methods, particularly for cryptic species (Brock 1982, Willis 2001, MacNeil et al. 2008). This indicates the effectiveness of the analysed information source for measuring fish species richness with a single data source.

In the Mediterranean, the littoral fish community is under various anthropogenic threats (Bianchi and Morri 2000, Claudet and Fraschetti 2010, Costello et al. 2010), but systematic monitoring is limited to a small number of commercial species or to protected areas (e.g. Coll et al. 2012, García-Rubies et al. 2013). Although overall littoral fish monitoring would be unrealistic to finance, many commercial groups, such as sparids, are not subject to assessment, so they are not under any systematic monitoring programmes, and this lack of information is hard to justify. The species recorded in this study that have a known status are mainly on the least-concern list: only one is endangered (*E. marginatus*) and one is considered vulnerable (*L. viridis*).

Among the results shown in this study, of particular relevance is the capacity of free-diving photography contests to count and assess cryptic species, which can be missed by underwater visual censuses (Ackerman and Bellwood 2000, Willis 2001, Smith-Vaniz et al. 2006). A total number of 32 cryptic species were photographed during these contests, a figure that is also higher than in any previous studies targeting this group based either on visual censuses (e.g. Macpherson 1994, La Mesa et al. 2006, Bussotti et al. 2015) or destructive techniques (Patzner 1999, Kovačić et al. 2012, Glavičić et al. 2016). Although they contribute greatly to the species richness of littoral systems, small cryptic species are perceived as minnows (small non-game or non-commercial fish species) due to their small size and cryptic behaviour. Consequently, they are generally overlooked or underestimated (Ackerman and Bellwood 2000, Willis 2001), and it is difficult to gain support for monitoring and conserving them (Sheldon

1988). Therefore, an important part of the strictly substratum-dependent littoral fish community is usually ignored, preventing any changes from being detected both in the community and in the microhabitat of the substratum.

Overall, the most abundant species were *C. julis*, *S. roissali*, *S. tinca*, *S. cabrilla*, *D. vulgaris* and *C. chromis*. In numerous studies these species have been reported as the most common in the rocky reef habitats of the Mediterranean littoral system (Bell 1983, García-Rubies and Zabala 1990, Claudet et al. 2006). However, the most abundant species differed between locations. The most abundant species in Mataró was *T. delaissi*. The preferred habitat of this species, flat, rocky and sheltered from wave action of the open sea (La Mesa et al. 2004), match the Mataró habitat location, a small, relatively flat, natural rock over 150 m in length. Further characteristics of this location, offshore at a minimum depth of 6 m that minimizes wave action and light, are also considered to be preferred by *T. delaisi* (Zander and Heymer 1970). Mataró's minimum depth also explained the absence of species typical of shallow waters, such as many species of blenniids (e.g. *A. sphynx*, *L. trigloides* and *M. canevae*), whose highest diversity (Kotrschal 1988, Macpherson 1994) is restricted to the narrow depth zone from 0 to 1 m (Illich and Kotrschal 1990) and the negative correlation between depth and blenniid diversity was recently proved (Tiralongo et al. 2016). This absence was also detected for some other species characteristic of shallow waters (*C. argentatus* and *Z. zebrus*). Another interesting observation in Mataró was the coexistence with high relative abundances of the four species of scorpenids (*S. maderensis*, *S. notata*, *S. porcus* and *S. scrofa*). On this basis, we could reject the mutual species displacements by interspecific competition for habitat or food resources that have been hypothesized for this group (La Mesa et al. 2004). However, based on the findings in the MPA, we cannot entirely reject this hypothesis. In the MPA, while *S. scrofa* was highly abundant, the other species of scorpaenids showed values generally below those observed in the non-protected locations. *S. scrofa* is bigger than the other species (Petrakis and Stergiou 1995, Ordines and Massuti 2009, La Mesa et al. 2010) and under the protection effect the species can get older and bigger (e.g. Halpern and Warner 2003) and easier to see.

The results on species richness and diversity index showed no positive effects on the MPA, as has been proposed in other studies (Guilhaumon et al. 2015). The results are indicative of the positive effect of the MPA on species of commercial interest, but not on all species, because responses to protection can be highly variable among fish taxa (Claudet et al. 2006). The positive effect of the MPA in contrast with the abundance observed in the closest comparable non-protected location (Palamós) was unquestionable for species of high commercial value. A total of eight species showed higher and significant values within the MPA: *C. conger*, *P. phycis*, *D. labrax*, *S. umbra*, *S. scrofa*, *E. marginatus* and *D. sargus*. The recovery of the dusky grouper in MPAs has been reported in numerous studies (e.g. Bell 1983, García-Rubies and Zabala 1990, Sahyoun et al. 2013) as has that of white seabream (Sahyoun et al. 2013). The positive effect in this particular MPA, the Medes Islands, has already been observed for most of the above-mentioned species (i.e. *D. labrax*, *E. marginatus*, *D. cervinus*, *S. aurata* and *S. umbra*) (García-Rubies et al. 2013). In addition, two species of labrids (*L. merula* and *L. viridis*) of no commercial interest but of recreational spear fishing interest were also favoured by the protection. *L. viridis* is considered vulnerable and its protection is recommended. The lack of information on fish sizes prevents us from analysing the well-known positive effect of MPA on this key indicator (Lester et al. 2009).

A different argument is that the response to protection can also be negative for certain taxa, and changes in fish assemblages in MPAs could occur because predation pressure is expected to be higher (Francour 1994, Ashworth and Ormond 2005). A potential negative effect on small cryptic species has been suggested (Willis and Anderson 2003, Claudet et al. 2006), but our results confirm that only two species were significantly higher at the unprotected site (*P. gattorugine* and *T. tripteronotus*). However, not all the differences in species abundance can be attributed to protection measures or to the type of data, because we also found a latitudinal gradient of variability that was unrelated to these factors. *S. melanocercus*, which prefers rocky areas and seagrass beds, decreases from north to south, while *S. cinereus* and *S. ocellatus,* which generally inhabits seagrass beds and sometimes soft bottoms and estuarine lagoons, increases from north to south. This partitioning is consistent with the previous categorization in the Catalan littoral system based on shore-based fishing tournament reports (Gordoa 2009, Boada et al 2017).

The type of information used in this study has several weaknesses, such as a potential underestimation of the most frequent species because the number of photographs per species and participant is limited to one, and the possibility that the same fish could be photographed by different participants. However, despite the potential weaknesses of photographic free-diving contest data, the results of this study show their effectiveness for evaluating species richness, in particular of cryptobenthic species, and for analysing littoral fish communities. Furthermore, it has also proved its worth as a tool for updating fish inventories. We can summarize that this type of contest enhances competition between participants to obtain maximum fish diversity. Additional positive aspects of this information source are that it is cost-effective, non-destructive, a potential observatory, an platform for interaction between scientists and free divers, and an alternative for spear fishers in MPAs. We conclude that the monitoring of photographic free-diving contests could be a complementary information source to scientific monitoring.

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