

**BASELINE ASSESSMENT OF BENTHIC INVERTEBRATE COMMUNITIES IN
THE SOUTHERN BENGUELA: INSIGHTS FROM TABLE MOUNTAIN
NATIONAL PARK MPA, ROBBEN ISLAND MPA, AND THE SEAS OF GOOD
HOPE EBSA.**

BY
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DECLARATION

I, Laureenne Snyders, declare that the contents of this dissertation represent my own unaided work, and that the dissertation has not previously been submitted for academic examination towards any qualification. Furthermore, it represents my own opinions and not necessarily those of the Cape Peninsula University of Technology.



L. Snyders

Signed

02 October 2024

Date

ABSTRACT

Areas afforded protection in the ocean space, for example, Marine Protected Areas (MPAs), assist in managing the marine environment by promoting fisheries sustainability, ensuring that marine ecosystems function effectively, and serve to protect the range of species living there. For these areas, baseline information is key in order to determine the range of organisms inhabiting these spaces. The aim of this thesis was to gain a better understanding of the benthic macro-invertebrate fauna of the southern Benguela ecoregion in South Africa, with a focus on the Table Mountain National Park Marine Protected Area (TMNP MPA), Robben Island Marine Protected Area (RI MPA) and the Seas of Good Hope Ecologically or Biologically Significant Area (EBSA). A comprehensive benthic survey was conducted from August to September 2019 to provide baseline information of the macro-benthic invertebrate fauna, including their diversity, assemblage patterns and the environmental variables that influence them. This study focussed on the visual assessments using two different camera systems: a simple Drop camera and a towed camera (Ski-Monkey).

A total of 48 stations were sampled and imagery software was used to analyse representative images per station. The habitat characteristics and species abundance of both hard and soft bottom substrata were recorded. Diversity indices selected for analyses and interpretation of data included species richness (S), abundance (N), Pielou's evenness index (J'), which measures diversity along with species richness and the Shannon-Weiner diversity index (H'), measuring the diversity of species in a community.

The species accumulation curves for the study area did not reach an asymptote, indicating that further sampling could reveal additional species and enhance the overall understanding of the area's biodiversity. Despite the latter, we have used the best available data from the survey to provide a better understanding and provide baseline information for the sampling areas. These data, while not exhaustive, offers valuable insights into macro-benthic invertebrate communities and serves as an important foundation for future research and conservation efforts in the southern Benguela ecoregion.

The drop camera sampled a higher mean percentage cover of benthos and rock, and a lower percentage of sand compared to the ski-monkey. The significantly higher percentage cover of sand in the Ski-Monkey samples is expected since a towed camera system is better suited for use on soft sediment. The mean species diversity from images collected by the drop camera

(i.e. richness, abundance, percentage cover, evenness), compared to that collected by the Ski-Monkey were found to be significantly different ($P < 0.05$) with the Ski-Monkey being the higher of the two. Large variations in diversity were found between the samples. Community structure was analysed and differences in community structure were found to be related to depth and substrate type.

Findings from this study has provided an important overview of the region's biodiversity, revealing a diverse range of species, shedding insight on diversity and community structure and providing a strong foundation on which to build future research efforts. Significant differences were found in species richness, diversity, and abundance between camera systems. When comparing sampling effort between the two camera systems, images collected by the Ski-Monkey captured a higher species richness, diversity and abundance compared to the Drop camera system. The study showed that the southern Benguela ecosystem is heterogeneous and the distribution of organisms varied spatially according to substrate type, location and depth. These results are important for MPA conservation and management, as well as for understanding benthic invertebrate diversity and abundance in the region. The study provides valuable baseline information for future research and highlights the need for continuous monitoring of the protected areas in the southern Benguela ecoregion. The high concentration of endemism and restricted species in the South-Western Cape is further highlighted by the lack of alien species in the research area, emphasizing the significance of protecting these ecosystems against invasive species. Baseline data on the benthic invertebrate fauna, related habitats, and environmental factors affecting their distribution and abundance in the southern Benguela ecoregion have been supplied by this thesis. The two environmental factors that have been found to best explain the observed community patterns are substrate type and depth. The adoption of long-term monitoring systems and the application of standardized efforts to guarantee the efficacy of conservation initiatives are among the recommendations for future research. Overall, this study makes a substantial contribution to our knowledge of the benthic ecosystems in the southern Benguela ecoregion and offers insightful data for the preservation and management of the marine biodiversity in the area. The findings of this research will also help shape decisions on marine spatial planning, which will support South Africa's efforts to preserve and manage its marine environments sustainably.

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BENTHIC INVERTEBRATE FAUNA, ASSOCIATED HABITATS AND THE ENVIRONMENTAL VARIABLES INFLUENCING THEIR DISTRIBUTION AND ABUNDANCE IN THE SOUTHERN BENGUELA ECOREGION.

CHAPTER 1: INTRODUCTION

1.1 Benthic ecosystems and species distribution

The ocean comprises 70% of the earth's surface, therefore despite numerous studies on the ocean environment, much remains to be discovered about the organisms that inhabit it (Widdicombe and Somerfield 2012). The ocean is a multi-dimensional environment, with a range of physical and biological features, including currents, temperature, light and substrate types which impact community structure and diversity (Valentine 2009). Biological features/interactions such as predation, trophic structure and symbiotic relationships are also factors that impact community structure and diversity within an environment and it is the combination of these physical and biological factors which drive diversity and patterns of biodiversity within the ocean (Miller 1999). Knowledge gaps, whether globally or locally, may be one of the biggest risks to marine biodiversity (Gibbons et al. 1999). Globally, it appears that marine habitat sampling is quite patchy, with significant gaps in both sampling effort and taxonomy (Costello et al. 2010). Making decisions on how valuable a site is to protect is made more difficult by these knowledge gaps. Gibbons (1999) suggests that the abundance of marine species in South Africa is relatively high. It is estimated that approximately 31% of the marine species found in South Africa are endemic species (Gibbons et al. 1999; Awad 2004; Griffiths et al. 2010). Furthermore, it is believed that South Africa has a greater understanding of marine fauna than other developing countries (Griffiths et al. 2010). Despite this, a significant portion of the ocean remains unexplored due to sampling restrictions such as the duration of time at sea and the expense of hiring research vessels and personnel. The current knowledge gaps which exist are of particular concern as ongoing and increasing anthropogenic activities affect species abundance (Costello et al. 2010).

Marine benthic ecosystems which consist of organisms living in association with a substratum on the sea floor (Thain and Hickman 2000; Ruiz et al. 2009), have received limited attention in comparison to other marine systems, despite their crucial role in the functioning of the

marine environment. Benthic invertebrates are described as animals without backbones that live at the bottom of the ocean or in silt. They are classified as either infauna because they dwell in sedimentary deposits, or as epifauna, because they reside on the surface of bedforms like rock, coral or sediment. Examples of epifauna are sponges, sea stars, hydroids, corals, crabs, bryozoans, and gastropods. These organisms belong to many feeding groups, including predators and filter feeders (Simpkins et al. 2003).

The health and maintenance of marine ecosystems are dependent on the diversity of benthic communities, which contributes to greater ecosystem services such as oxygenation of sediments and nutrient cycling (Snelgrove 1999; Boudreau and Worm 2012). Some benthic organisms, like large sponges, are habitat forming and therefore play a key role in structuring their environment and increasing diversity. By serving as an important food source for other ocean dwellers (e.g. fish), benthic organisms also increase the complexity of the food-webs of the environments they inhabit (Simpkins et al. 2003; Ross and Quattrini 2007; Buhl Mortensen et al. 2010; Smith et al. 2010).

Benthic marine ecosystems are particularly sensitive to anthropogenic stressors, like overfishing, pollution and habitat destruction as a result of mining. Mining activities destroy natural features and abolish functional habitats, displacing organisms from their habitats and disrupting processes. For example, silt plumes can result in the loss of benthic processes and the benthos which they support (Carriero-Silva et al. 2022). Because effects have not been thoroughly documented, the possible significance of alterations in benthic populations brought on by habitat disruption by commercial fishing is frequently downplayed (Hall 1994; Thrush et al. 1998) but certainly not undocumented (Collie et al. 2000; de Boer et al. 2001; Kaiser et al. 2003; Atkinson et al. 2011; Sink et al. 2012). Clark et al. 2016 concluded that in addition to harvesting target species, deep sea fishing may potentially unintentionally affect the environment, mostly through the deployment of heavy bottom trawls and to a lesser extent, bottom long lines. The majority of the benthic fauna is frequently removed by bottom trawling over hard sea beds. This leads to decreases in species richness, cover and abundance. Comprehending how fishing affects habitats is essential to implementing ecosystem based monitoring which takes into consideration the impacts on all parts of a marine ecosystem which experiences exploitation (Williams et al. 2011). These stressors emphasise the importance of understanding the ecology, diversity and functioning of benthic ecosystems so as to conserve and manage benthic invertebrate species and their habitats.

A variety of factors drive diversity and distribution in benthic ecosystems, such as abiotic environmental variables which include temperature, substrate type, depth and biotic variables which include productivity, predation and anthropogenic activities (Levin et al. 2001; Thrush et al. 2006; Glover et al. 2010). Benthic habitats are made up of various substrate types, namely sand, rock, reef, mud and/or a combination thereof that influence the diversity of organism's resident to a certain area (Thayer 1974; Smith et al. 2010). Many sessile epifaunal organisms such as mussels, sponges and sea anemones, require hard substrate attachment points while burrowing organisms such as worms, sand crabs and sea pens inhabit soft bottom subtidal environments (Atkinson et al. 2011). Studies have revealed dominant phyla for soft bottom subtidal habitats to be Polychaeta, Nemertea, Arthropoda and Mollusca (Thayer 1974; Commito and Ambrose 1985; Atkinson et al. 2011). In both Canada and the United States of America, Ruiz et al. (2009) found a greater number of species (sessile and mobile combined) on hard substrata versus a combination of hard and soft substrata. Hard substrata act as shelter, feeding, settling and nursery grounds, however, the number of mobile species in soft substrata was found to be higher than that found in hard substrate. In soft bottom benthic environments, silt-sized particles are often re-suspended by burrowing activities, resulting in less favourable living conditions for suspension feeders because suspended particles may clog feeding and respiratory structures of benthic organisms (Thayer 1974). Specifically in the southern Benguela, epifaunal assemblages showed greater differences in sand dominated areas with significantly reduced species diversity (Wozniak 2023). Furthermore, several epifaunal species were absent from these sand dominated areas compared to communities inhabiting hard substrate, highlighting the impact benthic substrata has on community composition (Atkinson 2009).

Depth variations play a leading role in both the diversity of marine communities as well as the structure thereof (Gray 2001; Celliers et al. 2007; Blamey and Branch 2012). Depth influences a variety of factors such as temperature, light penetration, nutrient availability, sedimentation and water movement (Bigelow and Maunder 2007; Gotz et al. 2009; Webb et al. 2010; Roberson et al. 2015). Changes in species composition and diversity are commonly linked to the strong bathymetric gradients such as temperature, pressure and food availability. The depth range in which benthic organisms are found is linked to their tolerance levels of specific environmental gradients (Clearly et al. 2005; Costello 2009; Rahman et al. 2009). In South Africa, the distribution of marine organisms along the coastline is also influenced by the diverse oceanic patterns present in the region (Roux et al. 2013). These include the dynamic

current system composed of the Agulhas Current along the east coast, which moves southwards transporting warm temperate water, and the Benguela current along the west coast, which moves northward and transports cooler water (Roux et al. 2013). The Benguela current is known to be synonymous with upwelling, which is regulated by weather systems and occurs as a result of the replacement of surface waters with cooler, nutrient-rich deeper water (Nelson and Hutchings 1983; Shannon and Nelson 1996). The South African west coast is characterized by high productivity and low benthic species diversity as a result of upwelling (Shannon 1985), in contrast to the east coast, which displays low productivity and high species richness (Button et al. 2021). Upwelling is defined as the process where cold offshore winds cause cold nutrient rich waters to rise and take the place of warm nutrient poor waters (Nelson and Hutchings 1983; Shannon 1985; Shannon and Nelson 1996; Hancke 2023). Due to a combination of oceanic and atmospheric processes, the south coast of South Africa also experiences upwelling (Goschen et al. 2012). This largely due to the strong, deep and warm Agulhas current flowing along the east coast of South Africa. One of the most powerful currents in the Southern Hemisphere (Nkwinkwa 2018). The Agulhas current is of great importance to South Africa as it influences rainfall and climate in southern Africa, contributes to global ocean circulation and creates an environment for local fisheries (Nkwinkwa 2018; Hancke 2023). Currents play an important role in distributing heat energy, regulating the weather and climate and cycling vital gases and nutrients which in turn contributes to benthic distribution patterns and species diversity/richness (Shannon 1985). Previous studies on benthos have focused mainly on commercially important species (Clark et al. 1996; Brill & Raemaekers 2013), leaving a gap in our understanding of the whole system, including sessile and semi-motile benthic organisms such as sponges, corals and echinoderms. These organisms play a crucial role in providing habitats and food resources for a diversity of marine life, and understanding their endemism and distribution patterns is crucial for effective conservation. In order to sustainably manage marine ecosystems, it is essential to have a thorough understanding of their biodiversity and distribution, as emphasized by Webb et al. (2010). According to Griffiths et al. (2010), the South African coastal zone is well documented, with a large number of samples being collected via dredge, grab or trawl. However, a majority of these samples were collected in the mesopelagic zone (0 – 1000 m), with 83 % being from the epipelagic zone (0 – 100 m), highlighting the need for infrastructure capable of sampling at greater depths (Griffiths et al. 2010).

There are several methods for gathering benthic samples: (i) Physical specimen collections include beach casting, intertidal collections, snorkelling, SCUBA diving (up to a depth of 30 m), dredging, trawling and grabs, and (ii) Visual assessments via in-situ images and video footage can be obtained from small boats and ships using remote underwater camera systems such as Remotely Operated Vehicles (ROV), Drop and towed camera systems.. Remote underwater camera systems have allowed us to explore the marine environment across a range of gradients. These tools are particularly valuable in high priority conservation areas such as MPAs (Cappo et al. 2003; Samaai et al. 2010; Mallet and Pelletier 2014), as they allow for continuous long-term monitoring and baseline information surveys for lesser-known areas without causing damage to the ecosystem (Jones et al. 2009). Additionally, these tools aid in the classification of habitats, which is essential for the mapping and comparative analysis of ecological data (Costello 2009). By using these technological advances we do not only contribute to the existing knowledge of site specific areas, but are more importantly able to identify current gaps in our knowledge of these areas (Webb et al. 2010).

Due to the amount of time, budget needed for research vessels and the costs surrounding the purchase of specialised equipment required to sample the depths at which certain communities occur, it is often difficult to determine which organisms exist in these environments and their community structure (Snellgrove 1999; Solana et al. 2003). Samples obtained from grabs, trawls and dredges are traditionally the methods used for benthic sampling. However, because these organisms are not viewed/examined in their natural habitat, these methods do not allow one to explore the underlying assemblage structure present in benthic environments (Solana et al. 2003). The use of underwater imagery can be described as ideal as it bridges the gap for many of the challenges involved in sampling benthic communities. It reduces time spent sampling in the field and avoids problems such as mixing of specimens from different sites or bottom types and in doing so, further improves sampling efficiency. Physical specimen sampling is notably more efficient on soft compared to hard substrata, but these methods are known to be destructive especially for large scale sampling and is particularly problematic in sensitive habitats (Strong et al. 2006; Sheehan et al. 2010; Williams et al. 2015). Bohnsack (1979) concluded that underwater photographic sampling in comparison to studies comparing conventional sampling techniques was best suited for sampling hard bottom communities and epifauna. Photographic sampling allows for quantitative information to be collected along with a good visual record of the study site. Conventional techniques are described as being less accurate and less precise by Foster et al. (1991) and according to Edmunds et al. (1998), is also

more time consuming in the field. Alternative non-destructive sampling techniques make it possible for patterns and characteristic relationships between benthic biota and their physical habitat to be detected and observed. The use of underwater imagery has therefore been considered ideal as it effectively allows us to develop insight into community structure (Solana et al. 2003; Williams et al. 2015) in a non-destructive manner which is particularly important in protected areas such as MPAs. Weaknesses in the method do exist since one might not always be able to identify the species visually, resulting in relying on good reference image databases which might not always be present. Furthermore the method relies on clean water to get good images but these weaknesses are common to many methods so these visual techs remain preferred.

1.2 Marine Protected Area's (MPAs)

Marine biodiversity is a significant contributor to the provision of essential ocean services like food security, recreation, pollution recycling, prevention of coastal erosion, climate management, and the production of building materials and medicines (Thorne-Miller 1999). Globally, factors that directly threaten marine biodiversity are overfishing, pollution, alien species, climate extremes, acidification, hypoxia, habitat degradation, species exploitation, and coastal run-off (Bohnsack and Ault 1996; Thorne-Miller 1999, National Research Council (NRC) 2001, Costello et al. 2010). Additionally, indirect effects of climate change, such as habitat loss owing to increasing sea levels, is also a threat (Widdicombe and Somerfield 2012). The aforementioned dangers also pose a threat to the marine biodiversity of South Africa where the major threats are pollution, overfishing, hypoxia, and alien species (Lombard et al. 2004; Costello et al. 2010). Changes in marine biodiversity, particularly through warming and climate variability, can be utilized as worldwide markers of climate change (Worm and Lotze 2009) and Marine Protected Areas (MPAs), with minimal anthropogenic stressors are often known as natural laboratories by which the effects of climate change on ecosystems/species can be explored.

In order to maintain ecosystem functioning in an undisturbed state and to reduce the environment's exposure to high levels of anthropogenic stressors, Marine Protected Areas (MPAs) provide a more regulated method of applying protection. Agardy et al. (2011) claim that MPA's are unquestionably the most effective tools available for fighting habitat degradation and over-exploitation. Marine Protected Areas also enable populations of

exploited invertebrate populations to recover (Attwood et al. 1997). In order for organisms to complete different stages of their life cycles in various ecosystems, the ocean's ability to act as a solvent guarantees the movement of biota and dissolved minerals between ecosystems (Kingsford and Battershill 1998; Thorne-Miller 2009). Therefore, one of the more significant advantages of species protection within MPAs is the potential for productivity of both fauna and flora, adult and juvenile, to spill over across protected area boundaries (Agardy 1994; Kerwath et al. 2013). The selection criteria and management approaches of MPAs vary greatly so that these areas can adequately meet the needs of fisheries and ecosystems (Turpie 2000). If the relationships between communities, habitats, and protected species are not completely understood, the creation of a MPA to protect representative biodiversity and the habitats of particular species may not be sufficient to protect other species as well as the habitats in which they live. When MPAs are managed separately from other areas, the integrity of their ecosystems become vulnerable to outside exploitation (Turpie et al. 2009). The use of MPAs to combat biodiversity loss is only one component of the solution and has its drawbacks (Agardy et al. 2011). Many MPAs are inadequately planned and managed, the area may fail due to degradation in the surrounding areas, and many MPAs appear to be offering protection when in reality they are not (Agardy et al. 2011). These drawbacks are frequently due to their ecologically insufficient design. Displacement of fishing activities, which increases pressure on the environment around a MPA is a common drawback (Agardy et al. 2011), particularly those that contain large no-take areas. Many offshore areas are still unprotected, while many MPAs have failed as a result of inadequate planning, a lack of local support, non-compliance and ongoing effects outside of MPA borders (Agardy et al. 2011). Declared MPAs must be regularly monitored and evaluated in order to make sure goals are being met and to allow for error correction. Costs related to economy and aesthetics could be negatively impacted by changes in ecosystem processes. Because ecosystems are dynamic, informed ecosystem management is crucial (Boero and Bonsdorff 2007). Rare species can become abundant in marine ecosystems, and abundant species can become rare. Species are always migrating, either permanently or temporarily between ecosystems (Boero and Bonsdorff 2007). Therefore, by enabling studies of the ecosystems inside MPAs and comparing results to its surrounding areas, MPAs could be a significant research tool in the assessment and monitoring of ecosystems.

In South Africa, MPAs have been established to manage marine resources effectively (Lombard et al. 2004). These MPAs were originally established established in accordance with

the Marine Living Resources Act (MLRA) (Act 18 of 1998). Their purpose is to ensure the preservation of marine biodiversity, habitats and ecosystems, as well as to enhance fisheries management and minimise user-conflicts. Several of South Africa's coastal MPAs were established through Section 43 of the MLRA. However, on May 16, 2014, the Marine Living Resources Amendment Act 5 of 2014 was enacted, repealing Section 43 of the MLRA on June 2, 2014. As a result, all MPAs declared under the MLRA in South Africa are now governed by the National Environmental Management: Protected Areas Act 57 of 2003 (NEM: PAA 57 of 2003). In addition, the management responsibility for MPAs was amended and transferred to the NEM: PAA through the Amendment Act 21 of 2014. Currently, MPAs in South Africa are declared and managed under the National Environmental Management: Protected Areas Act (NEM: PAA) 57 of 2003 (Kirkman et al. 2021).

Many South Africans' socioeconomic well-being is reliant on the marine biodiversity of the region, either directly or indirectly (Tunley 2009). Because overexploitation of South Africa's marine resources is one of the main threats to the region's marine biodiversity, MPAs were therefore required to help limit it (Lombard et al. 2004). The management of MPAs is not without flaws, despite reports that South Africa's coastline is well protected compared to other parts of the world (Tunley 2009). Currently, South Africa has declared 42 MPAs. In 2019 alone, 20 new or extended MPAs were added (Kirkman et al. 2021; SAEON 2023). Prior to 2019, the South African MPA network comprised of a total of 25 MPAs, nineteen being coastal, one a lagoon and four were islands. These MPAs were all situated within South Africa's mainland exclusive economic zone (EEZ). In addition to these, there was also the Prince Edward Islands MPA, which was located outside of the mainland EEZ (Lombard et al 2007). Zones within MPAs can be categorized in the following ways: (i) Restricted - a demarcated area within a MPA that prohibits disturbance, extraction or harvesting of marine resources and plant life, and (ii) Controlled - an open area within a MPA where the taking and harvesting of marine resources is permitted with a valid permit and subject to species, bag and size limitations. Scuba diving, rock and shore fishing, boat fishing, collecting bait, using jet skis, harvesting molluscs, professional photography, filming, and research are among the activities for which permits can be purchased. Numerous coastal MPAs were created in South Africa to safeguard economically and ecologically significant species (Attwood et al. 1997). However, many of the areas where MPAs have been established were not sufficiently surveyed to determine the availability of resources and habitats (Götz et al. 2009; Kirkman et al. 2021). For instance, the declaration of Robben Island MPA was primarily motivated by the desire to

protect an endangered species, such as the commercially important benthic abalone species, *Haliotis midae*, and its placement was contingent upon the MPA providing this species with sufficient protection (Attwood et al. 1997). However, as populations of protected predators may indirectly affect populations of grazers and abundance of algae within protected areas (Babcock et al. 2010), protection itself may contribute to a community shift in marine benthic communities (Parravincini et al. 2013).

Additionally, managing MPAs while coexisting with nearby coastal communities is a major challenge. Ideally, communities should be involved in the management of MPAs through projects, discussions on quotas and the implementation of management techniques (Sink 2016). Currently, in South Africa, the general public's awareness of MPAs, their role in preserving the health of the ocean and the socioeconomic benefits they offer is low. Government and non-governmental organizations must work together to change the public's perception of MPAs and to position them as the cornerstone of our marine ecosystems' long-term health and the ecosystem services that provide socioeconomic benefits to a large number of communities (Agardy et al. 2011; Sink 2016; Sink et al. 2019). This lack of public awareness of MPAs is partly due to South Africa's MPA boundaries being drawn unpredictably and inconsistently, which can lead to problems with enforcement. This may result in the general public not understanding MPA boundaries and/or not adhering to them. Standardized MPA boundary demarcation and MPA signage outlining the designated MPA zones and their GPS points (with links to download these points onto navigation devices) should be developed nationwide in order to address this problem. In terms of research, law enforcement and public involvement with marine conservation, the managers of MPAs are at the forefront. They need to be skilled in conducting research, maintaining and enforcing regulations, monitoring, and performing a variety of administrative duties, such as writing reports and interacting with the media. MPA managers require a comprehensive understanding of the MPA's legal compliance and governance framework in addition to the ecosystem approach to management (Sink et al. 2019). In order for managers to effectively engage with local communities and the public, they must have a clear understanding of the MPA's objectives, as the park becomes more integrated with society and local communities. When managed by competent managers, MPAs provide an opportunity to demonstrate good governance because they are frequently the one conservation tool that the general public and fishermen are most aware of and are therefore subject to frequent scrutiny (Sink 2016).

Apart from MPAs, marine areas classified as Ecologically or Biologically Significant Areas (EBSAs) are isolated regions that provide important services to species or populations within an ecosystem, or to the overall ecosystem. These areas are compared to nearby regions with similar ecological features, and can also satisfy at least one of the seven EBSA criteria: uniqueness or rarity, special importance for life-history stages of species, importance for threatened, endangered or declining species and/or habitats, vulnerability, fragility, sensitivity, or slow recovery, biological productivity, biological diversity, and naturalness (EBSA, 2018). EBSA's and MPAs are not equivalent in that they do not enjoy the same level of legal protection. While MPAs have legal recognition under South African law, EBSAs are defined by the criteria set forth by the Convention on Biological Diversity (CBD). Although the EBSA designation does not force these areas to become MPAs, many already-existing MPAs may be added to EBSA regions because of comparable selection standards (Sink et al. 2019).

Because many of the existing MPAs in South Africa were established with the primary goal being fisheries management or the conservation of species of high economic value, historically, there was a general disregard for non-commercial or non-focal species which led to a poor scientific base or understanding of whole ecosystems that occur within these protected areas. One of the main functions of priority areas such as MPAs and EBSAs is to conserve biodiversity, or habitats for focal species, but how do you conserve or protect what you do not know? A better understanding of the full range of organisms and habitats that these areas aim to conserve, or already are conserving, is required. Operation Phakisa, is a South African government initiative through which South Africa aims to more efficiently, quickly and better execute critical programs. It is a results driven initiative that includes defining precise goals and objectives, continuously tracking advancement and disclosing the outcomes to the public (Otto 2019). For this initiative, one of the key focus areas is Marine Spatial Planning which includes MPA expansion and for this objective to be met, South Africa needs to improve the science base for existing MPAs (Sink & Attwood 2008). Thus, whether MPA networks or EBSAs are meeting their objectives of biodiversity/habitat conservation, or a more popular term for MPAs in particular – being effective, is a question of high concern, making baseline assessments and continued monitoring particularly significant.

1.3 Aims and objectives

The aim of this thesis is to improve our understanding of the benthic invertebrate communities in priority areas, specifically MPAs and EBSAs, located within the southern Benguela ecoregion. The priority areas that this study focuses on are the long-standing Table Mountain National Park MPA (TMNP MPA), the newly gazetted Robben Island MPA (RI MPA) and a portion of the Seas of Good Hope Ecologically or Biologically Significant Area (EBSA) (referred to as “Cape Point EBSA” in this study). A comprehensive overview of the diversity and distribution of benthos, as well as the environmental factors influencing these communities that will directly inform management and conservation efforts.

The objectives:

- To conduct a comprehensive benthic invertebrate survey to examine the diversity and distribution of benthic invertebrates within the sampling area, focussing on data collected via underwater benthic camera systems.
- To determine if and how environmental variables influence the diversity and distribution of benthic communities, by adopting a multidisciplinary approach, examining variations in depth, substrate type, and oceanographic variables like temperature, salinity and oxygen.

Research Questions:

1. Which taxa of epifaunal benthic invertebrates are found in the southern Benguela ecoregion?
2. How does species diversity and abundance of epifaunal benthic invertebrate communities vary across the three priority areas within the southern Benguela ecoregion: Robben Island MPA, Table Mountain National Park MPA and the Cape Point EBSA?
3. How do environmental variables; depth, substrate type, temperature, salinity and oxygen influence the distribution and abundance of epifaunal benthic invertebrate communities within the sampling area?
4. How do the findings of this study contribute to the establishment and expansion of Marine Protected Areas in the southern Benguela ecoregion?

CHAPTER 2: MATERIALS AND METHODS

2.1 The Study Area

The sampling sites within this study fall within the southern Benguela ecoregion and includes Table Mountain National Park Marine Protected Area (TMNP MPA) and a section of the Seas of Good Hope EBSA (referred to here as the Cape Point EBSA), which extends eastward to Cape Agulhas, as well as the newly declared Robben Island MPA (RI MPA). Table Mountain National Park (TMNP) was established in 1998 and includes 1000 km² of ocean (Van Wilgen et al. 2012) (Fig. 2.1). The area was declared a MPA in 2004 to ensure sustainable ocean use for both commercial and recreational purposes (Brill & Raemaekers 2013). Both TMNP and Robben Island MPA are managed by the South African National Parks (SANParks). The TMNP MPA covers the entire Cape Peninsula, with most of it being a controlled zone with six restricted "no-take" zones (Brill 2012).

Robben Island was declared as one of South Africa's new MPAs under the government's Operation Phakisa initiative in May 2019. The MPA covers 580 km² (Sink et al. 2019; Kirkman et al. 2021; (National Gazette No 42478 of 23 May 2019, Volume 647) and falls within the south-eastern portion of the Cape Canyon EBSA (Sink et al. 2019). The main objective of the MPA is to protect the Island's endangered bird populations, but it is also a sanctuary for the recovery of abalone, *Haliotis midae*, and West Coast rock lobster, *Jasus lalandii*. There is very little information on the full scope of the benthic life and its associated habitats, which are vital in sustaining the Island's inhabitants (Sink et al. 2019). In 1999, the Island was declared a World Heritage Site (Underhill et al. 2001) and according to the South African National Heritage Resources Act of 1999, management decisions must be based on a specific conservation management plan for the Island (Deacon 2004). Presently, four distinct zones make up the Robben Island MPA, three are controlled zones and one a restricted zone.



Figure 2.1. Table Mountain National Park MPA and its Restricted Zones within the Control Zone (Adapted from Brill & Raemaekers 2013).

Proposed through a Marine Spatial Planning (MSP) process which include robust stakeholder engagement and negotiations among sectors, the newly proposed Cape Point EBSA (Fig. 2.2) is known to be of great significance because of the Agulhas and Benguela currents that meet offshore of its proposed boundary. This unique oceanic feature makes the area important for the study of benthic life and its associated habitats. The area is a habitat for a number of commercially important fish species, which in turn provides a key feeding ground for a variety of predators including seals and seabirds (EBSA 2018).

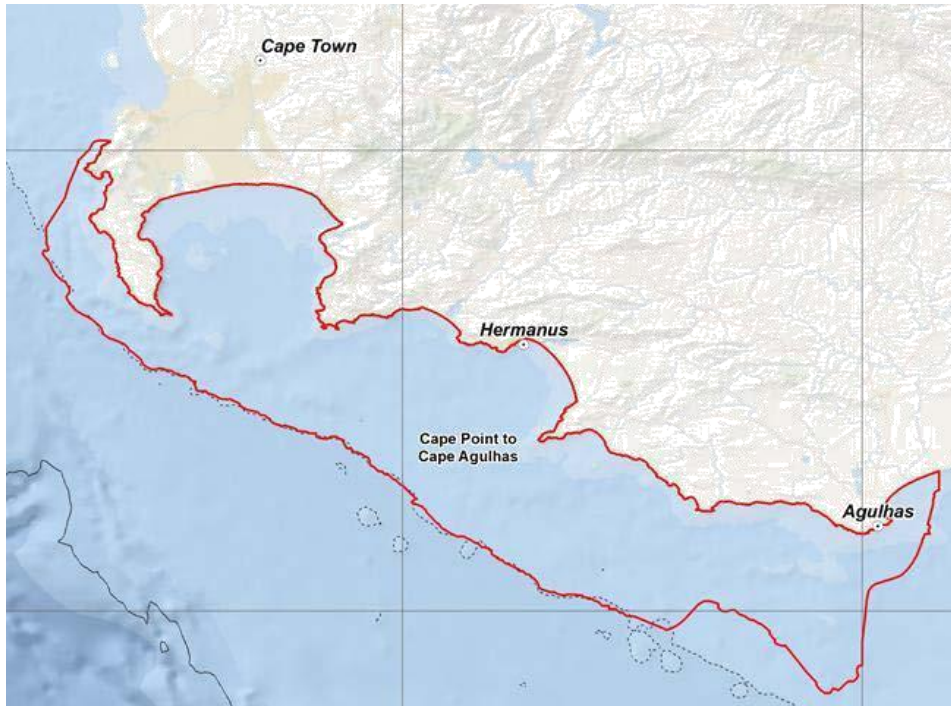


Figure 2.2. The newly proposed Cape Point EBSA which falls within the sampling area (EBSA 2018).

2.2 A review of the camera systems

For this study, we focussed specifically on two benthic camera platforms used by the Department of Forestry, Fisheries and the Environment (DFFE): the Ski-Monkey (a towed camera system), and a Drop camera system (Fig. 2.3A and B)

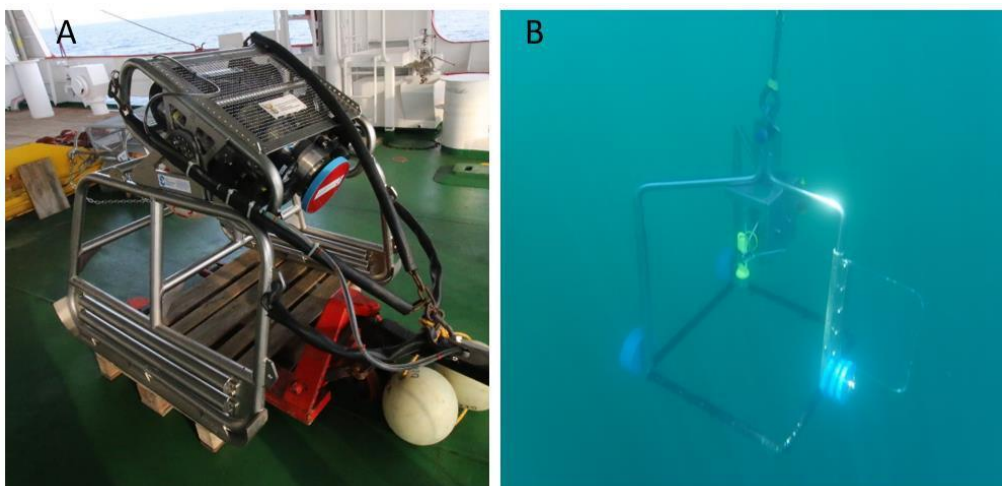


Figure 2.3. The benthic camera systems used by the DFFE: the Ski-Monkey (A) a towed camera system, and a Drop camera system (B)

These camera systems provide invaluable data that is not otherwise obtained from other benthic sampling approaches (such as trawl nets, dredges and grabs) and are the methods most widely used for collecting quality images of the benthic environment. These systems produce both video and still images. The DFFE Ski-Monkey (Fig. 2.4) is configured to collect data in either a towed position (positioned on skis) or a Drop camera (tri-pod legs) system and is pressure tested to 700 m. Finer components of the benthic camera include; three lasers which assist in determining the size of animals in the field of view, two flash lights and two 24 volt LED lights that illuminate the darker depths of the ocean. The system design includes both a detachable instrument that measures conductivity, temperature and depth (CTD), along with an altimeter that records point specific depth. Apart from these external units, the interior control panel links the camera to the observational room where scientists and technicians receive high-definition data through a live feed. Area specific geographic information is also recorded. However, despite the function of such a system in affording an array of data, and being less destructive in doing so, operational technicalities have proven to be challenging. The camera configuration for this study is a forward facing (oblique view) camera. The oblique angle of view improves species identification and gives a clear indication of the benthic habitat type. Fully automated photos are taken at regularly spaced, pre-defined intervals which allows for sampling bias to be avoided. Manually operated stills may also be taken opportunistically, allowing for quality images of target organisms or habitats. This may create bias and although such images should not be included in the statistical analysis, it can assist with organism identification when required.

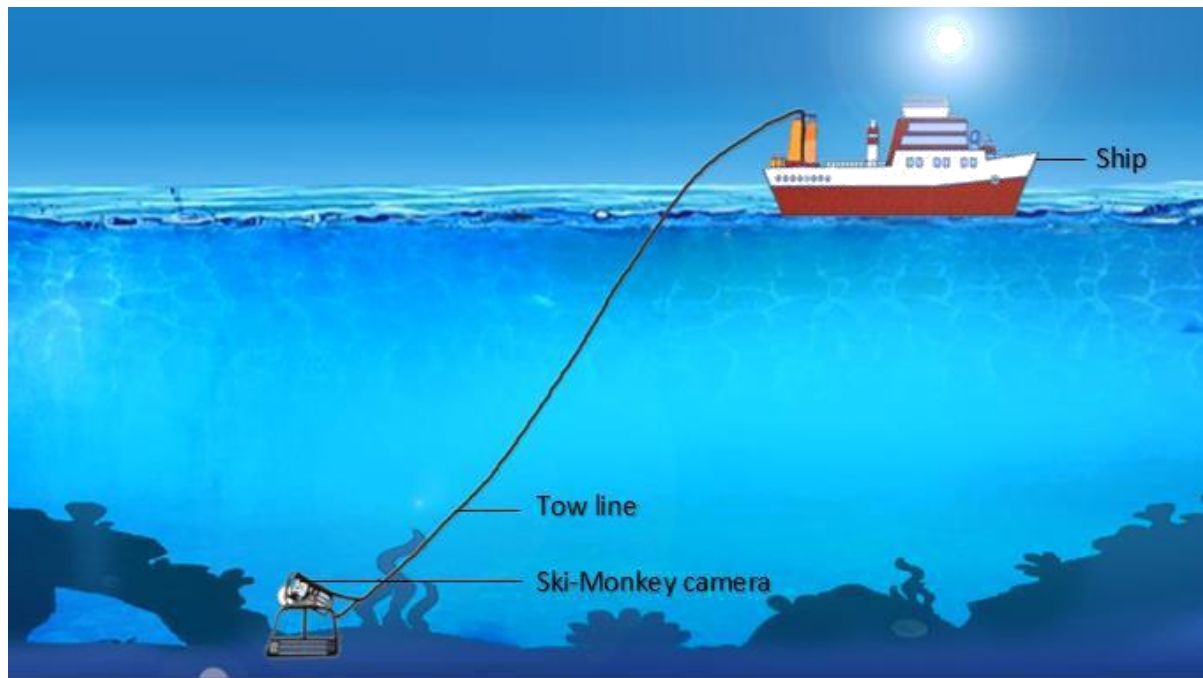


Figure 2.4. Ski-Monkey Tow camera illustration (Hudsonberg 2019).

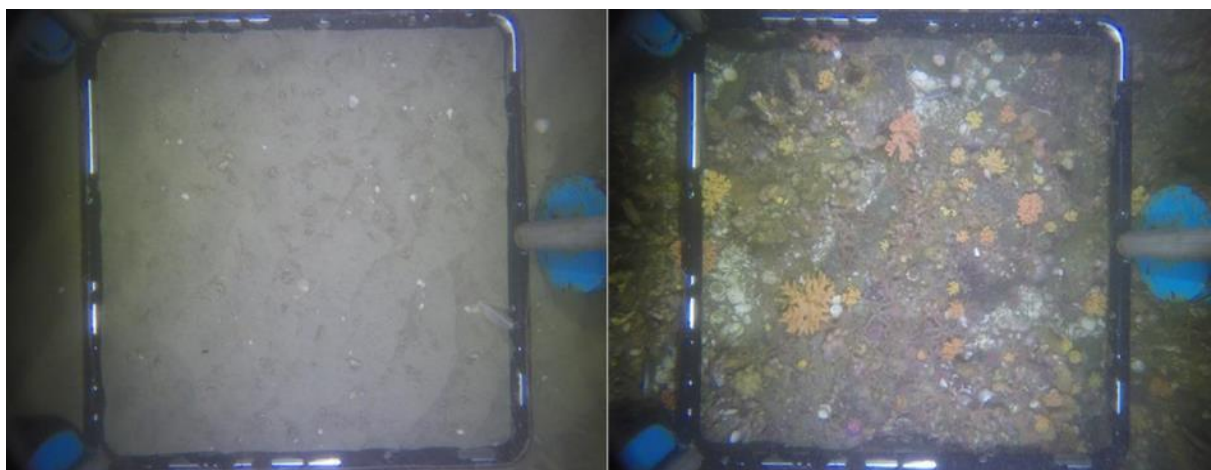


Figure 2.5. The Drop camera quadrat positioned on the sea floor.

The vertically deployed Drop camera captures benthic quadrat images (Fig. 2.5), used in this study, can isolate a standard unit area for study of the distribution of organisms over a large area. Its frame is used to provide the necessary protection and stability for the equipment tethered to the surface. It gives the equipment a platform to rest on while on the bottom and shields the equipment from the terrain while it is being lowered. The entire unit is built out of stainless-steel material. The bottom exterior frame is 60 60 cm and makes up the photo quadrat that is within the camera's field of view. The top of the frame contains mounts for a pressure tested (300 m) GoPro camera housing and two lights (pressure tested torches) (300m), which

should be oriented towards the center of the grid. The eye used for mounting the rope to which the frame is tethered to the surface is located on top of the frame. For size referencing, lasers could also be used (if needed in future). The camera is orientated downwards for quantifying and qualifying collected footage within the quadrat. The Drop camera differs from the Ski-Monkey (towed camera) in that it is a vertically deployed passive camera system suspended and deployed over the side of the ship or small boat, instead of being towed. The Drop camera however, can be used as a “jump camera”, “jumping” the frame across the sea floor while drifting in the current (Fig. 2.6) to obtain close up images of benthic fauna and sediment surfaces along with wide views of the habitat.

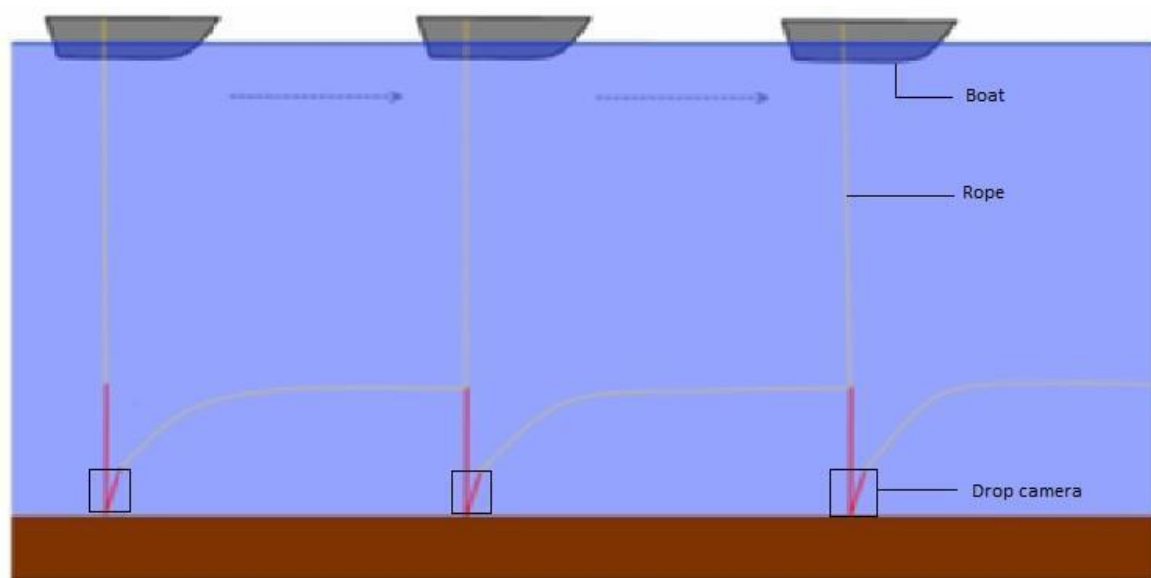


Figure 2.6. An illustration of the Drop camera sampling technique

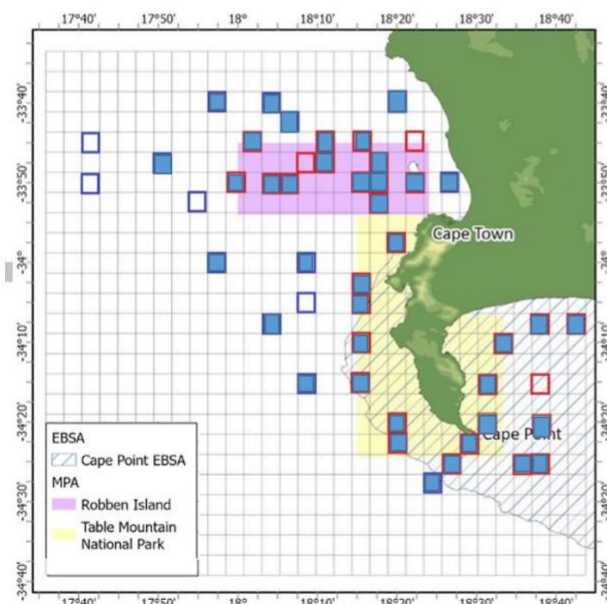
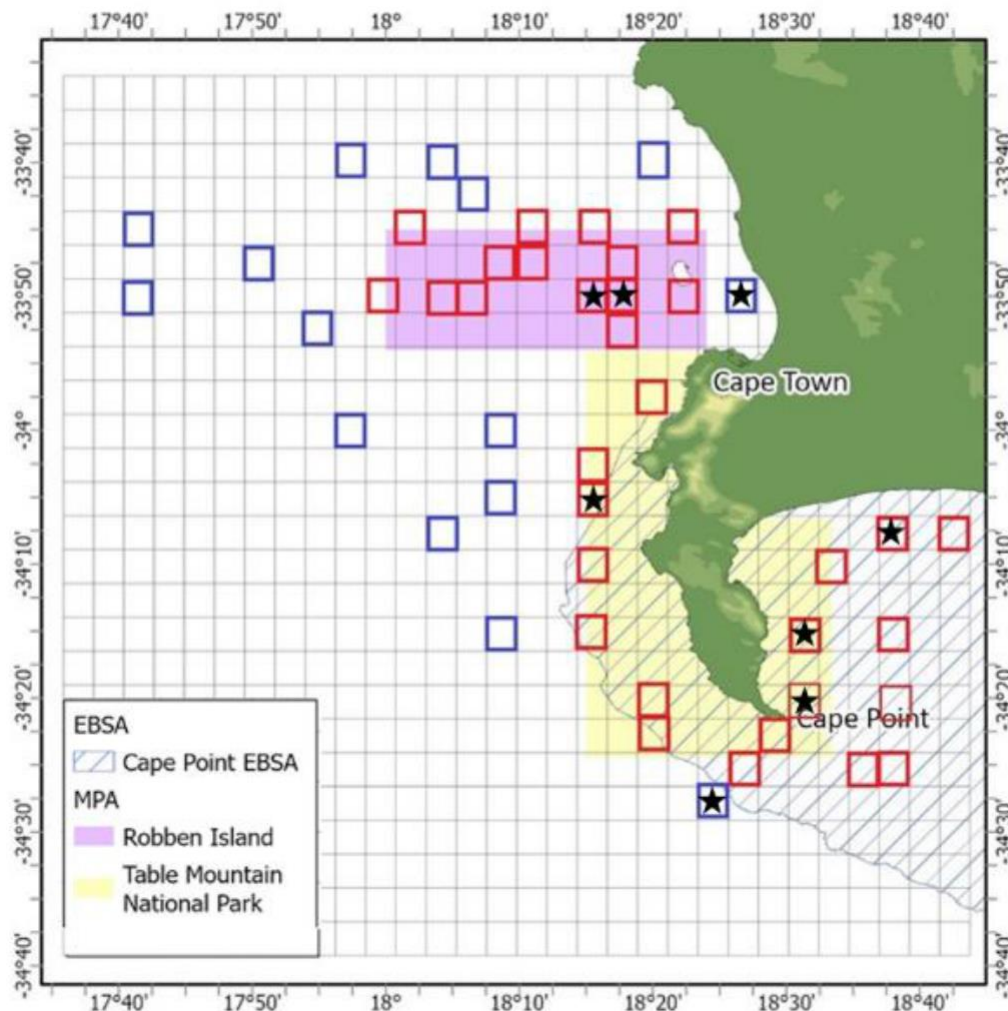
Alternatively, short transects can also be completed, provided that the site has a suitable terrain. If the visibility is poor or the intention is to obtain close-up photos of the benthos, then the frame can be lowered onto the sea floor (Appendix 1). On the other hand, to obtain a wide view of the benthic environment, the frame can be suspended above the sea floor (Appendix 2). The Drop camera can sample a greater variety of habitats including complex topographical reef and rock features (Appendix 12). However, since the camera is not always in contact with the sea floor, large swells can cause the camera height above the sea floor to be variable and as a result, the frame can collide against the sea floor, obscuring the view (Appendix 3), particularly in sandy, silty and muddy environments. Thus, sampling is only recommended under suitable sea conditions which are < 2m swells and < 10 knots wind speed if possible. Described below are the methods used to derive the environmental data and the numerical analyses used to identify the environmental variables responsible for the community patterns.

2.3 Sampling

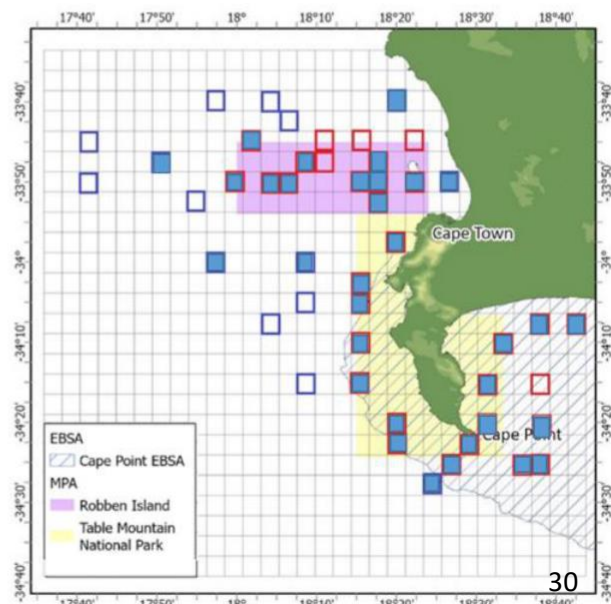
Data were collected during a cruise led by the Department of Forestry Fisheries and the Environment (DFFE) on-board the *RV Algoa* for ten days (29 August 2019 – 7 September 2019). In total, 62 stations were proposed for sampling but only 48 were successfully sampled (30 Drop camera stations and 18 Ski-Monkey stations) due to technical difficulties and time constraints. Sampling of the benthos and associated habitats/physical features was conducted at stations across a depth gradient via specimen/sample collections, visual surveys and the collection of oceanographic information. Taking into account depth restrictions for each piece of equipment, visual surveys of benthic fauna and associated habitats were conducted using both a Ski-Monkey and a Drop camera system and specimen/substrate collections were carried out using a Dredge and a Grab (Van Veen). The latter sampling was necessary to verify habitat/substrate types and also to provide a reference collection of specimens which assisted in the identification of taxa/ species in the photographs. This sampling was limited mainly to stations outside of priority areas because of their destructive nature. Oceanographic variables were collected at each station using the CTD.

In QGIS (version 3.4.2) a grid layer was superimposed over the study area in order to split the area into sampling units (henceforth referred to as cells) and to mitigate spatial autocorrelation (Figure 2.7) A grid with 1cm x 1cm cells was placed over the entire sampling area and each cell represented a station of 5 km² (Fig. 2.7). Stations to be sampled were pre-selected using a random number generator according to various depth bins for the sampling area: 20–50 m, 51–

80 m, 81–110 m, 111–140 m and 141–180 m. In addition, since Robben Island was only recently gazetted as an MPA, a set of stations unique to this relatively shallow water area and adjacent areas with depth bins: 20–50 m, 51–80 m, 81–110 m, 111–140 m and 141–180 m, were also sampled. Again, stations were randomly selected using a random number generator according to various depth bins.



Ski-Monkey stations.



Drop camera stations.

Figure 2.7. Sampling areas within the southern Benguela ecosystem, showing Robben Island and TMNP MPAs, and a section of the Cape Point EBSA which extends eastward to Cape Agulhas. Squares represent randomly selected stations within a grid at depths between 16–250m. Stations sampled within and outside of priority areas are indicated as red and blue squares, respectively. The ★ symbol represents stations where both cameras were successfully deployed and retrieved.

At each sampling station (Fig. 2.7) the benthos was surveyed visually using either the Ski-Monkey tow camera, the Drop camera system or a combination thereof. The dredge and a Van Veen grab were used to collect physical specimens along with sediment at selected stations outside of priority areas. A CTD was also deployed at each station to support the imagery data with oceanographic variables recorded. The shipboard CTD consisted of a big metal rosette wheel connected to a number of probes. A conducting cable that connects the CTD to a computer on board the ship allowed scientists to view parameters in real time while the rosette was lowered vertically on a cable to the seafloor. To prevent any form of bias, the order of sampling activities (i.e. CTD, Ski-Monkey, Drop camera, Grab and Dredge) remained consistent throughout the cruise, since one instrument could negatively impact the sample output of the next piece of equipment deployed. Operational times for each piece of sampling equipment remained consistent unless forced to be cut short due to technical issues experienced with the equipment or ship. Since this study focuses on benthic imagery data, only the protocols for the Ski-Monkey and Drop camera are explained in further detail below.

The Ski-Monkey operations took 60 minutes per station starting from the point that the camera was placed into the water until it was retrieved and back on the ship's deck. This time included deployment time to get to the sea floor, properly landing and towing the camera along the sea floor and retrieving the camera from the sea floor back to the ship. Deployment and retrieval speed of the camera was determined by the winch speed setting standard for the ship which is 1 m per second. Allowable depths range from 40 to 700 m. The distance covered by the Ski-Monkey is for a 10 minute tow is +/-500m. The camera was deployed using the vertical plankton winch which is a 10 millimetre (mm) high conductivity bunched, stranded copper core cable that is insulated and colour coded. Insulated cores are twisted and filled with a PVC bedding to give the cable a round finish. It is armoured with a galvanized steel wire. These cables allow for a live view to be possible. Prior to deployment camera connections were checked on the deck. This deck test involved testing the lasers, lights and camera live view

connections. The camera was deployed while the vessel was holding position and once on the seabed and the live video was properly focused, the camera team communicated with the vessel to move at a speed of 0.5 to 1.5 knots to pay out the cable. Once the tow angle was set, still images were taken manually for 10 minutes, followed by a video for a further 10 minutes when battery power allowed.

The Drop camera operations took 45 minutes per station starting from the point that the camera is placed into the water until it is retrieved and back on the ships deck. This time includes deployment time to get to the sea floor, sampling and retrieving the camera from the sea floor back to the ship. The camera was deployed off the aft of the ship using the small towing winch. This winch cable does not have live wire inside but is a 6 mm stranded galvanized steel wire. The camera is deployed while the vessel is holding position and was moved away from the ship off the side using the crane to prevent any complications between the cable and the propeller wash since it is a vertical dip. The camera housing allows for the set up to explore depths up to 300 m. Once on seabed, the camera team communicated with vessel (i.e. bridge) to hold position for 60 seconds (GoPro set to take stills every 5 seconds) after which the camera was lifted roughly 5m above substrate. The ship was then asked to drift (i.e. carried slowly by a current of air or water) within the position, hold position and the camera was dropped again. These steps were carried out 10 times per station after which the camera team communicated with the vessel to bring the Drop camera up to the surface. This was the first Drop camera deployment with the current set up to take place onboard the *RV Algoa*.

2.4 Sample analysis

Photo quadrats from images collected by both the Ski-Monkey and Drop camera system provided a permanent in-situ visual record along with a more precise estimate of percentage cover (i.e. compared to conventional techniques) of abundant benthic species (Preskitt et al. 2004). Coyer and Witman (2012) described a quadrat as a predefined area which is searched for biota and in determining which type of quadrat to be used in a specific study, statistical precision and cost efficiency are two major factors to be considered. Cost is proportional to the time spent examining the quadrat (Pringle 1984) and for statistical precision, a quadrat that would produce minimum Standard Error (SE) should be used (Krebs 1989). Statistical precision also increases when the edge effect is minimized, which occurs when an organism lies under the border of the quadrat resulting in the surveyor needing to decide whether that organism is within the quadrat. Issues around parallax are also factors which impact bias and

observer error based on the surveyor's position in relation to the quadrat, resulting in bias or error being more likely (Krebs 1989). Readily available open-source software packages may be used to process photoquadrat images. Image analysis software like Coral Point Count with excel extensions (CPCe) (Fig. 2.8; Fig. 2.9) was used to analyse images. This is a user friendly, open-source Windows-based software that provides a tool for the determination of benthic cover using photographs. This is a commonly used method for estimating species diversity and community structure of marine benthic environments (Kohler and Gill 2005). Foster (1991) determined that to simultaneously calculate percentage cover and abundance estimates, the use of 100 points per photograph is ideal (by photographing and mapping 30×50 cm areas in his study), allowing for faster processing and ultimately limits observer bias. One hundred intersection points were overlaid on each photographic sample via a uniform grid of ten rows by ten columns (see Fig. 2.8 & 2.9). The habitat characteristics for points landing on hard or soft bottoms was also recorded. Absence or presence and species abundance for each photographic sample was produced by observing, identifying and recording species located beneath the overlaid intersection points and guides were used to assist with species identification. By recording and summing the number of points over each individual species within a sample, percentage cover per organism can be calculated. The footage of the Ski-Monkey was analysed via grid lengths per tow. Five images along this grid length were used to determine the habitat type. The tow track was divided into five sections with snapshots taken at the midpoint of each of the five. The ten minute video tow was divided into five (which means it was broken into five two-minute clips with the sample image taken at the mid-point of these two minute clips to avoid any bias). Likewise, the Drop camera footage was also analysed by means of representative images per station and only all clear quality images per station was used. All viable photo quadrats (clear, with focal frame properly positioned within the frame of the photograph) collected were analysed and each photograph was treated as a sample within a sampling station. Only viable photographs (visually clear) per station were used and the number of samples per station therefore differed. A total of 509 photo-quadrats were analysed: RI MPA (176), TMNP MPA (183) and Cape Point EBSA (150). The Drop camera had 133 samples for RI MPA, 153 samples for TMNP MPA and 118 samples for Cape Point EBSA. The Ski-Monkey had 50 samples for RI MPA, 25 samples for TMNP MPA and 15 samples for the Cape Point EBSA. Each organism was identified and percentage cover of species were recorded for each photograph by means of photographic point sampling (i.e. 100 points per photograph).

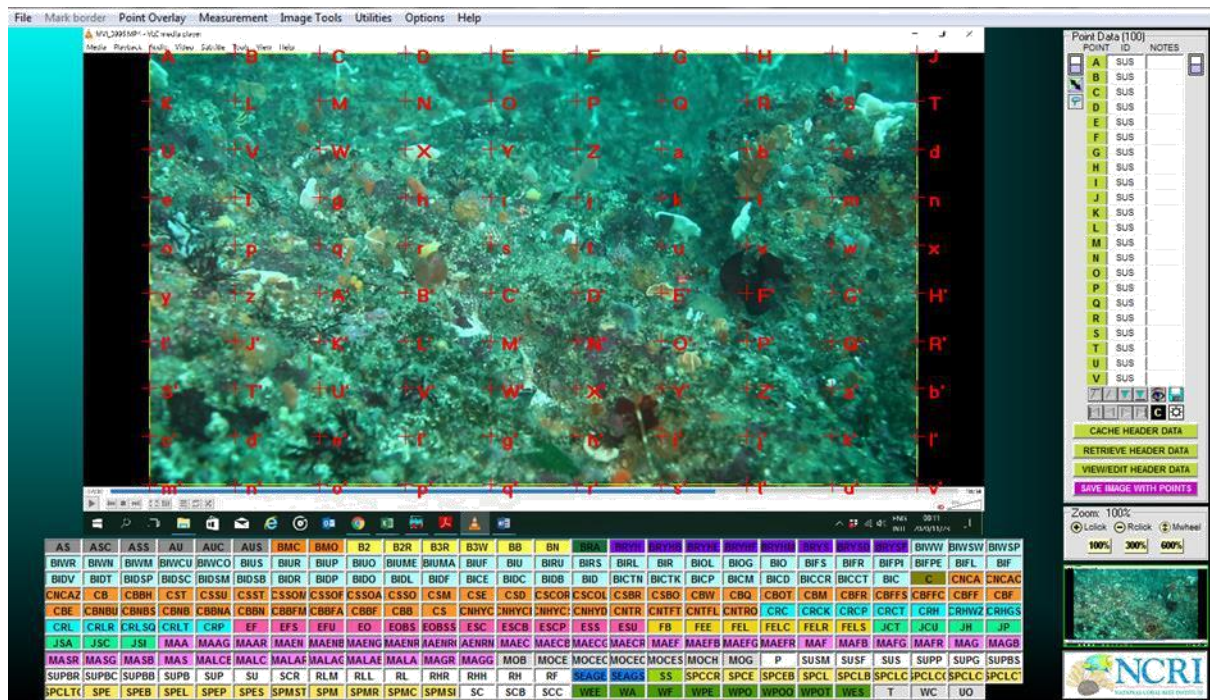


Figure 2.8. Example of an annotated oblique Ski-Monkey image in Coral Point Count with 100 grid point annotations.

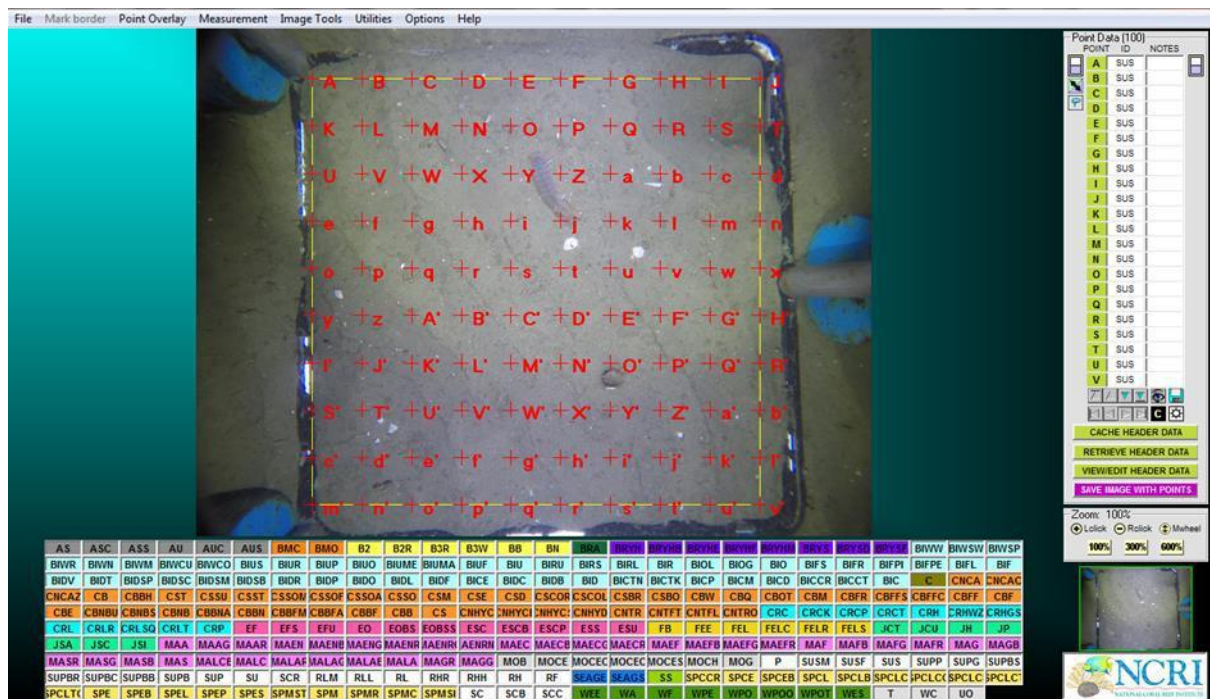


Figure 2.9. Example of an annotated Drop camera image in Coral Point Count with 100 grid point annotations.

2.5 Data analysis

Species were separated into seven broad taxonomic groups: ascidians, bryozoans, cnidarians, echinoderms, molluscs, poriferans and unknowns. A catalogue of all epifaunal species occurring within the in-situ photographs was developed by identifying and listing all the species observed. The catalogue including the in-situ photographs of each species observed was added to a database. This database was used to select and identify species from samples within CPCe. Species accumulation curves were plotted to test whether the sampling effort was adequate. Species accumulation curves record the number of species (cumulative) during the sampling period as a function of the sampling effort (cumulative) (Gotelli and Colwell 2001). The species accumulation curves were produced using PRIMER 6 and indicates S_{obs} (total number of species observed in samples) (Chazdon et al. 1998). By plotting the total number of species against the number of samples, the species accumulation curves were produced for both the camera systems. Smaller, less noticeable morphospecies, such as molluscs (bivalves, gastropods and cephalous), may be difficult to observe and recognize due to the presence of large, dominant, and noticeable organisms. Because these morphospecies may be obscured by algae and epibionts or concealed in cracks and crevices, they may have been under-sampled. In order to down-weight the presence of abundant morphospecies, the morphospecies abundance data were fourth-root transformed using Primer version 6 (Clarke et al. 2014). As a method of analysing benthic epifauna and to relate factors such as depth, substrate type, temperature, salinity and oxygen to biological data for the three sites (Robben Island MPA, TMNP MPA and the Cape Point EBSA), PRIMER 6 (Plymouth Routines in Multivariate Ecological Research) was used. The functions that were used were cluster analysis, species accumulation curves, analysis of variance (ANOVA), permutational multivariate analysis of variance (PERMANOVA) and a Similarity Percentage procedure (SIMPER). To determine the significance levels of each factor, the environmental variables were related to the macrobenthos using a distance-based linear model (DISTLM, PRIMER version 6 with PERMANOVA+ add-on). To illustrate the relationship between important variables and the biotic communities, a distance-based redundancy analysis (dbRDA) diagram was created (Appendix 9). The parameters for the DISTLM were set to AIC for selection criterion and the selection procedure was set to “Best” to select the model with the best combination of environmental variables that explain the biotic community patterns. The environmental variables that strongly correlate with the community resemblance were selected from the using the BIO-ENV analysis in PRIMER version 6. The PERMANOVA allows for testing of the null hypothesis that the species diversity of the epifaunal benthic invertebrate communities in the southern Benguela ecoregion

is not influenced by geological and physical factors such as habitat type, depth and temperature. The similarity matrix of sample station abundance data was used to calculate the Global R value in order to test for similarity between sample station location and substrate type. A Global R value of approximately 0 would suggest that similarities between sampling stations are generally the same, allowing for the null hypothesis to be accepted (Clarke and Warwick 2001). A significant difference in similarity between sampling stations would produce a Global R value closer to 1 (Clarke and Warwick 2001). To separately determine the species accountable for the similarity between locations, a Similarity Percentage procedure (SIMPER) was performed on fourth root transformed species abundance data. A SIMPER procedure highlights the species most influential in distinguishing between sites (Gray 2000). The procedure gives the contribution of each species to the overall similarities and /or dissimilarities that are as a result of variation in species abundances. One of the most common requirements for statistical test procedures is that the data used must be normally distributed. If t-test or an ANOVA is to be calculated, it must first be tested whether the data or variables are normally distributed. This is done with a test of normality. Normal distribution can be tested either analytically or graphically. The most common analytical tests to check data for normal distribution are the Kolmogorov-Smirnov Test, Shapiro-Wilk Test and Anderson-Darling Test. With all these tests, we test the null hypothesis that the data are normally distributed. For the graphical test, either a Quantile Quantile plot (Q-Q plot) or a histogram is used. In this study the latter was made use of. The null hypothesis is that the frequency distribution of the data fits the normal distribution. In order to reject or not reject the null hypothesis a P-value is obtained from each of the above mentioned tests. A P-value less than 0.05 is considered statistically significant meaning that the null hypothesis should be rejected. This indicates that the observed data is unlikely to have occurred under the null hypothesis. Homogeneity of variance between sample sites was determined by performing the Levene's Test for equality of variances.

In order to obtain natural groupings and to visually observe differences or similarities which may exist, Multi-dimensional scaling (MDS) ordinations were done on the frequency of occurrence of both camera systems. MDS Ordination was used on species abundance per sample to determine the effect of substrate type and location on the species abundance per sample. Data was fourth root transformed and the Bray Curtis similarity index was used. Location and substrate type were the factors used to indicate if natural groupings occurred in community structure between sand/mud stations and rock/reef stations and between stations of Robben Island MPA, TMNP MPA and Cape Point EBSA. A draftsman plot was produced to

investigate correlations between the environmental variables. This method can be used for looking at the interrelations between variables in multivariate data. The plot comprises of an array of two-variable scatter diagrams. To graphically present and to investigate the relatedness between the variables, a principal component analysis (PCA) was performed on the environmental variable data of both camera systems. Location and substrate type were the factors used to indicate if natural groupings occurred in community structure between sand/mud stations and rock/reef stations and between stations of Robben Island MPA, TMNP MPA and Cape Point EBSA. This was plotted onto the PCA plot to investigate the relationship between the environmental variables in relation to the samples.

Diversity indices in PRIMER 6 using the DIVERSE function was used on the species abundance data to obtain species richness (S), species diversity (H') and evenness (J').

Despite the total number of species being linked to species richness in an area, the number of species on its own is inadequate to describe assemblages as the sample number per species may differ. Species richness was therefore used along with species diversity to provide indications of connectivity and patterns in the area. Being the most frequently acknowledged index in biodiversity studies, the Shannon-Wiener index was used (Gray 2000). The index is calculated as:

$$H' = -\sum_{i=1}^S p_i \log p_i$$

Where S = total number of species obtained and p_i = the proportion of each species in the sample.

Evenness is a measure of the abundance of the variety of species making up the richness of an area and was calculated by means of the Pielou's evenness index (Clarke and Gorley 2006):

$$J' = \frac{H'}{\ln S}$$

J' is the number obtained from the Shannon-Weiner index with H' being the highest value scored with the Shannon-Weiner index. Evenness measures and indicates communities with low evenness as close to 0, and close to 1 as high evenness (Routledge 1980; Stirling and Wilsey 2001). Smith and Wilson (1996) describe patchy distributions to be linked with low evenness and equally distributed, associated with high evenness. The R statistical program (version 4.0.2) was used to test whether there were significant differences among the benthic epifaunal communities and their habitat types and to determine which habitat had the highest abundances. An Analysis of Variance (ANOVA) was used to determine whether there were

significant differences between species richness, abundance and diversity between locations, depth bins and the camera systems used. An ANOVA was done as the number of samples differed among both camera systems as well as the locations sampled. STATISTICA V7 was used to perform the ANOVA. The ANOVA takes individual samples per location, instead of groups, and provides significant differences between them (Shaw and Mitchel-Olds 1993). To further illustrate variations and differences between samples, plots of the means and standard deviations of the species diversity for the different camera systems were also used. To establish if any significant differences occurred between the species diversity, richness and abundance among stations, in terms of depth, an ANOVA was used. This was done in STATISTICA V7. For normally distributed data, the t-test is one of the most commonly used methods for comparing two independent groups (Glover et al. 2011). In theory, ANOVA samples are randomly collected from populations that have equal variances and are normally distributed (Zar 1999; Glover et al. 2011). Despite this, the test allows for minor deviations of unequal sample variances and normality (Zar 1999). Therefore, the ANOVA could be performed in this study even though the sample sizes were unequal, as deviations in variances were minor along with the fact that the data was fourth root transformed which narrows the skewness of data (Warwick 1988). The ANOVA was also used to test for significant differences between the species diversity and abundance using the two camera systems.

To establish whether or not the species richness means and number of individuals counted increased interdependently with temperature and depth, the correlation coefficient (r) was used (Clarke and Warwick 2001). Negatively correlated data is indicated by an R -value between the range -1 and 0 and positively correlated data between 0 and 1 (Clarke and Warwick 2001). $r =$ close to 1, would be indicative of a strong correlation (positive) between the data and that the species richness and number of individuals counted are interdependent of each other.

CHAPTER 3: RESULTS

A total of 72 species were visually observed in the samples collected across the study area. The most diverse epifaunal taxonomic group found in the Table Mountain National Park MPA was Cnidaria (30 %), followed by Porifera (21 %). In the Cape Point EBSA, the most diverse group was Mollusca (35 %) followed by Porifera (30 %) and for Robben Island MPA, it was Echinodermata (50 %) followed by Cnidaria (19 %). The mean percentage cover of benthic organisms for samples collected using the Drop camera method was 26.91 % (± 18.07 SD) for rock/reef and 37.4 % (± 19.76 SD) for sand/mud while for the Ski-Monkey, the percentage cover was 10.45 % (± 13.33 SD) and 75.17 % (± 16.74 SD), respectively (Appendix 12 and Appendix 12.1).

3.1 Sampling Effort

In terms of the sampling effort, species accumulation curves of all abundance data for both the Drop camera (Fig. 3.1) and the Ski-Monkey (Fig. 3.2) methods across the three sampling areas did not reach an asymptote. The fact that the curves did not reach asymptote, therefore suggests that the sampling effort was not sufficient to capture the full range of species present in the area. The species accumulation curves of the Drop camera method, however, appeared to be nearing an asymptote, suggesting a better indication of the total number of species present in the area. The asymptote of the curve indicates the total number of species in the area and serves as a measure of the sampling effort. The fact that the curves did not reach an asymptote suggests that the sampling effort was insufficient to capture the full range of species in the area.

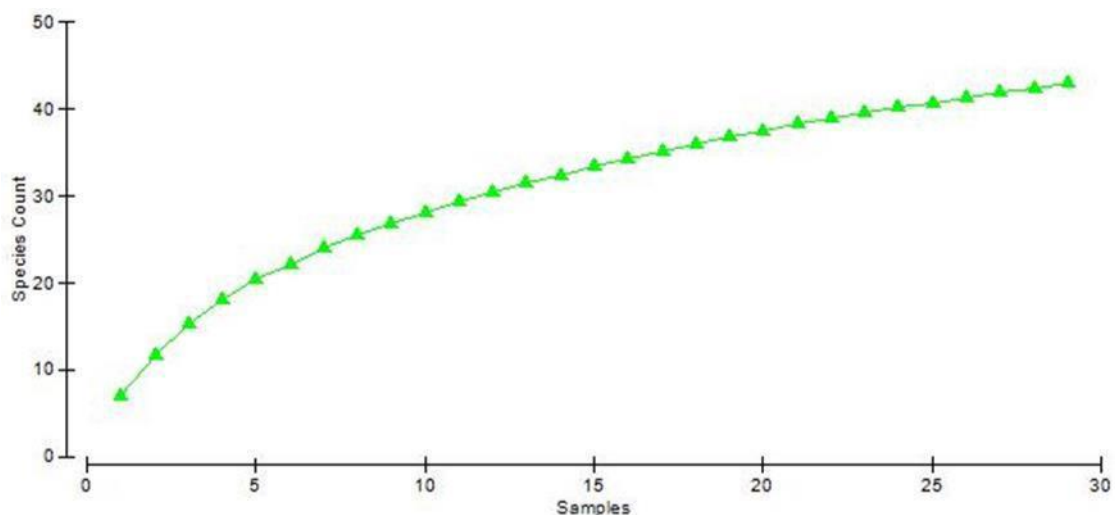


Figure 3.1. Drop camera species accumulation curve of all samples' abundance data, indicating the total number of species observed in samples (Sobs). Samples on x-axis represent the sampled stations.

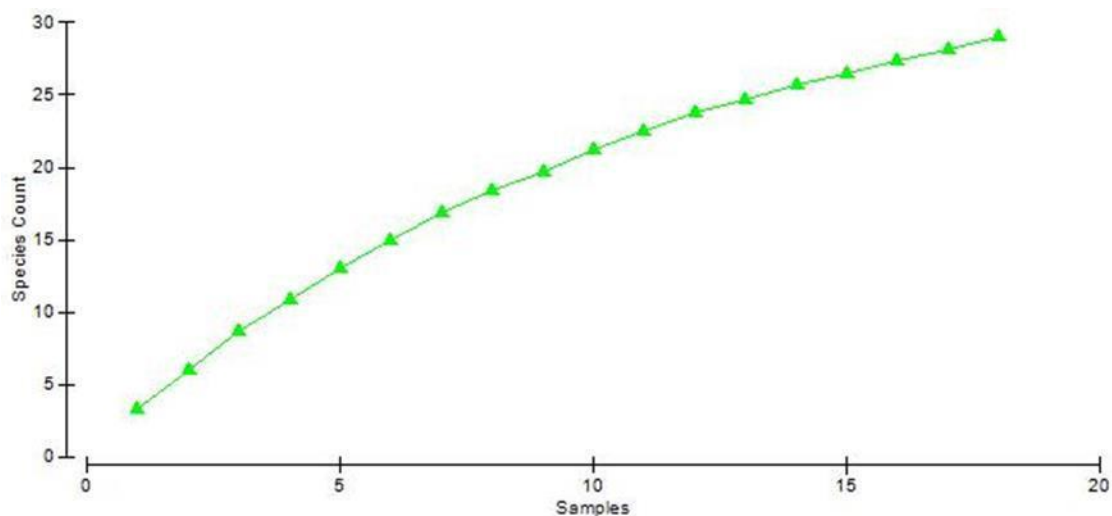


Figure 3.2. Ski-Monkey species accumulation curve of all samples' abundance data, indicating the total number of species observed in samples (Sobs). Samples on x-axis represent the sampled stations.

3.2 Diversity Indices

Diversity indices were calculated on all species' abundance data collected by the Drop camera and the Ski-Monkey. The mean species richness of samples within the study area for Drop camera sampling was $3.95 (\pm 1.08 \text{ SD})$, the mean abundance was $8.9 (\pm 1.64 \text{ SD})$ and the mean percentage cover measured $11.38 \% (\pm 14.21 \text{ SD})$ (Appendix 12.2). The mean evenness for all Drop camera samples was high and measured $0.95 (\pm 0.03 \text{ SD})$. The mean species diversity for all the samples was $1.20 (\pm 0.28 \text{ SD})$. For the Ski-Monkey method, the mean species richness of samples within the study area was $4.62 (\pm 1.12 \text{ SD})$, the mean abundance was $11.64 (\pm 1.52 \text{ SD})$ and the mean percentage cover measured $6.64 \% (\pm 9.58 \text{ SD})$ (Appendix 12.3). The mean evenness for all Ski-Monkey samples was high and measured $0.97 (\pm 0.02 \text{ SD})$. The species

diversity mean for all samples was 1.42 (± 0.21 SD), which in relation to the abundance can also be considered high.

The highest mean percentage cover of benthic organisms collected by the Ski-Monkey was (38.27%, ± 51.43 SD) and the lowest (0.20%, ± 0.45 SD) (Appendix 12.1). The Drop cameras' highest mean percentage cover on rock was (83.89 %, ± 23.66 SD) and lowest (1.88 %, ± 5.30 SD). Using the Ski-Monkey, the highest mean percentage cover on rock was (44%, ± 44.64 SD) and lowest (0.25 %, ± 0.56 SD). On sand, the highest mean percentage cover using the Drop camera was (99.73 %, ± 0.47 SD) and its lowest was (0.70 %, ± 1.06 SD). The highest mean percentage cover on sand, using the Ski-Monkey was (99.80 %, ± 0.45 SD) and its lowest was (9.00 %, ± 20.12 SD). Station 176 (Cape Point EBSA) possessed the highest means in terms of the diversity indices (H' and S). The highest means in terms of abundance and percentage cover was found at this station as well. Analysis of variance calculations of species richness, abundance and diversity between means revealed that there were no significant differences between all the sites ($P > 0.05$). The Ski-Monkey samples possessed the highest means in terms of the diversity indices (S , N , J , H ($\log_e$)) and the plots of the means and standard deviations of diversities of Drop camera samples and those of the Ski-Monkey camera samples, revealed an overlap in standard deviations between the cameras. The Drop camera samples possessed the highest means in terms of percentage cover. Analysis of Variance calculations of abundance, species richness and diversity between camera system means of all stations sampled, did not reveal any significant differences ($P > 0.05$).

The ANOVA of species richness (mean $S=3.95$ species for Drop camera and mean $S=4.62$ species for Ski-Monkey samples), diversity (mean $H' = 1.20$ for Drop camera and mean $H' = 1.42$ for Ski-Monkey samples) and abundance (mean $N = 8.88$ individuals in Drop camera and mean $N = 11.64$ individuals in Ski-Monkey samples) across all stations using the different camera systems revealed no significant differences ($P < 0.05$). The mean species diversity of samples collected by the Drop camera and Ski-Monkey ($H' = 1.20 \pm 0.28$ SD and $H' = 1.42 \pm 0.21$ SD respectively) were found to be significantly different ($P < 0.05$) and the large standard deviations on either side of the mean indicated a large variation in samples. The ANOVA also revealed that the mean abundance in Drop camera samples ($N = 8.88$ individuals ± 1.64 SD) was significantly lower ($P < 0.01$) than that of the Ski-Monkey samples ($N = 11.64$ individuals ± 1.52 SD). The mean species richness found in the Drop camera samples ($S = 3.95$ species $\pm$

1.08 SD) was significantly higher ($P < 0.05$) compared to that of the Ski-Monkey samples ($S = 4.62 \text{ species} \pm 1.12 \text{ SD}$), as revealed by the ANOVA.

Table 3.5. FACTORIAL ANOVA results indicating similarity within the study area (with regards to location) the ANOVA was performed on fourth root transformed data using Bray-Curtis similarity matrix.

FACTORIAL ANOVA						
Source of Variation	SS	df	MS	F	P-value	F crit
Sample	46861,30303	2	23430,65	7,230202	0,001537	3,150411
Columns	16450,9697	1	16450,97	5,07642	0,027922	4,001191
Interaction	11033,84848	2	5516,924	1,702406	0,190929	3,150411
Within	194439,8182	60	3240,664			
ANOVA						
Source of Variation	SS	df	MS	F	P-value	F crit
Between Groups	199,2389	2	99,61946	0,971661	0,040634	3,885294
Within Groups	1230,299	12	102,5249			
Total	1429,538	14				

3.3 Assemblage Composition

The Drop camera Cluster analysis (Fig. 3.3) of the mean abundances of species per station grouped according to location (Cape Point EBSA, TMNP MPA, Robben Island MPA), indicated a strong similarity between TMNP MPA and Robben Island MPA followed by a strong similarity between Cape Point EBSA and TMNP MPA. The strongest similarity in species abundance was seen when samples had a similar substrate type that is, rock/reef or sand/mud. These similar substrate types were often found in samples adjacent to each other. The cluster further reveals sand/mud benthic environments to possess a lower number of species present (Fig. 3.4). The Ski-Monkey Cluster analysis of the mean abundances of species per station grouped according to location (Cape Point EBSA, TMNP MPA, Robben Island MPA) (Fig. 3.5) indicated the strongest relationship to be between TMNP and Robben Island followed by Cape Point EBSA and TMNP MPA. This could be due to the TMNP MPA being linked to Robben Island MPA in the north and to Cape Point EBSA in the south hence it being the common denominator in the linkages. The Ski-Monkey cluster analysis based on substrate indicated a strong similarity between samples with rock/reef substrate followed by those with a sand/mud substrate (Fig. 3.6). The Drop camera and Ski-Monkey both indicated the strongest similarity to be between TMNP MPA and Robben Island MPA followed by Cape Point EBSA and TMNP MPA. Samples where the benthic environment were rock/reef showed a greater similarity to each other than samples which were solely reef or sand/mud, this could be a result of under-sampling of these substrate types (Fig. 3.4).

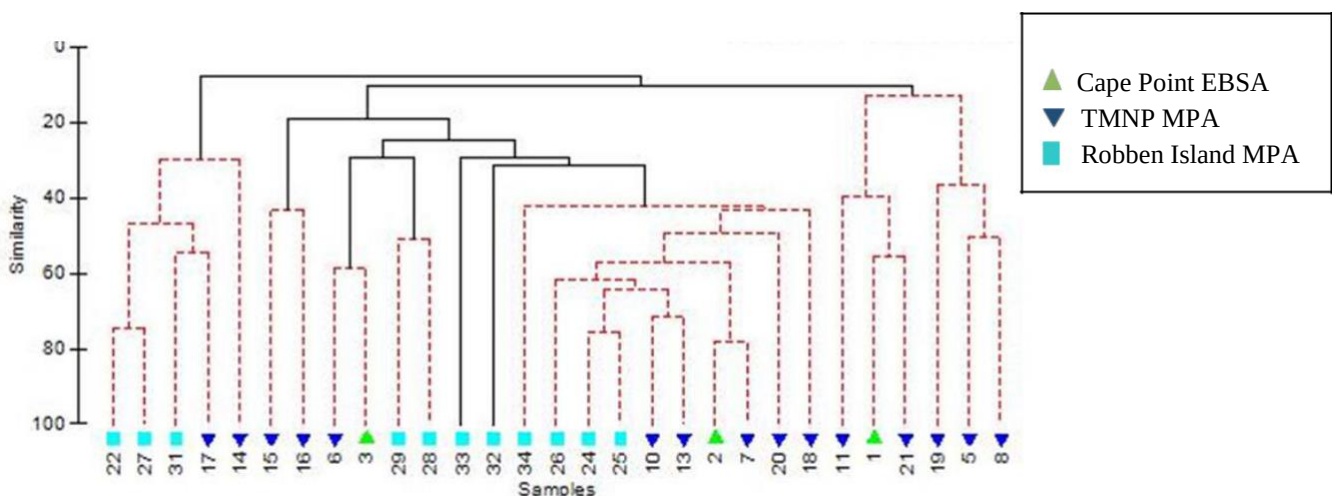


Figure 3.3 Dendrogram representing the cluster analysis of the mean abundances of species per station grouped according to location using the Drop camera.

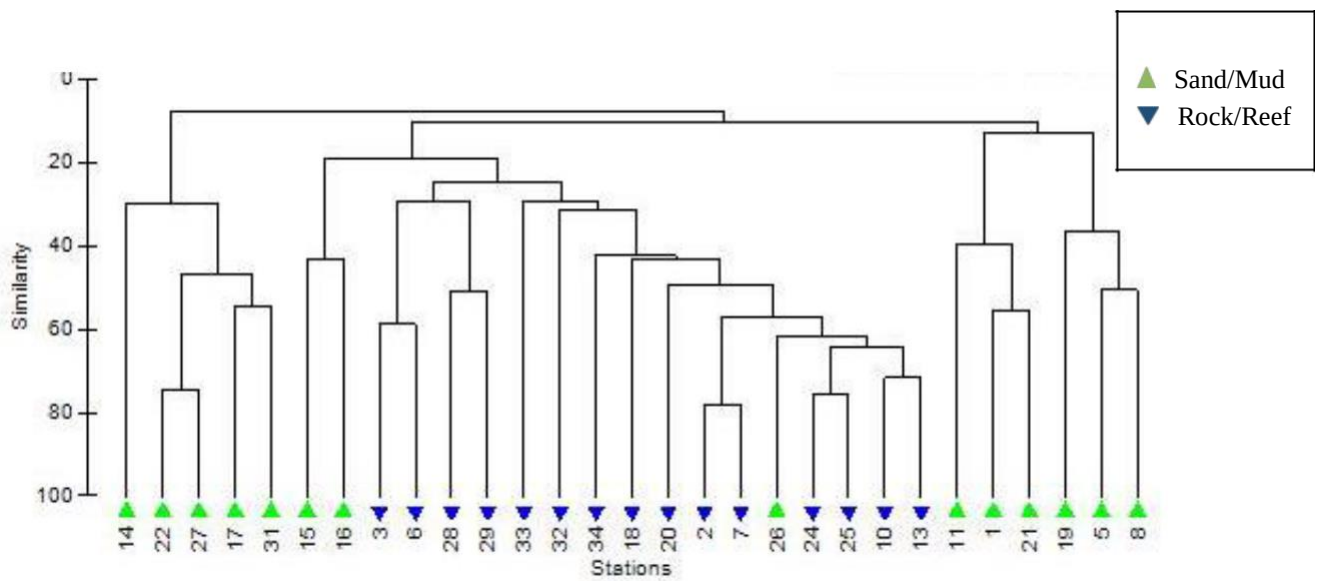


Figure 3.4. Drop camera Cluster analysis of the mean abundances of species per station grouped according to Substrate.

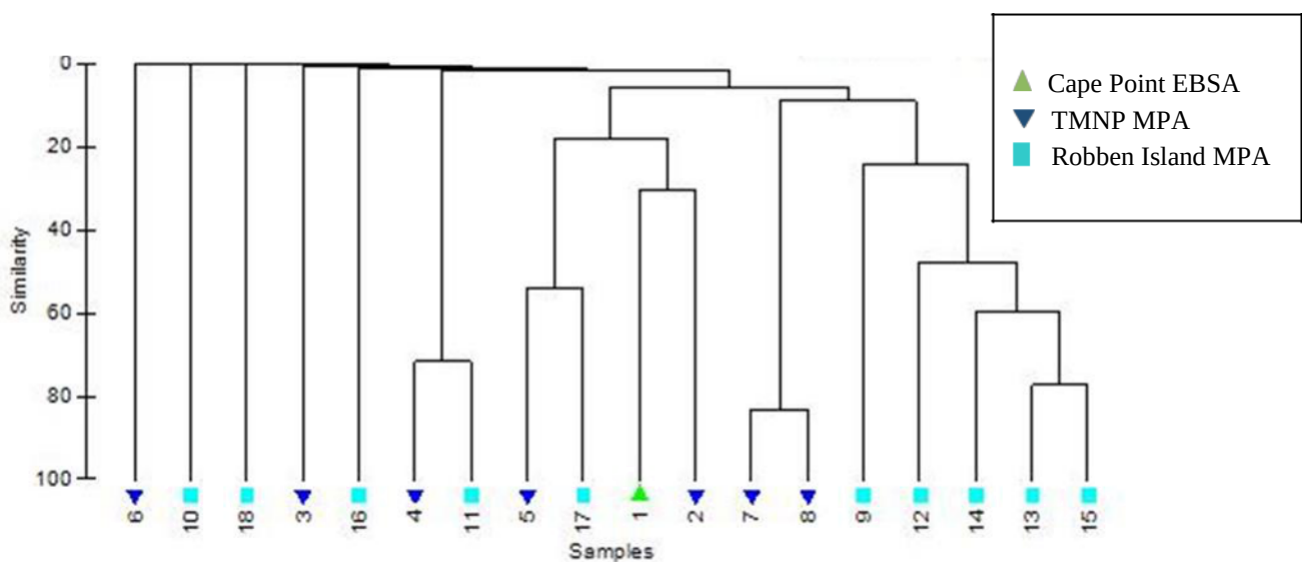


Figure 3.5. Ski-Monkey Cluster analysis of the mean abundances of species per station grouped according to location.

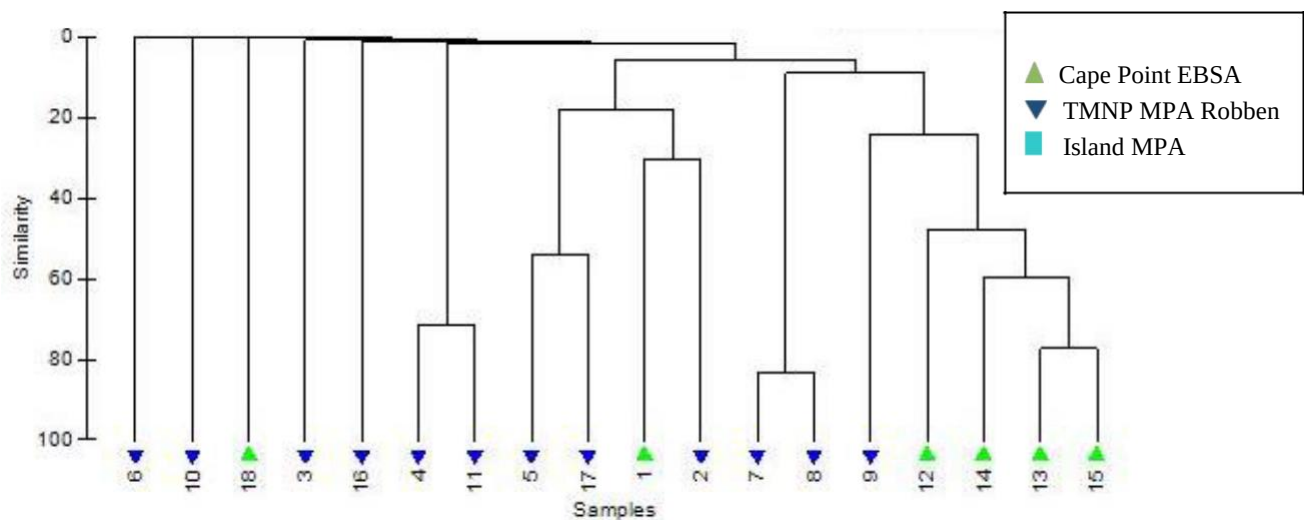


Figure 3.6. Ski-Monkey Cluster analysis of the mean abundances of species per station grouped according to Substrate.

The MDS ordination of the Drop camera species abundance per sample (2D Stress: 0.17) for substrate type and location (Fig. 3.7; Fig. 3.8) and MDS ordination of the Ski-Monkey species abundance per sample (2D Stress: 0.01) for substrate type and location (Fig. 3.9; Fig. 3.10), revealed natural groupings based on substrate type. In the case of the Ski-Monkey, the MDS revealed high clustering regardless of the location and substratum type hence a PERMANOVA was performed to further investigate this. (Appendix 6). The highest species abundance was found on the rock/reef substrate type. A PERMANOVA between the mean species' abundances for both location and substrate ($R^2 = 0.22$ and $R^2 = 0.21$) respectively did not reveal any significant differences. The uniformity across the MDS ordination for location and the interplay across the entire area indicates that the area is homogenous in terms of what we found. The MDS ordination (2D Stress: 0.17) of the abundance of species per sample for the Drop camera revealed that the three locations within the study site grouped separately, with some overlap between them. The MDS ordination (2D Stress: 0.01) for the Ski-Monkey and its cluster analysis of sampling stations' mean species abundance provided a clearer picture as the substrate types grouped separately with no overlap (Fig. 3.8). A PERMANOVA between location and substrate, as variables for the Ski-Monkey and Drop camera produced a R^2 value of 0.19 and $R^2 = 0.31$ respectively, indicating no significant difference between location and substrate type. The MDS, in summary, for both the Drop camera and Ski-Monkey indicate a

strong correlation between TMNP MPA and Robben Island MPA followed by Cape Point EBSA and TMNP MPA. Both also show species abundance is greater in the rock/reef substratum. Longitudinally from north to south the communities are the same but when split between benthic habitat types, the communities are different.

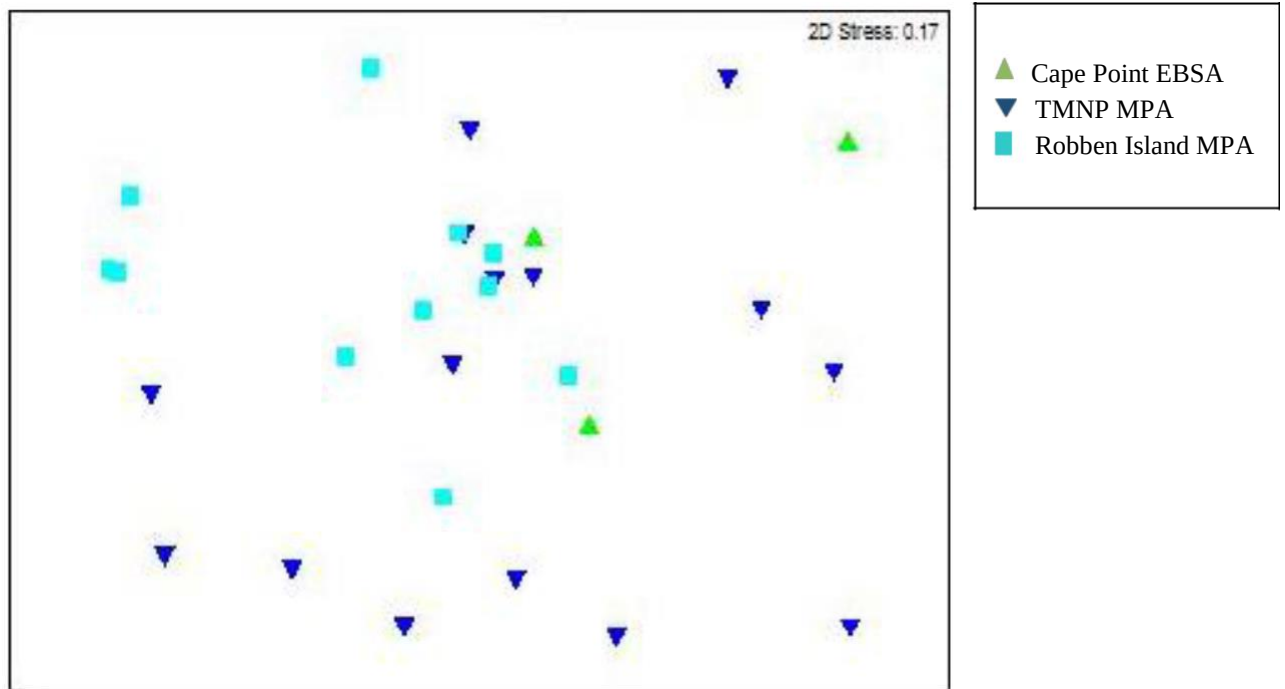


Figure 3.7. Drop camera MDS ordination showing similarity between species abundance in samples at different locations.

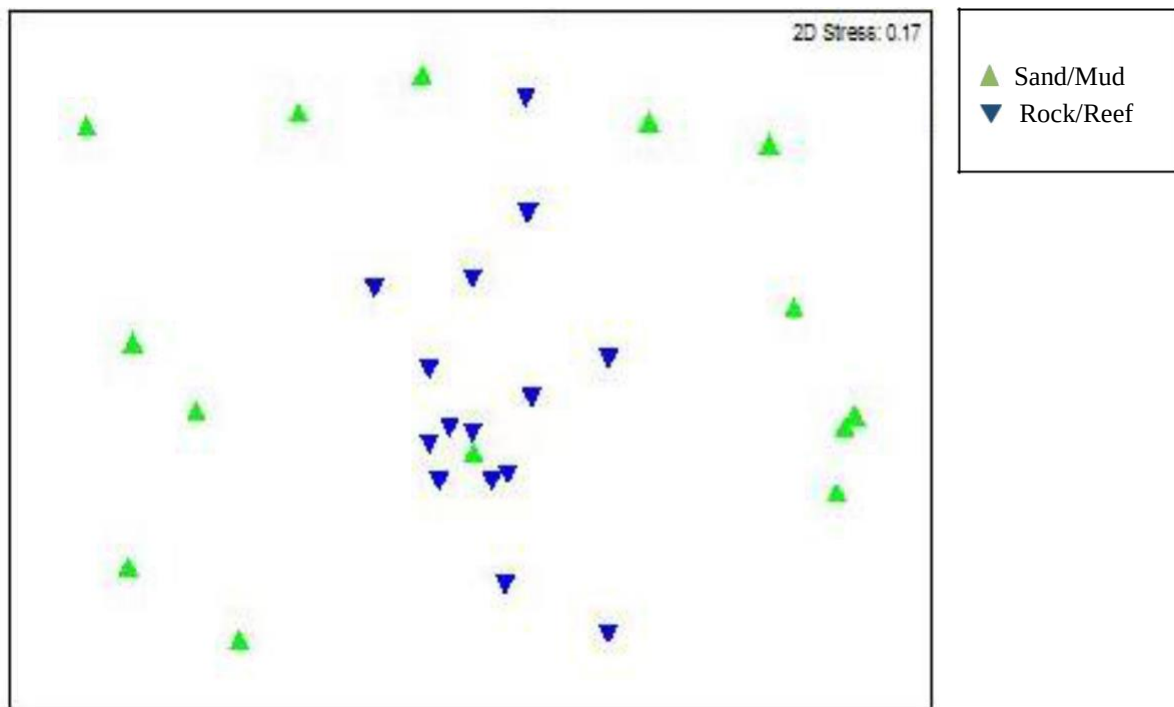


Figure 3.8. Drop camera MDS ordination showing similarity between species abundance in samples at different substrates

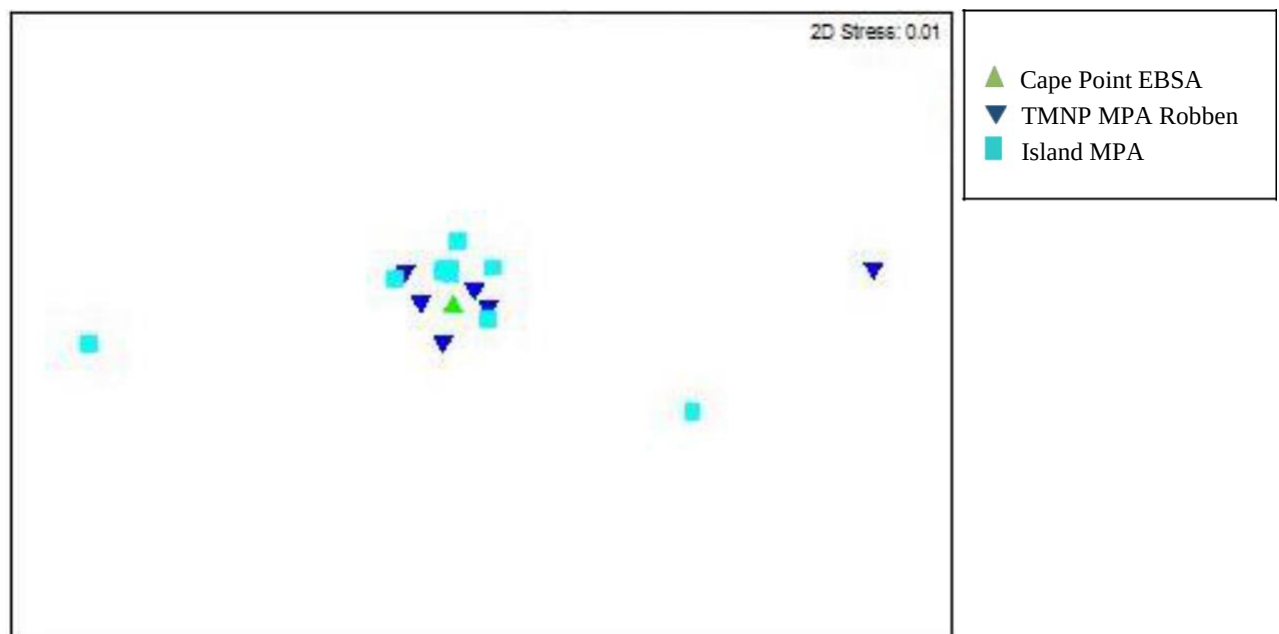


Figure 3.9. Ski-Monkey MDS ordination showing similarity between species abundance in samples at different locations.

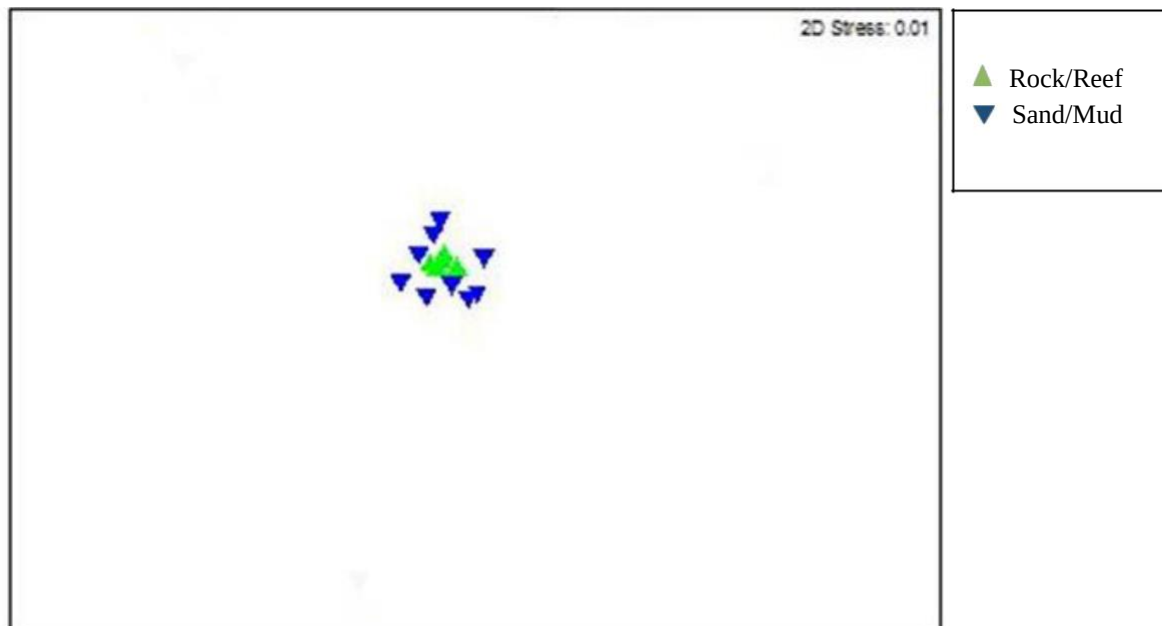


Figure 3.10. Ski-Monkey MDS ordination showing similarity between species abundance in samples at different substrates.

The ANOVA (Table 3.5) for total abundance per species indicated low similarities in samples of the same location but high similarity in samples of similar substrate type for both camera systems. The results revealed phyla with the highest percentage contribution for the Robben Island MPA, Cape Point EBSA and TMNP MPA to be Echinodermata (16.64 %), Mollusca (34.58 %) and Cnidaria (30.09 %), respectively. Ascidia (19.16 %) was most responsible for the high dissimilarity between the areas. The ANOVA procedures performed on the species abundance data for both camera systems revealed a higher dissimilarity than similarity in species abundance between stations according to both substrate type and location variables. Little to no species were present in sand dominated sampling stations (Fig. 3.16; Fig. 3.17). Rocky habitats were almost always inhabited by sessile benthic organisms. The mean species percentages were higher when there is a combination of rock and sand. The graph indicated that the Ski-Monkey is very restricted to sampling mainly sandy benthic environments in comparison to the Drop camera which can sample a greater variety of benthic environments. These are of course due to the operational differences which exist between the two, this will be discussed in detail in a closing chapter. The PCA's (Figs. 3.12 – 3.15) revealed depth to be a positive variable, and was the variable most responsible for the grouping of samples. As temperature decreased, it had less influence on the grouping of sites. Temperature decreased from shallow to deep stations and the Draftsman Plot (Fig. 3.11) also confirmed this. This provides further insight on how the environmental variables influenced the structure/grouping

of the samples. Depth and substrate appear to be the most important variables that are influencing the community structure of TMNP MPA and Robben Island MPA. The samples from Cape Point EBSA, although closely grouped with that of TMNP, appeared to be more influenced by longitude and temperature.

3.4 Community Composition

Cnidaria displayed the highest total abundance (N=70 individuals) per sample and Ascidia the lowest (N=5 individuals) within the study area. The two camera systems showed a large difference in species diversity between stations (Fig. 3.20) (Fig. 3.21) (Fig. 3.22). The mean abundance of organisms was highest in the Cape Point EBSA as highlighted by the Drop camera (Fig. 3.21). TMNP MPA yielded the highest number of representative phyla (Cnidaria, Porifera, Mollusca and Echinodermata) (Fig 3.20). In Cape Point EBSA, Mollusca (N=35 individuals) was the most abundant phylum. In TMNP MPA it was Cnidaria (N=30 individuals) and in Robben Island MPA it was Echinodermata (N=50 individuals). The Cape Point EBSA also had the station with the highest percentage cover of benthic species ($34.2\% \pm 21.51$ SD). The station with the lowest percentage cover of species ($0.20\% \pm 0.45$ SD) was located in the TMNP MPA. The stations revealing the highest abundance and percentage cover were stations that were rock/reef dominated and the station with the lowest percentage cover was sand/mud dominated as highlighted by the Ski-Monkey in Figure 3.21. The ANOVA procedure for similarity tests of mean abundance per samples, between the 3 locations indicated low similarity for the location but higher similarity for the substrate variable. Location and substrate were the variables which caused the highest similarity amongst the Robben Island MPA stations. The ANOVA procedures also revealed that dominant phyla Echinodermata, Mollusca and Cnidaria were most responsible for characterising similarity within the study area as they had the highest contributions in relation to abundance and evenness.

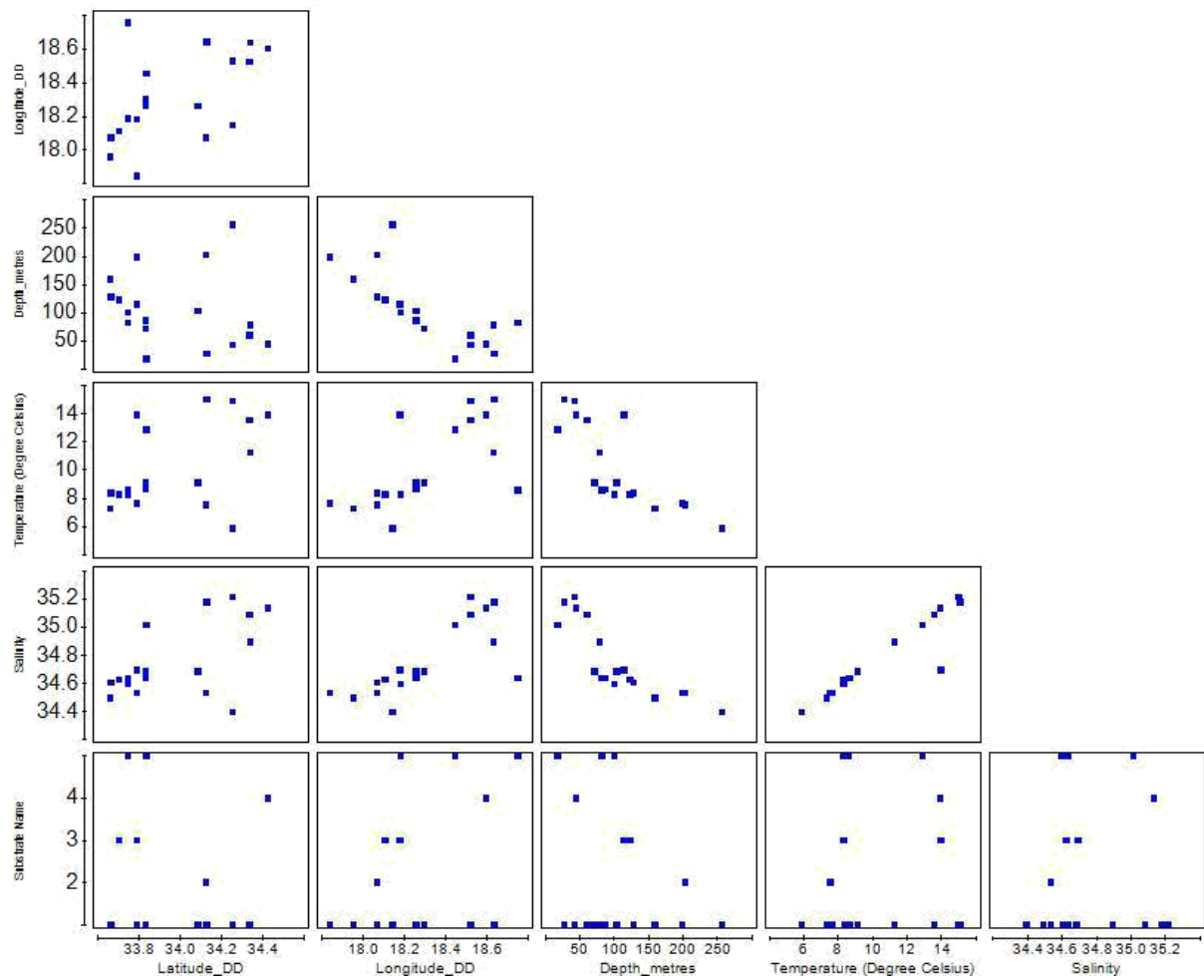


Figure 3.11. Draftsman plots indicating correlation between Temperature (Degree Celsius), Salinity, Depth and Location across all sampled stations.

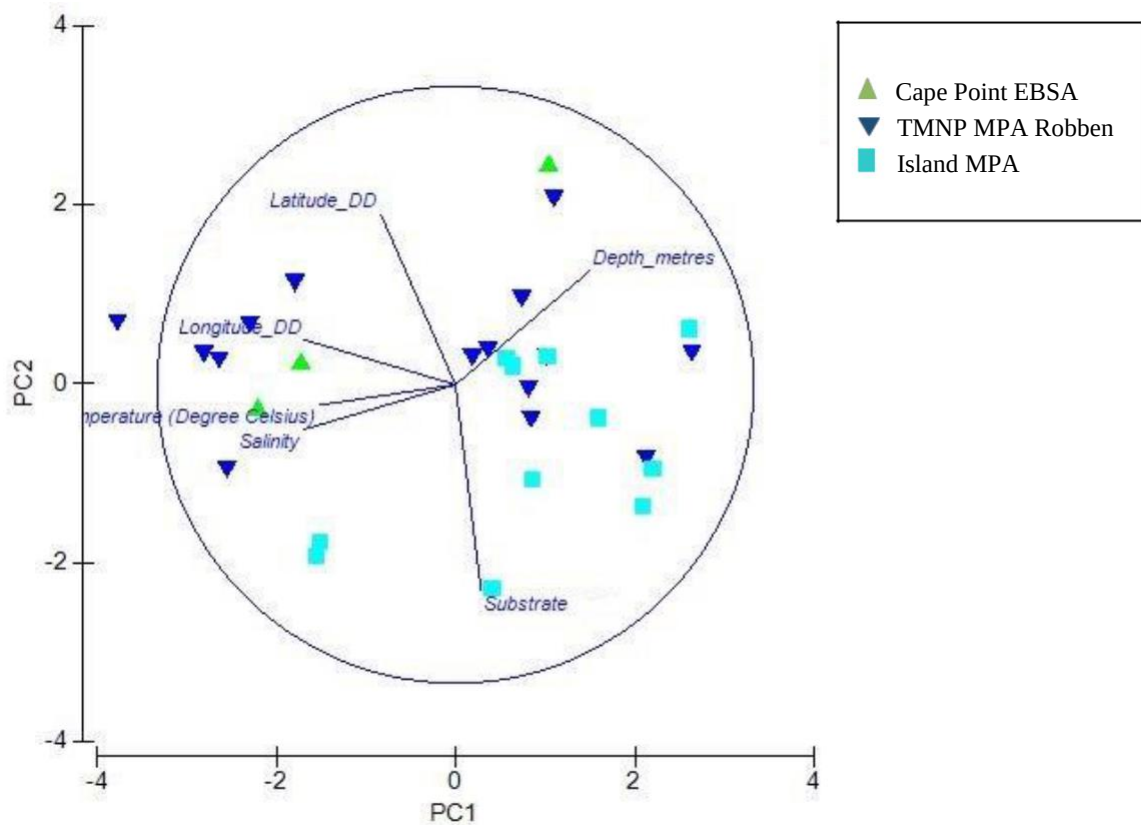


Figure 3.12. Drop camera PCA for location indicating Cape Point EBSA, TMNP MPA and Robben Island MPA using benthic biota abundance data with Depth, Salinity, Temperature, Substrate type, Latitude and Longitude as environmental variables. (PC1 37.4%; PC2 33.8%).

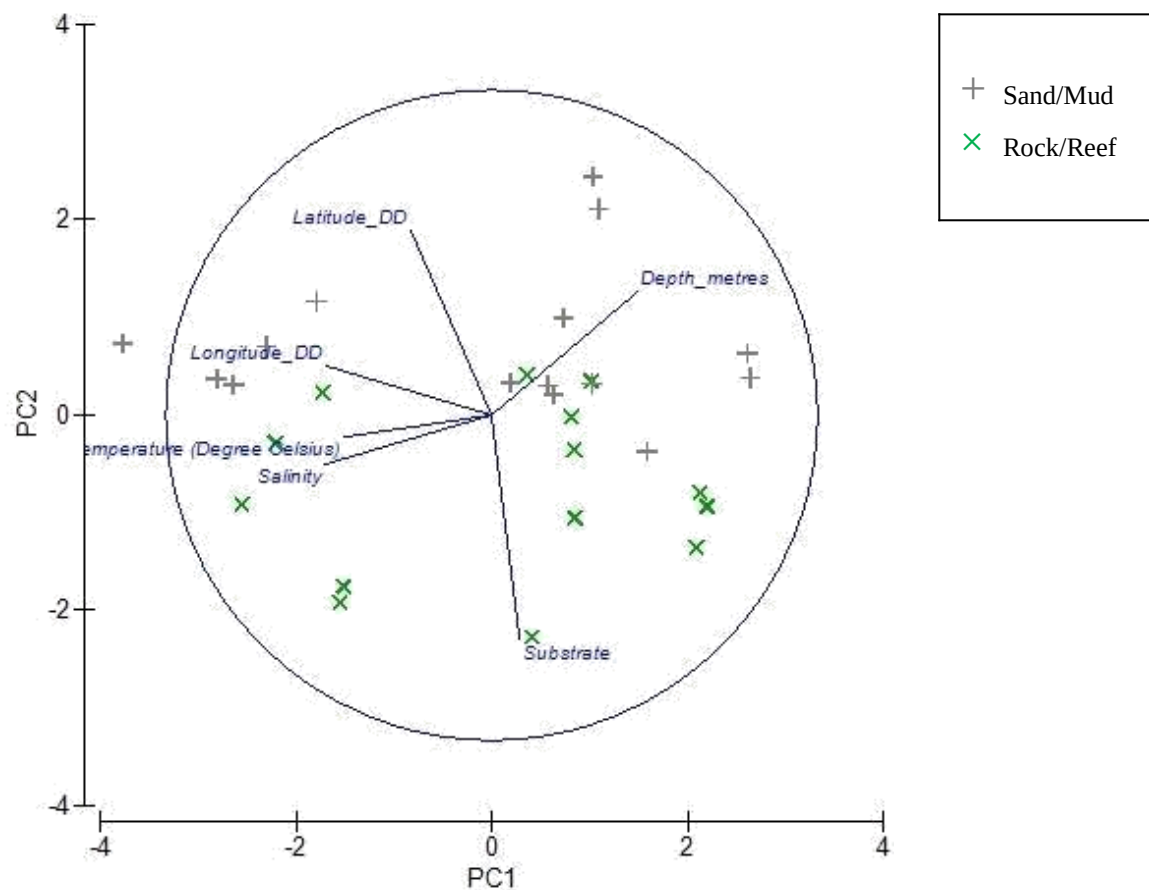


Figure 3.13. Drop camera PCA for Substrate indicating Sand/Mud and Rock/Reef using benthic biota abundance data with Depth, Salinity, Temperature, Substrate type, Latitude and Longitude as environmental variables. (PC1 37.4%; PC2 33.8%)

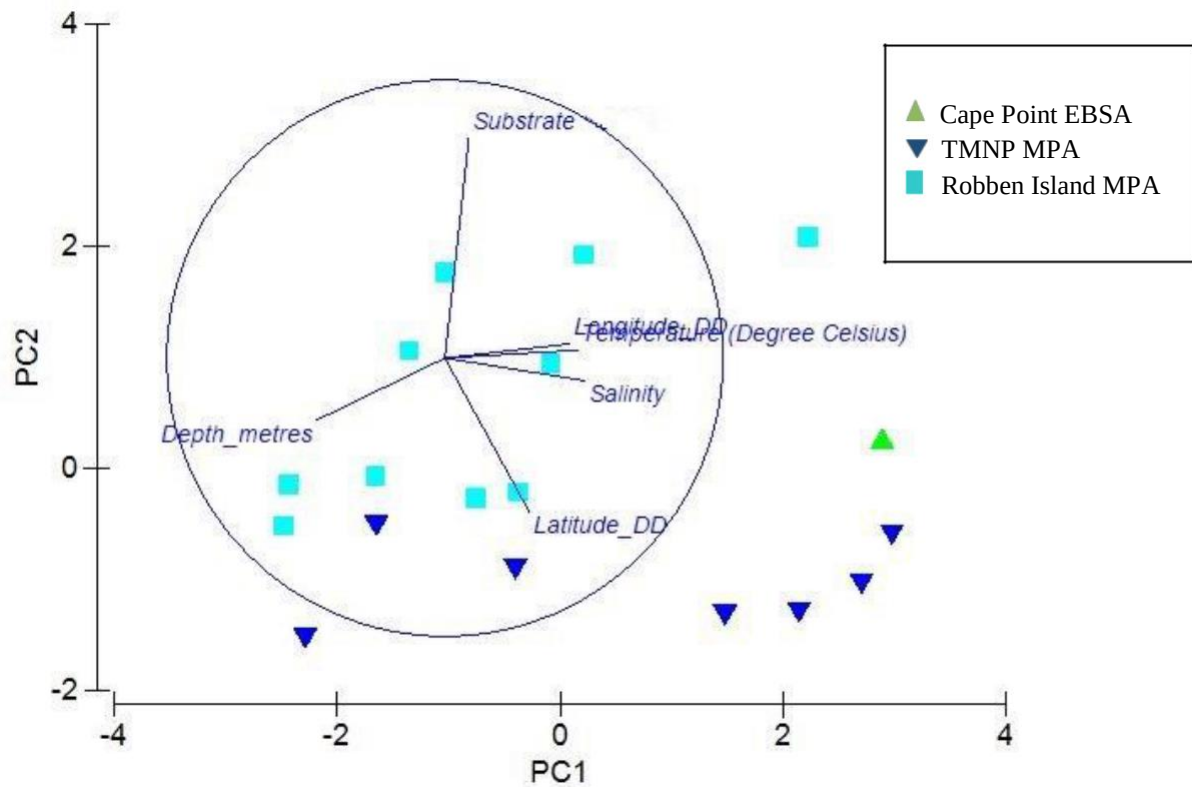


Figure 3.14. Ski-Monkey PCA for location indicating Cape Point EBSA, TMNP MPA and Robben Island MPA using benthic biota abundance data with Depth, Salinity, Temperature, Substrate type, Latitude and Longitude as environmental variables. (PC1 51.9%; PC2 36.1%).

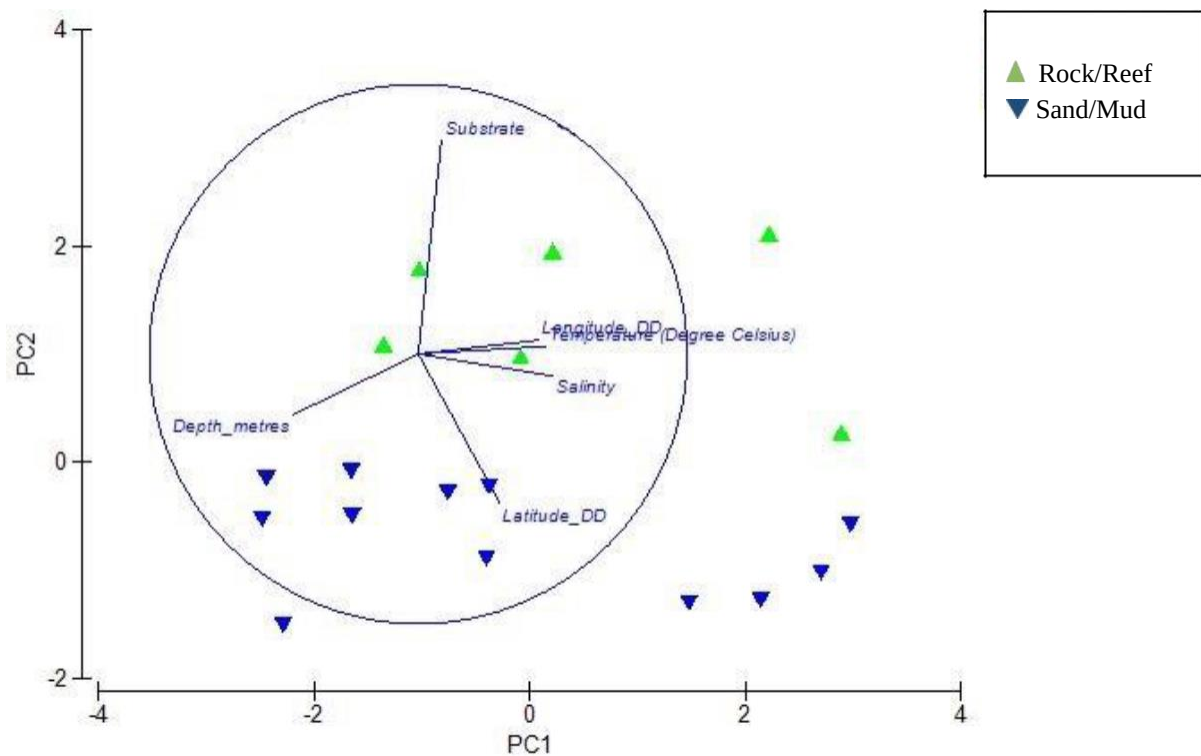


Figure 3.15. Ski-Monkey PCA for Substrate indicating Rock/Reef and Sand/Mud using benthic biota abundance data with Depth, Salinity, Temperature, Substrate type, Latitude and Longitude as environmental variables. (PC1 51.9%; PC2 36.1 %).

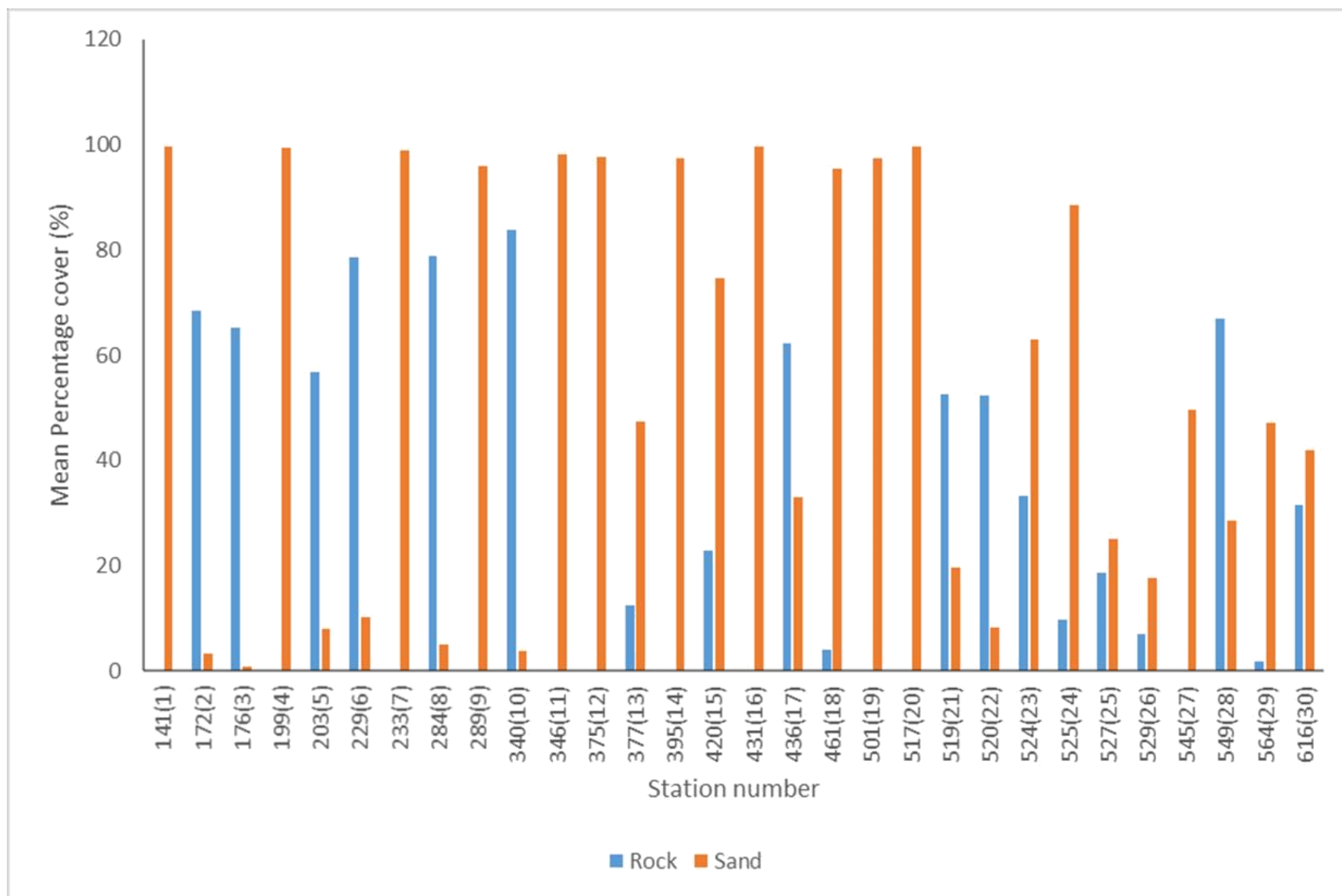


Figure 3.16. Drop camera percentage cover (%) for rock and sand substratum.

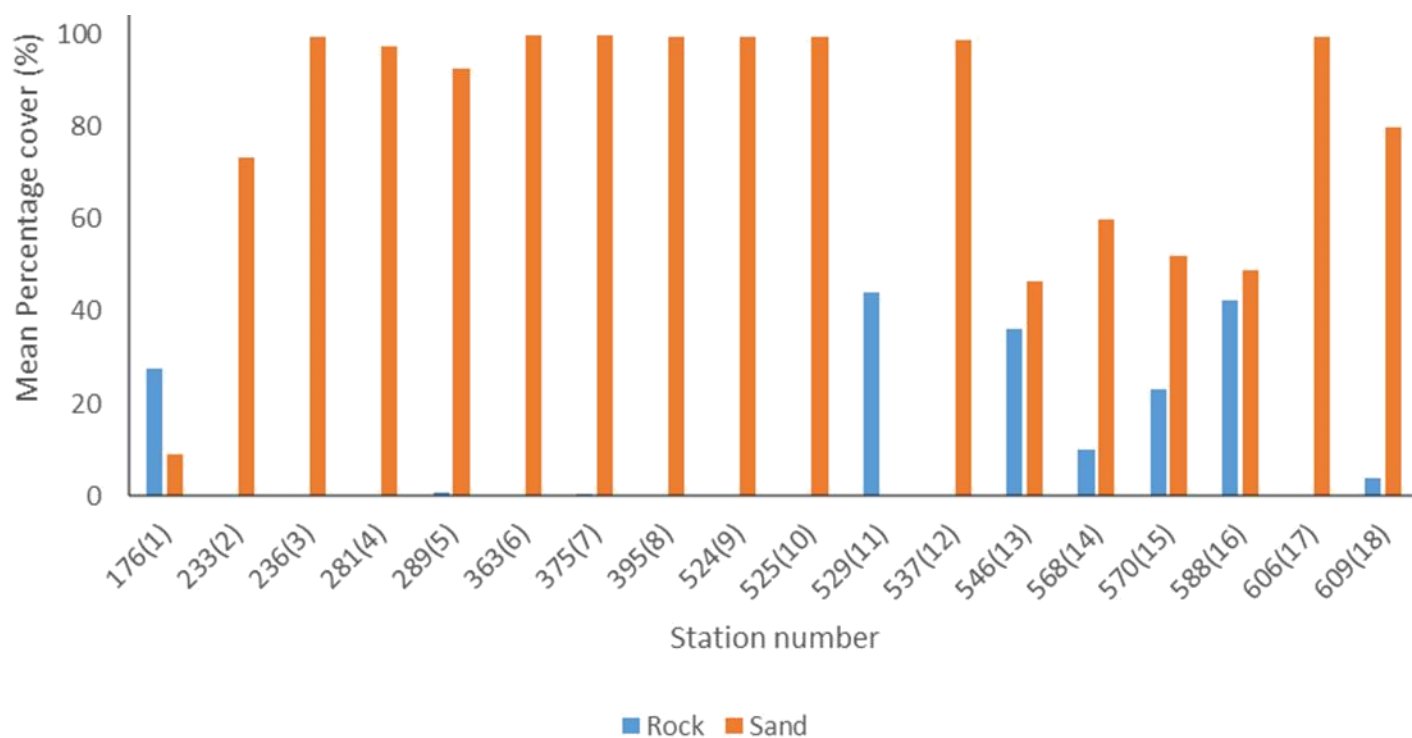


Figure 3.17. Ski-Monkey percentage cover (%) for rock and sand substratum.

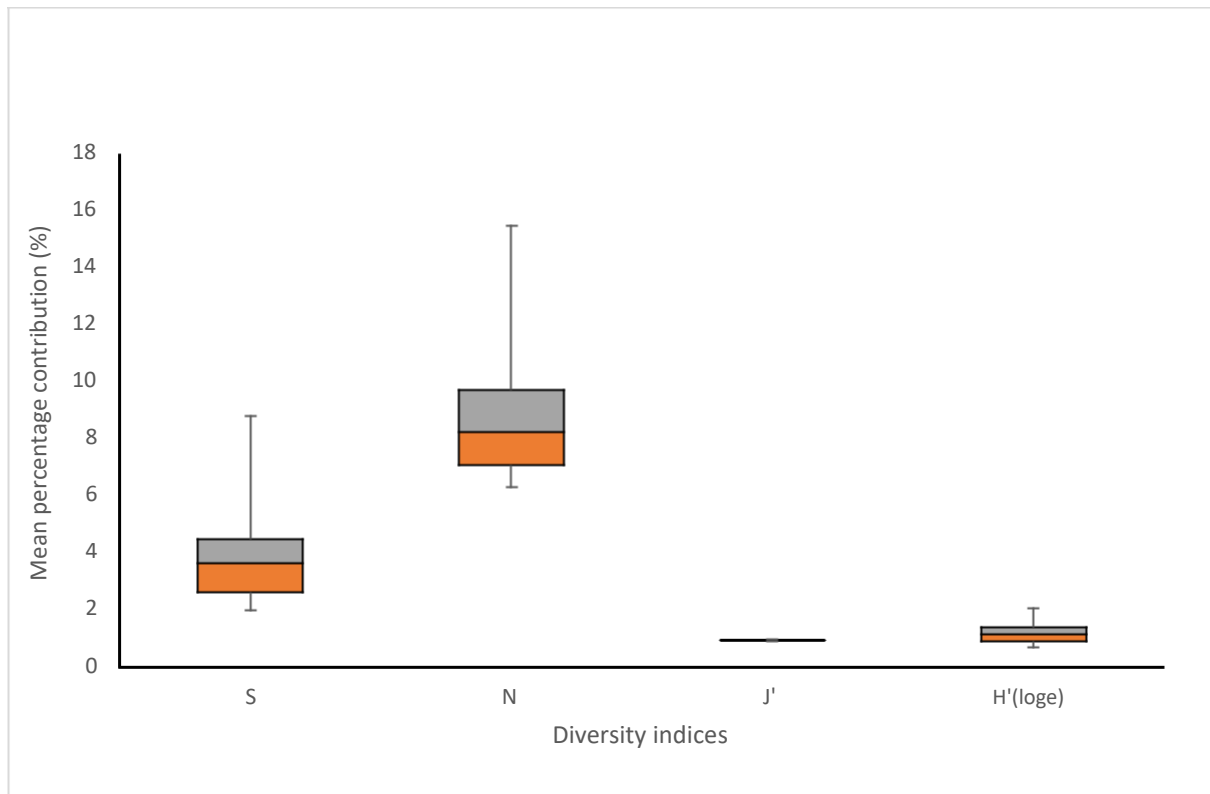


Figure 3.18. Graph showing the mean and standard deviation of Species richness (S), Abundance (N), Evenness (J') and Diversity (H'(Loge)) for all Drop camera stations.

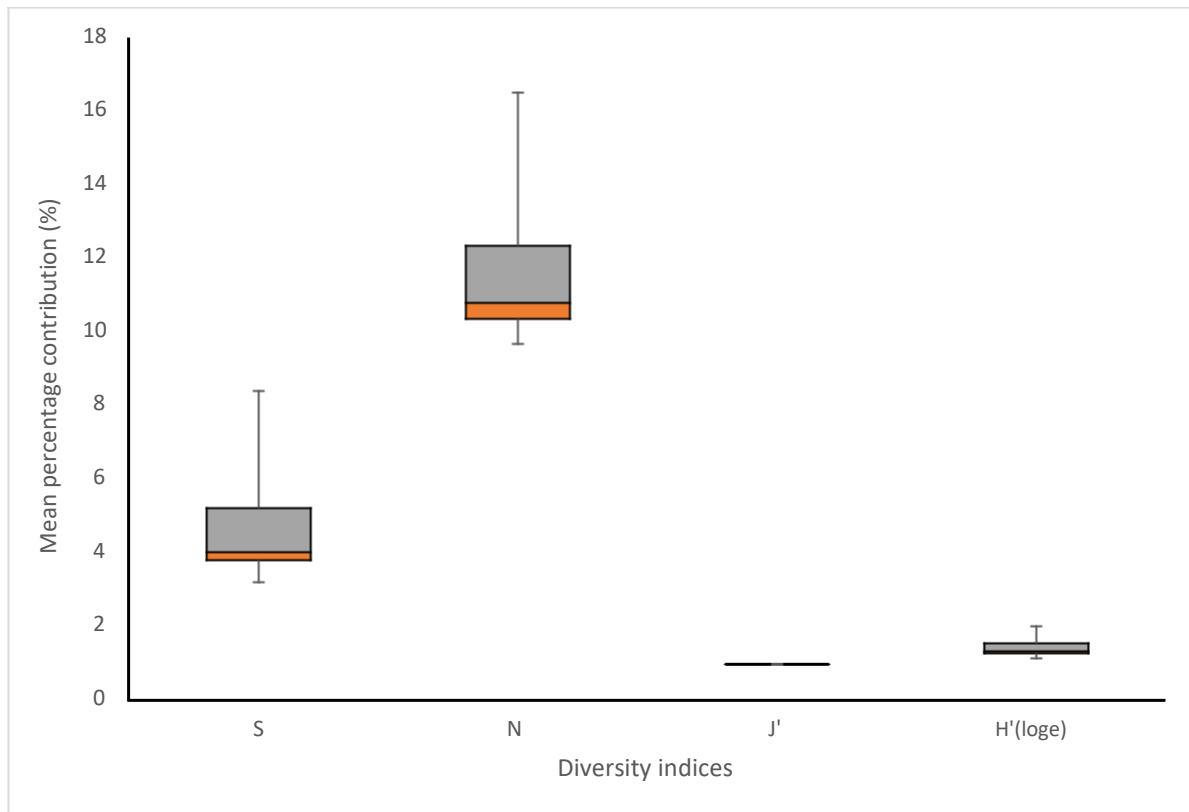


Figure 3.19. Graph showing the mean and standard deviation of Species richness (S), Abundance (N), Evenness (J') and Diversity (H'(Loge)) for all Ski-Monkey stations.

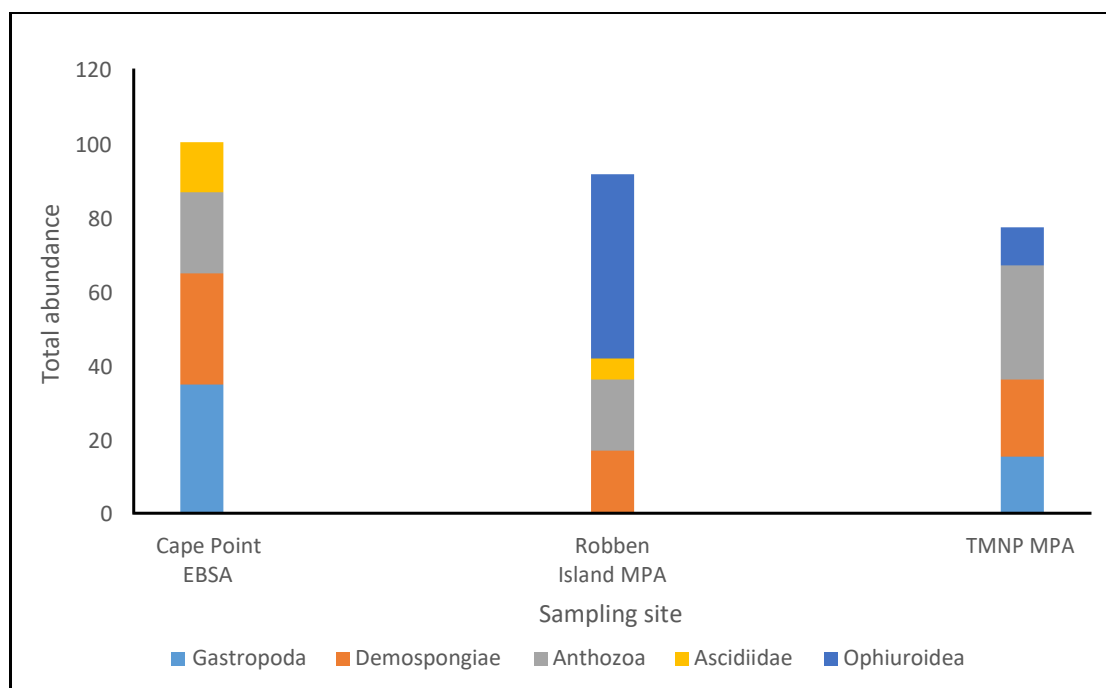


Figure 3.20. Drop camera total abundances of dominant phyla within the different sampling sites.

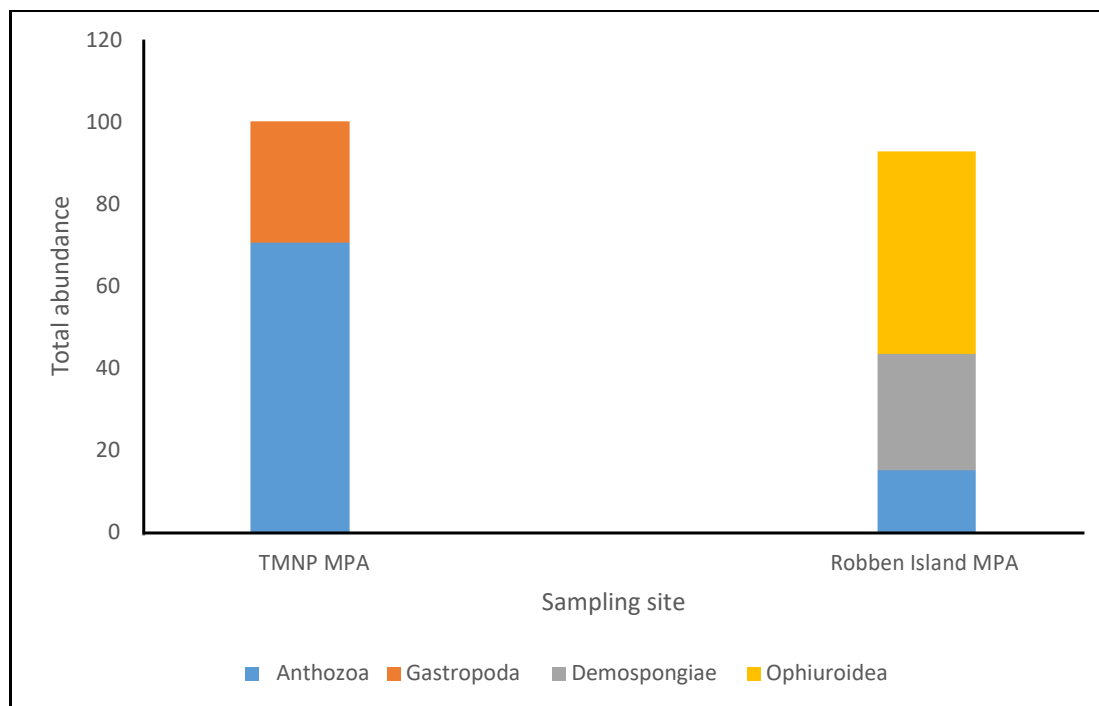


Figure 3.21. Ski-Monkey total abundances of dominant phyla within the different sampling sites. (Low successful Ski-Monkey tows for Cape Point EBSA).

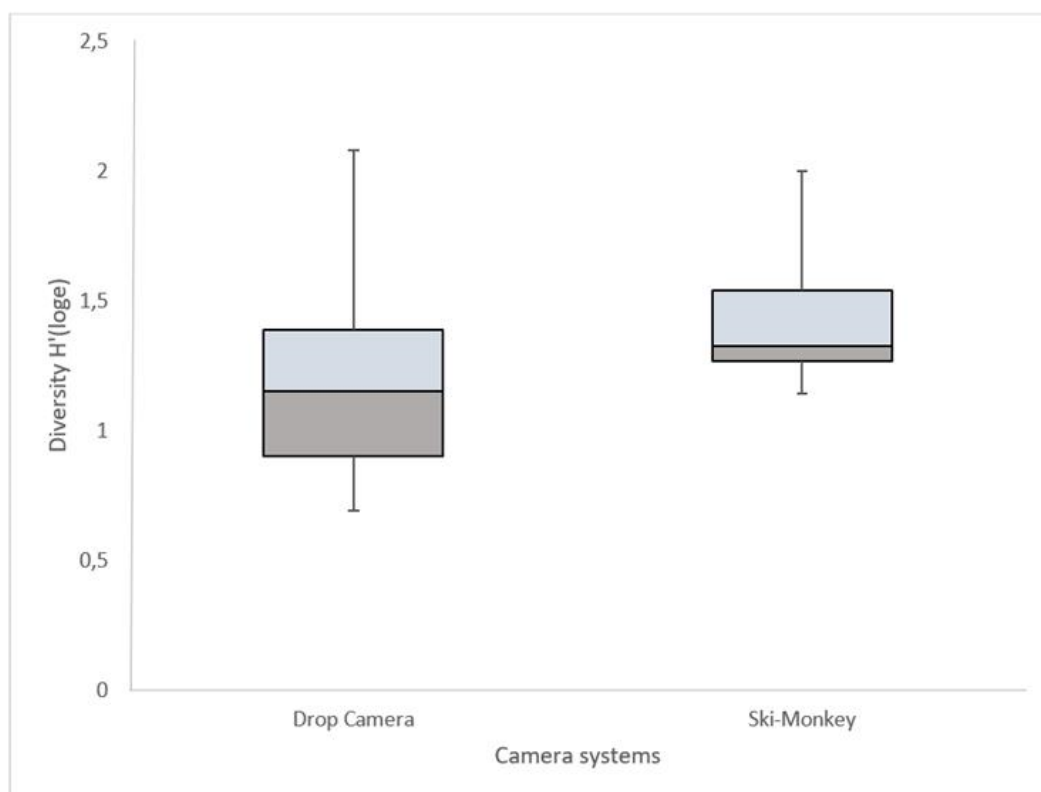


Figure 3.22. The mean and $\pm$ one standard deviation from the mean of the species diversity ($H'(\log e)$) between stations using the Drop camera and Ski-Monkey camera systems respectively.

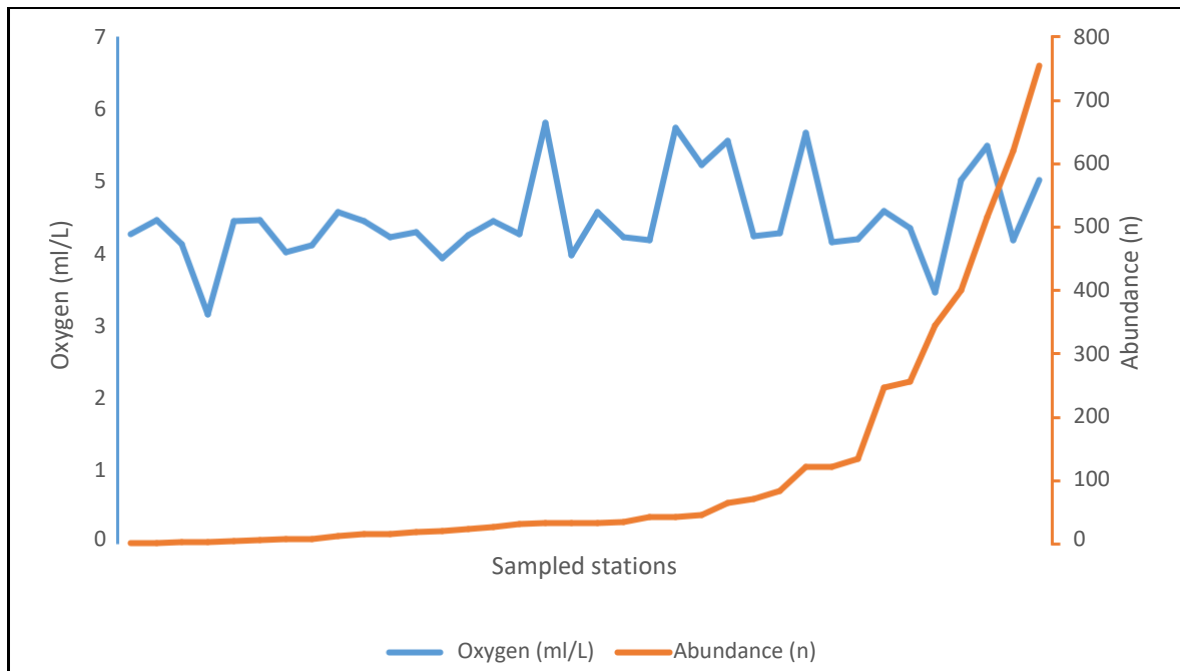


Figure 3.23. CTD results of measured oxygen (ml/l) at each station collected throughout the study area plotted against species abundance (n).

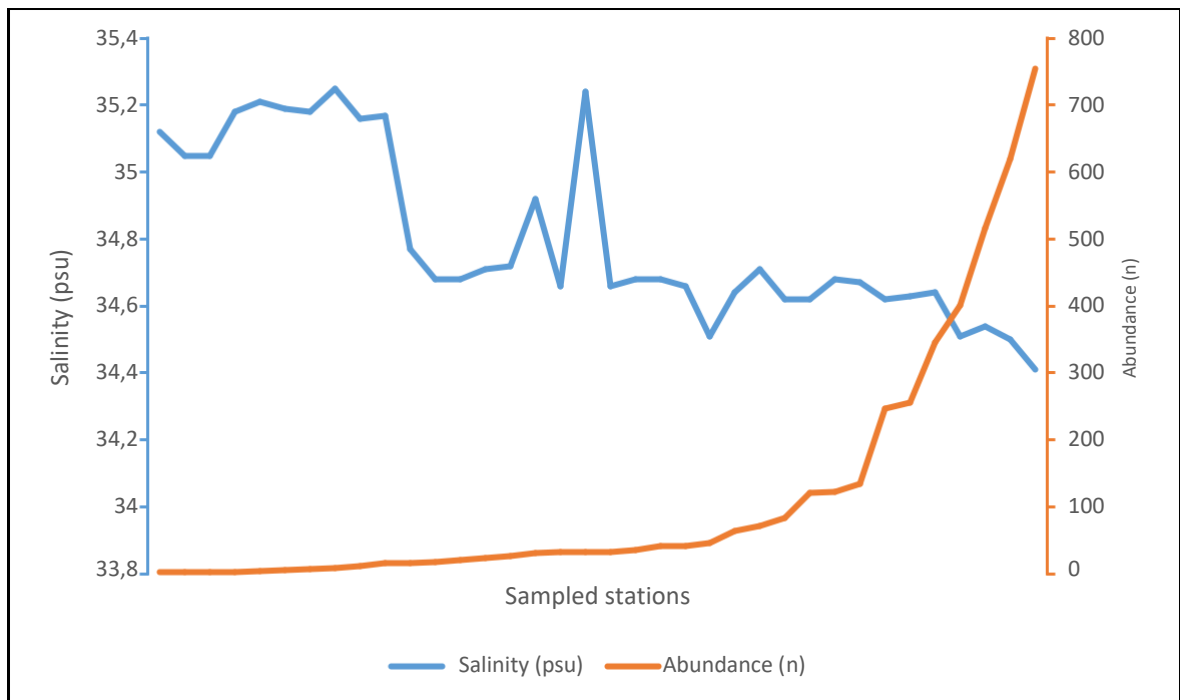


Figure 3.24. CTD results of measured salinity (psu) at each station collected throughout the study plotted against species abundance (n).

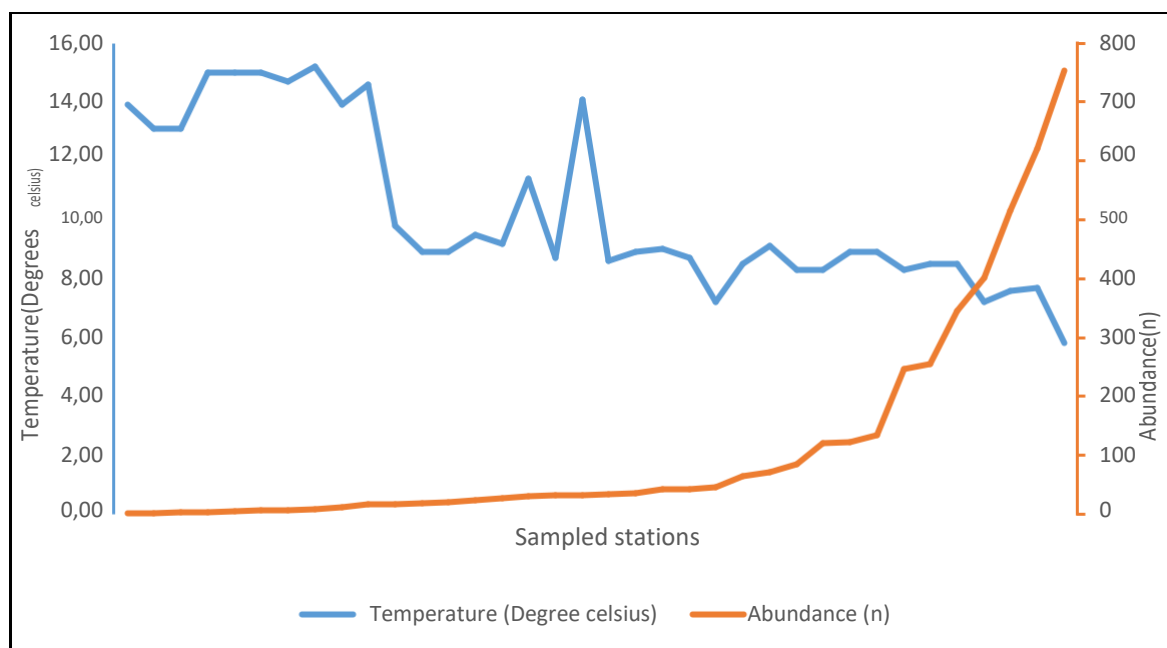


Figure 3.25. CTD results of measured temperature (Degree Celsius) at each station collected throughout the study plotted against species abundance (n).

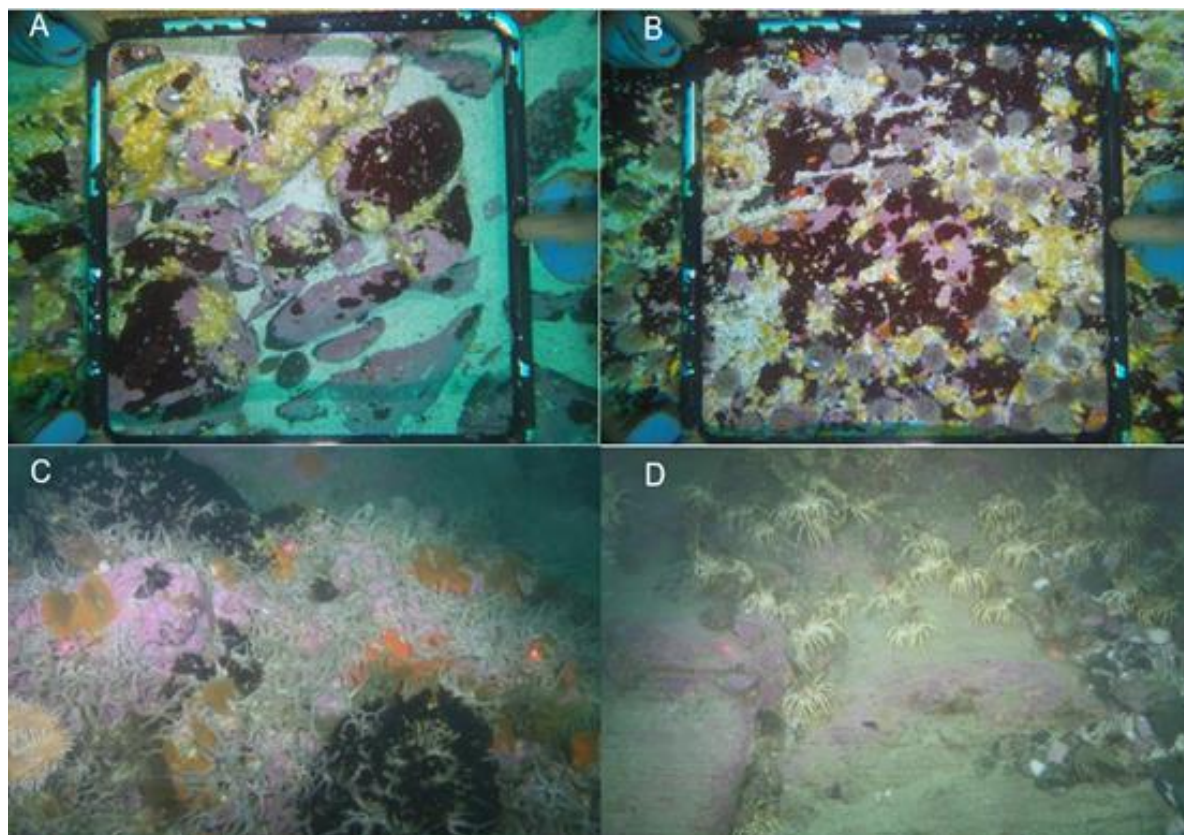


Figure 3.26. Drop camera images A and B, taken at 16m (Station SB527), depict patchy rocky-sand habitats that are home to sponges, coral aggregations, and sea urchin clusters. C and D:

Ski-Monkey photos of Station SB529 taken at 16 m reveal a significant number of rock lobsters as well as brittle star aggregations.

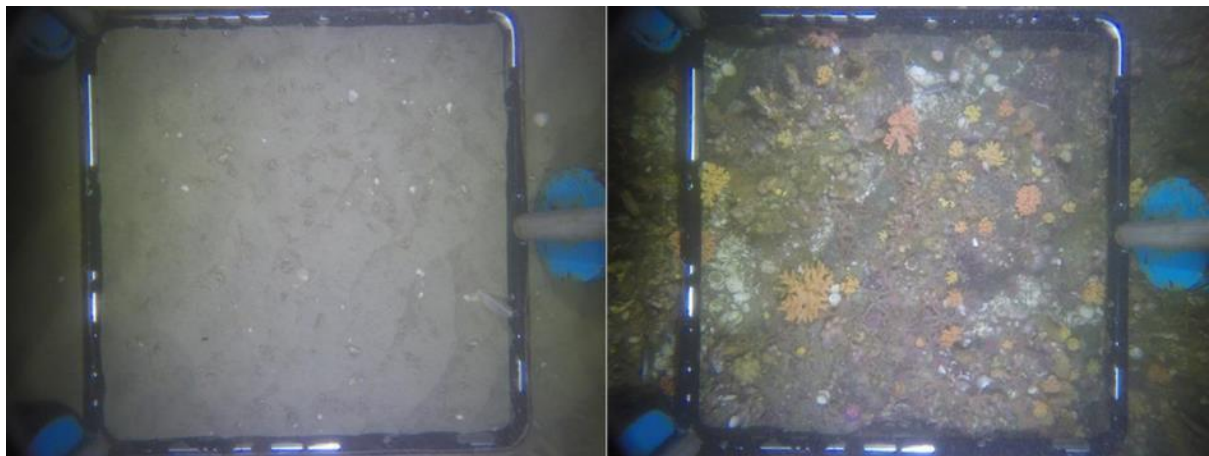


Figure 3.27. Drop camera visuals at 45-46m (Station SB616) showing sandy habitat types with an individual sea pen (left) and a rocky habitat type with aggregations of coral and brittle stars (right).

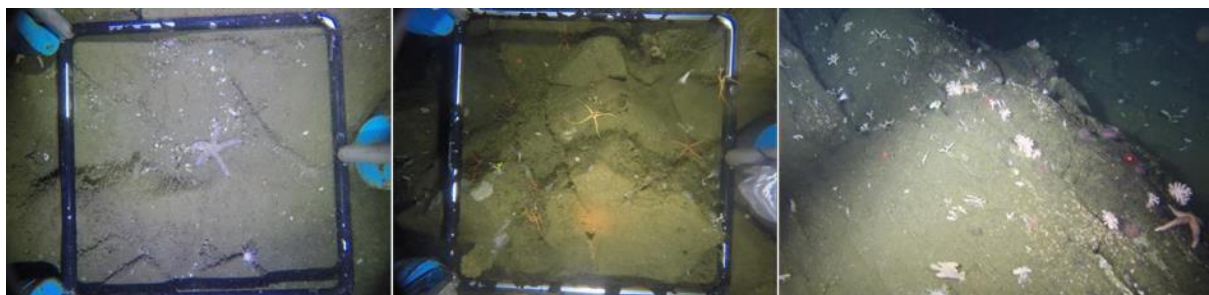


Figure 3.28. Drop camera photos show a brittle star and coral species living on boulders at 130m (Station SB520) and a spiny starfish on sandy/semi-rocky substrate at 86m (Station SB524). At station SB568, a Ski-Monkey image (far right) displays a variety of taxa groups (Cnidaria, Crustacea, Echinodermata, Polychaeta, and Mollusca) adhered to a solitary boulder at a depth of 104m.



Figure 3.29. Ski-Monkey images at 54m (Station SB461) showing a diverse boulder environment of Cnidarians and Poriferans.

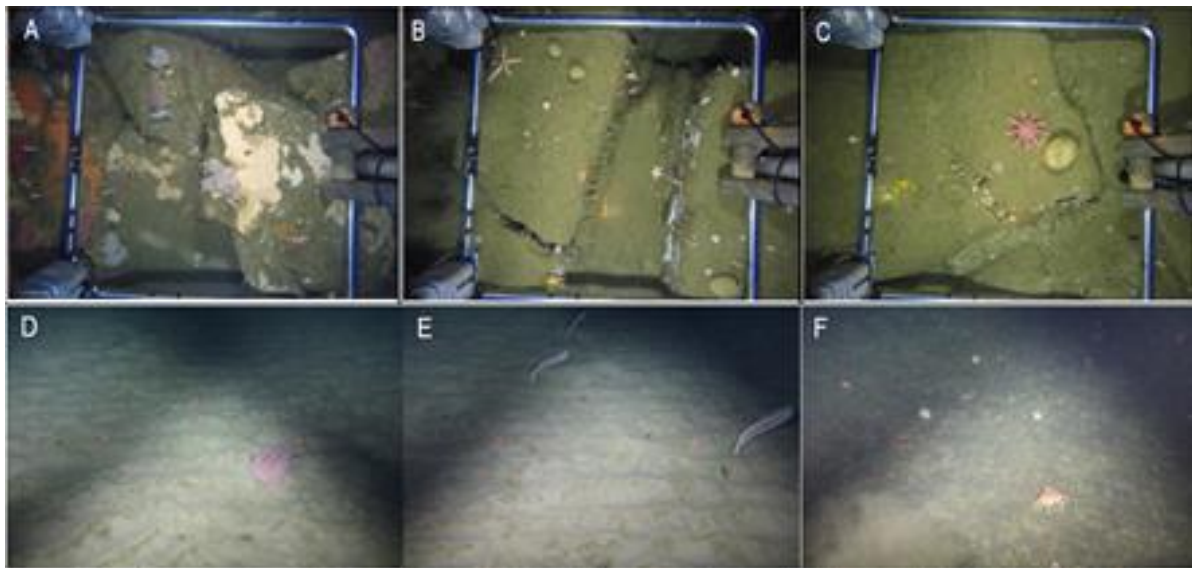


Figure 3.30. Images captured by the Drop camera at 93m (Station SB229) (A) and 117m (Station SB 284) (B and C) depict communities of starfish, ball sponges, hydroids, and lamp shells (B), as well as boulder habitats with soft coral and a variety of encrusting sponge species (C). Images from Ski-Monkey reveal sandy areas where cnidarians, including large sea fans (D), sea pens (E), and crustaceans, can be seen at 105m (Station SB395) (D and E) and 192m (SB363) (F).

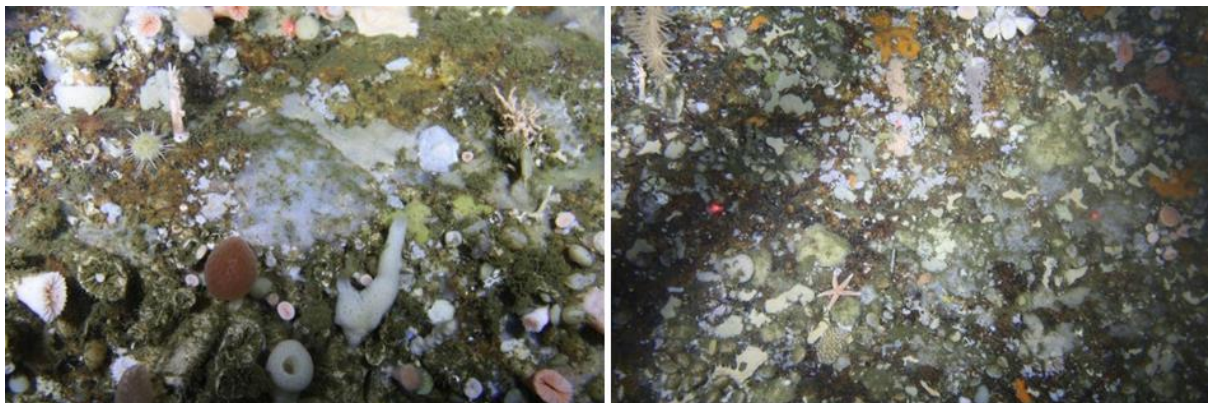


Figure 3.31. Ski-Monkey images at 124m (Station SB436) showing a diverse boulder environment of echinoderms, cnidarians, polychaetes and poriferans.



Figure 3.32. Sea anemones, sea fans, and hard coral can be seen in the Drop camera image on the left, while a rich diversity of feather stars, sea fans, and soft corals can be seen in the rocky habitat type at 27m (Station SB377) (Ski-Monkey image on the right).



Figure 3.33. A rich diversity of sponges and coral communities were visible on reefs at 44m (Station SB176) in both Drop camera (A) and Ski-Monkey (B) visuals. An image from the Ski-Monkey (C) depicts an assembly of screw shells on a sandy patch at 42m (Station SB289).



Figure 3.34. Both the Drop camera (left) and Ski-Monkey (right) visuals show the species poor sediment habitats at 94 m (Station SB177). A tiny white nudibranch (mollusk) is visible on the left.

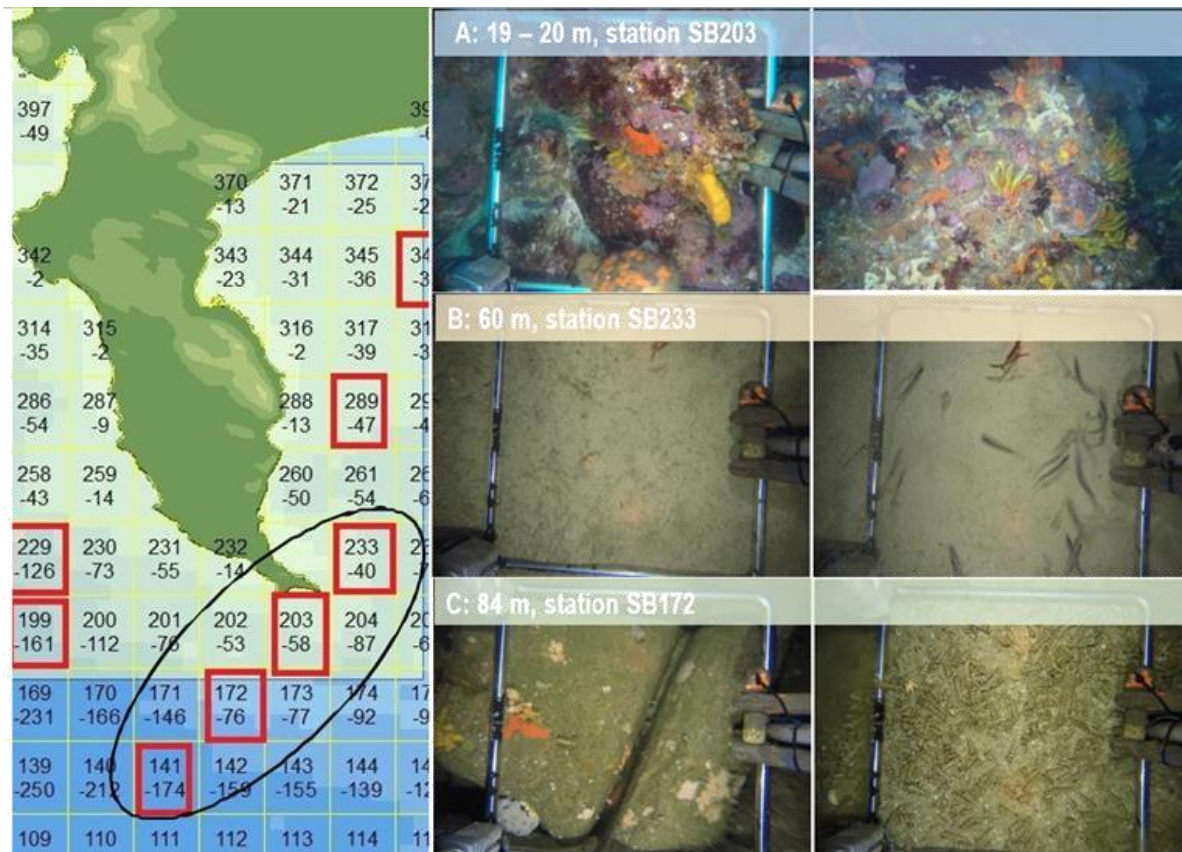


Figure 3.35. Visuals showing benthos and associated habitats at the shallow (A), middle (B) and deep waters (C) of Cape Point EBSA and surrounds.

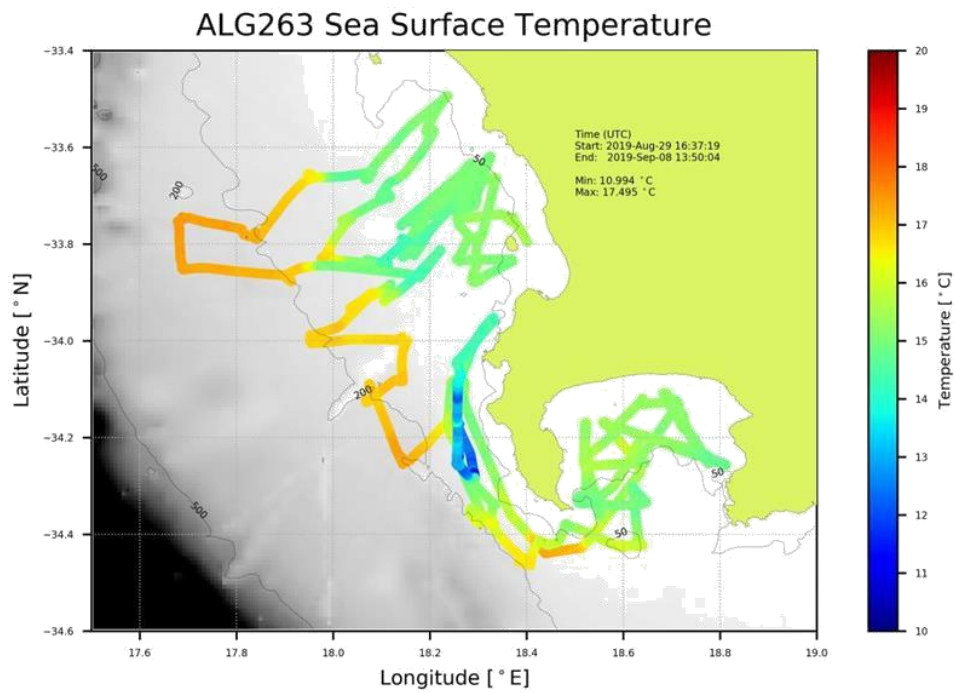


Figure 3.36. The study area's map displays the measured sea surface temperature, which is shaded in colour. The same image displays both bathymetry (grayscale and shaded) and isobaths (contoured-black) (S. Bergman, physics 2019).

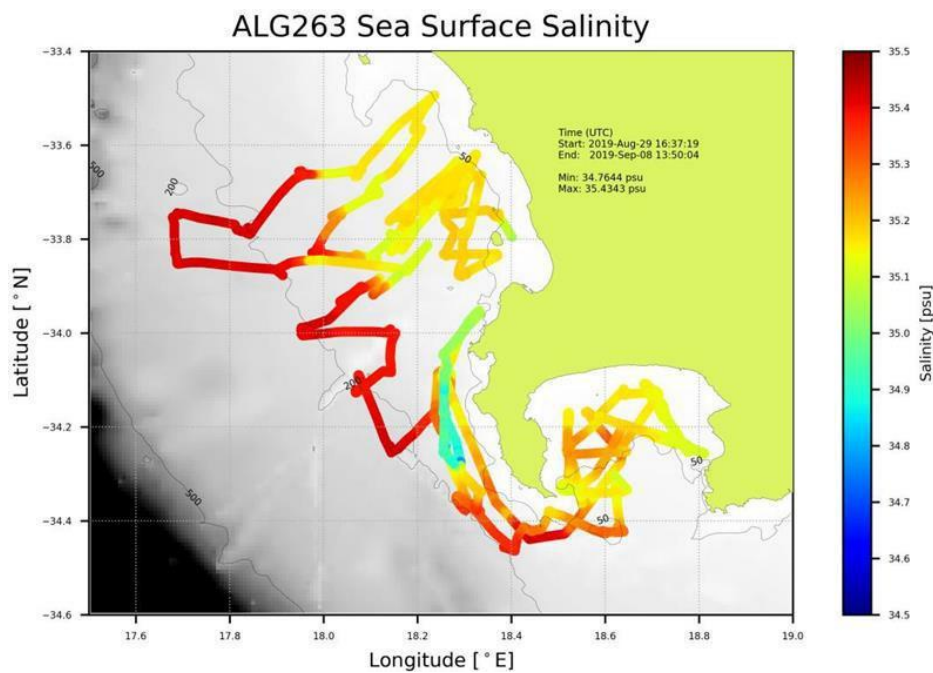


Figure 3.37. The study area's map displays the measured sea surface salinity, which is shaded in different colours. The same image displays both bathymetry (grayscale and shaded) and isobaths (contoured-black) (S. Bergman, physics.2019).

CHAPTER 4: DISCUSSION

The purpose of this study was to investigate the epifaunal benthic invertebrate communities and their associated habitats in the southern Benguela ecoregion. Specifically, this research aimed to examine the variations in species diversity across different areas within the ecoregion, and how factors such as habitat type, temperature, depth, salinity and oxygen levels influence their distribution and abundance. Additionally, we sought to determine how these findings can contribute to the establishment and expansion of Marine Protected Areas in the ecoregion. To achieve these objectives, we conducted investigations on diversity indices and the spatial patterns of these organisms (assemblage structure).

4.1 Sampling Effort

Prior to drawing conclusions from the results obtained, it was important to determine whether or not the sampling effort during the study was sufficient, as it is a known fact that the measurement of species richness for an area is mainly sensitive to the sample size (Foggo et al. 2003; Chao et al. 2014). By investigating the asymptote of a species accumulation curve, we can obtain a good measure of sampling effort as this provides the total number of species in the respective area as a measure of the sampling effort (Butler and Chazdon 1998; Chazdon et al. 1998). The graphs presented in Figure 3.1 and Figure 3.2 revealed an absence of an asymptote, which implied inadequate sampling effort. Since the asymptote of species accumulation curves fundamentally defines species richness within homogenous assemblages, the lack thereof may simply be indicative of a heterogeneous area and not an inadequacy in sampling effort (Chazdon et al. 1998). Accumulation curves were calculated across the entire datasets (samples from all locations grouped for each method). Within each method there was unbalanced sampling among locations (e.g. Ski-Monkey: Cape Point EBSA = 3, TMNP = 5, RI = 10), hence looking at the accumulations curves at the location level would help understand sampling completeness.

Due to the number of workable photographs obtained per station, the number of samples analysed and processed per station varied. The species accumulation curves produced indicated that < 25 photographs per station (Fig. 3.1) was sufficient to obtain a reasonable estimation of species richness in comparison to fewer photographs < 20 (Fig. 3.2) which provided lower species richness estimates. This further highlighted the sensitivity of species richness to sample size which at times causes surveys to be incomplete as a large number of species may go

undetected (Chao et al. 2014). More photographs per station were possibly necessary, but sampling offshore benthic environments is often both limited (weather conditions, time at sea, sampling equipment capabilities) and labour intensive (Colwell et al. 2012) as experienced in this study. Purvis and Hector (2000) stated that many diversity indices have been developed to convey the extent to which individuals are distributed evenly among species and that most but not all combine evenness with species richness, losing information by reducing two dimensions to one. Chazdon et al (1998) highlighted that species richness estimates are impacted by the species distribution within a specific area. A large amount of randomly distributed species within an area along with a high evenness causes for a quicker accumulation growth curve in comparison to species found to have a patchy distribution (Chazdon et al. 1998). A low number of dominant species in an area has proven to be linked with high evenness (Magurran 2004; Hewitt et al. 2008). A greater number of sampling stations and number of samples for this study would have been preferred but the challenges faced did not allow for this. The sampling restrictions along with the challenges faced (unreliable bathymetry data, weather conditions, time at sea, sampling equipment capabilities) increased the difficulty of data collection and overall species representation within the study area. This would have influenced the certainty around these observed patterns, so the results should be viewed with caution. However, the data are still valuable as very little is known about the target organisms within the study area.

4.2 Diversity Indices

The patterns within the MPAs were analysed by examining the species richness (S), abundance (N), diversity (H') and evenness (J') of the organisms recorded in the study area (Fig. 3.18) (Fig. 3.19). The global range of the Shannon-Wiener index is narrow, ranging from 1.5 to 3.5 (Magurran 2004). The obtained species diversity value (Appendix 12.2 and Appendix 12.3) fall slightly below this range ($H' = 1.20$), suggesting that the species diversity of the area is relatively low on a global scale. This could also be linked to the low sample size that led to low diversity. However, previous studies conducted adjacent to the TMNP MPA and its surrounding areas have also reported Shannon-Wiener diversity index values that fall below the global range of 1.5 to 3.5 (Roberson et al. 2015; Sanguinetti 2013). It is important for areas, particularly MPAs or protected areas, to have relatively high species diversity and evenness, as higher diversity promotes a healthier ecosystem (Thorn-Miller 1999; Purvis and Hector 2000). The area was also found to have high habitat heterogeneity according to Brill and Raemaekers (2013), and as described by Gotelli and Colwell (2001), more diverse habitats may

support more distinct species assemblages, thereby increasing the diversity of an area. The species diversity, richness and abundance across the sampling sites did not show significant differences. However, it should be noted that the results may have been affected by the unequal number of stations. The higher percentage of sand cover across stations, as described by Thrush and Dayton (2002), may indicate a different habitat type that supports fewer macro-benthic species compared to stations dominated by hard substrata (Atkinson et al. 2011).

4.3 Assemblage Composition

Comprehensive studies at a community level are necessary to make informed decisions about the organisation of benthic assemblages in marine environments (Sink 2016; Garrabou et al. 2002). To investigate community composition within an assemblage and visually represent the relationship between samples, a multivariate analysis such as Multi-dimensional scaling (MDS) ordinations can be used. The Drop camera MDS ordinations of both location (Fig. 3.7) and substrate (Fig. 3.8) species abundance data produced high stress values, suggesting that the data points are close to being randomly dispersed, which can lead to misleading interpretation according to Clark and Warwick (2001). The Ski-Monkey Cluster analysis of mean abundances of species per station grouped by location (Cape Point EBSA, TMNP MPA, Robben Island MPA) (Fig. 3.5) indicated that the strongest relationship was between TMNP and Robben Island, followed by Cape Point EBSA and TMNP MPA. This could be due to the TMNP MPA being linked to Robben Island MPA in the north and to Cape Point EBSA in the south hence it being the common denominator in the linkages. In the case of the Ski-Monkey, the MDS revealed higher clustering compared to the Drop camera regardless of the location and substratum type, hence a PERMANOVA was performed to further investigate this. (Appendix 6). No significant differences were found. The MDS ordinations of sampling species abundance data (Fig. 3.5 and Fig. 3.6) had lower stress levels, indicating a better ordination that allowed for interpretation of similarities between sampling stations. The simpler of mean species abundance for substrate type produced a non-significant Global R value and showed no natural groupings of location. The stronger clustering of groupings associated with substrate, as opposed to location, suggests that substrate differences have a stronger influence on community structure than the locations of these organisms within the sampling area. Sanguinetti (2013) previously found differences in community composition with benthic substrate type within the study area. This study also found that benthic substrate type directly influenced species abundance, with rock/reef substrate containing the greatest species abundance (Fig. 3.16; Fig. 3.17; Appendix 12; Appendix 12.1), similar to findings by Ruiz et

al. (2009), that reported a greater number of species in rock dominated environments and a lower number of species in sand dominated environments. The sand/mud assemblages identified in this study exhibited similarities to those found in the coastal shelf of Portugal by Gaudencio and Cabral (2007) with filter feeding assemblages dominating areas of high turbidity, strong currents and low light attenuation.

Benthic organisms are located in certain areas based on their tolerance to the abiotic factors (Garrabou and Harmelin 2002; Harley and Helmuth 2003). These factors, including temperature, light penetration, nutrient availability and sedimentation, are influenced by changes in depth (Garrabou and Harmelin 2002; Thorne-Miller 2009). The PCA analysis (Fig. 3.12- 3.15) indicated that depth is a positive variable that groups the samples. As temperature decreases, it has less influence on the grouping of sites. This temperature decrease from shallow to deep is confirmed by the Draftsman Plot (Fig. 3.11) and is further supported by Figure 3.25. These findings provided insights into how environmental variables affect the structure and grouping of benthic organisms.

De Vos (2021) also found differences in community composition with depth in the study area. Depth was the most significant predictor of differences in epibenthic megafauna and algal composition. The correlation between depth and these differences was weak but highly significant ($P = 0.01$). Another study conducted in the TMNP MPA found that species abundance significantly varied with depth, with higher abundances in deeper waters (Sanguinetti 2013). This finding is supported by other studies such as Blamey and Branch (2012), which observed a decrease in kelp communities with depth and a positive correlation between depth and diversity of understory algal growth. Depth is also correlated with light intensity and quality, which play a role in distinguishing between benthic communities (Thrush and Dayton 2002). Salinity, another important factor, directly affects the functioning of benthic communities due to its influence on tolerance levels. To gain a more comprehensive understanding of these communities, it is recommended that additional environmental variables be measured such as time of day, weather conditions, turbidity, current speed and visibility. It may also be useful to sample over a broader range of these variables to get a good understanding of their effect and overcome some of the confounding effects of covariation with depth. This will help identify and quantify the important variables that impact benthic community structure.

4.4 Community Composition

The western side of the TMNP consists mainly of rocky terrain, as indicated by previous studies labelling it as "sediment starved" (Van Zyl 2018). Similarly, the northern section of the Cape Point EBSA also has rock-dominated shorelines on its western side, but its northern coastline consists of sandy beaches (Clark et al. 1996). This area is home to commercially significant invertebrate species, such as sea urchins (*Parechinus angulosus*), abalone (*Haliotis midae*), and west coast rock lobster (*Jasus lalandii*) (Foster and Hodgson 1998; Mayfield et al. 2001; Van Zyl et al. 2003; Anderson et al. 2007). The main distinction between the priority areas is the large sand bank that extends into the Cape Point EBSA from the rockier habitats of the TMNP MPA.

The Robben Island MPA is considered a high priority area for study because it was recently established. The shallow areas of the MPA, which are less than 30 m deep, consists mostly of rocky benthic habitats with some patches of coarse sand. The dominant benthic organisms in these areas are Mollusca, Porifera, Cnidaria, and Echinodermata, which consists of soft corals, sponges, brittle stars, and sea urchins. Large populations of rock lobsters and aggregations of brittle stars are also common (Fig. 3.26). The mesophotic (or middle zone) contains organisms that depend on light or tolerate low light levels, encompasses a variety of habitats, each associated with specific benthic organisms (Button et al. 2021). In comparison to the higher diversity of taxa found in rocky reef systems, such as Cnidaria, Echinodermata, Porifera and Mollusca, these sandy habitats have fewer species and are dominated by groups like Gastropods (Fig. 3.27). The depth of stations sampled in the Robben Island MPA's deeper reaches and the nearby offshore area ranged from 71 to 250 m. In these areas, benthic habitats consisted of a mix of thick layers of fine sediment or rocky substrate covered by thin layers of fine sediment. There were also habitats that were less common, which consisted of boulders interspersed with sandy areas. Rocky substrata or boulders provided suitable habitats for many different taxonomic groups, with Mollusca, Echinodermata, Cnidarians, Porifera, and Crustacea being the most commonly seen. Examples of taxa found in these habitats included mantis shrimp, anemones, sea urchins, sea pens, cup corals, sea fans, sea slugs, hard and soft corals, gastropods, and starfish. (Fig. 3.27). In shallower areas of the TMNP MPA, the habitat was dominated by boulders. This made it challenging for camera deployments. It was found that kelp beds dominate at less than 30 m, with Ascidiaceans, Gastropods (Mollusca) and sea pens (Cnidaria) being quite common. The mesophotic zone was dominated by an assortment of sponges, ascidians, echinoderms, cnidarians and coralline algae (Fig. 3.28). In deeper waters,

the most common type of habitat was boulders and boulders with patches of fine to coarse sediment. The following epi-benthic taxa were also frequently observed: bryozoans, poriferans (with a high species number), gastropods (with an abundance of screw-shells), hydroids, sand anemones, and soft and hard coral species; echinoderms (brittle stars, starfish, and sea urchins). Polychaetes, ascidians, and crustaceans (hermit crabs and mantis shrimp) were found in small numbers at certain sites. (Fig. 3.29). The rich diversity occurring in the offshore regions of the TMNP MPA is further captured in the illustration Figure 3.30 and Figure 3.31. In the shallow reaches of the Cape Point EBSA, the average depth was 27 m. The most prevalent habitat types were patches of fine sand along with rocky outcrops and interspersed with coarse sand. The most prevalent benthic taxa found were echinoderms, cnidarians, molluscs, and poriferans. This included various crab species, soft corals, sea pens, sea slugs, sponges, sea fans, anemones, brittle stars, and feather stars. (Fig. 3.32). In the mesophotic zone, which included stations between 32 and 60 m, the habitats were either sandy or reef-like. Large clusters of screw shells were seen on sand patches. Reef habitats at these depths were teeming with life and included a rich diversity of molluscs, crustaceans (rock lobsters), poriferans, Urochordata (sea squirts), echinoderms (sea cucumbers), and cnidarians (anemones, coral, and sea fans) (Fig. 3.33). The deepest stations were found within depths of 78 and 96 m and consisted mainly of coarse sand. These areas had a low species diversity, with a scattering of gastropods such as nudibranchs and sea snails (Fig. 3.34).

When comparing benthic communities in deeper waters <100m to those in shallower areas >30, the former exhibited higher species richness and diversity. This is a characteristic feature of the rocky and boulder habitats found at deeper depths. The most prevalent and diverse taxonomic groups were Poriferans and Cnidarians, including sea fans and sea anemones. Less frequently observed were Arthropoda, such as mantis shrimp and rock lobsters and Chordata, including ascidians and bony fish (Fig. 3.33A).

The mesophotic zone had lower benthic abundance and species richness. The low species diversity and occurrence of benthic organisms such as screw shells (Gastropoda), hydroids, hard/soft corals (Cnidaria), and sponges (Porifera), can be attributed to the dominance of the fine sediment habitat (Soloana et al 2003, Van Zyl 2018) (Fig. 3.35B). The deepest areas of Cape Point EBSA ranged from 84 to 168 m and consisted of two coexisting habitat types. One type comprised patches of coarse sediment with large benthic communities of screw shells or none at all. The second type consisted of rocks and boulders serving as substrates for various

cnidarians (hydroids, soft/hard corals), echinoderms (starfish, sea urchins), and poriferans (Fig. 3.35C). Figure 3.36 portrays the variation in sea surface temperature across the sampled area. Notably, the inshore regions are colder, possibly due to upwelling events that bring in colder bottom waters (Nelson and Hutchings 1983, Goschen et al 2012, Hancke et al 2023). Comparatively, offshore sea surface temperatures are more stable than those in coastal waters (Thomas et al. 2002). Ocean surface waters generally have oxygen concentrations similar to the atmosphere. Cold waters typically contain higher oxygen concentrations than warm waters. As waters ascend towards the thermocline, organic matter from above results in oxygen consumption through metabolic processes of aerobic bacteria, leading to reduced oxygen concentrations. Therefore, the amount of oxygen in deeper water depends on the initial quantity at the surface less the amount consumed by deep-sea taxa.

Inshore areas have lower sea surface salinity compared to offshore areas, which may be attributed to freshwater runoff from the coast. The latter is noteworthy in Figure 3.37. Although variations in depth are correlated with variations in salinity (Fig. 3.24), the manner in which these variations occur depends on the location of the ocean's waters. The addition of fresh water to the ocean, lowers surface salinity, while evaporation increases it. Deeper waters maintain a more constant salinity because neither salt nor water can be introduced there (Nelson and Hutchings 1983).

The low abundance of algae in the study area may be attributed to the high abundance of herbivorous species that feed on them, such as the screw shell (*Turritella carinifera*), abalone (*Haliotis midae*) and Cape Sea urchin (*Parechinus angulosus*). These species were well-represented and documented in the area (Foster and Hodgson 1998; Mayfield et al. 2001; Bril 2012; Sink et al. 2019). This high abundance is not uncommon, as they have also been reported to be abundant on the South-West coast of Southern Africa (Field et al. 1980) and in TMNP MPA (Baliwe 2021). The low abundance of arthropods recorded in this study, could be attributed to the sampling method used, as arthropods are motile and can move out of camera view which affects the probability of that species being sampled (Brose and Martinez 2004). A far greater sampling intensity is required in a given area, for arthropods to be well represented (Brose and Martinez 2004). Chu and Leys (2010) used a fine resolution grid-point survey method and noted that crustaceans were more abundant in the presence of sponge beds than in the surrounding area. Sponges, which are colonising species (Van Soest 2007) and more resilient, usually display a much larger percentage cover in a sampled area (Arntz et al. 2006).

The high percentage cover of sponges (relative to other taxa in this study) further supports this. In a similar study conducted in a Canadian MPA, Novaczek et al. (2017) found that Porifera and Cnidaria were the most species rich groups. These taxa, along with Ascidiaceans are important as they form habitats for other benthic species (Di Camillo et al. 2013). Additionally, filter feeders, like Porifera play a crucial role in maintaining healthy marine ecosystems by improving water quality (Ostroumov 2005). Filter feeders filter and absorb minuscule organisms present in the water and in doing so, provide a link between suspended nutrients in the ocean and the benthos (Van Soest et al. 2012).

Porifera are efficient filter feeders and an important food source (Van Soest et al. 2012). Sponges also contribute to organic production through symbiotic relationships with nitrogen fixing organisms (Van Soest et al. 2012). Cnidaria and Echinodermata played the largest role in the similarity between sites within the study areas. The cnidarian *Anthopleura michaelseni*, and the echinoderm *Amphiura capensis* were the species most responsible for this. Out of the six cnidarian species recorded, four were sea anemones (*Anthostella stephensoni*; *Anthopleura michaelseni*; *Anthothoe chilensis* and *Pseudactinia flagellifera*). Sea anemones are highly abundant along the coast of South Africa with a total of 49 species recorded (Laird and Griffiths 2016). Sea anemones have a fairly high species richness in relation to the size of the study area. They can reproduce asexually, allowing for greater distribution (Gibbons et al. 2009; Boero et al. 1992; Darling et al. 2009). Sea anemones also form symbiotic relationships with fish and crabs and provide important benefits like camouflage, protection and a source of food (McLean and Mariscal 1973; Vader 1983; Ross 1984). The large contribution and high abundance of sea anemones in the study area is therefore significant. In the study area, echinoderms also played a major role in the sample similarity. The study represented three of the five classes of echinoderms namely *Amphiura capensis* (class: Ophiuroidea), *Tropiometra carinata* (class: Crinoidea) and *Austrofromia schultzei* (class: Asteroidea), highlighting the diversity of echinoderms found in the study area. Large aggregations of brittle stars were also observed in this study. Turon et al. 2000 observed brittle star aggregations at various depths and on various substratum types and their ecology is well documented (Turon et al. 2000; Stohr et al. 2012; Landschoff et al. 2015; Olbers et al 2015). Dense monospecific brittle star aggregations are known to be permanent features on the seafloor with seasonal variation in density (Dauvin et al 2013). Off the coast of South Dakota, USA, fossilised findings of such aggregations confirmed this occurrence to date back to the Late Cretaceous period with dense aggregations of ophiuroid *Brezinacantha tolis* dominating sand habitats (Thuy et al. 2018). Such —

aggregations have also been found to be linked to habitat forming sponge species as observed by Turon et al. (2000) who noted that juveniles preferred settling on certain species of sponge to facilitate feeding. Benthic ecosystems are complex ecosystems with a variety of habitats. When examining the processes responsible for the distribution and composition of benthic invertebrate fauna in the southern Benguela ecoregion, substratum type and depth were the two environmental variables which best described the observed community patterns. However, depth is a proxy for other processes at play such as turbidity, light and temperature. The unique physical hydrography of the southern Benguela ecoregion allows for persistence of a diverse benthic ecosystem. The known patterns of benthic communities in South Africa are limited and the understanding of the key environmental variables influencing their distribution are even scarcer. As a result, instruments measuring physical environmental parameters should accompany all underwater photographic surveys, despite the camera platform being used. Combining a variety of environmental variables together with photographic data analysis will allow researchers to determine which variables are important in determining the diversity of benthic ecosystems worldwide.

CHAPTER 5: CONCLUSION

Every underwater visual survey has built-in design limits, ranging from maximum operational depth restrictions of equipment to the resolution of information obtained by underwater photography. Despite these challenges, benthic ecologists continue to use underwater video and photos to derive biological patterns and fundamental biodiversity information. When contrasted with conventional macrofauna sample methods like benthic dredging, underwater video surveys typically overestimate abundance and underestimate richness (Williams et al. 2015). This is due to the fact that traditional sampling enables species-level identification, while camera data frequently combines multiple species into a single phototaxa to prevent misidentifications. Additionally, because they are easier to spot, large, noticeable species tend to prevail in camera surveys; in contrast, dredged material frequently contains smaller, less noticeable organisms that are missed by camera surveys (Williams et al. 2015). Rock/Reefs in the current study have a higher species richness than non-reef substratum. This was not a surprise.

The study has revealed that the southern Benguela ecoregion is heterogeneous and that macro epi-benthic organisms are distributed spatially according to substrate type, location and depth. The community shift with changes in substrate type which was revealed within the TMNP MPA, and concurs with the literature of previous studies conducted in this MPA (Brill and Raemaekers 2013; Roberson et al. 2015) and in doing so, represents a system equivalent to other benthic communities found on hard and/or soft substrata. The inclusion of the Robben Island MPA within the TMNP MPA ensures the connectivity of the coastal, offshore and inshore zones in this area. This inclusion however may not be sufficient to ensure the MPAs reliability to preserve biodiversity within the area. The difference between diversity based on substrate types within the study area indicates that the TMNP MPA is possibly conserving biodiversity on rock/reef within the southern Benguela ecoregion. In order to achieve more effective management of the benthic environment, a monitoring programme is required (Tunley 2009). By implementing long term monitoring of a benthic environment, we are able to determine the effects of both anthropogenic and natural processes on the environment (Schleyer et al. 2008). The expansion of this protected area to include the rock/reef outside the MPA may be considered prudent, as doing so will protect the integrity of the biodiversity of the area from potential future degradation. As seen during this study, a survey of only macro epi-benthic organisms is not conclusive enough for implementing management decisions for

the study area. Thus, continuous research is required in the study area and this particular study should accompany others to facilitate in improving informed decision making within the study area and surrounding areas by setting up sampling methods which allow for long term monitoring to be possible by using non-destructive equipment and pre-determined stations within the sampling grid.

This study has advanced our understanding of epifaunal benthic invertebrate communities and associated habitats within the southern Benguela ecoregion. Through a comprehensive examination of species diversity across different areas and an in-depth investigation into the influence of geological and physical factors on distribution and abundance, this study has yielded significant findings. Despite encountering limitations, particularly regarding sampling effort, the research contributes substantially to our knowledge of biodiversity patterns in Marine Protected Areas within the region. The identification of habitat heterogeneity as a key factor influencing species assemblages underscores the importance of MPAs in preserving biodiversity, with rock/reef substrates found to support greater species abundance compared to sandy habitats. Additionally, multivariate analyses have provided insights into community composition, highlighting the influence of substrate type and depth on benthic communities. The complexity of benthic ecosystems has been underscored, emphasizing the need for continued comprehensive studies to understand their distribution and composition. Substratum type and depth have emerged as crucial environmental variables shaping benthic communities, with depth serving as a proxy for other factors such as turbidity, light, and temperature. The unique physical hydrography of the southern Benguela ecoregion supports a diverse benthic ecosystem, emphasizing the importance of considering environmental parameters in future research efforts.

This study has provided baseline information on the benthic invertebrate fauna, associated habitats, and environmental variables influencing their distribution and abundance in the southern Benguela ecoregion. It has identified depth and substrate type as the two environmental variables that best explain community patterns observed. Regardless of camera position, I recommend continuing the use of Coral Point Count (CPCe) for benthic invertebrate picture annotation of underwater video and camera images. Despite the fact that CPCe is often utilized for overhead photography, this study showed that it can detect community patterns when applied to oblique data. To improve the benthic research capabilities and perhaps provide a better comparison of imagery from drop and towed camera systems, I also suggest adding a

separate downward-facing or an additional stereo camera system mounted to the camera frames. The downward-facing camera's configuration will not only make it easier to sample benthic invertebrates simultaneously, but it will also solve the scale problem that afflicts forward-facing and oblique camera orientation. The macrobenthos' volumetric measurements will be possible with the inclusion of a stereo camera setup similar to the Baited Remote Underwater Video Systems. This should make it possible to compare estimates of benthic biomass across various bathymetric zones and geographical areas. Recommendations for future research include the implementation of long-term monitoring programs and the use of standardized protocols to ensure the effectiveness of conservation efforts. Overall, this research contributes significantly to our understanding of benthic ecosystems in the southern Benguela ecoregion and provides valuable information for the management and conservation of marine biodiversity in the region. Results from this study will also inform marine spatial planning decisions, contributing to the protection and sustainable management of marine environments within South Africa.

REFERENCES

- Agardy MT. 1994. Advances in marine conservation: the role of Marine Protected Areas. *Trends in Ecology and Evolution* 9: 267 – 270.
- Agardy T, Notarbartolo di Sciara G, Christie P. 2011. Mind the gap: Addressing the shortcomings of Marine Protected Areas through large scale marine spatial planning. *Marine Policy* 35: 226–232.
- Anderson RJ, Rand A, Rothman MD, Share A, Bolton JJ. 2007. Mapping and quantifying the South African kelp resource. *African Journal of Marine Science* 3: 369-378.
- Anderson MJ, Walsh DCI. 2013. PERMANOVA, ANOSIM, and the Mantel test in the face of heterogeneous dispersions: What null hypothesis are you testing? *Ecological Monographs* 83: 557–574.
- Arntz WE, Thatje S, Linse K, Avila C, Ballesteros M, Barnes DKA, Cope T, Cristobo FJ, De Broyer C, Gutt J, Isla E, Pez-Gonza l, Montiel A, Munilla T, Ramos Espla AA, Raupach M, Raushert M, Rodriques E, Teixido N. 2006. Missing link in the Southern Ocean: sampling the marine benthic fauna of remote Bouvet Island. *Polar Biology* 29: 83–96.
- Atkinson LJ. 2009. Effects of demersal trawling on marine infaunal, epifaunal and fish assemblages: Studies in the southern Benguela and Oslofjord. PhD thesis, University of Cape Town. South Africa.
- Atkinson LJ, Field JG, Hutchings L. 2011. Effects of demersal trawling along the west coast of southern Africa: multivariate analysis of benthic assemblages. *Marine Ecology Progress Series* 430: 241-255.
- Attwood CG, Harris JM, Williams AJ. 1997b. International experience of Marine Protected Areas and their relevance to South Africa. *African Journal of Marine Science* 18: 311 - 332.
- Awad N. 2004. Bioactive Brominated Diterpenes from the marine red algae *Jania Rubens* (L.) Lamx. *Phytotherapy research* 18: 275-279.

- Awad A, Griffiths C, Turpie K. 2002. Distribution of South African marine benthic invertebrates applied to the selection of priority conservation areas. *Diversity and Distributions* 8: 129-145.
- Babcock RC, Shears NT, Alcala AC, Barrett NS, Edgar GJ, Lafferty KD, McClanahan TR, Russ GR. 2010. Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects. *Proceedings of the National Academy of Sciences of the United States of America* 107: 18256–18261.
- Baliwe NG. 2021. The effectiveness of Table Mountain National Park Marine Protected Area in the conservation of rocky shore biodiversity. MSc thesis, University of Cape Town. South Africa.
- Bergman S. 2019. Conductivity, Temperature and Depth graphs. [unpublished manuscript].
- Bernard A, Gotz A. 2012. Bait increases the precision in count data from remote underwater video for most subtidal reef fish in the warm-temperate Agulhas bioregion. *Marine Ecology Progress Series* 471: 235–252.
- Bigelow K, Maunder M. 2007. Does habitat or depth influence catch rates of pelagic species? *Canadian Journal of Fish Aquatics* 64: 1581-1594.
- Blamey LK, Branch GM. 2012. Regime shift of a kelp-forest benthic community induced by an ‘invasion’ of the rock lobster *Jasus lalandii*. *Journal of Experimental Marine Biology and Ecology* 420: 33–47.
- Boero F, Boullion J, Piraino S. 1992. On the origins and evaluation of hydromedusan life cycles (Cnidaria, Hydrozoa). Sex origin and evolution. *Selected Symposia and Monographs* 6: 59 - 68.
- Boero F, Bonsdorff E. 2007. A conceptual framework for marine biodiversity and ecosystem functioning. *Marine Ecology* 28: 134–145.

- Bohnsack JA. 1979. Photographic Quantitative Sampling of Hard-Bottom Benthic Communities. *Bulletin of Marine Science* 29: 242-252.
- Bohnsack JA, Ault JS. 1996. Management strategies to conserve marine biodiversity. *Oceanography* 9: 73-82.
- Boudreau SA, Worm B. 2012. Ecological role of large benthic decapods in marine ecosystems: a review. *Marine Ecology Progress Series* 469: 195-213.
- Bremner J, Rogers SI, Frid CLJ. 2003. Assessing the functional diversity in marine benthic ecosystems: a comparison of approaches. *Marine Ecology Progress Series* 254: 11 - 25.
- Brill, GC, Raemaekers. 2013. A decade of illegal fishing in Table Mountain National Park (2000-2009): trends in the illicit harvest of abalone *Haliotis midae* and west coast rock lobster *Jasus lalandii*. *African Journal of Marine Science* 35: 491-500.
- Brill G.C. 2012. The tip of the iceberg: Spatio-temporal patterns of marine resource confiscations in the Table Mountain National Park. MSc thesis, University of Stellenbosch. South Africa.
- Brose U, Martinez ND. 2004. Estimating species richness of species with variable mobility. *Oikos* 105: 292-300.
- Buhl-Mortensen L, Vanreusel A, Gooday AJ, Levin LA, Priede I, Buhl-Mortensen P, Gheerardyn H, King NJ, Raes M. 2010. Biological structures as a source of habitat heterogeneity and biodiversity on the deep ocean margins. *Marine Ecology* 31: 21-50.
- Butler, BJ, Chazdon, RL. 1998. Species Richness, Spatial Variation, and Abundance of the Soil Seed Bank of a Secondary Tropical Rain Forest. *Biotropica* 30: 214-222.
- Button RE, Parker D, Coetzee V, Samaai T, Palmer R, Sink K, Kerwath S. 2021. ROV assessment of mesophotic fish and associated habitats across the continental shelf of the Amathole region. *Nature Portfolio* 11: s41598.

Cappo M, Harvey E, Malcolm H, Speare P. 2003. Potential of video techniques to monitor diversity, abundance and size of fish in studies of marine protected areas. In J. P. Beumer, A. Grant, & D. C. Smith (eds.), *Aquatic protected areas. What works best and how do we know?* University of Queensland.

Carreiro-Silva M, Martins I, Riou V, Raimundo J, Caetano M, Bettencourt R, Rakka M, Cerqueira T, Godinho A, Morato T, Colaco A. 2022. Mechanical and toxicological effects of deep-sea mining sediment plumes on a habitat-forming cold-water octocoral. *Frontiers of Marine. Science*. 9:915650.doi: 10.3389/fmars.2022.915650.

Cary GA, Hinmann VF. 2017. Echinoderm development and evolution in the post-genomic era. *Developmental Biology* 427(2): 203-211.

Cardoso P, Silva I, De Oliviera NG, Serrano ARM. 2004. Higher taxa surrogates of spider (*Araneae*) diversity and their efficiency in conservation. *Biological Conservation* 117: 453-459.

Celliers L, Mann BQ, Macdonald AHH, Schleyer MH. 2007. A benthic survey of the rocky reefs off Pondoland, South Africa. *African Journal of Marine Science* 29: 65-77.

Chao A, Gotelli NJ, Hsieh TC, Sander EL, Ma KH, Colwell RK, Ellison AM. 2014. Rarefaction and extrapolation with Hill numbers: a framework for sampling and estimation in species diversity studies. *Ecological Monographs* 84: 45–67.

Chazdon RL, Colwell RK, Denslow JS, GaurIguata MR. 1998. Statistical methods for estimating species richness of woody regeneration in primary and secondary rain forests of North-eastern Costa Rica. In: Dallmeier F, Comiskey JA (eds): *Forest biodiversity, research, monitoring and modelling: Conceptual background and old world case studies. Man and Biosphere series* pp: 285-309.

Chu J, Leys SP. 2010. High resolution mapping of community structure in three glass sponge reefs (Porifera, Hexactinellida). *Marine Ecology Progress Series* 417: 97-113.

- Clark BM, Bennet BA, Lamberth SJ. 1996. Temporal variations in surf zone fish assemblages from False Bay, South Africa. *Marine Ecology Progress Series* 131: 35-47.
- Clarke KR, Gorley RN. 2006. PRIMER v6: User Manual/Tutorial. PRIMER-E: Plymouth.
- Clarke KR, Warwick RM. 2001. Change in Marine Communities: An Approach to Statistical Analysis and Interpretation (2nd Edition). PRIMER-E: Plymouth.
- Clarke KR, Gorley RN, Somerfield PJ, Warwick RM 2014. Change in marine communities: an approach to statistical analysis and interpretation. Primer-E Ltd.
- Clark MR, Althaus F, Schlacher TA, Williams A, Bowden BA and Rowden A. 2016. The impacts of deep sea fisheries on benthic communities: a review. *ICES Journal of Marine Science* 73(1): 51-69.
- Clearly DFR, Becking LE, De Voogd NJ, Renema W, De Beer M, Van Seost RWM, Hoeksema BW. 2005. Variation in the diversity and composition of benthic taxa as a function of distance offshore, depth and exposure in the Spermonde Archipelago, Indonesia. *Estuarine, Coastal and Shelf Science* 65: 557-570.
- Cochran SA. 2007. Benthic habitats and offshore geological resources of Kaloko-Honokōhau National Historical Park, Hawai‘i. U.S. *Geological Survey Scientific Investigations Report* 2006–5256.
- Collie J, Hall S, Kaiser M and Poiner I. 2000. A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology* 69: 785-798.
- Colwell RK, Chao A, Gotelli NJ, Lin SY, Mao CX, Chazdon RL, Longino JT. 2012. Models and estimators linking individual-based and sample-based rarefaction, extrapolation and comparison of assemblages. *Journal of Plant Ecology* 5: 3-21.
- Colwell, RK and Elsensohn, JE. 2014. EstimateS turns 20: Statistical estimation of species richness and shared species from samples, with non-parametric extrapolation. *Ecography*. 37. 10.1111/ecog.00814.

Commito JA and Ambrose WG. 1985. Multiple trophic levels in soft-bottom communities. *Marine Ecology Progress Series* 26: 289-293.

Costello MJ. 2009. Distinguishing marine habitat classification concepts for ecological data management. *Marine Ecology Progress Series* 397: 253-268.

Costello MJ, Coll M, Danovaro R, Halpin P, Ojaveer H, Miloslavich P. 2010. A Census of Marine Biodiversity Knowledge, Resources, and Future Challenges. *PLoS ONE* 5: e12110.

Dauvin JC, Mear Y, Murat A, Poizot E, Lozach S. 2013. Interactions between aggregations and environmental factors explain spatio-temporal patterns of the brittle star *Ophiothrix fragilis* in the eastern Bay of Seine. *Estuarine, Coastal and Shelf Science* 131: 171-181.

Darling K, Thomas E, Kasemann S, Wade C. 2009. Surviving mass extinction by bridging the benthic/planktonic divide. *PNAS* 106: 12629-12633.

De Boer W, van Schie A, Jocene D, Mabote AB and Guissamulo A. 2001. The impact of artisanal fishery on a tropical intertidal benthic fish community. *Environmental Biology of fishes* 61: 213-229.

Di Camillo C, Bartolucci I, Cerrano C, Bavestrello G. 2013. Sponge disease in the Adriatic Sea. *Marine Ecology* 34: 62–71.

EBSA Portal. Ecologically or Biologically Significant Marine Areas (EBSAs) in the Benguela Current Large Marine Ecosystem (BCLME). 2018. Available at <https://cmr.mandela.ac.za/EBSA-Portal>.

Edgar GD, Bustamante RH, Farina JM, Calvopina M, Martinez C, Toral-Granda MV. 2004. Bias in evaluating the effects of Marine Protected Areas: the importance of baseline data for the Galapagos Marine Reserve. *Environmental Conservation* 31: 1-7.

Edmunds PJ, Aronson RB, Swanson DW. 1998. Photographic versus visual census techniques for the quantification of juvenile corals. *Bulletin of Marine Science* 3: 937–946.

Field JG, Griffiths CL, Griffiths RJ, Jarman N, Zoutendyk P, Velimirov B, Bowes A. 1980. Variation in structure and biomass of kelp communities along the south-west Cape coast. *Transactions of the Royal Society of South Africa* 44: 145-203.

Foster MS, Harrold C, Hardin HH. 1991. Point vs. photo quadrat estimates of the cover of sessile marine organisms. *Journal of Experimental Marine Biology and Ecology* 146: 193-203.

Foggo A, Rundle SD, Bilton DT. 2003. The net result: evaluating species richness extrapolation techniques for littoral pond invertebrates. *Freshwater Biology* 48: 1756-1764.

Foster GG, Hodgson AN. 1998. Consumption and apparent dry matter digestibility of six intertidal macroalgae by *Turbo sarmaticus* (Mollusca: vetigastropoda: Turbinidae). *Aquaculture Journal* 167: 211-227.

Gaither MR, Bowen BW, Rocha LA. 2016. Fishes that rule the world: circumtropical distributions revisited. *Fish and Fisheries* 17: 664-679.

Garraboa J, Ballesteros E, Zabala M. 2002. Structure and Dynamics of North-western Mediterranean Rocky Benthic Communities along a Depth Gradient. *Estuarine, Coastal and Shelf Science* 55: 493-508.

Garrabou J and Harmelin JG. 2002. A 20-year study on life-history traits of a harvested long-lived temperate coral in the NW Mediterranean: insights into conservation and management needs. *Journal of Animal Ecology* 71: 966-978.

Gaudencio M, Cabral H. 2007. Trophic structure of macrobenthos in the Tagus estuary and adjacent coastal shelf. *Hydrobiologia* 587: 241-251.

Gibbons MJ, Abiahy BB, Angel M, Assuncao CML, Bartsch I, Best P, Biseswar R, Bouillon J, Bradford-Grieve JM, Branch W, Bureson E, Cannon L, Casanova JP, Channing A, Child CA, Compagno L, Cornelius PFS, Dadon JR, David JHM, Day J, Della Croce N, Emschermann P, Erseus C, Esnal G, Gibson R, Griffiths CL, Hayward PJ, Heard R, Heemstra P, Herbert D, Hessler R, Higgins R, Hiller N, Hirano YM, Kensley B, Kilburn R,

Kornicker L, Lambshead J, Manning R, Marshall D, Mianzan H, Monniot CF, Newman W, Nielsen C, Patterson G, Pugh P, Roeleveld M, Ross A, Ryan P, Ryland JS, Samaai T, Schleyer M, Schockaert E, Seapy R, Shiel R, Sluys R, Southward EC, Sulaiman A, Thandar A, van der Spoel S, van Soest R, van der Land J, Vetter E, Vinogradov G, Williams G, Wooldridge T. 1999. The taxonomic richness of South Africa's marine fauna: a crisis at hand. *South African Journal of Science* 95: 8-12.

Gibbons MJ, Richardson A. 2009. Patterns of jellyfish abundance in the North Atlantic. *Hydrobiologia* 616: 51-65.

Glover AG, Gooday AJ, Bailey DM, Billet DSM, Chevaldonna P, Colaco A, Copley J, Cuvelier D, Desbruyeres D, Kalogeropoulou V, Klages M, Lampadariou N, Lejeune C, Mestre NC, Paterson GLJ, Perez T, Ruhl H, Sarrazin J, Soltwedel T, Soto EH, Thatje S, Tselepidis A, Van Gaever S, Vanreusel A. 2010. Temporal Change in Deep-Sea Benthic Ecosystems: A Review of the Evidence from Recent Time-Series Studies. *Advances in Marine Biology* 58: 1-95.

Glover DM, Jenkins WJ, Doney SC. 2011. *Modelling Methods for Marine Science*. Woods Hole Oceanographic Institution. Available at: <https://doi.org/10.1017/CBO9780511975721>

Goschen W, Schumann E, Bernard K, Bailey S. 2012. Upwelling and ocean structures off Algoa Bay and the south-east coast of South Africa. *African Journal of Marine Science* 34(4): 525-536.

Gotelli NJ, Colwell RK. 2001. Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. *Ecology Letters* 4: 379-390.

Götz A, Kerwath SE, Attwood CG, Sauer HH. 2009b. A change in the seaward boundary of Goukamma Marine Protected Area could increase conservation and fishery benefits. *South African Journal of Marine Science* 105: 330 - 331.

Gray JS. 2000. The measurement of marine species diversity, with an application to the benthic fauna of the Norwegian continental shelf. *Journal of Experimental Marine Biology and Ecology* 250: 23-49.

Gray JS. 2001. Marine Diversity: the paradigms in patterns of species richness examined. In: Gili JM, Pretus JL, Packard TT (eds), *Scientia Marina* 65: 41- 56.

Griffiths CL, Robinson TB, Lange L, Mead A. 2010. Marine Biodiversity in South Africa: An Evaluation of Current States of Knowledge. *PLoS ONE* 5: e12008.

Hall SJ. 1994. Physical disturbance and marine benthic communities: life in unconsolidated sediments. *Oceanography and Marine Biology, an annual review* 32: 179-239.

Hancke L, Smeed D, Roberts M, Russo C, Rayner D, Jebri F. 2023. Atmospheric and advective forcing of upwelling on South Africa's central Agulhas Bank. *Deep Sea Research Part 2*. 1-16.

Harley CDG, Helmuth BST. 2003. Local-and regional-scale effects of wave exposure, thermal stress, and absolute versus effective shore level on patterns of intertidal zonation. *Limnology and Oceanography* 48: 1498–1508.

Hewitt JE, Joanne E, Thrush S. 2008. Multiple stressors, nonlinear effects and the implications of climate change impacts on marine coastal ecosystems. *Global Change Biology*. 22: 2665-2675.

Hoeksema BW, De Voogd NJ. 2012. On the run: free-living mushroom corals avoiding interaction with sponges. *Coral Reefs* 31: 455-459.

Howell KL, Davies JS, Allcock AL, Braga-Henriques A. 2019. A framework for the development of a global standardised marine taxon reference image database (SMarTaR-ID) to support image-based analyses. *PLoS ONE* 14: e0218904

Jones A, Haywood J, Boucher O. 2009. Climate impacts of geoengineering marine stratocumulus clouds. *Journal of geophysical research* 14: 1-9.

Kaiser M, Collie J, Hall S, Jennings S and Poiner I. 2003. Impacts of fishing gear on marine benthic habitats. 1-19.

Kerwath SE, Winker H, Götz A, Attwood, CG. 2013. Marine protected area improves yield without disadvantaging fishers. *Nature Communications*. 4: 1–6.

Kirkman SP, Mann BQ, Sink KJ, Adams R, Livingstone T-C, Mann-Lang J, Pfaff MC, Samaai T, van der Bank MG, Williams L, Branch GM. 2021. Evaluating the evidence for ecological effectiveness of South Africa's marine protected areas. *African Journal of Marine Science* 43:389-412.

Kingsford MJ, Battershill CN. 1998. Subtidal habitats and benthic organisms of rocky reefs. In: Kingsford MJ, Battershill CN (eds.): *Studying Temperate Marine Environments: A Handbook for Ecologists*. Christ Church, New Zealand: Canterbury University Press pp 84 - 114.

Kohler KE, Gill SM. 2005. Coral Point Count with Excel Extensions (CPCe): A visual basic program for the determination of coral and substrate coverage using random point count methodology. *Computers and Geosciences* 32: 1259-1269.

Krebs C. 1989. *Ecological Methodology*. New York Harper and Row.

Laird MC, Griffiths CL. 2016. Additions to the South African sea anemone (Cnidaria, Actiniaria) fauna, with expanded distributional ranges for known species. *African Invertebrates* 57: 15–37.

Landschoff J, Du Plessis A, Griffiths C. 2015. A dataset describing brooding in three species of South African brittle stars, comprising seven high resolution micro x-ray computed tomography scans. *Gigascience* 52: 1-4.

Levin LA, Etter RJ, Rex MA, Gooday AJ, Smith CR, Pineda J, Stuart CT, Hessler H, Pawson D. 2001. Environmental Influences on Regional Deep-Sea Species Diversity. *Annual Review of Ecology, Evolution, and Systematics* 32: 51-93.

Lombard AT, Strauss T, Harris J, Sink K, Attwood C, Hutchings L. 2004. *South African National Spatial Biodiversity Assessment 2004: Technical Report. Vol. 4: Marine Component*. Pretoria: South African National Biodiversity Institute. South Africa.

Lombard AT, Reyers B, Schonegevel L, Cooper J, Smith L, Nel D, Froneman P, Ansorge I, Bester M, Tosh C, Strauss T, Akkers T, Gon O, Leslie R, Chown S. 2007. Conserving pattern and process in the Southern Ocean: designing a Marine Protected Area for the Prince Edward Islands. *Antarctic Science* 19: 39-54.

Magurran AE. 2004. *Measuring Biological Diversity*. Oxford: Wiley.

Mallet D, Pelletier D. 2014. Underwater video techniques for observing coastal marine biodiversity: a review of sixty years of publications (1952 – 2012). *Fisheries Research* 154: 44-62.

Mah CL, Blake DB. 2012. Global Diversity and Phylogeny of the Asteroidea (Echinodermata). *PLoS ONE* 7: e35644.

Mayfield S, De Beer E, Branch GM. 2001. Prey preference and the consumption of sea urchins and juvenile abalone by captive rock lobsters (*Jasus lalandii*). *Marine and Freshwater Research* 52: 773-780.

Mclean RB, Mariscal RN. 1973. Protection of a hermit crab by its symbiotic sea anemone *Calliactis tricolor* (Le Sueur) in protecting hermit crabs from octopus predators *Calliactis tricolor*. *Journal of Experimental Marine Biology and Ecology*: 116: 128-130.

Meese RJ, Tomich PA. 1992. Dots on the rocks: a comparison of percent cover estimation methods. *Journal of Experiential Marine Biology and Ecology*. 165: 59-13.

National Gazette No 42478 of 23-May-2019, Volume 647 <https://cer.org.za/wp-content/uploads/2019/05/MPA-declarations.pdf>

National Research Council. 2001. *Marine Protected Areas: Tools for Sustaining Ocean Ecosystems*. Washington, DC: The National Academies Press. <https://doi.org/10.17226/9994>.

Nelson G and Hutchings L. 1983. The Benguela upwelling area. *Progress in Oceanography* 12: 333-356.

Nkwinkwa NAS. 2018. On the role of the Agulhas Current on weather and climate of South Africa. Faculty of Science. Department of Oceanography .<http://hdl.handle.net/11427/30111>.

Novaczek E, Devillers R, Edinger E. 2017. High-resolution seafloor mapping to describe coastal denning habitat of a Canadian species at risk: Atlantic wolffish (*Anarhichas lupus*). *Canadian Journal of Fisheries and Aquatic Sciences* 3: 1-34.

Olbers JM, Celliers L, Schleyer MH. 2008. Zonation of benthic communities on the subtropical Aliwal Shoal, Durban, KwaZulu-Natal, South Africa. *African Zoology* 44: 8-23.

Ostroumov SA. 2005. Some aspects of water filtering activity of filter-feeders. *Hydrobiologia* 542: 275–286.

Otto L. 2019. South Africa's Operation Phakisa: Development without security? *Scientia Militaria, South African Journal of Military Studies*. 47: 69-84.

Parravincini V, Micheli F, Montefalcone M, Morri C, Villa E, Castellano M, Povero P, Bianchi CN. 2013. Conserving biodiversity in a human-dominated world: Degradation of marine sessile communities within a protected area with conflicting human uses *PLoS ONE*. 8: 1-12.

Pringle JD. 1984. Efficiency Estimates for Various Quadrat Sizes Used in Benthic Sampling. *Canadian Journal of Fisheries and Aquatic Sciences* 41: 1485-1489.

Preskitt LB, Vroom PS, Smith CM. 2004. A Rapid Ecological Assessment (REA) Quantitative Survey Method for Benthic Algae Using Photoquadrats with SCUBA. *Pacific Science* 58: 201-209.

- Purvis A, Hector A. 2000. Getting the measure of biodiversity. *Nature* 405: 207-253.
- Rahman S, Tsuchiya M, Uehara T. 2009. Effects of temperature on hatching rate, embryonic development and early larval survival of the edible sea urchin, *Tripneustes gratilla*. *Biologia* 64: 768-775.
- Ramirez-Llodra E, Tyler PA, Baker MC, Bergstad OA. 2011. Man and the last great wilderness: human impact on the deep sea. *PLoS ONE* 6. e22588. 10.1371/journal.pone.0022588. 6: e22588.
- Roberson L, Winker H, Attwood C, D Vos L, Gotz A. 2015. First survey of fishes in the Betty's Bay Marine Protected Area along South Africa's temperate south-west coast. *African Journal of Marine Science* 37: 543–556.
- Roberts DE, Smith A, Ajani P, Davis AR. 1998. Rapid changes in encrusting marine assemblages exposed to anthropogenic point-source pollution: a 'Beyond BACI' approach. *Marine Ecology Progress Series* 163: 213-224.
- Roberts JM, Wheeler AJ, Freiwald A. 2006. Reefs of the deep: the biology and geology of cold-water coral ecosystems. *Science* 312(5773): 543-547. <https://www.researchgate.net/publication/287314060>.
- Routledge, RD. 1980. Bias in estimating the diversity of large, uncensused communities. *Ecology* 61: 276 - 281.
- Ross RM, Quetin LB. 1984. Depth distribution of developing *Euphausia superba* embryos, predicted from sinking rates. *Marine Biology* 79: 47-53.
- Ross WS, Quattrini AM. 2007. The fish fauna associated with deep coral banks off the south eastern United States. *Deep Sea Research* 54: 975-1007.
- Roux JP, Van Der Lingen C, Gibbons MJ, Moroff N, Shannon L, Smith A, Cury P. 2013. Jellyfication of marine ecosystems as a likely consequence of overfishing small pelagic fishes: lessons from the Benguela. *Bulletin of Marine Science* 89: 249–284.

Ruiz GM, Freestone A, Fofonoff PW, Simkanin C. 2009. Habitat Distribution and Heterogeneity in Marine Invasion Dynamics: the importance of hard substrate and artificial structure. *Marine Hard Bottom Communities*. pp 321-332.

Samaai T, Gibbons MJ, Kerwath S, Yemane D, Sink K. 2010. Sponge richness along a bathymetric gradient within the iSimangaliso Wetland Park, South Africa. *Marine Biodiversity* 40: 205-217.

Sanguinetti CAV. 2013. Patterns in reef fish assemblages as determined by baited remote underwater video (BRUV) along the western side of False Bay: effects of site, depth and protection. *African Journal of Marine Science* 37: 543–556.

Schleyer MH, Kruger A, Celliers L. 2008. Long-term community changes on a high-latitude coral reef in the Greater St Lucia Wetland Park, South Africa. *Marine Pollution Bulletin* 56: 493-502.

Shannon LV. 1985. The Benguela Ecosystem Part 1. Evolution of the Benguela, physical features and the processes. *Journal of Oceanography and Marine Biology: An Annual Review* 23: 105-182.

Shannon LV, Nelson G. 1996. *The Benguela: Large scale features and processes and system variability. The South Atlantic: Present and Past Circulation* pp 163 - 210.

Shaw RG, Mitchell-Olds T. 1993. Anova for Unbalanced Data: An Overview. *Ecology* 74: 1638-1645.

Sheehan EV, Stevens TF, Gall SC, Cousens SL, Attrill MJ. 2013. Recovery of a Temperate Reef Assemblage in a Marine Protected Area following the Exclusion of Towed Demersal Fishing. *PLoS ONE* 8: e83883.

Simpkins MA, Hiruki-Raring LM, Sheffield G, Grebmeier JM, Bengtson JL. 2003. Habitat selection by ice-associated pinnipeds near St. Lawrence Island, Alaska in March 2001. *Polar Biology* 26: 577-586.

Sink, K. & Attwood, C. 2008. Guidelines for Offshore Marine Protected Areas in South Africa. SANBI Biodiversity Series 9. South African National Biodiversity Institute, Pretoria.

Sink KJ, Wilkinson S, Atkinson LJ, Sims P, Leslie R and Attwood CG. 2012. The potential impacts of South Africa's demersal hake trawl fishery on benthic habitats: Historical perspectives, spatial analysis, current review and potential management actions. DOI: [sanbi.org/wp-content/uploads/2018/04/benthic-report-sadstia-Sink-et-al-2012](https://doi.org/10.17159/sajs.2016/a0179).

Sink K. 2016. The Marine Protected Areas debate: implications for the proposed Phakisa Marine Protected Areas network. *South African Journal of Science* 1: 112. 10.17159/sajs.2016/a0179.

Sink K, Harris J, Samaai T, Leslie R, Attwood C. 2019. Template for Submission of Scientific Information to Describe Ecologically or Biologically Significant Marine Areas.1-5. Smith B and Wilson JB. 1996. A consumer's guide to evenness indices. *Oikos* 76: 70 - 82.

Sink, K., Adams, R. Mann, J., Whitehead, O., Franken, M. & Maze, K. 2019. South Africa's New Marine Protected Areas. South African National Biodiversity Institute, an entity of the Department of Environmental Affairs, Pretoria.

Smith JE, Hunter CL, Smith CM. 2010. The effects of top-down versus bottom-up control on benthic coral reef community structure. *Oecologia*. 163. 497-507.

Snelgrove PVR. 1999. Getting to the Bottom of Marine Biodiversity: Sedimentary Habitats: Ocean bottoms are the most widespread habitat on Earth and support high biodiversity and key ecosystem services. *BioScience* 49: 129-138.

Solana M, Germanob J, Rhoads D, Smith C, Michaud E, Parry D, Wenzhofer F, Kennedy B, Henriques C, Battle E, Carey D, Locco L, Valente R, Watson J, Rosenberg R. 2003. Towards a greater understanding of pattern, scale and process in marine benthic systems: a picture is worth a thousand worms. *Journal of Experimental Marine Biology and Ecology* 7: 313 – 338.

South African Environmental Observation Network. 2023. Marine Protected Areas – South African Risk and Vulnerability Atlas. Available at (<https://www.sarva.saeon.ac.za/marine-protected-areas/php>).

South African National Parks. 2022. Table Mountain National Park. Available at (https://www.sanparks.org/parks/table_mountain/tourism/activities.php).

Stirling G, Wilsey B. 2001. Empirical relationships between species richness, evenness, and proportional diversity. *The American Naturalist* 158: 286 - 299.

Stohr S, Hara TD, Thuy B. 2012. Global diversity of brittle stars (Echinodermata: Ophiuroidea). *PLoS ONE* 7: e31940.

Thain M, Hickman M. 2000. *The Penguin Dictionary of Biology, Tenth Edition*. London: Penguin books.

Thayer CW. 1974. Morphologic Adaptations of Benthic Invertebrates to Soft Substrata. *Journal of Marine Research* 33: 177-189.

Thomas A, Byrne D, Weatherbee R. 2022. Coastal sea surface temperature variability from Landsat infrared data. *Remote Sensing of Environment* 81: 262-272.

Thorne-Miller B. 1999. *The living ocean: Understanding and protecting marine biodiversity*. Washington: Island Press.

Thrush SF, Hewitt JE, Cummings VJ, Dayton PK, Cryer M, Turner SJ, Funnel GA, Budd RG, Milburn CJ and Wilkinson MR. 1998. Disturbance of the marine benthic habitat by commercial fishing: impacts at the scale of the fishery. *Ecological Society of America*. 866-879.

Thrush S, Dayton P. 2002. *Disturbance to marine benthic habitats by trawling and dredging: implications for marine biodiversity*. Annual Review of Ecology and Systematics pp 449-473.

- Thrush SF, Hewitt JE, Gibbs M, Lundquist C, Norkko A. 2006. Functional role of large organisms in intertidal communities: community effects and ecosystem function. *Ecosystems* 9: 1029–1040.
- Tunley K. 2009. State of management of South Africa's Marine Protected Areas. *WWF South Africa Report Series: 2009* pp 1-209.
- Turon X, Codina M, Tarjuelo I, Uriz M, Becerro M. 2000. Mass recruitment of *Ophiothrix fragilis* (Ophiuroidea) on sponges: settlement patterns and post settlement dynamics. *Marine Ecology-Progress Series* 200: 201-212.
- Turpie JK, Beckley LE, Katua SM. 2000. Biogeography and the selection of priority areas for conservation of South African coastal fishes. *Biological Conservation* 92: 59 - 72.
- Turpie JK, Clark BM, Hutchings K, Orr KK, De Wet J. 2009. Ecology, value and management of the Kogelberg coast. *Environmental Development* 4: 1-30.
- Underhill LG, Whittington PA, Calf KM. 2001. Shoreline birds of Robben Island, Western Cape, South Africa. *Water Study Group Bulletin* 96: 37-39.
- Valentine JW. 2009. Overview of Marine Biodiversity. In: Witman JD, Roy K (eds.): *Marine Macroecology*. Chicago and London: The University of Chicago Press pp 3-28.
- Vader W. 1983. Prehensile pereopods in gammaridean Amphipoda. *Sarsia* 68: 139-148. <https://doi.org/10.1080/00364827.1983.10420565>.
- Van der Heever G, Atkinson L, Von der Meden C. 2021. Comparison of the particle size distributions of sediment collected from sandy seafloor using a Van veen grab and cone dredge. *African Journal of Marine Science* 43: 259-264.
- Van Wilgen B, Forsyth GG, Prins P. 2012. The Management of Fire-Adapted Ecosystems in an Urban Setting: The Case of Table Mountain National Park, South Africa. *Ecology and Society* 17: 1-10.

Van Soest RWM. 2007. Sponge biodiversity. *Journal of Marine Biological Association of the United Kingdom* 87: 1345-1348.

Van Soest RWM, Boury-Esnault N, Vacelet J, Dohrmann M, Erpenbeck D, De Voogd NJ, Santodomingo N, Vanhoorne B, Kelly M, Hooper JNA. 2012. Global diversity of sponges (Porifera). *PLoS ONE* 7: 1-23.

Van Zyl RF, Mayfield S, Branch GM. 2003. Aquarium Experiments Comparing the feeding behavior of Rock Lobster *Jasus lalandii* on Abalone and Sea Urchins at two sites on the west coast of South Africa. *African Journal of Marine Science* 25: 377-382.

Van Zyl WF. 2018. Geological mapping of the inner shelf off Cape Town's Atlantic Seaboard, South Africa. MSc thesis. University of Cape Town. South Africa.

Vevers HG, Phil DAM. 1951. Photography of the sea floor. *Journal of the Marine Biological Association of the United Kingdom*. 30: 1-64.

Warwick RM. 1988. The level of taxonomic discrimination required to detect pollution effects on marine benthic communities. *Marine Pollution Bulletin* 19: 259-268.

Webb JJ, Vanden Berghe E, O'dor R. 2010. Biodiversity's big wet secret: The Global Distribution of Marine Biological Records Reveals Chronic Under-Exploration of the deep Pelagic Ocean. *Plos One* 5: e10223.

Widdicombe S, Somerfield PJ. 2012. Marine biodiversity: it's past development, present status, and future threats. *Marine Biology* 3:1-35.

Williams A, Dowdney J, Smith ADM, Hobday AJ and Fuller M. 2011. Evaluating impacts of fishing on benthic habitats: A risk assessment framework applied to Australian fisheries. *Fisheries Research* 112: 154-167.

Williams R, Erbe C, Ashe E, Clark CW. 2015. Quiet(er) Marine Protected Areas. *Marine Pollution Bulletin* 100: 154-161.

Worm B, Lotze HK. 2009. Changes in marine biodiversity as an indicator of Climate Change In: Letcher TM (ed.): *Climate Change: Observed impacts on planet earth*. Oxford: Elsevier pp 263-279.

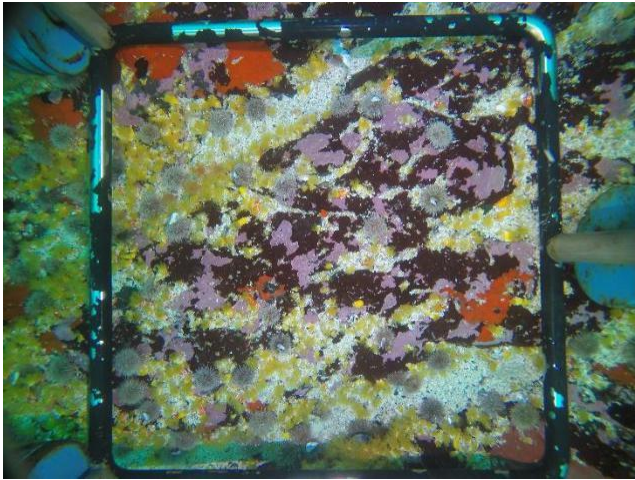
Wozniak D. 2023. Towards informing a data-driven approach to marine bioregionalization in South Africa: A case study using benthic epifaunal datasets from the southern Benguela shelf. MSc thesis. University of Cape Town. South Africa.

ZAR JH. 1999. *Biostatistical Analysis (4th edition)*. New Jersey: Pearson Education.

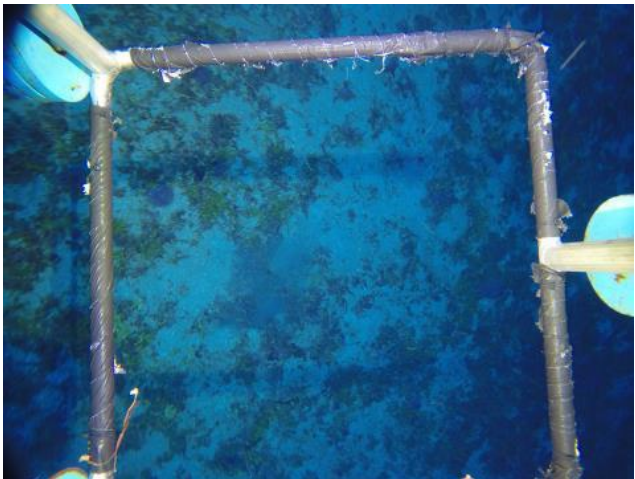
Zardus JD, Etter RJ, Chase MR, Rex MA. 2006. Bathymetric and geographic population structure in the pan-Atlantic deep-sea bivalve *Deminucula atacellana* (Schenck, 1939). *Molecular Ecology* 15: 639–651.

Appendices

Appendix 1



Appendix 2



Appendix 3



Appendix 4

Challenges and Recommendations

The current cruise was the first during which Drop camera operations were conducted from the *R/V Algoa*. During the first few attempts, it was noted that the apparatus was underweighted since slack was not detectable on the towing winch. Extra weights were therefore added to the steel frame and attached via cable ties (Appendix 4.1). These were however lost after deployment in rough terrain with large swells. New weights were added with duct tape and further secured with metal bands. It was recommended that extra weights be more firmly secured (e.g. via welding).



Appendix 4.1. Weights added and secured to the Drop camera system

Inadequate camera lighting was another challenge. Two sets of torches are available for Drop camera operations at different depths: one set operable under 100 m (Big Blue), and another has a 300 m (Scuba Pro) depth limit. Note that this was the first attempt at using the 300 m torches. The lightning on the first set of torches produced pictures of perfect quality. However, depths of up to 250 m were sampled and the powerful torches operable at depths of 300 m were too bright, resulting in poor light dispersal and images of poor quality. This was corrected by merely placing a plastic bag over the torches to filter light and increase dispersal for clearer images. It was recommended that filters be fitted to the torches for future operations and since torches are also very exposed and secured by only two clamps on the side, a housing should be built for extra protection which will minimize damage and prevent loss. Nonetheless, after all adjustments were made, apart from the CTD, the Drop camera proved to be the most successful

operation onboard with minimal time and effort needed to collect valuable information of the benthic environment.



Appendix 4.2. Left - Images taken by the 100 m torches, and 300 m torches (right)

Operating the Ski-Monkey Camera system proved more challenging. Problems encountered included:

- Connection issues (From Winch and GPS) during operations,
- Damaged cables causing connection issues (new underwater joint made (Appendix 4.3)),
- Poor battery power which only allowed for images and no video footage at some stations,
- Broken bolts,
- Cracked buoys needed for floatation,
- Altimeter (depth recorder) stopped working,
- Poor focus in silty environments,
- Unsuccessful landing attempts because of strong currents, Software problems, and
- The camera was generally inoperable in rough terrain (e.g. big boulders)



Appendix 4.3. A cable damaged in rough terrain

These challenges are not exclusive to this cruise, and although some problems encountered were addressed, i.e. repairs or alterations were possible, others will remain due to the overall design of the Ski-Monkey itself. It is recommended that the whole apparatus not be towed in the future, as this results in damaged cable and connection issues, but rather it be converted into a drop system similar to that of the Drop camera. Other recommendations include a stronger battery be purchased, and a device (e.g. an Android tablet) to provide live feed which the winch operator can use to have a visual of the entire operation at any given time.

Other general challenges and recommendations:

Underway measuring and recording scientific systems ran fairly well, apart for the vessel based weather station (WeatherPak), which malfunctioned at the onset of the voyage.

The General Oceanics pCO₂ measuring system also presented challenges, mainly the condenser which failed to cool down extracted atmospheric (ATM) and equilibrator (EQU) air. These systems should have been attended to after completion of the cruise.

While carrying out requisite maintenance activities, a total of 55 and 20 m of wire was cut from the Vertical Plankton Winch (VPW) and Hydrographic Winch (Hydro-winch) cables respectively,

Water draining from the air-conditioning unit in the vessels Acoustic lab frequently floods the area. With the new satellite communications equipment cabinet installed in close proximity to the air-con and an increasing demand for cooling in this lab, the drainage problem requires a permanent solution,

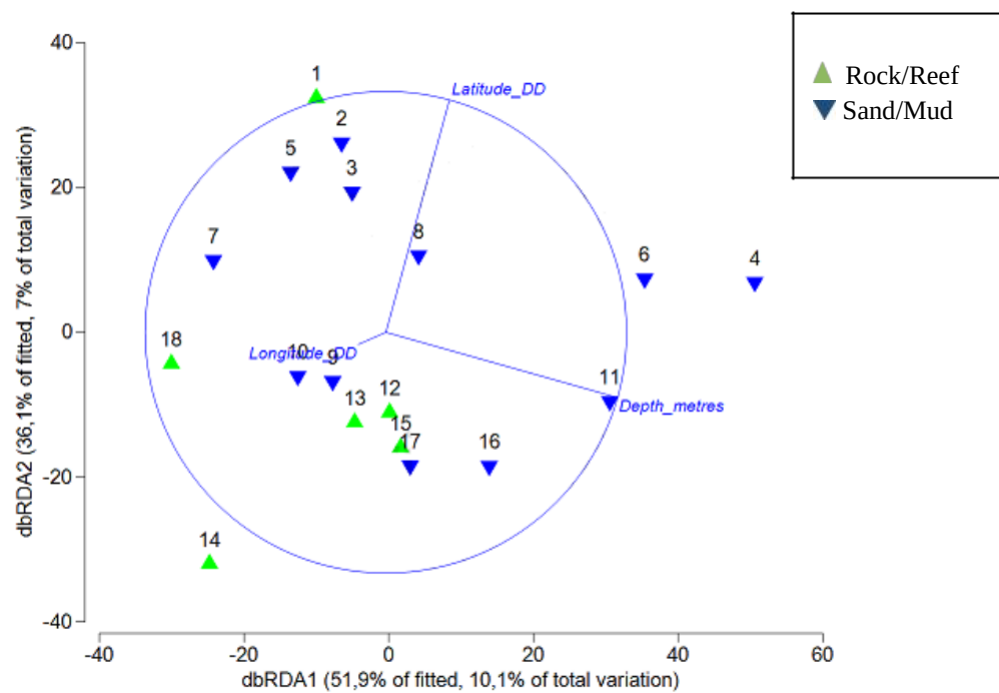
The lightning in the dry lab malfunctioned midway through the voyage and the ships engineers decided to resolve this problem once the vessel had returned safely to Port.

Appendix 5: Total number of photographic samples obtained for each station sampled.

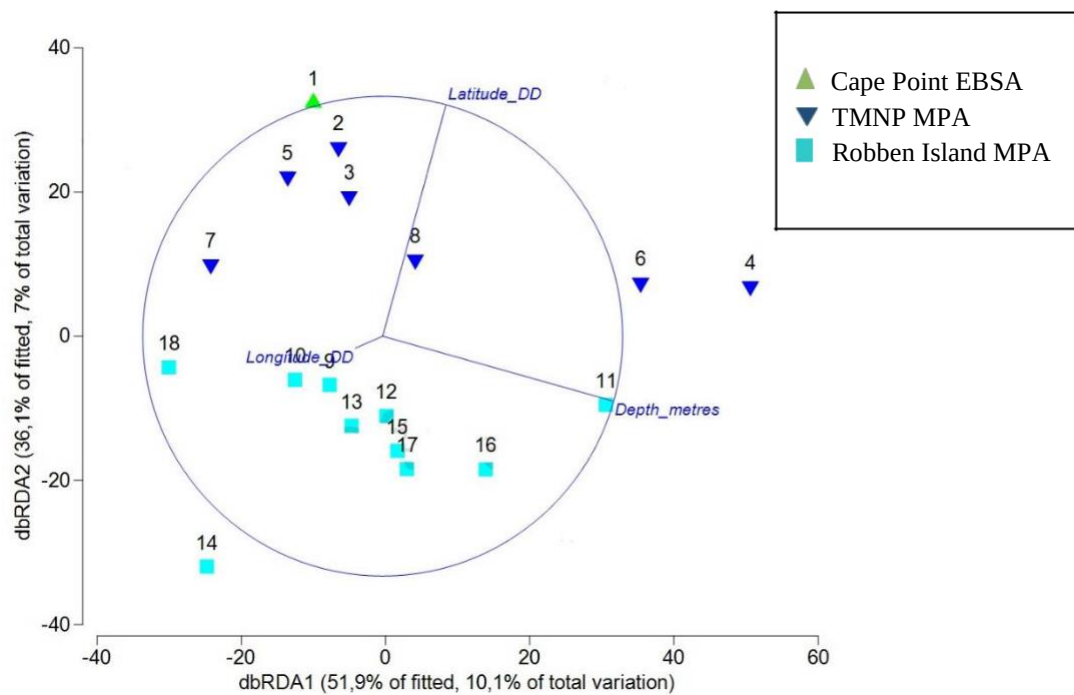
Station number	Number of samples
141	3
172	19
176	10
177	10
199	6
203	45
229	22
233	13
236	7
281	5
284	23
289	8
340	19
346	26
363	5
375	15
377	3
395	13
420	12
431	11
436	6
461	17
501	2
519	10
520	12
524	11
525	10
527	33
529	15
537	4

545	7
546	5
549	7
564	8
568	5
570	5
588	5
606	5
616	13

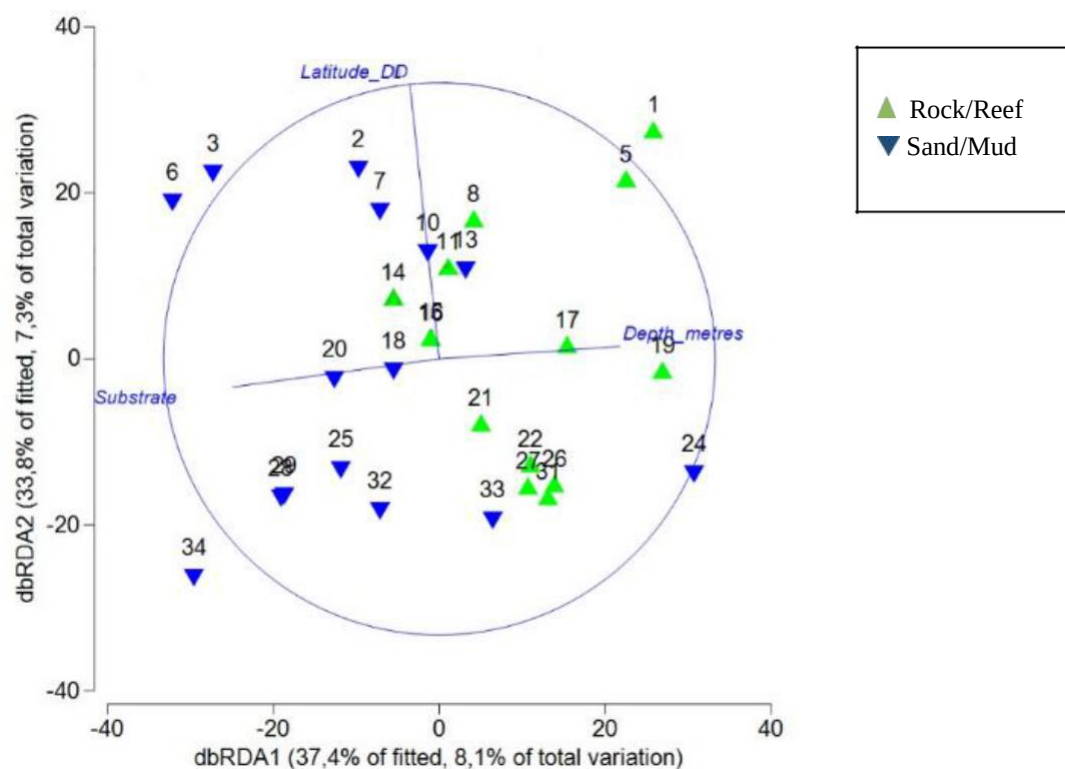
Appendix 6.



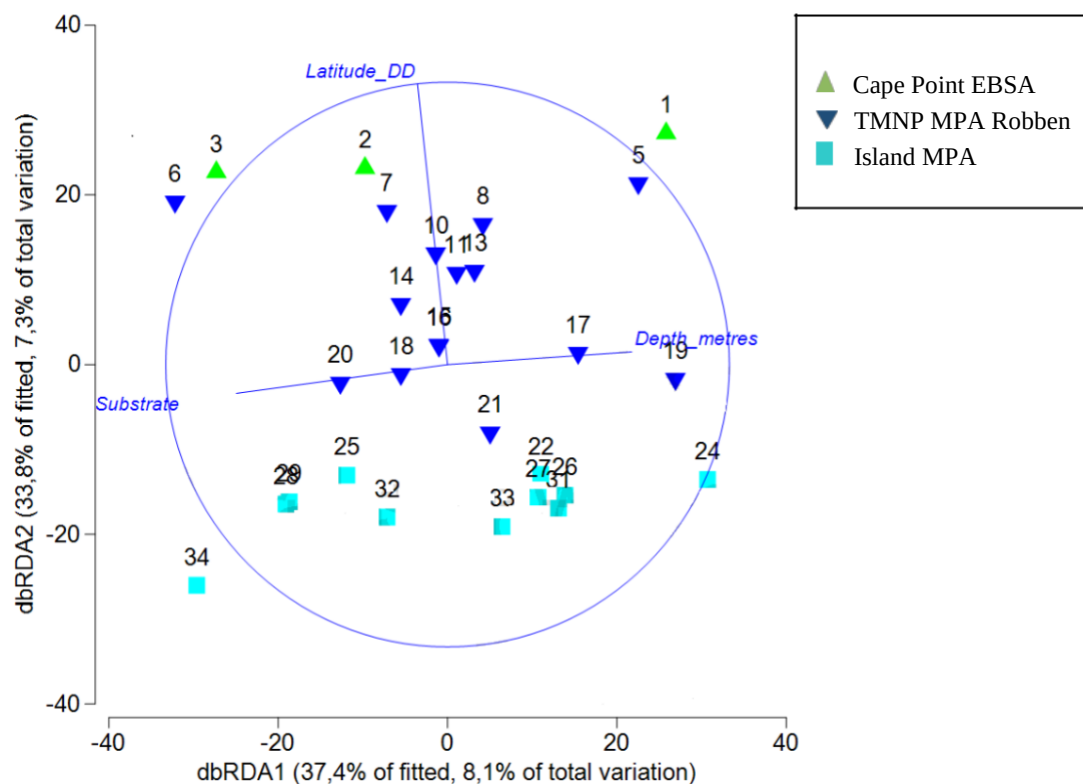
Appendix 6.1: Ski-Monkey PERMANOVA for Substrate indicating Rock/Reef and Sand/Mud using benthic biota abundance data with Depth, Salinity, Temperature, Substrate type, Latitude and Longitude as environmental variables.



Appendix 6.2: Ski-Monkey PERMANOVA for location indicating Cape Point EBSA, TMNP MPA and Robben Island MPA using benthic biota abundance data with Depth, Salinity, Temperature, Substrate type, Latitude and Longitude as environmental variables.



Appendix 7: Drop camera PERMANOVA for Substrate indicating Rock/Reef and Sand/Mud using benthic biota abundance data with Depth, Salinity, Temperature, Substrate type, Latitude and Longitude as environmental variables.



Appendix 7.1: Drop camera PERMANOVA for location indicating Rock/Reef and Sand/Mud using benthic biota abundance data with Depth, Salinity, Temperature, Substrate type, Latitude and Longitude as environmental variables.

Appendix 8: Similarity percentage analyses SIMPER for communities based on fourth root transformed abundance data, displaying the top three characteristic morphospecies groups.

Cape Point EBSA

Average similarity: 20.62

Morphospecies	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Gastropods	2.05	7.13	2.91	34.58	34.58
Sponge	2.00	3.91	0.58	18.94	53.53
Solitary Ascidian	1.29	2.79	0.58	13.53	67.05

TMNP MPA

Average similarity: 17.26

Morphospecies	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Anemonies	0.72	3.49	0.43	20.19	20.19
Sponge	1.07	2.88	0.54	16.68	36.87
Gastropods	0.74	2.60	0.34	15.07	51.95

Robben Island MPA

Average similarity: 25.52

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Brittle stars	1.46	3.09	0.46	12.11	41.49
sponge	1.07	2.82	0.75	11.05	52.54
Corals	0.93	2.33	0.58	9.11	61.65

Appendix 8.1: Similarity percentage analyses SIMPER for communities based on fourth root transformed abundance data, displaying the top five characteristic morphospecies groups between benthic substrate type.

Sand

Average similarity: 14.28

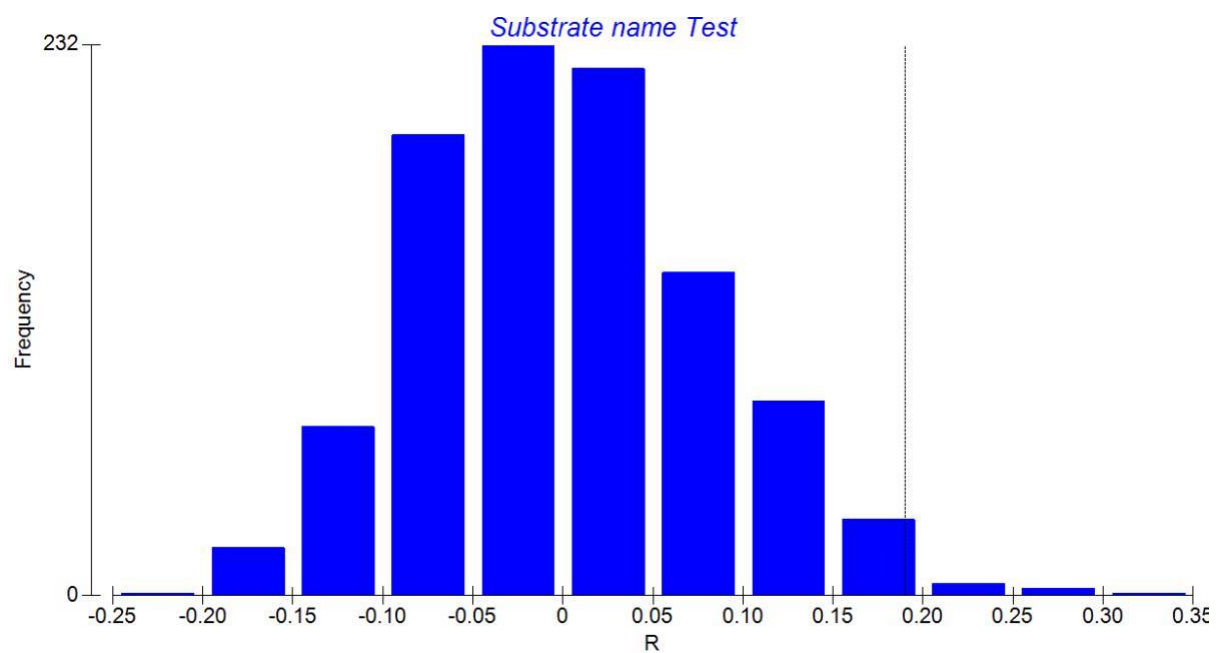
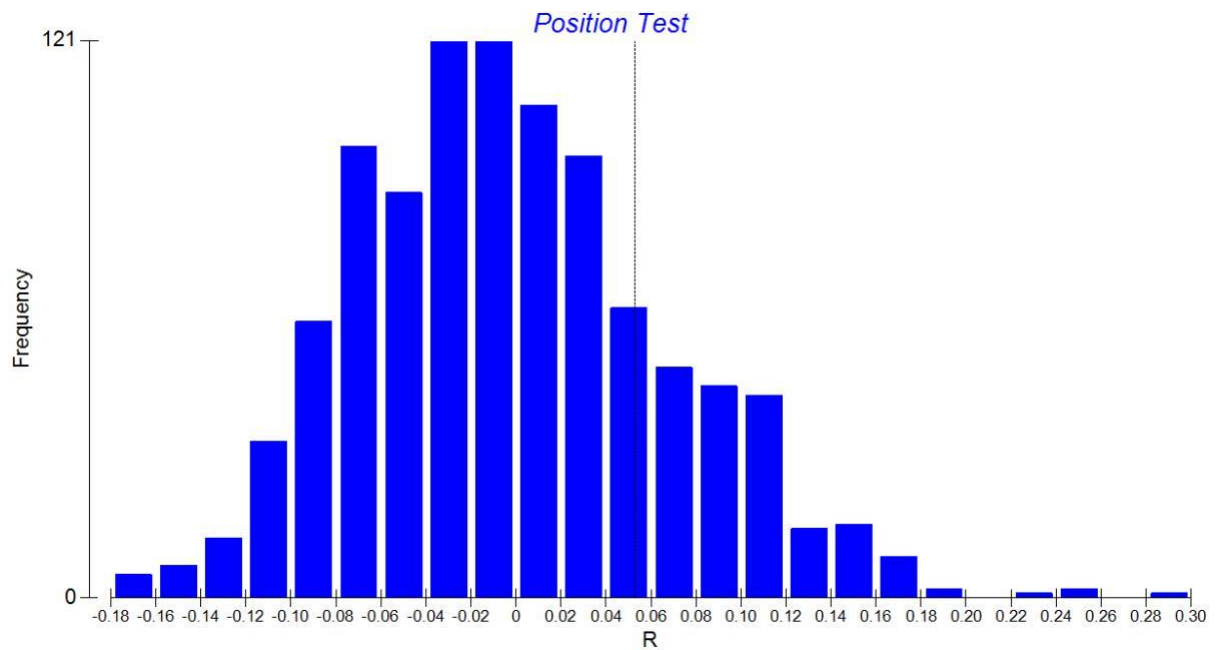
Morphospecies	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Gastropods	0.60	4.68	0.37	32.75	32.75
Sea stars	0.39	2.51	0.29	17.55	50.30
Seapens	0.41	2.46	0.30	17.25	67.56
Sponge	0.43	1.51	0.31	10.58	78.13
Anemonies	0.46	1.50	0.20	10.47	88.60

Rock

Average similarity: 39.32

Morphospecies	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Encrusting sponge	1.74	5.87	1.10	14.94	14.94
Solitary Ascidian	1.30	5.05	1.12	12.84	40.99
Sea urchins	1.30	4.63	1.19	11.78	52.78
Sea stars	1.05	4.19	1.18	10.65	63.43
Brittle stars	1.40	3.55	0.63	9.03	72.46

Appendix 9: Drop camera summary of statistical outputs (ANOSIM) for location and substrate followed by the respective Principal Component Analysis conducted using the environmental data.



PCA

Principal Component Analysis

Data worksheet

Name: Drop camera Location

Data type: Environmental

Sample selection: All

Variable selection: All

Eigenvalues

PC	Eigenvalues	%Variation	Cum.%Variation
1	3.38	56.4	56.4
2	1.17	19.5	75.9
3	0.861	14.3	90.2
4	0.394	6.6	96.8
5	0.133	2.2	99.0

Eigenvectors

(Coefficients in the linear combinations of variables making up PC's)

Variable	PC1	PC2	PC3	PC4	PC5
Latitude_DD	-0.250	0.572	-0.678	-0.056	-0.090
Longitude_DD	-0.508	0.151	-0.035	-0.315	0.569
Depth_metres	0.448	0.380	-0.219	0.466	-0.049
Temperature (Degree Celsius)	-0.457	-0.068	0.113	0.823	0.250
Salinity	-0.512	-0.151	-0.043	0.023	-0.759
Substrate	0.086	-0.692	-0.690	0.056	0.164

PCA

Principal Component Analysis

Data worksheet

Name: Drop camera Substrate

Data type: Environmental

Sample selection: All

Variable selection: All

Eigenvalues

PC	Eigenvalues	%Variation	Cum.%Variation
1	3.38	56.4	56.4
2	1.17	19.5	75.9
3	0.861	14.3	90.2
4	0.394	6.6	96.8
5	0.133	2.2	99.0

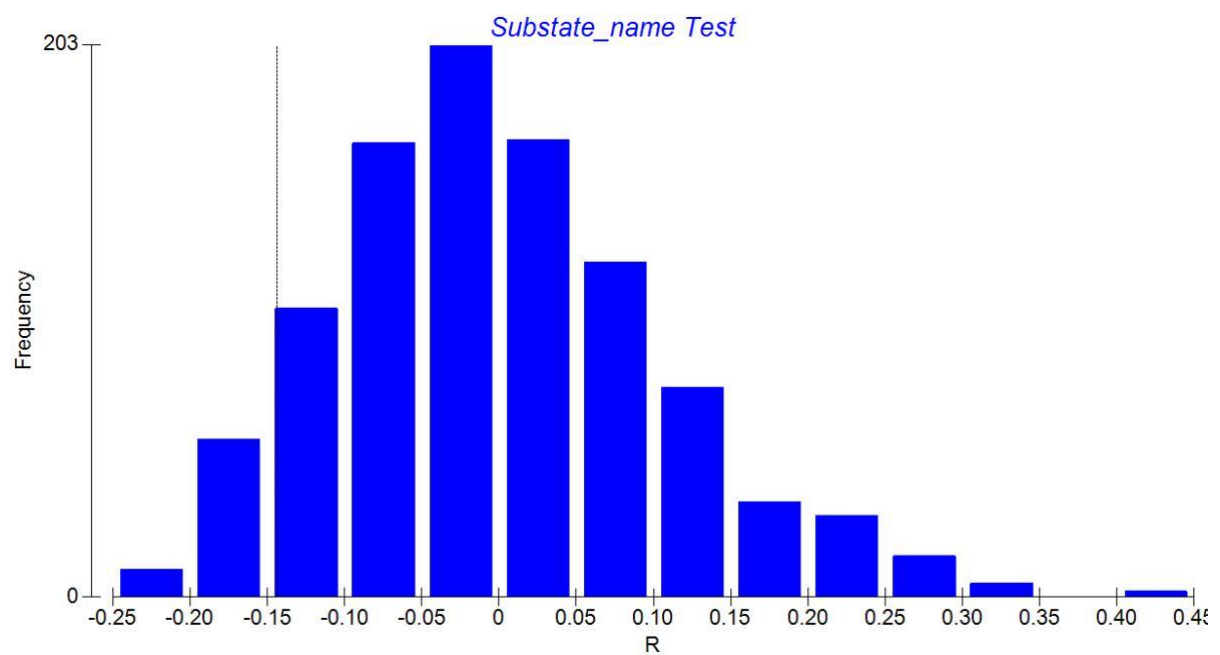
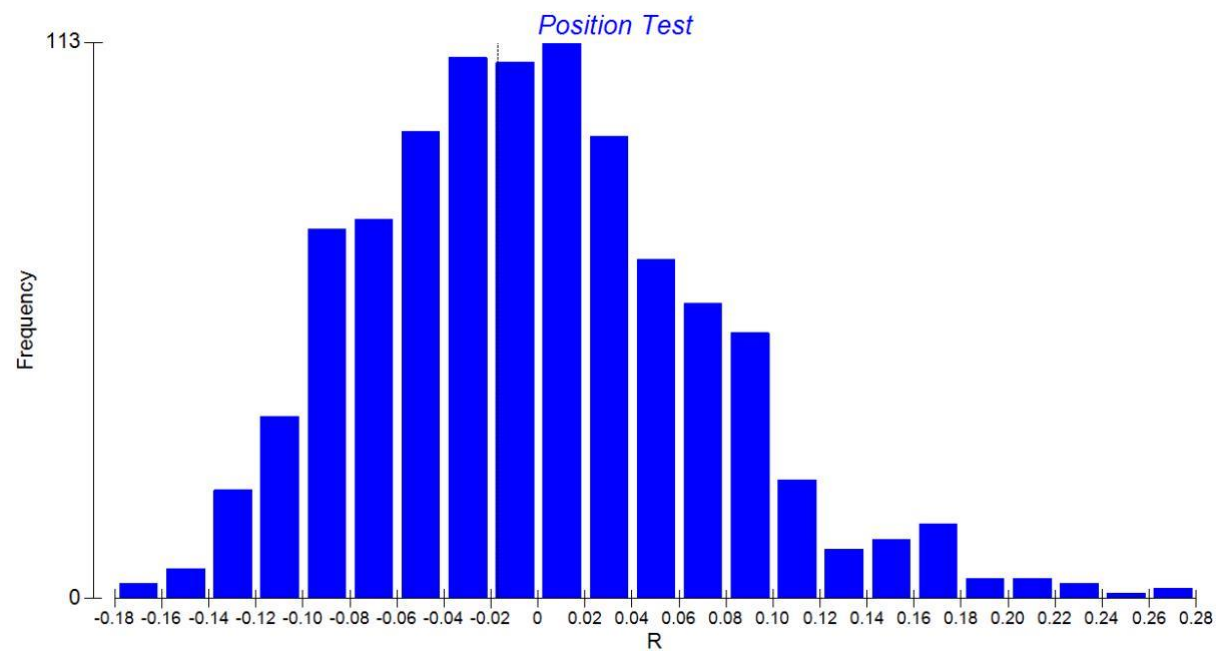
Eigenvectors

(Coefficients in the linear combinations of variables making up PC's)

Variable	PC1	PC2	PC3	PC4	PC5
Latitude_DD	-0.250	0.572	-0.678	-0.056	-0.090
Longitude_DD	-0.508	0.151	-0.035	-0.315	0.569
Depth_metres	0.448	0.380	-0.219	0.466	-0.049
Temperature (Degree Celsius)	-0.457	-0.068	0.113	0.823	0.250
Salinity	-0.512	-0.151	-0.043	0.023	-0.759
Substrate	0.086	-0.692	-0.690	0.056	0.164

Appendix 9.1

Ski-Monkey summary of statistical outputs (ANOSIM) for location and substrate followed by the respective Principal Component Analysis conducted using the environmental data.



PCA

Principal Component Analysis

Data worksheet

Name: Ski-Monkey Location

Data type: Environmental

Sample selection: All

Variable selection: All

Eigenvalues

PC	Eigenvalues	%Variation	Cum.%Variation
1	3.7	61.7	61.7
2	1.26	21.0	82.6
3	0.596	9.9	92.6
4	0.295	4.9	97.5
5	0.126	2.1	99.6

Eigenvectors

(Coefficients in the linear combinations of variables making up PC's)

Variable	PC1	PC2	PC3	PC4	PC5
Latitude_DD	0.305	-0.555	-0.622	-0.309	0.258
Longitude_DD	0.450	0.052	-0.361	0.718	-0.383
Depth_metres	-0.466	-0.225	-0.324	-0.229	-0.642
Temperature (Degree Celsius)	0.476	0.029	0.309	-0.432	-0.592
Salinity	0.504	-0.082	0.191	-0.200	0.142
Substrate	0.086	0.794	-0.495	-0.333	0.060

PCA

Principal Component Analysis

Data worksheet

Name: Ski-Monkey Substrate

Data type: Environmental

Sample selection: All

Variable selection: All

Eigenvalues

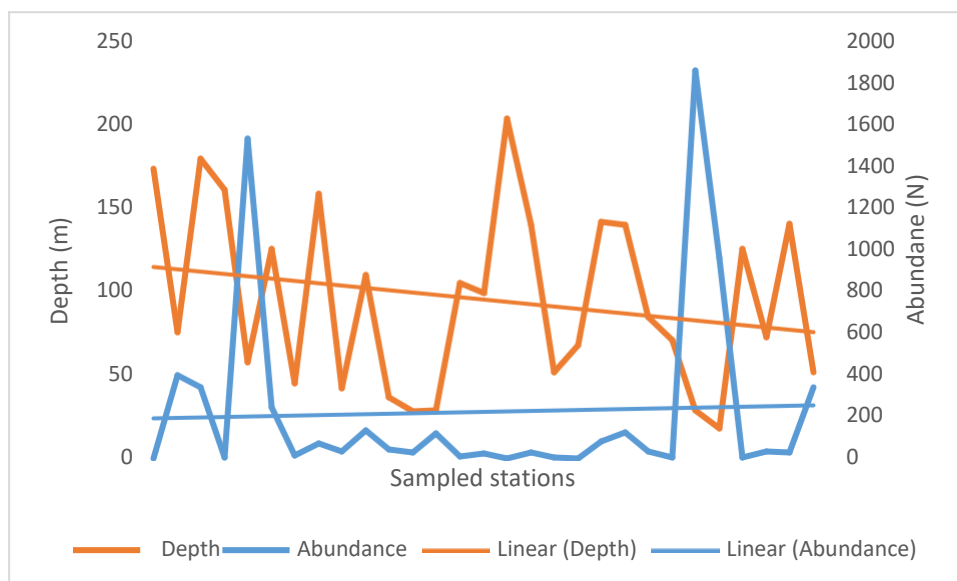
PC	Eigenvalues	%Variation	Cum.%Variation
1	3.7	61.7	61.7
2	1.26	21.0	82.6
3	0.596	9.9	92.6
4	0.295	4.9	97.5
5	0.126	2.1	99.6

Eigenvectors

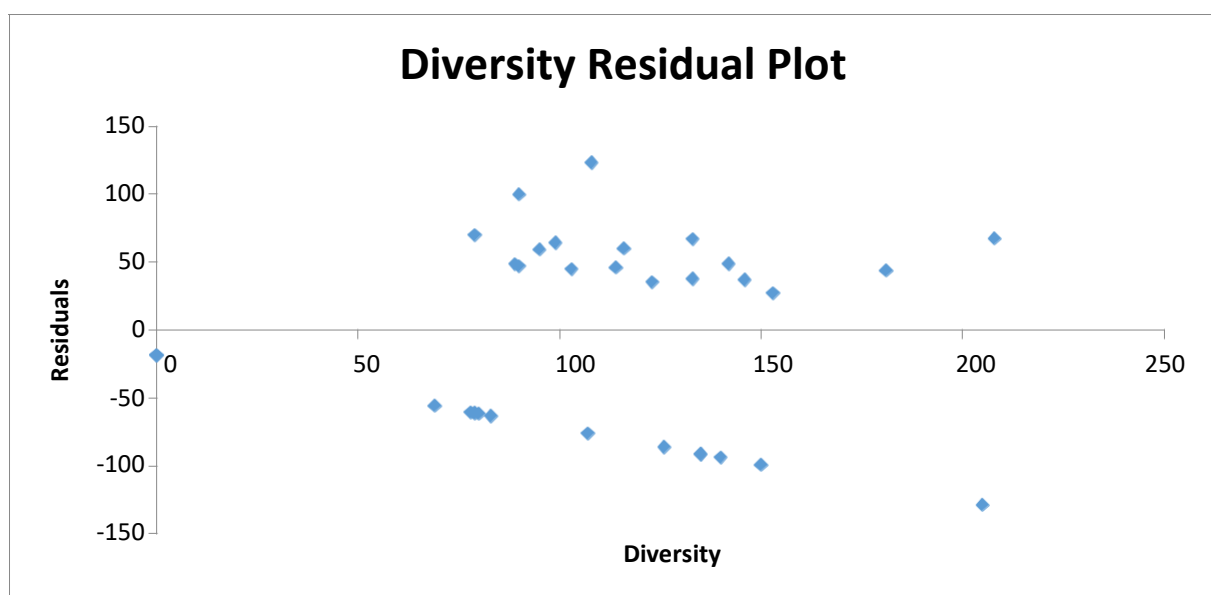
(Coefficients in the linear combinations of variables making up PC's)

Variable	PC1	PC2	PC3	PC4	PC5
Latitude_DD	0.305	-0.555	-0.622	-0.309	0.258
Longitude_DD	0.450	0.052	-0.361	0.718	-0.383
Depth_metres	-0.466	-0.225	-0.324	-0.229	-0.642
Temperature (Degree Celsius)	0.476	0.029	0.309	-0.432	-0.592
Salinity	0.504	-0.082	0.191	-0.200	0.142
Substrate Name	0.086	0.794	-0.495	-0.333	0.060

Appendix 10: Species abundance in relation to depth across all sampled stations.



Appendix 11: Diversity Residual Plot indicating the variation in the model.



Appendix 12: Drop camera Percentage cover (%) (means and standard deviations) of sediment type (rock and sand) and cumulative percentage cover (%) for all samples per station.

Sampling Site	Station	Percentage Cover (%)			
		Rock		Sand	
		Mean	SD	Mean	SD
Cape Point EBSA	141	0	0	99.67	0.58
Cape Point EBSA	172	68.53	35.84	3.37	7.30
Cape Point EBSA	176	65.10	8.08	0.70	1.06
Cape Point EBSA	199	0	0	99.33	1.21
Cape Point EBSA	203	56.91	13.57	7.96	8.49
Cape Point EBSA	229	78.53	32.68	10.23	25.64
Cape Point EBSA	233	0	0	98.85	1.68
TMNP MPA	284	78.91	37.23	5.04	20.67
TMNP MPA	289	0	0	95.88	3.36
TMNP MPA	340	83.89	23.66	3.79	12.09
TMNP MPA	346	0	0	98.23	2.50
TMNP MPA	375	0	0	97.73	4.56
TMNP MPA	377	12.33	18.82	47.33	12.74
TMNP MPA	395	0	0	97.54	2.33
TMNP MPA	420	22.92	41.50	74.58	41.50
TMNP MPA	431	0	0	99.73	0.47

TMNP MPA	436	62.17	48.28	33.00	48.28
TMNP MPA	461	4.00	6.92	95.53	7.83
RI MPA	501	0	0	97.50	0.71
RI MPA	517	0	0	99.71	0.49
RI MPA	519	52.50	45.54	19.60	41.33
RI MPA	520	52.25	42.79	8.25	28.58
RI MPA	524	33.36	46.38	62.91	49.88
RI MPA	525	9.80	30.99	88.40	31.24
RI MPA	527	18.52	18.49	25.03	27.23
RI MPA	529	7.00	9.00	17.67	26.91
RI MPA	545	0	0	49.72	53.19
RI MPA	549	67.00	45.86	28.43	48.55
RI MPA	564	1.88	5.30	47.10	33.79
RI MPA	616	31.62	31.20	41.85	48.62
	All Samples	26.91	18.07	370.40	19.76

Appendix 12.1: Ski-Monkey Percentage cover (%) (means and standard deviations) of sediment type (rock and sand) and cumulative percentage cover (%) for all samples per station.

Sampling Site	Station	Percentage Cover (%)			
		Rock		Sand	
		Mean	SD	Mean	SD
Cape Point EBSA	176	27.53	39.14	9.00	20.12
Cape Point EBSA	233	0	0	73.20	24.06
Cape Point EBSA	236	0	0	99.40	1.34
TMNP MPA	281	0	0	97.40	1.52
TMNP MPA	289	0.60	1.34	92.60	6.99
TMNP MPA	363	0	0	99.80	0.45
TMNP MPA	375	0.25	0.56	99.55	0.62

TMNP MPA	395	0	0	99.20	1.30
RI MPA	524	0	0	99.20	0.84
RI MPA	525	0	0	99.40	1.34
RI MPA	529	44.00	44.64	0	0
RI MPA	537	0	0	98.60	1.14
RI MPA	546	36.10	47.69	46.30	44.40
RI MPA	568	10.13	20.50	59.80	54.49
RI MPA	570	22.95	36.34	51.80	48.51
RI MPA	588	42.48	40.84	48.73	48.75
RI MPA	606	0	0	99.38	0.91
RI MPA	609	4.00	8.94	79.75	44.58
	All Samples	10.45	13.33	75.17	16.74

Appendix 12.2: Drop camera summary table showing the mean and standard deviation of Species richness (S), Abundance (N), Evenness (J') and Diversity (H' (Loge)) for all stations.

Station	S		N		J'		H'(loge)	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD
141(Cape Point EBSA)	2.33	0.58	6.63	0.57	0.97	0.05	0.79	0.18
172(Cape Point EBSA)	5.15	1.77	10.37	1.98	0.94	0.02	1.50	0.31
176(Cape Point EBSA)	8.80	1.23	15.27	1.63	0.96	0.01	2.08	0.14
177(Cape Point EBSA)	2	0	6.32	9.36	1	0	0.69	1.17
199(Cape Point EBSA)	2.33	0.52	6.71	0.60	0.97	0.04	0.80	0.17
203(Cape Point EBSA)	8.44	1.20	15.47	1.50	0.97	0.01	2.05	0.14
229(Cape Point EBSA)	4.41	1.47	9.35	1.63	0.94	0.02	1.35	0.27
233(Cape Point EBSA)	2.69	0.85	7.08	0.95	0.96	0.04	0.90	0.24

284(TMNP MPA)	4.04	1.58	8.57	1.78	0.93	0.03	1.23	0.35
289(TMNP MPA)	2.88	0.64	7.54	0.82	0.92	0.03	0.99	0.20
340(TMNP MPA)	4.89	1.85	9.76	2.05	0.94	0.03	1.42	0.36
346(TMNP MPA)	2.62	0.50	7.07	0.62	0.96	0.04	0.90	0.16
375(TMNP MPA)	2.87	0.92	7.33	1.19	0.95	0.04	0.95	0.26
377(TMNP MPA)	4.33	1.15	10.35	1.66	0.98	0.02	1.40	0.27
395(TMNP MPA)	3.08	0.76	7.60	0.94	0.94	0.03	1.03	0.21
420(TMNP MPA)	3.33	0.98	7.90	1.18	0.94	0.04	1.07	0.26
431(TMNP MPA)	2.27	0.47	6.60	0.46	0.97	0.04	0.78	0.14
436(TMNP MPA)	4.17	1.72	8.79	1.97	0.94	0.03	1.26	0.37
461(TMNP MPA)	2.41	0.71	7.00	1.14	1.00	0.02	0.83	0.24
501 (RI MPA)	4.50	0.71	8.80	0.70	0.90	0.003	1.35	0.15
519(RI MPA)	4.40	1.78	9.32	2.08	0.95	0.03	1.33	0.38
520(RI MPA)	5.33	1.56	10.43	1.87	0.94	0.02	1.53	0.30
524(RI MPA)	3.64	1.29	8.17	1.54	0.93	0.04	1.14	0.32
525(RI MPA)	2.60	0.52	7.03	0.64	0.96	0.04	0.89	0.17
527(RI MPA)	6.85	2.12	13.77	3.26	0.97	0.02	1.81	0.37
529(RI MPA)	4.67	1.29	10.59	1.70	0.97	0.01	1.46	0.24
545(RI MPA)	2.29	0.49	6.65	0.56	0.98	0.04	0.79	0.16
549(RI MPA)	3.29	1.11	7.93	1.36	0.95	0.04	1.08	0.31
564(RI MPA)	3.63	1.19	8.33	1.38	0.95	0.04	1.16	0.32
616(RI MPA)	4.23	1.36	9.64	2.12	0.95	0.03	1.33	0.34
All samples	3.95	1.08	8.88	1.64	0.95	0.03	1.20	0.28

Appendix 12.3: Ski-Monkey Summary table showing the mean and standard deviation of Species richness (S), Abundance (N), Evenness (J') and Diversity (H' (Loge)) for all stations.

Station	S		N		J'		H'(loge)	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD
176(Cape Point EBSA)	8.2	2.17	16.51	2.90	0.96	0.01	1.98	0.28
233(Cape Point EBSA)	6	1.58	13.77	2.63	0.96	0.02	1.67	0.27
236(Cape Point EBSA)	3.4	0.55	10.07	0.80	0.99	0.02	1.20	0.14
281(TMNP MPA)	4.2	0.45	10.88	0.46	0.95	0.01	1.36	0.08
289(TMNP MPA)	4.6	1.14	11.62	1.45	0.97	0.02	1.44	0.23
363(TMNP MPA)	3.2	0.45	9.67	0.44	0.99	0.03	1.14	0.09
375(TMNP MPA)	3.8	0.84	10.69	1.29	0.98	0.03	1.29	0.19
395(TMNP MPA)	3.4	0.55	9.95	0.63	0.98	0.03	1.19	0.12
524(RI MPA)	3.8	0.84	10.28	0.83	0.96	0.03	1.26	0.17
525(RI MPA)	3.2	0.45	9.75	0.58	0.99	0.02	1.15	0.11
529(RI MPA)	4	1.22	11.24	2.06	0.99	0.01	1.34	0.27
537(RI MPA)	4	0.71	10.54	0.71	0.95	0.03	1.31	0.14
546(RI MPA)	8.4	1.95	16.07	2.32	0.95	0.01	2.00	0.22
568(RI MPA)	4.6	2.07	11.60	2.93	0.97	0.03	1.41	0.38
570(RI MPA)	5.4	1.67	12.56	2.26	0.96	0.03	1.57	0.31
588(RI MPA)	5.4	1.52	13.10	2.28	0.97	0.02	1.60	0.27
606(RI MPA)	3.8	0.84	10.52	1.11	0.97	0.02	1.28	0.18
609(RI MPA)	3.8	1.10	10.66	1.61	0.98	0.02	1.28	0.24
All samples	4.62	1.12	11.64	1.52	0.97	0.02	1.42	0.21