Epibenthic biodiversity, habitat characterisation and anthropogenic pressure mapping of unconsolidated sediment habitats in Algoa Bay, South Africa

By Hannah Jessie Truter

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Submitted in fulfilment in the case of a dissertation for the degree of Master of Science in Biological Oceanography in the Faculty of Science to be awarded at the Nelson Mandela University

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Declaration

I, Hannah Jessie Truter, student number 217073948, hereby declare that the thesis "Epibenthic biodiversity, habitat characterisation and anthropogenic pressure mapping of unconsolidated sediment habitats in Algoa Bay, South Africa" for the Masters of Science in Biological Oceanography to be awarded is my own work and that it has not previously been submitted for assessment or completion of any postgraduate qualification to another University or for another qualification.

Signature: ______________________

Date: 8 March 2019

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Summary

Implementation of an ecosystem-based management approach for marine systems requires a comprehensive understanding of the biophysical marine environment and the cumulative human impacts at different spatio-temporal scales. In Algoa Bay, South Africa, this study describes the epibenthic communities occurring in unconsolidated marine habitats. It further investigates the potential abiotic factors that influence their distribution and abundance, compares epibenthic communities with existing habitat information and evaluates the protection status of the marine environment in the Bay. Seabed imagery, covering a total area of 171.4m^2, and sediment samples were collected from 13 stations from which 106 epibenthic species were identified. Multivariate analyses revealed two statistically distinct communities that did not align with the Algoa Bay benthic habitat types defined in the current National Biodiversity Assessment (NBA, 2012). Further assessment indicated that community differences were driven by the presence of rock substrate. A range of abiotic factors were tested against the epibenthic communities to explore patterns and identify potential drivers. The combination of abiotic factors depth, mean grain size, mean bottom temperature and mean bottom current explained 55% fitted variation in epibenthic data. The degree of long-term variability in several of these parameters were likewise identified as explanatory variables, including bottom temperature, current speed and dissolved oxygen. The link between abiotic factors and the epibenthic communities observed indicate that these variables can act as surrogates for habitat mapping in the future. The existing and proposed Marine Protected Area (MPA) in conjunction with the NBA 2012 habitat types does well in protecting the majority of habitats in the Bay, however there remain habitats that lack protection. Utilising the benthic communities and potential drivers identified in this study, the proposed MPA boundary delineations should be somewhat altered to include missing habitat types.

Key words: Unconsolidated marine sediment habitats, underwater imagery, epibenthic communities, environmental drivers, marine spatial planning, Algoa Bay

Chapter 1: Introduction and background

Overview

The ocean plays a fundamental role within the global economy and in supporting societies (Costanza *et al.,* 1997; Beaumont *et al.,* 2006; Worm *et al.,* 2006). Marine ecosystems supply services that sustain life on land by regulating ecosystem health and functioning (Snelgrove *et al.,* 1997; Duarte, 2000; Palumbi *et al.,* 2009; Costanza *et al.,* 2011; Böhnke-Henrichs *et al.,* 2013). These ecosystem services can be defined as the benefits that humans gain from effective operational ecosystems or ecological characteristics that directly or indirectly contribute to human well-being (Chee, 2004; Costanza *et al.,* 2011; Guisado-Pintado *et al.,* 2016). Ecosystem services, both terrestrial and marine, can be divided into four core groups: provisioning, supporting, regulating and cultural, each with dynamic interacting relationships (Millennium Ecosystem Assessment, 2005; Townsend *et al.,* 2011; Li *et al.,* 2016; Perrings, 2016). Coastal and benthic ecosystem services include the provision of food (from fisheries), oxygen production, energy production (such as wind and wave power), transportation (through shipping), mineral extraction (e.g. diamonds, gold, silver, oil, etc.) and water supply (through desalination of salt water for human consumption). Regulatory ecosystem services include flood prevention, seawater intrusion, erosion, climate regulation and waste water treatment (e.g. coastal plant life such as mangroves, wetlands and estuaries act as buffers and filters) (Thrush and Dayton, 2002; Borja, 2014; Handley *et al.,* 2014; Hattam *et al.,* 2015; Martin *et al.,* 2016). Ecosystem services support the maintenance of biodiversity, habitats and resilience of ecosystems while providing cultural services that include recreation (such as swimming, sailing, diving and kayaking), cognitive value (e.g. education and research) and cultural heritage (Chee, 2004).

The global human population strongly depends on the ocean as it provides as much as two thirds of the planet's ecosystem services natural capital (Millennium Ecosystem Assessment, 2005; Worm *et al.,* 2006; Nahuelhual *et al.,* 2017). However, ecosystem services often fall into the category of open access or public services, providing little incentive for beneficiaries to manage these services sustainably (Hardin, 1968; Chee, 2004). This has led to an increasing risk of habitat degradation, shifts in species distributions and loss of ecosystem function (Arkema *et al.,* 2015; Hattam *et al.,* 2015). Coasts are a focal point for human migration and economic activities and are exposed to increasing human induced pressures (Glaser and Glaeser, 2014). The rate of global change has greatly surpassed our policy response, adding to the importance of including ecosystem services as part of marine resource management, regardless of the complex and dynamic relationships in terms of trade-offs and synergies (Glaser and Glaeser, 2014; Li *et al.,* 2016). Growing evidence has shown that humanity is driving global change, pushing us into a new geological epoch referred to as the Anthropocene (Griggs *et al*., 2013; McCauley *et al.,* 2015). Recognising this, the introduction of global sustainable development goals (United Nations, 2012) with the aim to meet the needs of the present while safeguarding Earth's life-support system, on which the welfare of current and future generations depends have been recommended (Griggs *et al*., 2013; Cormier and Elliott, 2017). Sustainable Development Goal 14 focuses specifically on the conservation and sustainable use of the oceans, seas and marine resources. Goal 14 prescribes several targets for the next ten years including reducing marine pollution, addressing the impacts of ocean acidification, sustainably managing and protecting coastal and marine areas, and increasing scientific knowledge by developing research capacity (international sharing of technology) (United Nations, 2015).

As a result of increasing access and utilisation of marine resources it has become more important to improve our knowledge and understanding of marine ecosystems. In recent years the number of scientific publications focusing on marine ecosystems has increased (to between 1100-1500 publications per year in the past five years) and several grand challenges have been identified for future research (Borja, 2014). These challenges address complex problems such as understanding the role of biodiversity in maintaining ecosystem functioning (Thrush and Dayton, 2002; Thompson *et al.,* 2012; Belley and Snelgrove, 2016), the relationships between human impacts and ecosystems (Halpern *et al.,* 2008; 2015; Trebilco *et* al., 2011; Korpinen *et al*., 2012;2013), and the impact of global change on marine ecosystems (Occhipinti-Ambrogi, 2007; Poloczanska *et al.,* 2013; Magris *et al.,* 2014; Molinos *et al.,* 2016). These grand challenges also identified the need for ecosystem-based management and spatial planning to allow ecosystems to recover, and to improve marine protection for ecosystem service delivery (Crowder and Norse, 2008; Douvere, 2008; Borja *et al.,* 2010; Katsanevakis *et al.,* 2011; Townsend *et al.,* 2011). The increase in pressure on marine resources has resulted in a policy shift towards implementing ecosystem-based management at a global scale which, to be successful requires rigorous environmental and ecosystem level data (Guarinello *et al*., 2010; Bohnke-Henrichs *et al*., 2013; Hattam *et al*., 2015; Kirkman *et al*., 2016).

The Convention on Biological Diversity (CBD) signed by 150 countries, including South Africa, at the Rio summit (United Nations CBD, 1992), forms the foundation on which both marine and terrestrial ecosystem-based management was built (Beaumont *et al*., 2007; Sparks *et al*., 2011). This approach was further adapted in the marine environment to the Ecosystem Approach to Fisheries (EAF) management (Cochrane *et al*., 2004; Atkinson & Clark, 2005; Kleisner *et al*., 2015; Moore *et al*., 2016). EAF management aims to conserve ecosystem structure and functioning whilst maintaining sustainable ecosystem services and recognises various scales in both time and space (Garcia *et al*., 2003). The appropriate balance between conservation goals and biodiversity use is vital to the success of this approach (Garcia & Cochrane, 2005; Moore *et al*., 2016). Internationally there have been many attempts to develop rules of engagement at different levels for integrated ocean strategies (Pinarbasi *et al*., 2017). Although most strategies have widely diverse objectives, they all fundamentally lead to ecosystem-based management aimed at balancing multiple uses of marine space to achieve ecological, economic and social objectives (Pinarbasi *et al*., 2017). Ecosystem-based Marine Spatial Planning (MSP) has been considered a useful management tool as it tackles the heterogeneity of marine ecosystems in a practical manner, concentrates on influencing the behaviour of humans and their activities, while guiding single-sector management toward integrative decision making (Douvere, 2008; Foley *et al.,* 2010; Collie *et al.,* 2013; Qui and Jones, 2013; Dominguez-Tejo *et al.,* 2016).

Qui and Jones (2013) describe the two different over-arching views on sustainability in the MSP policy landscape. 'Soft' sustainability considers economic growth as the foundation for societal well-being (implemented as integrated-use MSP) while 'hard' sustainability is based on a foundation of healthy ecosystems (implemented as ecosystem-based MSP) as the foundation for the well-being of society. Ultimately, if ecosystems collapse, the two views respectively predict that society will either adapt or collapse. Based on growing evidence globally for the impacts of collapsed ecosystems on human well-being (Qui and Jones, 2013; Pinarbasi *et al*., 2017), and the recommendations of the SD Goal 14, the present study supports an ecosystem-based approach to MSP and the adoption of transdisciplinary approaches to reach this goal in South Africa.

A South African perspective

Governance of South Africa's marine and coastal environment utilises several legislative instruments that aim to achieve goals for both utilisation and protection of marine living resources in the continental and offshore (Prince Edward Islands) Exclusive Economic Zones (EEZs). These include the United Nations Convention on the Law of the Sea (UNCLOS, 1982), the Marine Living Resources Act (1998), the National Environmental Management Act (NEMA, 1998) with three supporting acts, the Protected Areas Act (2003), Biodiversity Act (2004), the Integrated Coastal Management Act (2008), and most recently the draft Marine Spatial Planning Bill (2017). The latter calls for biodiversity plans to be formulated from data-derived assessments such as the marine component of the National Spatial Biodiversity Assessment (Lombard *et al.,* 2004) and National Biodiversity Assessment (NBA, Sink *et al.,* 2012a), both focusing exclusively on the continental EEZ. The forthcoming NBA in preparation will also include the offshore Prince Edward Islands (NBA 2019 in prep). One of the products from the 2012 marine and coastal component of the NBA was an updated national-scale marine ecosystem classification and habitat map (Fig. 1.1) which incorporated existing data sets of geological features, substrate (digitised texture map and grain size) and wave exposure. The NBA (2012) also identified several marine research priorities that aimed to improve marine ecosystem-based management in South Africa (Sink *et al*., 2012a) as follows:

- Identifying marine ecosystem priority areas including sensitive habitats and key areas for resource recovery;
- Improving the science base for South Africa's Marine Protected Areas (MPAs) through species inventories, fine-scale habitat mapping, and coordinated monitoring initiatives;
- Improving the knowledge base to support the understanding of climate change in South Africa, particularly focusing on long-term monitoring;
- Refining the marine and coastal habitat classification and map based on testing the validity of the current classification, high resolution bathymetric mapping, and systematic marine biodiversity surveys across broad ecosystem groups.

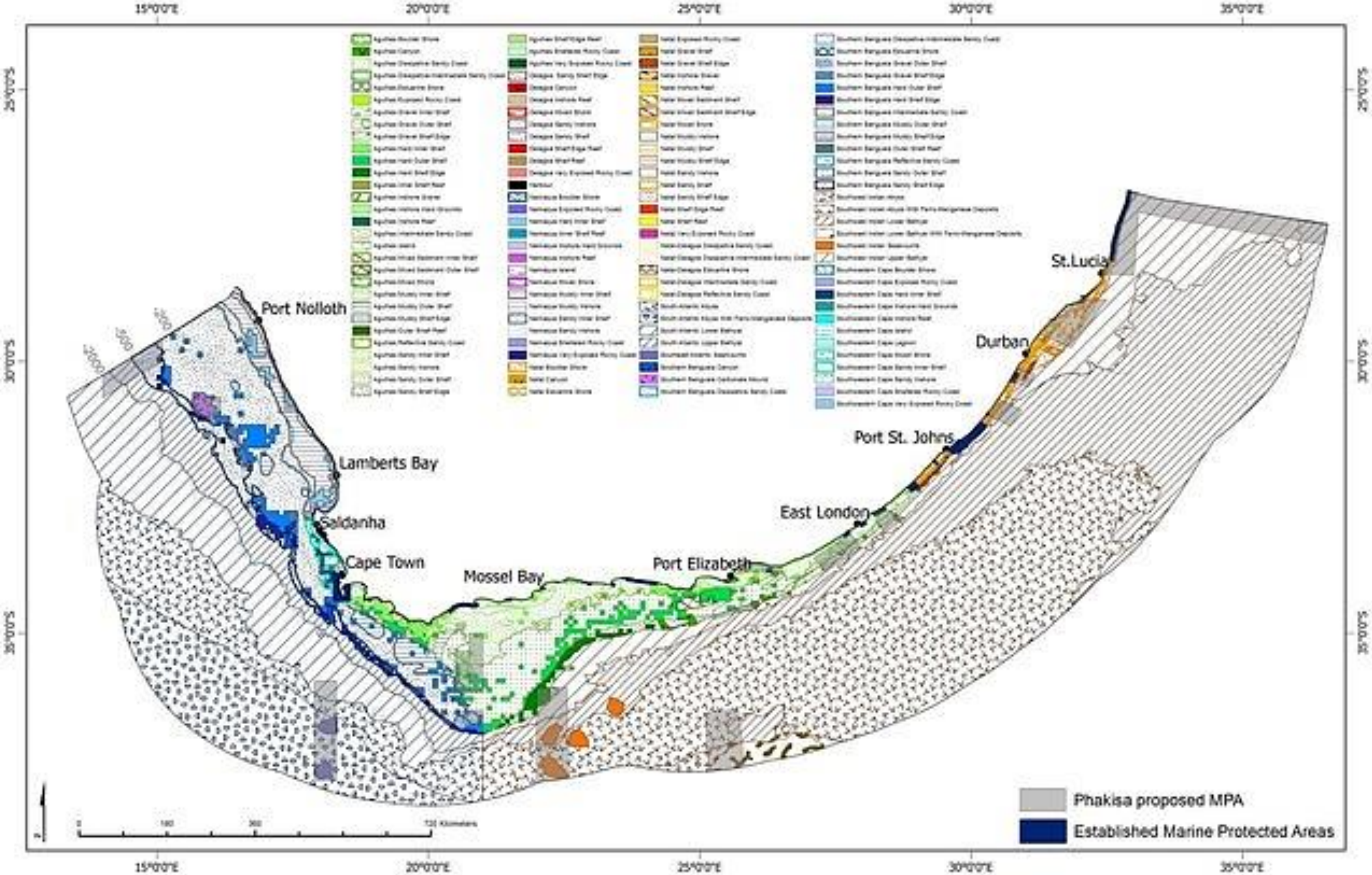

Figure 1.1: Coastal and offshore benthic habitat types in South Africa with the existing (in blue) and proposed (in grey) Marine Protected Area network for the continental Exclusive Economic Zone (adopted from Sink et al., 2012a and Harris et al., 2014).

The most recent legislation, the MSP Bill (2017) defines MSP as an iterative, phased process consisting of several steps including the development of a MSP framework, knowledge and information system, marine area plans and effective implementation, monitoring and evaluation of the marine area plans. The MSP framework (2017) proposed South Africa's ocean space be divided into smaller bio-geographic marine areas from which marine area plans will be prepared sequentially. The consequent experience gained from preparing each plan will further inform the subsequent plans (Dorrington *et al.,* 2018).

Algoa Bay is considered the best monitored coastal area in Africa and the Southern Hemisphere owing to the establishment of the Algoa Bay Sentinel Site for Long Term Ecological Research in 2007 (Atkinson *et al*., 2016; Dorrington *et al.,* 2018). Data arising from this Sentinel Site has provided a wealth of physical and environmental data to improve understanding of the oceanographic processes that occur within the Bay. Algoa Bay includes a small MPA, namely Bird Island, that forms part of a larger proposed MPA (Addo MPA). The proposed Addo MPA was formulated in response to Operation Phakisa (Department of Environmental Affairs, 2014b), a national initiative aimed at unlocking South Africa's economic potential from the ocean while also improving the protection level of the EEZ from less than 0.5% to 5%. Since ecosystem-based MSP and the establishment of effective MPAs requires a comprehensive understanding of how marine ecosystems function, fundamental steps are required to survey, classify and monitor marine ecosystems and environmental processes governing them in regions where MSP is to be undertaken. Subsequently, Algoa Bay has been identified as an ideal case study for MSP in South Africa (Dorrington *et al.,* 2018), owing to intensive research in the area, as well as extensive existing data sets existing and further research required. Understanding benthic communities in Algoa Bay has however been identified as a foundational knowledge gap and requires further research

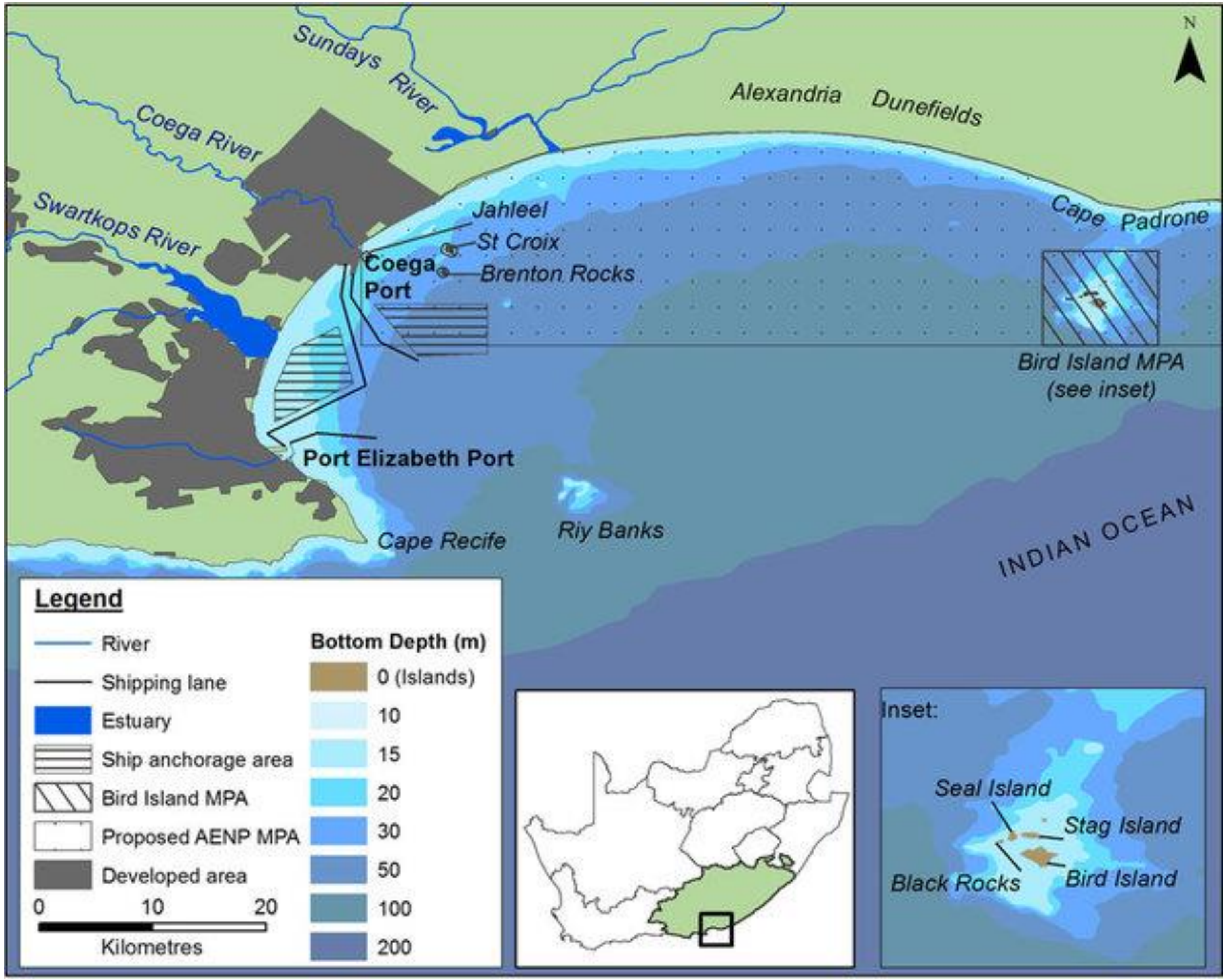

Figure 1.2: The position of Algoa Bay on the south-eastern Cape coastline of South Africa (after Melly et al., 2017). AENP= Addo Elephant National Park, referred to in text as Addo MPA.

Study Area

Algoa Bay (Fig. 1.2) is situated on the southern Cape coast of South Africa in the transition zone between the Agulhas current and the wide Agulhas bank (Goschen and Schumann, 2011; Goschen *et al*., 2012). Further oceanographic processes such as large solitary meanders, shear-edge eddies and plumes are known to influence the dynamics of the Bay often leading to enhanced upwelling (Goschen and Schumann, 2011; Goschen *et al*., 2012). Nearshore currents tend to follow the local bathymetry that slopes very gently towards the south southeast at a 0.15^0 and orientation of the shore line and harbour walls (Goschen and Schumann, 2011). The Bay is well known for the south-westerly winds that dominate throughout the year while in summer the south-easterly winds often drive upwelling at the headlands (Pattrick *et al*., 2013). Coastal upwelling is common within the Bay and is considered an important driver of marine diversity composition and distribution (Goschen and Schumann, 2011; Pattrick *et al*., 2013). The rivers flowing into Algoa Bay input minor amounts of fresh water only influencing the surface water with salinity remaining around the oceanic average of approximately 35.2 (Schumann *et al*., 2005). The Bay has been shaped and defined by the resistant quartzite of the Cape Supergroup geology, which play a fundamental role in wave action and sediment transport (Goschen and Schumann, 2011). Large areas of unconsolidated sediments, including sand and mud, along with several biologically important reefs, occur within Algoa Bay (Pattrick *et al*., 2013). Sand mobility is high in the region and, depending on wave and ocean conditions, volumes of sand can cover low reef areas (Goschen and Schumann, 2011). The construction of two harbours within Algoa Bay caused an interruption in the longshore transport of sediment along the coastline. This has led to the ongoing formation of considerable beach area on the southern side of both harbours and severe erosion on the northern side. Algoa Bay marine protection includes Bird Island MPA that consists of Bird, Seal, Stag Islands, Black Rocks and the surrounding area.

Algoa Bay hosts a rich array of marine biodiversity including several top predators such as vulnerable seabird species endemic to the Bay (Klages *et al.,* 1992; Crawford *et* al., 1983; 1995; Green *et al.,* 2015; Connan *et al.,* 2016; 2017), dolphins and whales (Karczmarski *et al.,* 1999a; 1999b; 2000a; 2000b; Reisinger and Karczmarski, 2010; Koper *et al*., 2016a; b; 2016b; Bouveroux *et al*., 2017, Melly *et al*., 2017). The oceanographic dynamics of the Bay support a productive ecosystem for pelagic fish and squid that serve as prey for top predators (Beckley and van Ballegooyen, 1992; Beckley and Hewitson, 1994; Pattrick *et* al., 2013; Pattrick and Strydom, 2014; Costalago *et al*., 2018). More recent research focusing on diatoms and bacterial communities in the Bay has led to discovery of high microbial diversity in the Algoa Bay system (Pitcher *et al*., 2014; Okaiyeto *et* al., 2015; Matobole *et al.,* 2017; Ntozonke *et al*., 2017; Waterworth *et al*., 2017). Benthic habitat research has centred on the reefs in the Bay with several new species of coral and sponges being described (Mcfadden and Ofwegen, 2012; Matcher *et al*., 2016).

Research rationale

Research within Algoa Bay has to date focused predominately on top predators, physical oceanography and pelagic fish species (Dorrington *et al*., 2018), however there has been limited research focused on the ecology of the dominant benthic habitat type (unconsolidated marine sediment habitats) and its epibenthic communities. Epibenthic organisms are defined as species that protrude from, live on or are attached to benthic substrates. These animals (often invertebrates) are favoured during biological monitoring owing to their often sedentary life style and longer life expectancy (Levin *et al.*, 2010; Rombouts *et al*., 2013; Siddig *et al*., 2016). Epibenthic species can increase or decrease in abundance, diversity and even size, depending on their tolerance to different human or environmental pressures (Olsgard *et al.,* 2003; Korpinen *et* al., 2013). Changes in abundance of key benthic species can trigger changes in the community assemblages, for example, a decline in lobster numbers are followed by an increase in sea urchins, causing a decline of kelp forests over time (Olsgard *et al.,* 2003; Hiscock *et al.,* 2004; Shannon *et al*., 2010; Ortiz *et al*., 2013). Owing to their responsiveness to environmental change, epibenthic species and communities are commonly used as indicators of ecosystem health (Shin and Shannon, 2010; Shin *et al.,* 2010; Shannon *et* al., 2010; Rombouts *et al*., 2013; Lockerbie *et al.,* 2016). In order to use epibenthic communities as indicators, extensive long-term surveys, inventories and baseline data should be gathered to define natural patterns of diversity, abundance and the relationship with environmental conditions

This study aims to describe the unconsolidated marine sediment habitat type and resident epibenthic communities in Algoa Bay, using data quantified from underwater imagery and sediment collections, and to investigate the possible abiotic factors that may influence species distribution patterns in the Bay. The study then aimed to undertake a spatial assessment of human pressures and marine protection in the Bay using a decision support tool to identify optimal boundaries for protection targets. This work directly addresses some priorities identified by the 2012 NBA, and generates fundamental ecological knowledge of dominant benthic ecosystems in Algoa Bay, in support of ecosystem-based MSP at a local scale. The research products contribute to an evolving spatial data set on benthic marine habitats for South Africa, and thus contribute towards informing national-scale MSP processes.

Chapter outline

As a result of the limited amount of information available on unconsolidated marine sediment habitats in Algoa Bay, Chapter Two focuses on describing epibenthic communities in unconsolidated habitats in Algoa Bay. Using seabed imagery, epibenthic species were identified and new community types were identified based on species composition and abundance, providing valuable baseline knowledge. The 2012 NBA benthic habitat types within Algoa Bay were compared with the statistically-defined epibenthic communities to assess the present NBA habitat types and further refine habitat classifications for future NBAs.

Chapter Three identifies the influences of potential physical drivers on observed patterns in unconsolidated habitat epibenthic communities in Algoa Bay. Several abiotic factors included in this analysis were derived from existing long-term monitoring stations within the Bay. These stations did not directly align with biological sampling stations and were interpolated accordingly. Both the long-term mean and variation of the interpolated drivers were considered during the analysis. The factors identified as prominent drivers of the epibenthic community patterns observed can be used to further inform habitat mapping in the Bay and potentially serve as a surrogate for unsampled and unmapped species distributions.

Human pressures and protection status directly linked to benthic habitats in Algoa Bay are described in Chapter Four. The position of the existing and proposed MPAs in the Bay were assessed on the protection status of each NBA-defined unconsolidated habitat to determine if protection targets (20% of each habitat type) have been met. Two alternative MPAs options were recommended using the Marxan decision support tool. Marxan utilises both habitat type and human pressure data, and aims to meet targets for biodiversity in areas of low cost (i.e. areas least utilised by humans). Given that this analysis focused only on the benthic component of the Bay, it serves to demonstrate how human–use data layers can inform MPA design, and should not be interpreted as a conclusive recommendation for MPA location.

The key outcomes, interpretation and implications of this study are presented in Chapter Five, including recommendations for future priority research in Algoa Bay. The integration of individual outputs from each of these Chapters will advance the understanding of unconsolidated marine sediment habitats, the potential drivers of epibenthic species distribution and abundance, and the methods available to manage these and other habitat types in Algoa Bay. This study demonstrates the distinct links between quantitative benthic biodiversity research and MSP processes that rely on accurate spatial biodiversity information.

Chapter 2: Epibenthic diversity of unconsolidated sediment habitats in Algoa Bay, South Africa

Introduction

Unconsolidated marine sediment habitats are one of the most expansive and oldest habitats globally, yet their patterns of species richness, and ecological functioning remain poorly understood (Snelgrove *et al.*, 1997; Leslie *et al.*, 2000; Gray, 2002; Thrush and Dayton, 2002; Karenyi *et al.*, 2016; Veiga *et al.*, 2017). Owing to the vast extent and inaccessibility of these habitats, it is believed that only a small portion of species that reside in and on marine sediments have been described (Snelgrove *et al.*, 1997; Van Hoey *et al.*, 2013; Karenyi *et al.*, 2016). Invertebrate communities that occupy unconsolidated habitats play an important role in ecosystem processes that can be directly or indirectly linked to the effect of their foraging efforts (Snelgrove *et al.*, 1997). Some of these ecosystem processes include nutrient cycling, pollution metabolism, dispersal and burial, and secondary production (Ellingsen, 2002). Epibenthic (invertebrates that live on top of the sediment) communities within these habitats often provide biogenic structures and nursery areas that sustain higher taxa including commercially important fish species (Meyer and Smale, 1991; Lindholm *et al.*, 1999; Thrush and Dayton, 2002; Tissot *et al.*, 2004; Belley and Snelgrove, 2016; Baux *et al.*, 2017).

The distributions of biological assemblages in unconsolidated habitats are often considered homogenous owing to the vast extents of the habitats. However, owing to various broad- and smaller-scale physical and biological features, these habitats appear to form a mosaic of habitat diversity that vary in grain size and include outcrops of hard ground features resulting in overall high species heterogeneity (Morrisey *et al.*, 1992; Gray, 2002; Thrush and Dayton, 2002). These mosaics are threatened by the direct and indirect impact of a global increase in anthropogenic activities along coastlines, bays and the open ocean. Some of these impacts include commercial fisheries, dredge dumping, anchor scour, sewage outflows and other pollution, as well as climate change (Warwick and Clarke, 2001; Garcia and Cochrane, 2005; Davis *et al.*, 2016). Commercial trawl fisheries targeting benthic species focus predominantly on unconsolidated habitats, in order to avoid their nets being damaged on hard grounds (Thrush and Dayton, 2002; Atkinson *et al.*, 2011; Ardron *et al.*, 2014). Trawling has the potential to modify and homogenise seafloor habitats, harming fragile species and altering their ecological functioning (Handley *et al.*, 2014). The impact on unconsolidated habitats increases in coastal regions near large ports and cities, where trawling is now limited, however, shipping activities are ever increasing (Garcia and Cochrane, 2005; Davis *et al.*, 2016). An improved understanding of unconsolidated habitats, their species assemblages, distribution and their relationship with the physical environment is needed for wise ocean management. Such information will support the long-term sustainability of fisheries, benthic ecosystems and their sound ecological functioning (Chapin *et al.*, 2000; Bax and Williams, 2001; Ellingsen, 2001; Sink *et al.*, 2012a; Belley and Snelgrove, 2016).

Indicator species that contribute to significant ecosystem processes can provide insights into the interactions between the marine environment and anthropogenic pressures (Shin *et al.*, 2010). Owing to the diverse roles invertebrates play in marine food webs and ecosystem functioning, changes in their abundance or diversity can provide a measure of anthropogenic pressures and indicate trophic cascades (Warwick, 1989; Warwick and Clarke, 1996; Shin *et al.*, 2010; Smale *et al.*, 2011). For example, several studies show that the size of species at a lower trophic level (mostly invertebrates), will often determine life history traits of higher organisms (Warwick, 1989; Warwick and Clarke, 1996; Gray and Elliot, 2009). Invertebrates can act as robust indicators of ecosystem health, providing useful information for long-term monitoring owing to their often sessile characteristics, sensitivity to changes in the environment and strong supportive functional roles (Tissot *et al.*, 2004; Lockhart and Jones, 2008; Van Hoey *et al.*, 2013). Improved knowledge of the patterns of invertebrate assemblage composition, abundance and distribution is fundamental to the establishment of a baseline or reference state against which future biodiversity changes can be measured (Underwood *et al.*, 2000; Bax and Williams, 2001; Cogan *et al.*, 2009; Costello, 2009; Lee *et al.*, 2015). Such information will further inform area-based management strategies, such as the establishment of Marine Protected Areas (MPAs), by identifying vulnerable ecosystems and measurements of ecosystem health.

Patterns and variability of benthic faunal communities and environmental conditions change with both temporal and spatial scale, therefore ecosystems should be described at various scales (Morrisey *et al.*, 1992; Underwood *et al.*, 2000; Ellingsen, 2002). Marine habitat mapping is a key component for marine management, and the scale at which an area is mapped should always be considered relative to the outputs required (Stevens, 2005). In South Africa marine habitat mapping has, to date, relied on abiotic surrogates such as sediment and bathymetry data, which are indicative of biotic distributions. However, the examination of *in situ* biological assemblages and their characteristics is vital to validation of surrogates where possible (Costello, 2009; Guarinello *et al*., 2010; McArthur *et al*., 2010; Van Hoey *et al*., 2013; Buhl-Mortensen *et al.,* 2015a; b). Within South Africa, information regarding unconsolidated habitats is spatially patchy and mostly limited to sediment properties, basic ecology, and the impacts of pollution on benthic communities (Leslie *et al.*, 2000; Sink *et al.*, 2012a).

South Africa's current benthic habitat map was produced during the National Biodiversity Assessment (NBA) process undertaken most recently in 2012 (Sink *et al.*, 2012a). Habitats were delineated using variables such as substratum type, depth, geology, grain size and terrestrial and benthic-pelagic connectivity at a coarse spatial scale, dividing the Exclusive Economic Zone into 136 habitat types (Sink *et al.*, 2012a). Priority actions identified by the 2012 NBA included the further "refinement of the marine and coastal habitat classification and map by systematic marine biodiversity surveys across broad ecosystem groups". This information is also required to address other priority actions such as improving the knowledge base to support the understanding of climate change (via long-term monitoring) and fine scale mapping for the implementation or refinement of MPAs (Sink *et al.*, 2012a).

Underwater imagery is an effective non-destructive method of surveying, describing and monitoring ecologically and economically important benthic habitats and biota, and can provide information on species abundance, diversity and behaviour (Diaz *et al*., 2004; Buhl-Mortensen *et al.*, 2015a; Lee *et al.*, 2015; Schultz *et al.*, 2015; Perkins *et al.*, 2016; Sheehan *et al.*, 2016). This method allows for the assessment of ecological condition in areas of potential concern, while improving the overall understanding of habitat and species distributions within an area. Several methods have been developed to gather underwater imagery, for example remotely operated vehicles (ROV), towed cameras, baited remote underwater videos (BRUV), jump cameras, side scan sonar, manned submersibles and autonomous underwater vehicle (AUV). Some of these tools require advanced skills to manoeuvre while others are simpler in design, each with their own advantages and disadvantages. Underwater imagery tools are often used in conjunction with the collection of physical variables such as sediment, specimens of interest seen during imagery collection, and physicochemical properties of the water. The advantages of non-destructive photographic surveys make the technique one of the most important emerging research tools for establishing baseline benthic habitat and species data in South Africa.

Research on the benthic habitats of Algoa Bay has largely been limited to reef habitats with very little work investigating the unconsolidated habitat types that dominate the Bay Algoa Bay has one small existing MPA and as part of Operation Phakisa's marine protection service and governance lab, a further extension of this MPA has been proposed (Department of Environmental Affairs, 2014b). This area requires extensive, minimally invasive surveys to better understand the functioning and diversity of benthic unconsolidated habitat communities. Therefore, this research aims to:

- Characterise patterns of epibenthic diversity and distribution in unconsolidated benthic habitats in Algoa Bay, thereby establishing baseline data for future long-term monitoring.
- Determine whether the spatial distribution of statistically defined epibenthic communities align with the nationally defined 2012 NBA (Sink *et al* 2012a) benthic habitats in Algoa Bay.

Methods

Station selection

Stations were semi-selectively identified and randomly stratified by two depth zones (inshore 30-50m and offshore 51-100m) to sample evenly across Algoa Bay. The NBA benthic habitat map (Sink *et al.*, 2012a) was used to guide the placement of stations with at least one or more station lying within each of the four defined unconsolidated habitats in Algoa Bay. The NBA-defined habitats assessed included:

- *Agulhas Sandy Inner Shelf*

  This habitat type was the most dominant in Algoa Bay and was mapped using a digitised geological map (Dingle *et al.*, 1987; Lombard *et al.*, 2004).
- *Agulhas Mixed Sediment Inner Shelf*

  This habitat type was also mapped using the digitised geological map, however, it was less dominant in the Bay.
- *Agulhas Hard Inner Shelf*

  This habitat boundary was delineated as a grid block of untrawlable grounds and was included as a mixed habitat for investigation during the present study.
- *Agulhas Island*

  Island habitats were classified as "minor" or "major" islands depending on seal and seabird colony densities. Buffer zones were created to define a zone of island influence (20km buffer around major islands and 10km buffer around minor islands). The classification of islands, buffer zones and associated habitats were identified as an aspect in need of revision during the next NBA.

Field sampling

Two different methods of image sampling were used owing to the poor visibility of images collected using the drop camera from three of the original twelve stations selected (Fig. 2.1 'removed stations' indicated as grey circles). Previously recorded ROV footage from four stations (ROV1_3 – ROV1_6) nearby these stations were used to replace the poor-quality imagery. The total number of thirteen stations were used during image analysis.

Drop camera

Images of the seabed were collected at twelve stations (Fig. 2.1) using a drop camera system consisting of a GoPro Hero 3+ camera and dive lights mounted on a circular frame (Fig. 2.2). The camera was deployed from research vessels (RV *Honckenii* or *uKwabelana*) and dropped directly onto the seabed. Using the onboard Global Positioning System, each station was sampled in a grid formation to collect a series of images from between 40-60 drops, each 50m apart (Fig. 2.3, Table 2.1). The camera was set to use interval photography, automatically capturing an image every five seconds during each drop. The camera remained on the substratum for an estimated 40-60 seconds per drop to allow disturbed sediment to settle, after which it was raised 10m and moved to the next point on the sampling grid. The coordinate position for each drop was recorded. The camera was checked once per station to confirm that the camera and lights were working properly. The seabed area captured for analysis per drop (image) was 0.28m^2, defined by the area of the circular drop-frame that contacted the seafloor. Owing to the method of interval photography used, numerous images of the same portion of seafloor were collected per drop. Only one of such duplicate images was selected for further processing. Images from only nine of the twelve stations sampled were suitable for processing

Remotely Operated Vehicle (ROV)

Video footage transects from four supplementary stations were collected using an ROV (Falcon Seaeye: 12177) fitted with a SubCControl 1 Cam (12.3-megapixel HD camera). A total of thirty minutes of video footage per station were analysed during which between 50-60 still frame images were captured using VLC media player. The camcorder was fitted with two parallel laser pointers set to project onto the seabed to provide a scale reference of 6.42cm. A predetermined scaling grid as per Wakefield and Genin (1987) was used to determine the area of the still frame images according to the laser positions. The position of the lasers allows for the calculation of the angle of the camera which in turn determines the area of the still frame image. Still frame images were captured when the position of the lasers aligned with the predetermined grid to maintain an estimated area of 0.28m^2 per image. This grid was then superimposed onto all still frame images to allow for the standardisation of area processed in each image.

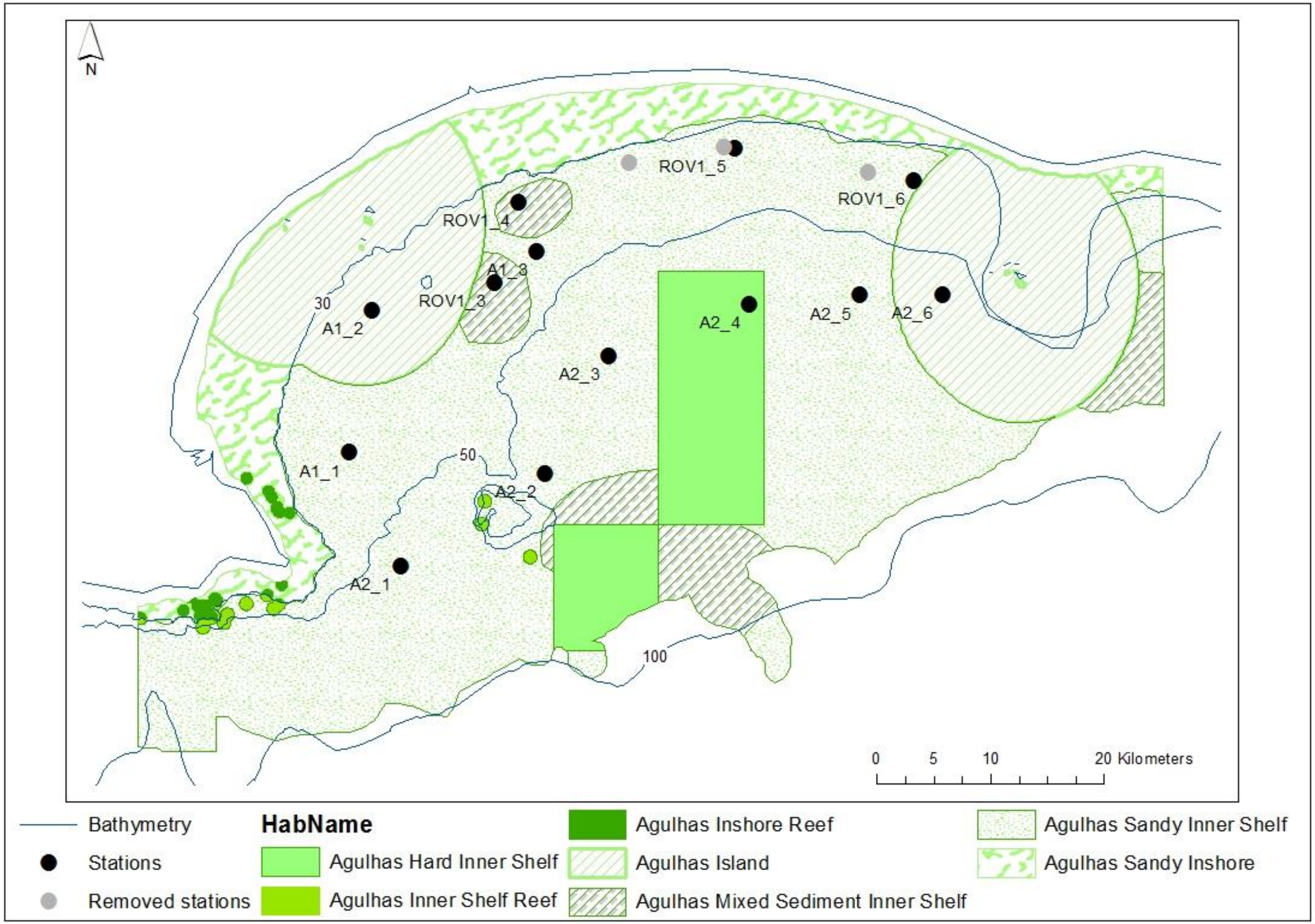

Figure 2.1: Distribution of 13 benthic sampling stations in Algoa Bay, superimposed on the National Biodiversity Assessment (2012) benthic habitat map. Black dots are stations used in the final analysis, and grey dots are stations excluded from the final analysis.

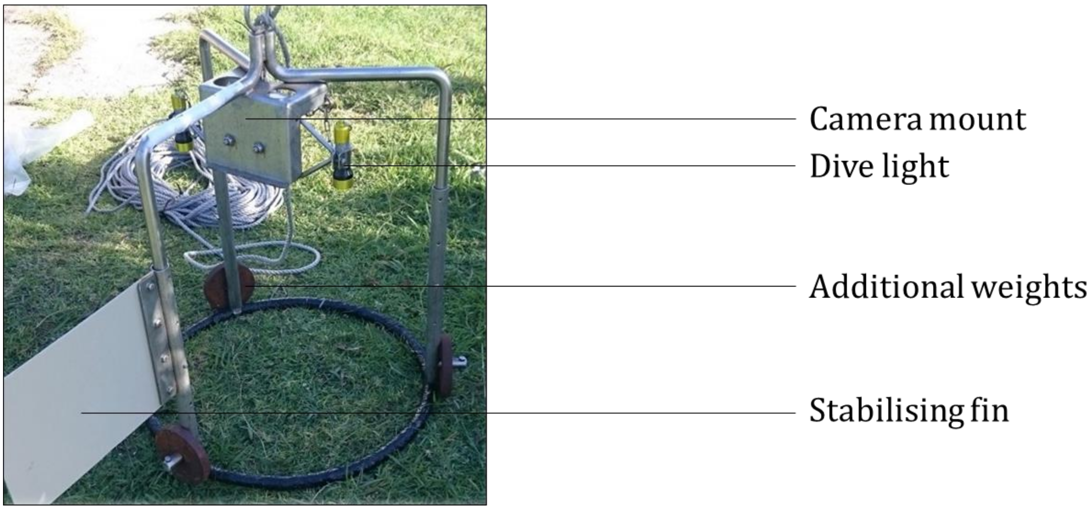

Figure 2.2: The drop camera frame with dive lights. The GoPro housing is placed in the rectangular holder (camera mount) between the two lights.

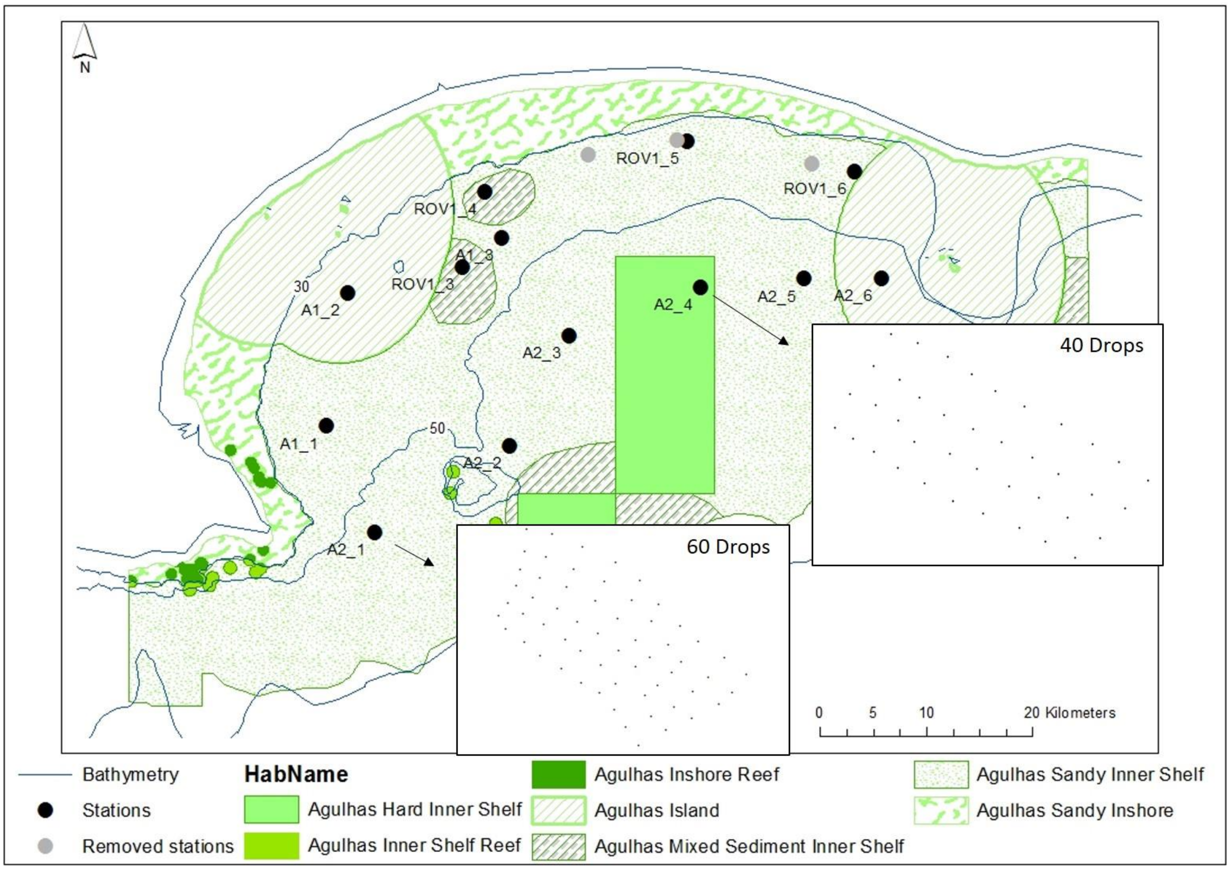

Figure 2.3: An example of the sampling grid used when deploying the drop camera. Seven stations collected 40 drop images (4 x 10), while two stations (A1_2 & A2_1) collected 60 drop images (6 x 10).

Table 2.1: Relevant information for each station sampled including latitude, longitude, sampling instrument, habitat type (as per National Biodiversity Assessment, Sink et al. 2012a), depth (in meters), depth category and number of images processed per station.

Station number	Latitude (decimal degrees)	Longitude (decimal degrees)	Sampling Instrument	Habitat type (NBA 2012)	Depth (m)	Depth category	# of images processed
A1_1	-33.9529	25.7556	Dropframe	Agulhas Sandy Inner Shelf	41.3	30-50m	40
A1_2	-33.8591	25.7735	Dropframe	Agulhas Island	41.4	30-50m	60
A1_3	-33.8201	25.9037	Dropframe	Agulhas Sandy Inner Shelf	46.6	30-50m	40
A2_1	-34.0279	25.7959	Dropframe	Agulhas Sandy Inner Shelf	60.8	51-100m	60
A2_2	-33.9671	25.9103	Dropframe	Agulhas Sandy Inner Shelf	52.8	51-100m	40
A2_3	-33.8893	25.9605	Dropframe	Agulhas Sandy Inner Shelf	66.3	51-100m	40
A2_4	-33.8552	26.0722	Dropframe	Agulhas Hard Inner Shelf	64.8	51-100m	40
A2_5	-33.8492	26.1594	Dropframe	Agulhas Sandy Inner Shelf	68.4	51-100m	40
A2_6	-33.8486	26.2248	Dropframe	Agulhas Island	64.7	51-100m	40
ROV_1_3	-33.8406	25.8704	ROV	Agulhas Mixed Sediment Inner Shelf	44.8	30-50m	52
ROV_1_4	-33.7875	25.8893	ROV	Agulhas Mixed Sediment Inner Shelf	34.9	30-50m	60
ROV_1_5	-33.7517	26.0607	ROV	Agulhas Sandy Inner Shelf	36	30-50m	50
ROV_1_6	-33.7731	26.2024	ROV	Agulhas Sandy Inner Shelf	34.5	30-50m	50

Image analysis

Species Identification and counts

All epibenthic species visible in images were identified (Fig. 2.4) to the lowest possible taxonomic level based on identification guides with assistance from taxonomic experts (Parker-Nance, pers. com.; Branch *et al.,* 2016; Atkinson and Sink, 2018). Absolute counts of all organisms, where possible (when clearly visible), were made in each image. Only epibenthic species appearing within or directly underneath the drop camera circular frame or ROV still frame image scaling grid, were counted.

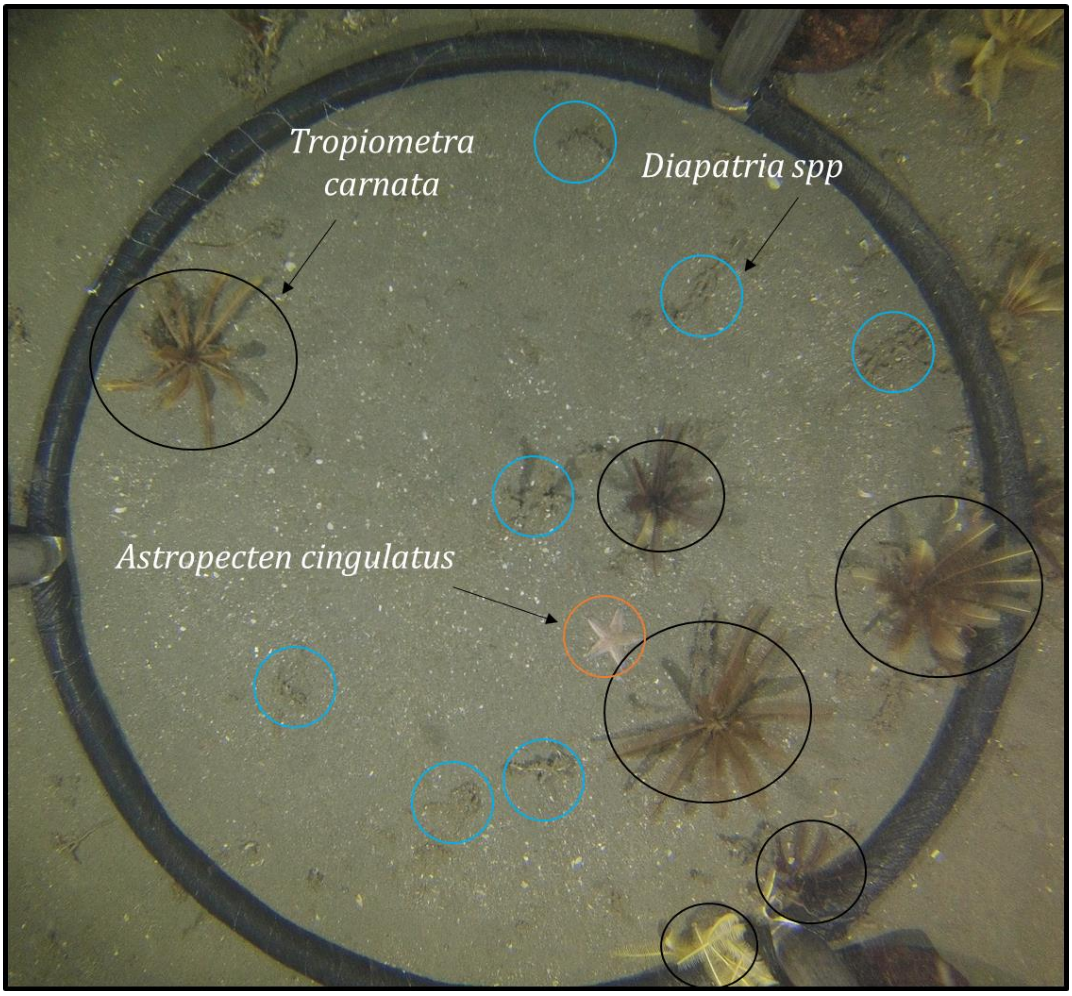

Figure 2.4: Processed benthic image (station 1_3) with fauna visible demarcated to illustrate the method employed.

Sand Shell Rock

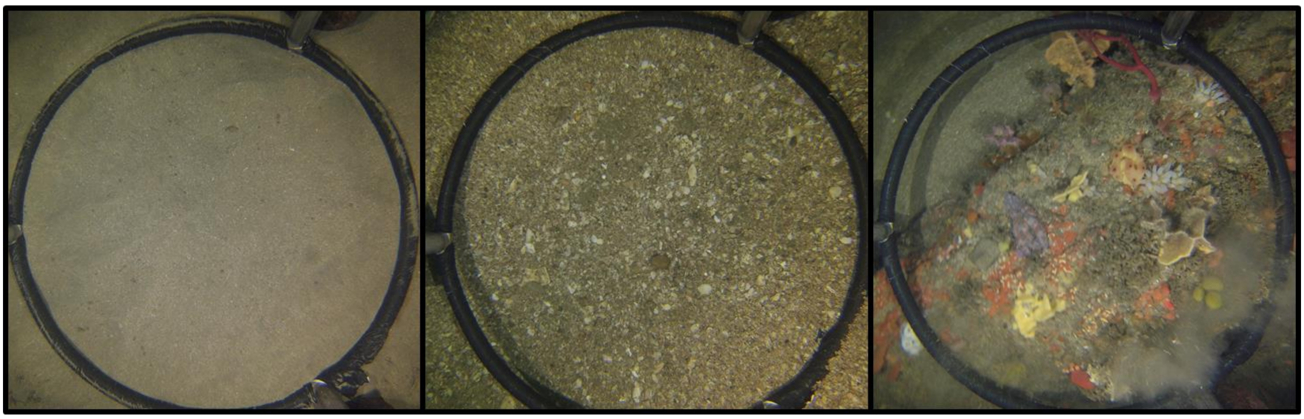

Figure 2.5: An example of the three classes of substratum used during the visual assessment

Table 2.2: List of species observed in drop camera frame and ROV imagery

Diapatria spp	*Halopteris tuba*	*Axinella spp3*	*Turritella declivis*	*Clavelina lepadiformis*
Astropecten cingulatus	*Crella spp2*	*Papilla sponge*	yellow worm/ Sea cucumber	*Eunicella papillosa*
Actinoptilum molle	*Trididenum cerebriforma*	Bryozoan 3	Unsure - Hydroid	Bryozoan 6
Hydroid 1	*Psammoclema spp*	*Homophyton verrucosa*	Anemone	*Aplidium spp*
Axinella spp1	*Axinella spp2*	*Cheilostomatida*	Robust mustard hydroids trees	Mollusca
Crella spp1	*Arcania spp*	*Balanophyllia bonaespei*	Algae	Helmet shell
Marthasterias africana	*Pycnoclavella filamentosa*	*Bryozoan grey*	*Seapen - Virgularia spp*	*Ovalipes tri*
Proteleia sollasi	*Astrocladus euryale*	*Bursitella spp*	Sand colonial ascidian	Cuttle fish
Bryozoan *3*	*Pink soft coral – Klyxum*	*Waltherarndtia caliculatum*	Red_white gorgonian	Squid
Parazoanthus spp	*Trichogorgia capensis*	*Clathria oxitoxa*	White coral_Algae	Gurnid
Pseudodistoma africanum	White hydroid feathers	*Tedania stylonychaeta*	*Isodictya elastica*	Club ascidian
Eudostoma spp	*Hymenaphia spp*	Unknown sponge 3 -pink	*Corallina officinalis*	Shy shark
Polyclinum isipingese	*Leptogorgonia gilchristi*	*Clathrina blanca*	*Corynactis annulata*	Unknown ascidian2
Tropiometra carnata	*Arcania spp2*	*Unknown polychaete*	*Phidoloporidae*	Finger sponge
Porifera 1	*Aplidium spp1*	Bryozoan 4 /algae	*Leptogorgia palma*	Hydroid spp3
Dideminium spp1	Hydroid spp2	Hermit crab	*Allopora nobilis*	Yellow ascidian
Porifera 3	*Conus spp*	*Gymnogongrus polycladus*	*Aplidium flavolineatum*	*Malacacanthus capensis*
Atriolum marionensis	Cucumber	Polychaete tube	Oyster	Gorgonian like hydroid
Pseudotrachya spp	*Euphrosine spp*	Sole (fish)	*Isodictya spp4*	Encrusting ascidian
Virgularia schultzei	*Aplidium spp2*	Brittle stars	Unknown bryozoan 5	*Caliaster baccatus*
Unknown ascidian - *Rhopalaea sp*	*Eunicella spp*	*Henricia ornata*		

Verification of sampling intensity

Species accumulation curves were constructed per station and per habitat type to determine whether sufficient samples (drop/ video frame-grab images) were processed. Sample level data were combined across stations within a given 2012 NBA habitat type to generate species accumulation curves per habitat type. Observed species richness (Sobs), Chao1, Jacknife1 and Bootstrap species abundance estimators were used for comparison (permutated values compared to extrapolated values). Sobs estimates the number of species by permuting observed species accumulation. Chao1 is frequently used for abundance data based on the notion that rare species play a greater role when informing the number of undetected species (Chao *et al.*, 2008, 2009). Chao1 uses the number of singletons (only one individual recorded of that species) and the number of doubletons (two or more individuals per species) to estimate species richness. Jacknife1 uses only the number of singletons to estimate species richness. The Bootstrap method was used to obtain approximate estimates of variances and confidence intervals for species richness (Chao *et al.*, 2009; Gotelli and Colwell, 2011; Gotelli and Chao, 2013). Points of deceleration were visually identified from species accumulation graphs as the approximate point at which a 20% increase in the number of samples results in a less than 5% increase in the number of new species observed (Hortal and Lobo, 2005).

Visual substratum assessment

Images were imported into Coral Point Count with Excel extension (Kohler and Gill, 2006) to visually assess the percentage cover of rock, shell/shale and sand in each image (612 images). The same area processed for species identification and counts was overlaid by a matrix of 100 equally-distributed points and the substratum type lying beneath each point was visually identified (saved as a .cpc file). Data from each individual image were then converted into Excel spreadsheets to produce percentage cover per image for further analyses. Average substratum cover was calculated per station and given a category according to the following criteria (Fig. 2.5):

- Sand – 100%
- Mixed – rock, sand and shell all present in varying percentages
- Sand-Shell – Sand>50% & shell<50%

Data analysis

Benthic species abundance data were analysed using a variety of univariate and multivariate analyses to assess patterns, differences and clustering among sampled stations and habitats. All univariate and multivariate analyses were performed using PRIMER (Plymouth Routines in Multivariate Ecological Research) version 6.1.18 with PERMANOVA+ version 1.0.8 extension (Anderson *et al.*, 2008).

Patterns of species richness

Species richness (S), number of individuals (N), Margalef's diversity index (D), Pielou's species evenness (J) and Shannon-Wiener diversity (H') were calculated using the DIVERSE function in PRIMER. Species were grouped according to taxonomic class or order and the percentage abundance per station was calculated. Species diversity indices and species community data were mapped to provide a spatial assessment of patterns of diversity in the Bay.

Multivariate analysis

Abundance data were forth-root transformed to reduce the masking effect of highly abundant species on less abundant species in the sample ordination (Field *et al.*, 1982). The Bray-Curtis similarity measure was used to generate a resemblance matrix of epibenthic abundance to determine the percentage similarity/dissimilarity among stations (Bray and Curtis, 1957). A dummy variable was included (zero adjusted Bray-Curtis) to account for the high number of zero abundance values recorded at most stations (Clarke *et al.*, 2006). An unconstrained multi-dimensional scaling (MDS) plot provided visual representation of the epibenthic community composition data. A two-way nested Permutational Analysis of Variance (PERMANOVA), based on the zero adjusted Bray-Curtis similarity matrix, was used to test for significant differences among NBA-defined habitat types (factor A) and stations (factor B) nested within habitat types.

To further examine epibenthic community patterns among stations, a cluster dendrogram, with Similarity Profile (SIMPROF), was performed on abundance data, summed per station. The SIMPROF analysis is a permutational test that indicates statistically significant evidence of genuine groupings which have non-random structure and warrant further detailed investigation (Clarke and Gorley, 2006; Clarke *et al.*, 2008). Additionally, a PERMDISP routine (distance to centroids using permutation of residuals, Anderson *et al.*, 2008) was used to test for significant differences in the dispersion and location of the groups identified by the SIMPROF analysis. To determine which species contributed most to the different communities identified by the SIMPROF analysis, a SIMPER analysis was conducted. The SIMPER analysis identifies characteristic and distinguishing species per group and provides their percentage contribution to the group.

Results

Epibenthic communities were characterised by two groups, but with little spatial pattern to the species richness. No significant differences were found between NBA habitats, however differences were observed between stations occurring within the same NBA habitat. Community groups were distinguished by different taxa with the polychaete *Diopatria spp.* identifying group A and the complete lack of the polychaete in group B. A visual assessment of the substratum further supported the split in community groups with stations in group A (except station ROV1_5) supporting 100% sand, while group B supporting a mixed substratum of rock, shells and sand.

Verification of sampling intensity

Species accumulation curves were plotted for each NBA-defined habitat type and per station to estimate total species richness per habitat and to assess if sufficient samples were processed. Sufficient sampling conducted is indicated by observational (Sobs) and extrapolated (Chao1, Jacknife1 and Bootstrap) curves reaching an asymptote (Chao *et al.*, 2009). The *Agulhas Island* followed by the *Agulhas Mixed Sediment Inner Shelf* habitat types were estimated to have the highest species richness among the four NBA defined habitat types assessed (Fig. 2.6). *Agulhas Island* and *Agulhas Mixed Sediment Inner Shelf* habitat types did not reach an asymptote indicating insufficient sampling effort (Fig. 2.6). It should be noted that only two stations were analysed for these habitat types. *Agulhas Sandy Inner Shelf* and *Agulhas Hard Inner Shelf* habitat types curves reached an asymptote with *Agulhas Hard Inner Shelf* also displaying the closest alignment between Sobs and the other species abundance estimators.

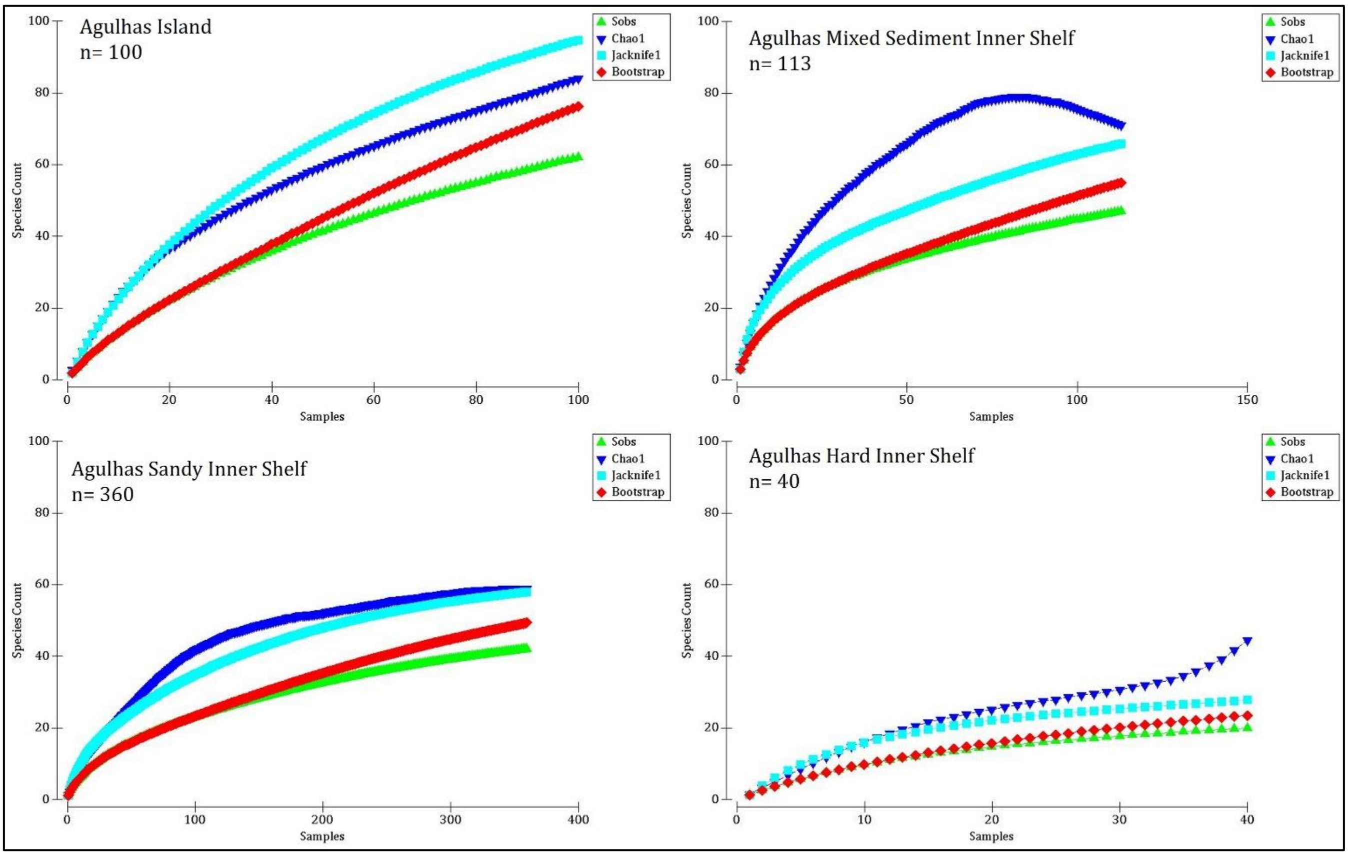

Figure 2.6: Species accumulation curves estimation of species richness per NBA-defined habitat type. The x-axis represents the number of camera drops (images) recorded and the y-axis indicates the number of new species recorded within that habitat type.

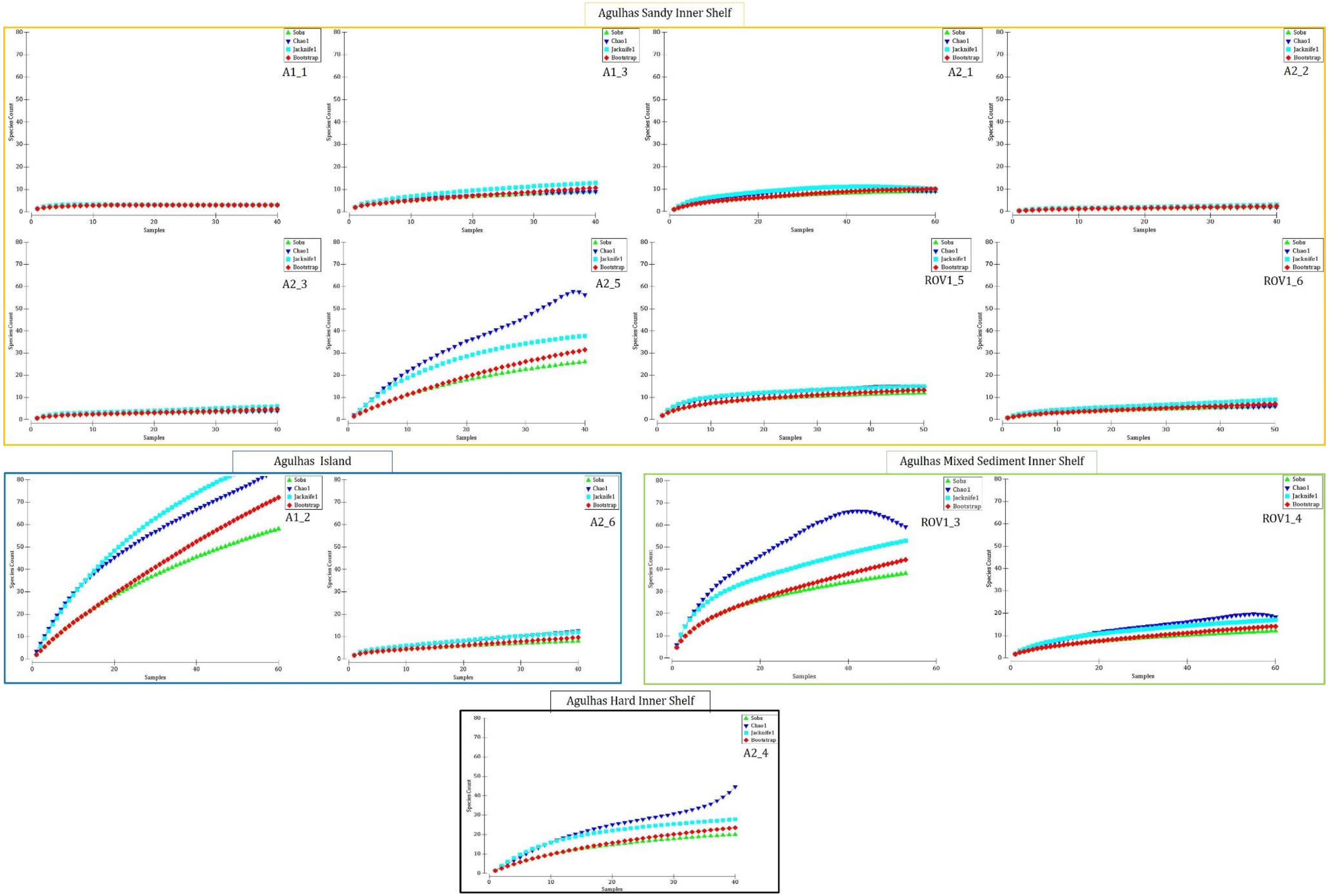

Figure 2.7: Species accumulation curves per station sampled. The x-axis represents the number of camera drops (images) and the y-axis indicates the number of new species recorded at that station. All Y- axes were standardised for comparison.

All species accumulation curves per station except two reached asymptotes and saw convergence between the Sobs and estimated curves (Fig. 2.7). The two exceptions were stations A1_2 and ROV1_3 and both had higher estimated species richness. Station A1_2 did not reach an asymptote by any species estimator tested (Fig. 2.7).

Visual Assessment

Of the 13 stations images processed, eight stations were identified as 100% sand cover with four stations displaying a mixed (varying percentages of rock, sand and shell at each station) substratum (A1_2, A2_4, A2_5 and ROV1_3), and one station, ROV1_5 was identified as sand-shell (95% sand, 5% shell).

Patterns of species richness

A total of 106 epibenthic species were recorded from images collected at 13 stations covering a sample area of 171.7m^2 in Algoa Bay. Relatively high numbers of species (S) (20-58) were recorded at four of the 13 stations, with each of these four stations occurring in different NBA-defined habitat types, distributed evenly across the Bay (Table 2.3, Fig. 2.8A). Higher numbers (>300) of individuals (N) were generally recorded towards the western and inshore sector of the Bay (Table 2.3, Fig. 2.8B) with station ROV1_3 recording the highest number (1354 individuals). The high number of individuals recorded at Station A2_6 (568) in the eastern, offshore region of the Bay was the only exception. Stations where more than 200 individuals were recorded (Table 2.3) were all located within or near to the *Agulhas Island* and *Agulhas Mixed Sediment Inner Shelf* habitat types. Stations A1_2 & A2_6 occur within the *Agulhas Mixed Sediment Inner Shelf* and, station A1_1 was located 6km away from this habitat type. Five of the six offshore stations and only two of the seven inshore stations recorded species evenness (J') greater than 0.6 (Fig. 2.8C). Shannon-Weiner diversity index values were greater than one for seven of the 13 stations sampled. These seven stations were evenly distributed across the Bay (Fig. 2.8D) with stations A1_2 (H': 3.077), A2_4 (H': 2.472) and A 2_5 (H': 2.809) recording the highest Shannon–Weiner diversity index values, again each occurring in a different NBA-defined habitat type.

Table 2.3: Epibenthic species diversity indices for all stations. [S – number of species, N- number of individuals, D- Margalef's diversity index, J'- Pielou's species evenness and H'- Shannon-Weiner diversity index].

Sample	S	N	D	J'	H'(log^e)
A1_1	3	203	0,3764	0,3957	0,4347
A1_2	**58**	316	9,903	0,7578	3,077
A1_3	9	384	1,344	0,5058	1,111
A2_1	9	163	1,571	0,5901	1,297
A2_2	2	22	0,3235	0,684	0,4741
A2_3	4	54	0,7521	0,6138	0,8508
A2_4	**20**	138	3,856	0,8252	2,472
A2_5	**26**	101	5,417	0,8622	2,809
A2_6	8	568	1,104	0,4017	0,8353
ROV1_3	**38**	**1354**	5,131	0,52	1,892
ROV1_4	12	266	1,97	0,318	0,7902
ROV1_5	12	190	2,096	0,6289	1,563
ROV1_6	6	66	1,193	0,4904	0,8787

Almost half of the communities sampled in the Bay were dominated by polychaetes (29% of total individuals observed – seven of 13 stations were made up of greater than 50% polychaetes per station), followed by Cnidaria (Anthozoa & Hydrozoa 30% of total individuals observed – three of 13 stations were made up of greater than 50%) and Ascidians (17% of total individuals observed). Crinoids were dominant at only one station, A1_3. Four stations, each occurring in a different NBA-defined habitat types, (A1_2, ROV1_3, A2_4 and A2_5) included few to no polychaetes, however had high species richness (Table 2.2, in bold). These stations instead included a high variability of taxa such as Anthozoans, Hydroids, Ascidians and Bryozoans and were not distributed with any obvious spatial patterns (Fig. 2.9 & 2.10). Phyla grouped into 'Other' included Arthropoda, Vertebrata (fish), Ophiuroidea (Echinodermata), Holothuroidea (Echinodermata) and Mollusca.

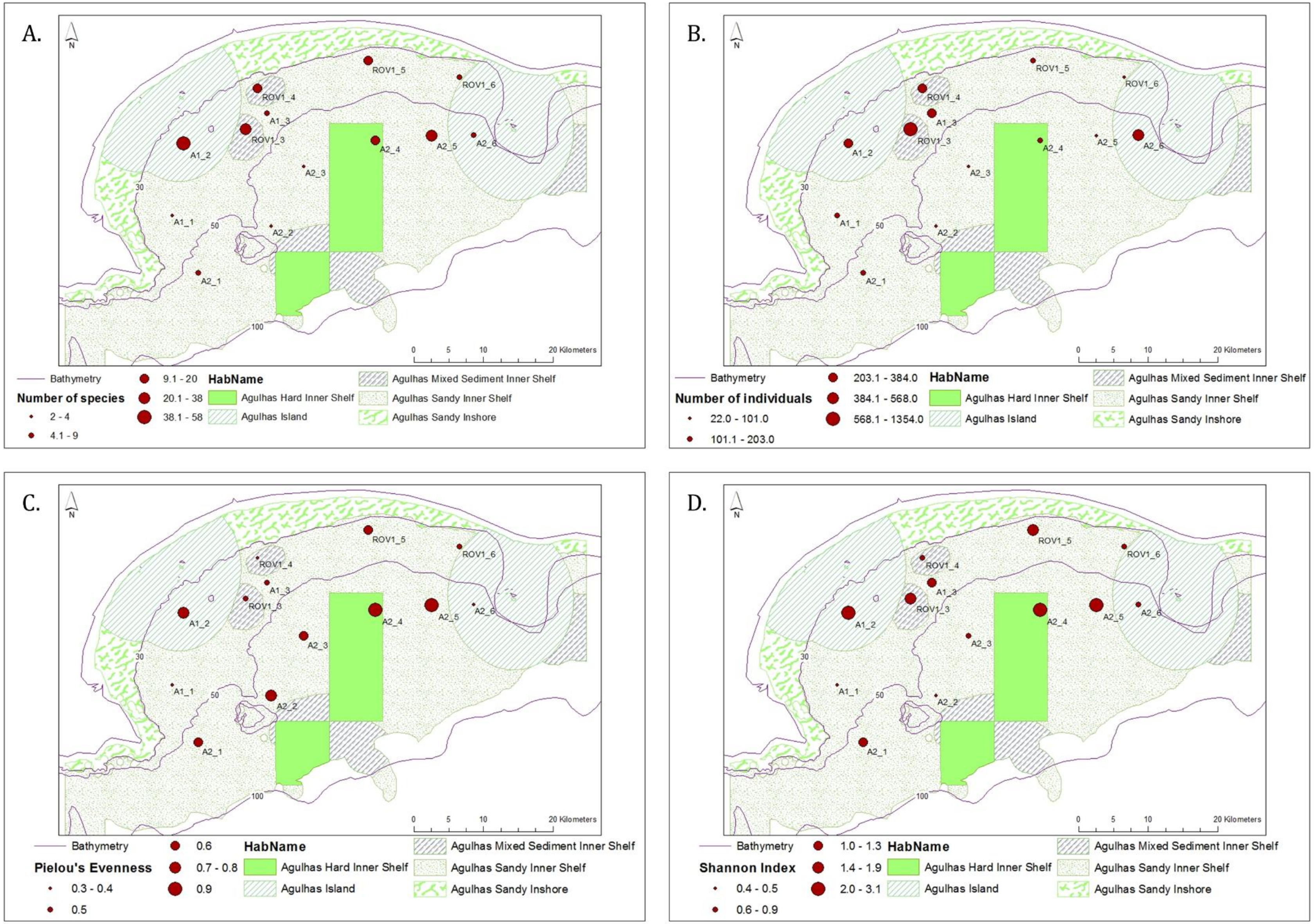

Figure 2.8: Species diversity indices represented spatially per station in Algoa Bay superimposed onto the 2012 NBA (Sink et al., 2012a) benthic habitat map (See Table 2.2 for data values). A. Total number of species (S), B. Number of individuals recorded(N), C. Species evenness (J') and D. Shannon-Weiner diversity index (H').

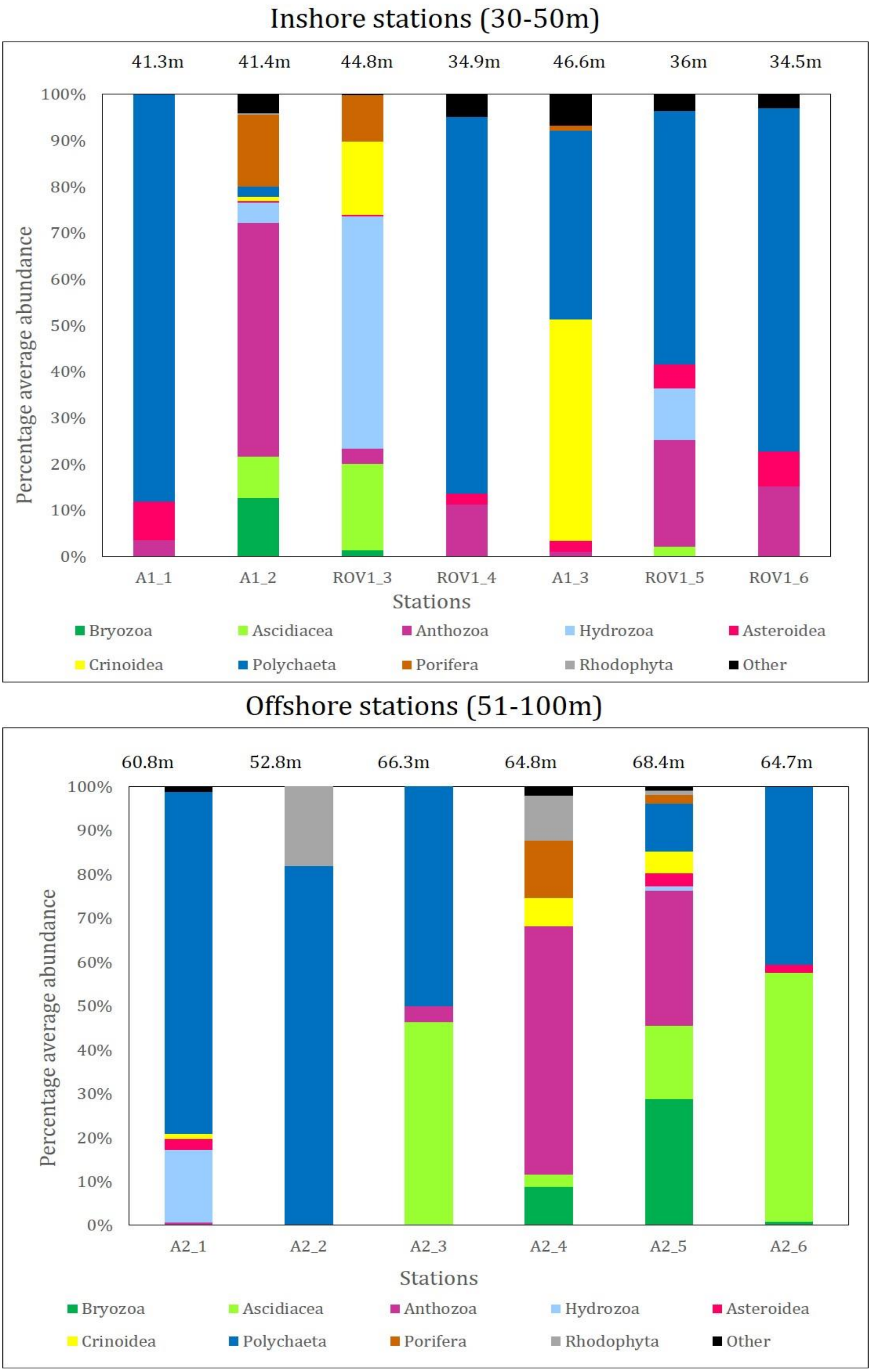

Figure 2.9: Percentage average abundance of dominant phyla in species assemblages per station. Results group stations per inshore (top) and offshore (bottom) and from West to East (left to right) across Algoa Bay.

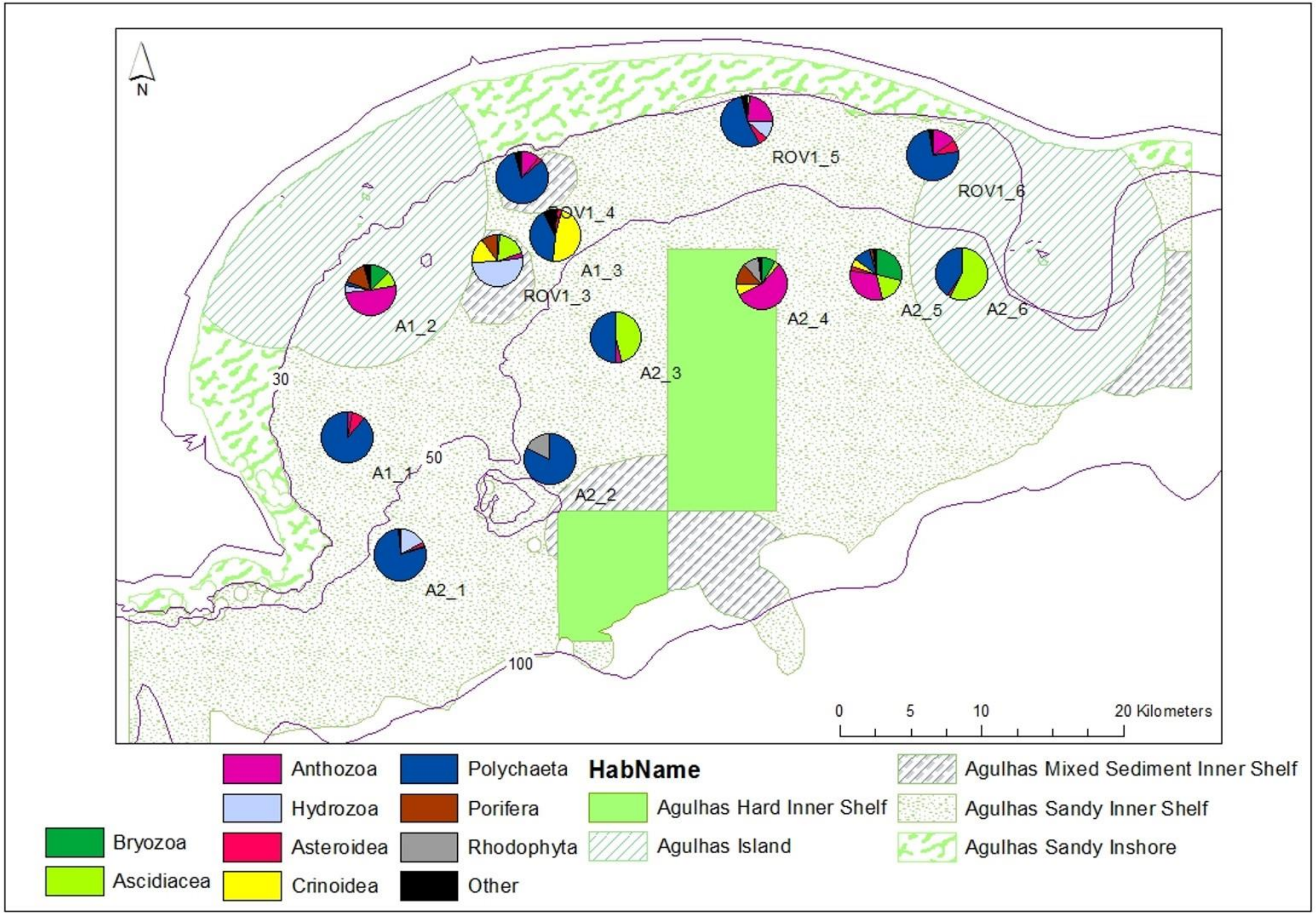

Figure 2.10: Percentage average abundance of dominant epibenthic taxa in species assemblages at stations sampled in Algoa Bay super imposed onto the 2012 NBA benthic habitat map (Sink et al., 2012a).

Multivariate analysis

The multi-dimensional scaling (MDS) plot (Fig. 2.11) shows a separation of the epibenthic species identified at each station into two groups, however it is not immediately apparent what is driving this separation, because both groups have samples from the same station (i.e. station A1_3 has sample points in both groups).

The nested PERMANOVA analysis indicated no significant difference among habitat types (p= 0.3362, pseudo $F = 1.176$, *d.f.* = 3; Table 2.3). There was a significant effect for the factor station nested within habitats, indicating high species assemblage variability among stations within the NBA-defined habitat types (p= 0.0001, pseudo F= 30,489, d.f. = 9; Table 2.3).

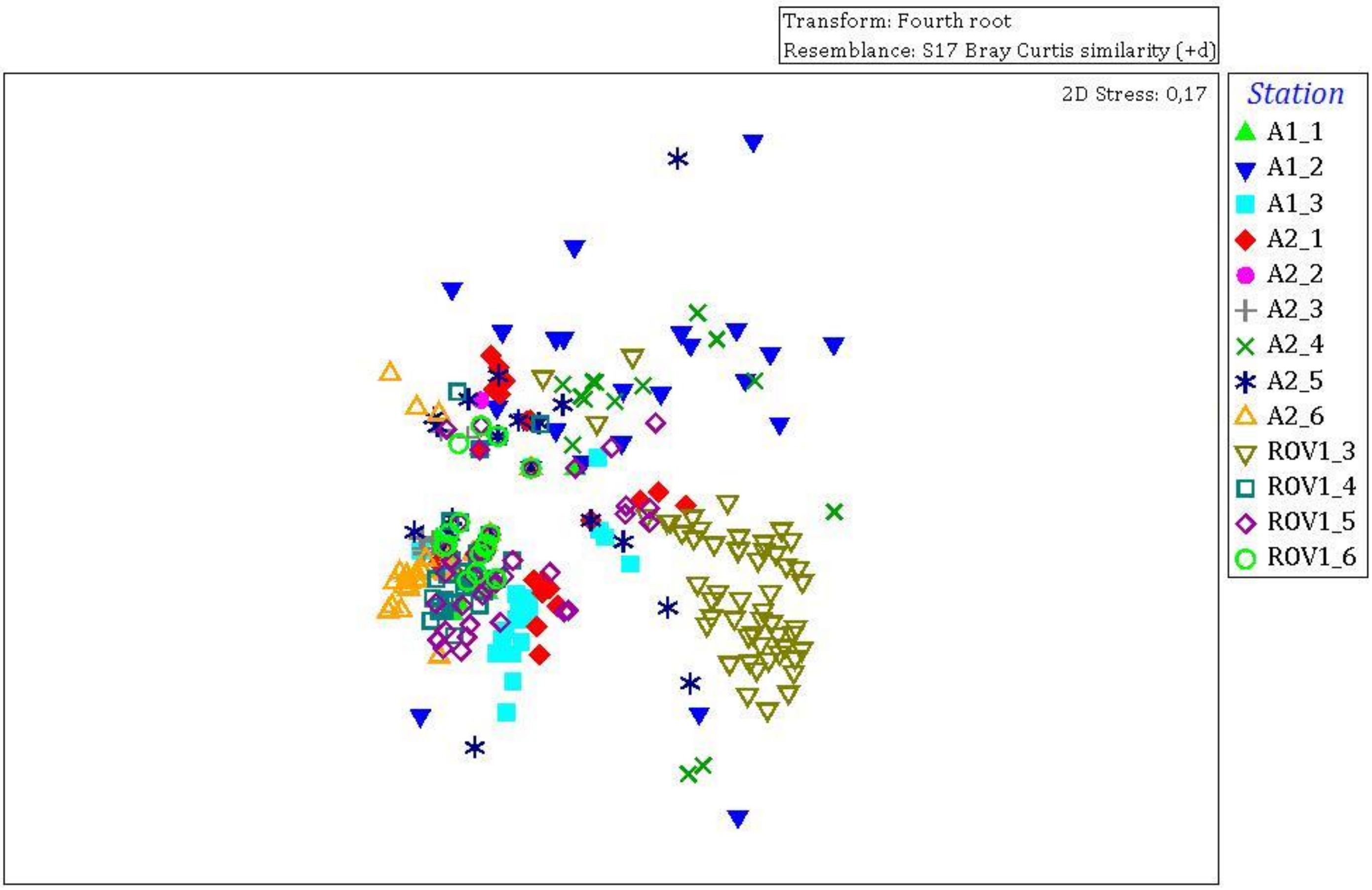

Figure 2.11: Multi-dimensional scaling plot showing the distribution of community composition for each sample per station after 4th root transformation and zero-adjusted Bray Curtis resemblance.

Table 2.4: Test statistics for main effects result from the PERMANOVA analysis of epibenthic species abundance among habitats and stations nested within habitats. The number of unique permutations possible exceeded 9000 in all cases. ***Bold values indicate*** *significance at p ≤ 0.05. (Df: degrees of freedom, SS: Sum of Squares, MS: Mean Square)*

Source of variation	Df	SS	MS	Pseudo-F	p-value	Unique perms
Habitat	3	1,0094E5	33648	1,176	0,3362	9940
Station (Habitat)	9	2,5357E5	28175	30,489	**0,0001**	9852
Residual	600	5,5446E5	924,1			

Epibenthic communities

The cluster dendrogram with similarity profile (SIMPROF) of epibenthic species abundance data from all stations shows two separate groupings at a similarity level of 33% (named group A & B respectively, Fig. 2.12). Seven of the nine stations within group A are classified as *Agulhas Sandy Inner Shelf* according to the NBA defined benthic habitat types, with the remaining two stations classified as *Agulhas Island* and *Agulhas Mixed Sediment Inner Shelf.* Group B includes only four stations, however, each station in this group is classified as a different NBA defined habitat type assessed. The visual assessment of the substratum averaged per station indicates that group B consists only of mixed substratum (rock, sand and shell), while group A consists of all sand stations and one station (ROV1_5) that had 95% sand and 5% shell. Results from the PERMDISP testing the dispersion and location effect of groups A (n= 420) and B (n= 193) were significantly different (F= 134.95, P= 0.0001). This can be visually observed in Figure 2.9 where two distinct clusters of samples are evident. The tight cluster of samples (lower left corner of plot) are mostly samples from group A while the more dispersed cluster of samples (upper and right hand of plot) are mostly samples from group B (as defined by SIMPROF). This reflects a higher similarity (lower diversity) in species present in group A (hence tightly clustered) while group B samples are more dispersed indicating a higher diversity in species composition associated with the mixed substratum.

Results from the SIMPER analysis (Fig. 2.13) indicate that group A was dominated by the polychaete *Diopatra* spp (92.67%) and had higher average abundance of the polychaete worm, *Diapatra spp*, two sea pen species (*Actinoptilum molle* and *Vigularia spp*), a starfish (*Astropecten cingulatus*) and colonial sand ascidians (*Molgula scutata*). Group B was dominated by an unknown hydroid (39.58%), a feather star, *Tropiometra carnata* (14.39%) and an ascidian, *Atriolum* spp (11.44%). When comparing species composition between the two groups, there is a clear separation between groups caused by the dominant *Diapatra* spp present in group A stations and an almost complete lack thereof occurring in group B stations. The average similarity within group B (5.07%) was far lower than that of group A (25.02%, Fig. 2.13).

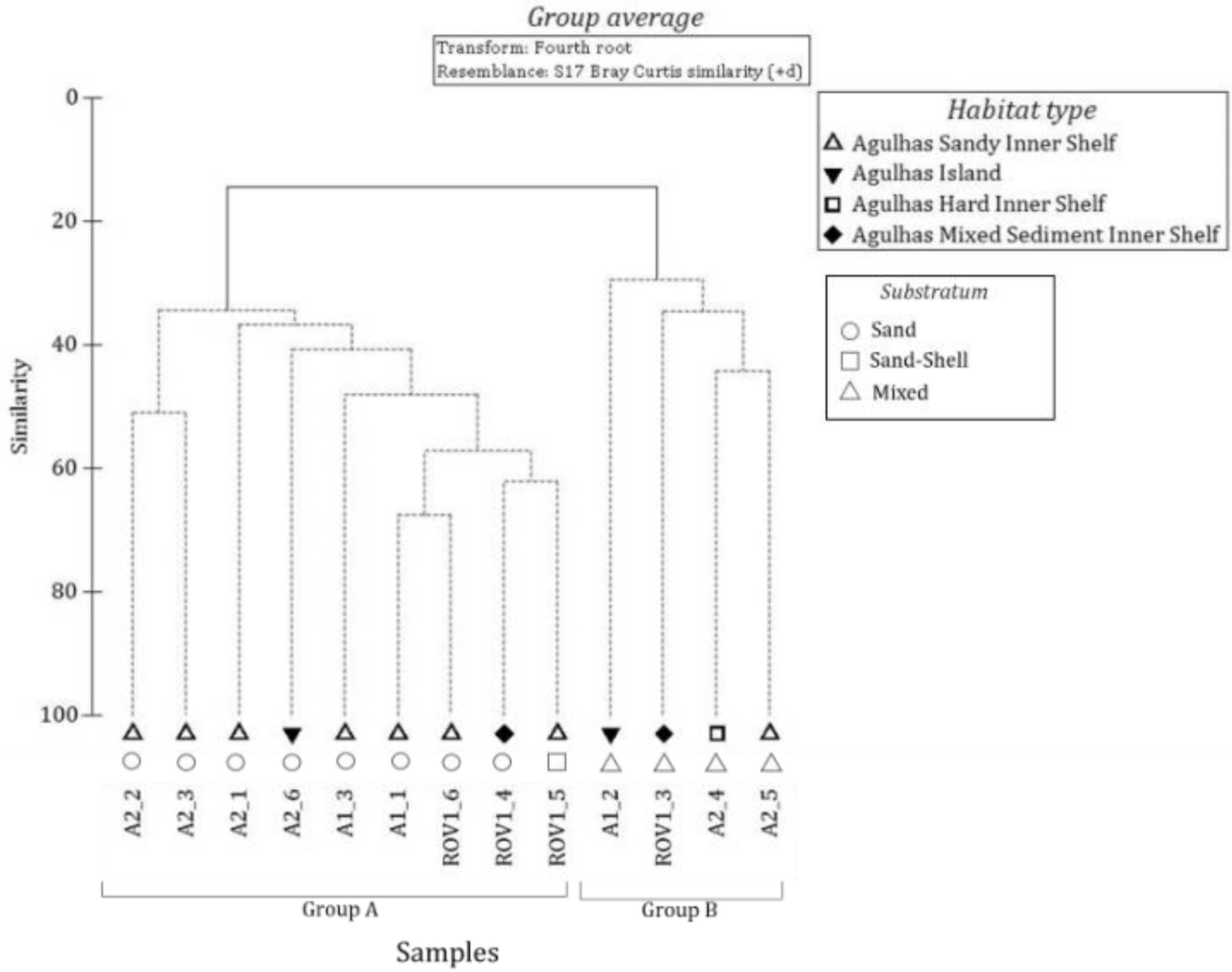

Figure 2.12: CLUSTER dendrogram with similarity profile indicating two groupings of species assemblages among stations, with each station's NBA defined habitat types and substratum., Groups were labelled as group A (left cluster) and B (right cluster) after 4th root transformation and zero-adjusted Bray Curtis resemblance. A solid black line on the dendrogram indicates significant similarities calculated by a SIMPROF analysis.

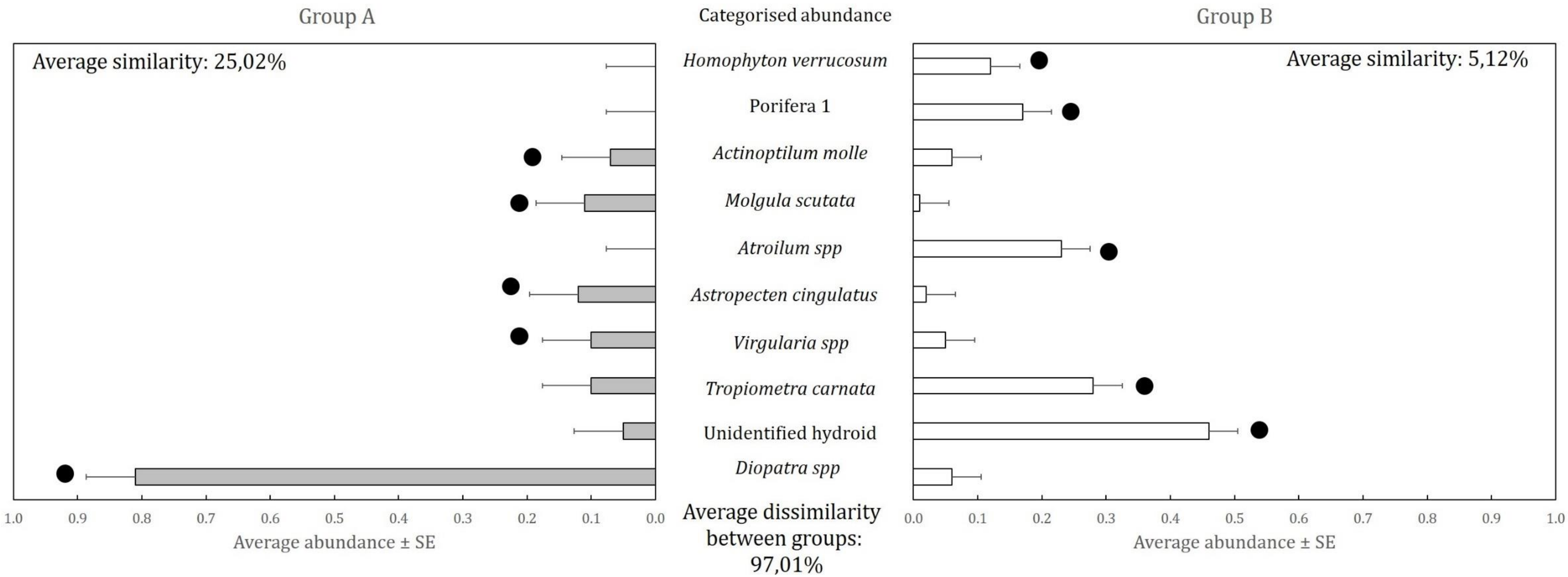

Figure 2.13: SIMPER results showing the top ten species contributing to group A and B identified in the CLUSTER dendrogram with SIMPROF. The black dots indicate which community/group had a higher average abundance of that species.

Discussion

Epibenthic characteristics and distribution in Algoa Bay

This study presents the first quantitative, visual survey of epibenthic communities in unconsolidated marine habitats in Algoa Bay. The species assemblages can be considered to be spatially patchy, divided into two distinct groups or community types. One community type (group A, nine out of 13 stations) is characterised by low species richness (<20 species/station) and is dominated by the polychaete *Diopatra* spp. In contrast, a second community was detected (group B, four out of 13 stations) and characterised by high species richness (>20 species/stations) and diversity indices (D>3) and featured few to no *Diopatra* spp. The four group B stations can be considered 'localised habitats', similar to the small patchy epibenthic communities, observed by Lange & Griffiths (2014) for unconsolidated marine sediment habitats of South Africa's West coast, where each were found to have their own distinctive biota. Similar local and international studies on epibenthic communities have identified depth as a major driver of community composition and distribution (Gray, 2001; Kruger *et al.*, 2005; Whittington *et al.*, 2006; McClain *et al.*, 2010; Brown and Thatje, 2014; Lange and Griffiths, 2014; Piacenza *et al.*, 2015; Serrano *et al.*, 2017a). Depth is often used as a surrogate for the spatial distribution of epibenthic communities when conducting marine habitat classifications (Post *et al.*, 2006; Williams *et al.*, 2009; McArthur *et al.*, 2010; Howell *et al.*, 2010). This, however, was not observed in the current study as the epibenthic groups A and B identified were distributed across both inshore and offshore depth zones. This finding indicates that the depth gradient at which samples were collected in this study was not broad enough to compare differences when addressing the biological communities alone. Studies conducted by Franken (2015) and Baux *et al*. (2017) in unconsolidated habitats, at similar scales to the current study, also established that depth did not play a major role in structuring the benthic community assemblages observed. These studies also explored the potential influence of other environmental factors, such as sediment types and oceanographic variables, in relation to benthic communities. Similarly, the influence of environmental drivers on epibenthic communities in Algoa Bay are explored in Chapter Three of this study.

Offshore stations had a higher species evenness (Table 2.3) when compared to inshore stations. Although several species within an ecosystem may fulfil the same function, a more even distribution of species within a community may allow the ecosystem to be more resilient to changes (Chapin *et al*., 2000; Hillebrand *et al*., 2008). The lower species evenness detected at inshore stations could be a result of the high number of anthropogenic activities along Algoa Bay's coastline (CLABBS Consortium, 1999; Department of Environmental Affairs, 2014a). Chapin *et al.* (2000) suggests that changes in species evenness occur more rapidly as a response to anthropogenic activities than species diversity and should therefore be monitored closely to identify when a habitat has been impacted. Measures of species evenness and other diversity indices should be collected for long-term monitoring in Algoa Bay.

The community identified as group A, although patchy, was mostly dominated by polychaetes, similar to work conducted by Ellingsen (2002) in unconsolidated habitats in Norway. The dominant polychaete in this study, *Diopatra* spp., is known to be a prominent biogenic habitat engineer, capable of building tubes covered in shells, algae, fibre and other objects (Berke *et al.*, 2010). These tubes stabilise sediment and enhance local diversity by providing attachment surface, and refuge from disturbance and predation (Berke *et al.*, 2010). Group A consisted of more stations than group B, however, although the presence of *Diopatra* spp. may lead to increased secondary diversity, lower total species richness per station (≤12 species) was observed. *A posteriori* visual observation of processed images from group A revealed the substratum was predominantly sandy with some shell fragments visible and low species diversity (<20 species). In comparison, stations grouped into group B had higher species diversity (>20 species) and mixed substrate (rock, sand and shell) was visible in the images processed. This was further reflected by more dispersed data for group B indicating mixed sediment habitats will support higher species diversity as they encompass a wider range of habitat types (Tew *et al*., 2004; Piacenza *et al.,* 2015; Yang *et al*., 2015). The presence of certain species such as *Homophyton spp.* (seafan) are known to require hard substratum for anchoring were only detected in stations from group B. This further supports the finding of two distinct groups of epibenthic communities being driven largely by substratum type, in Algoa Bay.

Alignment of 2012 NBA defined habitat types

The alignment of the 2012 National Biodiversity Assessment (NBA, Sink *et al*., 2012a) habitat types in Algoa Bay with the unconsolidated epibenthic communities identified in this study was investigated by comparing species assemblages within NBA-defined habitat types. The stations sampled in Algoa Bay in this study occurred in four defined habitat types (*Agulhas Sandy Inner Shelf, Agulhas Mixed Sediment Inner Shelf, Agulhas Island and Agulhas Hard Inner Shelf*). A cluster dendrogram with SIMPROF analysis grouped epibenthic assemblages into two significantly different communities (groups), each group including more than two of the four habitat types. Group B encompassed all four represented habitat types, with high within-group variability being observed (similarity observed: 5%). Group A included three habitat types having higher within-group similarity (25%) in species composition. The 2012 NBA habitat classification incorporated drivers such as substrate, depth, geology and biogeography at a coarse scale using the best available data (Sink *et al.*, 2012a). The patterns of epibenthic communities observed in the current study did not align with these broadly classified habitat types. The significant difference observed among stations nested within habitat types supports this, indicating that the stations allocated to each habitat type did not have statistically similar species assemblages. Owing to the higher resolution of this study than that conducted at a national scale, the quality of the data used in the NBA map both in accuracy and categorisation of the habitat type, and the use of hierarchy when surrogates were implemented, resulted in "false" habitats being created. As an example, the island influence zone buffer was given preference over the underlying benthic habitat creating a habitat type that was not represented by the epibenthic communities surveyed. The most important physical distinction detected in stations making up epibenthic groups A and B in this study is the presence of hard substratum, specifically rock, featuring in stations clustered in group B and their absence from group A station.

Stevens and Connolly (2004) and Williams *et al.* (2009) tested the utility of abiotic variables as predictors of habitat types and observed the concepts of false homogeneity and heterogeneity. These studies found that using only abiotic variables for habitat classification was less useful for marine benthic diversity mapping as it diminishes the recognition of important biodiversity features (false homogeneity). In contrast, however, Przeslawski *et al.* (2011) report that although abiotic variables (termed "seascapes") are not consistently useful surrogates, they are appropriate for detecting coarse scale benthic community patterns. Coarse scale habitat mapping can be useful at a national level to identify potential ecologically or biologically important areas (Przeslawski *et al.*, 2011). Similarly, South Africa's 2012 NBA makes use of abiotic variables to map habitat types at a coarse, national scale (Sink *et al.*, 2012a), however, when regional assessments for local management are required (e.g. for MPAs or marine spatial planning), fine to meso-scale habitat mapping (i.e. at a bay scale of 100m - 1km) is vital to inform meaningful management boundaries (zoned activities) within an area. Even the within-station scale (in this study this is at 100m resolution) can inform the management of ecosystem diversity and further refine habitat boundaries. For this reason, comprehensively surveying benthic habitats at a finer scale (100m – 1km), was identified as a high priority action in the 2012 NBA (Sink *et al.*, 2012a).

Internationally, coarse-scale mapping (10-100km) has been conducted, particularly in the northern hemisphere, (Howell *et al.*, 2007; Howell, 2010; Pickrill *et al.*, 2014; Reynolds *et al.*, 2014; Buhl-Mortensen *et al.*, 2015a; b; Bekkby, 2017), using both abiotic (e.g. multibeam surveys) and biotic (species assemblages) variables. Variables used for these mapping exercises included species assemblage information and extensive multibeam surveys that provide explicit physical seabed information. Collecting fine-scale habitat, environmental and species assemblage data throughout the current NBA-defined habitats has been identified as a national priority to enable further refinement of marine habitat types in South Africa. Several studies support using the relationship between environmental factors and benthic macrofauna to model habitats (McArthur *et al.*, 2010; Dutertre *et al.*, 2013; Huang *et al.*, 2014; Kaskela *et al.*, 2017). These studies have shown that using a combination of environmental factors and distributions of selected species can improve and validate existing habitat maps by predicting where habitat types should occur. The mismatch between species data and the 2012 NBA indicates that for local-level mapping and planning, high resolution sampling is required in order to refine national scale information layers. Finer-scale environmental and species assemblage data will enable improved modelling of South Africa's benthic habitats. The process of implementing multi-beam surveys (Council for GeoSciences) for Algoa Bay are underway and the data collected can be validated using the benthic imagery to help build future benthic habitat models (Dorrington *et al.,* 2018).

Limitations of this study

Several limiting factors should be considered when interpreting the results from this study, specifically logistic sampling restrictions owing to the limited availability of survey vessels and equipment, as well as dynamic weather conditions. These restrictions prevented sampling more stations at a greater sampling intensity (number of drops per station) within the Bay. More samples would have likely provided improved resolution (and thus knowledge) of the area surveyed.

Additionally, species abundance data were collected using two different visual survey tools (drop camera and ROV), each capturing different aspects of the epibenthic diversity (ROV captures mobile species). Although two different methods were used for image collection, similar species abundances were recorded in both drop images and ROV still frame images and the scaled ground area assessed remained similar ($0.28m^2$).

Conclusion

This study presents the first quantitative analysis of epibenthic community structure and patterns in unconsolidated benthic habitats in Algoa Bay. Two community types (groups A & B) were identified, but did not align with the current NBA-defined habitat types. Group A was distinguishable by a dominance of polychaetes, low diversity and sandy substratum compared with the highly diverse, mixed substratum community of group B. Analyses conducted in Chapter Three of this study further investigate the extent of environmental data as potential explanatory drivers of the diversity and patterns observed. Important habitats and species have been identified using these images, providing invaluable baseline knowledge prior to the proclamation and implementation of the proposed Addo MPA.

Chapter 3: Identifying potential environmental drivers of epibenthic biodiversity patterns in Algoa Bay

Introduction

One of the primary goals of ecological research is to understand and explain natural processes and the subsequent patterns of species distribution, abundance and behaviour (Underwood *et al.*, 2000). Marine habitats have been defined as "a recognizable space which can be distinguished by its abiotic characteristics and associated biological assemblages, operating at a particular spatial and temporal scale" (ICES, 2005). Observational studies, such as benthic habitat surveys, which form a component of habitat mapping, play a key role in both ecological research and the management of marine resources (Costello, 2009; Huang *et al.*, 2014; Henriques *et al.*, 2015; Buhl-Mortensen *et al.*, 2015a). These studies enhance our perception of ecosystem dynamics and the link between biota and their habitat, providing information towards predicting species occurrences, future pressures and evaluating the efficiency of management interventions (Guarinello *et al.*, 2010; Shumchenia and King, 2010). Depending on the management requirements, habitat mapping often requires extensive surveys at different scales (Diaz *et al.*, 2004; Stevens and Connolly, 2004). The lack of infrastructure available such as vessel access, funding and adequate equipment, and difficult sampling conditions at sea, often prevent comprehensive assessment of all aspects of a given marine system (Post *et al.*, 2006; McArthur *et al.*, 2010; Huang *et al.*, 2011; Przeslawski *et al.*, 2011). Considering such constraints, the use of abiotic factors as surrogates (substitutes) are often implemented to inform the distribution patterns of benthic communities as an approach for habitat classification and mapping (McArthur *et al.*, 2010; Shumchenia and King, 2010; Mellin *et al.*, 2011; Buhl-Mortensen *et al.*, 2012; Dutertre *et al.*, 2013; Douglass *et al.*, 2014; Schultz *et al.*, 2014; Buhl-Mortensen *et al.*, 2015a; Lacharité and Metaxas, 2018).

Abiotic factors frequently used as surrogates in marine ecosystems include measures of physical processes, structures (geology) and chemical composition of the marine system. These factors can act as environmental drivers that regulate habitat condition and influence species distribution and abundance (Douglass *et al.*, 2014). The relationship between the benthic communities and abiotic characteristics can be highly variable at different scales, both temporally and spatially (Underwood *et al.*, 2000; Boström *et al.*, 2011; Cooke *et al.*, 2014; Lecours *et al.*, 2015; Reiss *et al.*, 2015; Robuchon *et al.*, 2017). The strength of the relationship between abiotic factors and benthic community dynamics will thus be dependent on the scale at which the abiotic conditions act and the scale at which sampling was executed (Blanchard *et al.*, 2013; Porter *et al.*, 2017). The link between abiotic and biotic (biota) constituents of marine habitats has not been adequately defined in the literature, however, it is clear that this relationship plays an important role in driving the distribution and abundance of benthic assemblages (Gray, 2002; Bremner *et al.*, 2006; Gray and Elliott, 2009; Przeslawski *et al.*, 2011; Griffiths *et al.*, 2017). Understanding the role abiotic variables play in structuring habitat and biodiversity in an area will provide a good overview for marine resource management and further marine spatial planning.

The use of abiotic factors as surrogates for benthic habitat and biodiversity mapping has traditionally been based on acoustic backscatter imagery (Shumchenia and King, 2010). This delineates landscape level classes that are usually geophysical in origin and influenced by the density, slope and roughness of the seafloor (Gray, 2001; Kenny *et al.,* 2003; Hewitt *et al.,* 2004; Shumchenia and King, 2010; Brown *et al.,* 2011; Raineault *et al.,* 2012). In the absence of this technology, depth has served as a widely accepted surrogate due to its correlation with other influential factors such as light availability and food supply (Brandt *et al.*, 2009; Howell, 2010; Galparsoro *et al.*, 2012; Brown and Thatje, 2014). Bottom samples are also frequently collected to characterise the unconsolidated substrate and infaunal species assemblages that can provide information about higher trophic levels in the food web (Mumby and Harborne, 1999; Mumby and Edwards, 2002; Urbański and Szymelfenig, 2003; Shumchenia and King, 2010; Raineault *et al.*, 2012; Douglass *et al.*, 2014; Huang *et al.*, 2014; Buhl-Mortensen *et al.*, 2015a). Physical factors such as water temperature, dissolved oxygen, organic content, water column nutrients (silicate, phosphate, ammonia, nitrate, etc.), salinity, turbidity and pH are known to directly influence benthic species assemblages and are considered as potential abiotic drivers (Bergen *et al.*, 2001; Post *et al.*, 2006; Belley and Snelgrove, 2016).

Benthic communities often reflect broad-scale processes such as benthic-pelagic coupling, which is primarily the transfer of primary production to the seafloor, and water mass movement driven by currents (Blanchard *et al.*, 2013; Guerra-Castro *et al.*, 2016;

Griffiths *et al.,* 2017; Porter *et al.*, 2017). The variability of broad-scale abiotic processes (spatial and temporal), and localised events, may in turn influence the distribution and composition of benthic assemblages (Hobson *et al.,* 1995, Marcus and Boero, 1998, Blanchard *et al.*, 2013). Understanding the benthic-pelagic coupling between epibenthic communities and the biophysical environment can provide information relevant to the present state of the benthic environment. There is often a lag between physical processes and biotic responses, making long-term monitoring even more important to understanding the relationship between benthic communities and abiotic factors (Griffiths *et al.,* 2017). Several studies have collected abiotic data during biological sampling, however, these snap-shot data do not account for long-term or seasonal trends and variability, particularly abnormal events (Shumchenia and King, 2010; Blanchard *et al.*, 2013; Goschen *et al.,* 2015; Henriques *et al.,* 2015; Karenyi *et al.,* 2016). Long-term monitoring of environmental factors plays a fundamental role in understanding the dynamics of benthic communities and the potential shifts caused by global change or anthropogenic activities (Wolfe *et al.,* 1987; Hily *et al.*, 2008; Lindenmayer and Likens, 2009). Long-term monitoring is usually limited by the available time period of data and the area covered, and as such, the data gathered are often used to create oceanographic or biological models capable of describing the system across wider spatial and temporal scales.

Modelling has often been used to predict the spatial variability of environmental factors across a study area or subsequent species distributions. However, without *in situ* data (particularly long-term for temporal variability) to validate the model, the values predicted cannot provide a holistic assessment of marine ecosystems (Burkhard *et al.,* 2011; Huff *et al.,* 2012). One of the most frequently used models in the marine environment is the Regional Ocean Modelling System (ROMS) that is adapted according to the research objectives for a particular area. ROMS is a terrain (bathymetry) following, 3-dimensional global circulation model that allows for numerous oceanographic variables to be derived (Shchepetkin and McWilliams, 2005; Lett *et al.,* 2006; Filgueira *et al.,* 2012; Huff *et al.,* 2012; Lacharite *et al.,* 2016). Finer-scale models or spatial interpolation methods can be created using GIS software applying Ordinary Kriging or Inverse Distance Weighting (Verfaillie *et al.,* 2006; Jerosch *et al.,* 2006; Tittensor *et al.,* 2010; Fu-cheng *et al.,* 2012; Huff *et al.,* 2012; Jerosch, 2013; Moura *et al.,* 2013; Park and

Jang, 2014; Zarco-Perello and Simões, 2017). Such spatial interpolation models can be suitable for generating information from *in situ* sampling points, particularly in areas where sampling resolution or intensity is limited such as in unconsolidated marine sediment habitats.

Unconsolidated habitats include different types of substrate (mud, gravel, sand and little to no rock) that do not have clearly defined borders and are rather a mosaic of patches, often spanning across large areas. The abundance and distribution of biological assemblages observed in these habitats are often driven by the sediment characteristics such as grain size, sediment sorting and the amount of organic content available. These characteristics in the sediment create new niches for species at different levels in the food chain. Furthermore, depending on species tolerances, variation in several key oceanographic factors such as water temperature, pH, current speed and dissolved oxygen will determine what habitats/ species occur (Morrisey *et al.*, 1992; Ellingsen, 2001, 2002, Gray, 2001, 2002; Hewitt *et al.*, 2004; Stevens, 2005; Anderson, 2008; Chapman *et al.*, 2009; Escaravage *et al.*, 2009; Gray and Elliott, 2009; Thrush *et al.*, 2012; MacKay *et al.*, 2016; Serrano *et al.*, 2017a).

The majority of Algoa Bay benthic habitat types identified in the 2012 National Biodiversity Assessment (NBA, Sink *et al.*, 2012a) are classified as unconsolidated habitats. Algoa Bay forms part of the Algoa Bay Sentinel Site Pelagic Ecosystem Long-Term Ecological Research (PELTER) programme focused on recording oceanographic data within the Bay and neighbouring coastline (Atkinson *et al.,* 2016; Dorrington *et al.*, 2018). In the previous chapter, the epibenthic components of the unconsolidated habitats within Algoa Bay were quantified and described. The current chapter explores potential abiotic drivers of the epibenthic patterns observed. By understanding the potential drivers of unconsolidated habitats in Algoa Bay we will be able to improve habitat and species distribution mapping within the Bay, with implications for similar unconsolidated habitat types along other areas of South Africa's coastline.

The aims of this chapter are to investigate the relationships between key environmental data and patterns of epibenthic biodiversity in unconsolidated habitats of Algoa Bay. Specific objectives are to:

- Describe, interpolate and map key abiotic factors in Algoa Bay
- Test for the effects of the long-term mean values of abiotic factors (temperature, salinity, grain size, etc.) on epibenthic communities
- Test for the effects the long-term variability of abiotic factors on epibenthic communities

Methods

Data collection and processing

Species assemblages

Field sampling and image processing methods are described in Chapter Two. In summary, 13 stations (referred to as biological sampling stations) were surveyed using benthic imagery from a drop camera and Remotely Operated Vehicle (ROV). Images were standardised using a circular frame for the drop camera images and a scaling grid for the ROV. All epibenthic species visible in images from each station were identified to the lowest taxonomical level possible and abundance data were recorded.

Sediment data

Sample collection

A sediment sample was collected from each of the 13 biological sampling stations using a cone dredge (total volume of three litres, Fig. 3.1) towed along the surface of the sea floor for 3-5 minutes at low speed (< 2 knots). The sediment was transferred into jars and frozen for further laboratory analysis.

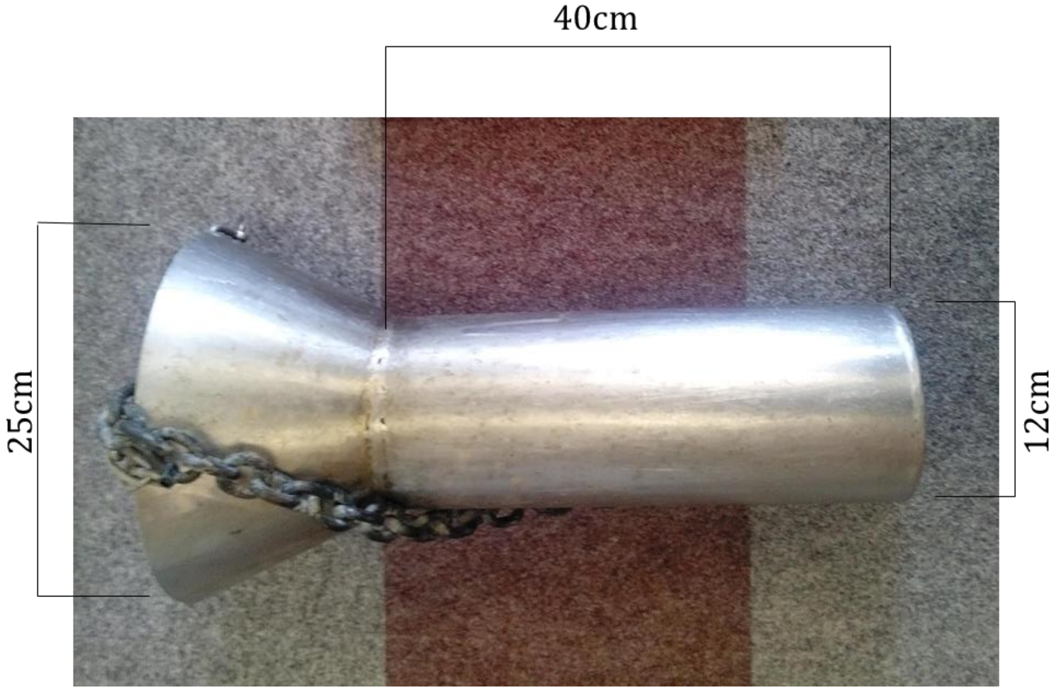

Figure 3.1: Cone dredge used to collect sediment from the top 6cm of the seafloor surface.

Particle size analysis

Sediment particle size was determined using a classical dry sieving method (Buller and McManus, 1979), separating sediment grains according to the intermediate axial length. To dissolve salt from the sediment samples and prevent salt particles influencing the particle size analysis, all sediment (approximately 750g) collected was rinsed using freshwater and left to soak for 24 hours. The sediment was rinsed a second time and left to soak for a further 24 hours, this time using distilled water. After soaking, the sediment was drained and placed in an oven for 24 hours at 70^{0}C to dry. The sediment was gently disaggregated using a mortar and pestle when necessary. Approximately 100-200g of sediment was passed through a sieve stack of decreasing mesh sizes by mechanical shaking for a standard duration of 15 minutes (King Test VB 200/300 Sieve Shaker). The mesh size of the sieve stack ranged from 2000µm to 63µm with a collection tray retaining all sediment smaller than 63µm. The distribution of the sieve stack fractions separates sediment into the widely accepted Udden-Wentworth size classification (Wentworth, 1922; Fig. 3.2). The mass of sediment retained in each sieve was weighed and presented as a proportion of the total mass of sediment processed. The sediment texture and grain size distribution were calculated and classified using Gradistat software version 4.0 as developed by Blott and Pye (2001) yielding the mean grain size (phi) and distribution of grain size across the sample including sorting, skewness and kurtosis.

Millimeters (mm)		Micrometers (µm)	Phi (ϕ)	Wentworth size class	
	4096		-12.0		
				Boulder	Gravel
	256		-8.0		
				Cobble	Gravel
	64		-6.0		
				Pebble	Gravel
	4		-2.0		
				Granule	Gravel
	2.00		-1.0		
				Very coarse sand	Sand
	1.00		0.0		
				Coarse sand	Sand
1/2	0.50	500	1.0		
				Medium sand	Sand
1/4	0.25	250	2.0		
				Fine sand	Sand
1/8	0.125	125	3.0		
				Very fine sand	Sand
1/16	0.0625	63	4.0		
				Coarse silt	Silt
1/32	0.031	31	5.0		
				Medium silt	Silt
1/64	0.0156	15.6	6.0		
				Fine silt	Silt
1/128	0.0078	7.8	7.0		
				Very fine silt	Silt
1/256	0.0039	3.9	8.0		
				Clay	Mud
	0.00006	0.06	14.0		

Figure 3.2: Sediment grain size classification according to the Udden-Wentworth size classification (Wentworth, 1922).

Organic content

Approximately 20g (wet weight) of sediment was removed from the initial wet sediment, for organic content processing before the particle size processing took place. This sediment was placed in a crucible (weighed before sediment added), weighed and dried at 70^{0}C for 24 hours. The crucible with dried sediment was weighed again (dry weight) and then placed in a furnace (ashing oven) at 550^{0}C for 8 hours. Thereafter, the mass of the crucible with sediment (ashed weight) was recorded to determine the percentage organic content from the sediment sample using the following equation:

$$\left(\frac{Md - Ma}{Md}\right) \times 100$$

Md: initial dry sediment mass

Ma: mass of sediment after ashing

This processing yielded a percentage of organic content from each sediment sample collected.

CTD and Nutrient data

Collection

Conductivity, Temperature and Depth (CTD) and nutrient data collected monthly at eight stations (Fig. 3.3) in Algoa Bay since 2010 were made available for this study from the Pelagic Ecosystem Long-Term Ecological Research Programme (PELTERP). PELTERP is a collaborative project between the South African Environmental Observation Network (SAEON) Elwandle node, the South African Institute for Aquatic Biodiversity (SAIAB) and Nelson Mandela University (NMU). This collaborative project aims to monitor the pelagic ecosystems within Algoa Bay and is the first Sentinel Site for long-term marine monitoring in South Africa (Atkinson *et al.*, 2016).

Each month, a SeaBird 19plusV2 CTD mounted to a SBE 55 (6 Niskin bottle) carousel is deployed off the RV *uKwabelana* (SAIAB coastal craft 13m LeeCat) from the starboard side (facing weather) with a davit and electrical winch, at each of the eight monitoring stations. The CTD was soaked for 1 min before deploying a vertical cast at 1m/sec with a sampling rate of 4Hz (approximately 4 scans/m). The sounder depth (minus 5m to avoid contact with sea-floor) ranged from 25-30m for stations 1-6 and 8, and 55-60m for station 7.

The CTD used included several additional sensors that recorded the following parameters: temperature, conductivity, pressure (depth), dissolved oxygen content (DO), pH, chlorophyll *a* (Chla)and turbidity. The Niskin bottles collect water during the CTD cast at 10, 20, 30 and 60m. The water is later analysed for nutrients including Ammonium

(NH_4), Silicate (SiO_4), Phosphate (PO_4) and Nitrate (NO_x). Raw CTD data and nutrient data (NH_4, Si, PO_4 and NO_x) were requested for the period 2012-2016 from SAEON Elwandle node for further processing for the purposes of this study.

Processing

Raw CTD data (.hex and corresponding .con files) were processed using SBE Data Processing Version 7.26.7 (Sea-Bird Scientific, 2017) software to remove outliers and abnormalities in the data. Monthly data collected over the four-year period (2012-2016) were exported and the bottom three meters of up-cast data were extracted. These monthly data (bottom 3m) were combined for the four-year period and descriptive statistics were calculated (mean, standard deviation and coefficient of variation). Key variables selected from the CTD data for further processing included salinity (derived from conductivity), turbidity, dissolved oxygen, chlorophyll-a and pH. Temperature data were extracted from permanently moored Underwater Temperature Recorders (UTRs) providing daily recordings rather than only monthly recordings (as obtained from the CTD casts).

Monthly nutrient data were provided in processed format from which the bottom 30 or 60 meters were combined per station and descriptive statistics were calculated (mean, standard deviation and coefficient of variation).

Current speed and temperature data

Collection

Acoustic Doppler Current Profilers (ADCP) and UTRs in Algoa Bay form part of the SAEON Continuous Monitoring Platform in Algoa and St Francis Bays (Atkinson *et al.*, 2016). Two ADCP recorders are permanently moored at 30m depth on either side of the Bay (Cape Recife and Bird Island, Fig. 3.3) with 10 UTR sites located near to the eight PELTERP stations and two additional UTRs around Bird Island. Shallow UTR moorings are deployed at 30m depth with the deeper moorings deployed between 60-80m bottom depth (Fig. 3.3). The UTRs are placed on the moorings at 10m depth intervals from the bottom until 10m below the surface and record temperature hourly (Goschen *et al.*, 2015). Currents are measured by a Teledyne RDI Express 600kHz ADCP placed in a frame secured to the sea floor.

Physical data obtained for this study included mean monthly bottom current speed and direction (from the ADCPs) and bottom temperature from the UTRs for 2009-2017. In addition, modelled bottom temperature and current speed values were generated for the present study's 13 biological sampling stations in the Bay using a third-order, upstream-biased advection scheme ROMS-based model. This model was provided by Mr Dylan Bailey (PhD candidate, Nelson Mandela University), focusing on fine scale modelling within the bay using nested domains of ¼°, 1/12°, 1/32°, and 1/108° spatial resolution to produce statistical means, minimums, maximums and standard deviation per month, season and year. The model was validated using *in situ* data generated by the PELTERP stations and permanent ADCP moorings in the Bay (Bailey, Goschen and Hermes, in prep).

Processing

Mean monthly data were collated, and descriptive statistics (mean, standard deviation and coefficient of variation) were calculated per station for further analysis.

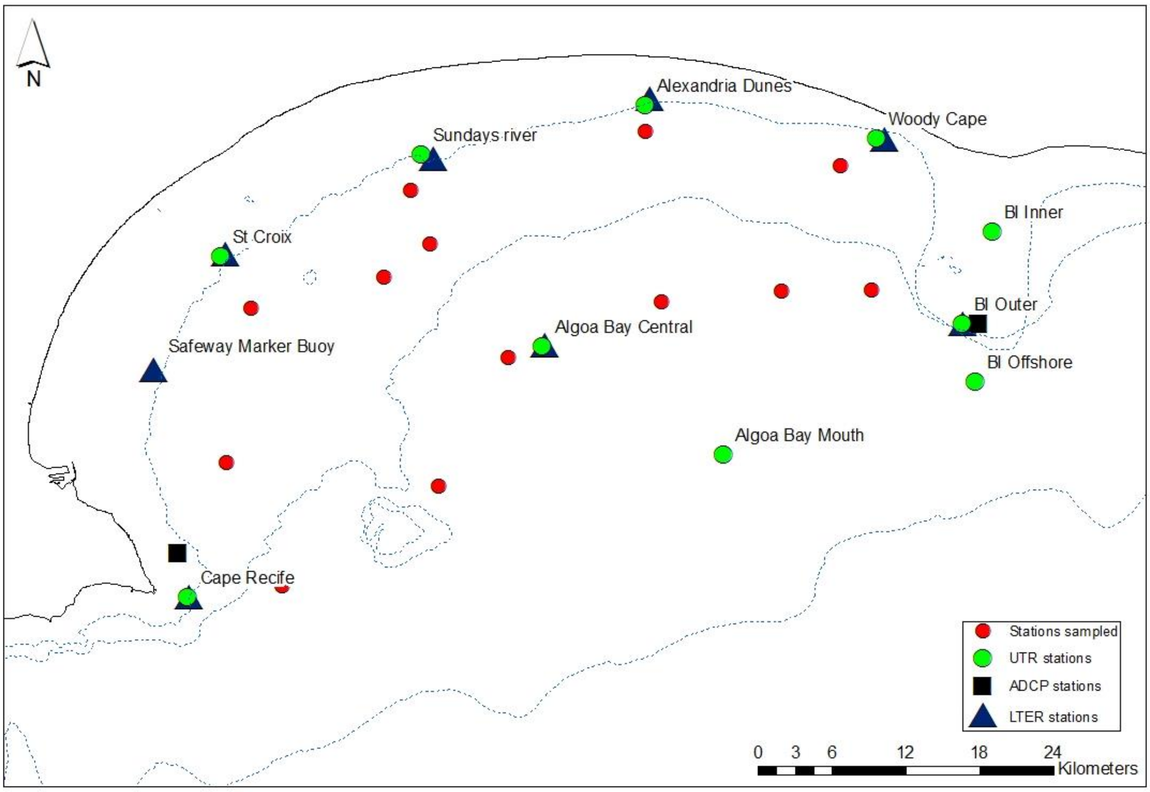

Figure 3.3: SAEON Algoa Bay Sentinel Site Long-term monitoring stations with biological sampling stations of the present study depicted by red dots.

Interpolation of abiotic data

Benthic sampling stations in the present study did not spatially align with oceanographic physical sampling locations. To obtain spatially paired data, the oceanographic variables were interpolated using ArcGIS (Esri, 2010) using ordinary kriging interpolation. The data were kriged to create a surface layer for the Bay using the ordinary kriging default parameters. This surface layer was created by considering the individual oceanographic variable and predicting (interpolating) the values in the area around each station. The higher the sampling intensity, the more accurate the interpolation. The values generated from interpolation are included in the attributes table of the 13 biological survey stations by using the prediction/ validation function from the surface layer created during kriging (Table 3.1). This interpolation and extraction were done for each variable using the CTD and UTR information. The ADCP data contained too few data points (only two points at either end of the Bay) and was therefore excluded from interpolation. The ROMS model output was used as an alternative to provide values for bottom temperature (to compare with the UTR kriged values), current speed and direction. Data from the ADCPs were only used for validation.

In addition to the sediment collected and processed for grain size at each of the biological stations, SAEON Elwandle also provided similar grain size data for a further 33 positions in Algoa Bay. A total of 46 sediment grain size positions were then used to create an interpolated sediment map for Algoa Bay using the same method of interpolation as described above.

Statistical analysis

Fifteen abiotic factors were selected based on their recognised potential to influence species distribution and assemblage composition in Algoa Bay (Table 3.1). All analyses were performed using PRIMER (Plymouth Routines in Multivariate Ecological Research) version 6.1.18 with PERMANOVA+ version 1.0.8 extension (Anderson *et al.*, 2008). Draftsman plots were generated for the mean of each factor and for the coefficient of variation to determine whether variables were directly correlated. When the draftsman plots indicated factors showing uneven distribution in the data, these factors were 4th root transformed. However, the transformed data remained unevenly distributed but were used if no strong correlation were found to another factor. Where factors exhibited a strong correlation with another factor (greater than 0.8 or normally distributed), only one of the factors was selected for further analysis based on the potential to drive species distribution and abundance (Table 3.1).

The relationships between epibenthic species patterns (biotic data) and the abiotic factors at each station were explored using a distance-based linear model (DistLM) and visualised using a distance-based redundancy analysis (dbRDA, Anderson *et al.*, 2008). The model assesses each factor against the biotic data and tests the variation explained by each abiotic factor alone, ignoring all other variables, resulting in marginal test outputs (Anderson *et al.*, 2008). The dbRDA visualises the spatial distribution of the DistLM result and generates the percentage explained by the fitted model and total variation explained in the resemblance matrix on each axis. Using the R^2 selection criteria the "step-wise" procedure begins with a null model adding and removing variables that will improve the selection criterion, and only stops when no further improvements can be made, resulting in sequential test outputs (Anderson *et al.,* 2008). Two DistLM procedures were run, one analysed the long-term mean data for each factor, while the other analysed the coefficient of variation.

Table 3.1: Full set of possible abiotic factors considered for DistLM analysis with an indication of the factors that were selected following a draftsman plot assessment.

Factor	***Source***	***DistLM analysis***	***Reason for use/disuse***
Depth	Station	Mean	Directly linked to station
Mean grain size (ϕ)	Station	Mean	Directly linked to station
Organic content	Station	Mean	Directly linked to station
Dissolved oxygen (DO)	LTER - CTD	Mean and CoV	No correlation
Chlorophyll a (Chla)	LTER - CTD	Mean and CoV	Selected to represent several correlated factors
Turbidity	LTER - CTD	CoV	Minimal variation across stations
Salinity	LTER - CTD	CoV	Minimal variation across stations
pH	LTER - CTD	CoV	Minimal variation across stations
Bottom temperature (Kriged UTR)	UTR mooring	No	Modelled data showed more normalised data
Bottom temperature (Model)	ROMS model	Mean	Only correlated with depth due to ROMS
Bottom current (Model)	ROMS model	Mean	Only correlated with depth due to ROMS
Ammonium (NH_4)	LTER - CTD	No	Minimal variation across stations
Silicate ($Si\ O_4$)	LTER - CTD	Mean	Minimal variation across stations
Phosphate (PO_4)	LTER - CTD	No	Minimal variation across stations
Nitrate (NO_x)	LTER - CTD	No	Minimal variation across stations

Results

Interpolated physical parameters proved to be homogenous across the Bay with slight change from west to east in mean and CoV for both Chla and turbidity. Mean grain size, bottom temperature and bottom current speed as well as coefficient of variation of bottom temperature, bottom current speed and dissolved oxygen explained the highest proportion of variation among stations.

Patterns observed from interpolated and mapped abiotic factors

Long-term means for most physical parameters from the PELTERP CTD stations were found to be homogenous across the Bay including salinity, bottom temperature, pH and DO (Fig. 3.4B, D, E & F). Mean Chla decreased from west to east with higher Chla evident near the harbour (Fig. 3.4C). Turbidity follows a similar trend as that of Chla with higher concentrations in the north-west portion of the Bay (Fig. 3.4A). The coefficient of variation (CoV) for factors sampled at the CTD stations also followed similar trends with the variation decreasing spatially from west to east in the Bay (Fig. 3.5). The CoV for turbidity and Chla showed greater variability in the western sector of the Bay (Fig. 3.5C). The salinity and dissolved oxygen concentrations showed a similar trend in the western portion of the Bay, although the differences in class values were minimal (Fig. 3.5B & E). Mean nutrients values displayed a decrease in mean from west to east for both Phosphate (PO_4) and Ammonia (NH_4, Fig. 3.6C & D), while Nitrate (NO_x) showed little variation in mean across the Bay (Fig. 3.6B). Higher mean silicate (Si) values appear to concentrate near the centre of the Bay and decrease in value towards the eastern po rtion of the Bay (Fig. 3.6A).

Interpolated mean grain size of Algoa Bay (Fig. 3.7) indicates three areas of course grain size (-2.54 – 0, Phi), specifically a small pocket off Cape Receife in the west, a small inshore area adjacent to the shore and Alexandria dunes, and dominating eastwards towards Bird Island. This pattern is similar to the silicate values observed from the interpolated map (Fig. 3.6A). Fine sediment (>2, Phi) appears to concentrate in the middle of the Bay between the 30 – 50m depth contour avoiding the smaller reef complexes and the areas adjacent to the shore between Alexandria dunes and Woody Cape. The majority of the Bay was classified as medium sand (0-1, Phi).

Analysis of potential drivers of benthic communities

Effects of long-term mean abiotic factors

Draftsman plots revealed a strong correlation between several of the abiotic factors considered as potential drivers of species abundance and diversity. Where the correlation value exceeded 0.8 between two factors, only one was selected for inclusion in the subsequent DistLM model analysis (Table 3.2). The modelled (ROMS) bottom current and temperature data were included for further analysis as the draftsman plots were more normally distributed than mean interpolated bottom temperature (kriged).

Table 3.2: Output from Draftsman plots indicating long-term mean abiotic factors with correlations greater than 0.8. Factors selected for further DistLM analysis are identified.

Abiotic factors	***Correlation value***	***Factor selected for DistLM***
Model bottom temp + depth	-0.96997	Depth
Model bottom current + depth	0.879916	Depth
NH_4 + Chla	0.96813	Chla
Turbidity + Salinity	-0.99385	Turbidity
Turbidity + pH	-0.99385	Turbidity
Model bottom temp + Model bottom current	-0.85473	Model bottom temperature

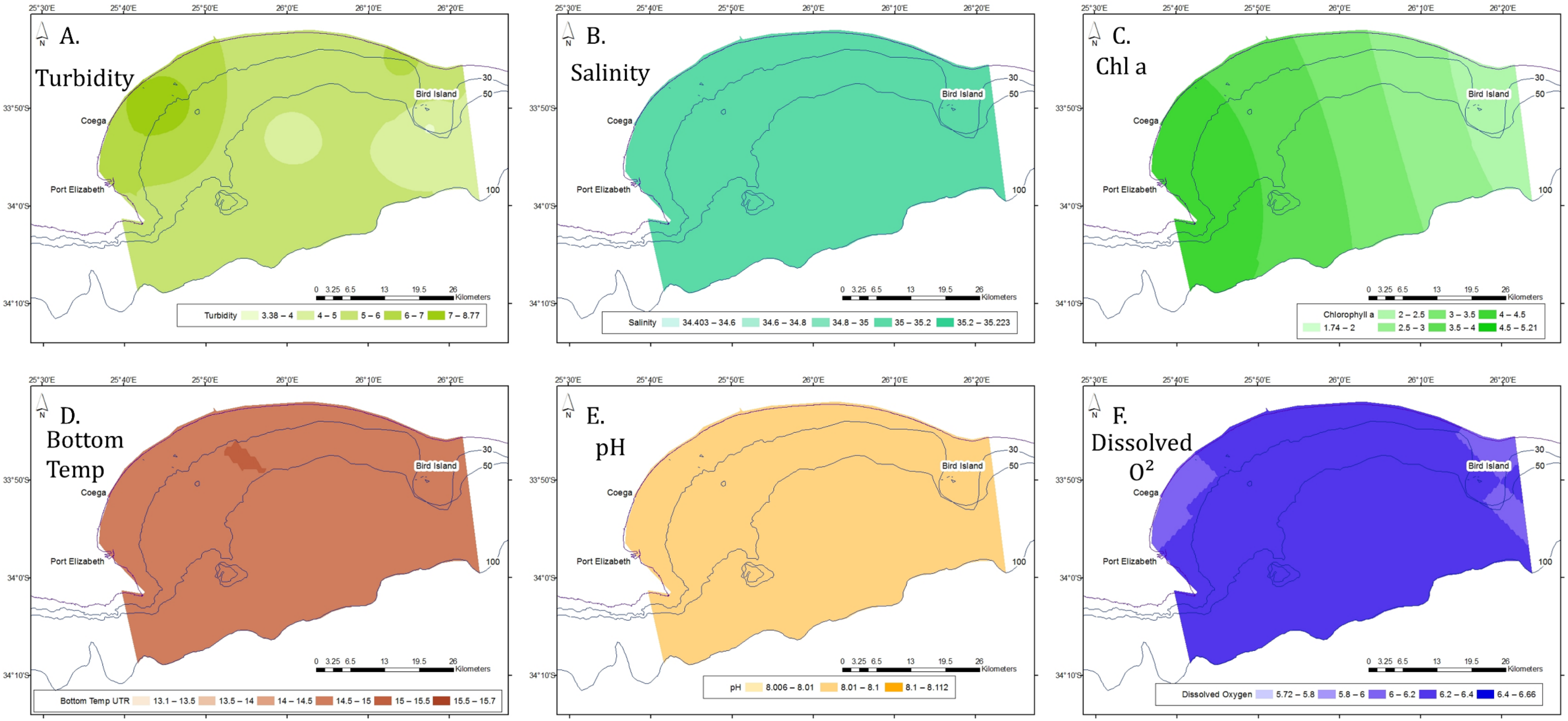

Figure 3.4: Interpolated long-term mean values for abiotic factors recorded at the PELTERP CTD and Continuous Monitoring platform, A.) turbidity, B.) salinity, C.) Chla, D.) bottom temperature, E.) pH, and F.) dissolved oxygen.

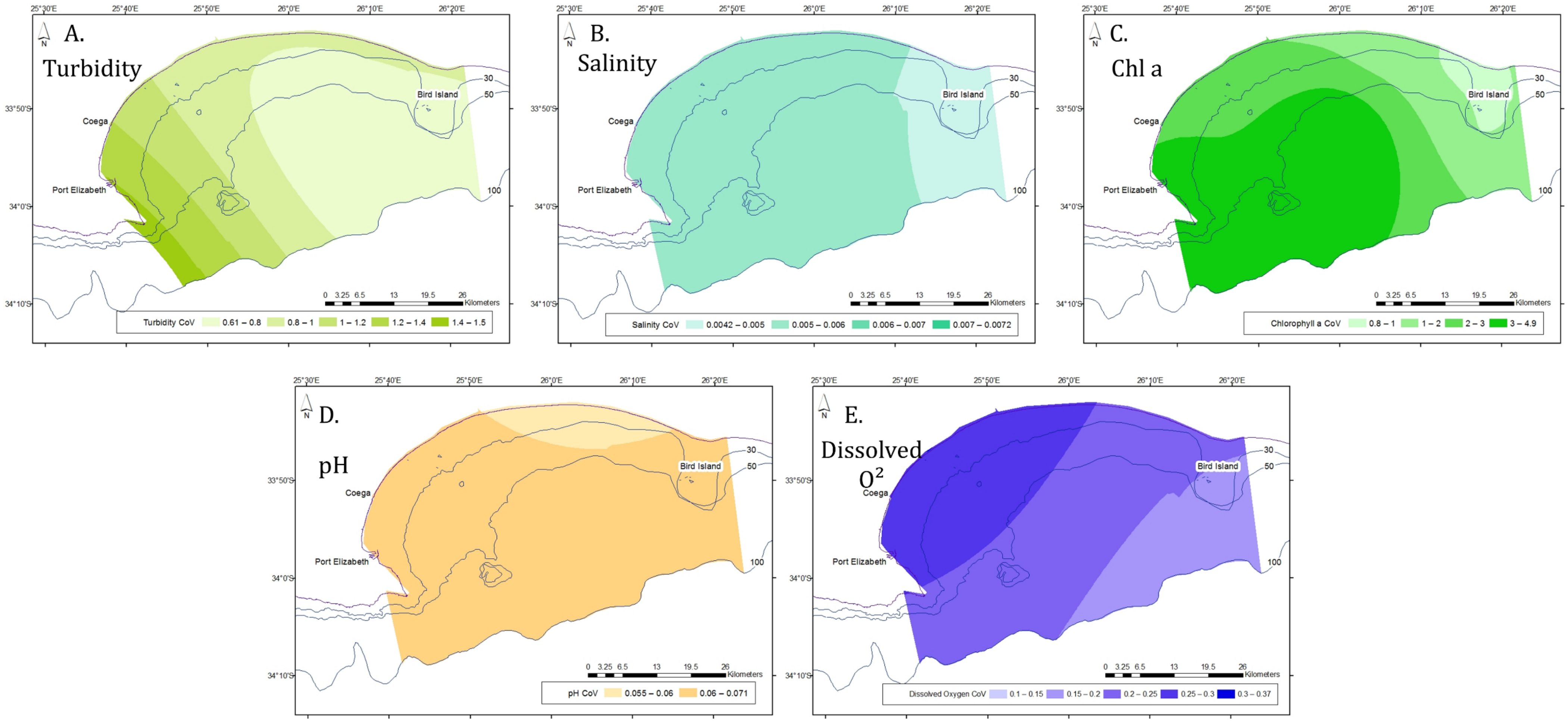

Figure 3.5: Interpolated long-term coefficient of variation values for abiotic factors recorded at the PELTERP CTD, A.) turbidity, B.) salinity, C.) Chla, D.) pH and E.) dissolved oxygen).

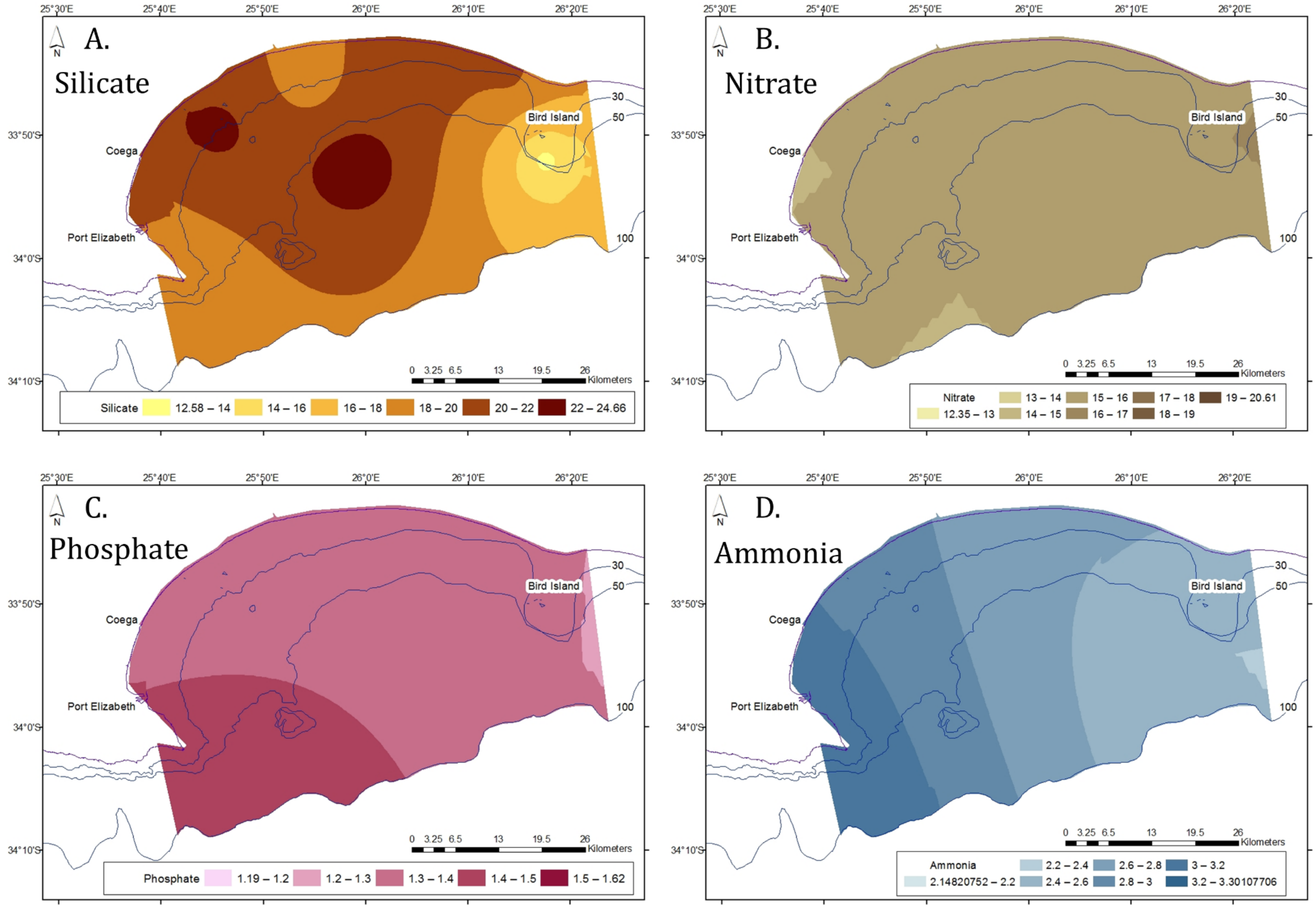

Figure 3.6: Interpolated mean nutrients values for factors recorded at the PELTER CTD, A.) Silicate, B.) Nitrate, C.) Phosphate and D.) Ammonium.

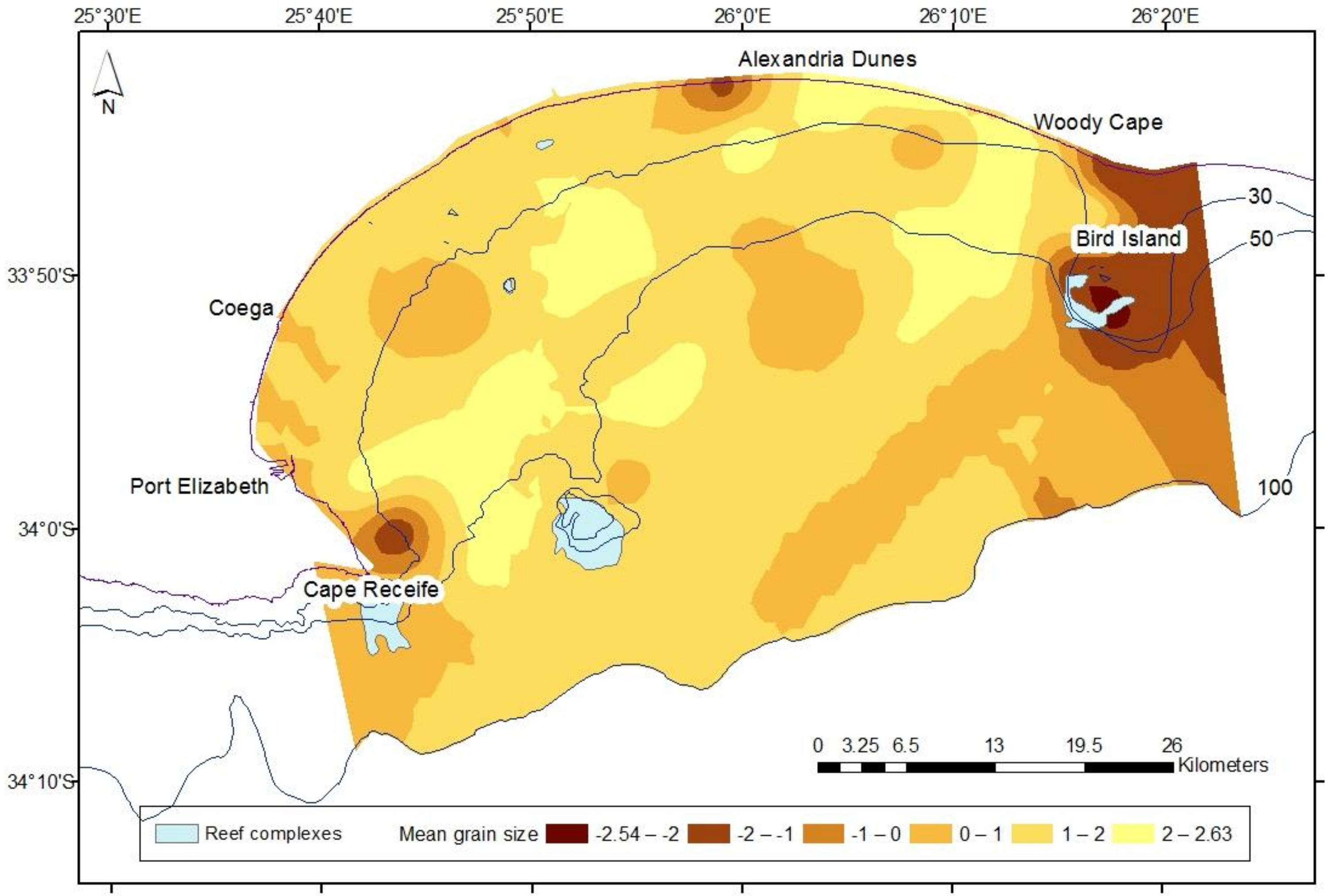

Figure 3.7: Interpolated mean phi grain size from 46 sediment samples collected within Algoa Bay including estimated areas of reef complexes. The lowest value (-) represents course sand and as the mean phi value increases, so the sediment grain size becomes finer (+) (refer to Wentworth table, Fig. 3.2).

Table 3.3: Mean of all factors considered for DistLM analysis. Values shaded in grey represent those generated by interpolation (Ordinary Kriging).

Stations	***Depth***	***Mean grain size***	***Organic content***	***Mean DO***	***Mean Chla***	***Mean Turb***	***Mean Salinity***	***Mean pH***	***Mean Temp***	***Mean Temp-model***	***Mean Current***	***NH4***	***Nox***	***PO4***	***SiO4***
A1_1	41.3	4.486	2.1429	6.0692	4.4861	6.5332	35.196	8.0689	14.9822	15.9921	0.0622	3.0182	15.119	1.4066	19.972
A1_2	41.4	-0.239	3.3641	6.0353	4.407	6.5404	35.196	8.0689	14.9416	16.1308	0.0459	2.9297	15.119	1.3789	21.902
A1_3	46.6	2.629	2.4532	6.0462	3.9999	6.532	35.196	8.0689	14.9822	15.2912	0.0573	2.7697	15.119	1.3655	20.802
ROV1_3	44.8	2.492	2.4405	6.043	4.1289	6.5329	35.196	8.0689	14.9822	15.5175	0.0588	2.813	15.119	1.3724	21.218
ROV1_4	34.9	2.322	2.3694	6.0499	4.0013	6.5364	35.196	8.0689	15.0276	16.3752	0.0461	2.7758	15.119	1.358	19.733
ROV1_5	36	1.265	0.6261	6.0721	3.3941	6.5284	35.196	8.0689	14.8400	16.3879	0.0433	2.6427	15.119	1.339	21.027
ROV1_6	34.5	2.417	1.9849	6.0237	3.108	6.7357	35.195	8.0671	14.8400	16.5358	0.0473	2.5645	15.514	1.3209	19.972
A2_1	60.8	2.394	2.9932	6.0844	4.383	6.5222	35.196	8.0689	14.7788	14.8688	0.0902	3.0222	15.119	1.4266	19.262
A2_2	52.8	0.652	0.8627	6.059	4.1055	6.5163	35.196	8.0689	14.8400	14.7720	0.0644	2.8271	15.119	1.4045	20.964
A2_3	66.3	2.231	1.2599	6.0378	3.8919	6.5197	35.196	8.0689	14.7788	14.5025	0.0673	2.7249	15.119	1.3788	23.153
A2_4	64.8	0.131	3.1074	6.0598	3.4787	6.5159	35.196	8.0689	14.8400	14.2540	0.0737	2.6008	15.119	1.3561	21.133
A2_5	68.4	2.137	0.3731	6.078	3.205	6.5133	35.196	8.0689	14.8400	14.1121	0.0796	2.533	15.119	1.3409	19.146
A2_6	64.7	1.384	1.9670	6.038	3.1687	6.7288	35.195	8.0671	14.8400	13.9397	0.0791	2.4824	15.514	1.3161	17.266

Table 3.4: Coefficient of variation for all factors considered for DistLM analysis. Values shaded in grey represent those generated by interpolation (Ordinary Kriging).

Stations	CoV DO	CoV Chla	CoV Turb	CoV Salinity	CoV pH	CoV Temp	CoV Temp-model	CoV Current
A1_1	0.259814	3.42937	1.215878	0.005746	0.066155	0.098625	0.10087737	0.584824
A1_2	0.281326	2.765136	1.023515	0.005722	0.063743	0.093049	0.110167841	0.600491
A1_3	0.263481	3.143513	0.820425	0.005516	0.061635	0.101316	0.098223425	0.57784
ROV1_3	0.266844	3.197844	0.869477	0.005576	0.062394	0.100046	0.09216323	0.559597
ROV1_4	0.272151	2.567508	0.827738	0.005539	0.061023	0.097559	0.104794637	0.565733
ROV1_5	0.241917	1.724672	0.77207	0.00523	0.059471	0.09897	0.103119014	0.568718
ROV1_6	0.215268	1.090134	0.779015	0.004967	0.060176	0.098331	0.107988304	0.630419
A2_1	0.235356	3.51765	1.290891	0.005689	0.067501	0.101012	0.101832288	0.58672
A2_2	0.230883	4.003057	0.958715	0.005513	0.065272	0.107298	0.08851715	0.601707
A2_3	0.236921	4.652126	0.790977	0.005414	0.06295	0.114307	0.083783361	0.551246
A2_4	0.220931	3.247465	0.691188	0.005198	0.061653	0.111789	0.081234513	0.544402
A2_5	0.207078	2.040589	0.679638	0.005049	0.061519	0.108044	0.082608434	0.552007
A2_6	0.201471	1.371431	0.693644	0.0049	0.061494	0.104546	0.098157202	0.618342

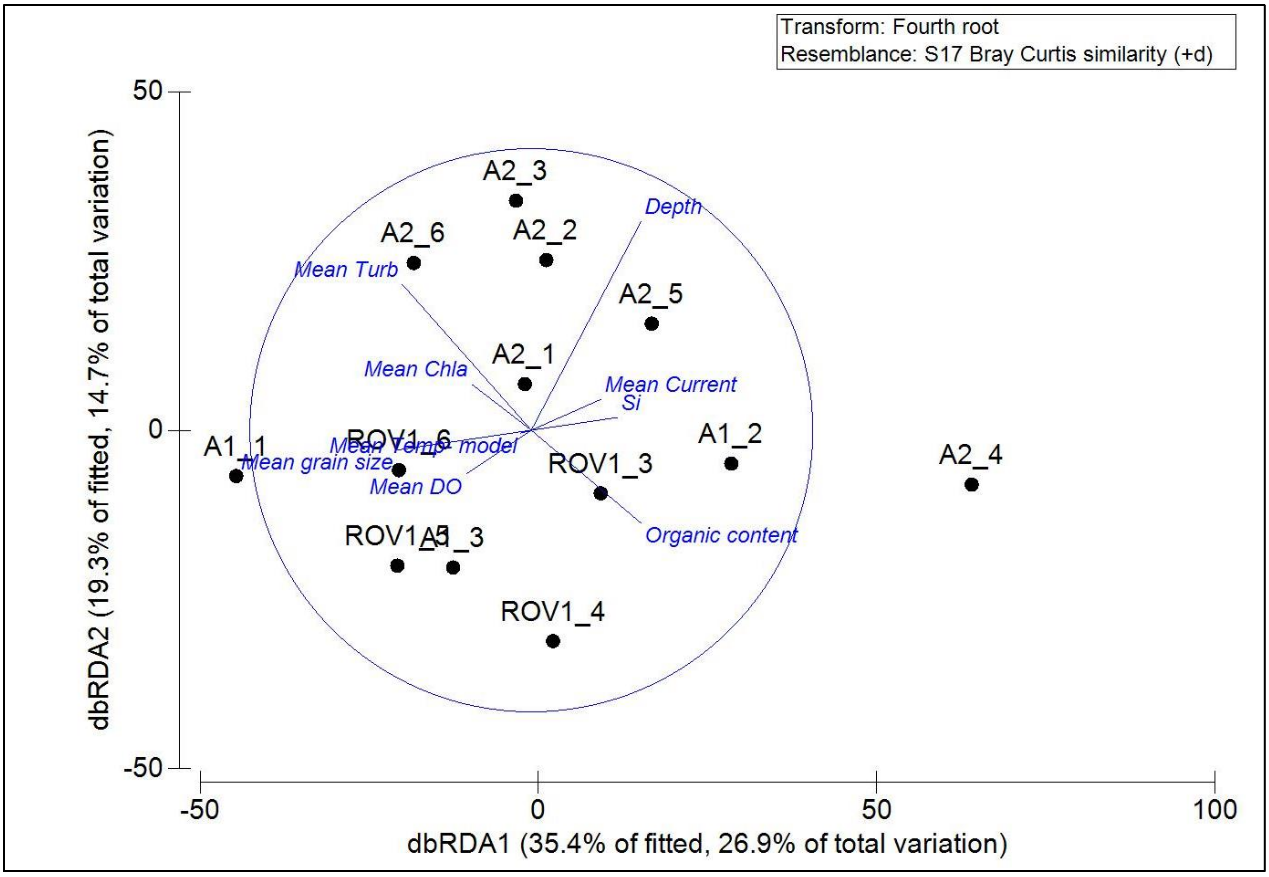

Figure 3.8: Distance-based redundancy analysis (dbRDA) plot visualising the DistLM analysis based on the mean abiotic factors fitted to the variation of species abundance and diversity patterns observed across the 13 biological stations sampled.

The distance-based linear model (DistLM) analysis of epibenthic communities at each station with mean abiotic factors selected for inclusion (Table 3.3 & 3.4) explained 54.7% of the total fitted variation and 41.6% of the total variation from both axes in the dbRDA (Fig. 3.8). Marginal test results showed that none of the individual abiotic factors measured contributed significantly to the separation among stations (P>0.05, Table 3.4). The 'best' model solution shows that depth (14.62%) provides the best fit of the data for Algoa Bay. It is also noted that mean grain size, mean temperature and mean bottom current speed explain a relatively higher proportion (>10%) of variation among stations.

Table 3.5: Marginal test statistics for Distance-based linear model (DistLM) analysis of long-term means based on Step-wise procedure and R2 criteria. Values in bold contribute greater than 10% variation. SS = Sum of Squares, RSS = residual Sum of Squares, R2 = RSS/SS.

	Abiotic factor	***SS(trace)***	***Pseudo-F***	***P***	***Prop.***
1	Depth	4711.6	1.8841	0.0733	**0.1462**
2	Si	2531.0	0.9378	0.4682	0.0786
3	Mean grain size	4098.7	1.6033	0.1185	**0.1272**
4	Organic content	2938.5	1.1039	0.3168	0.0912
5	Mean DO	1579.4	0.5670	0.8574	0.0490
6	Mean Chla	2242.0	0.8227	0.5732	0.0696
7	Mean turbidity	2068.4	0.7546	0.7371	0.0642
8	Mean bottom temp- model	4445.6	1.7607	0.0845	**0.1380**
9	Mean bottom current speed	3655.2	1.4076	0.1682	**0.1135**
Overall best solution:					
		R^2	RSS	No. Vars	Selections
Best		0.75827	7788.2	9	1-9

Effects of long-term variation in abiotic factors

Only modelled (kriged or using ROMS) abiotic factors were considered when calculating the coefficient of variation as a factor contributing to the variation of species diversity and abundance across stations (all long-term data). Draftsman plots revealed a strong correlation between several of the abiotic factors considered as potential drivers of species abundance and diversity. Where the correlation value exceeded 0.8 between two factors, only one was selected for inclusion in the subsequent DistLM model analysis (Table 3.6).

Table 3.6: Output from Draftsman plots indicating long-term variation of abiotic factors (CoV) correlating greater than 0.8 and which factor was selected for further analysis.

Abiotic factors	***Correlation***	***Factor selected for DistLM***
DO + Salinity	0.82911	DO
Turbidity + pH	0.88022	Turbidity
Temp + Temp model	-0.916	Temp model

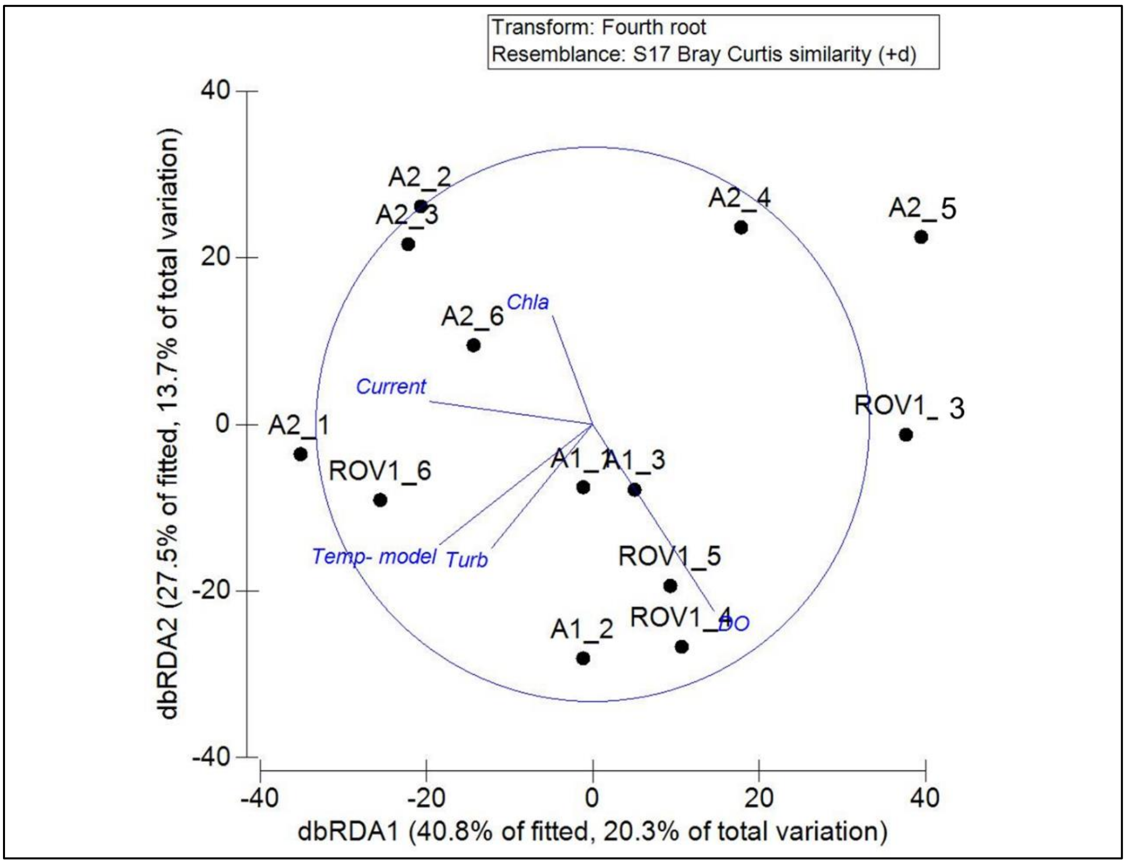

Figure 3.9: Distance-based redundancy analysis (dbRDA) plot representing the DistLM analysis of long-term variation of abiotic based on the abiotic factors (CoV) fitted to the variation of species abundance and diversity patterns observed across the 13 biological stations sampled.

The distance-based linear model (DistLM) analysis of epibenthic communities at each station with coefficient of variance (CoV) for abiotic factors selected for inclusion (Table 3.6) explained 68.3% of the total fitted variation and 34% of the total variation (Fig. 3.9). Marginal test results showed that none of the individual abiotic factors contributed significantly to the separation among stations (P>0.05, Table 3.7). Modelled CoV for bottom temperature (13.6%), bottom current speed (10.4%) and dissolved oxygen (10.4%) factors explain a relatively higher proportion (>10%) of variation among stations. In addition, sequential tests indicate the cumulative contribution of all abiotic factors explained 49.8% of the variation (Table 3.7). This is further supported by the best model solution indicated to include all factors addressed.

Table 3.7: Marginal test statistics for Distance-based linear model (DistLM) analysis of CoV data based on Step-wise procedure and R^2 criteria. Values in bold contribute a 10% or greater proportion when compared to other factors. SS = Sum of Squares, RSS = residual Sum of Squares, R2 = RSS/ SS.

Marginal tests							
	Abiotic factor	**SS(trace)**	**Pseudo-F**	**P**	**Prop.**		
1	DO	3351.2	1.2769	0.2221	**0.1040**		
2	Chla	2070.5	0.75545	0.6525	0.0643		
3	Turb	2748.7	1.026	0.3865	0.0853		
4	Temp- model	4380.4	1.7308	0.0953	**0.1360**		
5	Current	3362.4	1.2817	0.2164	**0.1044**		
Sequential tests:							
Abiotic factor	R^2	*SS(trace)*	**Pseudo-F**	**P**	**Prop.**	**Cumul.**	**res.df**
+Temp- model	0.13596	4380.4	1.7308	0.0928	0.13596	0.13596	11
+Current	0.23348	3142.1	1.2723	0.2327	9.75E-02	0.23348	10
+Chla	0.32934	3088.6	1.2864	0.2305	9.59E-02	0.32934	9
+DO	0.44354	3679.4	1.6418	0.1484	0.1142	0.44354	8
+Turb	0.49795	1753	0.75864	0.6014	5.44E-02	**0.49795**	7
Overall best solution:							
		R^2	RSS	No.Vars	Selections		
Best		0.75827	7788.2	9	1-9		

Discussion

When sufficient data are available and the correct modelling techniques applied, the spatial distribution of any ecological unit can be modelled to support marine management decisions (Gonzalez-Mirelis and Buhl-Mortensen, 2015). Interpolation of the abiotic factors measured within Algoa Bay can be considered a preliminary step towards generating point information linking physical variables with the biological epibenthic species component observed in Chapter Two. Interpolated maps generated for both the mean and coefficient of variation for several abiotic factors showed a spatial pattern of decreasing values from west to east in Algoa Bay. This pattern was evident for the long term means of turbidity, Chl*a*, SiO_4, PO_4 and NH_4 (Figs. 3.4A & C, & 3.6A, C & D), while all CoV factors showed also followed this pattern with a greater long-term variation in the western sector decreasing to less variable to the east (Fig. 3.5). A lower CoV indicates a lower variability in the abiotic factor considered and epibenthic communities with narrower environmental tolerances may prefer these areas (Costello *et al.*, 2015). The shape (log-spiral) and orientation (east-ward facing) of the Algoa Bay and the direction of the adjacent fast flowing Agulhas current (south-westward) are likely contribute to the patterns observed (Pattrick and Strydom, 2017). The Agulhas current brings nutrients into the Bay from offshore with the western part of the Bay being the last area the current reaches, retaining the nutrients for longer (Goschen and Schuman, 1994). This may influence the levels of variability between the east and western parts of the Bay. Additional mechanisms may also cause further variation of abiotic factors such as seasonal upwelling, meanders or episodic Natal pulses (Goschen *et al.*, 2015). Possible anthropogenic influences may also play a role in the nutrients and Chl*a* patterns observed as the western part of the Bay supports the highest human population owing to the position of the city of Port Elizabeth and the port, and thereby increasing waste water input into the Bay in this area (Adeniji *et al.*, 2017).

The lack of significant effects of the abiotic factors on the patterns of epibenthic communities observed suggests that no individual factor should be considered as the main driver, rather the cumulative effect of several factors play key roles in the epibenthic community distribution. Of all the factors considered as potential drivers, depth, mean grain size, modelled bottom current speed and temperature accounted similar proportions of epibenthic variation (14.6%, 12.7%, 11.4% and 13.8% respectively). As demonstrated in this study, depth and mean grain size are known globally to be drivers of species diversity patterns and distribution in unconsolidated habitats (Gray, 2001; Kruger *et al*., 2005; Whittington *et al*., 2006; McClain *et al*., 2010; Pitcher *et al*., 2012; Brown and Thatje, 2014; Lange and Griffiths, 2014; Piacenza *et al*., 2015; Serrano *et al*., 2017a; b). While mean grain size is considered appropriate to account for differences observed in unconsolidated habitats (exclusively soft sediment), it is unlikely to detect mixed substrates (mosaics) that include patches of hard ground (rock) as a result of the sampling tool used (i.e. grab or dredge). This finding was highlighted in Chapter Two where a visual assessment of each image showed that where hard ground (rock) and the associated fauna featured in the community assemblages were determined to be significantly different. Similarly, Stevens and Connolly (2004) found that using individual abiotic factors (depth, sediment and distance from river) did not discriminate sufficiently between soft bottom communities to be a reliable basis for mapping. as such, without rigorous biological surveys at the appropriate scale and level of resolution, little confidence can be placed in modelled marine habitat classifications (Verfaillie *et al.* 2006; Gonzalez-Mirelis and Buhl-Mortensen, 2015; Lecours *et al*., 2015). The interpolated mean grain size map of Algoa Bay (Fig. 3.7) does align to some extent with the current known reef complexes as coarse sand is observed closer to said reefs. The addition of multibeam side-scan sonar data would enable further refinement of the surficial layer. A comprehensive sediment map of Algoa Bay could then be utilised as a potential surrogate for habitat mapping, as is done in several other such studies (Howell, 2010; Pitcher *et al*., 2012; Hill *et al.,* 2014; Kaskela *et al.,* 2017; McHenry *et al.,* 2017).

The use of different descriptive statistics can provide insight into various aspects of long-term data usage. Considering only the mean of abiotic factors over a broad temporal scale removes comprehensive aspects of the factor, such as variability and extreme events, that may play an important role in understanding the patterns of epibenthic communities (Heath and Borowski, 2013). The mean and CoV of interpolated abiotic factors showed a similar west to east pattern of decreasing value, however the CoV showed that although small, there is distinct variation in pH, salinity and dissolved oxygen values across Algoa Bay. This would not have been detected if only the mean values of these abiotic factors had been considered. Stow *et al*. (1998) state that during environmental monitoring it must be recognised that many environmental characteristics (abiotic factors) are intrinsically highly variable and to assess environmental patterns, the variation of the data should be understood. Therefore, including the CoV does provide insight as a potential driver of epibenthic communities that is often excluded during such studies (Franken, 2015; Lacharite *et al.*, 2016; MacKay *et al.,* 2016; Makwela *et al.,*2017; Post *et al.,* 2017; Lacharite and Metaxas, 2018)

Limitations of this study

During this study, it was only possible to sample only 13 stations for epibenthic communities in Algoa Bay (total sampled area of $171.4m^2$). As with most subtidal and offshore benthic studies, sampling is logistically limited, therefore relatively small are sampled and these may not account for the full range of habitat heterogeneity. The approached used did however consider the known literature indicating less variability in unconsolidated habitats. The abiotic factors used during analysis were interpolated from eight stations in Algoa Bay, limiting the accuracy of interpolation and the potential utility of output data for explaining epibenthic patterns. Additional surveys are required in Algoa Bay to provide more detailed information on the substrate composition using different techniques to account for hard ground, such as side scan sonar and multi-beam assessments. Other variables should also be considered such as proximity to reef complexes (possible higher species diversity closer to reefs) and zooplankton (bentho-pelagic coupling) sampling. This additional information will support and enhance the ongoing fine-scale habitat mapping in Algoa Bay.

Conclusion

This study presents the first assessment of abiotic factors in Algoa Bay as potential drivers of epibenthic diversity and distribution. Although the study was limited in spatial extent by the number of stations sampled (13), some factors measured were found to contribute to the biological variation observed. Jointly, the long-term means of the abiotic factors examined accounted for almost 55% of the fitted variation while the coefficient of variation data for these factors accounted for as much as 68.3% of the fitted variation. Key abiotic factors (accounting for > 10% variation) included the mean grain size, depth, modelled bottom temperature and modelled current speed, while in terms of long-term variability (CoV) the key contributory factors were modelled current temperature, modelled current speed and dissolved oxygen.

The interpolated mean grain size layer for Algoa Bay's unconsolidated sediments was a key feature identified as a potential surrogate for epibenthic species distributions in the Bay. Using individual abiotic factors does not provide a complete understanding of the habitats observed, however with limited resources available for habitat mapping this factor can still be used on its own to provide some indication of the benthic habitats with less effort than most other sampling methods. This study provides a foundation for developing fine-scale habitat maps for unconsolidated sediment and has the potential to be implemented in similar habitats along South Africa's coast.

Chapter 4: Protection status of benthic biodiversity in Algoa Bay with recommendations for future management

Introduction

Over the past two decades there has been a shift in global attention towards the value of the ocean and the quality of its present state (Millennium Ecosystem Assessment, 2005; Bennett *et al.*, 2009; Carollo *et al.*, 2009; Bermas-Atrigenio and Chua, 2013; Kirkman *et al.*, 2016; Buhl-Mortensen *et al.*, 2017). Much of the ocean and its ecosystems are considered to be in a degraded state caused by the increasing demand for access and harvesting of marine resources (Rondinini and Chiozza, 2010; Boon and Beger, 2016). Degradation of marine ecosystems have generated raised global concern as the ocean plays a fundamental role in supporting economies and human well-being (Costanza *et al.*, 1997; Duarte, 2000; Balmford *et al.*, 2004; Worm *et al.*, 2006; Douvere, 2008; Palumbi *et al.*, 2009; Foley *et al.*, 2010; Domínguez-Tejo *et al.*, 2016). It is widely recognised that to ensure the sustainable use of marine resources and protection of vulnerable habitats, the entire ecosystem (including humans) should be considered when managing our oceans (Agardy *et al.*, 2003; Garcia *et al.*, 2003; Garcia and Cochrane, 2005; Backer *et al.*, 2010; Agardy *et al.*, 2011; Altman *et al.*, 2014). Ecosystem-Based Management (EBM) seeks to maintain ecosystems in a healthy, productive and robust condition to enable long-term provision of services supporting human well-being (Buhl-Mortensen *et al.*, 2015a; b; Kirkman *et al.*, 2016; Rodriguez, 2017). Implementation of an EBM approach requires a comprehensive understanding of both the marine environment and the cumulative human impacts at various spatial and temporal scales (Halpern, 2009; Ban *et al.*, 2010; Halpern *et al.*, 2012, 2015; Korpinen *et al.*, 2013; Jacobsen *et al.*, 2014; Kirkman *et al.*, 2016; Korpinen and Andersen, 2016). An ecosystem-based approach to management should form an integral part of the Marine Spatial Planning (MSP) process, particularly during the delineation and implementation of Marine Protected Areas (MPAs).

The increasing demand for access to and use of ocean space has left few suitable areas available for conservation (Rondinini and Chiozza, 2010). Consequently, areas identified for protection need to prioritize for the greatest ecological gains at the lowest cost to human activities (Naidoo *et al.*, 2006; Carwardine *et al.*, 2008; Rondinini and Chiozza, 2010; Agardy *et al.*, 2011; Chan *et al.*, 2011). The Convention on Biological Diversity (CBD) introduced biodiversity targets with the goal to increase marine habitat protection globally to greater than 10% for all marine ecosystems by 2020 (United Nations CBD, 1992; O'Leary *et al*., 2016). Biodiversity targets aim to ensure a portion of each ecosystem can remain intact and continue its ecological function should the majority of said ecosystem be impacted by anthropogenic activities and global change (Mazor *et al.,* 2014; Schmiing *et al.,* 2015). Achieving biodiversity targets requires a balance between protecting sufficient ecosystem area and the sustainable management of utilisation of ocean resources. (Huggett, 2005; Mcleod *et al.*, 2009; Klein *et al.*, 2013; Hameed *et al.,* 2017). Targets can be driven by either policy (United Nations CBD, 1992; Sink *et al.*, 2012a) or data, or both, depending on the amount of information available for the area or region in question (Agardy *et al.*, 2003; Svancara *et al*., 2005). Identifying where protection should be implemented such as vulnerable areas that are considered low cost to local anthropogenic pressures, less threatened or already degraded, will improve habitat recovery efforts. Furthermore, determining the appropriate scale of protection remains a challenge as it can be dependent on the classification of the protected area such as habitat types, ecosystem types and biozones. (Stewart and Possingham, 2005; Tear *et al.*, 2005; Campbell *et al.*, 2009; Schmiing *et al.,* 2015; Boon and Beger, 2016). Furthermore, there is need to establish how the benefits or effects of the protection will be measured and for how long, allowing for adaptive management (Tear *et al.*, 2005; Wood, 2011).

The frequent overlap of conservation objectives with other ocean uses and the inevitable conflicts that arise when trying to achieve a balance between the two, has resulted in the use of complex spatial analysis tools such as Marxan (Ball *et al*., 2009; Segan *et al.,* 2011) to provide support for the decisions made (Henriques *et al*., 2017). Marxan is globally the most widely used software tool for developing conservation plans and protected area networks (Smith *et al.,* 2008; Watts *et al.,* 2017). The goal of such decision support software is to present options for reserve design by selecting areas that represent the assigned biodiversity targets with the lowest possible socio-economic cost (Game and Grantham, 2008). Areas with lower socio-economic cost are sometimes also potentially areas in better ecological condition due to the likelihood of lower cumulative impact from human activities in these areas. However, this is not always the case because frequently the area supporting the resource (and hence higher socio-economic activity), is also one of high ecological importance and value. Aside from incorporating the biological and human pressure data, stakeholder engagement and participation also plays a vital role in successful MPA design by providing important knowledge of the area in question (Smith *et al*., 2008). This information can then be incorporated into the decision support tool for improved reserve selection.

South Africa's current marine protection includes 23 coastal MPAs and, more recently, one offshore MPA (Prince Edward Islands, declared in 2013) declared in terms of the Marine Living Resources Act (Marine Resources Act, 1998) and managed by various conservation agencies (CapeNature, South African National Parks, Eastern Cape Parks and Tourism Agency, Kwazulu Natal Wildlife, Nelson Mandela Bay Metropol and City of Cape Town). Designated MPAs in South Africa have slowly increased in number from 1964 when the first MPA (Tsitsikamma) was declared through to 2013 with the most recent MPA (Prince Edward Islands). Much of this growth in South Africa's MPAs has been related to the increase in national focus on marine protection and the improved global understanding of anthropogenic pressures on the marine environment. Declaring South Africa's MPAs was no simple task, particularly in terms of identifying suitable, agreeable areas. Furthermore, even once MPAs are designated, there may still be continuous stakeholder conflicts, as recently occurred with South Africa's oldest MPA, Tsitsikamma MPA (Chadwick *et al.*, 2014; Sink, 2016; Sowman and Sunde, 2018). A network of 22 offshore MPAs has been proposed through the Operation Phakisa Oceans Economy initiative (Department of Environmental Affairs, 2014b). These proposed MPAs aim to advance the current 0.4% protection of South Africa's Exclusive Economic Zone (EEZ) to 5%, somewhat closing the gap towards the 10% EEZ protection target set by the Convention on Biodiversity (United Nations CBD, 1992) and the target of 20% EEZ protection for offshore areas, set by the National Biodiversity Strategy and Action Plan (NBSAP, DEAT, 2008). The proposed offshore MPAs also aim to protect a representative 20% of each habitat type identified by the National Biodiversity Assessment (NBA, Sink *et al.*, 2012a) including important ecosystem features that are not known to occur anywhere else in South Africa's EEZ.

One of the proposed MPAs is that of Addo Marine Protected Area which lies within the study area of Algoa Bay (Fig. 4.1). The purpose of declaring this MPA is to contribute to a national and global representation of MPAs by providing protection for species, habitats and ecosystem processes in a biodiversity hotspot, to form a contiguous conservation area between marine, estuarine and terrestrial habitats (National Environmental Management: Protected Areas Act draft notice, 2016). This proposed MPA would encompass the existing Bird Island MPA, and would be zoned with areas of both restricted and controlled access. Restricted areas are considered 'no-take' areas where extraction and harvesting of all marine and plant life is prohibited, while controlled areas are considered 'open' where extraction and harvesting is dependent on issue of a permit and is limited to the following activities: spear fishing, angling, scuba diving snorkelling for mollusc extraction, boating, commercial dive salvage operations, commercial fishing, whale watching, shark cage diving or filming (per the Marine Living Resources Act, 1998). Algoa Bay has also been identified as a case study for the first South African Marine Area Plan (Dorrington *et al.*, 2018), in response to the recently published Marine Spatial Planning Bill (MSP Bill, DEA, 2017). The Marine Area Plan aims to achieve ecological, economic and social objectives, considering all relevant principles and factors set out in the MSP Bill. This includes sustainable use, growth and management of the ocean and its resources while minimising negative financial, social, economic or environmental impacts. Algoa Bay is considered appropriate for implementation of a Marine Area Plan (called the Algoa Bay Project) owing to the complex scales of governance, human use and biophysical environments within the area (Dorrington *et al.*, 2018). The Algoa Bay Project is currently in the first phase of development which includes the use of systematic biodiversity planning methods. The Project is based on three pillars centred on the aims identified by the MSP Bill, namely a bioregional plan (biodiversity focused), a governance framework and a socio-economic plan (Dorrington *et al.*, 2018). Previous Chapters in this study revealed that the benthic habitats in defined by the 2012 NBA for Algoa Bay do not align with the observed patterns in the unconsolidated epibenthic communities. Despite this mismatch, it is nonetheless useful to test the suitability of the existing and proposed MPAs using the best available broad-scale NBA-defined information. This chapter will assess competing options for meeting the 20% protection targets that are not included in the current proposed MPA for Algoa Bay, by identifying alternative spatial designs.

This study aims to assess the proportion of 2012 NBA-defined habitats, and of newly defined epibenthic communities are represented within the existing and proposed MPAs in Algoa Bay. In conjunction with these assessments, this study further aims to map and incorporate known anthropogenic activities directly associated with unconsolidated sediment habitats into a decision analysis (using Marxan) of alternative MPA designs scenarios.

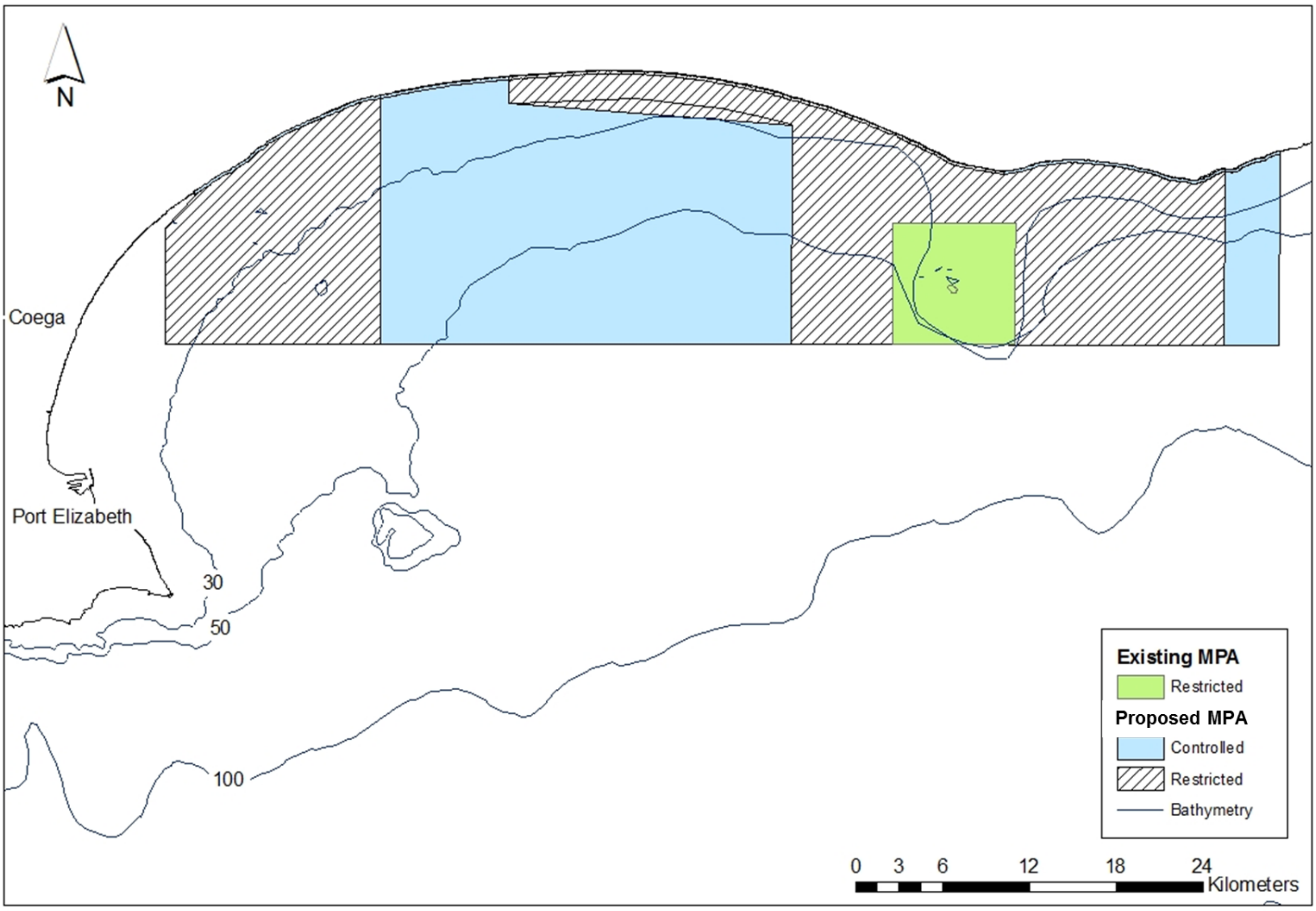

Figure 4.1: Existing and proposed Marine Protected Areas (MPAs), with restricted areas indicated in green (existing) and hatched shading(proposed), and controlled areas shaded in blue, in Algoa Bay.

Methods

Proposed MPA representation

To determine the area of only inshore and inner shelf NBA habitat (Sink *et al.,* 2012a) types within the study area, *Agulhas Hard Inner Shelf, Agulhas Inner Shelf Reef, Agulhas Inshore Reef, Agulhas Island, Agulhas Mixed Sediment Inner Shelf, Agulhas Sandy Inner Shelf, Agulhas Sandy Inshore*, a 1x1km planning unit (PU) was created using ArcGIS (ESRI). The NBA benthic habitat type map and the GIS shapefile for existing and proposed Algoa Bay MPAs were combined with the PU using the union function from the geoprocessing toolbox to create a single shapefile. This method conserves each layer's attributes, including the area of each planning unit in each habitat type, and the related protection status.

The total area for each NBA defined habitat and its protection status was calculated from the attribute table. Using the position of the 13 stations sampled in Algoa Bay as part of this study, and the protection status for each of the two previously defined epibenthic community types (Chapter Two) were calculated.

Pressure Map

The planning unit layer was overlaid on maps of anthropogenic pressures (layers) within Algoa Bay. The pressure layers were gathered from different sources and were measured at different scales. Sources included the NBA pressure layers (National scale, Sink *et al.,* 2012a), Offshore Marine Protected Area project (OMPA- National scale, Sink *et al.,* 2011) and local municipality information. Only those pressures with direct impacts on the benthic habitat were selected. The pressure data were all collected in different units, therefore for comparison, the data were normalised (values were scaled from 0- low impact to 1- high impact) and then summed per PU. These pressures included:

- Dredge dumping from Coega and Port Elizabeth harbours

  The deposition of sediment as a result from dredging the harbours in Algoa Bay will affect the in- and epifauna. New sediment deposited into the area increases the spatial extent of such habitat type and may extend the range of species exclusive to the harbour. The deposition of sediment may lead to an increase in turbidity and the likely burial of sedentary species causing possible interference in the animal's life history stages (Essink, 1999; Angonesi *et al*., 2006; Smith *et al.,* 2008). The harbour sediment may also contain buried contaminants such as heavy metals that could be exposed by dredging and further distributed into the Bay (Naser, 2013). The data provided the two ports in Algoa Bay, Port of Port Elizabeth and Port of Ngqura (Coega) only indicated the area of the allocated dumping sites for both harbours. Should these areas fall within a PU they were allocated a pressure value of 1.

- Waste water outfalls

  The discharge of effluent water into the ocean can alter the pH of the water and may cause eutrophication or harmful algal blooms as a result of the increase in nutrients in the water (Bellan *et al.*, 1999; Gucker *et al.*, 2006). Point co-ordinates of the discharge areas were provided by the Nelson Mandela Metropol municipality and the impact was determined using the PU and their distance from the source in 0.25 increments.

- Shipping anchor scour (CSIR)

  A ships anchor and chain can shift across the seabed causing damage to benthic habitats (Davis et al., 2016). Although this can be considered a localised impact, this repetitive disturbance can result in large areas being impacted. The original data was provided by CSIR for shipping time at anchor within Algoa Bay for one month.

- Historical trawling

  Historical trawl fishing was permitted to take place in bays and shallower than 30m depth. Permit restrictions were introduced in 1978 after the establishment of South Africa's EEZ preventing offshore trawling vessels from fishing shallower than the 110m depth contour (Attwood *et al.*, 2011; Sink *et al.*, 2012b). Parts of Algoa Bay were closed to trawling as early as 1935 owing to concerns of other inshore fisheries sectors (Sink *et al.,* 2012b). These areas can therefore be considered transformed owing to previous resource exploitation.

- Inshore trawling (OMPA & NBA)

  The current inshore trawl grounds lie beyond the 50m depth contour (i.e outside of Algoa Bay) however, they overlap the physical and biological processes that influence ecosystems within the Bay. Seabed trawling disturbs benthic ecosystems and can transform habitats by the removal of emergent fauna (Thrush *et al.*, 2006; Atkinson *et al.*, 2011). Inshore trawling pressure was measured by the number of trawling tracks per PU for the OMPA values and the NBA pressure values reflected the relative fishing effort for the inshore demersal trawling sector.

- Petroleum exploration (OMPA)
  Seismic surveys, the use of an array of airguns to generate an acoustic signal in order to find oil and gas deposits, are conducted in large petroleum lease areas and are known to negatively affect marine mammals and possibly fish and other species within the vicinity (Gordon *et al.*, 2003; Dunlop *et al.*, 2017) Exploratory well drilling, an intrusive method of determining the feasibility of drilling for economic development, also impacts benthic communities. The majority of Algoa Bay is currently part of a petroleum lease and pressure values were allocated as a combination of the lease and known oil well heads explained below.
- Oil well heads (NBA, 2012)
  The drilling of oil wells is highly intrusive and can have long term effects including the possibility of oil spills. The active drilling of well heads can cause increased sediment deposition that may result in the smothering of emergent fauna and may lead to the possible contamination of sediment from drilling muds exposed during exploration (Gray *et al.,* 1990; Kingston, 1992; Olsgard and Gray, 1995; Cranford *et al.,* 1999; Grant and Briggs, 2002). The pressure values were allocated as a combination of the known oil well heads and petroleum lease explained below.
- Shipping paths (NBA, 2012)
  Algoa Bay has two commercial ports resulting in high shipping traffic throughout the Bay. Depending on the size of the ship and the water depth, the ships movement has the ability to disrupt sediment, disturbing life on the seabed (Abdulla and Linden, 2008). Ships are also known to be vectors for introducing alien invasive species (hull fouling and ballast water) and deposition of pollution (Aronson *et al.*, 2011). Shipping pressure indicated the normalised density of vessel tracks at a national scale.
- Mariculture (NBA, 2012)
  The NBA 2012 identified the possible declines in water quality and associated pollution impacts caused from chemicals used in mariculture practices. Localised habitat alteration and the potential introduction of alien invasive species can also affect benthic habitats. Pressure values indicate the presence or absence of maricultural activities.

- Squid fisheries (NBA, 2012)
  The squid fishery has a low impact on the seabed, however there may be potential indirect impacts on benthic habitats caused by the use of bright lights at night and the possible trophic impacts on squid predator populations such as cetaceans and seabirds. Scaled pressure values that reflect the relative fishing effort.
- South Coast Rock Lobster fisheries (NBA, 2012)
  Biodiversity considerations for this fishery include the vulnerability of the species stock, incidental bycatch, loss of traps with potential associated ghost fishing and localised damage to the seabed (damage to epibenthic species) (Japp, 2004). Scaled pressure values that reflect the relative fishing effort.

Selected pressures were adjusted according to the following rules before each PU total pressure was calculated:

- The NBA inshore trawling pressure layer was excluded as it displayed catch per unit effort at a national scale. The OMPA inshore trawling pressure layer was used instead as it measured the number of tracks per planning unit at a finer spatial scale (see Fig. 4.4 f & h).
- The OMPA petroleum lease and NBA oil well heads pressures were combined into one pressure by adding each normalised value together and weighting the two values 0.5:1 respectively. The OMPA petroleum lease was given a lower weighting (0.5) as it did not identify any active oil well heads, only the broader area that could be explored (Fig. 4.4 b & d).
- Historical trawling grounds (Fig. 4.4e) were weighted 0.5 as these areas have not been trawled since the proclamation of the Marine Living Resources Act (1998).

The summed values, per grid cell, were used as a surrogate for direct, cumulative benthic pressures in Algoa Bay (Fig. 4.5).

Alternative MPA proposed

Marxan (Ball *et al.*, 2009; Segan *et al.,* 2011) input data files were created in ArcGIS (ESRI) using attribute tables from the 10 input layers (Table 4.1).

Table 4.1: Pressure data and its source used during Marxan analysis.

Data type	**Data source**
Dredge dumping	Port of Port Elizabeth
Waste water outfalls	Nelson Mandela Bay Metropol
Shipping anchorage	CSIR
Historical trawling grounds	Scott grounds, Sink *et al.,* 2012b
Inshore trawling	OMPA
Petroleum exploration	OMPA & NBA
Shipping intensity	NBA
Mariculture	NBA
Squid fisheries	NBA
South coast rock lobster fisheries	NBA

Information additionally provided into the Marxan software included the 2012 NBA habitat types within the study area (*spec.txt*), the area of each habitat type per planning units (*puspr.txt*), the cost and protection status of each planning unit (*pu.txt*) and the boundary length of each planning unit (*bound.txt*). The input file (*input.dat*) file was created following the recommendations of the Marxan user manual (Watts *et al*., 2009).

Two scenarios were developed as an alternative to the current proposed MPA using Zonae Cogito, the user interface programme to Marxan (Segan *et al*., 2011). Scenario 1 took into account the existing MPA while adding additional areas for protection to include 20% representation of each habitat type. Scenario 2 considered both the existing and proposed MPA with further areas for protection added to represent missing habitat type targets (20% target per habitat type).

Marxan clusters sites to reduce the boundary length for improved area selection, however, this may increase the overall cost of the reserve network. Site selection can be manipulated by altering the boundary length modifier (BLM) to maximize reserve clustering for the lowest cost. Each scenario included a calibration run of the analysis to determine the correct BLM and determine if all targets were met for each habitat type. Species Penalty Factors (SPF) were added to habitat types where the target fell short of the 20% protection target, specifically for habitat types Agulhas Inner Shelf Reef and Agulhas Inshore Reef in this study.

Results

The present protection status of NBA-defined habitat types in Algoa Bay includes only one habitat type (Agulhas Island) with no protection to any of the seven unconsolidated habitats listed. The proposed MPA protection represents 20% or more of all but three habitats. Marxan analysis concluded that in order to further protect the three additional habitat types, the proposed MPA should have an additional 6 small areas added to include missing habitat types. An alternative scenario proposes 6 small protected areas including the existing MPA that requires less area than the proposed MPA, however at the cost of connectivity.

Proposed MPA representation

The existing Bird Island MPA includes only one (Agulhas Island) of the seven NBA-defined habitat types identified in Algoa Bay. The proposed MPA represents all but three NBA-defined habitat types with 20% or more being protected (Fig. 4.2). It should be noted that the areas of each habitat type in Algoa Bay differ extensively, and two of the three habitats not represented within the proposed MPA (Agulhas Inshore Reef and Agulhas Inner Shelf Reef) have the smallest total areas (Table 4.2).

Table 4.2: Total area (m^2) per NBA-defined habitat type and the respective protection status within Algoa Bay.

Habitat type	**Existing Restricted (m^2)**	**Unprotected (m^2)**	**Proposed Controlled (m^2)**	**Proposed Restricted (m^2)**	**Grand Total (m^2)**
Agulhas Hard Inner Shelf		167,806,313	35,718,378		203,524,691
Agulhas Inner Shelf Reef		4,398,228.			4,398,228
Agulhas Inshore Reef		5,222,564			5,222,564
Agulhas Island	70,584,161	144,660,864	850,681	315748829	531,844,537
Agulhas Mixed Sediment Inner Shelf		94,542,123	36,853,053	5265567	136,660,744
Agulhas Sandy Inner Shelf		803,361,150	325,855,887	40110885	1,169,327,924
Agulhas Sandy Inshore		72,846,616	63,443,100	61228456	197,518,174

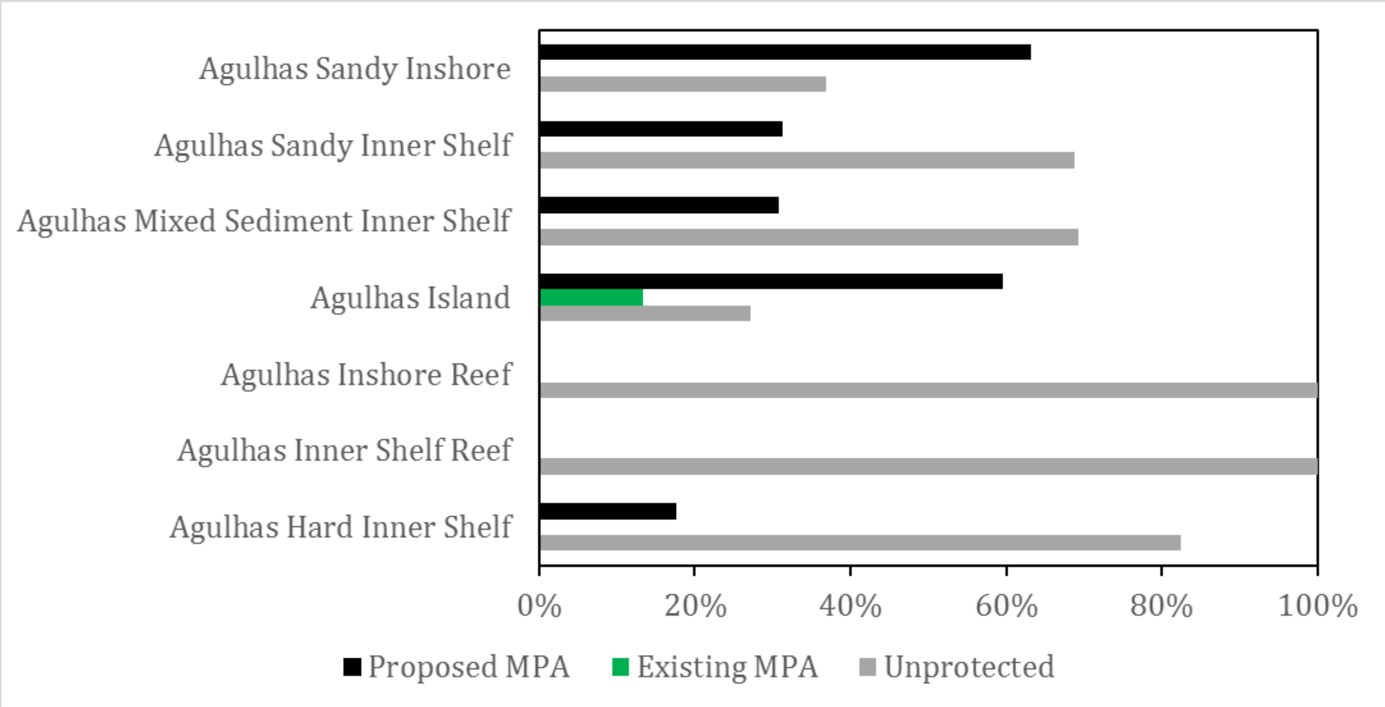

Figure 4.2: Percentage protection status per NBA-defined habitat type within Algoa Bay.

Analyses conducted in previous Chapters of this study characterised epibenthic species in unconsolidated habitats into two community types A and B. The spatial occurrence of these community types in proposed protected areas is shown in Figure 4.3. No station sampled in this study occurred within the existing MPA, but stations identified as community A and B were both well represented in the proposed MPA (56% & 100% of stations respectively) with only 30% of stations not included in the proposed MPA (Fig. 4.3).

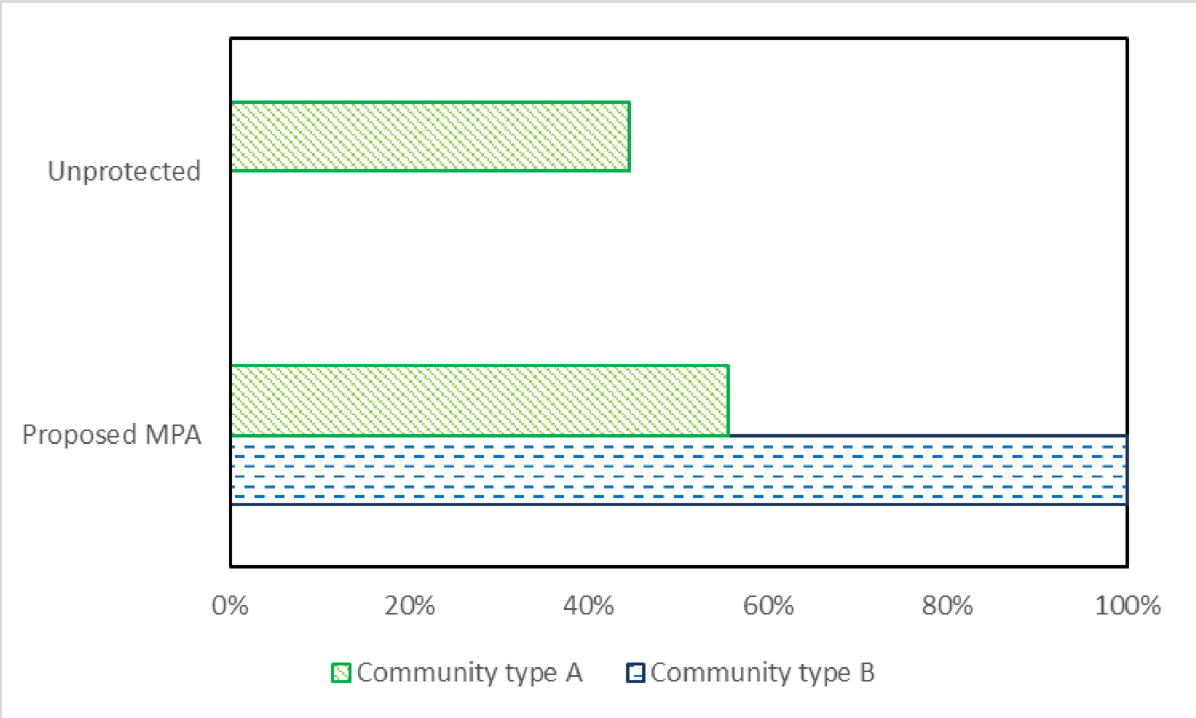

Figure 4.3: Protection status of stations surveyed representing two community types identified from epibenthic communities observed in Chapter Two.

Pressure Map

Individual anthropogenic pressures appear to vary in moderation across the Bay. Dredge dumping, waste water outfalls and mariculture pressures are situated towards the western side of the Bay, nearest the two main ports, Port Elizabeth and Coega (Fig. 4.4a, c & g). Cummulative pressures in Algoa Bay indicate a higher overall pressure in the western portion of the Bay (Fig. 4.5). There is also a small area towards the eastern portion of the Bay (offshore of Bird Island) that has greater anthropogenic pressures, possibly as a result of the high intensity of individual shipping pressures, South coast rock lobster and squid fisheriesin this area (Fig. 4.4i, j & l).

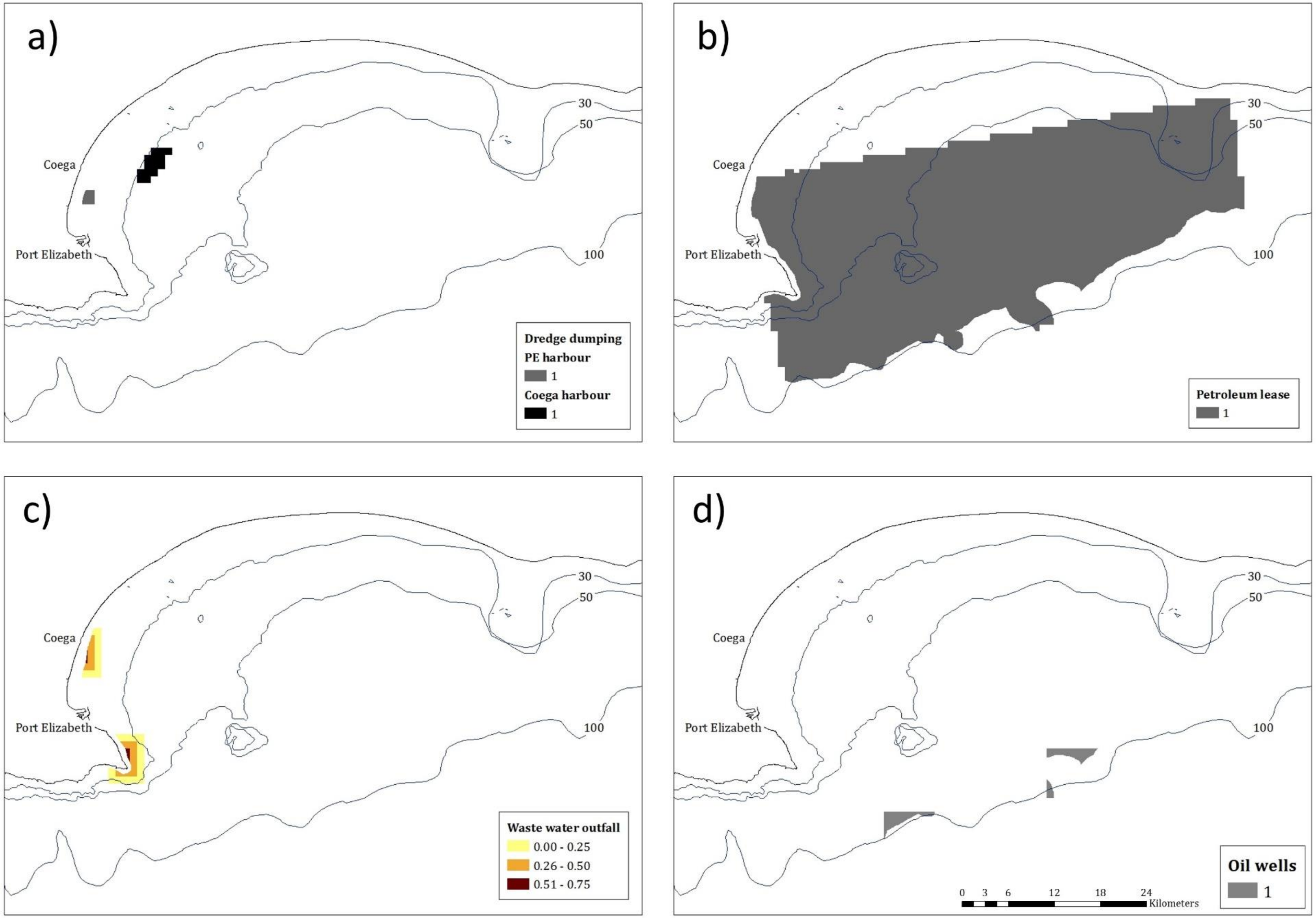

Figure 4.4: Individual anthropogenic pressures within Algoa Bay: a.) Dredge dumping, b.) petroleum lease, c.) waste water outfalls and d.) oil wells.

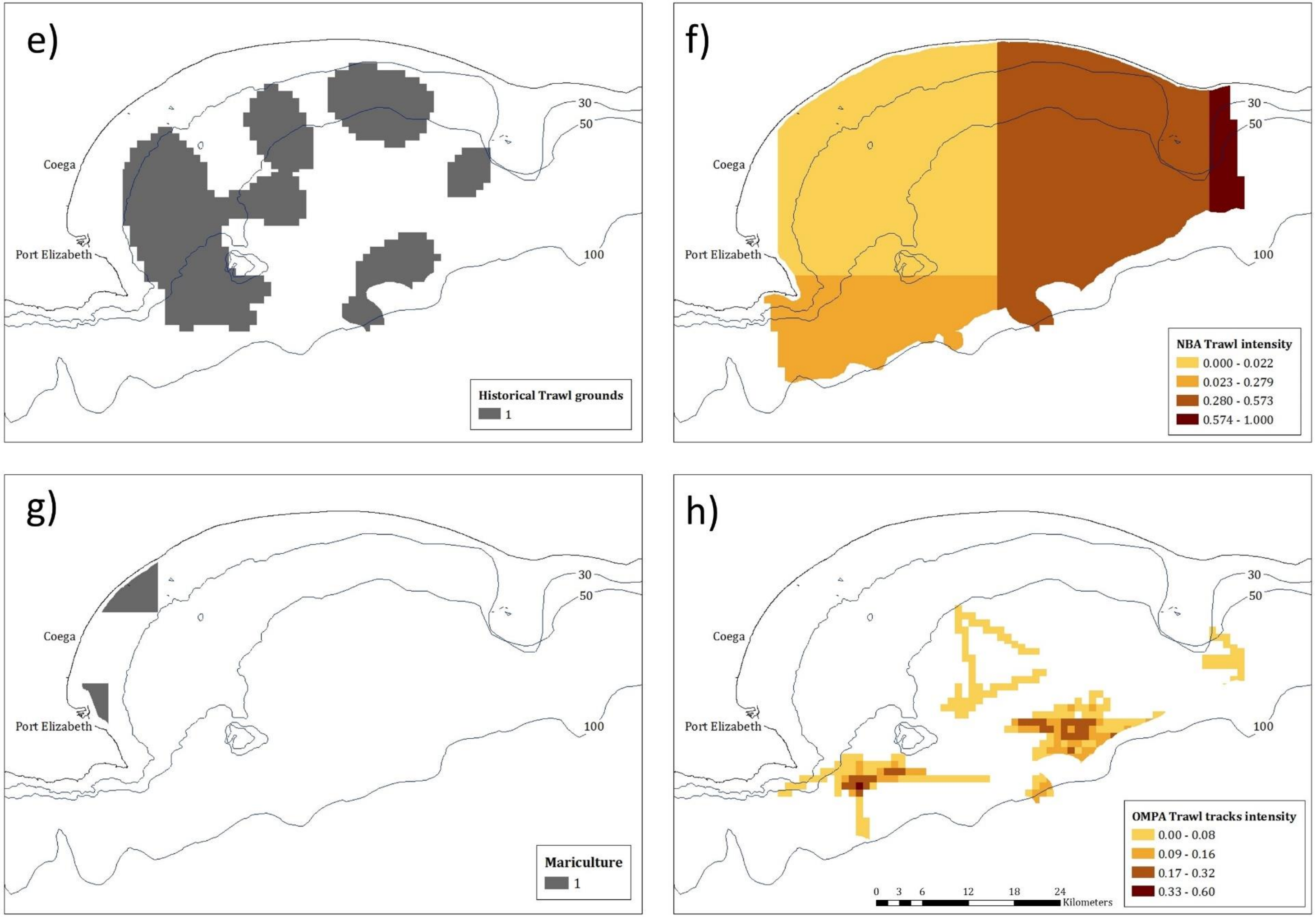

Figure 4.4 continued: e.) Historical trawling grounds, f.) NBA Trawling intensity, g.) Maricultural activities and h.) OMPA inshore trawling track intensity.

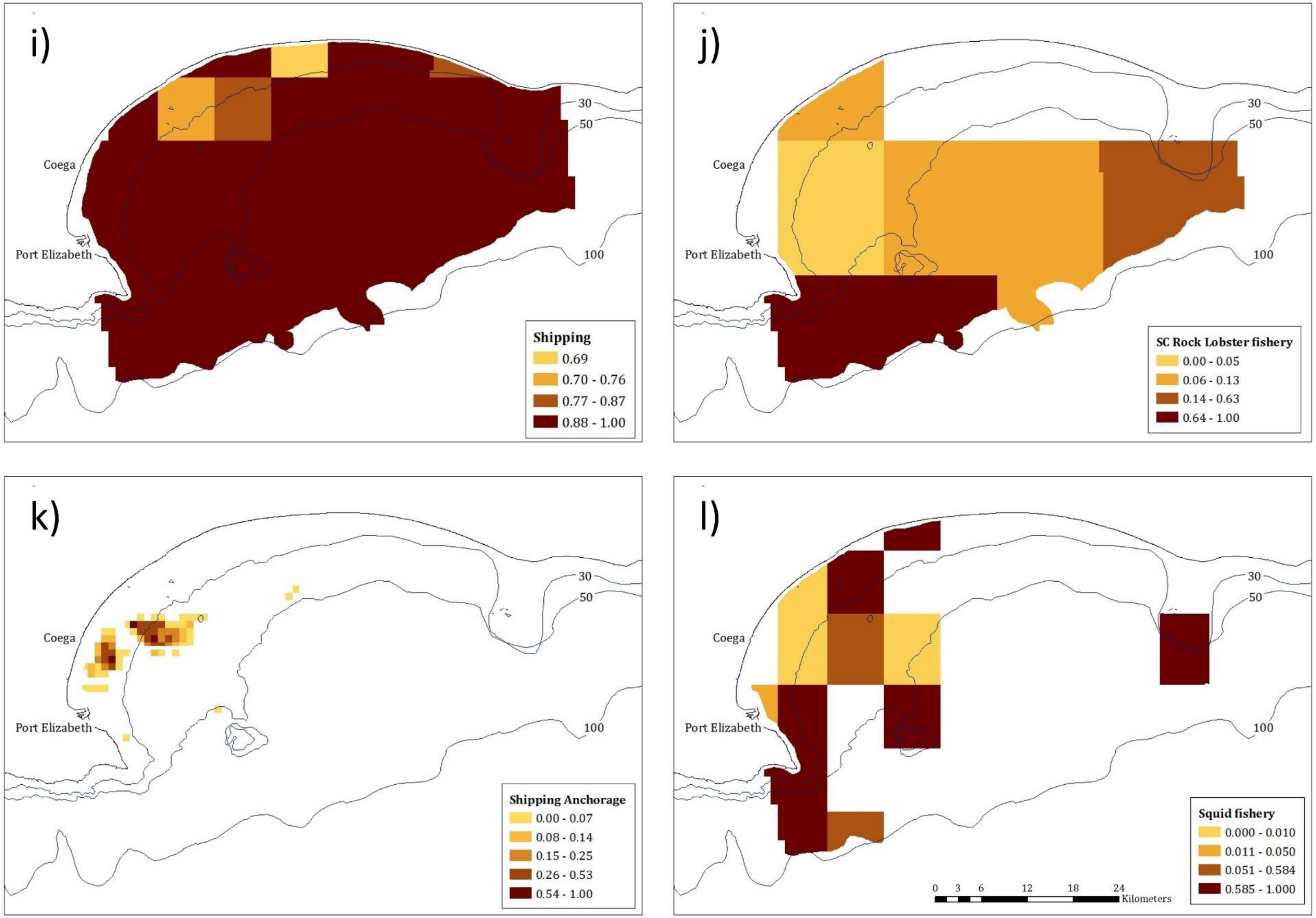

Figure 4.4 continued: i.) Shipping intensity, j.) South Coast rock lobster fishery, k.) shipping anchorage intensity and l.) squid fishery.

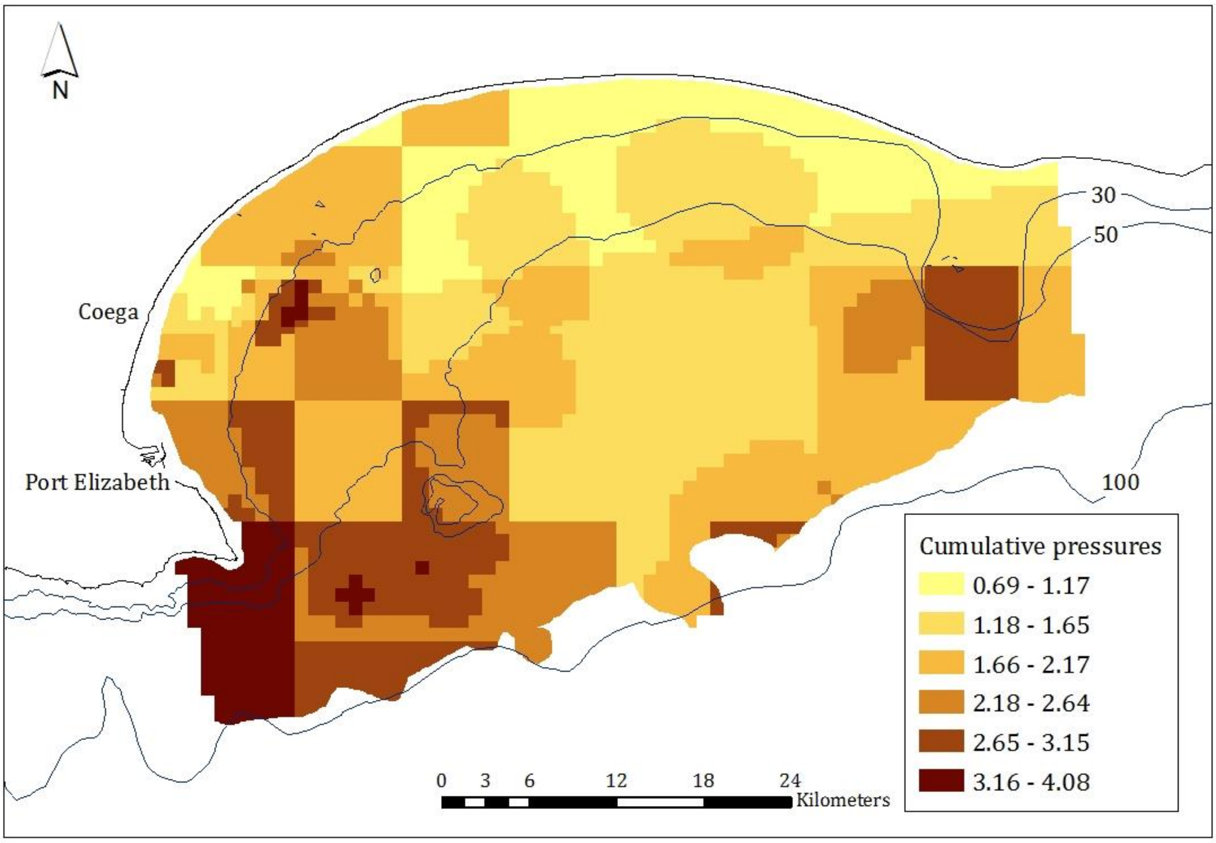

Figure 4.5: Cumulative anthropogenic pressures directly linked to benthic habitats in Algoa Bay.

Alternative proposed MPA

Marxan generated a best solution for Scenario 1 (Existing MPA with additional area selected to meet target) and 2 (Existing and proposed MPA with additional area selected to meet target) that BEST met biodiversity targets at the lowest cost value (in this case, cumulative pressure value). The scenario 1 best solution from Marxan produced a reserve network of six areas including the existing Bird Island MPA (Fig. 4.6). The best solution report (Table 4.3) indicates that the target of 20% per NBA-defined habitat type under protection was met with Scenario 1 (with Species Penalty Factors of 2 & 3 for Agulhas Inshore Reef and Agulhas Inner Shelf Reef respectively). Total area protected in scenario 1 was 447,236,536m^2 out of a total 2,248,496,864m^2 for the study area (19.9%).

Table 4.3: Best solution report from Scenario 1 run in Zonae Cogito indicating the area selected (amount held) as a possible marine protected area.

Conservation feature	**Target (m^2)**	**Amount held (m^2)**	**Target met (%)**
Agulhas Sandy Inshore	39,503,634	39,911,436	101
Agulhas Sandy Inner Shelf	233,865,584	234,180,115	100
Agulhas Mixed Sediment Inner Shelf	27,332,148	27,853,043	104
Agulhas Island	104,731,134	105,526,147	100
Agulhas Inshore Reef	1,044,512	1,182,752	113
Agulhas Inner Shelf Reef	879,645	896,851	101
Agulhas Hard Inner Shelf	40,704,938	40,923,032	100

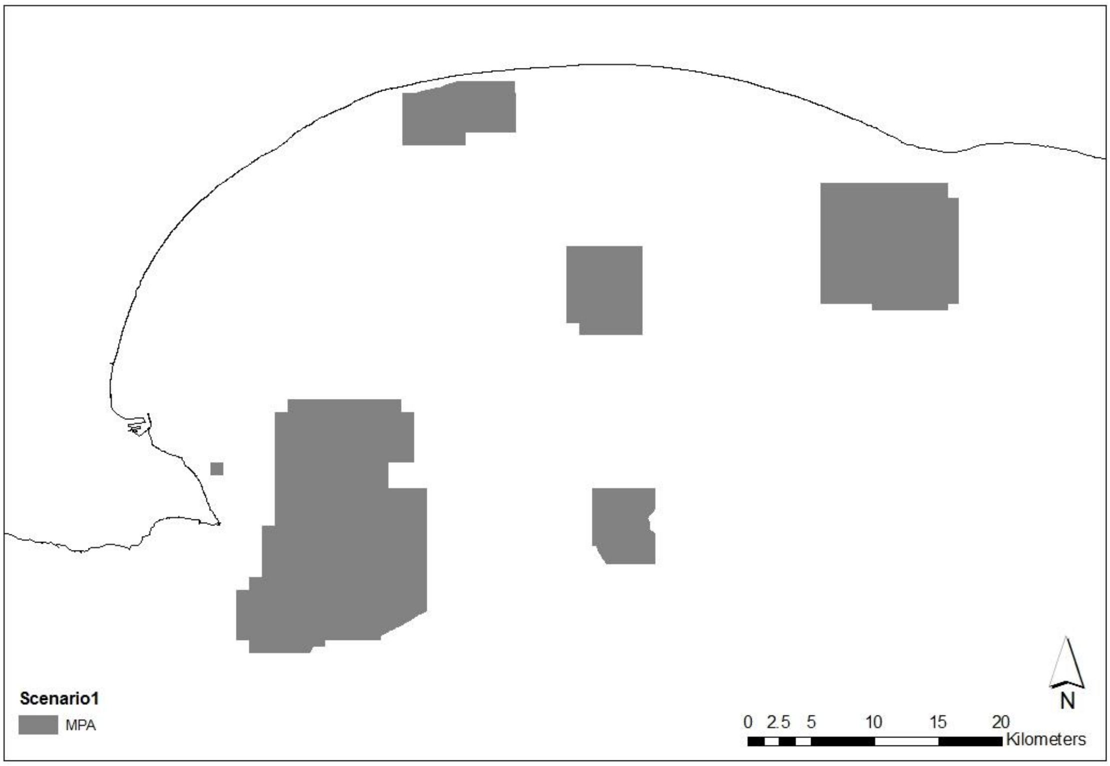

Figure 4.6: Scenario 1 best solution for an MPA network in Algoa Bay. This includes the existing Bird Island MPA (the eastern-most grey area) and additional areas to meet the 20% target for each NBA-defined habitat type.

The scenario 2 best solution retained the full extent of the proposed MPA with several smaller, additional planning units selected to meet the 20% targets for the NBA-defined habitats (Fig. 4.7). The best solution report (Table 4.4) for scenario 2 indicates that all targets were met except for Agulhas Inshore Reef which reached 97% of the area targeted (with Species Penalty Factors of 2 and 3 for Agulhas Inshore Reef and Agulhas Inner Shelf Reef respectively to best meet the target set). When comparing the total target area to the amount included in the proposed MPA in scenario 2, four of the seven conservation features (habitat types) surpassed the target by a large margin. The area of Agulhas Sandy Inner Shelf protected surpassed its target area by 56%, Agulhas Sandy Inshore by 215%, Agulhas Mixed Sediment Inner Shelf by 54%, Agulhas Island by 269%, and Agulhas Inner Shelf Reef by 15%. The total area protected in scenario 2 was 962,659,002 m^2 of 2,248,496,864m^2 of the study area (42.8%). Although scenario 2 percentage area was higher than that of scenario 1, it does not provide protection for the full 20% of Agulhas Inshore Reef habitat type.

Table 4.4: Best solution report from Scenario 2 run in Zonae Cogito indicating the area selected (amount held) as a possible marine protected area.

Conservation feature	**Target (m^2)**	**Amount held (m^2)**	**Target met (%)**
Agulhas Sandy Inshore	39,503,634	124,817,090	315
Agulhas Sandy Inner Shelf	233,865,584	366,789,976	156
Agulhas Mixed Sediment Inner Shelf	27,332,148	42,118,621	154
Agulhas Island	104,731,134	387,183,672	369
Agulhas Inshore Reef	1,044,512	1,017,219	**97**
Agulhas Inner Shelf Reef	879,645	1,014,044	115
Agulhas Hard Inner Shelf	40,704,938	40,718,978	100

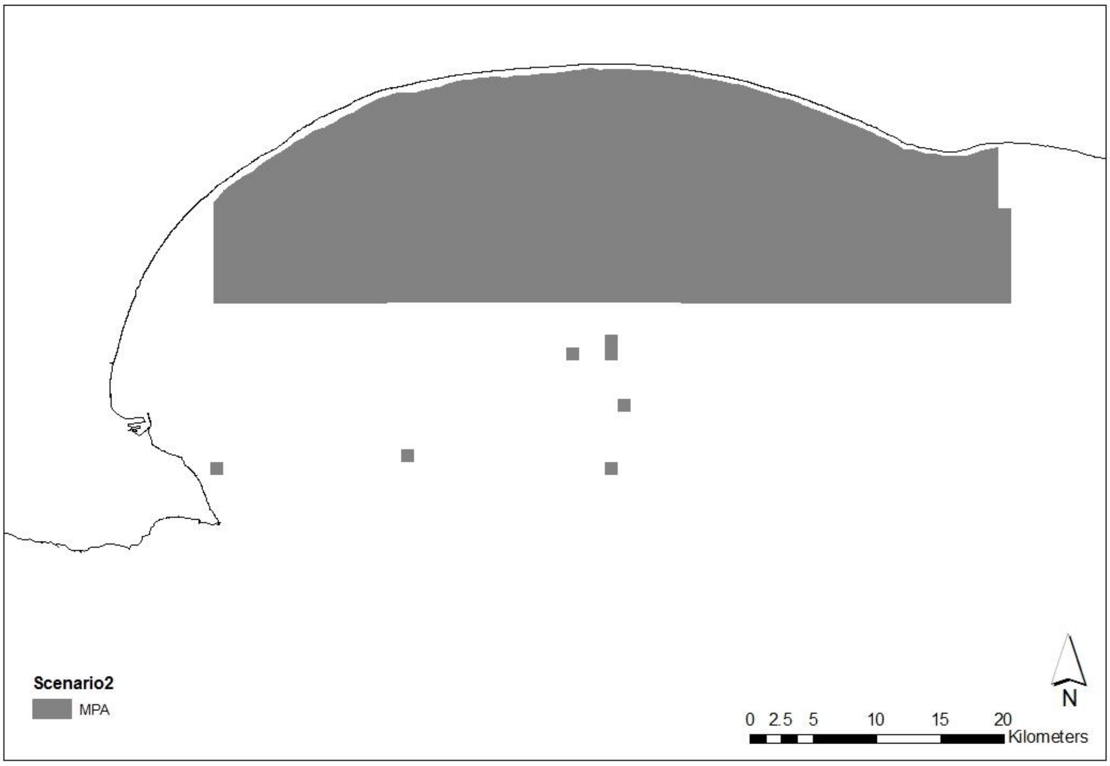

Figure 4.7: Scenario 2 best solution for an MPA network in Algoa Bay based on inclusion of both the existing and proposed MPA and additional areas to meet the 20% target for each NBA-defined habitat type.

Discussion

The existing MPA in Algoa Bay (Bird Island MPA) encompasses only one of the seven offshore unconsolidated NBA-defined habitats in Algoa Bay, namely Agulhas Island. The remaining six offshore habitat types identified in the Bay are currently not afforded any protection. The proposed, extended Addo MPA (Sink *et al*., 2011) rectifies this by increasing the number of habitats protected from one to five of the seven with 20% or more of the five habitats being afforded protection, if proclaimed. However, the proposed Addo MPA was planned as part of a national scale implementation and, not all offshore habitat types are represented, as those habitat types may be represented in existing or proposed MPA elsewhere.

The alternative proposed MPAs based on existing (scenario 1) and proposed (scenario 2) MPAs provide insight into a potential new reserve network for Algoa Bay. Although the current proposed MPA includes more area than required to obtain habitat protection targets for several habitats in the Bay, but not all seven. Scenario 1 could therefore present a better alternative, requiring less area to be set aside to reach set habitat targets and including all seven habitat types assessed. However, scenario 1 includes six smaller, separate protected areas (one of which is the existing Bird Island MPA) which could in turn pose a challenge in the connectivity and management costs of protected areas in Algoa Bay's benthic habitats. Connectivity is an essential process supporting the persistence, recovery and productivity of marine ecosystems (Lagabrielle *et al*., 2014). A combination of habitats should be included adjacent to each other to accommodate movement of mobile organisms and the potential shifts in distributions caused by seasonal fluctuations (Kendall *et al*., 2008) The largest of these potential six protected areas lies in an area with the highest cumulative pressures (nearest Port Elizabeth) and may lead to management and enforcement challenges owing to the high number of stakeholders (monitoring of activities both commercial and recreational with limited resources). Scenario 2 includes the whole proposed Addo MPA with seven additional individual planning units to reach the 20% targets for Agulhas Inshore Reef and Agulhas Inner Shelf Reef habitat types. These areas reflect a lower cumulative pressure value and the larger area of the proposed Addo MPA provides the necessary connectivity. The two scenarios presented in this study provide alternative considerations to the current proposed Addo MPA, however each option has advantages, scenario 1 utilising less area while scenario 2 conserves connectivity at lower pressure cost, and limitations, scenario 1 having less connectivity and in high pressure areas that increase management costs while scenario 2 utilises a larger area, that should be considered on implementing additional protected areas.

Algoa Bay is the only bay in South Africa containing two ports, a deep water industrial port (Coega) and a commercial port designed to include waterfront development and a marina (Port Elizabeth). These ports have led to major coastal development and increased intensity of human impacts on the marine environment. The Bay hosts several economically important species supporting both food supply (fish, rock lobster, squid/chokka) and tourism activities (dolphins, whales, endangered bird life). Considering the human impacts in the Bay and how they may affect the marine life, protected areas play a vital role in maintaining these ecosystem services. The pressures identified in this study only focused on those directly linked to benthic habitats. Several additional anthropogenic pressure layers should be considered if a complete pressure assessment is to be conducted in Algoa Bay such as, pelagic fisheries, boat-based tourism, alien invasive species, coastal development and disturbance, freshwater flow reduction, ocean noise and climate changes (Sink *et al*., 2012a). Shipping anchorage scour impacts, introduced for the first time in this study, added value to the cumulative pressure layer and should be included in future pressure assessments in South Africa. Davis *et al*. (2016) is one of few studies that focused on anchoring impacts. They identified large ships on anchor to pose a risk to the seafloor and its biota as a result of constant shifting of the anchor and its mooring chain as a result of ocean current movements in south east Australia. Based on Automatic Identification System (AIS) vessel tracking they were able to document the area affected by individual anchoring events and noted that some areas exceed 500m in diameter. AIS data can produce adequate information for Marine Spatial Planning including maritime traffic density, shipping lanes and navigation flows, but is limited to bigger vessels (Le Tixerant *et al*., 2018). By improving anthropogenic pressure data for Algoa Bay to better inform MPA proposals and designation, more effective protection can be provided, while avoiding areas of human use.

Analyses conducted in this chapter presents test case scenarios for what can be achieved when comprehensive information is gathered for Algoa Bay and combined to provide improved insight in support of MSP efforts. The alternative MPAs suggested herein would not necessarily be intended as exclusively a restricted area, but could rather include zoned protection levels according to the habitat type within the area. Zoning of the MPA would require further detailed analysis to determine the ecological value for the whole MPA area. Information presented in this study serves as a foundation for the larger Algoa Bay Project (Dorrington *et al*., 2018) currently underway that will provide more insight into marine spatial planning from multiple disciplines including maritime law, socio-economics, oceanography, marine biology and microbiology.

Chapter 5: Synthesis

This study utilised three approaches to investigate unconsolidated marine sediment habitats in the Bay. First, epibenthic communities and substratum are described using underwater seabed imagery and compared to the benthic habitat types of the National Biodiversity Assessment (NBA, Sink *et al*., 2012a, Chapter Two). This component provides baseline information of epibenthic communities in unconsolidated habitats in the Bay that have not yet been described. Second, abiotic factors (both station specific and long term interpolated factors) are tested as potential drivers of the epibenthic communities observed. These factors could be used as surrogates for further benthic mapping of similar habitat types. Finally, the representation of benthic habitats in Algoa Bay within the existing and proposed Marine Protected Areas (MPAs) is assessed using the national benthic habitat map, new data from this study, and mapping of anthropogenic pressures within Algoa Bay to generate alternative MPA designs. However, these designs are purely to illustrate a proposed methodology to improve the representation of unconsolidated sediment habitats in the MPA, and additional benthic sampling and pressure information is required before any alternative MPA designs can be recommended during the current Marine Spatial Planning (MSP) process.

Key findings

This study identified two significantly different epibenthic communities within Algoa Bay. As expected, these communities did not align with the national benthic habitats defined at a much broader scale in the NBA nor did they follow any clear spatial patterns. The visual assessment of the substratum and species diversity indicated that one community was dominated by sand and polychaetes, while the other community occurred exclusively on mixed substratum (rock, sand and shells) and included the presence of species commonly found in hard ground habitats. Habitats that include hard ground components (e.g. rock) are shown to host significantly different epibenthic communities with higher species richness than purely unconsolidated habitats (e.g. sand). This suggests that a visual assessment of substratum can effectively be used as a potential surrogate for mapping similar benthic communities.

A multivariate analysis revealed that a suite of abiotic drivers potentially explains the species diversity patterns of the epibenthic communities. Depth, mean grain size and, both the mean and variation of long-term interpolated bottom temperature, current speed and dissolved oxygen explain relatively high proportions (54.7% & 68.3% fitted variation respectively) of the variation in epibenthic species diversity in the Bay, but no individual factor can be considered as the primary driver of patterns in epibenthic communities in unconsolidated habitats. These factors act together as they are linked. Grain size can be dependent on the current speed as this will determine its movement and sorting structure. Furthermore, bottom temperature and current speed play a role in upwelling, should the current speed increase or decrease, the water temperature will fluctuate accordingly. In Algoa Bay water temperature may rise as the Agulhas current enters the Bay for short periods within the year. These links form part of the benthic-pelagic coupling and understanding how these links work will allow the modelling of habitat shifts. The spatial and temporal scale at which such factors are measured should also be considered before utilising them as surrogates for epibenthic habitat mapping. Long-term monitoring in Algoa Bay has provided insight into the oceanographic processes in the Bay allowing for the modelling of several parameters such as bottom temperature and current speed (Goschen *et al*., 2015; Bailey, Goschen and Hermes, in prep). In order to validate these models, *in situ* values are required and the current scale of sampling may require a higher resolution to accurately represent the highly dynamic Bay (Goschen *et al*., 2015). The scale at which epibenthic communities were surveyed requires a much higher resolution to assess the potential drivers of patterns of epibenthic communities owing to the high variability in epibenthic species observed in a small area sampled (171.7m^2). Measuring the abiotic factors represented in this study at a higher resolution may increase the proportion of variation explained by each factor, thus supporting the selection of a surrogate for epibenthic habitat mapping in the Bay.

Algoa Bay is considerably impacted by anthropogenic pressures owing to the large coastal metropolitan city of Port Elizabeth and the presence of two commercial ports in the Bay (Dorrington *et al*., 2018). The current protection status of benthic habitats in the Bay is limited to the area around Bird Island, with only one NBA habitat type included in this MPA. In response to Operation Phakisa, an additional MPA encompassing the existing MPA and other areas in the Bay was proposed. The assessment of the proposed MPA indicated an improved benthic NBA habitat protection, however two reef habitats identified within the Bay remained insufficiently protected. As a test case, two alternative MPA scenarios were proposed to represent 20% of all benthic NBA habitats in the Bay. Keeping the existing MPA in consideration, an alternative MPA was suggested that includes less area, but more habitat types than that of the current proposed MPA. The new design, however, purely demonstrates a potential method of delineating MPAs in the Bay, and do not consider important design aspects such as connectivity, or the representation of similar habitats in MPAs outside the study area. As an essential process supporting the persistence, recovery and productivity of marine ecosystems, connectivity should be a dominant focus during MPA delineation (Lagabrielle *et al*., 2014).

Limitations

The number of stations sampled was limited by time constraints and boat, financial and human resources, and was therefore restricted to only 13 stations. Owing to the nature of station selection that focused on spanning over two depth zones spaced evenly across the Bay, an uneven number of stations was sampled per NBA habitat type. Species accumulation curves indicated that two of the 4 habitat types sampled did not reach an asymptote, therefore additional sampling should be done in these areas to improve coverage (Gotelli and Colwell, 2011). The 13 stations sampled were not at the same locations as the seven oceanographic long-term monitoring stations (PELTER) as more biological stations were sampled than the monitoring stations and biological station selection required similar distribution of stations across the two depth zones identified (seven inshore and six offshore). Therefore, abiotic factors were interpolated from the seven PELTER stations in the Bay decreasing the accuracy of the abiotic factor value at each biological sampling station.

New techniques are available for habitat mapping in South Africa such as side-scan sonar and multi-beam mapping and in conjunction with visual assessment can be used to create habitat models for similar habitat types. These techniques were not available for this study, however, the Bay has been targeted for future sampling with these techniques as part of the Algoa Bay Project (Dorrington *et al*., 2018). Given that only13 stations have *in situ* unconsolidated sediment epibenthic data to validate future sonar and multibeam mapping, additional sampling will be required to develop a statistically acceptable habitat model for similar habitat types. Other abiotic factors could also be tested as potential drivers of benthic communities, including pelagic productivity (zooplankton) and upwelling indices.

Additional anthropogenic pressure data are required for a final MPA design and marine spatial plan for the Bay, including pressures such as commercial and recreational fishing activities. Stakeholder engagement should also be included as local knowledge will likely identify additional areas of ecological and economic importance (Smith *et al.,* 2008; Le Heron *et al.,* 2016).

Implications and recommendations

Habitat mapping plays a fundamental role in marine conservation and resource management (Stevens and Connolly, 2004; Cogan *et al*., 2009; Buhl-Mortensen *et al*., 2015a). South Africa's existing NBA, 2012 benthic habitat map was generated at a national scale often using broad-scale surrogates for areas not yet sampled. The misalignment of epibenthic communities identified in this study relative to the NBA, 2012 benthic habitat types supports the need for additional fine scale sampling and habitat mapping required for the Marine Area Plan in Algoa Bay and similar national priorities. The focus of this mapping should not be exclusively for Algoa Bay but rather include other areas known to play important ecological roles, particularly the additional proposed offshore MPAs. This will assist with future Marine Area Plans required by the MSP Act (2017). The use of surrogate abiotic data in models, to predict the presence of unconsolidated habitat, requires a suite of abiotic factors to be measured including mean grain size, depth, water temperature, current speed, etc. The use of both sediment samples and visual imagery of the benthic habitats in this study provided better insight into identifying the mosaics in unconsolidated sediment habitats and found them to host a different community of epibenthic species. Therefore, when additional sampling is implemented a similar method should be used to allow these mosaics to be meticulously described for fine-scale benthic habitat mapping in Algoa Bay.

Algoa Bay's benthic habitats are not exclusively unconsolidated marine sediment, but also include several high-profile reef complexes. Biodiversity varies among and within habitat types and there is an increasing consensus that higher biodiversity is often linked to increased ecosystem function (Hammill *et al*., 2018). The comparison of different habitat types within the Bay and assessment of their ecosystem functioning and services could provide improved understanding of the relationship between different habitat types and the potential impacts of different human activities on the benthic habitats to guide MSP.

Conclusion

While limited in station sampling density, two significantly different epibenthic communities were identified in the Bay potentially driven by a suite of abiotic factors including mean grain size, depth, bottom temperature and current speed. This information compliments the existing biotic and abiotic information collected in the Bay and will contribute towards baseline information underpinning demarcation of areas for protection, as well as validation of future benthic habitat models developed from sonar and multi-beam mapping products. Reassessment of the proposed MPA with fine-scale benthic habitat mapping is important for successful MSP implementation in Algoa Bay. The information gathered in this study represents an important contribution towards improved understanding of epibenthic community patterns in unconsolidated habitats which supports the implementation of the Algoa Bay Project Marine Area Plan.

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