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Effects of a spatial closure on highly mobile fish species: an assessment using pelagic stereo-BRUVs

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ABSTRACT

The effects of a spatial area closure on pelagic fish assemblages within the Houtman Abrolhos Islands were assessed using mid-water pelagic stereo-BRUVs. The spatial area closure within the Easter group of the Houtman Abrolhos Islands was found to have no significant effect on the species composition and relative abundance of pelagic fish assemblages. The most abundant demersal target species recorded was pink snapper (*Chrysophrys auratus*) and, individuals measured within the spatial closure were significantly larger than those sampled in the area open to fishing. This spatial area closure is of moderate size (22.3 km^2), but the spatial management of highly mobile species may require larger area closures than those for reef-associated species. The monitoring of pelagic species both in large and small spatial area closures is required in order to better understand how mobile species respond to this management strategy. Some species were only recorded in relatively low numbers using pelagic stereo-BRUVs. Moreover, some of the highly mobile pelagic fish species, such as tunas, mackerel and some shark species have proven difficult to measure, as these species were observed furthest from the camera systems. While pelagic stereo-BRUVs are an effective fishery-independent approach to monitor spatial fishing closures, improving the power of replicates by pooling individual deployments, and increasing the attraction rate of pelagic fish to the stereo-cameras, will enhance their performance in future studies.

1. Introduction

Spatial area closures such as marine reserves, fish habitat protection areas and other forms of spatial fishing closures have been widely implemented around the world in coastal areas to protect a broad range of species from exploitation and for biodiversity conservation (Dichmont et al., 2013; Edgar et al., 2014; Russ and Alcala, 2011). However, species with different life histories and ecological traits are likely to respond differently to protection (Claudet et al., 2010). There has been extensive work undertaken on assessing the effects of spatial area closures on demersal or sedentary species (Babcock et al., 2010; Ballantine, 2014; Claudet et al., 2008). However, the methods used have often missed or not targeted the mid-water and, consequently, the response of mobile pelagic species to spatial closures is the subject of ongoing debate (Claudet et al., 2010; Davies et al., 2012; Game et al., 2009; Gruess et al., 2011; Kaplan et al., 2010). Highly mobile species are not expected to respond positively to discrete spatial fishing closures because these are often of limited size and theoretical evidence suggests that effective protection requires the areas closed to fishing to be larger than the home range of the individuals targeted (Gruess et al., 2011; Palumbi, 2004; Walters et al., 2007). However, recent studies provide evidence suggesting that mobile species may benefit from spatial area closures (Bond et al., 2012; Claudet et al., 2010; Goetze and Fullwood, 2013; Jensen et al., 2010; Knip et al., 2012; Pichegru et al., 2010).

Spatial management of pelagic environments in the open ocean, such as closing areas of the high seas to fishing, is being increasingly discussed (de Juan and Lleonart, 2010; Game et al., 2009, 2010; Grantham et al., 2011; Hyrenbach et al., 2000; Kaplan et al., 2010; Mills and Carlton, 1998; Norse, 2005; Sumaila et al., 2007; White and Costello, 2014). Simultaneously, large offshore marine spatial area closures are also increasingly being established around the world (Davies et al., 2012; Kaplan et al., 2014; Notarbartolo-Di-Sciara et al., 2008; Sheppard, 2010). The main challenges for the implementation of pelagic spatial area closures include: (1) the need for large closures in order to cover the broad home ranges of pelagic species (Kaplan et al., 2010); and (2) difficulties associated with compliance because the majority of pelagic environments (64%) occur outside national jurisdictions making enforcement difficult and expensive (Sumaila et al., 2007; White and Costello, 2014). Moreover, pelagic fish hotspots can be difficult to identify and are likely to vary in space and time (Hyrenbach et al., 2000; Norse, 2005). Research on pelagic ecosystems is often data poor and there is a lack of demographic and baseline data for many pelagic species (Claudet et al., 2010; Freon and Misund,

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1999; Worm et al., 2005). Research on pelagic fish presents difficulties for the collection and interpretation of survey data, and often relies exclusively on fisheries catch data that can lead to sampling biases and is not suitable for sampling in areas that are closed to fishing (Heagney et al., 2007; Ward and Myers, 2007).

Spatial monitoring, both inside and outside spatial area closures, is needed to quantify the effect of protection on fish assemblages (Murphy and Jenkins, 2010). It is essential to improve our capacity to monitor changes in marine communities, and particularly in pelagic ecosystems, using non-destructive and fishery-independent techniques where applicable (Claudet et al., 2010). Combining traditional sampling methods and emerging technologies like remote sensing, echo-sounder transects and Baited Remote Underwater stereo-Video systems (stereo-BRUVs) has been discussed as being the most effective way to improve the quality of data for spatial management (Murphy and Jenkins, 2010). For instance, pelagic stereo-BRUVs may have the potential to monitor the effects of spatial area closures on pelagic fish (Claudet et al., 2010; Heagney et al., 2007; Santana-Garcon et al., 2014a, 2014c), in the same way that stereo-BRUVs have been widely used to study such effects on demersal fishes (Goetze et al., 2011; Harvey et al., 2012a; McLean et al., 2010; Watson et al., 2009).

Baited video techniques are increasingly used to obtain estimates of biodiversity, relative abundance, behaviour and, when stereo-cameras are used, size and biomass of a range of marine species (Bailey et al., 2007; Harvey et al., 2012b, 2013; Langlois et al., in press; Mallet and Pelletier, 2014; Santana-Garcon et al., 2014b; White et al., 2013). This method uses bait to attract individuals to the field of view of the cameras so that individuals can be counted, identified and accurately measured (Dorman et al., 2012; Hardinge et al., 2013). BRUVs have proven to be a robust method for assessing fish community structure in deep water (Bailey et al., 2007; Zintzen et al., 2012), estuaries (Gladstone et al., 2012), tropical or temperate reefs (Harvey et al., 2012a, 2012b; Langlois et al., 2010; Unsworth et al., 2014) and the pelagic environment (Heagney et al., 2007; Santana-Garcon et al., 2014a, 2014c). As opposed to the commonly used benthic deployments set on the seafloor, pelagic stereo-BRUVs set the camera systems at a predetermined mid-water depth, which allows the study of pelagic and mobile fish assemblages that inhabit the water column (Santana-Garcon et al., 2014a).

Monitoring the response of mobile species to spatial management in coastal areas can contribute to understanding the effects of protection in larger offshore spatial area closures and improve the management of pelagic species in the future. Studying these effects in nearshore spatial area closures is logistically and economically more feasible than monitoring large offshore fishing closures (Edgar et al., 2014; Sumaila et al., 2007). For instance, the Houtman Abrolhos Islands, located off the mid-west coast of Western Australia, comprise a series of well studied spatial area closures established to protect valuable and vulnerable reef fish species (Dorman et al., 2012; Fitzpatrick et al., 2013; Harvey et al., 2012a; McLean et al., 2010; Nardi et al., 2004; Sumner, 2008; Watson et al., 2007, 2009, 2010). The Abrolhos are comprised of 4 island groups (North Island, Wallabi group, Easter group, and Pelsaert group) and each group has one area closed to fishing (these spatial area closures are termed Reef Observation Areas; ROAs). These areas have been closed to scalefish fishing since 1994, thus catching fish by line, spear or any other method is prohibited, but lobster pots are permitted. Periodic monitoring of the effects of protection on the fish assemblages and, particularly, on the demersal target species has shown a variety of responses over the years including an increase in abundance (Harvey et al., 2012a; Nardi et al., 2004; Shedrawi et al., 2014; Watson et al., 2007), increase in average fish size or biomass due to protection (McLean et al., 2010; Watson et al., 2009), or no significant difference between the assemblages inside and outside the fishing closures (Dorman et al., 2012). However, none of the studies in the area have targeted their sampling to the pelagic environment, thus the effect of these spatial area closures on highly mobile species remains largely unknown.

We sampled fish assemblages in the water column in areas open and closed to fishing using pelagic stereo-BRUVs. The aims of this study were to (1) explore the effects of protection on pelagic species in the Houtman Abrolhos Islands; (2) assess the potential of pelagic stereo-BRUVs as a monitoring technique for spatial area closures; and (3) contribute to the ongoing debate on the use of spatial area closures for the conservation and management of highly mobile species both in coastal waters and the high seas.

2. Materials and methods

2.1. Study area

The Houtman Abrolhos Islands are located 60 km offshore from the mid-west coast of Western Australia between 28° 15′ S and 29° 00′ S. In this study, pelagic stereo-BRUVs were used to survey fish assemblages in the mid-water at 6 sites in the Easter group (Fig. 1). Sites were selected haphazardly, off the reef edge (in the pelagic environment above the base of the reef slope) between 30 and 35 m deep. Three sites were within the area closed to fishing and three in areas where fishing is permitted. The Easter group ROA covers 22.29 km^2 and is the second largest spatial area closure at the Houtman Abrolhos Islands.

2.2. Sampling technique

The pelagic stereo-BRUVs used in this study were designed to be deployed, anchored and to remain at ~10 m depth in the water column (Fig. 2). The bait arm acts as a rudder and keeps the camera system facing downstream of the current. The use of ballast and sub-surface floats effectively reduces movement from surface waves and allows for control over the deployment depth. A full description of the deployment method is provided in Santana-Garcon et al. (2014a). The camera systems consisted of two Sony CX12 high definition digital cameras mounted 0.7 m apart on a steel frame and converged inwards at 8° to allow the measurement of fish length (Harvey et al., 2010). The bait consisted of 800 g of crushed pilchards (*Sardinops sagax*) in a wire mesh basket suspended 1.2 m in front of the cameras.

2.3. Experimental design

The experimental design consisted of 2 factors: *Status* (2 levels, fixed: Open and Closed to fishing) and *Site* (3 levels, nested in Status, random). Twelve replicate 2-hour deployments were conducted at each site. This sampling effort was considered suitable following the results obtained from Santana-Garcon et al. (2014a) that estimated the optimal soak time and replication required for sampling using pelagic stereo-BRUVs. Over 4 days of sampling, 72 mid-water stereo-camera systems were deployed allowing a minimum distance of 500 m between replicates sampled simultaneously, thus minimizing the potential for overlap of bait plumes and to reduce the likelihood of fish moving between replicates. Further research on bait plume dispersion in the mid-water is needed in order to confirm the minimum distance that should be allowed between deployments and to estimate the sampling area (Heagney et al., 2007; Rizzari et al., 2014). Samples were collected during daylight hours, allowing at least 1 h between sampling and sunrise or sunset, to avoid possible crepuscular variation in fish assemblages (Birt et al., 2012). Sampling was interspersed so that all 6 sites were being sampled at the same time in order to avoid differences between sites being confounded by the temporal variability of pelagic fish assemblages.