A critical approach to the use of published data for baseline characterisation of marine fish assemblages: An exercise on Portuguese coastal waters

Miguel Pessanha Pais [a],[*], Sofia Henriques [a], Maria José Costa [a], Henrique Nogueira Cabral [a]

[a] Centro de Oceanografia, Faculdade de Ciências, Universidade de Lisboa, Campo Grande, 1749-016 Lisboa, Portugal

A R T I C L E   I N F O

A B S T R A C T

Policy-makers are becoming increasingly aware of the importance of coastal fish communities for the integrity of marine habitats. However, initial assessments are often done prior to any major funding and available published data is sometimes seen as a low-cost approach to this problem. The present work constituted an exercise on marine fish assemblage characterisation, by relying only on published data from mainland Portugal, covering soft substrate areas and near-shore rocky reefs. Regardless of the characterisation itself, this exercise aimed to critically approach these procedures, by pinpointing problems and highlighting achievements. Criteria were defined for the inclusion of published studies and included datasets were standardised in terms of species proportions, spatial and temporal scales. Species were assigned to ecological guilds and all analyses were performed separately for species and guild data. The influence of season, depth and latitude were analysed through multivariate analysis of variance using permutations and distance-based linear models and assemblages were characterised based on multivariate ordination and hierarchical classification methods. Significant differences were found between studies employing different methods in rocky reefs and outliers were not used in assemblage characterisation. Due to limitations in spatial and temporal detail in published data, only a rough, large scale characterisation of assemblage types was accomplished, with datasets divided in rocky reefs, shallow soft-substrate (0–20 m), intermediate soft-substrate (20–100 m) and deep soft substrate (100–200 m). No seasonal differences were detected, but a non-linear relationship with latitude was found in deep soft-substrate assemblages. During the exercise, spatial and temporal scales were forced by the quality of the data and some analyses failed to detect known patterns. Differences between methods, designs and types of data required assumptions to be made regarding data comparability. Overall, the exercise provided a hands-on insight into the use of published data for planning in a coastal management context, pinpointing data comparability, scale and information gaps as severe limitations of these approaches. In order to improve planning and design, some proposed measures were highlighted, such as the establishment of national databases and the use of pilot studies.

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1. Introduction

Due to the consequences of increasing anthropogenic pressure on the marine environment, several national and international policies are addressing the need to maintain the integrity of marine ecosystems (Borja et al., 2008; Katsanevakis et al., 2011; Ricketts and Harrison, 2007). In fact, in Europe, the importance of coastal fish communities for the stability of the ecosystem and their role in supporting stocks is being increasingly addressed by both international policies and local management requirements (Borja, 2006; EC, 2008; Stål et al., 2008). Moreover, the high socio-economic value of fish, allied to their relative easiness of identification, diversity of ecological guilds, longevity, among others, are important advantages of using them as quality indicators for water bodies (Harrison and Whitfield, 2004; Whitfield and Elliott, 2002).

On the marine environment, most of the work has been centred on the impact of fishing on exploited fish species (e.g. Rice, 2000; Sainsbury et al., 2000) or, more recently, on an ecosystem approach to fisheries management (e.g. Browman and Stergiou, 2004; Jennings, 2005) that constitutes a holistic approach to fisheries but leaves a minor role to other anthropogenic impacts (Dethlefsen and Tiews, 1985). Adding the human dimension to the ecosystem is now deeply embedded in European policy, with the inter-connection of all species and habitats being a keystone

* Corresponding author. Tel.: +351 217500826.
  E-mail address: mppais@fc.ul.pt (M.P. Pais).

principle in Integrated Coastal Zone Management. In fact, there is increased awareness that the effects of humans on exploited resources are reciprocal (Forst, 2009). With this in mind, the implementation of the Marine Strategy Framework Directive in 2008 (EC, 2008) has set the goals for improving the state of European marine waters, but the scientific community is yet struggling to understand some of the underlying fundamental differences between marine habitats that define the "best" potential community a given area can support.

So far, the most common fish-based tools created in response to policy requirements in streams and estuaries (Cabral et al., 2012; Roset et al., 2007) and, more recently, in coastal waters (Henriques et al., 2008a,b) are multimetric indices, which rely mainly on functional guilds, by comparing measurable aspects of a community with "reference" values that correspond to a certain "status" in a given "type" of habitat. However, these (and likely other) tools have requirements that need to be fulfilled before they can be successfully applied in a monitoring context, namely the definition of habitat "types" and the understanding of how the relative importance of each functional unit in a community varies between habitats (Henriques et al., 2008b).

The downside is that initial assessments are often done prior to any major funding and, more often than desirable, have no choice but to rely on available published data compiled in a database of previous studies for a given area (Aiken et al., 1999; Elliott et al., 1999). Because published data have served different purposes, often there is valuable information missing regarding habitat characteristics that would prove useful in defining habitat types (Olenin and Dawnys, 2004). However, since what is ultimately measured is the supported community, some information can be obtained by looking at the assemblages themselves. Assuming that in a baseline characterisation of coastal areas, data should represent the "natural" state of a system, surveys performed in heavily impacted sites should be avoided, as they are likely to support "unnatural" communities (Guidetti et al., 2003; Smith et al., 1999). Another important aspect when characterising assemblages in a monitoring context is that species lists are insufficient, as important variations in abundance would pass unnoticed (Hewitt et al., 2005; de Jonge et al., 2006). On the other hand, it is known that different sampling designs and methods can lead to different abundance estimates for a given area (Pasquaud et al., 2012; Willis et al., 2000), which, coupled with the fact that raw abundances are not frequently published (e.g. Olenin and Dawnys, 2004), can prevent the use of abundance values for analysis.

The present work constitutes an exercise of marine fish assemblage characterisation using only published data as a basis. As a case study, we will focus on the coastal waters of mainland Portugal down to 200 m deep, covering soft substrate areas and near-shore rocky reefs, in an attempt to understand the main gradients and factors delimiting fish assemblages, not only in terms of species composition, but also using ecological guild data. The study ultimately aims to critically approach such procedures, by pinpointing problems and highlighting achievements.

2. Materials and methods

2.1. Study area

The Portuguese coast is located in western Iberian Peninsula and extends from the Minho river mouth southwards along the 9°W meridian, and then eastwards at cape São Vicente (Fig. 1). The continental shelf is relatively narrow and its most conspicuous irregularity is the Nazaré Canyon. Reaching depths of around 5000 m, this depression divides the western shelf in a northern, flatter section up to 70 km wide, and a southern, steeper section up
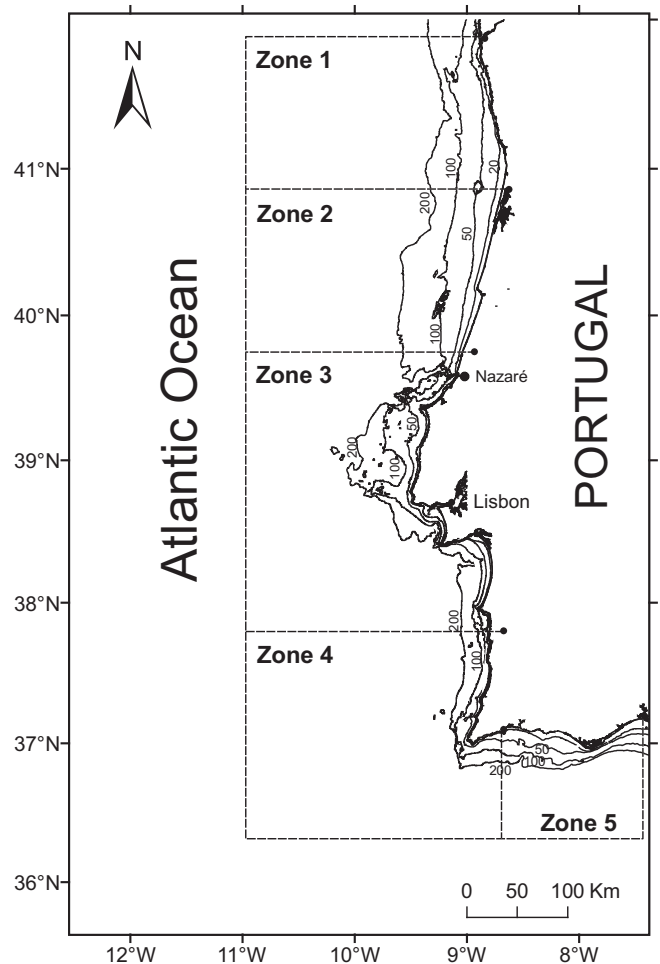

Fig. 1. Map of the study area with coastal zones delimited by IPIMAR (1980, 1981a,b, 1982, 1984) based on hydro-morphological criteria. Only the 20 m, 50 m, 100 m and 200 m isobaths are represented.

to 20 km wide until cape São Vicente, then reaching a width of about 30 km in the south coast (Gomes et al., 2001). Over the shelf, upwelling occurs during warm months (April–September) due to predominant northern winds. In winter, only intermittent and weaker upwelling periods are observed. Average sea surface temperature usually stands in the 14–21 °C range (Lemos and Pires, 2004).

2.2. Data collection and assumptions

For the purpose of this study, no privileged access to databases was used and only publicly available data was considered. In general, it was found that data on composition and abundance of fish assemblages in the study area is not easily accessible, being mostly scattered across several sources in research papers, academic theses and technical reports. Nevertheless, an effort was made to compile data from various locations, depth ranges, seasons and substrates.

The first step in data collection was to define the basic criteria that any published dataset should fulfil to be included in the exercise, by defining a simple decision tree illustrated in Fig. 2. This step was crucial in order to ensure an acceptable degree of comparability between studies that are likely to differ in their design and goals.

In order to minimise the introduction of samples from highly impacted assemblages, near-shore sites were given priority when
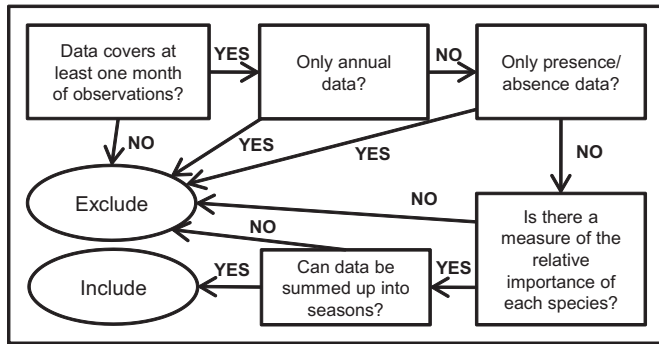
Fig. 2. Decision-tree illustrating the selection criteria for data to include in the exercise.

located near or inside protected areas. Moreover, given the unavoidable differences in abundance values between sampling methods and designs and to maximise the available data, relative abundances were calculated for every dataset to conserve the proportion of the assemblages represented by each species or guild. Another important criterion was the possibility to organise data into seasons in order to introduce seasonal variability into the analysis.

For each site and season, data were aggregated into a single data point and were assumed representative of the proportion of species and guilds on that site and season. As a result, by the end of the data collection process, each entry in the database represented the relative proportion of species in an area of the coast in a particular season, delimited by depth ranges. In order to avoid confusion regarding the concept of sample, data points are referred in the text as "datasets", since they are the result of aggregated samples in a dataset.

Most of the data used for soft substrate assemblages along the coast were gathered from technical reports of the trawl surveys performed by the Portuguese Institute for Fisheries and Sea Research (IPIMAR) that are grouped according to pre-defined depth ranges and coastal sections (zones 1 to 5 in Fig. 1). This zonation was indivisible in the published data, however, it is based on hydro-morphological criteria and zones can be roughly corresponded to homogenous fish communities, considering the large spatial scale (Gomes et al., 2001). Therefore, in order to overcome this constraint and standardise latitudinal location across the database, all collected datasets were assigned to these zones.

Concerning sampling procedures on the collected datasets (Table 1), otter trawl was the most frequent method used on deeper soft substrate, complemented with beam trawl data to characterise

some areas approximately 10–30 m deep (Abreu, 2005) and beach seine fisheries data for some areas shallower than 10 m (Cabral et al., 2003). Beach seine, despite not being intentionally performed with the purpose of surveying fish assemblages, provides rather complete data due to the low selectivity of the fishing gear (Cabral et al., 2003). All rocky reefs were sampled using underwater (SCUBA) visual census methods that varied from timed random paths to fixed distance transects and stationary observations.

Compiled studies were organised into 69 datasets (Table 1) and the taxonomy was updated and corrected according to FishBase online database (Froese and Pauly, 2011).

2.3. Guild classification

One of the advantages of using fish as ecological indicators is the large variety of ecological guilds, which can respond more predictably to alterations on the ecosystem (Elliott et al., 2007; Halpern and Floeter, 2008). Therefore, not only is there a need for guilds to play an important role in monitoring and assessment tools (Henriques et al., 2008b; de Jonge et al., 2006), but also the baseline characterisation of habitats and assemblages should take into account the distribution of these guilds regardless of individual species. For this reason, all species were incorporated into a total of 37 ecological guilds from seven categories (Table 2), based on previous work by Henriques et al. (2008a,b).

On substrate dependence guilds, species were considered "resident" when a particular substrate is needed for settlement, life and reproduction to occur and "dependent" when a particular substrate is needed to partially fulfil the requirements of the species life cycle (e.g. food, reproduction, protection, etc.). The term "offshore" was used when species inhabit or depend on deeper waters, mostly indifferent to the type of substrate beneath.

Migration and trophic guilds were based on the review on estuarine fish guilds by Elliott et al. (2007), with some adaptations to marine communities. Species were considered "invertebrate feeders" when they feed mostly on non-planktonic invertebrates, otherwise being considered "zooplanktivore", along with other zooplankton feeders (e.g. species that feed on hydroids and fish eggs/larvae). "Herbivore" species feed mostly on benthic and planktonic macro and microalgae and macrophytes. Detritus and opportunistic feeders were included along with other "omnivore" species. "Macrocarnivores" feed both on macroinvertebrates and fish and species that feed almost exclusively on fish were included on the "piscivore" guild.

Habitat guilds were adapted from Holthus and Maragos (1995) and resilience guilds were based on the estimated minimum

Table 1
Summary of the references from which the data were collected. See Fig. 1 for coastal zone limits.

Reference	Sampling years	Substrate	Latitude zones	Seasons sampled	Depth range (m)	Sampling method/gear	Nr. of datasets
IPIMAR, 1980	1979	S	1, 2, 3, 5	Su	20–200	Otter trawl	8
IPIMAR, 1981a	1979	S	1, 2, 3, 4, 5	W	20–200	Otter trawl	9
IPIMAR, 1981b	1980	S	3, 4, 5	Sp	20–200	Otter trawl	6
IPIMAR, 1982	1980	S	1, 2, 3, 4, 5	Su	20–200	Otter trawl	10
IPIMAR, 1984	1980	S	1, 2, 3, 4, 5	W	20–200	Otter trawl	10
Henriques, 1993	1993	R	3	Sp/Su/A/W	1–15	VC random	4
Rodrigues, 1993	1993	R	3	Sp/Su/A/W	10–25	VC transect	4
Almeida, 1997	1997	R	5	Su	17–25	VC stationary	1
Almada et al., 2002	1999	R	3	Sp/Su/A/W	1–20	VC transect	4
Cabral et al., 2003	1999	S	3	Sp/Su/A	0–20	Beach seine	3
Prista et al., 2003	2001	S	3	Sp/Su/A/W	10–30	Otter trawl	4
Almada et al., 2004	2004	R	3	A	1–20	VC transect	1
Gonçalves, 2004	2002	R	4	Su	19–23	VC random	2
Abreu, 2005	2005	S	5	Sp	10–30	Beam trawl	1
Maranhão et al., 2006	2004-05	R	3	Su	1–15	VC random	2

VC, visual census; S, soft; R, rock; Sp, spring; Su, summer; A, autumn; W, winter.

Table 2
List, by category, of the ecological guilds used in the analysis.

Category	Guild	code
Substrate dependence	S resident	Sres
	O resident	Ores
	R resident	Rres
	I resident	Ires
	S dependent	Sdep
	O dependent	Odep
	R dependent	Rdep
	I dependent	Idep
Mobility	High	himob
	Medium	medmob
	Territorial	terr
	Sedentary	sed
Habitat	Demersal	dem
	Pelagic	pel
	Reef-associated	reef
	Bathydemersal	batd
	Bathypelagic	batp
	Benthopelagic	bentp
Migration	Non-migratory	nomig
	Oceanadromous	oce
	Catadromous	cat
	Anadromous	ana
	Anfidromous	anf
Trophic	Invertebrate feeder	inv
	Omnivore	om
	Macrocarnivore	mac
	Zooplanktivore	zoo
	Piscivore	pi
	Herbivore	he
Resilience	Very low	VLresi
	Low	Lresi
	Medium	Mresi
	High	Hresi
Spawning season	Spring	SPspn
	Summer	SUspn
	Autumn	AUspn
	Winter	WIspn

I, rocky intertidal; S, soft substrate; R, rocky substrate; O, offshore. See section 2.3 for a detailed description and references.

population doubling time (Musick, 1999) and classified as "high" (up to 1.4 years), "medium" (1.4—4.4 years), "low" (4.5—14 years) and "very low" (more than 14 years) according to FishBase data (Froese and Pauly, 2011). The proportion of individuals that fit each guild by category (hereafter designated "guild data") was calculated for each dataset, constituting a separate data matrix.

2.4. Data analysis

In order to study the differences between groups in terms of guilds or species composition, both types of data were used in the analyses. On all initial analyses, soft and hard substrates were separated, since otherwise the expected differences between them would dominate multivariate analyses.

2.4.1. Minimising the effect of sampling methods
Prior to characterising the assemblages, the multivariate methods described below were used to analyse potential confounding variation due to differences in sampling methods, by isolating zone 3 (Fig. 1), which presented the highest variability in terms of methods for rocky reefs and shallow soft substrate areas (Table 1). Methods in zone 3 that were significantly dissimilar and considered less representative of the assemblage (lower number of

species counted in average) were marked for exclusion. Unconstrained ordination and classification methods were then used to compare these deviant datasets with all data available and exclude similar datasets from other coastal zones.

2.4.2. Assemblage characterisation
All multivariate analyses were based on Bray—Curtis dissimilarities (Bray and Curtis, 1957) and no further transformations were performed besides the standardisation per total in each dataset, since variations in the proportions of the most common species were important, considering the relatively short coastal area and depth range covered.

In order to test the effects of categorical factors in the assemblage structure using Bray—Curtis dissimilarities and not having to comply with normality assumptions, multivariate analysis of variance using permutations (PERMANOVA) was applied. This method partitions the total sum of squares based on the distance measure of choice and calculates a pseudo-F statistic, analogous to the F statistic in traditional ANOVA (Anderson, 2001; McArdle and Anderson, 2001). Unlike in a typical assessment, where experiments are designed a priori, the use of published data often leads to heavily unbalanced data (unequal number of samples within each level of a factor), and complex designs are sometimes impossible to test due to lack of replication and coverage of all possible combinations. However, using one-way PERMANOVA will perform the correct tests with minor concerns, under the assumption that individual data points are permutable under the null hypothesis (Anderson et al., 2008). Therefore, for the purpose of this exercise, all factors were analysed separately, with possible interactions being merely observed in multivariate space or minimised by selectively removing confounding datasets for each analysis. In the case of depth ranges and latitude zones on soft substrate datasets, PERMANOVA was used to test their effect as qualitative (categorical) predictors and distance-based linear models (DISTLM; Legendre and Anderson, 1999; McArdle and Anderson, 2001) to test their effects as quantitative (gradient) predictors (using the average depth per dataset).

Another important factor when dealing with unbalanced data from various sources is the differences in multivariate dispersion between groups, to which PERMANOVA is sensitive (Anderson et al., 2008). Therefore, a test of homogeneity of dispersions (PERMDISP; Anderson, 2006) based on distances from centroids was used to complement the analysis.

In order to observe data in unconstrained multivariate space based on the Bray—Curtis dissimilarities, Principal Coordinates Analysis (PCO; Gower, 1966; Torgerson, 1958) was used.

However, since in many cases the differences between groups are not evident when visualising axes that maximise total variance, a canonical analysis of principal coordinates (CAP; Anderson and Robinson, 2003; Anderson and Willis, 2003) was used to find axes in multivariate space that best discriminate between groups of interest.

Some observations were made concerning the species and guilds responsible for the observed patterns by representing Spearman's rank correlations of the original variables with PCO and CAP axes and group cohesion was analysed by calculating classification success rates through a "leave-one-out" procedure (Lachenbruch and Mickey, 1968).

P values were obtained using 9999 permutations. In PERMANOVA analyses, whenever the number of unique permutations available did not reach 100 due to lack of replicates, P-values were based on the Monte Carlo method proposed by Anderson and Robinson (2003). All analyses were performed using PRIMER v6 with PERMANOVA + package and the level of statistical significance adopted was 0.05.

Table 3
Within-substrate comparison between different sampling methods used in shallow areas (less than 50 m deep) of coastal zone 3 (see Fig. 1) with the average number of species (and standard deviation) calculated from *n* available datasets. Average Bray−Curtis similarities are shown for both species (in bold) and guild data. All PERMANOVA pairwise comparisons showed significant differences between methods.

	Method	*n*	Nr. of species	Average Bray−Curtis similarity				
				OT	BS	VCR	VCT	VCTb
Soft substrate	Otter trawl (OT)	4	21 ± 7	**51.21**	–	–	–	–
				74.50				
	Beach seine (BS)	3	38 ± 8	**18.32**	**39.77**	–	–	–
				48.33	69.63			
Rocky reefs	Random path (VCR)	6	32 ± 11	–	–	**65.92**	–	–
						91.10		
	Transect (VCT)	5	49 ± 7	–	–	**25.63**	**52.12**	–
						78.24	81.72	
	Benthic transect (VCTb)	4	13 ± 2	–	–	**2.31**	**8.09**	**59.33**
						52.60	60.81	86.37

3. Results

A total of 201 species from classes Chondrichthyes and Actinopterygii were counted on all 69 datasets included in the exercise.

3.1. Minimising the effect of sampling methods

Despite the criteria adopted when selecting published data, some unwanted variation due to sampling methods was only visible when comparing datasets. Due to the diversity of sampling methods in shallow areas of coastal zone 3 (Fig. 1, Table 1), comparisons were only made within this zone.

PERMANOVA found significant differences between methods using species and guild data for both substrates, yet leaving a single method per substrate in this case would lead to the loss of a large number of datasets and prevent further characterisation of the assemblages. Therefore, the criteria adopted were the average number of species counted by each method (per dataset) for this coastal zone and the within and between-group similarities (Table 3).

For soft substrates, otter trawl had less species per dataset, in average, than beach seine, which is probably related to the fact that seine nets sample a wider depth amplitude and have a smaller mesh size (Cabral et al., 2003). However, otter trawl in this case can be seen as complementary in terms of depth and habitat, as it can sample areas that are deeper and adjacent to rocky coastal areas (Prista et al., 2003), therefore supporting the decision to maintain both methods for this depth range. When including the remaining dataset from Abreu (2005), located in the south coast (zone 5) and sampled with beam trawl, in a cluster analysis by group average, all datasets were clustered at 8.4% similarity using species data and at 46.7% using guild data. Upon removal of the latter dataset, clustering level improved to 17.6% similarity for species data, but maintained at 46.7% for guild data, which led to the decision of not discarding any datasets for functional characterisation. However, all comparative analyses relying on soft substrate data for this depth range were limited to zone 3 (see section 3.2).

On rocky reef datasets, however, a study by Rodrigues (1993) (see Table 1) applied a visual census method with a clear focus on benthic species, which led to an overall lower number of species counted and low average similarities with other methods (Table 3), some of them performed on the exact same site (Table 1). This led to the decision of excluding these datasets and cluster analysis, along with unconstrained ordination (Fig. 3) were used to decide on the exclusion of data from other coastal zones. All rocky reef datasets clustered at 4.7% similarity with species data and 54.05% with guild data. Since data from zone 5 published by Gonçalves (2004) (see Table 1) clustered with the datasets to be removed

when using both species and guilds (Fig. 3), these datasets were also excluded. Upon removal of all datasets with benthic-oriented methods, all data clustered at 15.4% using species data and 67.4% using guild data.

3.2. Assemblage characterisation

After excluding rocky reef datasets with deviant sampling methods, the resulting data were mostly from coastal zone 3 (see Fig. 1) and depths were not evenly distributed across sites (see Table 1), thus not allowing any tests regarding latitude or depth. Only seasonal variation was tested, showing no significant differences when using species (pseudo-$F = 0.317$, $P = 0.9763$) and guild data (pseudo-$F = 0.283$, $P = 0.9974$).

For soft substrate assemblages, latitude and depth were tested as gradient and categorical predictors, while excluding confounding datasets in each case, which would bring additional, unequally distributed variance across levels of the factor under test (Table 4). Depth showed significant results both as a quantitative and a categorical predictor for coastal zone 3, while latitude showed significant differences only when treated as categorical. These observations support the hypotheses that functional and compositional changes in fish assemblages on the study area may occur along a depth gradient but do not seem to have a linear relationship with latitude (Hewitt et al., 2005). In fact, pairwise tests overall tend to show a difference between the central zones 3 and 4 with northern and southern areas (zones 1 and 5, respectively) (Table 4). Multivariate dispersion was significantly different between depth ranges in zone 3 using species ($F = 49.542$, $P = 0.0003$) and guild ($F = 27.490$, $P = 0.0002$) data, with deep (100−200 m) assemblages being significantly less variable in multivariate space. Overall, no significant differences were found between the available seasons (spring, summer and winter).

Given that depth ranges were indivisible in some datasets (IPIMAR, 1980, 1981a,b, 1982, 1984), depth was treated as categorical and the effects of season and latitude were tested separately within each depth range (Table 5). PERMDISP found differences in multivariate dispersions between seasons in intermediate depths (20−100 m), but PERMANOVA did not find enough evidence to reject the null hypothesis. Latitudinal location (coastal zones), however, showed significant differences on the guild and species composition of soft substrate assemblages sampled 100−200 m deep.

The pattern responsible for this difference can be seen on the CAP plot (Fig. 4) for the 100−200 m depth range, where the canonical axes are drawn to best discriminate between latitude zones as categorical predictors. By looking at the plot, it is evident that datasets located to the right of the plot are mainly from central
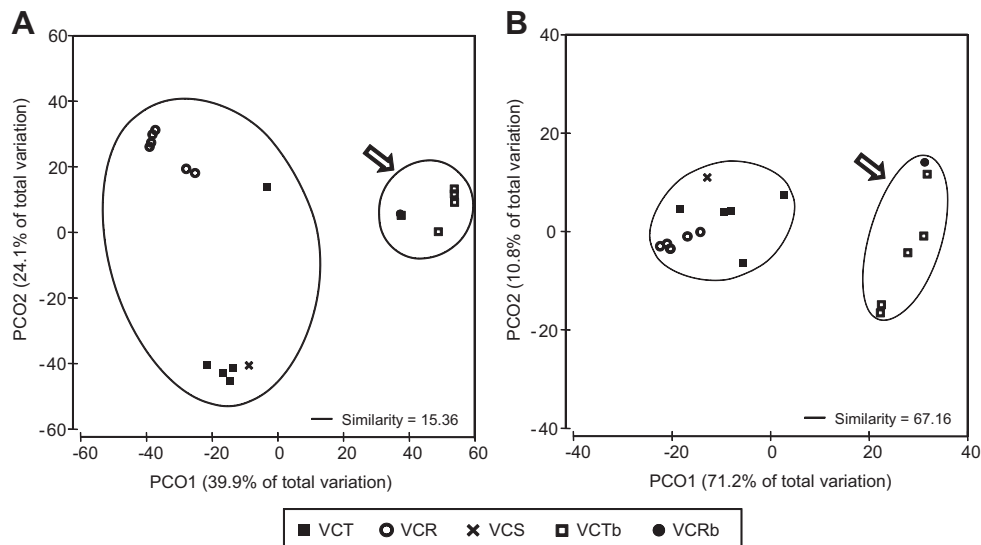
Fig. 3. Principal coordinates analysis (PCO) using species (A) and guild (B) data. The group signalled by an arrow was excluded from further analyses due to underestimation of assemblage composition (see Table 3). Cumulative explained variation was A) 87.80% and B) 77.68%. Similarity contours represent the level at which the accepted datasets were clustered by group average. For guild code correspondence see Table 2, for species codes see Appendix A.

locations on the coast (zones 2, 3 and 4) and dominated by two species of snipefish (*Macroramphosus gracilis* and *Macroramphosusscolopax*), whereas datasets to the left are mainly constituted by hake (*Merluccius merluccius*) and other species. When plotting guild data (not shown), the results were similar, since datasets to the right of the plot were dominated by all the guilds represented by snipefish.

After analysing substrates independently, datasets were separated into four basic assemblage "types", according to the results obtained: rocky subtidal (R; permanently submerged rocky reefs down to a depth of 20 m), shallow soft-substrate (SS; soft substrate down to 20 m deep), intermediate soft-substrate (IS; soft substrate 20−100 m deep) and deep soft-substrate (DS; soft substrate 100−200 m deep). The distribution of all datasets was then analysed using species and guild data in unconstrained multivariate space using PCO (Fig. 5a,c), where the distinction between depth ranges in soft substrate areas was once again overruled by the presence of datasets dominated by a few species of the genus *Macroramphosus* (Fig. 5a). However, when looking at the variation along the vertical axis, assemblage types are not separated, being dispersed along a gradient, with a clear overlap between depth ranges that becomes even clearer when looking at functional guild composition (Fig. 5c).

In order to look at the multivariate distribution of the established types without the strong influence of atypical datasets, a CAP analysis was used to find the axes that best discriminate between

the defined assemblage types (Fig. 5b,d). In Fig. 5b, a distinction between types became evident, with the "leave-one-out" allocation procedure correctly classifying 69.8% of the datasets. The most distinct type were rocky reefs, with 100% correct classifications, and IS assemblages, with 87.5%. The only clear overlap when using species data was between IS and DS assemblages, where 9 out of 21 IS datasets were allocated to the DS group and 5 out of 22 DS datasets were misclassified as IS.

When using guild data (Fig. 5d), groups were more distinct, leading to 79.9% of correct allocations. Classification success for R assemblages was 91.7%, with one dataset misclassified as SS. DS assemblages had 81.8% correct classifications, with 4 datasets classified as IS, and SS had 62.5% success, overlapping with both R and IS assemblages. The largest functional overlap was observed in IS assemblages (47.6% success), with 8 datasets classified as IS, and 3 as SS.

The objective of this exercise is to attempt a functional characterisation from published data that mostly focus on species, so individual species will deliberately not be given much attention, serving occasionally as a justification for functional aspects of the assemblages. However, by comparing Spearman's correlations of species proportions with the PCO and CAP axes when using species data (Fig. 5a,b), it is evident that the influence of highly dominant gregarious species like *Macroramphosus* spp. was minimised, with a gradient of species that characterise each assemblage type

Table 4

Influence of latitude, depth and season on soft substrate assemblages using species and guild data. For depth as a quantitative gradient, the average depth per dataset was used, whereas in the categorical test, depth ranges corresponded to shallow (S; 0−20 m), intermediate (I; 20−100 m) and deep (D; 100−200 m) areas. *P*-values were bolded when considered significant and underlined if significantly different multivariate dispersions were found in PERMDISP analyses. For pairwise tests, significantly different pairs are shown in brackets.

	Data type	Quantitative (DISTLM)		Categorical (PERMANOVA)		
		pseudo-*F*	*P*	pseudo-*F*	*P*	Pairwise
Latitude (20−200 m)	Species	2.0637	0.0949	2.7401	**0.0039**	(1,3)(1,4)(5,3)(5,4)
	Guilds	2.8159	0.0728	3.2628	**0.0073**	(1,3)(1,4)(1,5)
Depth (zone 3)	Species	9.5534	**0.0002**	5.9709	<u>**0.0002**</u>	(S,I)(I,D)(S,D)
	Guilds	10.474	**0.0013**	5.3630	<u>**0.0050**</u>	(S,D)(I,D)
Season	Species	−	−	1.1079	0.3248	−
	Guilds	−	−	0.8998	0.4991	−

Table 5
Influence of latitude zones and seasons within depth ranges on soft substrate assemblages. Data for areas shallower than 20 m were not available for all coastal zones. PERMANOVA tests using species and guild data. P-values were bolded when considered significant and underlined if significantly different multivariate dispersions were found in PERMDISP analyses.

	Data type	Depth range					
		0–20 m		20–100 m		100–200 m	
		pseudo-F	P	pseudo-F	P	pseudo-F	P
Latitude	Species	–	–	1.3225	0.2068	4.6681	**0.0002**
	Guilds	–	–	1.5845	0.1675	6.2984	**0.0004**
Season	Species	0.4537	0.9640	1.8793	<u>0.0701</u>	0.3470	0.9372
	Guilds	0.2813	0.9512	2.5696	<u>0.0568</u>	0.3041	0.8966

standing out in multivariate space, from seabream (*Diplodus* spp.) in rocky reefs and shallow sandy areas to pilchard (*Sardina pilchardus*) and horse mackerel (*Trachurus trachurus*) in shallow and intermediate soft substrates and hake (*M. merluccius*) and blue whiting (*Micromesistius poutassou*) in intermediate and deep soft-substrate assemblages.

For the same reasons, when using guild data, Spearman's correlations of guild proportions with the canonical axes of CAP (Fig. 5d) provide a clearer functional characterisation of the assemblage types. Winter spawners are mostly associated with deeper assemblages, while species from shallow soft substrates and rocky reefs are mostly summer spawners. Pelagic species characterise soft substrates and get more abundant in deeper areas while benthopelagic species are characteristic of shallow soft substrates. On rocky reefs most species are reef-associated. On the trophic category, omnivores and herbivores are more associated with reefs and zooplanktivores with shallow soft substrates. The average

Fig. 4. Canonical analysis of principal coordinates (CAP) of species data using datasets from the 100–200 m depth range. Axes are drawn to maximise the distinction between latitude zones 1 (north) to 5 (south). 83.34% variation was explained by $m = 2$ PCO axes. Correlations with canonical axes are only shown when $\rho > |0.6|$. For species code correspondence see Appendix A.

number of species per dataset (standard deviation in brackets) was 38 (14) for rocky reefs, 33 (16) for shallow soft substrates, 24 (8) for intermediate soft substrates and 24(9) for deep soft substrates.

4. Discussion

4.1. Comparability of methods and designs

This study was an exercise that aimed to characterise marine fish assemblages according to species composition and functional guilds, by relying exclusively on published data, an approach intended as a hands-on confrontation with the problems arising from such procedures.

The first issue when compiling existing data is the comparability of different sampling methods. In fact, the published data gathered comes from samples collected with pure ecological studies in mind, which most likely differ in design and purpose (Elliott et al., 1999; Hewitt et al., 2005; Olenin and Dawnys, 2004). On fish assemblage studies, it has been shown that sampling gear type and selectivity (Pasquaud et al., 2012; Willis et al., 1993), observational focus in visual census techniques (De Girolamo and Mazzoldi, 2001) and even the observer's experience (Kulbicki and Sarramégna, 1999) can significantly influence species composition estimates at a local scale. Likewise, aspects such as the number of replicates (Samoilys and Carlos, 2000), sampling periodicity (Pasquaud et al., 2012), variability of habitats sampled (Jones, 1988) and the spatial scale considered (Anderson and Millar, 2004; García-Charton et al., 2004; Jones, 1988) can affect sample variability and hence the power to detect trends and patterns.

In the present study, finding a common ground in terms of spatial scale and sampling-related issues was no easy task, with datasets having to be standardised by the smallest indivisible spatial (coastal zones) and temporal (seasons) scales found in published literature. Another important aspect is the type of compositional data that best fits the objective of each study, with fisheries-related data often being reported in biomass (Cabral et al., 2003), random path visual census in frequency of occurrence (Henriques, 1993; Maranhão et al., 2006) and even sometimes using a categorical scale of abundance (Almada et al., 2004). These cases were the exception, rather than the rule, but the bias due to the overestimation of large fish in biomass data and to the underestimation of rarer gregarious species in frequencies of occurrence had to be assumed, despite the fact that these datasets did fit the overall pattern in multivariate analyses. Given this, in order to make datasets comparable, species data were standardised by total and all measures were assumed to represent the relative "importance" of each species for that particular study. All the above mentioned adjustments led to a very limiting starting point, forcing conclusions to be drawn from large coastal areas and masking within-season variation into single entries in the database.

By looking at the comparison of methods for areas shallower than 20 m (see section 3.1), it is evident that achieving unquestionable comparability between datasets was an impossible task, and too much strictness in the exclusion criteria would have prohibited any characterisation. However, datasets from rocky reefs that came from studies with a clear focus on benthic fish (Rodrigues, 1993; Gonçalves, 2004) counted significantly less species, making them unsuited for assemblage characterisation, even at a large scale. The problem of the observer's focus in visual census techniques has been pointed out by many authors (Harmelin-Vivien et al., 1985; Willis, 2001), since focusing on demersal fish underestimates cryptobenthic fish and vice-versa. The results observed in the present study show that the lack of a consensual practice regarding visual census methods can be a strong argument against the use of published data for rocky reef
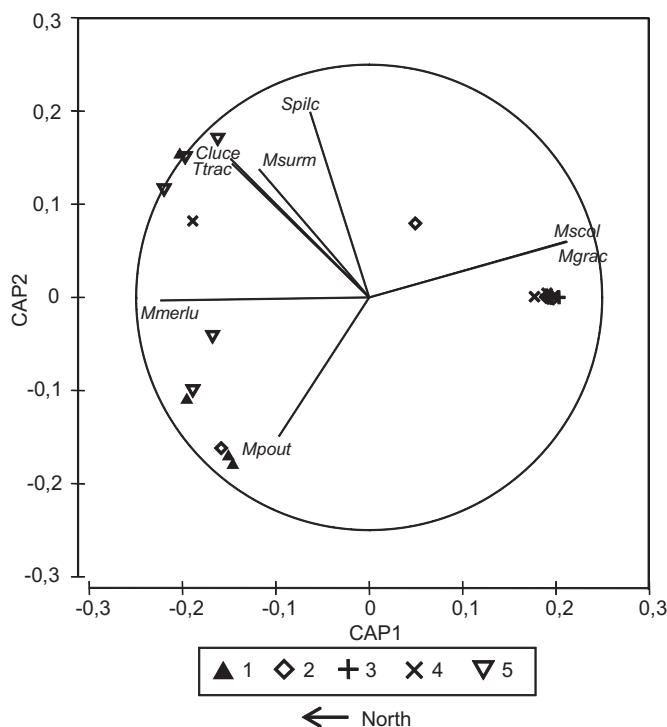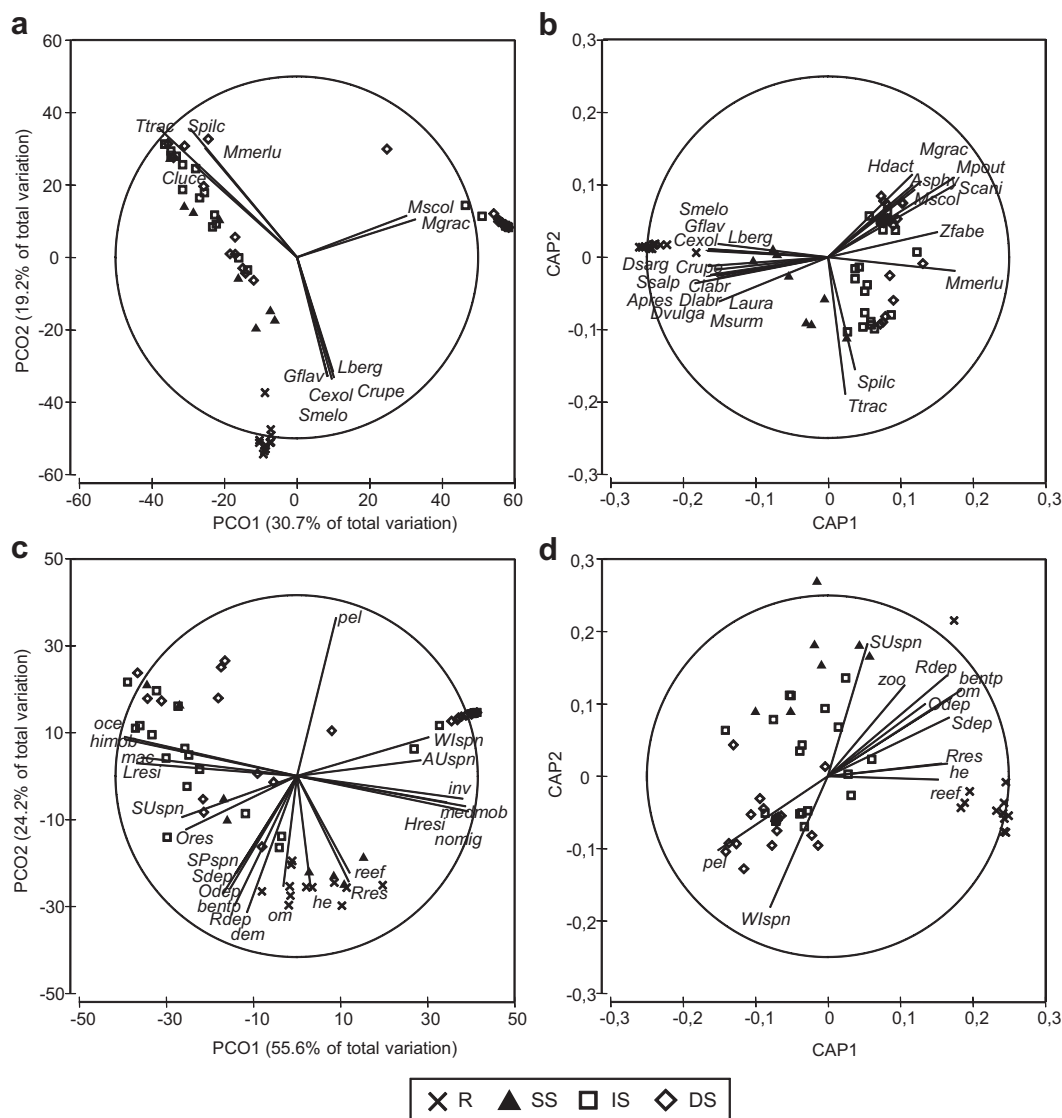
Fig. 5. Unconstrained (PCO) and constrained (CAP) ordination plots to discriminate among the defined assemblage types, using species (a,b) and guild (c,d) data. Cumulative explained variation was a) 49.89% c) 79.84% b) 99.48% by $m = 6$ PCO axes and d) 49.89% by $m = 3$ PCO axes. Correlations with canonical axes are only shown when $\rho > |0.6|$. For guild code correspondence see Table 2, for species codes see Appendix A.

characterisation purposes, and an approach that covers substrate, crevices and water column (e.g. De Girolamo and Mazzoldi, 2001) should always be the choice when characterising assemblages in a monitoring context.

Given the overall differences found between methods within the same coastal area, the decision to exclude datasets based on the average number of species, coupled with multivariate classification and ordination methods assured a certain level of functional and compositional similarity between datasets to be included in the exercise.

4.2. Characterisation of assemblage types: problems and achievements

Even considering all limitations, the compiled datasets allowed a rough functional characterisation to be achieved, by using multivariate analysis to test how different substrates, depth ranges and seasons affected both species and guild data.

Overall, substrate and depth were identified as the main factors responsible for differences in assemblage distribution at this spatial scale. The difference between soft substrates and rocky reefs is known and was expected prior to data analysis (e.g. Pihl and Wennhage, 2002), but it is also known that different structural characteristics within the same substrate, like different types of sediment (Demestre et al., 2000) or rocky reef areas of different complexity (García-Charton and Pérez-Ruzafa, 2001; Jones, 1988) can greatly affect the supported fish assemblage. These factors could not be tested from the published data analysed, even though it can be argued that a balance has to be achieved between the assessment scale and the applicability of the tools and monitoring procedures, in order to optimise costs (Caughlan and Oakley, 2001). Indeed, the coverage of a wide range of small-scale habitat variability with sufficient compositional homogeneity to compare relatively distant locations has been found in many studies (García-Charton et al., 2004; Jones, 1988), but the main issue here is the context and purpose of the management objectives. In fact, if a monitoring programme aims to detect anthropogenic impacts on coastal fish communities, the scale must be chosen so that the natural variability inherent to habitat types is smaller than the expected variation due to impacts (Osenberg et al., 1994).

Depth could not be tested for rocky reef data due to lack of detailed information, but incorporating this variability into an overall habitat unit at this spatial scale was acceptable. Within soft substrates, the range covered was much larger, and significant functional and compositional differences were found when treating depth as a quantitative gradient and a categorical factor. However, the overlap between depth ranges was evident in multivariate ordination, so the establishment of strict depth boundaries in a gradient can be an arbitrary decision with little ecological meaning. As depth increases, changes occur in water temperature, salinity, pressure, light intensity and other factors that affect fish distribution according to ecological needs and physiological tolerances (Rice, 2005). Demestre et al. (2000) and Catalán et al. (2006) observed that depth was the main limiting factor for species distribution on soft substrate of the north-western Mediterranean continental shelf and the studies on demersal assemblages by Gomes et al. (2001) and Sousa et al. (2005) also identified depth as one of the main factors influencing the distribution of fish, crustaceans and cephalopods on the Portuguese shelf and upper slope.

Based on the ordination plots, Bray–Curtis similarity values and the "leave-one-out" allocation success, it is evident that the most pronounced differences between assemblages occur when species data is used. This is due to the fact that species are directly affected by small-scale habitat characteristics (Rice, 2005), while guilds tend to suffer smaller variations in abundance as some species are replaced by others of the same guild. The fact that proportions of individual species are very sensitive to environmental variation makes the distinction between natural and pressure-induced changes very difficult. In addition, individual species, except in the case of indicator species, provide little information about the state or "health" of an assemblage, compared to ecological guilds (Elliott et al., 2007). However, as observed on within-group similarity values, though guild data can distinguish different substrates and depths at a relatively large biogeographic scale, smaller variations are probably more difficult to detect, and thus a careful selection of the guilds that are affected by each source of anthropogenic pressure is key for successfully managing coastal fish assemblages (Henriques et al., 2008a).

4.2.1. Rocky reefs (0–20 m deep)

Rocky reefs identified in the present study display typical warm-temperate characteristics (Almada et al., 1999; Henriques et al., 1999). In these areas, the increase in turbulence and the decrease in water temperature, photoperiod, prey availability, among other factors, in autumn and winter, favour spring and summer spawning (Almada et al., 1999). Due to the high productivity and complexity of rocky reefs, most species are very linked to the substrate throughout their whole lifecycle (Almada et al., 1999; García-Charton and Pérez-Ruzafa, 2001; Henriques et al., 1999; Pihl and Wennhage, 2002), hence the abundance of rock residents is characteristic of this substrate, making it vulnerable to impacts that negatively affect habitat integrity (Guidetti et al., 2002).

The herbivores trophic guild was also characteristic of rocky reef assemblages. However, this is mainly due to *Sarpa salpa*, the only species, among the most common, whose adults are almost exclusively herbivore. The occurrence of few herbivore species on temperate rocky reefs might be related to the seasonal variability of algal biomass not favouring exclusive herbivory (Horn and Ojeda, 1999), however, as Bellwood et al. (2003) have pointed out in a much more diverse system, a single species can dominate a key function in the community, thus reinforcing the need to use abundance data rather than species lists and to always assess functional integrity.

Due to the a higher exposure to dominant winds and wave action (Sousa et al., 2005), rocky reefs in the north coast of Portugal

(zones 1 and 2) are difficult to sample using underwater visual census (Henriques et al., 1999) and very few data were found for these areas, except for occasional species lists. This limitation led to year-round rocky reef data being mostly available in the centre and south, in areas that are sheltered from the predominant north-western winds (Lemos and Pires, 2004). Effects of latitude in rocky reefs were therefore not addressed in this exercise, but variation is known to exist due to the location of the Portuguese coast in the gradient between Mediterranean and Northern-Atlantic waters, and even climatic oscillations have been shown to affect the predominance of species with cold-temperate or warm-temperate affinities (Henriques et al., 2007).

4.2.2. Shallow soft substrate (0–20 m deep)

Multivariate analyses revealed a certain degree of similarity between shallow soft substrates and rocky reefs, probably due to factors associated with coastal productivity and to the frequent occurrence of shallow sandy areas near rocky reefs, with species known to occur on both substrates (Demestre et al., 2000; Prista et al., 2003). However, shallow soft-substrate assemblages were characterised by the co-occurrence of resident benthic species like *Mullus surmuletus* with highly mobile pelagic species like *T. trachurus* and *S. pilchardus*, as observed by Catalán et al. (2006) near the Guadalquivir river mouth on the Gulf of Cadiz. The fact that pelagic zooplanktivores like *S. pilchardus* exhibit gregarious behaviour explains the abundance of the zooplanktivore guild on these assemblages.

4.2.3. Deep and intermediate soft substrate (20–200 m deep)

The demersal soft-substrate surveys conducted by the IPIMAR were published in technical reports with detailed abundance data (although along fixed coastal zones and depth ranges) only for the period used in this exercise (1979–1980), being subsequently included in integrative studies and published in journal articles by Gomes et al. (2001) (1985–1988) and Sousa et al. (2005) (1989–1999), where a division of assemblage types was accomplished based on species composition. Since the present work was intended as an exercise, using twenty-year-old data was not a major concern, as it were the only usable data to cover the whole continental shelf, with winter, summer and spring surveys and different depth ranges.

The most abundant species belonging to deeper assemblages are more independent from the substrate and exhibit gregarious behaviour. The occurrence of gregarious species dominated assemblages 20–200 m deep, and the use of proportions emphasised these differences. Although the use of proportions was the only way of standardising data in the present study, standardising by area or using raw abundances should always be favoured. Another solution is to exclude pelagic species from datasets (Gomes et al., 2001; Sousa et al., 2005), thus revealing the underlying patterns of less abundant species but losing information regarding functional guild composition.

Besides the pelagic guild, winter spawners dominated intermediate and deep soft-substrate assemblages, as pelagic species on upwelling systems tend to spawn when offshore transport is minimal, with planktivore juveniles feeding during the summer upwelling period (Santos et al., 2001).

Assemblages from intermediate depths (20–100 m) were dominated by the pelagic species *T. trachurus* and *S. pilchardus* and deeper assemblages (100–200 m) were characterised by *M. merluccius*, a species occupying higher trophic levels. This increase in trophic level in offshore waters is typical of coastal upwelling systems, since offshore transport of primary productivity leads to a distribution pattern where species that feed on primary producers (e.g. *S. pilchardus*) are closer to the coastline and higher

trophic levels place further away (Gomes et al., 2001; Vinogradov and Shushkina, 1978).

Although deep soft-substrate assemblages are known to be more stable (Sousa et al., 2005), there is a known influence of seasonality on rocky reefs (Almada et al., 1999), shallow soft-substrates (Cabral et al., 2003) and intermediate soft-substrates (Gomes et al., 2001). However, no significant differences between seasons were found on the species and guild composition of these assemblages. This is likely due to the fact that each sample in the analysis represented a whole season, with within-season variation being represented by different studies, with different methods, in different locations. This likely led to too much variation and not enough replicates to achieve sufficient power to detect differences. Another possible explanation, as found by Pihl and Wennhage (2002), is that seasonal differences can affect the total number of individuals, thus the use of abundance proportions may mask those effects.

A non-linear relationship with latitude zones was found in deep soft-substrate assemblages, with snipefish (*Macroramphosus* spp.) dominating zone 3. The explanation for this fact was attributed by Marques et al. (2005) to the presence of the Setúbal Canyon, but also the Cascais and Nazaré Canyons might have an important role in extending the distribution of these species into areas closer to the coast. Moreover, *T. trachurus* and *M. merluccius* were less abundant on central coastal zones at this depth range, which has been attributed by Marques et al. (2005) and Sousa et al. (2005) to a predator–prey relationship, since the main prey of these species, *M. poutassou*, occurs mainly deeper than 200 m in the region off Lisbon.

An important aspect when using old datasets to characterise assemblages has to do with highly dominant species, which can change the functional characteristics of a community when a particular year is less favourable. In fact, the data used in the present study (1979–1980) correspond to a period of very high abundance of snipefish compared to the present state, and Marques et al. (2005) have pointed out a significant population decline due to unsuccessful recruitment in the year 2000 which, according to recent surveys, was maintained until present.

While the abundance of snipefish in some datasets dominated unconstrained ordination, the use of CAP attenuated those effects, thus allowing the characterisation of each habitat type without having to remove species or use data transformations, as the relative proportion of each species was deliberately conserved. Moreover, the inclusion of guild data on multivariate analysis provided important information on marine fish assemblages and established a link between the definition of management units and the development of monitoring tools.

5. Conclusions

Collecting published data can be useful in the case of social or economic indicators and statistics, or even ecological data which can be assumed somewhat stationary. In some cases, a few summarised values may be enough to support the design of a management plan. However, when dealing with data that can be highly variable in space and time, with very complex interactions, our study has demonstrated that summarising data masks a great deal of complexity. In this exercise, although assemblage types could be characterised to a certain degree using published data, it was not without a significant amount of doubtful assumptions and important information gaps.

Although we considered an extreme case, where no access to raw data is available, it served as a case study to highlight several problems that can be faced by coordinating entities, regardless of the type of monitoring objectives and ecosystem parameters to be measured. The main issues found can be summarised in the following topics:

1. Spatial and temporal scales were forced by the quality of the data, as important information regarding habitat complexity and environmental gradients are lost when data is summarised for publication. Some analyses failed to detect patterns that have been observed at regional and local scales.
2. The stricter the requirements for including datasets in the analyses, the more comparable they become, but the less the information available to support conclusions.
3. Different methods, designs and data types prevented the comparability of some studies and forced questionable assumptions (sections 2.2 and 3.1).
4. Missing data for some areas and time periods limited the analyses of spatial and temporal patterns.
5. Some distribution patterns could only be related to complex interactions between hydrology, habitat structure and biotic variables (such as predator–prey relationships).
6. Some older datasets were found to have different species compositions when compared to more recent observations of the same sites.

In order to avoid or minimise some of the problems demonstrated in this study, a number of guidelines that have been referred by other authors are highlighted:

1. The quantity and quality of available data should be exhaustively assessed before deciding to use them to support management decisions. The creation and maintenance of national databases of publicly available data can greatly improve this process (Hiscock et al., 2003).
2. Species lists are not useful for most monitoring and assessment programmes (de Jonge et al., 2006; Halpern and Floeter, 2008) and information on species abundances should always be favoured.
3. Besides species composition, functional aspects of the communities must be assessed when analysing patterns, defining management units and developing monitoring tools (de Jonge et al., 2006)
4. Spatial and temporal scales adopted have to depend on monitoring objectives and environmental and habitat variability must be quantified in order to permit the distinction between natural and anthropogenic variation (García-Charton and Pérez-Ruzafa, 2001; Anderson and Millar, 2004).
5. Pilot studies should always be a part of every planning phase, since sampling methods and designs can greatly influence the power to detect changes (Osenberg et al., 1994). All aspects, from reference values to warning thresholds should take into account a standard sampling design and should be able to adapt to changes in objectives or funding constraints.
6. Policy requirements should be viewed as an opportunity to acquire solid scientific knowledge on ecosystem patterns and responses (Elliott et al., 1999; de Jonge et al., 2006).

In the present study, even in a relatively small area, with all the effort employed in finding and filtering available data, an extensive list of issues was found. This can be seen as a wakeup call, stressing the importance of following the proposed guidelines and ultimately the need for funding in the planning phase of new management challenges, so that problems in monitoring designs can be anticipated, rather than corrected at a greater expense. Managing an area based on an inappropriate scale or using insufficient sampling can lead to assessment errors that ultimately carry environmental and socioeconomic consequences. It is on the best

interest of regions and countries to actually ensure environmental quality and sustainability, rather than fulfil requirements with the sole purpose of fulfilling them.

Acknowledgements

Appendix A. Correspondence between codes in the ordination plots (Figs. 4 and 5a,b) and species names.

Code	Species name
Apres	*Atherina presbyter* Cuvier, 1829
Asphy	*Argentina sphyraena* Linnaeus, 1758
Cexol	*Centrolabrus exoletus* (Linnaeus, 1758)
Cluce	*Chelidonichthys lucernus* (Linnaeus, 1758)
Crupe	*Ctenolabrus rupestris* (Linnaeus, 1758)
Dlabr	*Dicentrarchus labrax* (Linnaeus, 1758)
Dvulga	*Diplodus vulgaris* (Geoffroy Saint-Hilaire, 1817)
Dsarg	*Diplodus sargus* (Linnaeus, 1758)
Gflav	*Gobiusculus flavescens* (Fabricius, 1779)
Hdact	*Helicolenus dactylopterus* (Delaroche, 1809)
Laura	*Liza aurata* (Risso, 1810)
Lberg	*Labrus bergylta* (Ascanius, 1767)
Mgrac	*Macroramphosus gracilis* (Lowe, 1839)
Mmerlu	*Merluccius merluccius* (Linnaeus, 1758)
Mpout	*Micromesistius poutassou* (Risso, 1827)
Mscol	*Macroramphosus scolopax* (Linnaeus, 1758)
Msurm	*Mullus surmuletus* Linnaeus, 1758
Scani	*Scyliorhinus canicula* (Linnaeus, 1758)
Smelo	*Symphodus melops* (Linnaeus, 1758)
Spilc	*Sardina pilchardus* (Walbaum, 1792)
Ssalp	*Sarpa salpa* (Linnaeus, 1758)
Ttrac	*Trachurus trachurus* (Linnaeus, 1758)
Zfabe	*Zeus faber* Linnaeus, 1758

References

Abreu, S., 2005. Caracterização da ictiofauna associada a substratos móveis costeiros ao largo do Algarve Central. Thesis, University of Algarve, Portugal (in Portuguese).

Aiken, K.A., Kong, G.A., Smikle, S., Mahon, R., Appledoorn, R., 1999. The queen conch fishery on Pedro Bank, Jamaica: discovery, development, management. Ocean Coast. Manage. 42, 1069–1081.

Almada, V.C., Henriques, M., Gonçalves, E.J., 1999. Ecology and behaviour of reef fishes in the temperate north-eastern Atlantic and adjacent waters. In: Almada, V.C., Oliveira, R.F., Gonçalves, E.J. (Eds.), Behaviour and Conservation of Littoral Fishes. ISPA, Lisbon, Portugal, pp. 33–68.

Almada, V.C., Gonçalves, E.J., Henriques, M., 2002. Inventariação e ecologia da ictiofauna do substrato rochoso da costa Arrábida/Espichel. Instuto Superior de Psicologia Aplicada/Instituto da Conservação da Natureza, Lisbon, Portugal (in Portuguese).

Almada, V.C., Henriques, M., Gonçalves, E.J., Garcia, J., Henriques, S., 2004. Caracterização da Situação de Referência e Avaliação de Impactes para a Central Térmica de Ciclo Combinado proposta para a área de Sines. Estudo de Impacte Ambiental, Componente de Biologia Marinha. Hidroprojecto, Portugal (in Portuguese).

Almeida, O., 1997. Estudo comparativo das comunidades ictiológicas de um redife artificial e um recife natural vizinho na costa sul algarvia. Thesis, University of Algarve, Portugal (in Portuguese).

Anderson, M.J., 2001. A new method for non-parametric multivariate analysis of variance. Austral Ecol. 26, 32–46.

Anderson, M.J., 2006. Distance-based tests for homogeneity of multivariate dispersions. Biometrics 62, 245–253.

Anderson, M.J., Millar, R.B., 2004. Spatial variation and effects of habitat on temperate reef fish assemblages in northeastern New Zealand. J. Exp. Mar. Biol. Ecol. 305, 191–221.

Anderson, M.J., Robinson, J., 2003. Generalized discriminant analysis based on distances. Aust. NZ J. Stat. 45, 301–318.

Anderson, M.J., Willis, T.J., 2003. Canonical analysis of principal coordinates: a useful method of constrained ordination for ecology. Ecology 84, 511–525.

Anderson, M.J., Gorley, R.N., Clarke, K.R., 2008. PERMANOVA+ for PRIMER: Guide to Software and Statistical Methods. PRIMER-E, Plymouth, UK.

Bellwood, D.R., Hoey, A.S., Choat, J.H., 2003. Limited functional redundancy in high diversity systems: resilience and ecosystem function on coral reefs. Ecol. Lett. 6, 281–285.

Borja, Á., 2006. The new European marine Strategy Directive: difficulties, opportunities, and challenges. Mar. Pollut. Bull. 52, 239–242.

Borja, Á., Bricker, S.B., Dauer, D.M., Demetriades, N.T., Ferreira, J.G., Forbes, A.T., Hutchings, P., Jia, X., Kenchington, R., Marques, J.C., Zhu, C., 2008. Overview of integrative tools and methods in assessing ecological integrity in estuarine and coastal systems worldwide. Mar. Pollut. Bull. 56, 1519–1537.

Bray, J.R., Curtis, J.T., 1957. An ordination of the upland forest communities of southern Wisconsin. Ecol. Monogr. 27, 325–349.

Browman, H.I., Stergiou, K.I. (Eds.), 2004. MEPS Theme Section: perspectives on ecosystem-based approaches to the management of marine resources. Mar. Ecol. Prog. Ser. 274, 269–303.

Cabral, H., Duque, J., Costa, M.J., 2003. Discards of the beach seine fishery in the central coast of Portugal. Fish. Res. 63, 63–71.

Cabral, H.N., Fonseca, V.F., Gamito, R., Gonçalves, C.I., Costa, J.L., Erzini, K., Gonçalves, J., Martins, J., Leite, L., Andrade, J.P., Ramos, S., Bordalo, A., Amorim, E., Neto, J.M., Marques, J.C., Rebelo, J.E., Silva, C., Castro, N., Almeida, P.R., Domingos, I., Gordo, L.S., Costa, M.J., 2012. Ecological quality assessment of transitional waters based on fish assemblages in Portuguese estuaries: the Estuarine Fish Assessment Index (EFAI). Ecol. Indic. 19, 144–153.

Catalán, I.A., Jiménez, M.T., Alconchel, J.I., Prieto, L., Muñoz, J.L., 2006. Spatial and temporal changes of coastal demersal assemblages in the Gulf of Cadiz (SW Spain) in relation to environmental conditions. Deep Sea Res. II 53, 1402–1419.

Caughlan, L., Oakley, K.L., 2001. Cost considerations for long-term ecological monitoring. Ecol. Indic. 1, 123–134.

De Girolamo, M., Mazzoldi, C., 2001. The application of visual census on Mediterranean rocky habitats. Mar. Environ. Res. 51, 1–16.

de Jonge, V.N., Elliott, M., Brauer, V.S., 2006. Marine monitoring: its shortcomings and mismatch with the EU Water Framework Directive's objectives. Mar. Pollut. Bull. 53, 5–19.

Demestre, M., Sánchez, P., Abelló, P., 2000. Demersal fish assemblages and habitat characteristics on the continental shelf and upper slope of the north-western Mediterranean. J. Mar. Biol. Assoc. UK 80, 981–988.

Dethlefsen, V., Tiews, K., 1985. Review on the effects of pollution on marine fish life and fisheries in the North Sea. J. Appl. Ichthyol. 1, 97–118.

EC, 2008. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008, establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive). Off. J. EU L 164, 19–40.

Elliott, M., Fernandes, T.F., de Jonge, V.N., 1999. The impact of European Directives on estuarine and coastal science and management. Aquat. Ecol. 33, 311–321.

Elliott, M., Whitfield, A.K., Potter, I.C., Blaber, S.J.M., Cyrus, D.P., Nordlie, F.G., Harrison, T.D., 2007. The guild approach to categorizing estuarine fish assemblages: a global review. Fish Fish. 8, 241–268.

Forst, M.F., 2009. The convergence of integrated coastal zone management and the ecosystems approach. Ocean Coast. Manage. 52, 294–306.

Froese, R., Pauly, D. (Eds.), 2011. FishBase. World Wide Web Electronic Publication. www.fishbase.org (version 10/2011).

García-Charton, J.A., Pérez-Ruzafa, Á., 2001. Spatial pattern and the habitat structure of a Mediterranean rocky reef fish local assemblage. Mar. Biol. 138, 917–934.

García-Charton, J.A., Pérez-Ruzafa, Á., Sánchez-Jerez, P., Bayle-Sempere, J.T., Reñones, O., Moreno, D., 2004. Multi-scale spatial heterogeneity, habitat structure, and the effect of marine reserves on Western Mediterranean rocky reef fish assemblages. Mar. Biol. 144, 161–182.

Gomes, M.C., Serrão, E., Borges, M.F., 2001. Spatial patterns of groundfish assemblages on the continental shelf of Portugal. ICES J. Mar. Sci. 58, 633–647.

Gonçalves, L., 2004. Caracterização da ictiofauna criptobêntica da Ponta da Baleeira, Sagres. Comparação entre dois métodos de censos visuais. Thesis, University of Algarve, Portugal (in Portuguese).

Gower, J.C., 1966. Some distance properties of latent root and vector methods used in multivariate analysis. Biometrika 53, 325–338.

Guidetti, P., Fanelli, G., Fraschetti, S., Terlizzi, A., Boero, F., 2002. Coastal fish indicate human-induced changes in the Mediterranean littoral. Mar. Environ. Res. 53, 77–94.

Guidetti, P., Terlizzi, a, Fraschetti, S., Boero, F., 2003. Changes in Mediterranean rocky-reef fish assemblages exposed to sewage pollution. Mar. Ecol. Prog. Ser. 253, 269–278.

Halpern, B.S., Floeter, S.R., 2008. Functional diversity responses to changing species richness in reef fish communities. Mar. Ecol. Prog. Ser. 364, 147–156.

Harmelin-Vivien, M.L., Harmelin, J.G., Chauvet, C., Duval, C., Galzin, R., Lejeune, P., Barnabe, G., Blanc, F., Chevalier, R., Duclerc, J., Lasserre, G., 1985. The underwater observation of fish communities and fish populations. Methods and problems. Rev. Ecol. Terre Vie 40, 467–539.

Harrison, T.D., Whitfield, A.K., 2004. A multi-metric fish index to assess the environmental condition of estuaries. J. Fish. Biol. 65, 683–710.

Henriques, P., 1993. Estrutura e variabilidade da comunidade ictiológica do Carreiro da Fortaleza (Berlenga) e bio-etologia da reprodução de Parablennius pilicornis

Cuvier 1829 (Pisces: Blenniidae). Thesis, University of Lisbon, Portugal (in Portuguese).

Henriques, M., Gonçalves, E.J., Almada, V.C., 1999. The conservation of littoral fish communities: a case study at Arrábida coast (Portugal). In: Almada, V.C., Oliveira, R.F., Gonçalves, E.J. (Eds.), Behaviour and Conservation of Littoral Fishes. ISPA, Lisbon, Portugal, pp. 473—519.

Henriques, M., Almada, V.C., Gonçalves, E.J., 2007. Rapid shifts in a marine fish assemblage follow fluctuations in winter sea conditions. Mar. Ecol. Prog. Ser. 340, 259—270.

Henriques, S., Pais, M.P., Costa, M.J., Cabral, H., 2008a. Efficacy of adapted estuarine fish based multimetric indices as tools for evaluating ecological status of the marine environment. Mar. Pollut. Bull. 56, 1696—1713.

Henriques, S., Pais, M.P., Costa, M.J., Cabral, H., 2008b. Development of a fish-based multimetric index to assess the ecological quality of marine habitats: the Marine Fish Community Index. Mar. Pollut. Bull. 56, 1913—1934.

Hewitt, J.E., Anderson, M.J., Thrush, S.F., 2005. Assessing and monitoring ecological community health in marine systems. Ecol. Appl. 15, 942—953.

Hiscock, K., Elliott, M., Laffoley, D., Rogers, S., 2003. Data use and information creation: challenges for marine scientists and for managers. Mar. Pollut. Bull. 46, 534—541.

Holthus, P.F., Maragos, J.E., 1995. Marine ecosystem classification for the tropical island Pacific. In: Maragos, J.E., Peterson, M.N.A., Eldredge, L.G., Bardach, J.E., Takeuchi, H.F. (Eds.), Marine and Coastal Biodiversity in the Tropical Island Pacific Region. Species Management and Information Management Priorities, vol. 1. East-West Center, Honolulu, Hawaii. USA, pp. 239—278.

Horn, M.H., Ojeda, F.P., 1999. Herbivory. In: Horn, M.H., Martin, K.L.M., Chotkowsky, M.A. (Eds.), Intertidal Fishes-Life in Two Worlds. Academic Press, San Diego. USA, pp. 197—222.

IPIMAR, 1980. Estudo dos Recursos Demersais da Costa Continental Portuguesa. Cruzeiro N/790604, N/E "Noruega". Junho 1979. INIP, Lisboa, Portugal (in Portuguese).

IPIMAR, 1981a. Estudo dos Recursos Demersais da Costa Continental Portuguesa. Relatório da 2a Campanha "Noruega". Outubro/Novembro 1979. Relatório do Instituto Nacional de Investigação das Pescas n° 2, Lisboa, Portugal (in Portuguese).

IPIMAR, 1981b. Relatório do Cruzeiro tipo "Demersal" 020130380 Realizado a bordo do Navio "Noruega" entre 5 e 17 de Março de 1980. Relatório do Instituto Nacional de Investigação das Pescas n° 4, Lisboa, Portugal (in Portuguese).

IPIMAR, 1982. Estudo dos Recursos Demersais da Costa Continental Portuguesa. Relatório da 3a Campanha N/02016058 "Noruega". Maio/Junho 1980. Relatório do Instituto Nacional de Investigação das Pescas n° 13, Lisboa, Portugal (in Portuguese).

IPIMAR, 1984. Estudo dos Recursos Demersais da Costa Portuguesa. Relatório da 5ª Campanha, N/E "Noruega". Cruzeiro 020221080. Outubro 1980. Relatório do Instituto Nacional de Investigação das Pescas N° 28, Lisboa, Portugal (in Portuguese).

Jennings, S., 2005. Indicators to support an ecosystem approach to fisheries. Fish Fish. 6, 212—232.

Jones, G.P., 1988. Ecology of rocky reef fish of north-eastern New Zealand: a review. New Zeal. J. Mar. Fresh. 22, 445—462.

Katsanevakis, S., Stelzenmüller, V., South, A., Sørensen, T.K., Jones, P.J.S., Kerr, S., Badalamenti, F., Anagnostou, C., Breen, P., Chust, G., D'Anna, G., Duijn, M., Filatova, T., Fiorentino, F., Hulsman, H., Johnson, K., Karageorgis, A.P., Kröncke, I., Mirto, S., Pipitone, C., Portelli, S., Qiu, W., Reiss, H., Sakellariou, D., Salomidi, M., Hoof, L., Vassilopoulou, V., Fernández, T.V., Vöge, S., Weber, A., Zenetos, A., Hofstede, R.T., 2011. Ecosystem-based marine spatial management: review of concepts, policies, tools, and critical issues. Ocean Coast. Manage. 54, 807—820.

Kulbicki, M., Sarramégna, S., 1999. Comparison of density estimates derived from strip transect and distance sampling for underwater visual censuses: a case study of Chaetodontidae and Pomacanthidae. Aquat. Living Resour. 12, 315—325.

Lachenbruch, P.A., Mickey, M.R., 1968. Estimation of error rates in discriminant analysis. Technometrics 10, 1—11.

Legendre, P., Anderson, M.J., 1999. Distance-based redundancy analysis: testing multispecies responses in multifactorial ecological experiments. Ecol. Monogr. 69, 1—24.

Lemos, R.T., Pires, H.O., 2004. The upwelling regime off the West Portuguese Coast, 1941—2000. Int. J. Climatol. 24, 511—524.

Maranhão, P., Franco, J., Castanheira, F., Castro, N., Rodrigues, N., 2006. Peixes da Reserva Natural das Berlengas (Portugal). XIV Simpósio Ibérico de Estúdios de Biología Marina, 12—15 September, Barcelona, Spain (in Portuguese).

Marques, V., Chaves, C., Morais, A., Cardador, F., Stratoudakis, Y., 2005. Distribution and abundance of snipefish (Macroramphosus spp.) off Portugal (1998—2003). Sci. Mar. 69, 563—576.

McArdle, B.H., Anderson, M.J., 2001. Fitting multivariate models to community data: a comment on distance-based redundancy analysis. Ecology 82, 290—297.

Musick, J.A., 1999. Criteria to define extinction risk in marine fishes. Fisheries 24, 6—14.

Olenin, S., Dawnys, D., 2004. Coastal typology based on benthic biotope and community data: the Lithuanian case study. In: Schernewski, G., Wielgat, M. (Eds.), Baltic Sea Typology. Coastline Reports, vol. 4. EUCC, Germany, pp. 65—83.

Osenberg, C.W., Schmitt, R.J., Holbrook, S.J., Abu-Saba, K.E., Flegal, A.R., 1994. Detection of environmental impacts: natural variability, effect size, and power analysis. Ecol. Appl. 4, 16—30.

Pasquaud, S., Brind'Amour, A., Berthelé, O., Girardin, M., Elie, P., Boët, P., Lepage, M., 2012. Impact of the sampling protocol in assessing ecological trends in an estuarine ecosystem: the empirical example of the Gironde estuary. Ecol. Indic. 15, 18—29.

Pihl, L., Wennhage, H., 2002. Structure and diversity of fish assemblages on rocky and soft bottom shores on the Swedish west coast. J. Fish. Biol. 61 (Suppl. A), 148—166.

Prista, N., Vasconcelos, R.P., Costa, M.J., Cabral, H., 2003. The demersal fish assemblage of the coastal area adjacent to the Tagus estuary (Portugal): relationships with environmental conditions. Oceanol. Acta 26, 525—536.

Rice, J.C., 2000. Evaluating fishery impacts using metrics of community structure. ICES J. Mar. Sci. 57, 682—688.

Rice, J.C., 2005. Understanding fish habitat ecology to achieve conservation. J. Fish. Biol. 67 (Suppl. B), 1—22.

Ricketts, P., Harrison, P., 2007. Coastal and ocean management in Canada: moving into the 21st century. Coast. Manage. 35, 5—22.

Rodrigues, S., 1993. Distribuição espacial e temporal da comunidade ictiobentónica da Baía da Fortaleza de S. João Baptista (Reserva Natural da Berlenga). Thesis, University of Lisbon, Portugal (in Portuguese).

Roset, N., Grenouillet, G., Goffaux, D., Pont, D., Kestemont, P., 2007. A review of existing fish assemblage indicators and methodologies. Fish. Manag. Ecol. 14, 393—405.

Sainsbury, K.J., Punt, A.E., Smith, A.D.M., 2000. Design of operational management strategies for achieving fishery ecosystem objectives. ICES J. Mar. Sci. 57, 731—741.

Samoilys, M.A., Carlos, G., 2000. Determining methods of underwater visual census for estimating the abundance of coral reef fishes. Environ. Biol. Fish. 57, 289—304.

Santos, A.M.P., Borges, M.F., Groom, S., 2001. Sardine and horse mackerel recruitment and upwelling off Portugal. ICES J. Mar. Sci. 58, 589—596.

Smith, A.K., Ajani, P.A., Roberts, D.E., 1999. Spatial and temporal variation in fish assemblages exposed to sewage and implications for management. Mar. Environ. Res. 47, 241—260.

Sousa, P., Azevedo, M., Gomes, M.C., 2005. Demersal assemblages off Portugal: mapping, seasonal, and temporal patterns. Fish. Res. 75, 120—137.

Stål, J., Paulsen, S., Pihl, L., Rönnbäck, P., Söderqvist, T., Wennhage, H., 2008. Coastal habitat support to fish and fisheries in Sweden: integrating ecosystem functions into fisheries management. Ocean Coast. Manage. 51, 594—600.

Torgerson, W.S., 1958. Theory and Methods of Scalling. John Wiley & Sons, New York.

Vinogradov, M.E., Shushkina, E.A., 1978. Some development patterns of plankton communities in the upwelling areas of the Pacific Ocean. Mar. Biol. 48, 357—366.

Whitfield, A.K., Elliott, M., 2002. Fishes as indicators of environmental and ecological changes within estuaries: a review of progress and some suggestions for the future. J. Fish. Biol. 61, 229—250.

Willis, T.J., 2001. Visual census methods underestimate density and diversity of cryptic reef fishes. J. Fish. Biol. 59, 1408—1411.

Willis, D.W., Murphy, B.R., Guy, C.S., 1993. Stock density indices: development, use, and limitations. Rev. Fish. Sci. 1, 203—222.

Willis, T.J., Millar, R.B., Babcock, R.C., 2000. Detection of spatial variability in relative density of fishes: comparison of visual census, angling and baited underwater video. Mar. Ecol. Prog. Ser. 198, 249—260.