



Topographic complexity and the power to detect structural and functional changes in temperate reef fish assemblages: The need for habitat-independent sample sizes



Miguel Pessanha Pais^{a,*}, Sofia Henriques^a, Maria José Costa^{a,b},
Henrique Nogueira Cabral^{a,b}

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

^b Departamento de Biologia Animal, Faculdade de Ciências, Universidade de Lisboa, Campo Grande, 1749-016 Lisboa, Portugal

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ABSTRACT

Functional approaches have shown promising results to detect degradation in marine fish assemblages. However, background variability significantly affects the amplitude of change that is detectable by a monitoring plan, and failing to detect such changes can have devastating consequences and carry aggravated recovery costs due to unnoticed degradation. The present study aimed to understand the relationship between topographic complexity in temperate reefs and the power to detect variations in fish-based metrics. Underwater visual census of fish assemblages was performed using strip transects and a Monte Carlo simulation approach was used to generate a large number of replicates and simulate three alternative hypotheses representing different magnitudes of change. Statistical power to detect differences between null and alternative hypotheses was estimated through 10,000 Mann–Whitney tests for numbers of replicates ranging from 2 to 15. Power tended to vary with topographic complexity particularly with small and medium changes in metric values and when using small sample sizes. While power increased with complexity for most metrics, some showed decreasing trends. With a large effect, 5–15 transects were needed, depending on the metric, to stabilise power above 0.80 independently of habitat features. A power of 0.95, however, could not be achieved for most metrics in all sites, even when using 15 transects. The observation that the power to detect degradation and recovery in temperate reef fish seems to vary with habitat features means that a monitoring programme that is correctly planned for a particular area may not be directly applicable in a nearby reef. Adding to the need to maximise power in monitoring, this study highlights the need to take into account habitat variability in these calculations and estimate habitat-independent sample sizes that are appropriate for the scale and location of interest.

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1. Introduction

The increasing anthropogenic pressure on marine habitats has stressed the need to develop tools and methodologies to monitor and detect degradation in time to take appropriate measures. This is now a requirement of several international policies, as well as local management plans (e.g. Borja et al., 2010; Rogers and Greenaway, 2005). Fish assemblages in near-shore reefs are particularly vulnerable to anthropogenic pressure sources such as fishing, chemical and organic pollution and habitat loss, among others (e.g. Henriques et al., 2013a; McKinley and Johnston, 2010).

Due to the role of fish in the ecosystem, as well as their socio-economic importance (Holmlund and Hammer, 1999), failure to detect and act upon anthropogenic impacts can have devastating consequences. This has led to the development of tools and indicators that aim to constitute short-term warnings of community degradation (Pérez-Domínguez et al., 2012), and a move from single-species approaches to community-based assessments (Nicholson and Jennings, 2004; Pais et al., 2013b). In addition, fish species can be assigned to functional guilds that have been shown to respond more predictably to stress, aiding in the distinction between natural and human-induced change (Elliott et al., 2007). However, these tools and measures are often not linked to standardised sampling methods and the adequacy of the applied methodology to the location to be monitored or the issue to tackle is rarely assessed (de Jonge et al., 2006).

* Corresponding author. Tel.: +351 217500826; fax: +351 217500207.
E-mail address: mppais@fc.ul.pt (M.P. Pais).

When establishing a monitoring programme, the goal is to be able to detect a signal in a pool of background noise. In the case of fish assemblages, a signal can be a change in abundance of a particular species, an increase or decrease in species richness, or a change in abundance of a functional guild (e.g. [García-Charton et al., 2000](#); [Guidetti et al., 2002, 2003](#); [Henriques et al., 2013a](#); [Maxwell and Jennings, 2005](#); [McKinley and Johnston, 2010](#)). Besides natural variation, sampling error also plays an important part in background noise. In underwater visual census, the most common sampling method for reef fish, this includes systematic errors due to low detectability, and random errors, mainly due to the mobility of fish around the sampling units ([Irigoyen et al., 2013](#); [Samoilys and Carlos, 2000](#)).

When detecting an impact on fish communities, the same principles of any statistical test must be applied, and the concepts of type I and type II errors must be brought into context, as well as their practical consequences ([Peterman, 1990](#); [Fairweather, 1991](#)). In a monitoring context, the probability of committing a type I error (α) translates as the probability of detecting an impact where there is none, while the probability of a type II error (β) is the probability of failing to detect an impact that is occurring ([Fairweather, 1991](#); [Mapstone, 1995](#); [Osenberg et al., 1994](#)). A type I error will lead to managers taking action where none is needed, with consequences for industries that constitute the main sources of impact, while a type II error can be devastating for marine communities and the costs of recovery will be aggravated as a result of long periods of unnoticed degradation ([Maxwell and Jennings, 2005](#)). Therefore, failing to detect an impact carries a cost that is usually higher than the cost of wrongly detecting an impact ([Fairweather, 1991](#); [Mapstone, 1995](#)).

Statistical significance, however, is not a synonym of biological relevance ([Mapstone, 1995](#); [Steidl and Thomas, 2001](#)), and there is a need to define the magnitude of change that is relevant in a monitoring context. This magnitude can be defined as the minimum impact threshold that would require management actions and must therefore be detected by a monitoring plan ([Mapstone, 1995](#); [Munkittrick et al., 2009](#)). The link between statistical significance and biological relevance is in practice established by $1 - \beta$, or the probability of detecting a true (biologically relevant) impact ([Thomas and Juanes, 1996](#)). In statistical terms, this is known as power, a probability that can be increased by increasing the number of replicates, reducing among-sample variability (e.g. increasing the size of each transect), increasing the minimum range of variation to be detected (effect size) or increasing the probability (α) of committing a type I error ([Steidl and Thomas, 2001](#)). Establishing a balance among all the variables that interact with power is the goal of statistical power analysis ([Cohen, 1988](#)).

For parametric tests, an estimation of power is relatively straightforward to obtain based on values for α , variance and expected effect size ([Cohen, 1988](#)). Thus, even a simple exercise based on previous studies can help establish a more suited sampling effort. However, ecological data are known to often fail to fulfil assumptions for parametric tests ([Olden et al., 2008](#)) and the popularity of non-parametric or semi-parametric alternatives is growing ([Anderson and Millar, 2004](#)). For non-parametric approaches there is considerably less information available on the estimation of power, and not even [Cohen \(1988\)](#) mentions power calculations for such tests, which may be part of the reason why so many experiments and monitoring programmes are ignoring β ([Maxwell and Jennings, 2005](#)).

In the assessment of reef fish assemblages through underwater visual census there is no agreement regarding the number of replicates or the size of each transect (e.g. [Mosquera et al., 2000](#)). In addition, several environmental features are known to affect fish assemblage characteristics in temperate reefs, particularly reef complexity (e.g. [García-Charton and Pérez-Ruzafa, 2001](#); [Gratwicke](#)

[and Speight, 2005](#); [Pais et al., 2013a](#)). However, the effect of habitat complexity on the variability of estimates has only been scarcely addressed in the literature (e.g. [Cote et al., 2013](#); [Mouillot et al., 1999](#)) and, to the authors' knowledge, there is no study that directly relates habitat complexity with statistical power, particularly in light of the recent adoption of functional approaches, which are expected to reduce the sampling effort necessary to detect change ([McClanahan et al., 2007](#)).

The present study analysed the relationship between topographic complexity and the power to detect structural and functional changes in temperate reef fish assemblages along the Portuguese coast. The main goals were (1) to identify trends in the relationship of individual fish community metrics with topographic complexity and (2) to calculate the number of replicates needed to achieve enough power to detect meaningful changes.

2. Materials and methods

2.1. Study area

Rocky reefs located on the Portuguese coast (SW Europe) are typical of warm-temperate regions and are affected by summer upwelling ([Almada et al., 1999](#); [Henriques et al., 2007](#)). With both Northern-Atlantic and Mediterranean influence, the area provides conditions for a relatively large number of fish species to occur, and thus it can be used to study the influence of local environmental factors on settlement and survival, ultimately determining the best potential assemblage supported by a given habitat. For this study, 14 sites were sampled along approximately 300 km in the centre and south coasts ([Fig. 1](#)), located at depths between 3 and 11 m and covering a wide range of environmental conditions and topographic complexity, from flat layers of rock to large blocks with caves and crevices. In order to minimise the effect of anthropogenic disturbance on the results, care was taken to avoid sites under direct influence of impact sources.

2.2. Fish assemblage surveys and topographic complexity

Sampling was performed during summer in order to optimise the potential of each site to support fish assemblages ([Henriques et al., 2013b](#)). Data were collected through SCUBA-diving visual census during daytime and replicates within each site covered several tidal states in order to reflect local variability. In order to minimise underestimation of fish abundances, each replicate consisted of two passes, one directed to demersal species (50 m × 2 m) and another to cryptobenthic species (50 m × 1 m) ([Cheal and Thompson, 1997](#); [De Girolamo and Mazzoldi, 2001](#)). The first pass was performed while unreeling a 50 m thin rope, with an average swimming speed of 8 m per minute, with cryptobenthic fish sampled in the opposite direction, while reeling the transect, by searching in crevices and under cobbles ≤ 20 cm in diameter (average speed of 3 m per minute). On cryptobenthic transects, the families Blenniidae, Bothidae, Batrachoididae, Callionymidae, Congridae, Gadidae (subfamilies Lotinae and Phycinae), Gobiiesocidae, Gobiidae, Muraenidae, Scorpaenidae, Scophthalmidae, Soleidae, Syngnathidae and Tripterygiidae and the species *Ctenolabrus rupestris* and *Labrus mixtus* were counted. Due to ontogenic variations in behaviour, fish smaller than 5 cm (total length) from the genus *Symphodus* were also counted on cryptobenthic transects. All others were counted on demersal transects. For each replicate, both passes were merged by standardising abundance per square metre of transect. A total of six transects were performed per site by two experienced observers (three by each observer), starting each time at a random point and allocated to pre-determined depth intervals (3–6 m and 8–11 m), according to each site's

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