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An effective method for assessing effects of no-take zones in reef fish species: Applied in a MPA in Tanzania.

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Mestrado em Biologia Marinha

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Resumo

Os estudos de monitorização de Áreas Marinhas Protegidas (MPA) em países em desenvolvimento nos trópicos, onde ocorre a maior proporção de recifes de coral do mundo, tendem a ser demasiado ambiciosos no início, confiando em recursos financeiros e humanos tão elevados que, quando o financiamento parar, o estudo acabará por não ser continuado e não se obterão resultados reais. Além disso, a variabilidade na selção de diferentes técnicas de amostragem, bem como a temporalidade e área abrangida, de acordo com a experiência e preferências do investigador neste tipo de estudo de monitorização, significa que os resultados dos estudos em MPAs estabelecidos em habitats semelhantes, sob as mesmas ameaças e condições de gestão, não são comparáveis. Para além da obtenção de dados com melhor valor estatístico para caracterizar as MPA, nas áreas protegidas estabelecidas para proteger os recursos vivos de artes de pesca destrutivas e sobre-exploração, o objectivo é obter dados científicos fiáveis sobre o estado da área em questão para a gestão da pesca. Esses dados servem para ajudar gestores responsáveis pela fixação de restrições dentro da área e para que, a longo prazo, a área em questão abasteça os recursos pesqueiros locais e, portanto, a população local cuja principal fonte de proteínas provém dos recursos marinhos, através da exportação de adultos.

Este estudo teve por objetivo desenvolver e demonstrar uma técnica fácil de usar, barata e precisa para a avaliação das populações de espécies de peixes associados a recifes importantes para a pesca local. Para isso, avaliamos factores que podem interferir com a amostragem e a obtenção de resultados fiáveis para melhoramento, e descrevemos as características estatísticas dos dados obtidos de forma prática numa área marinha protegida na ilha de Pemba, Tanzânia, denominada 'Kwanini Marine Protected Area' (KMPA). Com os resultados obtidos, procurámos responder a duas questões: (i) houve variação na abundância e/ou biomassa desde a imposição de proibições à colheita de recursos marinhos de todos os tipos no âmbito do KMPA? E, (ii) um ano após a imposição da lei de não utilização é tempo suficiente para ver as diferenças entre a KMPA e as áreas não geridas circundantes? A amostragem foi levada a cabo por dois mergulhadores utilizando scuba em circuito aberto. Nadando a uma determinada profundidade e a uma velocidade constante, o primeiro mergulhador levou a câmara GoPro e uma extremidade de uma fita métrica. O segundo mergulhador levava a outra extremidade da fita e alertava o primeiro mergulhador quando alcançava o limite de transecto para parar a

gravação. Foi deixada uma distância de 10 m entre os transectos para evitar a duplicação de informação. Desta forma, foram obtidos vídeos de 50 cm x 5 cm transectos dentro e fora da KMPA e a diferentes profundidades para quatro campanhas: Julho 2019, Outubro 2019, Janeiro 2020 e Novembro 2021. A fim de padronizar o método, foi feita uma pré-selecção de espécies importantes na pesca e associadas aos recifes de coral KMPA: *Lethrinus harak* (Lethrinidae), *Macolor niger* (Lutjanidae), *Parupeneus macronemus* (Mullidae), *Chlorurus sordidus* (Scaridae) e *Cephalopholis miniata* (Serranidae). Desta forma, o investigador apenas teria de procurar indivíduos desta espécie, contá-los e classificá-los em classes de tamanho pré-determinado (pré-adulto, adulto e veterano, de acordo com os registos da literatura). Algumas espécies de peixes pequenos recorrentes nos vídeos e associados aos recifes de coral foram utilizados como referências de tamanho quando havia qualquer dúvida na classificação de tamanho de um determinado indivíduo. A necessidade de peritos em censos de peixes por parte dos colectores de dados foi assim minimizada em comparação com as técnicas de observação in-situ utilizadas noutros estudos. Posteriormente, utilizando a abundância de indivíduos, o comprimento total médio (cm) das categorias de tamanho de cada espécie e as relações comprimento-peso específico de cada espécie obtidas em estudos anteriores, foi possível calcular a biomassa de cada espécie por transecto (g/m2).

As espécies mais amostradas neste estudo foram *P. macronemus* (N=207) e *C. sordidus* (N=238). Os dados de *M. niger* foram ignorados na análise estatística devido ao baixo número de indivíduos amostrados (N=9). Os resultados da análise estatística dos dados de biomassa para cada espécie mostraram diferenças estatisticamente significativas entre as campanhas (Kruskal-Wallis, P<0,05) para a espécie *C. sordidus*. Os resultados mostraram respostas negativas às perguntas sobre efeitos de gestão, selecionadas como objetivos de estudo. No entanto, pensa-se que isso pode ser devido à curta escala temporal, dado que a amostragem durou apenas 2 anos e que se pode ver nos resultados uma relação positiva entre o tamanho da amostra e as diferenças entre estações de amostragem, prevemos com alguma confiança que no futuro serão obtidas diferenças significativas para o resto das espécies pré-seleccionadas, especialmente para *P. macronemus*, se algumas fontes de erro, tais como condições de sazonalidade e visibilidade forem minimizadas na concepção da amostragem futura e a recolha de dados continuar a aumentar.

Em conclusão, o método proposto neste estudo é suficientemente preciso para caracterizar o estatuto relativo do KMPA e outros MPAs semelhantes, e fornecerá à equipa de gestão do MPA conteúdo científico suficiente para ajudar nas decisões futuras relativas às limitações impostas, encorajar mais apoio financeiro, justificar o financiamento e inspirar outras organizações a criar novos MPAs. Além disso, o aperfeiçoamento dos potenciais erros de amostragem encontrados ao longo dos seus testes permitirá que o método descrito neste estudo seja exportado para outras regiões, de modo a que a comunidade científica possa estabelecer uma rede de AMP cujo acompanhamento proporcione resultados comparáveis, não só para aprender com as realizações uns dos outros, mas também para ajudar na recuperação dos recifes de coral que estas medidas legais protegem e para mitigar, tanto quanto possível, os efeitos das alterações climáticas neste rico habitat.

Abstract

In tropical developing countries, reef monitoring programmes inside protected areas (MPAs) suffer from being unrealistically large and sophisticated. In addition, the lack of consensus among the scientific community on the experimental design to be used makes the standardisation of a method to compare results between different locations of the globe imperative. Our objectives were to describe an easy, affordable, and precise method for determining reef fish abundance and biomass as indicators of reef health and to evaluate the usefulness of this method to address questions of relevance for MPA management. On Pemba Island (Tanzania), within the no-take Kwanini Marine Protected Area (KMPA) and its buffer zones, diver operated video (DOV) samplings were conducted utilising a single camera along 50x5 cm belt transects. A list of reef fish species targeted by the local fishery was previously compiled, and then the video footage was played back on a computer to search for the selected species, count them, and classify them into pre-determined size categories. No experience with fish identification was needed throughout the sampling. From the numerical abundance data, average lengths, and species-specific length-weight relationships from literature, we were able to determine biomass (g/m2) results per species for different campaigns. The 2 years of data available were insufficient to detect significant changes but this approach provided valuable insights into the population variability of previously over-exploited species. The method is accurate enough to provide efficient monitoring information, if the sampling continues in the long-term, a relative status of not only the KMPA but other areas under similar restrictions, helping with future management decisions, encouraging more support, justifying investment and gaining a broader understanding of how MPAs in the western Indo-Pacific are helping coral reef habitats not only to recover from destructive fishing practises but also to cope with the effects of climate change in the near future.

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List of Abbreviations, Acronyms, and Symbols

BACI – Before-and-After Control Impact design

BRUV – baited remote underwater video

BRUVS – Baited Remote Underwater Video Stations

CPUE – Catch Per Unit Effort

DOV – Diver-Operated Video

KMPA – Kwanini Marine Protected Area

MPA – Marine Protected Area

ns – No significance within the statistical hypothesis test (P>0.05)

RUV – remote underwater video

RVT – Rapid Visual Technique

SCUBA – Self-Contained Underwater Breathing Apparatus

SPC – Stationary Point Counts

TL – Total Length

UVC – Underwater Visual Census

* – Significance within the statistical hypothesis test (P≤0.05)

CHAPTER 1. GENERAL INTRODUCTION

Marine fisheries in Tanzania

Tanzania is a country home to approximately 58 million people (NBS, 2020) and consists of mainland Tanzania and the Zanzibar Archipelago, Pemba Island and Unguja Island. The territorial waters – immediately adjacent to the shores of the state and thus subject to the territorial jurisdiction of that state – are estimated to be 4,450 km^2 and form a major fishing area (Feidi, 2005). Recent changes in the demography, economy and level of technology in Tanzania have led to the fast development of its fisheries from 'traditional and non-monetized' to 'artisanal and cash-oriented' (Jiddawi and Öhman, 2002), affecting the marine environment and transforming the ecosystems (Katikiro *et al.*, 2012). However, despite these changes, coastal communities continue to depend on fisheries for food security culture and livelihoods (Dyck and Sumaila, 2010; Teh and Pauly, 2018). In general terms, since fish are the main source of dietary protein on the island of Zanzibar (Johnstone *et al.*, 1998), the coral reefs of Pemba Island play a crucial role in the welfare of the majority of the coastal communities.

The Tanzanian fishers cannot usually access offshore areas due to lack of seaworthy vessels and gear to retain large fish (Jiddawi and Öhman, 2002), so most fishing is carried out relatively close to shore (within 5 miles of the shore) and the nearby reefs are thus intensely fished (Feidi, 2005; Grimsditch *et al.*, 2016), ultimately leading to a state of localized overexploitation. In the late-1980s, the fisheries changed fundamentally and deleteriously in character (Katikiro *et al.*, 2012). Beach seines, drag nets, and dynamite fishing are typical of the destructive methods in the area that cause significant damage to the coral reef structure and populations (Jiddawi and Öhman, 2002; Grimsditch *et al.*, 2016). Most of the destructive methods are prohibited by law but continue to be used due to lack of surveillance, enforcement, and public awareness (Jiddawi and Öhman, 2002).

Dynamite fishing, one of the most destructive fishing techniques, has been practiced in Tanzania for over 4 decades (Samoilys and Kanyange, 2008). A typical dynamite explosion kills all fish within a 50-70 m radius (Alcalá and Gómez, 1987) by sending a powerful shock wave through the water that bursts a fish's internal organs while the fish's skeleton suffers hundreds of small fractures (Ronquillo, 1961). Although the Tanzanian government banned dynamite fishing in 1970, it has continued largely unabated since that time (Guard and Masainganah, 1997; Wells, 2009; Braulik *et al.*, 2015) and on some parts of the coast it is said that is becoming 'fishing as usual' (Slade and Kalangahe, 2015). Moreover, since COVID-19 hit the Tanzanian tourism economy and forced many locals to fish for a living, fishing using explosives is now more common again along the Tanzanian coast.

Figure 1.1. Comparison between a coral reef in favorable condition (left) and a reef in degraded condition (right) in Kwanini Marine Protected Area, Pemba Island, Tanzania. Copyright Kwanini Foundation.

Coral reefs and biomass associated

Coral reefs are located along about two thirds (600 km) of Tanzania's continental shelf (Wagner, 2004). Pemba, the second main island of Zanzibar, is estimated to have 1,100 km of reef, representing 45% of the reefs of Tanzania (Wagner, 2004). On the gradient of diversity, they are among the 'second tier' of the most diverse coral reefs in East Africa and locally extremely important since the local human population relies on them heavily for food security and income from fishing, aquaculture and increasingly SCUBA diving tourism (Grimsditch *et al.*, 2016). According to the IUCN report, dynamite fishing and overfishing are the greatest threats to Pemba's coral reefs (Grimsdith *et al.*, 2009). More recent observations from North Pemba confirm that the situation has not changed and that these remain the primary threat to reef integrity (Barratt, *pers.com.* 2022)

Coral reefs have high productivity and biodiversity and are considered key ecosystems (Hunter Jr and Gibbs, 1996). Although coral reefs cover much less than one percent of the ocean floor, they support between a quarter and a third of all species of marine fish (Moberg and Folke, 1999). Even if not targeted, corals are destroyed by the explosion of fishing blasts, usually leaving behind a large crater of coral rubble surrounded by a band of standing dead coral (Alcalá and Gómez, 1987; Darwall *et al*., 1996) (Fig. 1). In addition, Nzali *et al*., (1998) found that, before the reef was short-term destroyed, the damage was so severe that the reef's ability to recover is hampered by the absence of nearby coral colonies that can serve as recruits to recolonize the exposed substrata. Benthic disturbances produced by dynamite fishing and other destructive fishing techniques alters fish assemblages that rely on corals for survival by affecting availability of food and shelter (Wilson *et al*., 2006; Pratchett *et al.*, 2011), and may result in reduced abundance, local extinctions, diversity loss (Graham *et al*., 2006; Holbrook *et al*., 2015; Newman *et al*., 2015) and declining fisheries productivity (Rogers *et al*., 2014). Also, shifts in the diversity of fish and invertebrate communities due to overfishing affect fundamental structural processes such as calcium-carbonate accretion by the increase bioerosion of coral reefs (Glynn *et al.*, 1979; McClanahan and Muthiga, 1988; McClanahan, 1992). The natural recovery of the reef is very slow or inexistent; Alcalá and Gómez (1987) demonstrated that hard corals regenerate minimally following dynamite blasting, even after 40 years. Thus, the human intervention is needed to regenerate these areas.

No take zones as the answer to the problem

In response to the many threats biodiversity and ecosystem services of coral reefs face nowadays, no-take marine protected areas (MPAs), where all forms of extraction are banned, are often used as a management tool as, inside their limits, undisturbed habitats are maintained, populations increase in size and diversity, and individuals live longer, grow larger and develop increased reproductive potential, while providing a recruitment source for surrounding areas and restocking through emigration (Bohnsack, 1990; Bohnsack, 1998; Kamukuru *et al*., 2004). Also, they provide a protected environment for testing reef restoration techniques to try to recover coral populations from destructive fishing practices, and thus the biomass associated with healthy reefs. This is especially relevant in developing island nations where animal protein is often sourced from coastal coral reefs (Cabral and Geronimo, 2018), as it is the case of Pemba Island.

Fish surveys on uninhabited, remote coral reefs (Friedlander and DeMartini, 2002; Stevenson *et al*., 2007) support historical reports of great fish abundance and predator domination that characterized coral reefs before degradation by extensive and destructive fishing efforts. A primary aim of most MPAs is to increase biomass of targeted fish stocks inside and eventually outside MPA boundaries through two mechanisms: net migration of adults and juveniles across borders, termed ‘spillover’, and export of pelagic eggs and larvae (Gell and Roberts, 2003). The creation of MPAs may involve negative impacts on local fisheries as a consequence of the fishing regulations (Guidetti and Claudet, 2010) or the subtraction of fishing grounds (McClanahan, 1999). However, the results obtained after eight years since the Mombasa Marine National Park in Kenya became fully protected proved the theory, when catches nearby the reserve boundaries reached three times more than those further away (McClanahan and Kaunda-Arara, 1996; McClanahan and Mangi, 2000). Density-dependent spillover from marine reserves has been also documented by other studies (Abesamis and Russ, 2005). Furthermore, the establishment of MPAs may increase the resilience of reefs by decreasing sensitivity of corals to warming (Wooldridge, 2009) and facilitating recovery of the reef (Mumby and Harborne, 2010; Olds *et al*., 2014) when removing one anthropogenic stress (i.e., fishing) (see Roberts *et al*., 2017), as shown in the study of Grimsditch *et al*., (2009) about the assessment of coral reef resilience of the Pemba Channel Conservation Area. That study recorded the highest hard coral cover (86%) and the highest coral diversity (42 genera) in the no-take zone at Misali island, while degraded sites such as Paradise and Fundo Outer had low coral cover (3% and 5% respectively), low coral diversity (23 and 33 genera respectively) and were dominated by rubble and turf algae. Moreover, the study of McClure *et al*. (2020), proved the effectiveness of no-take MPAs in the Central Visayas region of the Philippines, where total biomass of all reef fish over 10 cm total length was 135-173% higher inside the MPAs than fished areas, which may provide a source of larvae to replenish fish populations, inside and outside the MPA. Therefore, spillover is a density-dependent mechanism where an increased competition within MPA triggers a movement/transfer of individuals/biomass towards adjacent areas outside MPAs (see Di Lorenzo *et al*., 2016).

Inlight of this, the Kwanini Foundation and Manta Resort, with the support from the local community and the government, turned 1 km of coastline into an MPA called the Kwanini Marine Protected Area (KMPA) in 2013 on Pemba Island, Tanzania. In 2018, the Bylaw designated the KMPA as a ‘No take zone’, where fishing, gathering, harvesting, collection and cultivation of marine species are permanently prohibited. The management plan which governs the implementation of the Bylaw, states that the Foundation will use fish biomass as one of the indicators to assess the ecosystem integrity after improving 5,200 ha of coral reef ecosystems by protecting/enhancing ecosystem functioning, processes, and services. After 10 years, an increase of 10% from baseline data is anticipated.

Methods to assess reef fish abundance in no-take MPAs

Management of MPAs, either with partial or total protection from harvesting, require non-destructional observational methods that provide accurate and precise population data on the resident species to quantify the effect of protection (Edgar and Barrett, 1997; Murphy and Jenkins, 2010). Size-specific data with interpretations in units of biomass, calculated from numerical abundance and species-specific length-weight relationships, are extremely important for the study of fishes, given the indeterminate nature of fish growth and size-specific variation of fish functioning (e.g., fecundity, predation threat) (Helfman *et al*., 2009). Survey data can also be used to estimate the number of individuals of each species of interest within a defined area, making it possible to derive estimates of density (numbers of individuals per unit area) and diversity for the sampled taxonomic group (Brock, 1954, Helfman *et al*., 2009).

Destructive sampling methods like collecting using explosives (Russell *et al*., 1978; Williams and Hatcher, 1983), gill-nets (Howard, 1989), traps (McClanahan and Mangi, 2000; Chapman and Kramer, 2000; Kaunda-Arara and Rose, 2004), trawling (Jones and Derbyshire, 1988; Watson and Goeden, 1989; Wassenberg *et al*., 1997; Cappo *et al*., 2004) or hook-and-line (Zeller *et al*., 2003; Taylor and McIlwain, 2010), even providing more accurate estimates of fish density than observational methods, should not be considered to assess any trend within MPA's limits because of legal and also ethical issues, but also because of its low precision for large areas and size selectivity (Edgar *et al*., 2004; Cappo *et al*., 2004; Cote and Perrow, 2006). The observational methods most currently used in marine spatial monitoring surveys include fishery-dependent and fishery-independent methods (Murphy and Jenkins, 2010). Fishery-dependent methods rely on the data of catch, effort and catch per unit effort (CPUE) obtained from the extraction of marine resources, either commercial or recreational (Pennington and Strøme, 1998; Hubert and Fabrizio, 2007; Murphy and Jenkins, 2010), but it represents the same problem as destructive samplings methods since they cannot be used within a fishery exclusion area (Murphy *et al*., 2010), but they can be used to provide information on fish stocks occurring outside the boundaries of the MPA (Alcala *et al*., 2005).

There are many methods independent of fisheries' catches to obtain accurate and precise population data on target and non-targeted fish species to monitor no-take MPAs' protection over fish populations. Among them, we can find underwater visual census (UVC), diver-operated video (DOV) census, baited remote underwater video stations (BRUVS), remote sensing and spatial monitoring (Mumby *et al*., 1999), hydroacoustic (Cote and Perrow, 2006; Egerton, 2018; Hossain *et al*., 2018) and environmental DNA (Maruyama *et al*., 2014). Looking for the cost-effectiveness and replication in developing areas, where coral reefs mostly occur, UVC and its variants DOV and even BRUVS are the monitor techniques more suitable to be used.

The most common method of surveying and monitoring shallow water fish assemblages is underwater visual census (UVC) (Caldwell, 2011), a method of estimating number and weight of reef fish by means of submarine counts and length estimates (Brock, 1954). This technique requires minimal equipment that is cheap and easily accessible (pencil, slate, and tape measure), is not disruptive to the habitat and it is not time-consuming after the sampling (Holmes *et al.,* 2013). In addition, they are generally less selective than most other sampling techniques (Brock, 1954). However, UVC techniques have their own problems and limitations which ultimately lead to under and overestimating fish abundance detectability biases (Bozec *et al*., 2011). In the first place, since these techniques are usually employed in situ, it requires participants to use SCUBA, so depth and observation time are often limiting factors in their implementation (Bortone *et al*., 1992). Moreover, this technique usually requires a high degree of observer expertise regarding fish identification to the species level and the accuracy and precision of size data gathering, which may lead in observer biases (Edgar *et al.,* 2004). Despite this, UVC has been commonly employed by fish monitoring programs around the world, particularly by research institutions (Holmes *et al.,* 2013).

In UVC surveys, three techniques seem to be the most widely used in the research community to survey reef fishes: the belt transect, stationary point count (SPC), and rapid visual technique (RVT) (Table 1.1). The belt transect (Brock, 1954) consists of a team of divers swimming along a transect and tallying the fishes observed in a predetermined swath. The most common transect length is 25 or 50 cm, and the transect swath, between 2 and 5 m. The transects usually run parallel to the shore and follow a uniform depth and habitat type. In the SCP method (Bohnsack and Bannerot, 1986), at each sampling point the diver remains stationary while counting all the fishes in an imaginary cylinder of warter from surface to the bottom for 5 minutes. The radius of the cylinder is measured by using a tape measure and must be set as a constant for the project, whose size depends on the visibility and the targeted fish species (Caldwell, 2011). The study of Samoilys and Carlos (2000) showed that no significant difference is shown when measuring fish densities between belt transects and SPC, so the choice of either survey approach is typically dependent on the researcher's experience, common sense, personal bias, and previously employed methods (Samoilys and Carlos, 2000). The RVT (Jones and Thompson, 1978) use time as the constant and the targeted species need to be large, usually greater than 20 cm. The diver does not follow an established route but is allowed to free swimming for the previously set search time interval within the area of study, being the species recorded only once (Jones and Thompson, 1978). With this technique, an index based on scores calculated from the rank order of species encountered is used to define reef fish assemblages (DeMartini and Roberts, 1982) and allows to cover a large amount of reef area (see Caldwell, 2011).

To solve the expert-requirement of the UVC method, as well as a solution to the difficulty on censusing fish in coral reef environments due to the structural complexity of the habitat and the mobility, diversity, and high abundance of reef fishes (Russell *et a1*., 1978), video techniques have increasingly been used and perfected, as a complement or alternative. Like UVC, underwater video provides data on species composition and abundance, lengths and records habitat features (Murphy and Jenkins, 2010). In diver operated video (DOV) surveys, a diver swims at a constant pace and holds a video system which films a defined area, that may vary in size and shape (Alevizon and Brooks, 1975; Mallet and Pelletier, 2014). The major advantage of DOV surveys is that the footages can be stored indefinitely and can be repeatedly analyzed on computer for any verification, identification or to collect more information (Holmes *et al*., 2013; Bennet *et al*., 2016). In stereo-DOV surveys, by using two cameras set in a stereo configuration, the fish lengths can be precisely estimated from 3-dimensional images, when these images are cross-checked from ad hoc software (Harvey and Shortis, 1995; Mallet and Pelletier, 2014). It allows robust evaluation of size distributions and biomass from accurate estimates of fish lengths that are converted to biomass by length/weight relationships (Harvey *et al*., 2001a,b, 2002, 2004). There are also other modifications of video surveys, which can use a single or a stereo camera, such as remote underwater video (RUV) and baited remote underwater video (BRUV). RUV systems exhibit different designs and technical features and can be linked or autonomous (see Mallet and Pelletier, 2014). BRUV systems uses a single camera or a stereo configuration filming the area surrounding a bait bag (normally filled with pilchards, *Sardinops sp*.) used to attract fish (Mallet and Pelletier, 2014) for a deployment time of 50 to 60 minutes (Willis and Babcok, 2000; Watson *et al*., 2005). The using of a bait has shown to be useful in many studies (Willis *et al*., 2000, Watson *et al*., 2005), but the attraction to the bait depends on the biology and behavior of the species (Willis *et al*., 2000; Harvey *et al*., 2007; Stobart *et al,* 2007; Wraith, 2007), so only a specific part of the population is surveyed by using this method. A comparison between data collection using DOV, RUV and BRUV made by Watson *et al*., 2005, showed that DOV was the only one which recorded small, cryptic, cave-dwelling species, and that DOV and RUV recorded greater species richness and relative densities of the most common reef-associated species, whilst BRUV mainly recorded large predatory fish species. Overall, processing of videos is time-consuming and still requires a degree of observer expertise/training, cryptic species may be underestimate, some fish may not be measured if not entering completely the window or if the fish abundance in the shot is too much for the software to detect some individuals obscure by others, and also limitations imposed on the clarity of video imagery by current technological restrictions can result in fish not being observed (Holmes *et al*., 2013; Mallet and Pelletier, 2014; Harasti and Malcolm, 2013). Also, the heavy financial cost for the purchase/hire of the required hardware (e.g., digital video cameras, high quality computers, software, data storage) is also a factor that must be considered when designing the experiment to-be-followed, since MPA monitoring requires a long-term sampling to provide consistent trends of fish stocks.

Table 1.1. Summary of underwater visual census (UVC) techniques used in spatial management survey

Methodology	Target fish	Advantages	Disadvantages	Biases	Reference(s)
Belt transects	Herbivores, conspicuous, sedentary, territorial species	Requires minimal equipment that is cheap and easily accessible (pencil, slate and tape measure), is not disruptive to the habitat and it is not time consuming after the sampling. Generally less selective. Not affected by habitat variation.	Need trained observers in fish ID, which may be time consuming, and optimal conditions of high visibility. Limitations from SCUBA (depth and time). Surveys can be time consuming.Small fishes are poorly detected by the diver.	Errors of estimating size (observer error), errors of estimating abundance by overlooking when swimming fast, errors of estimating abundance by fish reaction to the presence of the diver	Brock (1954) Thresher and Gunn (1986) Edgar *et al*. (2004) Watson and Harvey (2007) Caldwell (2011) Lindfield *et al*. (2014)
Stationary Point Counts (SPC)	Mobile and sedentary fish species	High replication, need only one diver to conduct survey, faster than transects. Can detect small fishes. Unbiased among sites.	Need trained observers in fish ID and high visibility. Limitations from SCUBA (depth and time). Surveys can be time consuming.	Errors of estimating size (observer error) and density, influence of diver on fish behavior. Errors in area estimation.	Bohnsack and Bannerot (1986) Thresher and Gunn (1986). Edgar *et al*. (2004) MacNeil *et al*. (2008). Caldwell, 2011 Lindfield *et al*. (2014)
Timed Visual Assesstment	Larger species of fish (>20 cm) that are usually targeted by fisheries	This technique allows to cover large areas of reef, by using tools such as tow boards or unserwater propulsion devices.	Limitations from SCUBA (depth and time). Small and/or cryptic species are not detected.	Errors of estimating size (observer error) and density. Influence of diver on fish behavior. Underestimate cryptic species.	Caldwell, 2011 Bozec *et. al.* 2011 Lindfield *et al*. (2014)
Rapid Visual Technique (RVT)	Cryptic species	Obtains data on shy and cryptic species missed by othe UVC. Decreases observer bias. Less time-consuming.	Less replicable and less accurate. Does not provide a complete species list. It is inaccurate because the importance of each species is scored solely on the basis of order of encounter. Ignores the varying spatial distribution of different species. Surveys can be time consuming. Cannot be appliet to more of one habitat per survey.	Errors of estimating abundance between patchy species and widespread but yet relatively rare species.	Jones and Thompson (1978) Sanderson *et al.* (1980) DeMartini and Roberts (1982)

Experimental design to assess the MPA trend

The main objectives in conducting reef fish censuses are to: (i) compare fish populations between protected and non-protected areas to determine the MPA measures' effect, and (ii) quantitatively monitor reef fish composition and relative or absolute abundance over time (Bohsnack and Bannerot, 1986). Decision-making in the development of a marine research and monitoring programme aimed at detecting spatial and temporal changes in the abundance, diversity and biomass of fish communities in MPAs and its surroundings is based initially on the level of precision required to detect changes, the repeatability of the method, the environmental conditions under which assessments are to take place and, most importantly, the human and financial resources available for the collection and analysis of data (see Holmes *et al*., 2013). In addition, while surveying an MPA, destructive effects to habitats and species must be minimized till the minimum, and thus extractive and destructive techniques should be dismissed.

During the MPA research and monitoring survey method development, four ecological assessment designs can be contemplated: (i) samples taken only within the MPA before its establishment ("Impact only"), (ii) samples taken both within and outside the MPA after its establishment ("Control-Impact"), (iii) samples taken within the MPA before and after its establishment ("Before-After") and (iv) samples taken within and outside the MPA, before and after its establishment ("Before-After-Control-Impact", BACI). In the study of Russ (2002), it is suggested that a study on the effects of MPAs on fish communities should last over a time period of 5 to 20 years and include a BACI design, considered superior for detecting environmental impacts (Underwood, 1994), with replicate sampling units in replicate sites duplicated regionally. In addition, to achieve a broad understanding of the performance of the MPA, a complete study would factor out habitat effects, measure fishing mortality, conduct a capture-recapture study, and take large transects that transverse both controls and reserves taken. However, since coral reefs occur primarily in developing countries (Munro and Williams, 2006), and thus the MPAs that protect them, large-scale studies that consider many aspects of marine protection are not carried out due to limited resources. Also, all that information is usually not needed for the purpose of successful resource management at a local scale in an area dominated by artisanal fisheries. On the contrary, in these areas the information needed about the MPA performance is less academic and more about the subsistence of local fishers (Kamukuru *et al*., 2004).

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CHAPTER 2. SCIENTIFIC MANUSCRIPT

An effective method for assessing effects of no-take zones in reef fish species: Applied in a MPA in Tanzania.

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ABSTRACT

In tropical developing countries, reef monitoring programmes inside protected areas (MPAs) suffer from being unrealistically large and sophisticated. In addition, the lack of consensus among the scientific community on the experimental design to be used makes the standardisation of a method to compare results between different locations of the globe imperative. Our objectives were to describe an easy, affordable, and precise method for determining reef fish abundance and biomass as indicators of reef health and to evaluate the usefulness of this method to address questions of relevance for MPA management. On Pemba Island (Tanzania), within the no-take Kwanini Marine Protected Area (KMPA) and its buffer zones, diver operated video (DOV) samplings were conducted utilising a single camera along 50x5 cm belt transects. A list of reef fish species targeted by the local fishery was previously compiled, and then the video footage was played back on a computer to search for the selected species, count them, and classify them into pre-determined size categories. No experience with fish identification was needed throughout the sampling. From the numerical abundance data, average lengths, and species-specific length-weight relationships from literature, we were able to determine biomass (g/m2) results per species for different campaigns. The 2 years of data available were insufficient to detect significant changes but this approach provided valuable insights into the population variability of previously over-exploited species. The method is accurate enough to provide efficient monitoring information, if the sampling continues in the long-term, a relative status of not only the KMPA but other areas under similar restrictions, helping with future management decisions, encouraging more support, justifying investment and gaining a broader understanding of how MPAs in the western Indo-Pacific are helping coral reef habitats not only to recover from destructive fishing practises but also to cope with the effects of climate change in the near future.

KEYWORDS

Abundance, biomass, method, MPA, overfishing, conservation, coral reef, Tanzania, Pemba Island.

1. Introduction

Marine conservation programmes in MPAs located in developing countries suffer from being unrealistically large and complex, and impossible to be sustained indefinitely with locally available funds and human resources (Danida, 2000). For the purpose of successful resource management at a local scale in these areas, where fisheries are mainly artisanal, the information needed about the MPA performance is less academic and more about the subsistence of local fishers (Kamukuru *et al*., 2004). When collecting ecological data, it is crucial that the experimental design and the sampling technique chosen is capable of detecting change over an appropriate spatial and temporal scale (Batch *et al*., 2019; Holmes *et al.,* 2013). However, many programmes that aim to capture effects of marine reserves on the biota contained within them over time, collapse almost immediately when the donor funding ceases because they are designed at a cost-level that will never be sustained (Danielsen *et al*., 2003). In these cases, the strength of a simple and cost-effective method does not lie in detecting trends with a statistically acceptable degree of confidence (Danielsen *et al*., 2003). Rather, it lies in its proven effectiveness, even in the short-term, in strengthening local management of the resources and dealing with threats to biodiversity (ETFRN, 2002; Nordeco and DENR, 2002), when based on the reliability of comparisons between datasets by enhancing data standardization, which is ultimately defined by the absence of variation in the survey methodology and in the easiness of data collection.

The economic and human resources limitations on marine conservation are particularly acute at a time when there is an urgent need to increase the effectiveness of conservation efforts in the tropics (Achard *et al*., 2002). Increasing overexploitation of limited resources, habitat loss and degradation by destructive fishing practices, together with climate change effects, remain the biggest threats to coral reefs and its associated biota. In Indo-pacific developing countries, fishing is not subsidized and access to modern fishing gear is very limited due to limited stock and high unit prices (Katikiro *et al*., 2014). The fishers cannot usually access offshore areas due to their boats' constraints, so most fishing is carried out relatively close to shore and the nearby reefs are thus intensely fished (Grimsditch *et al*., 2016). Large-bodied predatory species are typically targeted by these fishers, which results in declines in target species size, density and biomass (Jennings and Kaiser, 1998; Russ, 2002), and which may further result in trophic cascades, whereby abundance of prey species increases, reducing the nature and quality of primary production (i.e., top-down effect) (Pinnegar *et al*., 2000, Dulvy *et al*., 2004; Mumby *et al*., 2006).

Beach seines, fishing using explosives and gill nets are typical of the destructive methods in these areas that cause significant damage to the coral reef structure and populations as it destroys important habitats for fish and other organisms, while at the same time, there is a long-term trend of overharvested fishery resources (Grimsditch *et al.*, 2016; Jiddawi and Öhman, 2002). Destructive fishing practices, especially dynamite fishing, reduce coral reef resilience over time, and lead to trajectories of ecological degradation that can have negative socio-economic implications as ecosystem services associated with coral reefs are reduced and eventually lost (e.g., tourism, carbonate budget, fisheries, shoreline protection, sand production, etc.) (Rogers *et al.*, 2015; Grimsditch *et al.*, 2016). Coral reefs are also highly susceptible to the effects of climate change; Scleractinian corals – the foundation species of tropical reef ecosystems – are living close to their thermal threshold (Hoegh-Guldberg, 1999). Such benthic disturbances alter fish assemblages that rely on corals for survival by affecting availability of food and shelter (Wilson *et al.*, 2006; Pratchett *et al.*, 2011), changing the population to fish groups that preferentially forage on dead coral surfaces (e.g., parrotfish, algal farming damselfish and detritivorous surgeonfish) (see McClure *et al.*, 2020). Loss of habitat, combined with overfishing, is therefore expected to have a severe impact on coral reef fish assemblages (Wilson *et al.*, 2010).

Natural recovery of the damaged reef is very slow or inexistent (Alcalá and Gómez, 1987). Against this issue, the implementation of a no-take marine protected area (MPA), where any kind of marine resource harvesting is banned, consistent with specialized scientific knowledge becomes an essential conservation tool to address unsustainable exploitation of marine resources (Gray, 2010; Raycraft, 2018), especially in developing island nations where animal protein is often sourced from coastal coral reefs (Cabral and Geronimo, 2018). MPAs have previously shown to increase the biomass and size of coral reef-associated fish (Roberts and Polunin, 1991; McClanahan *et al.*, 2006) and density of some target fish families (Samoilys *et al.*, 2007), whilst the net active movement of commercially exploitable individuals from MPAs to fished areas (usually defined as spillover; Rowley, 1994) is the process leading to the most direct and palpable local fisheries effects: the increase in yields close to the MPA boundaries (Abesamis and Russ, 2005; Di Lorenzo *et al.*, 2016). From a management perspective, conserving herbivores is clearly important for keeping reef seaweeds in check (Mumby *et al.*, 2007(a,b); McClanahan *et al.*, 2012), while commercially important predatory fish (e.g., groupers, snappers and emperors) not just keep population numbers in check but are good indicators of fishing pressure (Africa *et al.,* 2009). Moreover, many fish species have been regarded as allochthonous ecosystem engineers, including the parrotfishes (Scaridae), which significantly contributes to coral reef sedimentation (Rotjan and Lewis, 2005). In the case of goatfishes (Mullidae), they may offer significant ecosystem services too, such as resuspension and development of mixed species foraging associations, due to their extremely active foraging behavior and the vigorous stirring up of sediments by their barbels and mouth (See Uiblein, 2007).

The island of Pemba lies 50 km off the northern Tanzanian Coast in the Indian Ocean, and forms part of the Zanzibar Archipelago. Even not being in the center of coral diversity for the Western Indian Ocean, on the gradient of diversity the coral reefs of Pemba are among the 'second tier' of the most diverse coral reefs in East Africa (Grimsditch *et al.*, 2016). Moreover, the local human population relies on them heavily for food security and income from fishing, aquaculture and increasingly SCUBA diving tourism (Grimsditch *et al.*, 2016). Inlight of this, the Kwanini Foundation and Manta Resort, with the support of the local community and the government, turned 1 km of coastline into an MPA called the Kwanini Marine Protected Area (KMPA) in 2013 on Pemba Island, Tanzania. In 2018, the Bylaw designated the KMPA as a 'No take zone', where fishing, gathering, harvesting, collection and cultivation of marine species are permanently prohibited.

Ecosystem-based management of the fisheries requires information on the source (Jiddawi and Öhman, 2002) to integrate that information into decision making while deciding the MPA's boundaries and restrictions. Size-specific data with interpretations in units of biomass have been recommended for monitoring fish stocks because they represent the distribution of energy given the indeterminate nature of fish growth and size-specific variation of fish functioning (e.g., fecundity, predation threat) (Helfman *et al.*, 2009; Jennings and Dulvy, 2005; Shin *et al.*, 2005). The repeatability of the method for data collection, the environmental conditions under which assessments take place and, also, the human capital and financial resources available are the most important pieces of information to consider when designing a study (see Holmes *et al.*, 2013).

An easy, economic, precise, and non-destructive estimate of fish biomass has been a concern to marine biologists for a long time. The preference of underwater visual censuses (UVC) by the scientific community are based on several features (see Murphy and Jenkins, 2010). However, regarding Bortone and Kimmel (1991), virtually every researcher employing a UVC technique has used different sample design parameters with regard to the area, amount of sample time, and survey protocol, which ends up making effective comparisons between studies by different authors nearly impossible (Bortone *et al*.,1992). Here we describe a simple and cost-effective method for obtaining data on the numerical abundance and biomass of reef associated fish, evaluate it, and report statistical characteristics of spatial and temporal monitoring of species exploited by local fisheries that can also be used as indicators of of the health and integrity of the ecosystem (Uiblein, 2007). In the same line, we qualitatively evaluate advantages and disadvantages of the method compared to commonly used survey methods based on experience and available literature. Reef fish numerical abundance and biomass will be used as reliable indicators of fish and ecosystem integrity while evaluating the success of no-take measures. Moreover, the research will be actively contributing to the data that the Kwanini Foundation needs to collect for the Global Fund for Coral Reefs to justify their financial support, which highlight the importance this project may have in terms of Pemba Island's marine environment active conservation.

This study has the objectives: (i) to describe the DOV methodology developed to standardize the sampling criteria, (ii) to evaluate the utility of the sampling methodology and the statistical characteristics of the data collected, to answer scientifically based management questions, so that appropriate sampling strategies and analytical methods can be improved and used in future studies. The outputs from the statistical analysis will be used to answer two questions: (a) has there been any increase in numerical abundance and/or biomass since the no-take measures started within the KMPA? And (b) is one year after implementation of the no-take zone sufficient to detect the expected changes resulting from protection? The study provides a 'proof of concept' for a method which could be used to assess fish stocks trends inside the MPA over time and to evaluate the differences between the stocks inside the MPA and the non-protected surrounding areas, so-called buffer zones.

2. Material and Methods

The study did not involve any experiment or collection of animals.

Study area

The KMPA is located in the Indian Ocean, within Pemba North Region boundaries and 56 km off the coast of mainland Tanzania (Fig. 1). The KMPA covers an area of 450,000 m^2, of which approximately 320,000 m^2 is lagoon area comprising sandy bottoms, reef flat and soe small patches of seagrass meadows. The remaining 130,000 m^2 encompasses the fringing coral reef. While there are areas of the KMPA where the habitat condition is favorable, much of the fringing reefs, which start at around 5 m, have been heavily damaged due to destructive fishing practices and as a result the reef slope is very unstable and comprised primarily of coral rubble.

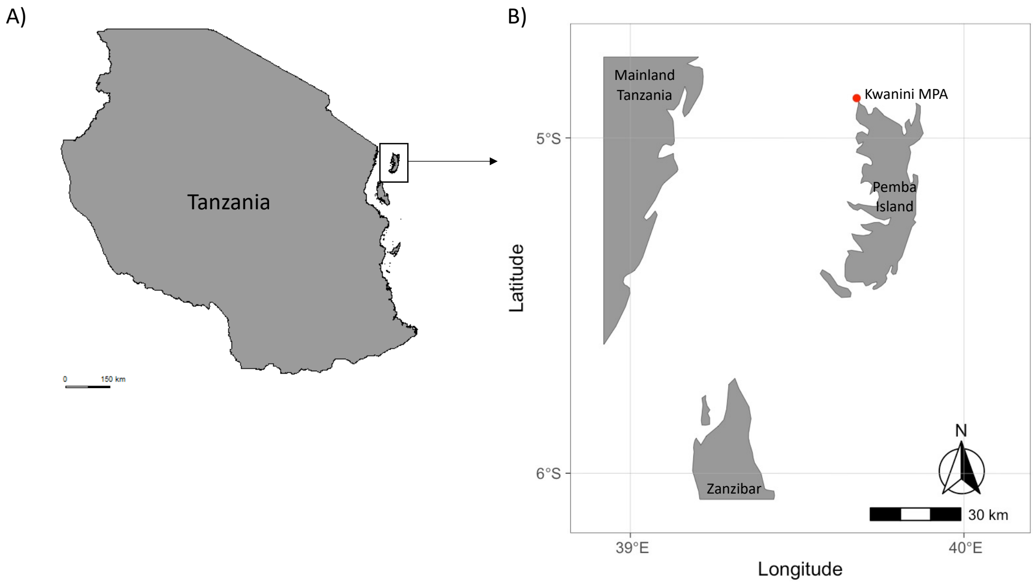

Figure 2.1. Map of the study region. A) Map of Tanzania (GADM, 2012), noting the position of Pemba Island in the rectangle. B) Pemba Island, marking the position of the Kwanini MPA with a red dot.

Underwater sampling design

To demonstrate the impact of MPAs on the density/biomass of target species, the study must include both spatial and temporal comparisons (Russ, 2002). Therefore, the underwater surveys were conducted in five different campaigns: July 2019, October 2019, January 2020, and November 2021, in both the interior of the KMPA and its surroundings, the so-called 'buffer zones'. The recordings from April 2022 were discarded because of their bad visibility. As non-destructive sampling is essential within the protected area, the equipment used for these surveys consists of using a single GoPro HERO7 Black, open-circuit SCUBA gear, a compass and a tape. The surveys were undertaken along ten consecutive transects, measuring a length of 50 metres with a width of 5 metres within the MPA and at two depth zones (Langlois *et al*., 2010; Holmes *et al*., 2013; Wartenberg and Booth, 2015; Andradi-Brown *et al*., 2016). For the buffer zones, the same length and width transects were completed, however due to the smaller size of these zones, less transects were done (four transects, two for the Southern part and other two for the Northern part of the buffer zone). A spacing of 10 m between each transect was maintained throughout to ensure that no fish from a previous transect were double counted. The total quarterly survey area for these fish surveys were 7,000 m^2. Of this, 5,000 m^2 were within the MPA at the deep and shallow sites with the remaining 2,000m^2 area consisting of buffer zone transects. Due to the nature of the coral reefs of Pemba Island, straight-line transects on a compass heading would cover too many depth differences and substrate variances, therefore surveys followed the natural depth contours of the reef, at the discretion of the lead diver.

The video surveys were completed at deep and shallow transects within the KMPA as well as deep and shallow transects in the buffer zones (North and South) stretching approximately 100 m outside of the KMPA. All surveys were undertaken between the hours of 7 am and 4 pm to reduce the impact of diurnal variations throughout the surveys (Watson *et al.,* 2010; Willis *et al*., 2006). Surveys were also completed during times where the visibility was deemed suitable and therefore at no less than 5 m (Tessier *et al.*, 2005), with the exception of the survey in November 2021 when the visibility during the survey was between 2 and 3 m. Deep transects were undertaken between 14 m and 16 m with shallow transects being conducted at between 6 m and 8 m. The DOV transects were swum at a consistent swimming pace and at a height of no more than 1 m above the substrate. The DOV camera faced directly ahead and avoided sudden sideway movements to include fishes that may not have been detected in the forward-facing field-of-view, as per Holmes *et al.* (2013).

A dive team of two, using a standard open-circuit (OC) SCUBA gear undertook these surveys, with the lead diver being responsible for the video filming and ensuring the surveys followed the depth contours of the reef. The second diver remained at the start point of the transect and via the use of a tape measure, ensured the survey distance was correct. This diver was also responsible for alerting the lead diver when the 50 m had been completed (or 10 m for the transect intervals). At the end of survey transects, and within the 10 m intervals, the camera stopped recording to ensure distinction between survey transects and other periods of the dive.

By allowing the diver with the video equipment to be the first to swim along the coral reef ensures that the footage captured is the most representative of reef conditions and avoids other divers and/or equipment disturbing the fish prior to filming. GPS coordinates were taken on the dive boat at the start and end of every survey dive to provide an approximate transect coverage area.

Fish species selection

The most efficient, time- and money-saving ecosystem monitoring and management can be accomplished by using groups of widely distributed, easily accessible species that partially combine the traits of indicator and key species (Nicholls, 2002). Once the randomly chosen transects were recorded and before the analysis of the footage, a predetermined list of target fish species was selected. The study region makes it imperative to select reef associated fish, thus the groups selected are regularly encountered on all reefs and are therefore considered to be key components of the ecosystem and of great socio-economic value for Pemba Island. The list is composed mainly by representative species of families which are important food species. The selected families are Lethrinidae (emperors), Lutjanidae (snappers), Mullidae (goatfish), Scaridae (parrotfish) and Serranidae (groupers). This list will be limited to one species per family since we aim to prove the method.

The results of presence/absence from surveys already done by the Kwanini Foundation crew in previous years, together with the catch rate information from literature (Jiddawi and Stanley, 1999; Jiddawi *et al*., 2002; Jiddawi and Öhman, 2002; Mgimwa *et al.,* 1999), and the relative economic (for food purposes only) and environmental importance, were used to define a list of diurnally active reef-associated species. The target species of each family chosen were the Thumbprint emperor *Lethrinus harak* (Forsskål, 1775), the Black and white snapper *Macolor niger* (Forsskål, 1775), the Long-barbel goatfish *Parupeneus macronemus* (Lacepède, 1801), the Daisy parrotfish *Chlorurus sordidus* (Forsskål, 1775) and the coral hind *Cephalopholis miniate* (Forsskål, 1775).

Data collection

Since the project aims to optimize the method for repetition, and because of the lack of stereo-video footages, the use of an expensive and non-easily accessible software was not possible. Instead of this, the footage recorded by the video operator were played back on a computer screen to capture the abundance of each of the five reef fish species at the same time each was classified into a size category.

For the consistency of the methodology, the total length (TL, cm) of the individuals sampled were estimated and classified in three species-specific size categories: (i) 'Sub-adult' (above 10 cm to TL at first maturity), (ii) 'Mature' (TL at first maturity to TL at which the species is commonly caught by fisheries) and (iii) 'Veteran' (over maximum TL recorded for the species) (Table 1). Only for the Daisy parrotfish *Chlorurus sordidus* and the Black-and-white snapper *Macolor niger* were found phenotypical differences on their life cycle that could give us a clue of their size when visualizing them in the videotape. Thus, for the Daisy parrotfish, the sub-adult shows dark-brown body color with two rows of white spots whilst the mature and veteran's body color is green, with green or yellow cheeks and caudal fin with pink submarginal bands. In the case of the Black-and-white snapper, the subadult is completely colored in black and white with 4 to 7 spots on the dorsal surface, whilst the mature and veteran's body is silvery grey strongly blotched with blackish overall color. In this specimen, also, the morphology changes with maturity. For the others, no differences on coloration nor in shape were found.

When in doubt on the size of those species which did not change their morphology throughout their life cycle or differentiate between the size categories "Adult" and "Veteran", reference fish species were used for size as they occur broadly in the samples collected. The proposed reference species are *Chromis dimidiata* (Max length 9 cm TL male/unsexed) (Lieske and Myers, 1994), *Chromis opercularis* (Max length 17 cm TL male/unsexed) (Allen and Erdmann, 2012), *Dascyllus trimaculatus* (Max length 14 cm TL male/unsexed) (Bacchet *et al*., 2006) and female *Pseudanthias squamipinnis* (Max length 7 cm TL female) (Lieske and Myers, 1994).

The mean length of each species-specific size category was used to calculate wet weight. Length estimates from each fish have been converted to weights using the equation: $W = a \cdot L^b$; Where 'L' represents the fish length in centimeters, 'W' the wet weight in grams and ‘a’ and ‘b’ published species-specific conversion constants from Fishbase (Froese and Pauly, 2022). When multiple length-weight relationships existed for a species, priority was given to metrics based on greatest number of replicates and geographical region closest to the study area (Clement *et al.,* 2012). The estimates of the weights were obtained by calculating the average weight of a fish from the midpoint of the fish size categories (‘L’), multiplying this average weight (‘W’) by the number of fish per size category censused and these values, when summed, the total weight of fish observed within each transect (50 m x 5 m), as per Brock (1954), which is the biomass per transect.

Table 2.1. Size categories in cm (TL) and species-specific weight-length relationship for the selected species to assess biomass.

Species	Family	Size category (TL, cm)			W-L Relationship	References
		Sub-adult	Mature	Veteran		
Lethrinus harak	Lethrinidae	10-25	25-50	>50	$W= 0.0195 \cdot L^{3.01}$	Kishimoto *et al*., 1987; Carpenter and Allen, 1989; Kulmiye *et al.*, 2002; Froese and Pauly (Eds.), 2022
Macolor niger	Lutjanidae	15-30	30-60	>60	$W= 0.0217 \cdot L^{2.97}$	Anderson, 1968; Longenecker *et al*., 2013; Kamikawa *et al*., 2015
Parupeneus macronemus	Mullidae	5-12	12-40	>40	$W= 0.0065 \cdot L^{3.27}$	Allen and Steene, 1988; Randall, 1997; Froese and Pauly (Eds.), 2022
Chlorurus sordidus	Scaridae	5-15	15-40	>40	$W= 0.0191 \cdot L^{3.09}$	Randall *et al*., 1990; McIlwain *et al*., 2009; Froese and Pauly (Eds.), 2022
Cephalopholis miniata	Serranidae	15-25	25-40	>40	$W= 0.1200 \cdot L^{3.22}$	Agembe *et al*., 2010; Allen and Erdmann, 2012; Froese and Pauly (Eds.), 2022

Statistical analysis

The dataset was first subset per species. The species *M.niger* was dismissed from the statistical analysis because of its low total number of individuals recorded (N=9). As the assumptions of ANOVA were not met by any of the group levels of the factors (“Campaign” and “Area”) influencing the quantitative variables (“Numerical abundance” and “Biomass”), the differences between means of samples were assessed using the non-parametric counterpart of ANOVA, the Kruskal-Wallis test, choosing for the whole analysis a 95% confidence level. The statistical package used was RStudio software version 2022.07.0. Significant differences between group levels were further investigated with pairwise Wilcoxon rank sum test using Bonferroni correction for adjusting p-values.

3. Results

The biomass data provided more accurate information than numerical abundance in terms of differences in means, proving the effectiveness of the method used. When analysing the video footage of each campaign, fish of all target species were identified, counted, and classified into size categories, thereby obtaining the species' numerical abundance per transect. When we were not sure of the identification, because of bad visibility or image quality, we did not count the individual. The most frequent species encountered throughout the experiment were the Daisy parrotfish, *Chlorurus sordidus* (N = 238), followed closely by the Longbarbel goatfish, *Parupeneus macronemus* (N = 207). Then, the numerical abundance and the average length of each size category were used to calculate the species biomass per transect (g/m^2), as described before.

Question a). Has there been any increase in numerical abundance and/or biomass since the no-take measures started within the KMPA?

Neither numerical abundance nor biomass data showed growing tendencies over the surveyed period (2 years) for any of the target species. Regarding the numerical abundance trend for the target species within the KMPA (Fig. 2.2), only *C. sordidus* showed differences close enough to the selected threshold of 0.05 for a 95% confidence level (Kruskal-Wallis, $P \leq 0.05$). The pair-wise comparison test revealed seasonal differences between the campaigns in October 2019 and January 2020, when the population decreased, and a less important but growing difference between July 2019 and January 2020.

In respect of the biomass (g/m^2) trend for the target species within the KMPA (Fig. 2.3), *C. sordidus* showed statistically significant differences between campaigns (Kruskal-Wallis, $P \leq 0.05$). The pair-wise comparison test revealed statistically significant differences ($P \leq 0.05$) between the campaigns of October 2019 and January 2020, the last with greater biomass, and between January 2020 and November 2021, this time with decreased biomass. *L.harak* showed differences close to the selected threshold between the campaigns in October 2019 and November 2021, decreasing, and January 2020 and November 2021, with the same effect.

Question b). Is one year after implementation of the no-take zone sufficient to detect the expected changes resulting from protection?

It was also analyzed the differences in biomass between the fish surveyed inside the KMPA ("control") and ones from the surrounding non-protected buffer zones ("impact"), a year later the no-take measures were implemented in the KMPA (July 2019). The results showed no differences between areas, so one year was not sufficient to detect the expected changes resulting from protection for either of the target species.

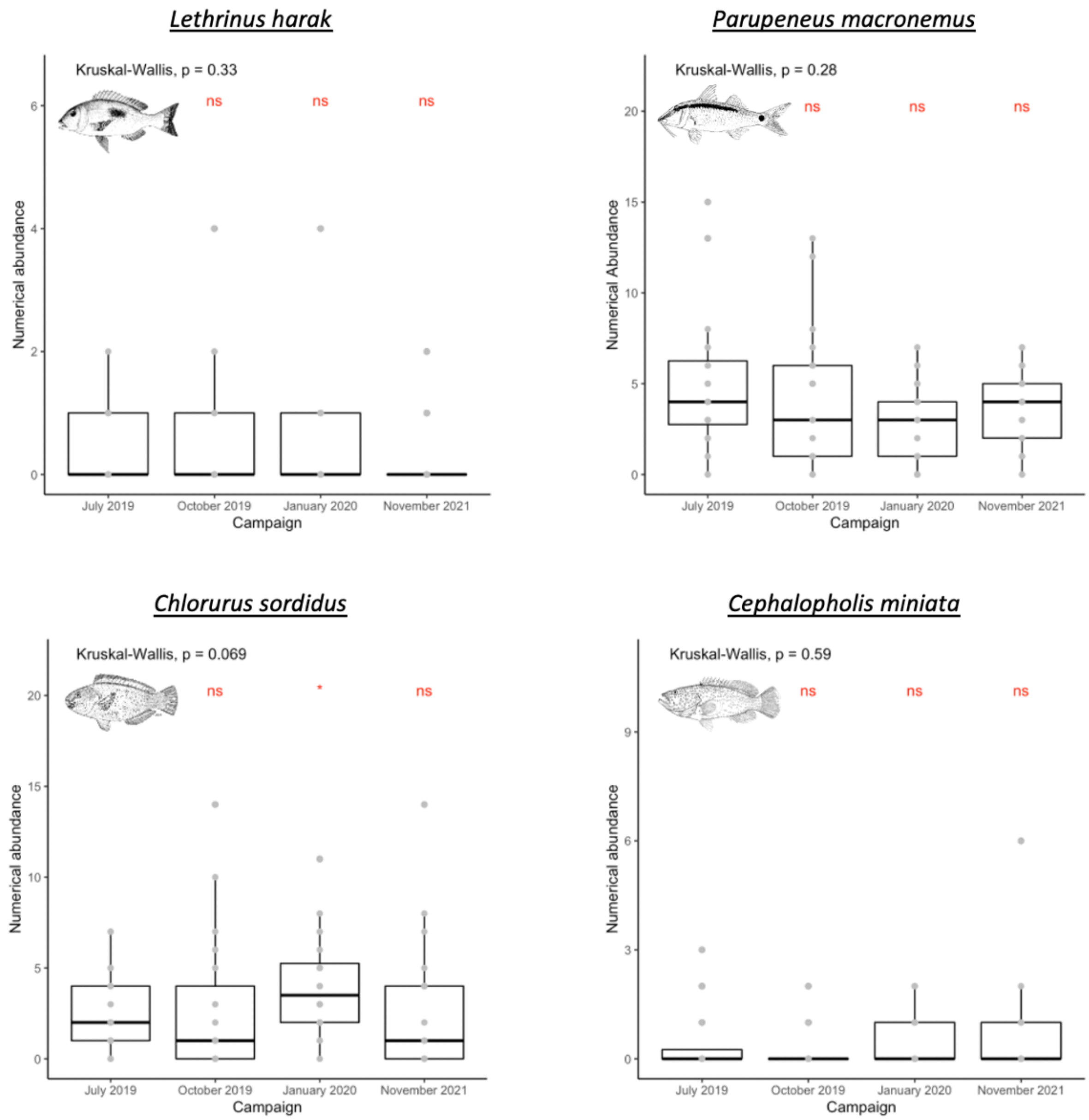

Figure 2.2. Numerical abundance trend of the species *Lethrinus harak* (N=40), *Parupeneus macronemus* (N=207), *Chlorurus sordidus* (N=238) and *Cephalopholis miniata* (N=38), over time under no-take measures inside de KMPA. Symbols in red indicate statistical significance of differences of means ('ns': P>0.05, '*': P≤0.05), being the first campaign, July 2019, the control.

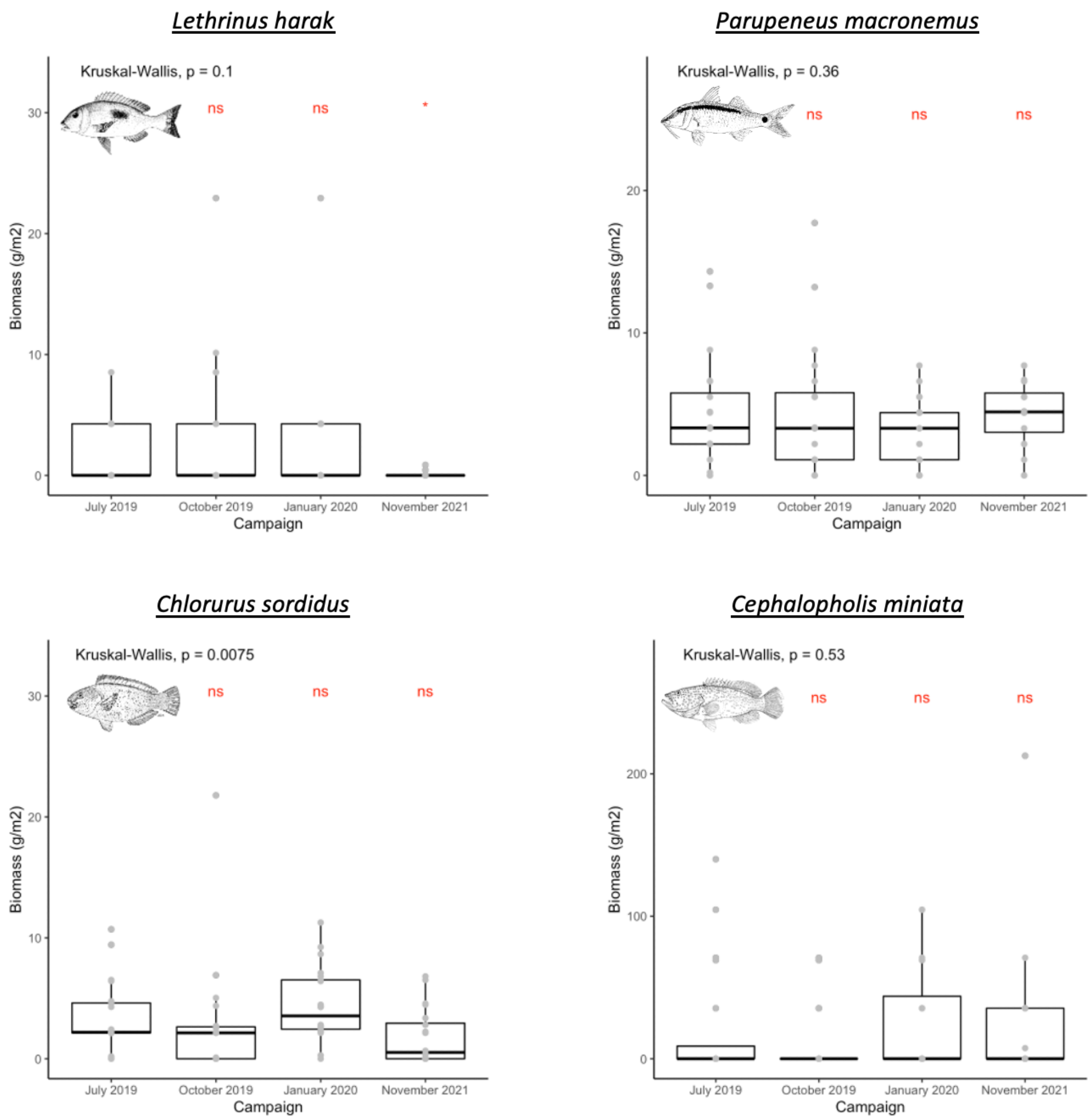

Figure 2.3. Biomass (g/m^2) trend of the species *Lethrinus harak, Parupeneus macronemus, Chlorurus sordidus* and *Cephalopholis miniata*, over time under no-take measures, inside de KMPA. Symbols in red indicate statistical significance of differences of means (‘ns’: *P*>0.05, ‘*’: *P*≤0.05), being the first campaign, July 2019, the control.

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4. Discussion

Standardizing fishery-independent methods based on *in situ* surveys to make reliable comparisons between fish stocks (i.e., abundance, size structure, and species composition) seems crucial at a time when fish populations linked to reefs are declining due to anthropological pressures and the effects of climate change on reef health. Efforts are being made around the globe to document the current status of reefs and their associated fish populations, and to identify the factors that influence their condition. However, the lack of consensus on methodology by reef fish ecologists (Sale, 1991), with different spatial and temporal scales, ultimately ends up making the results of these studies impossible to compare (see Caldwell, 2011). Agreeing with Caldwell (2011), the process of developing and adopting standardised methods should start on a regional basis. Here, we recommend the use of a standardised video technique not only in Tanzania but also in those tropical areas where financial resources and the scientists' availability are limited to easily assessing the increase or decrease in biomass inside and outside the area of interest.

The method here described, counted, and classified into size categories the targeted species. Moreover, it was shown to be effective and sensitive to identifying changes in some species' abundance and biomass in different seasons and years, but a reserve effect was not observed after 2 years, pointing to slow recovery rates of fish stocks. This lack of observation of statistically significant recovery after only 2 years is not unexpected. Due to the differences in their life histories from pelagic or temperate fish, coral reef fish are more vulnerable to overfishing (Russ, 1991; Roberts, 1995). They exhibit complicated sexual patterns and reproductive behaviors (Shapiro, 1987), narrow home ranges (Sale, 1978) and high levels of habitat selection (Sluka and Sullivan, 1996). Additionally, certain species have extended life spans, low rates of natural health, and late maturation (Manooch, 1987; Choat and Axe, 1996, Pears *et al.*, 2006). The species chosen for the characterization of the fishery targeted stocks within the KMPA were those that are known to be the most intensively fished in Pemba Island by artisanal fisheries, which had previously been overexploited and suffered from habitat degradation in the surveyed area before the KMPA was established. The removal of fish mortality of the target species by the closure of the KMPA in 2018 will in theory alter the size and age structure of the population with time (Getz and Haight, 1989), and this will be reflected in larger and more abundant adults of the previously exploited species (Roberts and Polunin, 1991; Wantiez, 1997; Russ and Alcala, 2011, McClanahan *et al*., 2006) and thus, in the biomass when the duration of protection increases.

After one year the KMPA was closed to fisheries, there were no changes between the protected area and the buffer zones surrounding it. Despite the fact that the recovery period was so brief, we did not anticipate it to work the other way around. However, findings on biomass' variability inside the KMPA over time provide an essential benchmark to assess ongoing status, both inside and outside the KMPA, to determine the response to protective management from both biomass trends and adult exportation (i.e. spill-over) to the surrounding areas. This last is important not just for the marine ecosystem but for the fishers community of Pemba Island. Moreover, according to Christie (2004), there is a substantial relationship between social and biological success, with social factors influencing long-term biological success. Especially in coastal communities located in developing countries like Tanzania, the export of fish biomass should, in the long term, at least compensate for the loss of fishing areas (Russ and Alcala, 1996a).

We did not apply the BACI design here, as suggested by Russ (2002), because surveys of the substrate over a 9 km extent of comparable coastline show that there are reefs in favorable and unfavorable condition both inside and outside the boundaries the KMPA. This indicates that the KMPA reefs were representative of the reefs within the vicinity and not in any better condition when the MPA was established. Furthermore, at least 30% of the KMPA reefs are in bad condition having been subjected to blast fishing in the past. Thus, even though we did not obtain differences one year after the implementation, we can expect reliable data in the future from future comparisons of both by a 'Control-Impact' long-term monitoring.

Even though not having obtained statistically significant results for the trend inside the KMPA, the present study demonstrated that the proposed method of estimating biomass from video footage can be used to assess the fish trends inside any MPA and compare results between protected and non-protected areas. Throughout the description and the evaluation of the method, some conclusions about its pros and cons can be highlighted. On the one hand, the sampling technique was proven simple, fast, and easy to replicate, whether repeated at a different time of the year or on a different type of substrate. The sampling strategy could be used in all tested habitats, from very complex reef formations to flat sandy areas, an issue found by many investigations since complex reefs' associated fish quantification remains difficult

(Gratwicke and Speight, 2005a) and requires long sampling times (Bohnsack and Bannerot, 1986) by UVC methods, while time is a limiting factor in this type of underwater survey. The sampling and analysis system provided a permanent record of the survey that could be viewed repeatedly to verify existing data and provide new information by combining information from multiple observers (Wilson *et al*., 2018), greatly improving data collection (diving time is used as efficiently as possible) and giving important conclusions to use afterwards in management decisions and justify funding. Also, the sampling did not need any experienced scientists involved, by selecting targeted species, there was no room for incorrect fish identifications, only trained divers with a relative control of their buoyancy to keep the sampling at the same depth. The belt transect technique modified in this study was relatively less invasive and more representative, as the diver went first with the camera and thus minimised the disturbance to the targeted fishes, as proved by the spotting of numerous groupers before hiding. Traditional belt transects often require a diver to deploy the tape before the survey begins, further disrupting the fish assemblages. This is frequently followed by two other divers recording the species, which in turn leads to more interruptions and can cause the second diver to miss his/her target species, especially if he/she is blocked by his/her dive partner. However, this would be an interesting topic to work on further surveys.

In addition, the method developed in this study covered a larger area of the reef in a single dive compared to stationary UVC methods, as the diver is only focused on filming and swimming, so no individuals within the study area were missed while writing on the slate or due to other possible distractions. Moreover, when analysing the video footage, the method offers the opportunity to focus on target species or the entire fish population and provides accurate fish counts and relative information on the size of the fish and the quality of the substrate. Fish specialists are not required to record the data, only people familiar with the method, and the person analysing the data does not need to be on site and, if unsure about ID, they can use books and online resources, with no need to rely on a quick underwater description or sketch. Finally, the method provides a substantial, permanent video archive that can be reviewed at a later date if needed by the same or a different study and can be independently verified by specialists if required. Arguably, standard techniques relying on in-water identification that cannot be independently verified potentially have errors because the level of accuracy in estimating individual fish size, identification down to species and ability to ignore distractions is directly dependent on the experience of the surveyor. This cannot be factored into any analysis.

In contrast, processing of videos at first was indeed time-consuming, as previously mentioned by Holmes *et al*. (2013), but it became quicker with experience. This last was the most time consuming part since the identification was relatively easy as we had only to look for 5 species. Experience from practise and the use of reference species made it progressively quicker. Even if the analysis of video footage still requires a degree of observer expertise (Holmes *et al*., 2013) or at least being in charge of the training and supervision, by using a single underwater camera, it provides a method that can easily be taught to non-scientific diving support staff and could also be adopted by a citizen science programme to monitor fish in a wide variety of different locations, even if they have no experience with fish taxonomy, making it truly cost-effective (Mallet and Pelletier, 2014).

With regard to the species chosen for this study, all but the snapper *Macolor niger*, which was only sometimes observed during the survey period and only in its sub-adult stage, were relatively abundant (especially *C. sordidus* and *P. macronemus*) and widely distributed, showing their tolerance of a wide range of environmental conditions, that react vehemently and visibly to specific natural or human-induced stimuli or alterations (Uiblein, 2007). Thus, the measuring of *L. harak*, *P. macronemus*, *C. sordidus* and *C. miniata* responses based on counts, weight and size, can be used as indicators for reefs ecosystems monitoring and management (Nicholls, 2002).

The statistical study revealed biomass and abundance variations for the Daisy parrotfish *C. chlorurus*, most of which were seasonal. The species' biomass showed a substantial annual increase in November 2021 as compared to January 2020. Since the species' abundance did not follow this outcome, a possible explanation for this drop in biomass is that there are fewer large adults but more small sub-adults, an indication that the stock is being replenished. Another factor that may be crucial to take into account in future studies of the KMPA status is the competition between the herbivorous *C. sordidus* and the grazer sea urchins, which, when faced with low predation as a result of the removal of large predatory fishes, reduce food sources below the level that parrotfishes can tolerate (McClanahan, 1992).

Top predators belonging to the families Serranidae, Lutjanidae, and Lethrinidae are, in general, the larger fish species and thus the most targeted by artisanal fishers (Munga *et al*., 2011). Groupers (Serranidae) and emperors (Lethrinidae) are of great importance to both recreational and artisanal fisheries (Carpenter and Allen, 1989; McClanahan *et al*., 2007), and their high susceptibility to overfishing and habitat loss is likely due to their biology and lifestyle (Hackradt *et al., 2014*). Their high site fidelity, moderate to high longevity, late age at maturity, protogynous sexual pattern, and low resilience make them one of the most vulnerable species on the reefs (Hilomen, 1997; Kramer and Chapman, 1999; Ebisawa, 2006; Sadovy *et al*., 2012). Neither the coral hind *C.miniata* nor the thumbprint emperor *L. harak*' data of abundance and biomass collected inside the KMPA differ between seasons or years, which may be explained by the species' small sample size (*C. miniata*, N = 38;. *L. harak*, N = 40), but also by their slow recovery due to their biological characteristics previously mentioned. It is worth keeping these species under control to see if the population is actually being restored under no-take measures as reported by other studies for the taxa (Galal *et al*., 2002; Starmer *et al*., 2008; Francis *et al*., 2015) and if, over time, the flux of adults through KMPA limits will benefit the community of Pemba, as opposed to ignoring this species for future studies like the decision made for *M. niger* (Lutjanidae).

Ideally, a study on the effects of MPAs on fish communities should last over a time period of 5 to 20 years and include a BACI design (Russ ,2002). Such ideal data availability is rare, however, and was not available here. Since we only have data from before the no-take measures were imposed within the MPA, a BACI design, where it is studied the samples taken within and outside the MPA, before and after its establishment, was not viable. It is important to continue monitoring and adding more data to increase the power to detect an effect of no-take measures within the MPA if it exists. This study is a contribution to start a database of information that will allow a future "Control-Impact" analyses after several years of data are accumulated in protected and non-protected areas.

The identification of sampling biases and their factorization into resulting estimates, provide margins of error in the conclusions, or at least give an insight in terms of the reliability of the conclusions obtained (Usseglio, 2015). To reinforce the conclusions exposed in this section of the report and also with the objective of improve the method, we looked into the factors that may have influenced the results obtained.

Fish behavior - The coral hind *Cephalopholis miniata* showed a clear tendency to avoid the diver, looking for shelter when spotting the bubbles, potentially generating underrepresentation. This has been also observed in other studies (e.g., Kulbicki, 1998; Feary *et al*., 2010; Lindfield *et al*., 2014) as avoidance behavior. In order to determine the extent of avoidance behavior, Lindfield *et al.* (2014) compared UVC findings from inside and outside MPAs using bubble-maker OC SCUBA and bubble-free closed-circuit rebreather (CCR). Fish surveys inside the MPA using CCR recorded equivalent community parameters to surveys using OC SCUBA. In contrast, outside the MPA, CCR recorded 48% more species and up to 260% greater fish abundance. Even so, CCRs are largely inaccessible due to their high operational cost, more demanding logistics (Usseglio, 2015) and need specialized divers to use them, which would not make it an option. In the present study, we minimize the error derived by this flight response by extending the sampling distance as the camera facing towards while recording gave more probability to detect the object of interest and to identify it and estimate its length afterwards when analyzing the records. Disagreeing with Samoilys and Carlos (1992), whose study concluded that cryptic Serranids' biomass could not be assessed accurately using UVC, our method proved, by the observation in several records of the *C. miniata* looking for shelter, that the Serranid could be detected before it hided and thus classify in a size category. On the contrary, some species tend to be double counted. *P. macronemus* and *C. sordidus*, potentially generated overrepresentation by passing within detection windows multiple times. Several studies have shown that some fish species have the tendency to be attracted to diver presence and noise (see Usseglio, 2015); the study of Chapman *et al*., (1974) showed this tendency and conclude that certain species correlated with time the association between the noise of the demand valve and the availability of food organisms. Little can be done to avoid this source of bias but try to identify the aggregations already counted and add it to the margins of error of the method.

Species' biology and lifestyle– Even though there were very few counts of *M. niger* sub-adults, there were none on the categories of adults and veterans. However, divers associated with this study did spot schools off the edge of the reef, towards the deeper areas of reef in the more built-up areas (nearer the northern parts of the MPA) or in the deeper parts of the reef outside the MPA, far from the areas surveyed. Compared to some of the deeper parts of the reef, the MPA and the fish surveys (7m/15m) were relatively shallow, so it could be that the juveniles might have used that area as a nursery/safe area and then moved towards the edges/deeper reefs as they matured. This spill-over effect would explain their presence in the video footage as juveniles and the difference on the habitat preference depending on the life cycle can explain the lack of information as well for the other two mentioned species. Further studies regarding the home range of the species chosen would be necessary to determine if they are, as believed, performing their vital functions inside the boundaries of the KMPA or if, on the contrary, at some point of their life stages they cross the borders. This data would help to understand further fluctuations in abundance and biomass inside and outside the no-take zone.

Underwater Visibility – The GoPro image quality was found to be enough to identify and count the species when the diver swam at a height of no more than 1 m above the substrate. However, when poor visibility (<5m) due to varying conditions on lighting (deeper areas surveyed) and water visibility in the presence of particles in the water column, fish detectability decreased dramatically. Records from deepest parts of the reef, and also due to oceanographic conditions in November 2021 showed that the method may not be accurate in not optimal conditions since some individuals could not be identified to the species level due to the impossibility of visualizing their color patterns or morphological characteristics when entering the window at more than 2 m from the camera. This bias, however, can be prevented by, for the deepest areas, using an underwater torch, and for the days when water turbidity is not ideal for sampling simply procrastinating the survey when bad conditions are found in the field the day planned. This last is a possibility due to the easiness of the deployment of the survey.

Seasonality – Even though tropical marine ecosystems are in general characterized by minor seasonality or even aseasonality, ignoring the existence of any seasonal change in East Africa may lead to a source of error. In this study, the surveys were done indistinctly because of lack of personnel to run the surveys, having campaigns of different years in different seasons that may make not statistically valid to compare the results on species abundance and biomass. In fact, East Africa areas experience great seasonality, and it is greater distinguished between wet and dry seasons that in many other tropical areas. The study of McClanahan (1988) showed that seasonality in South Kenya and Tanzania is a major factor affecting biological processes on a yearly basis, which include (i) decreased density and activity due to a deeper thermocline and cooler waters in the southeast monsoon (March to October) (Morgans, 1962) and (ii) increased reproduction due to high water temperatures associated with the northeast monsoon (October to March) for demersal fish (see McClanahan, 1988), among others. This may be the explanation for why the black and white snapper *Macolor niger* was found in some campaigns and was missing in others, even though only sub-adults of this species were found. Thus, even this study proved that important data on the characterization of reef associated fish can be calculated by using the developed method, a specific season should be selected to have reliable data on the trend the MPA is following.

Distance – Even not directly observed during the data analysis, distance to fishes is a common bias on transactional observational methods that must be considered. Choat and Bellwood (1985) noted that 5 m distance in UVCs underestimated the abundance and tended to include larger fish in the transect, which in fact were outside it. Underestimating the distance results in larger areas being surveyed and thus, more fish being counted (Harvey *et al*., 2004). Since it is a bias related to the observer, it can be extrapolated to our study.

Some biases associated with the method can be reduced through changes in species targeting (i.e., poor representation of the species), available technology (i.e., using a torch) and picking the optimum survey conditions (i.e., bad visibility), but other biases cannot (i.e., fish behavior to the diver, distance). However, if these last biases are considered consistent throughout the data collection, then it is possible to compare datasets (Buckland *et al*., 1993). Then, we can conclude that our method indicated relative rather than absolute differences between campaigns and sites, but even so the outcome from the analysis of the data collected is accurate enough to provide a relative state-of-the-community and create a basis for determining the usefulness of the KMPA, justify funds, and to support future management decisions.

5. Conclusions

The method developed and tested in this study proved to be simple, fast, easy to use, reliable and repeatable for assessing trends in reef fish populations under protected measures, even with different levels of substrate complexity. It provides a standardized means of comparing reef fish communities and reduces some of the biases of traditional visual sampling methods. It also provides quantitative data on abundance, length and biomass. Reef fish assemblages are quickly recorded, and many samples can be easily obtained for statistical analysis. Although we have recorded certain commercially significant species, the method can be modified to obtain information of all observable taxa and even establish the quality of the substrate.

The method was efficient to compare data between years and sites, but the data available after 2 years of monitoring was insufficient to demonstrate statistically significant effects of the management protection, which is not unexpected given the life cycles and generation times of the species involved. This study is a first step in what is expected to become a long-term monitoring plan which can be used to conduct further tests and analyses of trends after many years of data to assess the efficiency of the management.

Despite its great advantages, the method showed some biases related to visual surveys, such as fish behavior and visibility. This method does not aim to provide the most accurate data, but to find differences between protected and non-protected areas over time to help organizations with limited resources make management decisions. The description, evaluation and interpretation of the statistical characteristics of the data collected made throughout this study provide the basis of a new method to-be-used for further monitoring and assessment programmes in the KMPA.

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