Baited remote underwater video survey of reef fish in the Stilbaai marine protected area, with an assessment of monitoring requirements.

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ABSTRACT

Long-term monitoring of changes in species abundance and community composition within marine protected areas (MPAs) is essential to assess whether conservation goals are being reached. The costs, logistics and sampling biases inherent to traditional monitoring methods limit sustainable monitoring in all MPAs along the South African coastline. Baited remote underwater video (BRUV) technology offers non-extractive monitoring with lower labour and cost requirements, whilst eliminating inter-observer variability and increasing statistical power. Species richness and relative abundance were assessed employing BRUV technology in the Stilbaai MPA in the Western Cape, South Africa. Multivariate analyses showed that patterns of species distribution and abundance across the MPA were best explained by variations in depth, sea temperature and reef profile. This result corroborated findings from traditional underwater visual census (UVC) and controlled angling surveys, confirming BRUV technology as a sound monitoring tool. Power analyses indicated the number of deployments required to detect an annual significant ($\alpha = 0.05$) doubling and a significant 20% change in population abundance with 80% power. Ubiquitous species such as the roman (*Chrysoblephus laticeps*) require 8 samples to detect a population doubling, whilst rare species such as the dageraad (*C. cristiceps*) require 135 samples. Species accumulation curves showed that a deployment time of 49 minutes was sufficient to sample 95% of species diversity in the Stilbaai MPA. The maximum number of fish of any species captured in a single video frame, referred to as Max N, was used as an index of abundance. Species-specific accumulation curves based on Max N data highlighted behavioural differences in approaches to the bait and species accumulation over time. BRUV technology is recommended for the establishment of an annual monitoring programme in the Stilbaai MPA, with a view to extending the use of this methodology to MPAs along the South African coastline.

CHAPTER 1

Literature Review

Overexploitation of fish stocks through fishing and the degradation of marine habitats is increasing in parallel with growing human population size, burgeoning development and the consequent anthropogenic impact on the oceans (Dulvy et al. 2003; Sala & Knowlton 2006; Worm et al. 2006; Götz et al. 2011). This depletion of ocean resources has linked biological and socio-economic implications that stem from accelerated biodiversity loss and impaired ecosystem service delivery (Lester et al. 2009; Levin et al. 2009). In light of increasing population pressure, human dependence on the ocean cannot be diminished; only managed (Pomeroy et al. 2006). As a result, the call for effective fisheries' management and conservation strategies has heightened interest in the use of marine protected areas (MPAs) as a spatial and temporal tool to mitigate the collapse of fish stocks, initiate the recovery of ecosystems and increase their resilience (Attwood et al. 1997; Sala et al. 2002; Carr et al. 2003; Lester et al. 2009).

Marine protected areas: the need for effective monitoring

MPAs facilitate the recovery of species age, sex and size structure, population abundance and delay species maturation and age at sex-change (Conover & Munch 2002; Halpern 2003; Barrett et al. 2007; Lester et al. 2009). MPAs have been shown as integral to the recovery of vulnerable and exploited species (Bennett & Attwood 1991; Willis et al. 2000; Follesa et al. 2008; Götz et al. 2009a). However, the role of MPAs extends beyond biodiversity conservation to include habitat protection, education, recreation and economic ventures, research comparisons and fisheries management (Hockey & Branch 1997; Dayton et al. 2000; Sala et al. 2002; Lubchenco et al. 2003; McGilliard et al. 2011).

In accordance with global trends, South Africa's marine biodiversity is threatened primarily by fisheries exploitation (Attwood et al. 2000; Lombard et al. 2004). The current MPA network encompasses a variety of management-types; namely, no-take zones, multi-purpose MPAs, RAMSAR sites, World Heritage Sites and a UNSECO biosphere reserve (Tunley 2009). However, South Africa once again mirrors global trends in that the delineation of many MPAs was *ad hoc* and opportunistic, and not based on a goal-orientated strategic assessment (Hockey & Branch 1994; Pressey 1994; Hockey & Branch 1997; Palumbi 2003; Turpie et al. 2000).

Additionally, consideration of reserve connectivity has been largely ignored, despite significant implications for genetic diversity, evolutionary potential and genetic resilience in light of climate change (Hauser et al. 2002; Palumbi 2003; McLeod et al. 2009). Given that reserve design should consider size and shape (Diamond et al. 1976; Simberloff & Abele 1976; McLeod et al. 2009), regions of endemism (Myers et al. 2000), species richness (Simberloff & Abele 1976), as well as irreplaceability and vulnerability to threat (Cowling et al. 1999), it is important to monitor existing underwater MPAs and ensure that their existence in their current format is defensible and effective (Colton & Swearer 2010).

The opportunistic design and delineation of protected areas can also increase the cost of implementing and monitoring reserves (Pressey 1994). With some South African MPAs well-established for at least 47 years, and a growing network already in place (Tunley 2009), it is essential that they be monitored to assess their efficacy in achieving biodiversity conservation, fisheries management goals and addressing non-extractive human use for tourism and bequest value (Kelleher 1996; Hockey & Branch 1997; Turpie et al. 2000).

The need for an additional monitoring technique in South Africa

MPA monitoring to assess the state of biological conservation encompasses the measurement of a range of physico-chemical variables, as well as biological variables such as species diversity and abundance patterns (Attwood 2003; Götz 2008; Tunley 2009). To this end, a variety of techniques have been developed to assess reef fish assemblages, from controlled angling and trawls to variations on underwater visual censuses (UVC) (Brock 1954; Willis 2001; Cappo et al. 2004; Götz et al. 2011).

Monitoring techniques should ostensibly evaluate the efficacy of a MPA by investigating the presence of keystone, indicator or vulnerable species, the abundance of commercially targeted species relative to nearby exploited areas, changes in the sex and size structure of a species' population and habitat diversity (Hockey & Branch 1997). Fundamental to the management of a MPA is the assessment of species conservation status, as well as where species are located (Colton & Swearer 2010). An effective monitoring solution should therefore at least provide a scientifically sound measure of species composition, relative or absolute species abundance and species size structure (Willis et al. 2000).

A variety of widely-accepted methods have been employed to document reef fish assemblages. However, biases exist in the selectivity of these sampling methods, highlighting why no one technique has emerged as an all-encompassing procedure (Stobart et al. 2007; Colton & Swearer 2010; Watson et al. 2010). It would appear, therefore, that the sampling technique chosen depends on the type of ecological questions posed, the management framework that guides research or the biology of the species under study (Willis et al. 2000; Watson et al. 2010). The existing suite of MPA monitoring techniques express biases that are the result of the behaviour of the species under observation, the diving experience and identification skills of the observer and factors intrinsic to the actual technique (Stobart et al. 2007; Colton & Swearer 2010).

UVC conducted during SCUBA dive transects can provide one of the more efficient means of obtaining absolute counts (Stobart et al. 2007). When a known area is divided into transects, a higher abundance of cryptic species is often recorded because directed searching is possible (Watson et al. 2005; Colton & Swearer 2010). However, the misidentification of species, incorrect estimation of species abundance and length, and inter-observer biases have been highlighted as shortcomings of non-destructive visual surveys (Brock 1982; Harvey et al. 2002; Harvey et al. 2004; Watson et al. 2010).

Advances in underwater filming have facilitated the development of diver-operated video (DOV) surveys which overcome most of these identification biases (Watson et al. 2010). However, the use of SCUBA divers, whether for UVC or DOV surveys, is fraught with problems (Stobart et al. 2007; Colton & Swearer 2010). The differential attraction to, and deterrence from, SCUBA divers by different fish species may bias both UVC and DOV surveys (Watson et al. 2010). Logistically, SCUBA transects are limited by depth and the amount of time spent underwater (restricting observation time and sampling units per day), as well as the number of skilled divers available for work (limited by scientific diving regulations) (Stobart et al. 2007). Safety issues, diver competency and the availability of days with suitable diving conditions further hinders the ease with which SCUBA monitoring can be organised regularly and sustainably (Stobart et al. 2007).

Surface-based monitoring avoids, to a greater extent, influencing fish behaviour by excluding observer presence (Willis et al. 2000). This is of particular interest in reef environments where species are adapted to fishery exploitation and predator behaviour, and where, as a result, data may be collected through remote techniques where many dive-based survey techniques are considered unsuitable (Bennett & Attwood 1991; Willis et al. 2000). However, the use of ichthyocides and explosives is both destructive to the habitat sampled and harmful to the species within it (Ackerman & Bellwood 2000; Cappo et al. 2004). The high post-release mortality rates associated with controlled angling (Götz et al. 2007) and fish trapping (Bernard pers. comm., SAEON) also raise questions as to their applicability in MPAs (Willis et al. 2000). This may conflict with MPA objectives and would be unsuitable for species-specific sampling where population numbers are either unknown, or low enough to be of conservation concern (Ackerman & Bellwood 2000; Willis et al. 2000).

Compromises in the data collected using any one of the afore-mentioned techniques are unavoidable, and biases are regarded as being within acceptable limits (Ackerman & Bellwood 2000). It is oftentimes for this reason that a combination of several methods is recommended for effective MPA monitoring (Cappo et al. 2004; Colton & Swearer 2010; Watson et al. 2010).

However, efforts to establish on-going MPA monitoring must consider several factors alongside the scientific integrity of the technique. The availability of skilled manpower for DOV and UVC surveys, and the extractive nature of angling, trapping and trawling, impacts the repeatability of monitoring efforts (Willis et al. 2000; Stobart et al. 2007). Additionally, monitoring efforts must be cost-effective and defensible to provide long-term data sets and contribute to the transparent protected area management essential to increase stakeholder support (Beaumont 1997; Hockey & Branch 1997; Rotheram et al. 2007; Charles & Wilson 2009). It is these logistical issues, rather than the scientific biases inherent in each method, that impedes their regular and sustainable use across all South African MPAs.

A technique that addresses the logistical problems, rather than the scientific biases in existing methods, would make a useful addition to South African MPA monitoring. Studies contest a basic dichotomy: a standardised monitoring methodology can be employed to compare data across different spatial and temporal scales (Kelleher 1996). However, it has also been suggested that, in place of a standardised methodology that assesses relative density for all species in an MPA, techniques should be tailored to account for known variability in the behaviour and habitat association of different species (Willis et al. 2000). Thus, a study's focus on reef fish assemblages must consider that these species are exploited by fisheries, adapted to predator behaviour, oftentimes resident and territorial (Attwood 2003).

Baited remote underwater video assessments

Baited remote underwater video (BRUV) surveys have become increasingly established as a non-destructive reef fish sampling technique, particularly in regions inaccessible to divers or where extractive sampling would compromise management objectives (Willis et al. 2000; Cappo et al. 2004; Watson et al. 2010). The system operates on the basic premise that reef fish are attracted to an area within the field of view of a remotely-controlled underwater camera using bait and their presence, numbers and behaviour are recorded for later analysis on land (Harvey et al. 2002; Cappo et al. 2004; Cappo et al. 2007).

The general BRUV system comprises a camera, mounted facing horizontally from the apex of a weighted, stainless steel tripod. A steel rod, acting as a bait arm, extends 1 m from the tripod to hold a bait canister 1 m above the ground in the camera's field of view. The tripod is connected to the boat by means of a rope, and a cable connects the camera to a video recording monitor which generates a live-feed on the boat. The camera is operated remotely using a surface control box.

In previous studies BRUV sampled higher species richness, a wider size range of families and a higher abundance of large-bodied and targeted species (Cappo et al. 2004; Watson et al. 2010). Whilst the technique is more limited by water clarity than angling or fish-trapping

(Cappo et al. 2004), BRUV operates in as little as one metre visibility – an improvement on visibility requirements for DOV and UVC. It has been suggested that stereo-BRUV sampling may merge some of the advantages of angling and UVC (Cappo et al. 2004). Studies have highlighted advantages of the technique as including the reduction in observer-biases and the collection of data with lower variance, increasing the statistical power with which a survey can detect differences in measured assemblage variables across space and time (Langlois et al. 2010).

Aside from the method's scientific integrity, cost-benefit analyses showed BRUV sampling as being more time and cost efficient than dive-transects, obtaining higher statistical power with lower manpower and boat requirements (Cappo et al. 2003; Watson et al. 2005; Langlois et al. 2010). The retention of video footage allows for independent re-analysis and also provides a permanent record against which long-term ecosystem and fish-assemblage change can be monitored (Parker et al. 1991; Langlois et al. 2010).

BRUV footage archiving has implications that extend beyond monitoring to include defensible future MPA design and current network expansion, public awareness and education. Underwater video footage is more accessible to the public than the traditional scientific output of graphs and statistics and may well prove a useful conservation tool in promoting MPAs (du Plessis pers. comm., Cape Nature). Fishermen may also be more likely to accept assessments based on visuals than obscure indices of abundance. Opportunities to increase support for MPAs outside the scientific community are pertinent most particularly in light of a global movement towards the increased use of MPAs as a conservation tool (Sanchirico et al. 2002; Halpern 2003; Palumbi et al. 2003; Dalton 2005; Helvey 2004).

Studies have highlighted the post-sampling analysis time involved with BRUV surveys as a disadvantage (Willis et al. 2000). This objection seems insufficient to detract from the gains in terms of time and cost efficiency during fieldwork compared to SCUBA transects (Langlois et al. 2010). The accuracy of size structure data is also questioned, but may be corrected by utilizing stereo-BRUV equipment that facilitates the use of digital measurement software (Abdo et al. 2006; Harvey et al. 2003). Time-related biases can arise where sites are not surveyed simultaneously (Willis et al. 2000). However, this may be remedied by developing low-cost equipment such that several cameras can be buoyed off and left to collect data simultaneously (Cappo et al. 2007). An absolute count of fish density cannot be obtained using BRUV surveys, because there is no effective means of quantifying the size of the survey-area, the extent of the bait plume and the sensory capacity of every species that visits the camera system (Colton & Swearer 2010). Instead, measures of fish density are given as relative abundance and are not unlike most catch-per-unit-effort (CPUE) measures (Willis et al. 2000; Cappo et al. 2004; Watson et al. 2010).

Reef fish: a target for conservation

The propensity for commercial and recreational fisheries to target reef fishes makes many of these species the subject of significant conservation concern (Götz et al. 2008). Life history characteristics of many South African reef fish species such as slow growth, sex-reversal, high longevity, residency and territoriality compounds their vulnerability to stock depletion and changes in assemblage composition through overfishing (Buxton 1993). However, obtaining data on these species is oftentimes problematic as a result of insufficiencies in sampling techniques (Willis et al. 2000). BRUV reef fish surveys are particularly efficient at assessing predatory and exploited fishes, recording higher elasmobranch species richness than other methods (Cappo et al. 2004; Stobart et al. 2007; Colton & Swearer 2010). Herbivorous fish attracted to activity around the bait canister, or entering the BRUV field of view, are also recorded (Cappo et al. 2003; Cappo et al. 2004).

Assessing patterns of habitat association

Stereo-BRUV surveys also include habitat analysis, and may be used to correlate patterns of reef fish species richness, abundance and composition with habitat type (Watson et al. 2010). Reef fish assemblages may be determined by the quality of available habitat for a variety of reasons: some features may provide respite from physical stressors; others may act as refugia from predators and competitors, whilst certain characteristics will determine the availability of food (Buxton & Smale 1989; Friedlander & Parrish 1998). It has been shown that rugosity (variation in the height of the seafloor) and profile will influence species richness and abundance (Gratwicke & Speight 2005). In a patchy reef mosaic, depth, temperature and relief were shown to be the most important predictors of reef fish distribution (Buxton & Smale 1989).

Thus, the structure and complexity of the available habitat may determine the heterogeneity of fish communities (Friedlander & Parrish 1998; Gratwicke & Speight 2005). Profile may play a role in determining food availability and, as a study on roman and red steenbras (*Petrus rupestris*) showed, in providing cover for predators and prey (Buxton & Smale 1989). Whilst depth appears to impact the availability of suitable prey, temperature may influence fish mobility and determine whether species emerge or seek refuge (Buxton & Smale 1989).

Not only does investigation of reef fish species habitat association contribute to scientific understanding, but it may aid more effective future MPA design and guide the establishment of a representative MPA network (Friedlander & Parrish 1998). The distribution of fish species across space may influence the significance with which changes in density can be detected in an area (Willis et al. 2000). For this reason, it may prove useful to establish the level of sampling effort required to detect within-reserve variability in relative fish abundance before comparisons are made with areas outside MPAs (Willis et al. 2000).

A cost-effective, repeatable monitoring technique will contribute to more efficient and sustainable monitoring of fish assemblages in South African MPAs. Currently, monitoring efforts are hindered by cost and the logistical problems associated with limited manpower in terms of commercial divers, skippers and dive supervisors. The correlation of habitat features with patterns of species abundance and occurrence can be used to inform the future design of MPAs in terms of their location, distance from neighbouring MPAs, size and shape.

CHAPTER 2

Characterising the Stilbaai Marine Protected Area: reef fish composition, relative abundance and distribution

INTRODUCTION

Marine protected areas (MPAs) have been shown to protect remaining adult fish populations and spawner biomass, and in so doing, augment populations outside protected areas by replenishing stocks (Roberts et al. 2005; Follesa et al. 2008; McLeod 2009). For this reason, they are considered useful as both a conservation and fisheries management tool (Conover & Munch 2002; Roberts et al. 2005; Gaines et al. 2010). This is of particular interest when one considers the sea bream family (Sparidae), one of South Africa's most dominant endemic fish families (Turpie et al. 2000). Their longevity, propensity to change sex and hold territories, coupled with their preference for water shallower than 200 m, makes them particularly vulnerable to overfishing by the hook and line fishery for which they are prized (Tilney et al. 1996; Turpie et al. 2000; Griffiths & Wilke 2002).

Proclaimed in 2008, Stilbaai is one of South Africa's youngest MPAs (Tunley et al. 2009) and is therefore requires a baseline assessment of its reef fish assemblage. Situated on the south-western Cape coast, where a paucity of suitable dive-days reduces the potential for monitoring using traditional underwater visual census (UVC) (du Plessis pers. comm., Cape Nature), the MPA is well suited to testing a new technique that would alleviate the logistical problems associated with MPA monitoring (Tunley 2009; Colton & Swearer 2010). Additionally, the popularity of Stilbaai as a tourism destination opens opportunities to archive BRUV footage for viewing in local tourism centres to publicize the functioning and value of the MPA (du Plessis pers. comm., Cape Nature).

Prioritising reef fish conservation

Reef fish are important targets for commercial and recreational fisheries in South Africa (Buxton 1992; Götz et al. 2008) and historic over-exploitation has consequently been detrimental to their persistence (Buxton 1992; Buxton 1993). The South African commercial and recreational fisheries have caused the near-collapse of several reef fish stocks (Griffiths 2000; Griffiths & Lamberth 2002; Sauer et al. 2006). Therefore, proper assessment of MPA efficacy in terms of reaching biodiversity conservation and fishery management goals should consider the presence and relative abundance of highly exploited fishery target species that show dwindling numbers (Stobart et al. 2007; McLeod et al. 2009).

Studies on several South African reef fish highlight their vulnerability and the difficulties in monitoring population responses to fishing pressure in a multi-user fishery where catches are often grouped together as 'redfish' (Buxton 1992, Götz et al. 2009b). Many sea breams tend to be highly resident in an area, increasing their vulnerability to overfishing outside MPAs (Potts & Cowley 2005), but recommending them for protection and monitoring in MPAs (Bennett & Attwood 1991).

Aside from their economic value, reef fish play an important ecological role in their environment (Hixon & Beets 1993; Götz et al. 2009b). Many of the exploited species prized by fisheries play an important trophic structuring role in the reef habitat (Götz et al. 2009b). Studies have highlighted differences in community composition between protected and exploited sites, with non-fishery species occurring in lower abundance inside MPAs when they share dietary preferences with fishery species (Götz et al. 2009b). The conservation of reef fish in MPAs therefore determines not only the continued economic viability of fisheries, but the overall functioning of South Africa's temperate reef systems (Götz et al. 2009b).

Understanding patterns of reef fish distribution and the implications for MPAs

Several studies on temperate reef fish link their distribution in a region to the influence of reef depth and profile (Choat 1982; Buxton & Smale 1989). Sea temperature and visibility have been shown to influence the activity rates of fish, as well as their chosen patterns of movement (Buxton & Smale 1989). The distribution and movement patterns of predatory species such as roman (*Chrysoblephus laticeps*), red steenbras (*Petrus rupestris*) and dageraad (*C. cristiceps*) have been explained by the abundance of suitable prey which are dependent on these environmental factors (Buxton & Smale 1989).

An understanding of the distribution of species within a MPA, in conjunction with their relative abundance and community composition, is important for effective management (Colton & Swearer 2010). The distribution of fish in a MPA can influence the detection of changes in species abundance, but if patterns of species distribution according to reef depth and profile are known, monitoring efforts can be directed, efficient and properly interpreted (Willis et al. 2000).

This study provides a first-time assessment of species diversity and relative abundance since the closure of the Stilbaai MPA. The aim of this chapter is to assess whether measured environmental variables can explain patterns of reef fish distribution, abundance and community composition within the MPA. This information can be reviewed to guide future management decisions and inform area-selection for possible expansion of the MPA.

MATERIALS AND METHODS

Study area

Situated west of Mossel Bay in the Western Cape, the Stilbaai MPA (Figure 1) is featured on the warm-temperate south coast and encompasses the Goukou estuary, sandy beaches, rocky shores and a shallow sandstone shelf (Tunley 2009). The MPA protects a coastline of 13.5 km from Noordkapperspunt to the historical fish traps, and 15.7 km of the Goukou estuary. The high-water mark defines the MPA boundary on land. The seaward boundary is delineated by lines running eastward from Noordkapperspunt to a point 4.2 km offshore, and back to the coast at the historical fishtraps. Skulpiesbaai, the Geelkrans reef and the Goukou estuary are no-take zones where all commercial, recreational and subsistence fishing is prohibited (hereafter referred to as restricted zones).

Stilbaai protected area management (Cape Nature) currently monitors estuarine fish populations and salinity, marine and terrestrial biodiversity, sea surface temperatures and human use of the MPA (du Plessis pers. comm., Cape Nature). Marine biodiversity is sampled using SCUBA UVC transects, controlled angling and underwater photo quadrates.

GPS-linked echosounder data from transects across the Stilbaai MPA provided spatially referenced depth measurements. Depth measurements were interpolated using the Very Important Point (VIP) method for ArcMap™, identifying VIP cells in the depth data matrix by calculating how well their value could be predicted by the values of their neighbour cells (Goodchild & Kemp 1990). Each point was taken to have eight neighbouring cells that form four opposite pairs; these were analysed by connecting a straight line between paired cells and calculating the distance to the central cell (Goodchild & Kemp 1990). The four resultant values were averaged and the central cell was classified as a VIP if this final value was larger than a set threshold. This was used to create a three-dimensional bathymetric contour map in ArcMap™ (ESRI 2006). Based on this information, a study area was selected within the restricted zone of the MPA and encompassed 11.3 km^2. Depth ranged from 5 m – 41 m, with sites shallower than 5 m excluded from sampling because the BRUV could not be deployed safely in the surf-zone. Sites deeper than 41 m lay outside the MPA.

Selection of sampling sites

A random selection of 50 paired latitude and longitude values were plotted within the delineated study area. Sea conditions and the available sampling time limited the number of samples that could be achieved within the study's timeframe. Thus, a list of sites in random order was generated and sites were sampled sequentially from this list. All sites were at least 200 m apart to prevent site-replication and allow for boat-drift on the anchor line at each site, maintaining independence of samples (*sensu* Langlois et al. 2010).

To avoid bias in sample site selection, a random approach that precluded *a priori* assumptions about reef fish habitat association was adopted. This study focused on refining a monitoring system that could be conducted in any of South Africa's existing or future MPAs where little may be known of reef profile and where no bathymetry data are available.

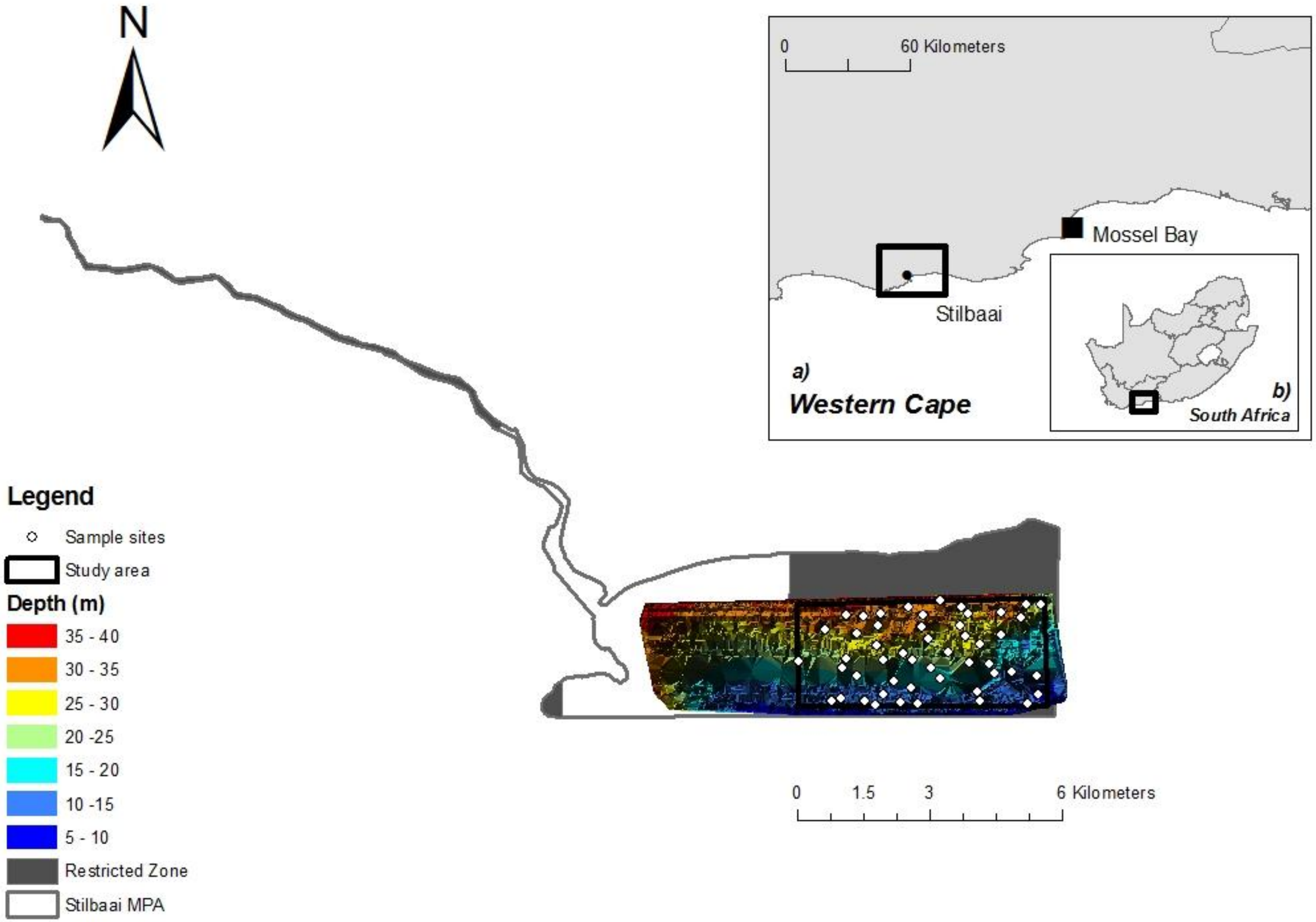

Figure 1. The Stilbaai Marine Protected Area (MPA) in the context of the Western Cape (a) and South Africa (b). Shaded regions indicate the restricted (no-take) zones of the MPA. Interpolated depth is graded and ranges from 5 m to 37 m. The study area is indicated by a grey rectangular outline and sampling sites are indicated in white.

BRUV tripod

A standard definition camera was mounted facing horizontally from the apex of a weighted, stainless steel tripod (20 kg). A stainless steel rod extended 1 m from the tripod and held a perforated PVC bait canister (130 mm X 110 mm with 10 mm perforations) 1 m above the ground in the camera's field of view. A rope maintained connection with the tripod and a cable connected the camera to a video recording monitor which generated a live-feed on the boat. The camera was operated remotely using a surface control box.

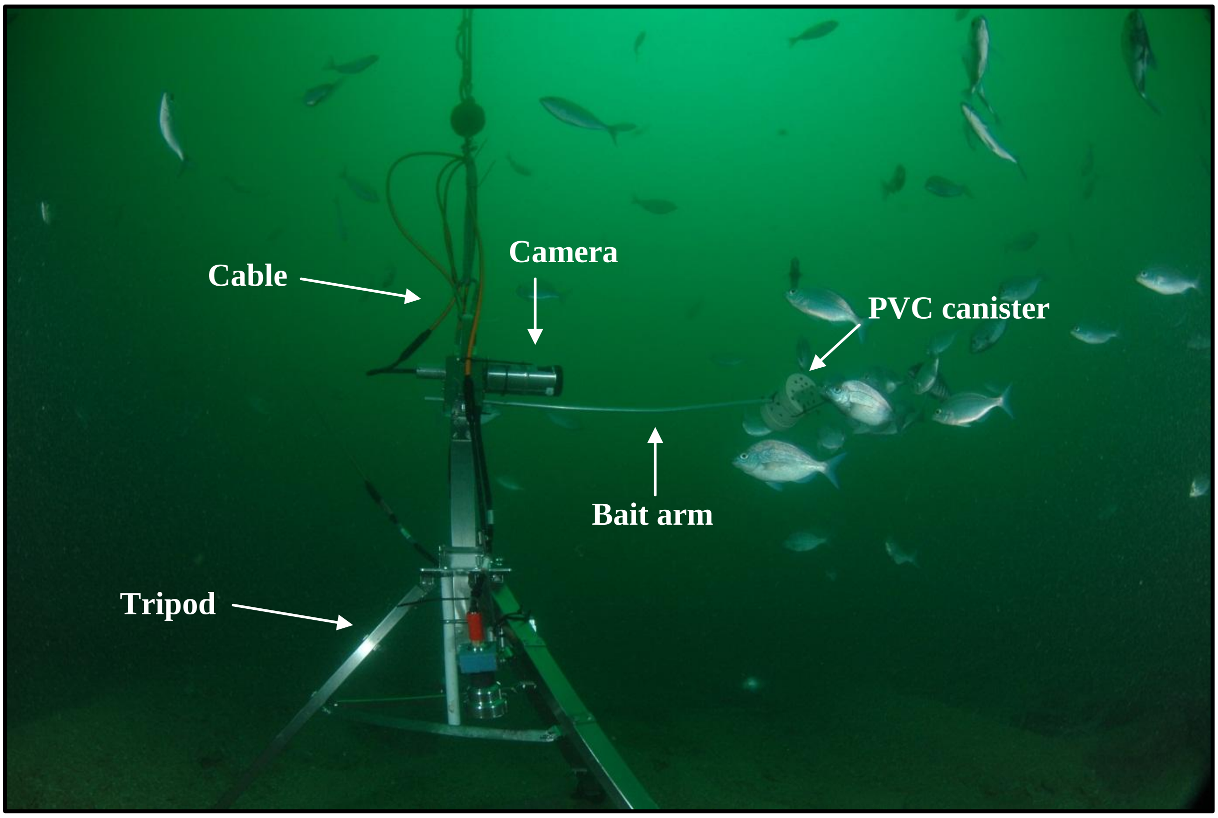

Figure 2. The deployed BRUV system with bait arm and PVC canister attached. Video camera focal length = 35 mm (54.4° horizontal angle of view; Bernard 2012).

Data collection

Environmental variables

Two temperature loggers attached to the BRUV tripod logged sea temperature every five minutes for the study's duration. Sea temperature at a sampling site was determined from known start and end times for each deployment and the median value taken for the duration of a deployment. Visibility was measured in metres with a secchi disc deployed from the boat before each BRUV deployment and depth was read from the boat's echo sounder and verified with map bathymetry data.

BRUV deployment

The BRUV tripod was assembled on land to ease operations at sea. Recording started when the tripod had settled on the seafloor. One kilogram of pilchard (*Sardinops sagax*) homogenate was used as bait (Cappo et al. 2004). Each deployment lasted 1 h to record between 90 % and 95 % of species, based on species accumulation curves from Castle Rock and Tsitsikamma MPAs (Bernard 2012) and a collection of recent studies (Watson et al. 2010; Colton & Swearer 2010; Langlois et al. 2010). Deployments were conducted from "Nkwazi", a 5.5 m Bill Fish (Hull design) boat with 2 x 60 hp outboard motors. A minimum of three persons was required for BRUV operation and deployment.

Habitat assessment

Reef profile and bottom sediment-type were described at each sampling site. The study's aim being to investigate a non-diving method, no SCUBA habitat assessments were conducted. A sediment grab proved inefficient at sampling sediment-type. Therefore, a GoPro® HD camera (Woodman Labs 2009) was set to video-setting and fitted using a GoPro® bicycle clamp to a stainless steel trident. This frame was lowered by rope to the sea floor and the downward-facing camera filmed the sediment for five seconds. Additionally, the frame was tilted forwards and then backwards, utilizing the GoPro's® wide-angle lens to capture broader habitat footage to assess reef profile and obtain closer images of the sediment-type.

Data analysis

Video analysis

To maintain consistency in identification, one researcher analysed videos using Apple QuickTime 7.7.1. All species in a video were recorded and a Max N measure was obtained for each species at every site. Max N is the maximum abundance of a species in any one frame for the duration of a video, to avoid recounting individuals that swim in and out of the camera's field of view (FOV) (Willis et al. 2003). Videos taken at sites with poor visibility (<3 m) were viewed in Final Cut Pro© where exposure and image colour and contrast could be adjusted, so that the shape and movement of different species could best be determined.

Habitat classification

Profile and bottom sediment-type were determined from footage recorded by both the BRUV and GoPro® system at each site. Profile was classified as either high or low for MDS and Cluster analyses. For subsequent BIOENV analysis, a profile score between zero and ten was assigned to each site (Table 1). The video screen was divided into three horizontal sections and a score assigned to each site based on reef height, using the proportion of reef that filled each of these three strata to assess relative height.

Table 1. Factors distinguishing profile and scores assigned according to reef profile grade.

Description	Height	Classification	Score
Reef higher than bait canister in BRUV FOV	>1m	High profile	6-10
Reef lower than bait canister, but visible in BRUV FOV	<1m	Low profile	3-5
Reef invisible in BRUV FOV, detected with GoPro®	<0.5m	Low profile	1-3
Sand invisible in BRUV FOV, detected with GoPro®	-	Low profile	0

Species summary table

To assess species ubiquity, the number of samples in which a species was recorded out of the total 29 samples was calculated and tabulated as 'frequency'. To illustrate the average maximum abundance which a species was recorded at, 'Mean Max N' was calculated as the sum of Max N counts per species divided by the number of deployments where the species was present to produce a Max N value. Shoaling behaviour would impact the variation in Max N values between sites for certain species, and so the standard deviation was calculated to assess the level of variation, or dispersion, from each Mean Max N value for each species. Relative abundance differed from 'Mean Max N' by taking the sum of Max N counts for each species and divided by the total number of sites sampled (29) (Colton & Swearer 2010). This accounted for possible over-representation of shoaling versus non-shoaling species and created an index showing the proportion in which species occurred relative to other species.

To assess what proportion of the species assemblage in Stilbaai was attracted to the BRUV's field of view, species were divided into four feeding guilds.

Table 2. Description of feeding guilds used to divide species according to their feeding biology (Smith & Heemstra 2003; Heemstra & Heemstra 2004; Branch et al. 2010).

Feeding guild	Feeding preferences
Omnivore	Benthic invertebrates, fish, algae
Invertebrate carnivore	Molluscs, worms, crustaceans, hydrozoans
Piscivore	Fish
Herbivore	Algae

Patterns of species' habitat association

Patterns of habitat association were investigated by analysing differences in species composition between sites in PRIMER-E version 6 (Clarke & Gorley 2006). Depth was split into three factor levels: 'shallow' (0 - 14 m), 'medium' (15 - 30 m) and 'deep' (31 - 45 m) and profile was recorded as 'high' or 'low'. The BRUV field of view was typically saturated with individuals of a species, with more individuals of that species in the area but unaccounted for in the camera's field of view. As such, the BRUV Max N method down-weighted superabundant species automatically through frame-saturation, making data transformation unnecessary.

A Bray Curtis cluster analysis and multidimensional scaling (MDS) plot assessed the similarity of species composition among the 29 sites, and ANOSIM tests on depth and profile assessed the significance of these two variables' influence on species composition. Data were transformed for a SIMPER analysis to explore which species' presence contributed to differences in species composition between different depth categories and profiles, and to assess which species characterised each of the depth strata and profiles. Transformation was necessary because the presence of dominant and ubiquitous species (steentjie and roman) could overshadow lesser abundant species which could be indicators of habitat type. Profile was graded for the BIOENV procedure that assessed which combinations of environmental variables best explain differences in species composition among sites (Clarke & Warwick 2001). Environmental data were normalised to ensure comparability of measures obtained in different metrics.

RESULTS

1. Environmental variables and fish species recorded in Stilbaai

A total of 29 BRUV stations were completed during seven fieldtrips to Stilbaai during the period 11 October to 30 November 2011. Site depths ranged from six to 38 m, averaging 22 (± 9.6 SD) m. Only one site was classified as sand, whilst 17 sites represented low profile reef and 10 sites were classed high profile reef. Visibility ranged from 2.5 to 10 m and averaged 6.2 (± 1.6 SD) m. Water temperature varied from 15 to 20°C, with an average of 17 (± 1.2 SD) °C.

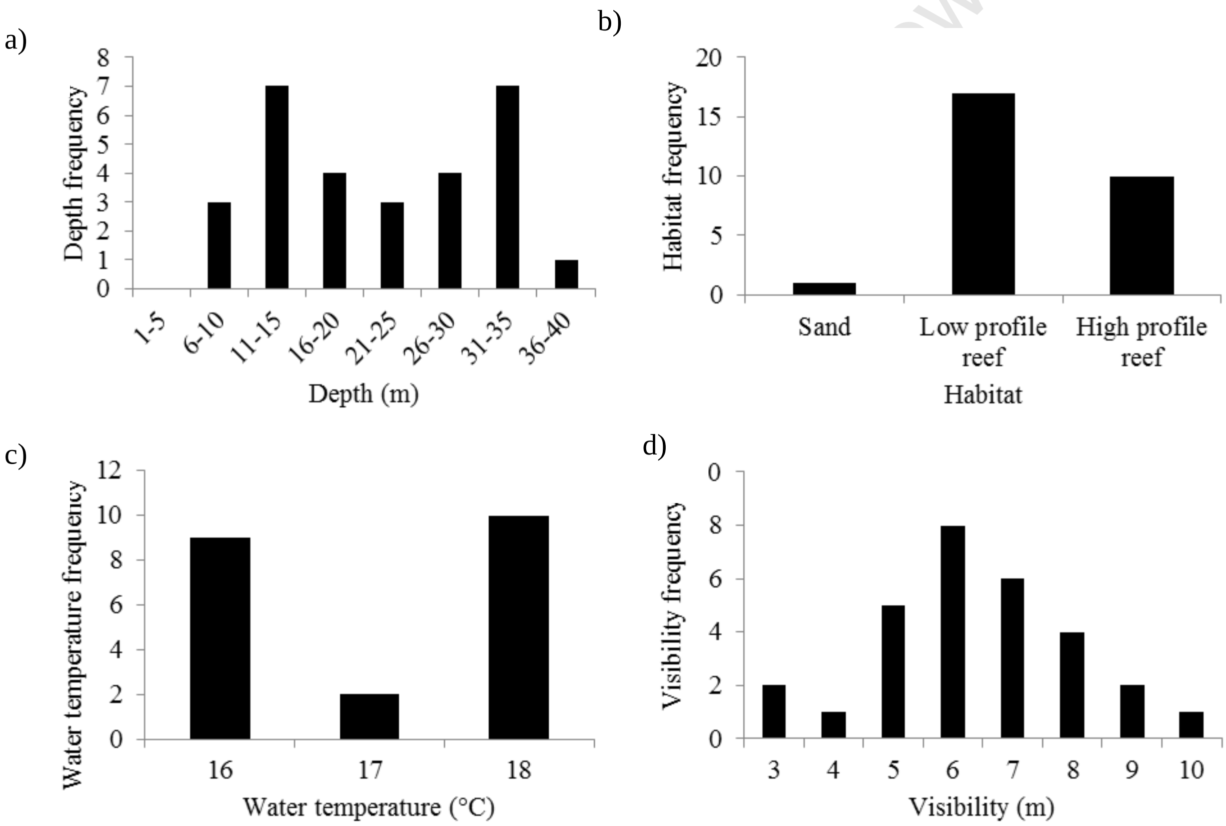

Figure 3. Environmental variables depth [m] (a), bottom habitat-type (b), water temperature [°C] (c) and visibility [m] (d) measured in Stilbaai MPA over the study period.

Table 3. The species recorded employing BRUV in the Stilbaai MPA. Species are ordered according to descending relative abundance.

Feeding Guild	Family	Species	*Scientific name*	Frequency	Max N.				
					Mean	*SD*	*Max.*	*Min.*	*Relative abundance*
Invertebrate carnivore	Sparidae	Steentjie	*Spondyliosoma emarginatum*	28	13.07	8.58	31	2	12.62
Invertebrate carnivore	Sparidae	Roman	*Chrysoblephus laticeps*	29	5.14	3.37	16	1	5.14
Herbivore	Sparidae	Strepie	*Sarpa salpa*	3	43.33	15.57	58	27	4.48
Invertebrate carnivore	Sparidae	Fransmadam	*Boopsoidea inornata*	24	2.50	1.93	7	1	2.07
Omnivore	Sparidae	Santer	*Cheimerius nufar*	28	1.79	1.20	5	1	1.72
Invertebrate carnivore	Sparidae	Blue hottentot	*Pachymetopon aeneum*	21	1.90	1.26	5	1	1.38
Omnivore	Tetraodontidae	Evil-eye pufferfish	*Amblyrhynchotes honckenii*	19	1.95	1.27	5	1	1.28
Invertebrate carnivore	Triakidae	Smooth-hound shark	*Mustelus mustelus*	23	1.57	0.73	3	1	1.24
Invertebrate carnivore	Sparidae	Blacktail	*Diplodus capensis*	15	2.27	2.58	11	1	1.17
Omnivore	Sparidae	Panga	*Pterogymnus laniarius*	11	2.18	1.54	6	1	0.79
Invertebrate carnivore	Sparidae	Zebra	*Diplodus hottentotus*	20	1.15	0.37	2	1	0.79
Omnivore	Ariidae	White seacatfish	*Galeichthys feliceps*	11	2.00	1.00	4	1	0.76
Piscivore	Scyliorhinidae	Leopard catshark	*Poroderma pantherinum*	19	1.12	0.32	2	1	0.72
Piscivore	Sparidae	Red steenbras	*Petrus rupestris*	17	1.12	0.33	2	1	0.66
Piscivore	Scyliorhinidae	Pyjama catshark	*Poroderma africanum*	13	1.30	0.48	2	1	0.59
Omnivore	Sparidae	Red stumpnose	*Chrysoblephus gibbiceps*	15	1.07	0.26	2	1	0.55
Invertebrate carnivore	Oplegnathidae	Cape knifejaw	*Oplegnathus conwayi*	8	1.38	0.52	2	1	0.38
Piscivore	Carcharhinidae	Bronze whaler	*Carcharhinus brachyurus*	10	1.00	0.00	1	1	0.34
Invertebrate carnivore	Cheilodactylidae	Two-tone fingerfin	*Chirodactylus brachydactylus*	6	1.67	0.82	3	1	0.34
Piscivore	Triakidae	Soupfin shark	*Galeorhinus galeus*	9	1.00	0.00	1	1	0.31
Invertebrate carnivore	Sparidae	Dageraad	*Chrysoblephus cristiceps*	4	2.00	0.82	3	1	0.28
Invertebrate carnivore	Sparidae	John brown	*Gymnocrotaphus curvidens*	7	1.14	0.38	2	1	0.28
Piscivore	Squalidae	Shortnose spiny dogfish	*Squalus meglops*	3	2.33	0.58	3	2	0.24
Omnivore	Sparidae	Black musselcracker	*Cymatoceps nasutus*	4	1.25	0.50	2	1	0.17

Piscivore	Triakidae	Spotted gully shark	*Triakis megalopterus*	4	1.25	0.50	2	1	0.17
Omnivore	Sparidae	White musselcracker	*Sparodon durbanensis*	3	1.33	0.58	2	1	0.14
Invertebrate carnivore	Sparidae	Cape stumpnose	*Rhabdosargus holubi*	2	1.50	0.71	2	1	0.10
Piscivore	Scyliorhinidae	Dark shyshark	*Haploblepharus pictus*	3	1.00	0.00	1	1	0.10
Invertebrate carnivore	Chaetodontidae	Doublesash butterflyfish	*Chaetodon marleyi*	3	1.00	0.00	1	1	0.10
Herbivore	Sparidae	Bronze bream	*Pachymetopon grande*	2	1.00	0.00	1	1	0.07
Omnivore	Serranidae	Catface rockcod	*Epinephelus andersoni*	2	1.00	0.00	1	1	0.07
Omnivore	Serranidae	Yellowbelly rockcod	*Epinephelus marginatus*	2	1.00	0.00	1	1	0.07
Invertebrate carnivore	Dasyatidae	Blue stingray	*Dasyatus chrysonota*	1	1.00	NA	1	1	0.03
Invertebrate carnivore	Myliobatidae	Eagle ray	*Myliobatis Aquila*	1	1.00	NA	1	1	0.03
Piscivore	Scyliorhinidae	Puffadder shyshark	*Haploblepharus edwardsii*	1	1.00	NA	1	1	0.03
Piscivore	Odontaspididae	Ragged-tooth shark	*Carcharias Taurus*	1	1.00	NA	1	1	0.03
Invertebrate carnivore	Cheilodactylidae	Redfinger	*Cheilodactylus fasciatus*	1	1.00	NA	1	1	0.03
Piscivore	Dasyatidae	Short-tailed ray	*Dasyatis brevicaudata*	1	1.00	NA	1	1	0.03

Thirty-eight species belonging to 14 families were recorded. Of these, roman (frequency = 29.00), steentjie (frequency = 28.00), santer (frequency = 28.00), fransmadam (frequency = 24.00), smooth-hound shark (frequency = 23.00), blue hottentot (frequency = 21.00) and zebra (frequency = 20.00) were recorded most frequently. Strepies (mean Max. N = 43.33, max. = 58.00) and steentjies (mean Max. N = 13.07, max. = 31.00) appeared in the highest numbers at any one site. Overall, steentjies appear as the most abundant species (Relative abundance = 13.00). Of the recorded species, 11 species were piscivorous (including red steenbras, bronze whaler and ragged-tooth shark). Most recorded species were omnivores (19 species), whilst 15 species consumed invertebrates. Two species (bronze bream and strepie) were classified as herbivorous, although juveniles of the latter species eat crustaceans (Heemstra & Heemstra 2004).

2. Patterns of species distribution in the Stilbaai MPA

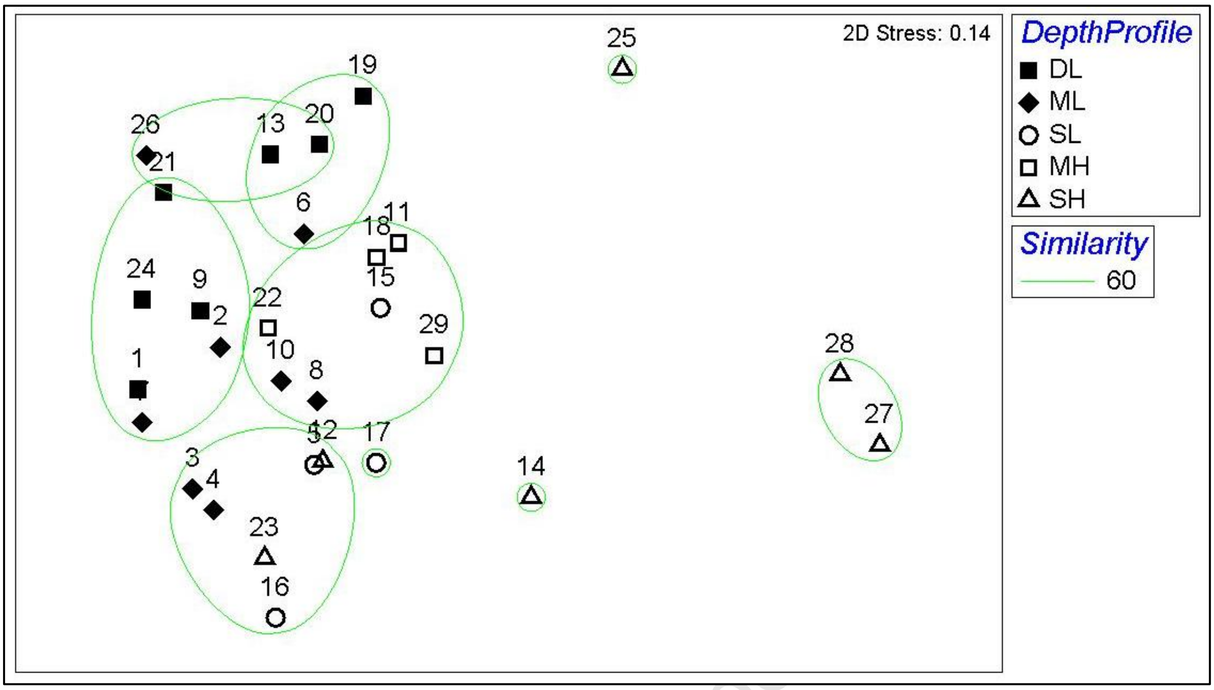

Figure 4. Similarity among sites based on species composition. Sites are characterised according to depth strata [S = shallow (0-15m), M = medium (15-30m), D = deep (30-45m)] and profile-type [L (low) and H (high) profile]. Elipses indicate groups at the 60% similarity level.

Sites similar in species composition at the 60% level tended to belong to the same depth strata or profile type. Sites differing in depth separated along the x-axis (Figure 4). Four shallow, high profile sites emerged as the least similar to the rest of the study sample (sites 14, 25, 27 and 28). Sites 27 and 28 were the shallowest sampled (6 and 6.5 m respectively), with site 25 the third shallowest site sampled (9 m). All four of these sites were high profile and assigned the highest score (7) relative to other sites in profile grading, with the highest water temperatures recorded during sampling (+ 17°C). Of these four sites, 14 had the overall highest species count (23 species), 27 had the second highest species count (21 species) and 28 had the fourth-highest species count (18 species).

Of the measured predictor variables (depth, profile, sea temperature and visibility), depth emerged as a significant predictor of site similarity (ANOSIM, $\alpha = 0.001$).

Table 4. The five combinations of environmental variables (out of 10 selections) that best explain variation in species assemblage (relative abundance) among 29 sites (BIOENV, $\alpha = 0.001$).

Selections	Variables	Correlation
Depth, sea temperature, profile	3	0.43
Depth, sea temperature	2	0.42
Depth	1	0.39
Depth, profile	2	0.39
Sea temperature, profile	2	0.37

Sites within the same depth strata were most similar to one another among 29 sites (Figure 4), and the ANOSIM analysis assessed depth as a significant predictor of site-similarity in species composition. The combined effects of depth, sea temperature and reef profile best explain observed differences in species composition between sites (Table 4). Depth appears most consistently as an explanatory variable in subsequent BIOENV selections, contributing to four combinations, whereas sea temperature and profile contribute to three each. Visibility was of least consequence in explaining variation among sites, appearing only in the 6th combination (not shown), in conjunction with all three other environmental variables.

Table 5. A list of species which dominate the communities in three depth categories is shown in the shaded diagonal boxes. Species which differentiate communities among different depth categories are shown in unshaded boxes off the diagonals. Differentiating species are common in the depth category listed in the column but rare in the depth category listed in the row.

		Common		
		Shallow	Intermediate	Deep
Rare	Shallow	Steentjie Roman Evil-eye pufferfish	Blue hottentot Steentjie Pyjama catshark	Panga White seacatfish Blue hottentot
	Intermediate	Evil-eye pufferfish Cape knifejaw Blacktail	Steentjie Roman Santer	Panga White seacatfish Spiny dogfish
	Deep	Blacktail Evil-eye pufferfish Cape knifejaw	Leopard catshark Pyjama catshark Evil-eye pufferfish	Steentjie Roman Santer

Steentjie and roman characterise sites across all three depth strata (Table 5). Whilst santer characterised both intermediate and deep strata, evil-eye pufferfish abundance was highest in shallow sites and replaced santer as a common species in that depth category. The abundance of cape knifejaw and blacktail in shallow sites also distinguished this depth category from deep and intermediate sites. The presence of panga and white seacatfish at deep sites differentiated this depth category from shallow and intermediate depths. The presence of catshark species (pyjama and leopard) distinguished intermediate sites from both shallow and deep sites.

Table 6. A list of species which dominate the communities in two profile categories is shown in the shaded diagonal boxes. Species which differentiate communities between profile categories are shown in unshaded boxes off the diagonals. Differentiating species are common in the profile listed in the column and rare in the profile listed in the row.

		Common	
		Low	High
Rare	Low	Steentjie Roman Santer	Blacktail Evil-eye pufferfish Strepie
	High	Panga Blue hottentot Steentjie	Roman Steentjie Fransmadam

Steentjie and roman are characteristic of both high and low profile (Table 6). Fransmadam was characteristic of high profile reef, and the presence of this species contributed to differentiating high profile from low profile sites. Santer was common in low profile sites. The abundance of blacktail, evil-eye pufferfish and strepie was higher in high profile areas and explained differences in composition between high and low profile sites. Panga, blue hottentot and steentjie were common in low profile sites.

DISCUSSION

This chapter corroborates findings that temperate reef fish distribution is best predicted by depth, sea temperature and reef profile (Buxton & Smale 1989; Buxton 1993). Sites classified as shallow, intermediate or deep were characterised by different species assemblages – a result which may be explained by differences in the abundance and diversity of food available at different depths (Buxton & Smale 1989). Furthermore, the results suggest that BRUV systems do sample higher species richness compared to other monitoring methods such as UVC and controlled angling (Cappo et al. 2004; Bernard 2012). This finding recommends the inclusion of BRUVs as a standard part of monitoring techniques for future reef fish assessments.

Patterns of reef fish distribution in the Stilbaai MPA

Whilst depth appeared to be the single-most important predictor of species distribution and abundance, it was the interaction between depth, sea temperature and reef profile that emerged as the best explanatory combination of variables in this and previous studies (Buxton & Smale 1989). It should be noted, however, that was no significant fluctuation in temperature during the study period. This may explain why depth, and to an extent, profile, appeared the relatively more important predictor variables. The reason for this finding may be elucidated by what this combination of variables offers fish for feeding opportunities, shelter and mobility (Buxton & Smale 1989; McCormick 1994; Friedlander & Parrish 1998).

Shallow, high profile sites attracted the highest species diversity and abundance, especially when higher sea temperatures prevailed. These results parallel findings from surveys using traditional monitoring methods (Buxton & Smale 1989; Götz et al. 2008). Controlled angling and UVC surveys in the Goukamma MPA showed that more roman were caught or observed in shallow, warm water with low turbidity (Götz et al. 2008). UVC counts in the Castle

Rocks MPA demonstrated that False Bay reef fish abundance was highest in higher water temperatures (Lechanteur 2000). The impact of sea temperature on fish metabolism has been shown to affect species' mobility, with certain species more active as sea temperatures increase (Buxton & Smale 1989). This would in turn influence BRUV monitoring by increasing the abundance of species moving into the BRUV's field of view during deployments at sites with higher sea temperatures.

UVC surveys found that species favour shallow reefs (< 20 m) of the Cape Pensinsula (Lechanteur 2000). Shallow and high profile sites in this study were dominated by invertebrate carnivores, particularly evil-eye pufferfish, cape knifejaw, roman and blacktail. These results corroborate findings from controlled angling and UVC surveys in Goukamma MPA that highlighted invertebrate carnivores such as steentjies, fransmadam, two-tone fingerfin and blacktail as most abundant on high profile reefs (Götz 2006). This pattern is attributed to the higher abundance of prey, as well as shelter from predation available at shallow, high profile sites (Buxton & Smale 1989; Friedlander & Parrish 1998). Additionally, dietary studies on blacktail and zebra showed that higher algal productivity in shallow sites facilitates diverse benthic invertebrate production, creating an important food source for juvenile fish, accommodating generalist and specialist feeders and species such as strepies and blacktail with different dietary requirements at various life stages (Mann & Buxton 1992).

The prevalence of blue hottentot at intermediate and deep sites mirrors findings from Goukamma, suggesting that this species' diet of hydrozoans facilitates its exploitation of a specialist niche at depths where algal productivity is limited by light attenuation (Götz 2006). Of more interest to this study, however, is the confirmation that BRUV monitoring is able to detect the same broad trends in species distribution as the more established methods. This recommends BRUV for monitoring not only where the logistical constraints of traditional monitoring should be addressed, but as part of a long-term sampling protocol.

Implications of species distribution patterns for monitoring

Understanding that species composition differs with depth, sea temperature and profile is important to direct future monitoring efforts, and will assist in a meaningful interpretation of monitoring data (Colton & Swearer 2010). Steentjies and roman were present across all depth strata and profile-types, which should qualify these species as suitable representatives around which to design a sampling protocol. However, the SIMPER results showed that certain species favour particular depth strata and profiles. By way of example, the distribution of black musselcracker, targeted in the recreational angling sector, is influenced by depth (wherever a broad array of benthic prey is available) and reef profile (with solitary, large adults in low abundance) (Buxton & Clarke 1989). The understanding that some species are confined to favoured habitats is essential to ensure that monitoring is representative across habitats (Colton & Swearer 2010).

Several reef fish may be of special interest for monitoring in the Stilbaai MPA because they face fishing pressure outside MPAs (Buxton & Clarke 1989; Buxton 1993). To detect significant change in these species' abundance over time, an understanding of how environmental variables influence their distribution can direct monitoring efforts towards their preferred habitats. This is an important consideration for monitoring to be effective within a reasonable annual timeframe and with limited resources.

Species diversity

BRUV sampling in Stilbaai obtained a higher estimate of species diversity than UVC and controlled angling surveys in the Castle Rocks, Goukamma and Tsitsikamma MPAs

(Lechanteur 2000; Götz 2006; Bennett et al. 2009; Götz 2009). This finding highlights the benefit of BRUV monitoring for achieving broad spectrum surveys. Twenty-eight species representing 11 families were recorded by UVC surveys in the Castle Rocks MPA (Lechanteur 2000). Given biogeographical influences on species diversity, one would anticipate more species to be recorded in the Stilbaai MPA (Turpie et al. 2000). This study recorded 38 species representing 14 families. It is worth noting, however, that these findings may also point to the ability of BRUV monitoring to overcome differential attraction to and deterrence from SCUBA divers (Watson et al. 2010).

Tsitsikamma has been closed to fishing for 26 years (Buxton 1992). It is therefore conceivable that species diversity should be higher in this MPA than in the more recently proclaimed Stilbaai MPA, based on the population recovery period available to species in each MPA (Bennett & Attwood 1991). However, the angling survey in Tsitsikamma recorded 14 species and UVC recorded 17 species (Bennett et al. 2009). It is unlikely that the Stilbaai MPA is more diverse than Tsitsikamma, given patterns of biogeography along the South African coastline (Turpie et al. 2000) and that the latter has a longer history of protection (Buxton 1992) and encompasses more high profile reef (Bernard 2012). These results therefore more likely corroborate international findings that BRUV surveys record higher species richness, a wider size range of families and a higher abundance of predators than traditional monitoring methods such as UVC (Cappo et al. 2004; Watson et al. 2010).

To assess the degree of species richness represented by the Stilbaai MPA, a more comparable result is perhaps that from a BRUV survey conducted in the Tsitsikamma MPA which recorded 39 species employing the exact same material and method (Bernard 2012). This potentially indicates that the diversity protected in the Stilbaai MPA is high, given that it was only closed to fishing three years ago (Tunley 2009). Interpreting differences in species diversity recorded at Castle Rocks, Goukamma, Tsitsikamma and Stilbaai relies on the understanding that biogeography has a strong influence on species diversity along the South African coastline (Turpie et al. 2000). For this reason, Goukamma and Tsitsikamma should represent higher species diversity than Stilbaai (Turpie et al. 2000). An explanation for the lower estimates of diversity recorded using UVC and angling is therefore to be found in BRUV's capability as a more broadly representative monitoring method.

Two important considerations in the selection of survey methodologies include sampling efficiency and the representation of habitats and species (Bennett et al. 2009). BRUV deployments are both efficient and representative in terms of species diversity when compared to surveys in other South African MPAs using traditional monitoring methods. In total, 88 point counts, 44 UVC transects and 10 angling hours at 16 stations were conducted to achieve the Tsitsikamma estimates (Bennett et al. 2009). When this effort is compared to the six sampling days in Stilbaai to achieve 29 samples and an overall higher species diversity estimate, it is clear that BRUV monitoring is more time-efficient.

Species composition

Steentjie and roman were the most abundant species recorded in this study. This result mirrors findings from the BRUV survey in Tsitsikamma (Bernard 2012). Controlled angling in Tsitsikamma also recorded roman and steentjie as most abundant, whilst UVC recorded highest numbers of blue hottentot and fransmadam (Bennett et al. 2009). UVC surveys in Goukamma recorded fransmadam and steentjie as the most abundant species, whilst roman and fransmadam were the most abundant species recorded using controlled angling (Götz 2006). Importantly, these four species were amongst the six most abundant for this survey. Overall, species composition obtained using UVC, controlled angling and BRUV are similar across Goukamma, Tsitsikamma and Stilbaai (Bennett et al. 2009; Götz et al. 2009b; Bernard 2012). This finding strengthens recommendations to apply BRUV as a monitoring technique.

Soupfin, smooth-hound and spotted-gully sharks feature more frequently in this study than in other UVC surveys (Bennett et al. 2009; Götz et al. 2009b). This may be the result of certain features of the study area itself that make it an attractive habitat for these species, but more likely points to the finding that BRUV systems record a higher presence of elasmobranch species due to bait attraction (Stobart et al. 2007; Colton & Swearer 2010). This is important in light of understudied and overexploited elasmobranch species that are deterred by divers and underrepresented in UVC surveys (Colton & Swearer 2010). Controlled angling surveys in Goukamma also sampled smooth-hound sharks, most likely as a similar result of bait attraction. However, the opportunity to obtain measures of a commercially exploited species without using an extractive method is more suited to research and monitoring for exploited species in MPAs (Willis et al. 2000).

This study corroborates findings that BRUV samples herbivorous species, but may underrepresent their diversity (Watson et al. 2010). The only truly herbivorous species recorded were strepies and bronze bream. This would confirm findings that species' behaviour impacts what portion of the assemblage is sampled, and that certain species may be overlooked in BRUV surveys (Harvey et al. 2007; Stobart et al. 2007; Watson et al. 2010). However, despite the use of sardines for bait, not only piscivorous species were recorded. The most abundant species (steentjie) was classified an invertebrate carnivore. It is important to note that the majority of species recorded were omnivorous rather than piscivorous. Monitoring that detects omnivorous species would cover a diversity of habitats and encompass a wider variety of prey. However, monitoring would benefit from recording specialist feeders too, which may indicate the protection of specific habitat or prey, or the recovery of more vulnerable species.

Conclusion

The position of a MPA along the coastline, together with the type of habitat and depth range sampled during surveys, will influence the portion of the species assemblage recorded by BRUV systems. Therefore, an understanding of species' habitat association will facilitate the correct interpretation of future surveys. Additionally, species behaviour towards bait will predispose certain species towards representation using BRUV surveys and exclude others. This is particularly useful for assessing vulnerable elasmobranchs that are underrepresented by UVC surveys and exploited species that are targeted by fisheries.

The implications of these results must be assessed in light of the viability of long-term BRUV monitoring in South Africa. This study suggests that BRUV can detect broad patterns of species composition and abundance across different habitats, but that it offers the advantages of doing so with lower manpower requirements, in a wider range of ocean and weather conditions, at greater depths and without compromising MPA objectives by extracting species (Willis et al. 2000; Langlois et al. 2010; Watson et al. 2010). Moreover, BRUV monitoring records higher species richness and a wider representation of families. A monitoring technique should demonstrate sound data collection with low variability and high statistical power. As such, it is shown to be an effective and sustainable monitoring tool to aid long-term monitoring in South Africa, especially where traditional monitoring methods are insufficient (Becker et al. 2010; Bernard 2012).

CHAPTER 3

Optimal baited remote underwater video sampling design for long-term reef fish monitoring in the Stilbaai marine protected area

INTRODUCTION

For MPAs to be properly managed for biodiversity conservation and to be incorporated into an expanding network, effective monitoring should be sustainable over the long-term (Gislason et al. 2000; Jones 2002; Colton & Swearer 2010). There is a need to objectively assess the progress of existing MPAs towards achieving biodiversity conservation, fisheries management and meeting non-extractive human-needs (Hockey & Branch 1997; Tunley 2009).

Whilst baited remote underwater video (BRUV) monitoring, like SCUBA and controlled angling, has its inherent biases (Colton & Swearer 2010; Langlois et al. 2010), its clear advantage for South African MPAs lies in its cost-effectiveness, modest requirements for skilled labour and low environmental impact. BRUV monitoring has been extensively developed and tested internationally, particularly in Australian waters (Cappo et al. 2003; Harvey et al. 2007; Watson et al. 2010). The development of BRUV systems from a purely research-oriented technique to a sustainable monitoring solution has arisen after studies found several key advantages over traditional monitoring techniques (Willis et al. 2000; Stobart et al .2007; Langlois et al. 2010).

The technique requires lower manpower, time and boat requirements to collect sound scientific data with higher statistical power and lower variability (Langlois et al. 2010). The system is operational where SCUBA techniques are considered unsafe, increasing the underwater data-collection time and extending monitoring scope to deeper waters (Cappo et al. 2004; Stobart et al. 2007; Watson et al. 2010). As a non-extractive method, it falls in line with MPA objectives, and the retention of footage for independent re-analysis also opens opportunities for use in long-term ecosystem comparisons and public awareness (Parker et al. 1991; Willis et al. 2000; Langlois et al. 2010).

Catch-per-unit-effort (CPUE) is one of the most commonly used indices of fish abundance in fisheries management (Maunder & Punt 2004), but CPUE data from the linefishery has low information content because it is unclear from the data which species were targeted. A zero in the data could indicate either that there were no fish, or that there was no attempt to catch that fish species. SCUBA counts tend to sample the proportion of fish assemblages attracted to divers, and underestimate species deterred by diver presence (Watson et al. 2010). BRUV overcomes this problem by providing a broad-spectrum view of fish abundance (Cappo et al. 2003).

BRUV does not escape the problems posed by variable dispersion among the different species (Willis et al. 2003). Species of a territorial nature are likely to be evenly distributed and should require a lower sampling effort. Others are either shoaling species, or are known to aggregate around specific features (Gascon & Miller 1982), leading to abnormally high variance and requiring the use of over-dispersion parameters in statistical models. Sampling requirements are therefore likely to vary enormously among species.

As South African MPAs are poorly resourced (Tunley 2009), there is a clear need to develop cost-effective yet scientifically credible monitoring techniques that can be applied across a number of MPAs. The aim of this study is to use BRUV data from Stilbaai MPA to estimate the minimum length of camera deployment and number of deployments to achieve effective monitoring of reef fish abundance.

METHODS

One hour long BRUV samples were taken from 29 randomly selected sites in a portion of the restricted zone of the Stilbaai MPA over the period 11 October to 30 November 2011. The area selected was 11.3 km^2 and covered a depth range from 5 m to 41 m. The habitat was almost exclusively temperate reef, most of which was low profile. The maximum number of fish of any species captured in a single video frame, referred to as Max N, was used as an index of abundance (Chapter 2). 38 species from 14 families were recorded. Thirty nine percent of species made up 90% of the abundance. Species composition differed among three depth categories and between high and low profile reef.

Power analysis

In considering calculations of statistical power, it was necessary to consider the nature of temporal comparisons that might be done in future. Generally speaking, higher statistical power can be achieved by regressing multiple repeated surveys against time. However, for the purpose of this analysis a comparison of two surveys was considered, for successive years or after an interval of several years. In considering effect sizes, the time elapsed between surveys becomes important. An effect size of two (i.e. a doubling), for example, might be appropriate for surveys taken at intervals of five years or more, whereas for annual surveys an effect size greater than 1.2 would be unrealistic for long-lived, slow-growing species.

Power analyses were used to assess the number of samples needed to detect a doubling (k_1 = 2) (*sensu* Edgar & Barrett 1999) and a 20% change (k_2 = 1.2) of a species' population) at the 0.05 significance level with 80% power. Six species were chosen for analysis, based on their ubiquity across sites in the Stilbaai MPA, or on their conservation and fisheries importance (Table 1).

Table 1. Species selected for power analysis.

Species	Scientific name	Selection status
Steentjie	*Spondyliosoma emarginatum*	Abundant
Roman	*Chrysoblephus laticeps*	Abundant; fisheries
Santer	*Cheimerius nufar*	Abundant
Red steenbras	*Petrus rupestris*	Conservation; fisheries
Dageraad	*Chrysoblephus cristiceps*	Conservation; fisheries
Black musselcracker	*Cymatoceps nasutus*	Conservation; fisheries
Red stumpnose	*Chrysoblephus gibbiceps*	Conservation; fisheries

The equation given by Willis et al. (2003) for calculating statistical power in tests for differences between two means, based on count data, was used. This equation was most appropriate because the very low counts for the majority of species included a high number of zeros, which meant that the parametric equations for statistical power provided by Zar (1984) were inappropriate.

The equations for statistical power are (Willis et al. 2003):

eq.1

where k is the effect size (μ_1/μ_2), μ_1 and μ_2 are the specified means for surveys 1 and 2, ø is the overdispersion parameter, n is the sample size for surveys 1 and 2 and Z_α and Z_β are the normal deviates indicating the likelihood of Type I and Type II errors. The power of the test is equal to 1 – β.

Since equation 1 uses the Poisson distribution, there was no need to separately specify the variance (variance and mean are equal in Poisson distribution). However, many distributions of fish count data are over-dispersed (i.e. they tend to have a higher frequency of zeros than predicted by the Poisson distribution). It was therefore necessary to estimate the degree of over-dispersion for each species. The most convenient method to calculate over-dispersion was to fit a General Linear Model (GLM) to the Max N data without offering any independent variables, other than a mean. No attempt was made to stratify the environment in terms of depth or habitat. The approach is based on the most conservative case in which there is no *a priori* information on either the habitat or fish habitat associations. GLMs were fitted in R version 2.13.0 using the Poisson distribution (R Development Core Team 2011).

Effect sizes of 2 and 1.2 were used. The terms of equation 1 were rearranged to calculate n at a power of 0.8 (β) and significance of 0.05 (α), to illustrate sampling requirements for species with differing abundance and over-dispersion parameters.

Optimal deployment times for recording diversity

For monitoring scenarios where no prior habitat stratification has been conducted, optimal deployment duration across all depth strata and reef profiles was assessed using the cumulative count of species arriving within the BRUV field of view over the 60 minute deployment. The cumulative proportion of the total number of species observed in any one sample at five minute intervals was calculated. These values were averaged across all 29 samples to give an average species accumulation curve with respect to time of deployment.

A sigmoid curve was fitted to the species accumulation curve data to estimate the time at which 50% and 95% of species were recorded. The sigmoid curve is given by:

eq. 2

where P_t is the modelled rate of increase of the cumulative diversity observed after t minutes in the BRUV video versus the total diversity observed in the video, t_{50} is the time at which half of the species have been detected in the BRUV video and delta is a parameter determining the steepness of the slope.

and

eq. 3

where $t_{95\%}$ is the predicted time in minutes required to record 95% of the total expected diversity, t_{50} is the time at which half of the species have been detected in the BRUV video and delta is a parameter determining the steepness of the slope.

Equation 2 was fitted to the ratio of the cumulative diversity observed after t minutes in the BRUV video versus the total diversity observed in the video by minimising the sum of squares and estimating the values of t_{50} and delta.

Equation 3 was used to estimate the time taken to record 95% of the total expected diversity.

Multivariate analyses based on species composition across 29 sites indicated that intermediate depth sites tended to group with either deep or shallow sites, depending on whether they were closer to 15 m (shallow) or 35 m (deep) (Chapter 2). Therefore, depth was split into two categories: shallow (0 – 20 m) and deep (20 – 40 m) so that these intermediate sites fell into either one or the other. Replicate sites were required to compare species accumulation curves across depth and profile. A cluster analysis (Clarke 1993) was used to distinguish six deep, low profile sites that were similar to one another at the 60% level and five shallow, high profile sites that were similar to one another at the 60% level. Sigmoid functions were compared among depth strata and profiles.

Deployment time for Max N assessments

Videos were analysed in five minute segments. The maximum number of fish of a particular species in a single frame was recorded as N_t for time segment t. N_t measurements were taken for every five minute segment of all 60 minute videos for each of the seven species used in the power analysis (Table 1). The maximum N_t value for each species in each video was the Max N value used in the power analysis. N_t was averaged for each five minute segment for each species across all videos in which that species was recorded. For each video the relative increase in N_t over time was calculated as N_t/Max N, until Max N was attained. Thereafter a value of 1.0 was used to indicate that Max N had already been attained regardless of the possibility that N_t might have declined subsequent to the Max N recording. N_t/Max N values were averaged across all sites where the species was recorded at least once. A sigmoid function was fitted to the average Nt/Max N values to estimate the video duration corresponding to 50% and 95% of the time taken to record the Max N value:

eq. 4

where R_t is the relative increase in the average number of fish of a particular species observed in a frame in each successive five minute segment in BRUV videos in which the species was recorded, t is the time in minutes at the end of each segment, t_{50} is the time at which half of Max N was attained in the BRUV video and delta is a parameter determining the steepness of the slope and

— **eq. 5**

where $t_{95\%}$ is the predicted time in minutes required to record 95% of the expected abundance, t is the time in minutes at the end of each segment, t_{50} is the time at which half of Max N was attained in the BRUV video and delta is a parameter determining the steepness of the slope.

Equation 4 was fitted to the average N_t/Max N values by minimising the sum of squares and estimating the values of t and delta.

Equation 5 was used to estimate 95% of the time taken to record the Max N value.

Incidental species recordings

To assess the effectiveness of BRUV in recording species from different feeding guilds and trophic levels, the behaviour of each species with respect to the bait was recorded. For each species, the behaviour of individual fish was recorded as either approaching and feeding, or not approaching the bait (Watson et al. 2010).

For each species, the number of BRUV recordings without approaching was divided by the total number of times that species was recorded during the study and expressed as a percentage.

RESULTS

1. Sample size required to detect changes in species abundance

Dispersion parameters differed for each species (Table 2). Dispersion values were highest for the ubiquitous steentjie ($\sigma = 6.09$) and roman ($\sigma = 2.2$), as well as the infrequently sampled dageraad ($\sigma = 2.04$). Conversely, dispersion values less than one were obtained for red steenbras ($\sigma = 0.58$) and red stumpnose ($\sigma = 0.6$). Dispersion values for black musselcracker ($\sigma = 1.3$) and santer ($\sigma = 0.9$) were close to one.

Table 2. Results from the power analysis detailing the required sample size (required n) to detect a doubling of the population ($k_1 = 2$) and a 20% change ($k_2 = 1.2$) at $\alpha = 0.05$ and a power of 0.8. N refers to the total number of sites where a species was sampled.

			Required n	
Species	N		k_1	k_2
Roman	29	2.2	11	186
Santer	28	0.9	13	218
Steentjie	28	6.1	12	209
Red steenbras	17	0.6	22	381
Red stumpnose	15	0.6	27	466
Black musselcracker	4	1.3	181	3193
Dageraad	4	2.0	182	3209

Sampling requirements across species increased with decreasing abundance. The most abundant species in the study (roman and steentjie) had the lowest sampling requirements (Table 2). Black musselcracker and dageraad were detected infrequently and required the greatest number of samples to detect an annual increase of 20% (3193 and 3209 samples respectively).

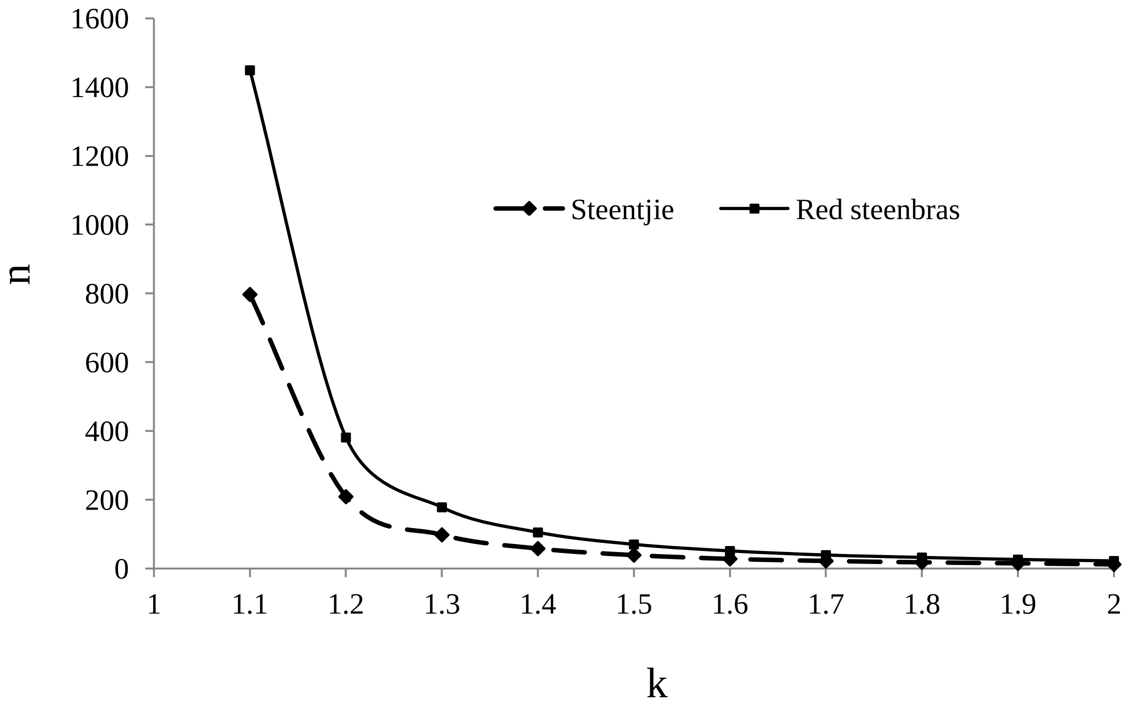

Figure 1. The number of samples (n) required to detect an effect size (k) with 80% power at a significance level of 0.05 for two species with different over-dispersion parameters.

The number of samples (n) required to detect changes in species abundance decreased with an increasing effect size (k). The number of samples required is high at k = 1.1 and gradually becomes lower from k = 1.3 to k = 2.0. The required n at k = 1.2 is 17 times greater than at k = 2.0.

2. Optimal deployment time

Species diversity

Deep, low profile sites require 48 minute deployments whilst shallow, high profile sites require 51 minute deployments. This makes the requirements across habitats remarkably similar for BRUV deployment times to record 95% of species diversity in the Stilbaai MPA. For an unstratified sampling design, it was found that a deployment of 49 minutes should record 95% of the species present across all depths and profiles in the MPA.

Max N assessments

Black musselcracker, red steenbras, red stumpnose and dageraad were recorded too infrequently at deep, low profile sites to assess their deployment times in this habitat. In shallow, high profile sites, black musselcracker and dageraad were sampled infrequently (Table 2) but required the shortest deployment times to record Max N (Table 3). Steentjies required the longest deployment time (Table 3).

Table 3. Optimal deployment times to record 95% of the species diversity represented in the Stilbaai MPA across all depths and profiles, and 95% of Max N abundance for species of monitoring interest in the Stilbaai MPA.

Species	Deep, low profile sites	Shallow, high profile sites
Roman	54.5	52.3
Santer	27.7	52.5
Steentjie	55.7	55.4
Black musselcracker	-	30.7
Red steenbras	-	50.1
Red stumpnose	-	54.2
Dageraad	-	39.1

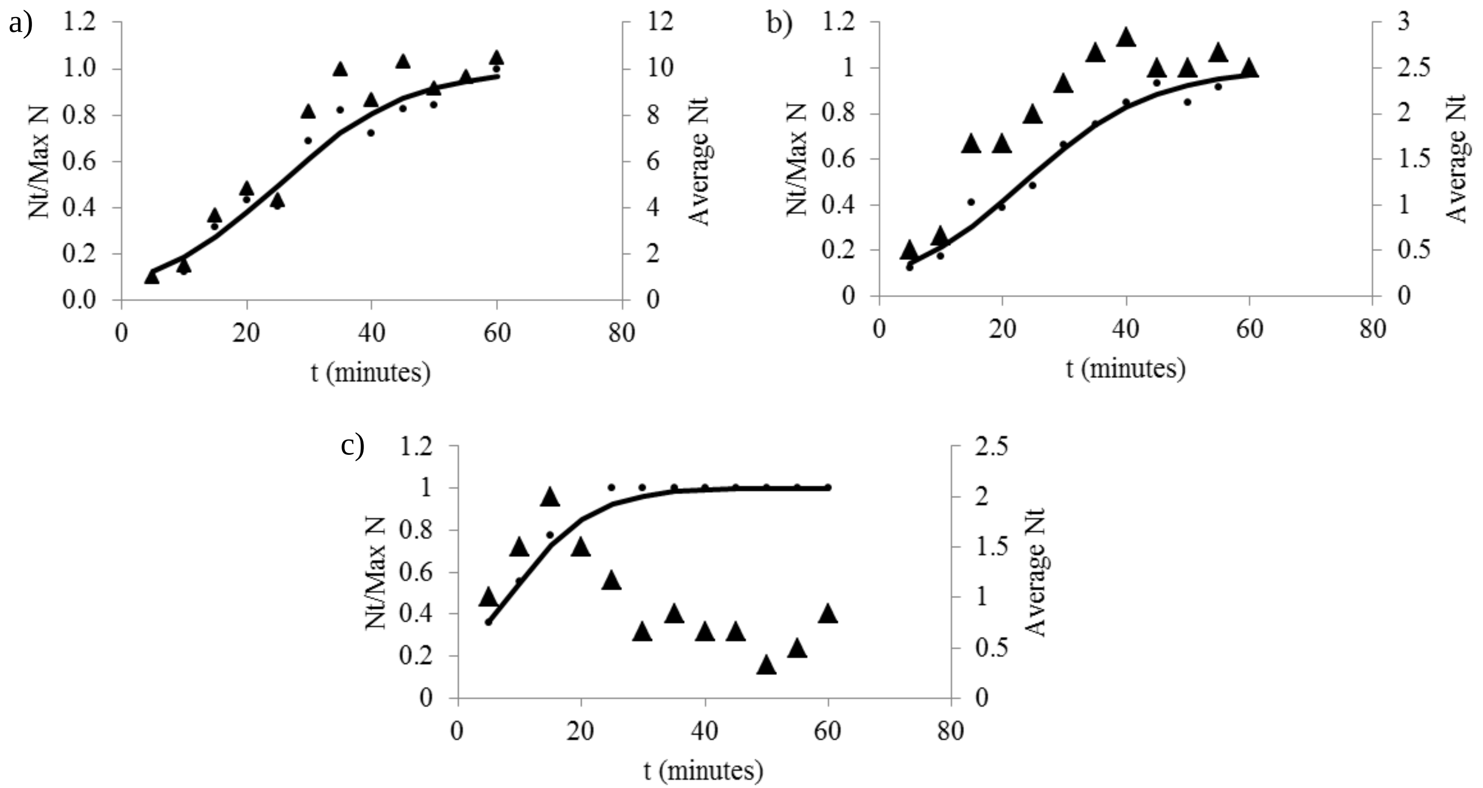

Figure 2. Average N_t (solid triangles) for steentjie (a), roman (b) and santer (c) throughout a 60 minute BRUV deployment in deep, low profile sites. A sigmoid curve (solid line) is fitted to N_t/Max N (solid circles).

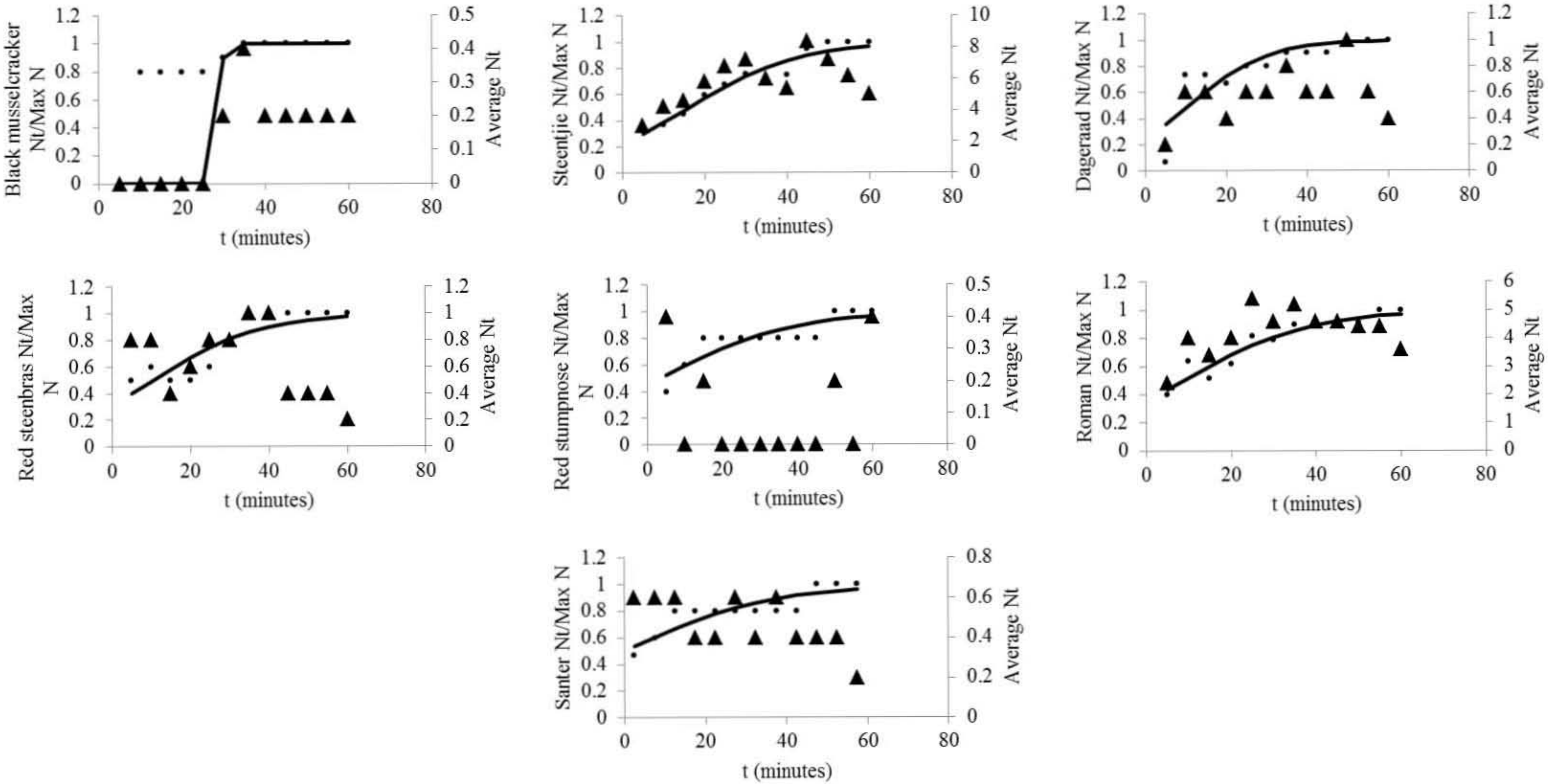

Figure 3. Average N_t (solid triangles) for seven species throughout a 60 minute BRUV deployment in shallow, high profile sites. A sigmoid curve (solid line) is fitted to N_t/Max N (solid circles).

N_t/Max N and N_t values were initially low but accumulated for steentjie and roman throughout the 60 minute BRUV deployment (Figure 2). Average N_t was initially high for santer, but dropped off just before 20 minutes as the numbers of steentjie and roman continued to increase.

Optimal deployment time decreased for roman, stayed remarkably similar for steentjies and increased for santer in shallow, high profile sites (Table 3). Steentjie and roman show the same continuous, gradual increase in abundance throughout the 60 minute deployment as shown in Figure 2 (Figure 3). Santer abundance (N_t) is initially high, but declines from about 20 minutes (Figure 3). Black musselcracker numbers (N_t) are initially low, but reach Max N after 20 minutes, whereafter N_t is maintained at abundance slightly lower than Max N for the remainder of the deployment.

N_t fluctuates throughout the 60 minute deployment for red steenbras and dageraad, with a decrease in abundance towards the end of the video (Figure 3). Red stumpnose abundance remains low for the duration of the video, with higher N_t values attained at the start and end of the deployment. N_t gradually increases for dageraad throughout the video,

Incidental species recordings

Many species were recorded by the BRUV without feeding at the bait canister. Instead, these species were recorded as filming started, or when they swam into the camera's field of view during the deployment without ever approaching the bait.

Many of the species that never fed were infrequently recorded e.g. yellowbelly rockcod (2 sightings), doublesash butterflyfish (*Chaetodon marleyi*) (3 sightings) and cape stumpnose (2 sightings). Other non-feeders were recorded frequently, but never approached the bait canister e.g. blacktail (15 sightings), cape knifejaw (8 sightings) and zebra (20 sightings).

Some ubiquitous species usually recorded as feeding were also recorded in samples as not feeding. These occasions were less frequently recorded e.g. steentjie (1 visit, no steentjies fed at the canister out of 28 sightings), roman (3 samples, no romans fed at the canister out of 29 sightings) and santer (6 visits, no santer fed at the canister out of 29 sightings).

Table 4. The number of approaches by each species to the BRUV without feeding, out of the total frequency (frequency) with which that species was recorded. Species are ordered according to the percentage of approaches without feeding with which they were recorded across 29 sites.

Feeding Guild	**Family**	**Species**	**Scientific name**	**Frequency**	**# approaches no feeding**	**% approaches no feeding**
Invertebrate carnivore	Sparidae	Zebra	*Diplodus hottentotus*	20	20	100.0
Invertebrate carnivore	Sparidae	Blacktail	*Diplodus capensis*	15	15	100.0
Omnivore	Sparidae	Red stumpnose	*Chrysoblephus gibbiceps*	15	15	100.0
Invertebrate carnivore	Oplegnathidae	Cape knifejaw	*Oplegnathus conwayi*	8	8	100.0
Invertebrate carnivore	Sparidae	John Brown	*Gymnocrotaphus curvidens*	7	7	100.0
Invertebrate carnivore	Cheilodactylidae	Two-tone fingerfin	*Chirodactylus brachydactylus*	6	6	100.0
Piscivore	Triakidae	Spotted gully shark	*Triakis megalopterus*	4	4	100.0
Invertebrate carnivore	Chaetodontidae	Doublesash butterflyfish	*Doublesash butterflyfish*	3	3	100.0
Herbivore	Sparidae	Strepie	*Sarpa salpa*	3	3	100.0
Omnivore	Sparidae	White musselcracker	*Sparodon durbanensis*	3	3	100.0
Invertebrate carnivore	Sparidae	Bronze bream	*Pachymetopon grande*	2	2	100.0
Invertebrate carnivore	Sparidae	Cape stumpnose	*Rhabdosargus holubi*	2	2	100.0
Omnivore	Epinephelinae	Catface rockcod	*Epinephelus andersoni*	2	2	100.0
Omnivore	Epinephelinae	Yellowbelly rockcod	*Epinephelus marginatus*	2	2	100.0
Invertebrate carnivore	Dasyatidae	Blue stingray	*Dasyatus chrysonota*	1	1	100.0
Invertebrate carnivore	Myliobatidae	Eagle ray	*Myliobatis aquila*	1	1	100.0
Piscivore	Scyliorhinidae	Puffadder shyshark	*Haploblepharus edwardsii*	1	1	100.0
Piscivore	Odontaspididae	Ragged-tooth shark	*Carcharias taurus*	1	1	100.0
Invertebrate carnivore	Cheilodactylidae	Redfinger	*Cheilodactylus fasciatus*	1	1	100.0
Piscivore	Scyliorhinidae	Leopard catshark	*Poroderma pantherinum*	19	15	79.0
Piscivore	Scyliorhinidae	Dark shyshark	*Haploblepharus pictus*	3	2	66.7

Invertebrate carnivore	Sparidae	Fransmadam	*Boopsoidea inornata*	24	15	62.5
Piscivore	Carcharhinidae	Bronze whaler	*Carcharhinus brachyurus*	10	6	60.0
Invertebrate carnivore	Sparidae	Blue hottentot	*Pachymetopon aeneum*	21	12	57.1
Piscivore	Triakidae	Soupfin	*Galeorhinus galeus*	9	5	55.6
Omnivore	Sparidae	Black musselcracker	*Cymatoceps nasutus*	4	2	50.0
Invertebrate carnivore	Triakidae	Smooth-hound shark	*Mustelus mustelus*	23	11	47.8
Piscivore	Sparidae	Red steenbras	*Petrus rupestris*	17	8	47.1
Omnivore	Sparidae	Panga	*Pterogymnus laniarius*	11	5	45.5
Omnivore	Sparidae	Santer	*Cheimerius nufar*	28	6	21.4
Omnivore	Tetraodontidae	Evil-eye pufferfish	*Amblyrhynchotes honckenii*	19	4	21.1
Omnivore	Ariidae	White seacatfish	*Galeichthys feliceps*	11	2	18.2
Piscivore	Scyliorhinidae	Pyjama catshark	*Poroderma africanum*	13	2	15.4
Invertebrate carnivore	Sparidae	Roman	*Chrysoblephus laticeps*	29	3	10.3
Invertebrate carnivore	Sparidae	Steentjie	*Spondyliosoma emarginatum*	28	1	3.6

DISCUSSION

Monitoring changes in abundance

It is anticipated that the protection afforded by the Stilbaai MPA, proclaimed in 2008, will result in increases in a number of reef fish species over time, judging from responses of reef fish to closures in other MPAs (Bennett & Attwood 1991; Willis et al. 2003). Monitoring of these increases in abundance is desirable for management to assess the efficacy of a MPA in achieving conservation goals (Kelleher 1996; Hockey & Branch 1997; Turpie et al. 2000). To date, there is no evidence that the Stilbaai MPA is adequately designed, positioned and enforced to achieve recovery in reef fish and it is therefore also necessary to consider the possibility of monitoring declines in abundance.

The potential for population increase from low levels of abundance is limited by the intrinsic rate of increase for a species (r), the rate of decrease in fishing mortality and the abundance of the remnant population in a MPA (Jennings 2001). For many species, this r value typically falls in the region of 0.05 to 0.15 per year (Buxton & Clarke 1989). These low rates of increase are typical of long-lived reef fish and, importantly for monitoring, equate into effect sizes as follows:

$$k \approx 1 + \text{lambda} = e^r \text{ for } r << 1.0 \qquad \textbf{eq. 5}$$

The effect size of 1.2 used as a reference in this study is marginally higher than the maximum expected rate of increase for long-lived reef fish (Jennings 2001). Immigration or unusually good recruitment can result in greater than typically expected population increase, but such events are likely to be rare (Jennings 2001). However, this effect size is adequate to measure declines in population abundance, because decreases can be significantly more rapid than increases (Jennings 2001). This is particularly true for exploited species, where fishing mortality rates greater than r are typical. Indeed, fishing rates of 0.2 and 0.3 are common along the South African coastline (Mann 2000).

Whilst the detection of effect sizes greater than 1.2 annually was possible for species such as steentjies and roman, less abundant species such as dageraad and black musselcracker required a greater sampling effort. Differences among species in the sampling effort required to detect changes in abundance are primarily the result of two factors; the overall abundance of a species, and the way individuals of a species are distributed across the MPA.

Given that the population doubling time can be estimated from $\log_e(2)/r$ (Jennings 2001), it follows that a doubling time of five years can be expected from a typical r-value of 0.16. Therefore, for an effect size of two to be a meaningful target, comparisons of abundance would need to take place over longer periods of time than one year. It is thus important to maintain annual monitoring in the Stilbaai MPA, but to expect to detect changes over longer time intervals. If sampling is annual, the results from the power analysis in this study represent a conservative estimate of required sample size. Power analysis of regressions and GLMs could show more feasible annual sampling requirements but these analyses were beyond the scope of this thesis.

Dispersion parameters

An understanding of the way species are distributed across an area is important for monitoring surveys which aim to be representative in terms of relative abundance (Colton & Swearer 2010). Territorial species tend to be uniformly spaced across a habitat, whereas migratory, nomadic and shoaling species have a clumped distribution which is sometimes related to specific habitat features (Pielou 1960; Connell 1963).

Dispersion parameter estimates may indicate the behavioural tendencies towards certain distribution patterns for each species. In accordance with findings from Goukamma, steentjies showed the highest dispersion parameter values (Götz 2006). As a shoaling species, steentjies were observed in high numbers at most sites, but in low numbers at others. The attraction to a site offered by the bait canister would likely draw steentjies from a large area, whereas territorial species would be less likely to leave an area despite detecting bait (Bernard 2012). A patchy distribution may mean that steentjie abundance is overestimated relative to those species with dispersion parameters close to one.

Red stumpnose and red steenbras are solitary large reef fish species with possible territorial behaviour (Buxton & Smale 1989) that may be more realistically represented by BRUV surveys as a result of their more uniform distribution across an MPA. These species had dispersion values less than one. This would mean that the density represented at a particular site for either of these species would be low relative to the abundance observed for steentjies because additional individuals are either not drawn from adjacent areas, or individuals exhibit more migratory movement across the MPA. Therefore, their abundance could actually be more comparable to steentjies' when assessed across the entire MPA, despite strong differences in Max N.

The problem of variable distribution based on different species behaviour will affect the representivity of BRUV monitoring, such that species composition is not necessarily accurately recorded. A similar problem caused by attraction to bait would be encountered in controlled angling. However, in addition to this bias, angling methods have a hook-selectivity bias (Otway & Craig 1993) whilst BRUV surveys are likely more representative of the entire size spectrum. BRUV will remain a relative measure such as CPUE, whilst UVC may provide absolute measures of abundance (Cappo et al. 2004; Colton & Swearer 2010). The relationship between CPUE and abundance is described by the catchability coefficient q. This is described in (Maunder & Punt 2004):

$$CPUE = qN \qquad \textbf{eq. 6}$$

BRUV theory is not developed sufficiently to describe an equation relating Max N to abundance. For the purposes of this study it is assumed to be linear but species-specific. This assumption, however, will not hold true for all species should species abundance increase.

Biases in detecting changes in abundance

Should the density of fish increase, so should Max N as more individuals are immediately in range of the BRUV. However, BRUV could suffer from the problem of saturation, where the abundance of species to be detected becomes limited by factors such as the camera's field of view. This problem is highlighted in controlled angling surveys, where CPUE monitoring is limited by long handling times relative to search times (Maunder et al. 2006). In BRUV surveys, abundant species such as steentjies aggregate at the bait canister in numbers great enough to fill the entire field of view. Should the population increase, the camera's field of view is already filled with individuals and so the species becomes largely insensitive to detecting changes in population abundance over time.

Investigating the steepness of the accumulation slope may overcome these problems. For instance, steentjie abundance starts off low and accumulates gradually over the duration of the video because individuals are attracted from other areas and take longer to arrive at the sample site. The steepness of the accumulation slope should increase at sites where the number of individuals in close proximity to the camera has increased and the time-elapse before fish are drawn from adjacent areas is reduced. Improvements in affordable camera technology also present the option of deploying fish-eye or wide-angle cameras that offer a wider field-of-view to delay saturation effect issues.

Species abundance

Another consideration when interpreting differences in required sampling effort among species is the variation in population size. Since the protection afforded by the Stilbaai MPA is relatively recent, it is expected that species will differ in their respective rates of population recovery because of differences in life-history traits among species (Buxton 1992). As a result, long-lived, slow-growing, sex-changing reef fish species may be present in lower numbers for a longer period before increases are detectable, relative to species capable of more rapid reproduction.

Recording species abundance

The problem that variable dispersion presents for effective monitoring protocol design is evident in the differences observed among species for deployment times to record 95% of the expected abundance. The long deployments required for roman and steentjies as a result of their continued accumulation in the BRUV field of view throughout the video duration may be explained by the attraction of individuals from a larger area. Considering their initial low abundance and gradual accumulation during a deployment, it is reasonable to assume that these species are not necessarily highly abundant at a particular site, but are attracted to the site as the bait plume disperses across a wider area over time.

Conversely, black musselcracker and dageraad required the shortest deployment times to record 95% of the expected abundance. Given that both species had dispersion values close to one, this may indicate that individuals are not attracted from a wider area by the presence of bait and that the measure of abundance to be expected at one site for both species will necessarily be low. Black musselcracker utilises shallow reefs as nursery grounds, whilst solitary adults occur in low abundance on deeper reef systems (Buxton & Clarke 1989). This result may therefore corroborate findings that the behaviour and life history of species influences the sampling effort required (Watson et al. 2010; Bernard 2012).

Alternatively, this result may also indicate that both species are present in low abundance in the Stilbaai MPA, so that Max N will necessarily be low and therefore recorded in a shorter time period. Given that both species are long-lived, slow-growing reef fish it is conceivable that population-level response to protection will be slow. Indeed, studies have shown that dageraad numbers are severely depressed as a result of exploitation by commercial and recreational fisheries (Buxton 1992; Heemstra & Heemstra 2003) and that its life history slows its population-level response to protection (Buxton 1992).

Sample size requirements

The influence of species distribution and abundance on sampling effort is apparent in differences among required sample sizes obtained in the power analysis. Red stumpnose and red steenbras both required more samples to detect a significant change in abundance than either roman or steentjies. The solitary behaviour and more uniform distribution of red stumpnose and red steenbras across the MPA would result in low abundance at any one particular site. This increases the required sampling effort required to detect significant changes over time. However, uniform distribution may actually lower data variability and increase statistical power. Given that the recording of their distribution over space is likely more realistically captured by BRUV than that of steentjies or roman, these sample sizes represent the protocol required to record targeted, territorial species' relative abundance at a sample site.

Roman and steentjie require the least samples to detect significant changes in abundance, a finding that points to their ubiquitous distribution across the MPA. Whilst steentjies are shoaling and therefore clumped in their distribution across sites, they differ from the other shoaling species recorded in this study (strepies) in that they are common to all depth strata and profile types. High abundance and uniformity are two important contributing factors to the high power of these BRUV data.

Selecting ubiquitous species for monitoring protocols is desirable in order to minimise sampling effort, in particular when effect sizes are low. Steentjie and roman require 209 and 186 samples to detect a 20% change in abundance, respectively. The improvement in affordable, high quality camera technology extends the potential of BRUV sampling to include the deployment of multiple cameras that record data simultaneously at different sites. However, these sample sizes are more realistically achievable over longer time periods, particularly given the accumulated quantity of footage that has to be analysed after each survey. It is therefore more realistic to monitor annually over at least five years, whereafter the latest can be compared with the first sample and a more significant result can be detected.

The practicality of annual BRUV monitoring over long timescales becomes apparent when considering sampling requirements for species that were recorded infrequently. Studies have recommended rarer species as models for monitoring-design so that the most conservative sampling regime is selected and the detection of other more frequently observed species will automatically be encompassed in sampling (Watson et al. 2010; Bernard 2012). This is of particular importance where target species are concerned and a representative monitoring protocol would therefore need to consider an upper estimate of sample size. For dageraad, this translates to 267 samples monthly and becomes logistically unachievable in the short-term.

The required sample size for each species assessed in the Stilbaai MPA differed from the results obtained for the same species from the Tsitsikamma MPA (Bernard 2012). However, these differences were slight, with roman requiring 11 rather than eight samples, red steenbras requiring 22 rather than 27 samples and steentjie requiring 12 rather than 11 samples to detect a population doubling in Stilbaai. The ubiquitous, abundant roman and steentjie required the lowest sampling effort - a pattern concordant with that found in Tsitsikamma (Bernard 2012).

Whilst the required sample sizes appear particularly high for rare, territorial, nomadic or migratory species, it is important to assess these results in light of required sampling effort using traditional monitoring techniques. A power analysis investigating UVC surveys of roman in the Tsitsikamma MPA showed that 15 samples were necessary to detect an effect size greater than two (Bennett et al. 2009). It is expected that sampling effort to detect changes in roman abundance using UVC would be higher than for BRUV or controlled angling, because the attraction to bait would draw individuals from a larger area (Bernard 2012).

Given that controlled angling and BRUV sample a very similar proportion of the species assemblage but that BRUV does so non-extractively, it is perhaps most important to investigate differences in sampling effort between these two methods. The required sample size of 12 for controlled angling surveys of roman to detect an effect size of k = 2.0 gives power of 55% (Bennett et al. 2009), whilst a sample size of 11 for roman in Stilbaai gives power of 80%. This point is corroborated by studies that showed BRUV sampling obtained data with higher statistical power, lower variance with lower sampling effort relative to other monitoring techniques (Langlois et al. 2010). Furthermore, BRUV sampling is less biased by observer skill and decreases the length of time required to sample an MPA effectively relative to controlled angling surveys.

Recording species diversity

A BRUV deployment 13 minutes longer than the 36 minute minimum suggested by Watson et al. (2007) is necessary to record 95% of species present in the Stilbaai MPA. These findings may point to differences in deployment times between temperate and tropical reefs. However, extending deployments to a more conservative 60 minutes (Watson et al. 2007) would sufficiently record both species diversity and abundance in Stilbaai. Optimal CPUE sampling was calculated as two angler hours for temperate reef fish (Bennett et al. 2009). This is represents greater sampling effort relative to BRUV deployment time requirements. Interestingly, the 49 minute optimal deployment time obtained for Stilbaai is lower than the 57 minutes obtained for Tsitsikamma (Bernard 2012). A greater proportion of high profile reef represented in Tsitsikamma provides hiding places for fish and may therefore have contributed to the longer deployment time estimates.

Differences in the type and abundance of species recorded in different habitats may influence the required deployment time (Watson et al. 2010). This may be the result of higher species diversity at high profile sites caused by the availability of additional prey and refugia (Buxton & Smale 1989). However, the marginal difference in deployment times for sampling deep, low profile reef and shallow, high profile reef in Stilbaai suggest that an overall deployment time of 49 minutes is suitable across both habitats. For the sake of consistency and comparability of results in the long-term, it is advisable to adhere to the standard 60 minutes suggested in previous studies (Cappo et al. 2004, Watson et al. 2007).

Species interaction

Species respond differently to bait in BRUV sampling (Watson et al. 2010). Studies have suggested that competition at the bait canister may influence the proportion of the species assemblage recorded using BRUV (Armstrong et al. 1992). It was often observed that the arrival of certain species deterred other species from the bait canister. This was particularly evident for the arrival of certain shark species, but the quantitative analysis of these interactions lay beyond the scope of this thesis and did not fulfil primary aims.

Whilst roman and steentjie accumulated gradually in abundance over the entire deployment interval, so that their numbers were highest towards the end of a deployment, santer accumulated rapidly initially and then decreased in abundance as other species arrived at the bait canister. This may be attributed to dominance of other species (Watson et al. 2010). It was evident during footage analysis that the accumulated presence of roman coincides with the stage at which santer decreased. This may be the result of roman territoriality and the inherent aggressiveness necessary to defend an area. Results highlight the potential application of BRUV data for behavioural observations. This would be of particular interest where species composition in a MPA changes over time due to recent protection or lack thereof, and the abundance of dominant species vacillate.

Incidental species recordings

The BRUV deployment did well to sample predatory and elasmobranch species recording three catshark species along with ragged-tooth, soupfin, smooth-hound and bronze-whaler sharks. This parallels findings that BRUV monitoring samples a higher abundance of predatory species and targeted species (Cappo et al 2004; Colton & Swearer 2010; Watson et al 2010). This is an important feature of a monitoring technique, since predatory species are often the first to be exploited in an area (Pauly et al. 1998). The ability of BRUVs to record predators usually deterred by SCUBA diver presence (Stobart et al. 2007) or recorded through extractive angling surveys (Bennett et al. 2009) beneficially addresses gaps in traditional monitoring techniques (Willis et al. 2000; Cappo et al. 2003; Cappo et al. 2004).

A representative monitoring technique should record as wide a variety of species as possible (Stobart et al. 2007; Watson et al. 2010). This study supports findings that predatory species attracted to feed at the bait canister are not the only species recorded (Watson et al. 2010). This presents an improvement on CPUE measures of species diversity, where only species attracted to bait would be sampled. Zebra, blacktail and red stumpnose were frequently recorded species across all sites, but were not once recorded feeding at the bait canister. It is possible that some species are attracted within the BRUV's field of view by the activity of other fish feeding at the bait canister, and that some species were not deterred by the dominant behaviour of species monopolising the bait (Watson et al. 2005; Harvey et al. 2007; Watson et al. 2010).

Conclusion

A study by Cappo et al. (2004) concluded that BRUV facilitates a more precise assessment of species distribution and abundance relative to traditional monitoring techniques. This study suggests that differences in the way species distribute themselves across an area will influence the accuracy with which their abundance is assessed. This finding explains the considerable variation observed among species in the required sampling effort to detect temporal and spatial changes in abundance.

The detection of several important fisheries target species; notably, dageraad, roman, black musselcracker and red steenbras supports findings that BRUV monitoring is particularly useful for the assessment of exploited species (Stobart et al. 2007; Colton & Swearer 2010) and recommends BRUV surveys for future monitoring. The result is expected, because exploitable fish species would be attracted to bait. These species are also of particular monitoring interest because the nature of their life-histories slows their population recovery once afforded protection (Jennings et al. 1998).

The detection of significant changes in abundance for rare, nomadic or migratory species is unrealistic on an annual monitoring timescale because the sample sizes required to accurately assess changes are impractically large. However, the achievement of required sample sizes becomes increasingly realistic if monitoring is conducted over longer timescales. This is made even more pertinent considering that the detection of population increases in slow-growing, long-lived reef fish species will only be achieved over a timescale of at least five years (Barrett et al. 2007). When assessing the viability of BRUV for long-term monitoring along the South African coastline, it is important to consider the other advantages this technique offers.

BRUV is a non-extractive method that is particularly adequate in assessing predatory and fisheries target species in a MPA, and has been shown to detect a broader range of species than just those attracted to the bait. Archived video footage presents a unique opportunity to raise public awareness and support for MPAs and conservation, as well as for introducing considerable transparency in MPA monitoring and data collection. The required deployment lengths still reduce sampling time relative to controlled angling and increase data collection time relative to SCUBA surveys while recording higher fish diversity. Future examination of the effect of species saturation and agonistic interactions would further refine this methodology to address challenges with the accurate detection of species abundance over time.

CHAPTER 4

Study review

CONCLUSIONS

The methods outlined in this study are highly repeatable, and provide a transparent methodology for assessing fish abundance and species richness in MPAs. Variation in reef fish abundance and distribution can be explained by depth, sea temperature and reef profile (Chapter 2). These relationships aid future monitoring efforts in the Stilbaai MPA by guiding effective habitat stratification during sampling design, and allows for focused monitoring and the correct interpretation of data. Additionally, this information will guide any future considerations to adjust the borders of the Stilbaai MPA. The compilation of species richness and relative abundance data within the MPA establishes a baseline against which the results from future monitoring efforts can be compared to confirm the functioning of the MPA and describe the impact of long-term climate change.

For the sake of comparability across South African MPAs, a standardised methodology is preferable (Chapter 3). The overall optimal deployment time of 60 minutes, suggested by international studies and results from Tsitsikamma, is supported by this study and is a conservative measure to record 95% of species captured by BRUV. As a more realistic measure to detect abundance changes in slow-growing sea breams that represent a significant number of species of monitoring concern in Stilbaai, sample sizes to detect a 20% annual change in abundance are recommended. Significant changes in abundance would be anticipated not annually, but over longer time intervals. Annual monitoring will, however, establish a monitoring regime in the Stilbaai MPA that ensures management presence in the area for law enforcement and awareness-raising, as well as the collection of long-term datasets that can then be used to compare across a period of at least five years.

The detection of species depressed by fishing pressure (such as roman, dageraad, black musselcracker and red steenbras) is an encouraging sign for a recently-promulgated MPA, and changes in their abundance should be monitored. Moreover, the agreement of these findings with those obtained using traditional monitoring methods and results from similar habitats (Tsitsikamma) bodes well for the continued use of BRUV technology in South Africa.

TECHNICAL SUGGESTIONS

Aspects of the BRUV tripod set-up could be improved to ease handling. The tripod's high centre of gravity made it susceptible to falling over in strong bottom-surge, and shifting currents often dragged the entire set-up from its place of settlement. Fortunately, the camera monitor allowed for remote-viewing on the boat. Irregularities were quickly corrected during a deployment, either by pulling the tripod upwards by means of the rope to re-settle, or by re-anchoring so that the current stopped dragging on the anchor and the BRUV tripod. An amended tripod design that lowers the tripod's centre of gravity may eliminate these issues.

The major advantage of BRUV monitoring is data collection with lower manpower and costs. However, the camera used in this study remains outside most MPA budgets and was on loan from the South African Environmental Observation Network (SAEON). For a national rollout, a refined BRUV system should explore improving camera technology and affordability. Reducing the input costs for the system will make it more feasible to implement as a monitoring tool and not simply as a scientific experimental system.

A measure of bottom visibility should be obtained using two parallel laser pointers attached to the BRUV tripod for each deployment to estimate horizontal underwater visibility, which are cheaper and more effective than a turbidity meter. This will refine the measurement of visibility.

FUTURE RESEARCH

Having established a baseline assessment of species distribution, richness and relative abundance in the Stilbaai MPA, the establishment of an annual monitoring programme using BRUV technology is now feasible. Annually collected data should be archived and compared with this initial assessment to monitor change in terms of species recovery, shifts in species composition and relative abundance, and long-term habitat alteration. Monitoring should consider assessments over seasonal change, and compare species abundance and richness inside and outside the MPA. An interesting addition to monitoring will include the use of archived video footage in education of local fishers and visiting tourists.

A national rollout of BRUV technology for MPA monitoring along the South African coastline will address the logistical issues impeding regular, sustainable assessments. The refinement of the BRUV system should consider GoPro© HD cameras for improved image resolution (particularly helpful when freezing video frames for correct identification) and cheaper camera technology. Lowering camera costs will afford annual MPA budgets the use of multiple camera systems. Future camera set-ups need not be connected to the boat, but can be buoyed off, facilitating simultaneous data collection at multiple sites. This will increase time-efficiency and the quantity of data collected. In this way, obtaining sample sizes in excess of 300 deployments to detect significant changes in rare and target-species abundance becomes feasible on an annual timescale. Additionally, laser pointers or stereo-cameras should be considered to collect visibility and fish size estimates. Size estimates will be useful for comparisons inside and outside MPAs, where this is likely the parameter most sensitive to changes through exploitation (Leaman 1991).

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Glossary of statistical acronyms

Acronym	Explanation
MDS	Multidimensional scaling
ANOSIM	Analysis of Variance
SIMPER	Analysis of dissimilarity
BIOENV	Best of Environmental Variables

APPENDIX I

Table 1. Environmental variables measured at 29 sites in the Stilbaai marine protected area

Site	Visibility (m)	Depth (m)	Water Temp (°C)	Reef profile
1	5.00	38.00	15.29	Low
2	5.00	34.00	15.39	Low
3	8.00	30.00	15.53	Low
4	5.50	28.00	15.70	Low
5	5.00	13.00	15.96	Low
6	4.00	28.00	15.82	Low
7	5.00	25.00	15.94	Low
8	3.00	18.00	15.87	Low
9	2.50	35.00	15.75	Low
10	6.50	16.30	17.65	Low
11	7.50	26.00	17.53	High
12	6.00	15.00	17.63	High
13	7.00	33.00	17.51	Low
14	6.00	15.00	17.61	High
15	6.00	13.00	18.25	Low
16	6.00	12.00	18.34	Low
17	7.50	14.00	18.27	Low
18	8.00	17.50	18.22	High
19	10.00	31.00	17.94	Low
20	8.50	32.00	18.03	Low
21	7.00	35.00	16.73	Low
22	6.00	22.00	17.37	High
23	7.00	13.00	17.72	High
24	7.00	33.00	16.73	Low
25	5.50	9.00	18.84	High
26	6.50	22.00	17.49	Low
27	5.00	6.00	19.65	High
28	6.00	6.50	19.53	High
29	8.50	18.00	17.44	High