Benthic algal communities of shallow reefs in the Eastern Cape: availability of abalone habitat.

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Abstract

Marine ranching has been identified as an alternative to traditional aquacultural rearing and growing organisms for consumption. In the Eastern Cape, abalone ranching is a new and experimental industry. The aims of the research were to: first develop a GIS model to assist management in site selection for abalone seeding; and secondly to develop and standardize the sampling methodology in order to ground truth the sites, and assist in the monitoring and habitat identification of abalone.

The GIS model developed in Chapter 3 was created using an unsupervised classification and fuzzy logic approach. Both vector and raster datasets were utilized to represent 7 different layers. Predominantly satellite imagery was used to classify the different substrate groups according to pixel colour signatures. The basic process was to apply a fuzzy rule set (membership) to rasters which gave an output raster (Fuzzification). The membership output rasters were overlaid which creates a single model output. It was found that model accuracy increased significantly as more layers were overlaid, due to the high variability within each of the individual layers. Model ground-truthing showed a strong and significant correlation (r^2 = 0.91; $p < 0.001$) between the model outputs and actual site suitability based on in situ evaluation.

Chapter 4 describes the investigation towards the optimal sampling methods for abalone ranching habitat assessments. Both destructive sampling methods and imagery methods were considered as methods of data collection. The study also evaluated whether quadrat and transects were going to be suitable methods to assess sites, and what size or length respectively they should be to collect the appropriate data. Transect length showed great variation according to the factor assessed. A transect of 15 metres was found to be optimal. Abalone counts showed no significant ($p = 0.1$) change in the Coefficent of Variance (CV) for transect lengths greater than 15m, and had a mean of 0.2 abalone per metre. Quadrat size showed a significant difference in functional group richness between quadrat sizes of 0.0625m^2, and 0.25m^2 but no difference between 0.25m^2 and 1m^2 quadrats for both scape and photographic quadrats. It was also found that between 5 and 10 replicates ($p = 0.08$) represents the functional groups appropriately using quadrats and that a 0.25m^2 quadrat is most suitable for sampling.

Chapter 5 describes the benthic community structure of Cape Recife shallow water reefs. Using the standardized methodology previously mentioned, 45 sites were assessed to identify the community structure. These sites were grouped into 5 different groups influenced by depth and substrate, as well as functional group composition according to a Wards classification. The community structure showed that depth and substrate play a significant role ($p < 0.05$) in the community type. There is also a significant relationship ($p < 0.05$) between complexity, rugosity, abalone presence and substrate.

During this study the basic protocols for site selection and benthic community monitoring have been developed to support the abalone ranching initiative in the Cape Recife area. It has also provided a baseline of the benthic community in the ranching concession area which will be used as a benchmark for future monitoring efforts. The site selection, sampling, and monitoring methods developed during the course of this work have now been rolled out as Standard Operating Procedures for the ranching programme in this area.

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Chapter 1. Introduction

The oceans make up more than 71% of the earth's surface coverage (Durack et al., 2012), with most of the productivity restricted to the continental shelf environment, extending down to a depth of 200 metres and only making-up about 7% of the oceans floor and as little as 0.5% of the oceanic volume (Chena and Borges, 2009). Modern man relies heavily on the ocean. Globally it was estimated that in the early 80's more than 60% of people live on the coastline (Branch and Branch, 1981). In South Africa it is estimated that for each person there is currently less than 10 cm of coastline available (Branch et al., 2010).

The Eastern Cape is well known for its diverse and productive coastline. Often referred to as the South Coast, the area is unique in temperature and the species found in the waters (Branch et al., 2010; Branch and Branch, 1981; Lubke, 1988; Smit et al., 2013). The South coast is influenced by two major currents, the warm Agulhas current which is located approximately 17 to 81 nautical miles off of the coast (Bryden et al., 2005) and often brings eddies, or pockets of warm water, in to the shallow water systems (Branch et al., 2010; Branch and Branch, 1981). The cold Benguela current is the second driver of this unique coastline. It pushes in with the long shore current that hugs the coast (Branch and Branch, 1981).

Much attention has been placed on the ocean's resources and related activities in South Africa, particularly with the implementation of operation "*Phakisa*". Aquaculture is of rapidly growing interest. Today the supply of marine resources is increasing at a rate of 3.2 percent annually (FAO, 2014). Edwards (2015) is of the opinion that aquaculture has the potential to expand and meet demands of the growing human population to supply cheap and accessible protein but only if more widespread ecologically sustainable aquaculture is introduced and the cost of the ecological services are incorporated into facility accounts. This seemingly beneficial view of the supply of protein sources by aquaculture is held by a number of authors (Frankic and Hershner, 2003; Gjedrem et al., 2012; Grigorakis and Rigos, 2011; Sapkota et al., 2008). This is mainly due to a drive to increase sustainability, preserve stocks and find alternative food sources (Liao et al., 2003).

One way of solving the continuous demand of fish, and a possible way to ensure the livelihoods of many subsistence fisheries, is the introduction of stock enhancement and ranching programs. Bartley and Bell (2008) regards the success of any program to fall on three pillars where the goals for the program are explicitly stated with regards to what the purpose of the program is, and the benefits that it may have, and the programs scientific defensibility, and adaptability as the project progresses and new information is obtained.

Haliotis midae L. is one of South Africa's most valuable export products from the aquaculture industry, and is the largest aquaculture subsector contributing more than 51% (DAFF, 2011). The market for abalone is vast but is very driven by the colour, texture, and shape of the abalone supplied (Oakes and Ponte, 1996). These are often heavily affected by the artificial feed that the abalone are fed and can greatly affect the profit margin that a facility can make (Fleming and Hone, 1996).

Ranching and stock enhancement is believed to be an alternative to this as it provides a platform whereby the abalone can be naturally reared to harvestable size, allowing for the greatest likelihood of these aforementioned prerequisites are met (Liao et al., 2003). While an argument can be made for ranching activities, there are risks associated with ranching activities, including the impact of released stocks on the gene pool of the wild stocks (Araki and Schmid, 2010; Evans et al., 2004; Hara and Sekino, 2007; Roodt-Wilding, 2007), as well as the ecological impact that may occur (Bartley and Bell, 2008; Bartley and Leber, 2004; Bell et al., 2008; Loneragan et al., 2013; Mustafa et al., 2003).

The first step in the project was to develop the structural support systems by which the operation could take place. Also due to the experimental nature of ranching in the Eastern Cape, knowledge availability on the effects of ranching is deficient. Understanding the community structuring and impacts is critical, as abalone are mostly opportunistic feeders, they do not tend to effect the community structure at increased densities (Hart et al., 2013a), but long term effects may be seen as *Haliotis midae* is predominantly a drift algae feeder, but does show limited grazing and only long term observation may reveal any ecological changes that may occur (Zeeman et al., 2014, 2012).

It was for this reason that the aims and objectives of the study were to:

Aim 1. Map and prioritize the coast for abalone ranching concession area 2;

Objective 1. Develop a GIS map of habitat, resource availability, constraints and opportunities for the Eastern Cape in terms of abalone ranching.

Objective 2. Develop a user friendly model to predict habitat suitability and availability, as well as model possible change to habitats if ranching operations were to occur

Objective 3. Assess areas with highest rating in terms of Objective 1.1 using a standardized methodology (see Aim 2)

Aim 2. Develop a standardized and effective methodology to assess abalone habitat for concession area 2

Objective 1. Determine the optimum transect length required for effectively sampling benthic communities and possible abalone habitat on temperate reefs along the Eastern Cape coast

Objective 2. Determine the optimum quadrat size required for effectively collecting data and specimens on benthic communities and possible abalone habitat of temperate reefs along the Eastern Cape coast

Aim 3. Describe the habitat types and communities of the Cape Recife area.

Objective 1. Describe the influence of depth on habitat structure and changes in dominance between functional groups.

Objective 2. Determine the relationship between cover estimates and biomass

Objective 3. Determine environmental and ecological associations between functional groups and plots identified in the ecosystem.

Predominantly this project has been driven to assist the abalone ranching project under operation “*Phakisa*” in their operational choices and decisions. Opportunity has arisen to allow for the assessment and identification of the main seaweed species of the Port Elizabeth reefs, as well as for the assessment of the ecology and functioning of the shallow subtidal reef systems of the area.

Chapter 2. Literature Review

2.1 Introduction and history of the commercial abalone sector in South Africa

The South African traditional abalone fisheries are riddled with a complex and unique history. Preceding the development of the abalone commercial industry, this fishery was limited to small communities that would fish the abalone, and was very sustainable (Raemaekers et al., 2011), but very little record of this has been documented and it was only after the establishment of the commercial industry that this all changed (Raemaekers et al., 2011; Raemaekers and Britz, 2009). Influenced majorly by the colonial and apartheid eras, laws such as the Black Land Act 27 of 1913, the Development Trust and Land Act 18 of 1936, the Coloured Labour Preference Policy, and the Group Areas Act 41 of 1951 drove a great deal of inequality in these fisheries (Hauck and Sowman, 2001). This helped to drive the predominantly coloured communities in the western cape into the fisheries industry (Sowman, 2006). Then with the demolishment of the apartheid regime and implementation of democracy in 1994 came the rise and dominance of organised crime and syndicates (De Greef and Raemaekers, 2014).

The industry itself was unrestricted when it initially commenced in 1949 (Goga, 2014), and during this time teams of divers on surface supply (referred to as Hookah diving) would collect the abalone with no limitations (De Greef and Raemaekers, 2014). This unsustainable harvesting peaked at more than 2800 tons per annum in 1965 (De Greef and Raemaekers, 2014) This drove the implementation of seasonal quota, which was only introduced in the 1970’s (Steinberg, 2005) which assisted in stabilizing the harvest to approximately 700 tons per annum (De Greef and Raemaekers, 2014). In the late 1980’s a division of the western cape coast into seven different fishing zones was implemented allowing each fishery a Total Allowable Catch (TAC) for their permitted area, based on stock assessments completed by relevant authorities (De Greef and Raemaekers, 2014), while the Eastern Cape remained relatively untouched and untapped due to it being deemed too patchy and scarce (De Greef and Raemaekers, 2014; Raemaekers and Britz, 2009). One of the major changes in the industry was the disbandment of the apartheid era which left many of the coloured divers in the Western Cape working for white owned groups (De Greef and Raemaekers, 2014). This drove the development of syndicates and gangs in many of these communities, which developed structured and organised networks that collected, transported, weighed, processed and illegally exported the product to the markets in Asia (Brill and Raemaekers, 2013; De Greef and Raemaekers, 2014; Raemaekers et al., 2011; Raemaekers and Britz, 2009; Steinberg, 2005).

Another aspect that drove and continues to fuel poaching was the weakening of the Rand (ZAR) to the Dollar (USD). In 1995 a diver could get approximately 40-50 ZAR a kilogram equating to approx. 2000 USD per bag (≈40kgs) (Hauck and Sweijd, 1999). In 2009 Port Elizabeth divers were getting approximately 40 ZAR/kg which at the market exchange rate was approximately 7.4 ZAR = 1 USD, meaning a 40 kg bag would give a diver approximately 11 840 ZAR. Today abalone are estimated to fetch up to 4500 ZAR per kg on the illegal market, with a diver will only obtain 300 ZAR/Kg (Roelf, 2014) however the market is very specific in size and quality of the abalone (i.e. the animal must be larger than a certain size and cannot have any damage to the foot) (To et al., 2006). In 2014 divers obtained about 300 ZAR/Kg (Roelf, 2014), and had to pay their helpers according to agreed verbal contracts, (pers. comm. Tom Swartz (TTF)), as well as for maintenance of equipment, boats, fuel etc. which directly impacted the profit margins for a poacher (De Greef and Raemaekers, 2014).

In the early 2000's illegal abalone exports were estimated to be about 2000 tons per annum, whereas the legally recorded catch was only approximately 300 tons per annum (De Greef and Raemaekers, 2014), and illegally poached abalone made up an equivalent of 43% of the labelled "cultured abalone" supplied to Hong Kong (To et al., 2006). This is also true for South Africa, which is one of the largest exports of dried abalone to Hong Kong, exporters contributing 19% to the market in 2002 (To et al., 2006) and an conservative estimate of 1723 tons per annum in 2012 (De Greef and Raemaekers, 2014). This rampant illegal traded forced the South African Government in 2005 to implemented "Operation Neptune" and "Operation Trident" to combat poaching and illegal harvest of abalone, with special designated environmental courts and officers (Goga, 2014). Although successful, these programs were short lived (Goga, 2014). Then in 2007/2008 abalone harvesting was totally suspended by the minister (Marthinus van Schalkwyk) of the Department of Environmental Affairs and Tourism (DEAT) (Raemaekers et al., 2011). This did not however stop the poaching syndicates and also forced many of the valid subsistence permit holders to turn to poaching in order to survive (Goga, 2014), and drove syndicates to launder the abalone through neighbouring countries in order to export to Asia (De Greef and Raemaekers, 2014). To curb this laundering abalone was listed in CITES APPENDIX III but due to logistical issues, abalone was then removed in 2010 from the list and has not been re-added to date (De Greef and Raemaekers, 2014).

The Eastern Cape abalone is generally smaller than its neighbouring population and the shell develops a very dark red colour as the main source of food is red seaweeds (as discussed later) (Wood, 1993). Poaching of this abalone has been intensive, and difficult to stop as the fisheries in this area have never really truly developed and no legal markets have been developed (Raemaekers and Britz, 2009). Godfrey (2003) stated that due to the rampant poaching, emergent stock had been possibly depleted beyond recovery and that the stocks of abalone would collapse.

While stocks have not yet collapsed, they are considered to be severely depleted in the Cape Recife area (DAFF, 2014a; Joemat-Pettersson, 2010). With the implementation of operation Phakisa in 2013 rights to ranch abalone have been given to Ulwandle Fishing (Pty Ltd) through a concession area extending from the Skoenmakerskop MPA to Cape Recife for a pilot ranching project (Figure 4) (Joemat-Pettersson, 2010). This area has been demarcated for the development and implementation of a sea ranching program of *Haliotis midae* (Abalone). In partnership with Lidomix investments Pty Ltd; Wildcoast Abalone, Rhodes University, Tactical Task Force, and with logistic support from the Nelson Mandela Metropolitan University, the program has successfully seeded more than 1 million abalone on the reef systems around Thunderbolt and Cape Recife reefs in 2015. The program due to this success has now started to shift its focus from Cape Recife to other areas within the designated zone, as well as made the focus on seeding areas that are less than 5 metres in depth.

Ultimately the project Phakisa is aimed at building a sustainable fisheries in areas that will benefit through exports as well as benefit the local community (Benkenstein, 2015). According to President Jacob Zuma the project is aimed at bringing the public and private sector together. It is currently headed by the Department of Environmental Affairs and is to focus on utilising the vast resource of our oceans more effectively and potentially could contribute up to one hundred and seventy-seven billion rand to GDP by 2033. In comparison

South African ocean resources only contributed around fifty-four billion in 2010 (H.E. President Jacob Zuma, 2014).

Currently there are a number of legal permit holders around the coast and they are allowed to harvest a predetermined TAC of abalone per annum determined by DAFF (DAFF, 2014b). According to a DAFF (2014b) media statement, the maximum tonnage is 25 tons for Areas A and B, while areas C and D may not be harvested in 2014/15. The Minister of Agriculture, Forestry and Fisheries Mr Senzeni Zokwana, also made a statement regarding long term permit expiries that right holders, who had abalone fishing rights in the 2013/2014 fishing season, were exempted to afford current right holders opportunity to catch uncaught 2013/14 abalone allocation and secondly right holders have exemptions until July 2015 (Palesa Mokomele, Spokesperson of the Minister of Agriculture Forestry and Fisheries, 2014).

In the early 1990’s abalone farming was initiated in South Africa in an attempt to ease the pressures on abalone (Hauck and Sweijd, 1999) with the earliest successful spawning events being recorded in 1981 (Sales and Britz, 2001). Then in the mid to late 1990’s ranching efforts were being made (Goga, 2014; Hauck and Sweijd, 1999; Liao et al., 2003; Raemaekers et al., 2011; Sales and Britz, 2001).

Initially, according to Sales and Britz (2001) a collaboration between Rhodes University, The University of Cape Town, the Council for Scientific and Industrial Research and three fisheries companies started the pilot farm and from there 12 more farms were built and made operational by the late 1990’s/early 2000’s (Hauck and Sweijd, 1999; Sales and Britz, 2001). By 2011, 14 farms were operational between East London and Port Nolloth (Sales and Britz, 2001), which includes two open water based farms, one being a cage farm facility and the other a ranching operation (DAFF, 2011). Most recently, in the Eastern Cape, a new ranching operation has been initiated by Wild Coast Abalone at Cape Recife in the Port Elizabeth. In comparison China in 2013 had approximately 300 operating abalone farms, with 95% of these farms being sea-caged-based and the largest of these farms producing more than 1000 MT of abalone per annum (Cook, 2014).

2.2 Stock Enhancement, Restocking and Ranching

Many fisheries are under threat and species exploited by these industries are severely depleted (Bartley and Bell 2008). In an attempt to recover and replenish stocks many of the industries have started programs that according to Bartley and Bell (2008) include:

1) Stock Enhancement
2) Restocking
3) Ranching

These three terms were interchangeable and synonymously used for many years, which caused a great amount of confusion to the actual meaning of each term (Mustafa, 2003).

Bartley and Bell (2008) define (1) Stock Enhancement as the augmentation of juveniles into wild stocks in order to increase productivity of an operational fishery (Bartley and Bell, 2008). It is the manipulation of the physical or biotic environment through the introduction of new stock to supplement natural recruitment for later harvesting (Booth and Cox, 2003). The main goal of Stock Enhancement is collectively to form larger populations by combining introduced juveniles into the wild stocks biomass so that they can be harvested commercially in the future (Mustafa et al., 2003).

(2) Restocking is defined as the release of cultured juveniles to restore severely depleted wild stock biomass and to increase the level of spawning individuals so as to provide regular, substantial yields in the future (Bartley and Bell, 2008). Restocking can also include re-introductions into areas in which a species historically occur (Molony et al., 2005).

(3) Ranching can be defined as a "put-grow-take" system (Bartley and Bell, 2008). It is similar to stock enhancement and restocking as it is the input of hatchery bred juveniles into the environment but with the goal of harvesting the stock after a defined period of growth and no intention of interbreeding or supplementing the wild stocks (Loneragan et al., 2013).

Success of these three types of approaches has been varied according to species used, release time, environmental conditions, numbers being released and area in which they are released in (Bartley and Bell, 2008). Success of these approaches is also largely associated with the capacity that a natural system has to support released individuals produced through aquacultural practices (Loneragan et al., 2013). These approaches are also driven by a demanding and rigorous scientific monitoring programs whereby pilot projects are run and follow-up counts are done periodically (Bartley and Bell, 2008; Bartley and Leber, 2004; Molony et al., 2005).

In Australia attempts have been made with various success on releases of invertebrates such as the saucer scallops (*Amusium balloti*), greenlip abalone (*Haliotis laevigata*), Brown tiger prawns (*Penaeus esculentus*) and trochus (*Tectus niloticus*) (Loneragan et al., 2013). The largest study of these was the tiger prawn and the project failed due to feasibility of the project as not enough funds were available to produce enough seeded to make harvestin viable. Hart et al. (2013a, 2013b) found that for *H. laevigata* ranching was feasible as long as the stocking densities remained below that of the environmental carring capacity which means that little or low environmental impact will occur.

Hart et al. (2013a, 2013b) are of the opinion that ranching and stock enhancement are feasible options to increase and manage the fishery in Australia. In New Zealand Roberts et al. (2007) also found that the abalone stock enhancement could yield an increase of up to 10% per annum increase in harvest, but only if correct habitat and sites are selected and suitable size animals are used.

2.3 The Shallow inshore Agulhas reef system community structure

2.3.1 Biological Structure and Seaweed Communities

2.3.1.1 Biodiversity

Globally it is estimated that there is a total number of 72,500 alga species with approximately 20000 diatoms, 2500 species of dinoflagellates, with approximately 500 remaining to describe; and 2000 marine and freshwater algae, with approximately 1000 still to be described (Guiry, 2012). Along the 3100 km stretch of relatively pristine Southern Africa coast (Branch et al., 2010), more than 12 900 marine species have been described (Griffiths et al., 2010), which makes it about 6% of the known coastal marine species worldwide (Branch et al., 2010) and hosts approximately 33% being endemic to the South African coast (Griffiths et al., 2010). It is also estimated that more than 7500 species are yet to be described (Griffiths et al., 2010).

There are estimated to be about 400 species of seaweeds in the Western Cape and more than 800 species for the entire south African coast, however there is still great uncertainty in the matter (Stegenga et al., 1997). The Eastern Cape is a transitional area between South Africa's warm sub-tropical waters of Kwazulu-Natal and the cold temperate waters of the Western Cape (Lubke, 1988). It forms the majority of the South Coast which typically ranges from between Cape Point to the Mbashe River (Bolton and Stegenga, 2002; Sink et al., 2004). According to Bolton and Stegenga (2002) more than 803 species have been identified for the South African coast with 101 Ochrophyta, 149 Chlorophyta and 553 Rhodophyta being identified (Figure 1).

It is estimated that approximately 58% of the Rhodophyta, 33% of the Ochrophyta and approximately 28% of the Chlorophyta are endemic to South African waters (Stegenga et al., 1997).The East coast is mainly dominated by the chlorophyta (Bolton and Stegenga, 2002) Anderson et al. (2005) counted as many as 104 seaweed taxa for 25 quadrats at Sodwana on the East Coast, while Bolton and Stegenga (1987) found 128 species in the Hluleka flora in the Transkei on the east coast.

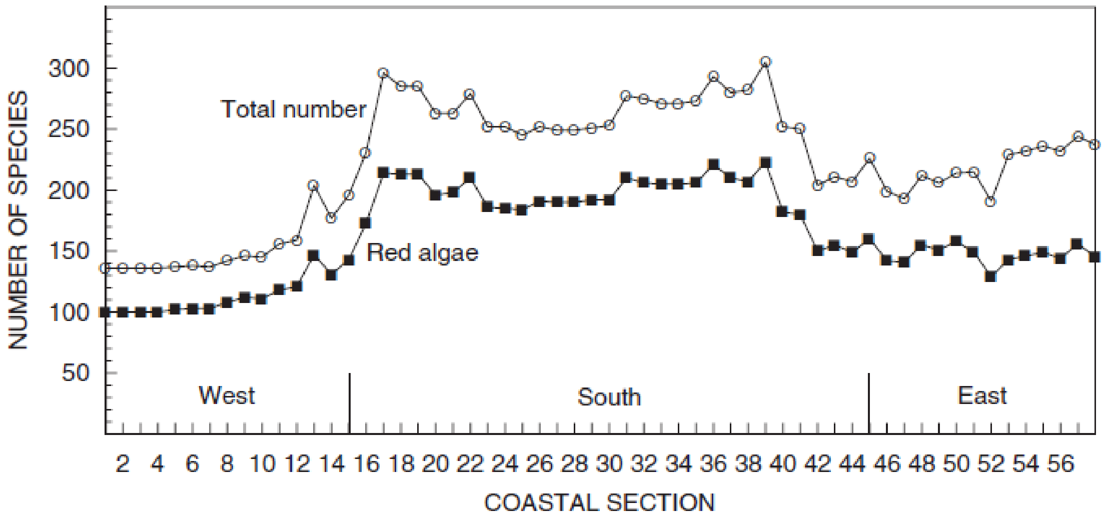

Figure 1: The seaweed species richness for the Coast of Southern Africa from Bolton and Stegenga (2002)

In Port Elizabeth estimates of the number of species have ranged depending on the study. At Bird Island a total of 122 species across 31 quadrats has been identified (Anderson and Stegenga, 1989), while in the western sector of Algoa Bay Knoop (1988) identified 36 seaweed species and identified *Plocamium corallorhiza* as the dominant seaweed in shallow subtidal regions, while Munnik (1987) identified 98 species at Noordhoek, but was focused on the intertidal regions. Bolton and Stegenga (2002) state the highest diversity of seaweeds is found between the Port Elizabeth and Port Alfred (Sections 36 to 39; Figure 1).

2.3.1.2 Biotopes

In 1877 a German biologist Karl Möbius used the term “biocönose” to describe an oyster bank he was requested to study (Olenin and Ducrotoy, 2006). This relates to the community patterns formed by these beds. This was however not formally recognised until Dahl (1908) coined the term “biotope”, stating that “so kann man die deutschen Worte Gewässer und Geländerearten als Biotope wiedergeben” which is roughly translated to the combination of the waters (“Gewässer”) and the type of terrain (“Geländerearten”) reproduces (“wiedergeben”) the term “biotope” (Dahl, 1908) but only refers to the physical environment interaction on species and not interspecies interaction (Olenin and Ducrotoy, 2006).

This terminology of a “biotope” was not widely used or accepted until the early 1990’s (Olenin and Ducrotoy, 2006). Today it is often used synonymously with habitat (Connor et al., 2004) due to the use by policy makers, scientists and conservation managers (Olenin and Ducrotoy, 2006). According to Brodie et al., (2009) a biotope is an algal assemblage that may occur repeatedly over a larger region. It is also influenced by the physical environment in a “subsidy-stress” manner specific to the species found in the assemblage (Lugo et al., 1988). Biotopes are considered to be the interaction between habitats and their associated communities (Sotheran et al., 1997). For example Hansen and Snickars (2014) identified biotopes in the northern Baltic Sea, and found the shallow sheltered soft- bottom biotopes, which are rich in plant and invertebrate diversity, were also are important for the recruitment of fish.

Biotopes have been used by ecologists to assess a population’s network of interaction and the biotic and abiotic components of an ecosystem, through the use of functional groups to better understand this relationship at a hierarchical and finer scale (Olenin and Ducrotoy, 2006).

Biotopes are important in the monitoring of environmental health in benthic ecosystems (Caeiro et al., 2005). For example, Caeiro et al., (2005) used seven benthic biotopes to map the estuarine environment of the Sado estuary in Portugal. These biotopes were selected using multivariate analysis (Caeiro et al., 2005), mostly between community structure and the environmental variables of the region (Ducrotoy, 2010), as well as the use of indices to quantify the biotope parameters (Caeiro et al., 2005). This can then be coupled with remote sensing data and other imaging techniques to detect changes in the environment, composition and health of an area’s ecosystem (Caeiro et al., 2005).

2.3.2 Physical aspects

2.3.2.1 Temperature

Stephenson and Stephenson (1949) were the first authors to suggest the zonation patterns seen on rock shores around the world, particularly the influence of temperature on the structuring found between the high and low tidal marks. Branch and Branch (1981) depicted the south African coast into three major temperature regions, with the cool West Coast, temperate South Coast and the warm temperate/subtropical East Coast. It is argued that 6 biogeographic regions exist but the general pattern of cooler temperatures in the west and warmer temperatures in the east is prevalent (Branch et al., 2010).

Algoa Bay and the surrounding area both fall in the unique temperature region called the South Coast. Driven by the warm Agulhas current which can be found on average between 38km from shore at Cape Padrone and 51km at Cape Recife with a temperature range of between 20°C and 24°C (Goschen and Schumann, 2011). Knoop (1988) found the mean annual temperature at 15 m was 18.4°C with a mean fluctuation range of 1.8°C and range from more than 23°C down to about 11°C. This is typical of the south coast which has the greatest temperature variation of the three main coastal regions. An evident annual cycle ranging between 14-15°C (±1°C) and 20-22°C can be found in the surface waters (Schumann et al., 2005). This however has been questioned as Smit et al. (2013) showed that there is significant difference between high resolution satellite images and in-situ measures and states the in-situ measures are recommended for inshore and shallow water system measurements, which can vary as much as 5°C. Temperature ranges can be as low as 9°C and can, at distances as far out as 190km from the shore reach 22-23C (Ismail et al., 2015). Seasonal thermoclines are prominent in the Algoa bay region (Goschen et al., 2015, 2012, Goschen and Schumann, 2011, 1995; Schumann et al., 2005) and can regularly decrease temperatures to below 10°C (Goschen et al., 2015, 2012, Goschen and Schumann, 2011, 1995; Schumann et al., 2005).

2.3.2.2 Wind

The wind in Algoa Bay and surrounding areas is highly variable but is mostly dominated by westerly and easterly winds (Schumann et al., 1991). In the Cape Recife area a westerly or south-westerly wind drives the water away from the shore causing cold nutrient rich water to replace it (Goschen and Schumann, 1995). Wind is highly important as a marine system driver as wind-driven upwelling events drive and directly impact frequency of nutrient availability, water temperature fluctuations and ecosystem health (Goschen et al., 2012; Schumann et al., 1991). The wind in Algoa Bay is most dominant during the changing seasons, and mainly over the months of May, June and July having the lowest wind speeds and the maximum wind speeds being recorded between September and November (Schumann et al., 1991). On average, speeds range between 5 and 18 knots with maximum speeds being recorded at over 50 knots (Schumann et al., 1991).

2.3.2.3 Waves and Wave Action

The south coast is predominately influenced by kelvin waves (Goschen and Schumann, 2011, 1995; Schumann et al., 2005). Waves originate both from offshore origins, including depressions and the Agulhas current as well as local conditions, mainly wind driven (Mallory, 1974). Kelvin waves normally occur over flat bottom sea floors and headlands such as that of Cape Recife and play an important role in the upwelling process (Goschen and Schumann, 1995). Waves play an important role in the turbidity and general water moment and disturbance of shallow water systems (Clark, 1997). It has been shown that waves can increase the available space for seaweeds and algae, as well as sessile animals, which in turn increases the competition of the exposed areas (Leigh et al., 1987). Most exposed shores tend to be dominated by filter feeders while sheltered areas tend to have higher seaweed biomass, diversity and more microhabitats (Stegenga et al., 1997).

2.3.2.4 Substrate

Substrate type has long been known to influence and determine community structure (Garner, 2013). Most seaweeds prefer solid rock substrates, and inundation due to sediment deposition tends to damage and even kill many seaweed species (Garner, 2013; Piazzi et al., 2002). In South Africa there are three main types of substrates including consolidated rock, mixed unconsolidated substrate and unconsolidated sandy substrate (Garner, 2013; Sink et al., 2011). It has been suggested by Rust (Rust, 1991) that the geological knowledge of the South African coastline is meagre and requires more in-depth attention from researchers.

2.3.2.5 Tides

South Africa experiences tidal amplitudes classified as micro-tidal ranges on a semi-diurnal cycle and a height range of less than 0.5 metres (neap cycles) and exceeding 2 metres during spring tides (Schumann et al., 2005). Historically abalone were found in the intertidal and very shallow subtidal regions, with very large individuals being found at greater depths (approximately 10m) (Wood, 1993; Wood and Buxton, 1996), where juveniles tend to prefer the very shallow, high water movement, wave impacted, high crevice and cryptic substrate, and intertidal regions (Wood and Buxton, 1996). Due to the demand for this species coupled with its preferred shallow habitat, it has been highly impacted by recreational and commercial harvesting (Hauck and Sweijd, 1999).

2.4 Abalone feeding biology and habitat requirements

2.4.1 Habitat

Haliotis midae (Abalone) are very cryptic animals especially during the juvenile stages and tend to demonstrate a very strong inclination towards microhabitats (Wood, 1993). They are normally found on consolidated substrate in the very shallow subtidal to intertidal fringes in very wave exposed environments (Wood, 1993; Wooldridge and Coetzee, 1988). Abalone habitat is defined by an abundance of overhangs and crevices, with good availability of food and encrusting corallines and sponges (Wood, 1993). There appears to be a strong relationship with sea urchins as juveniles as this relationship provides the abalone with protection (Day and Branch, 2002a). de Waal (2005) although indicated that while sea urchins can indicate suitable habitat they are not the exclusive indicator.

2.4.2 Feeding

Abalone along the South African coast are strictly herbivorous (Barkai and Griffiths, 1986). Being herbivorous abalone feed by two means, firstly through active grazing and by trap feeding (Wood and Buxton, 1996). Abalone along with other grazers such as sea urchins and alikreukel, can enhance algal productivity and recruitment even if biomass reduction occurs in algal composition, meaning there are more species with lower biomass for each individual (Iken, 2012). Grazing can impact seaweed sporlings establishement, but if seaweed sporlings do establish then grazing pressure can become less (Branch and Branch, 1981).

Wood (1993) identified thirty-seven seaweed species in the stomach contents of *H. midae*, with the most important of these being *Hypnea spicifera* (Suhr) Harvey, *H. rosea* Papenfuss, *Plocamium corallorhiza* (Turner) Harvey, *Ralfsia verrucosa* (Areschoug) Areschoug, *Neoralfsia expansa* (J.Agardh) P.-E.Lim & H.Kawai ex Cormaci & G.Furnari and *Calliblepharis fimbriata* (Greville) Kützing. In the Eastern Cape it seems that the juveniles showed the greatest selectivity is towards *N. expansa* and *Ralfsia verrucosa* (Areschoug) Areschoug which they seem to graze on in the cover of darkness (Wood, 1993), while adult abalone showed the greatest selection towards *Ulva* species and *R. verrucosa* (Wood and Buxton, 1996). At Cape Hangklip the diet of abalone incorporated more *Plocamium*, while at Marcus Island the dominant food source was *Ecklonia maxima* (Osbeck) Papenfuss (Figure 2) (Barkai and Griffiths, 1987). *Plocamium* could make up to 45.9% of the gut content at Hangklip, while *Ecklonia* could make up 46.4% and 80.6 % for Marcus Island depending on the season (Barkai and Griffiths, 1986). Although species such as *Pachymenia orbitosa* (Suhr) L.K.Russell, *Pterosiphonia cloiophylla* (C.Agardh) Falkenberg, *Phyllymenia* spp. and *Ulva* spp. were also found in relatively high quantities in the stomachs of the abalone (Figure 2) (Barkai and Griffiths, 1986).

Species of seaweed that were totally avoided in the Eastern Cape included *Amphiroa ephedraea* (Lamarck) Decaisne, *Arthrocardia carinata* (Kützing) Johansen, *Corallina* sp., *Mesophyllum engelhartii* (Foslie) W.H.Adey, and *Ecklonia radiata* (C.Agardh) J.Agardh (Wood and Buxton, 1996).

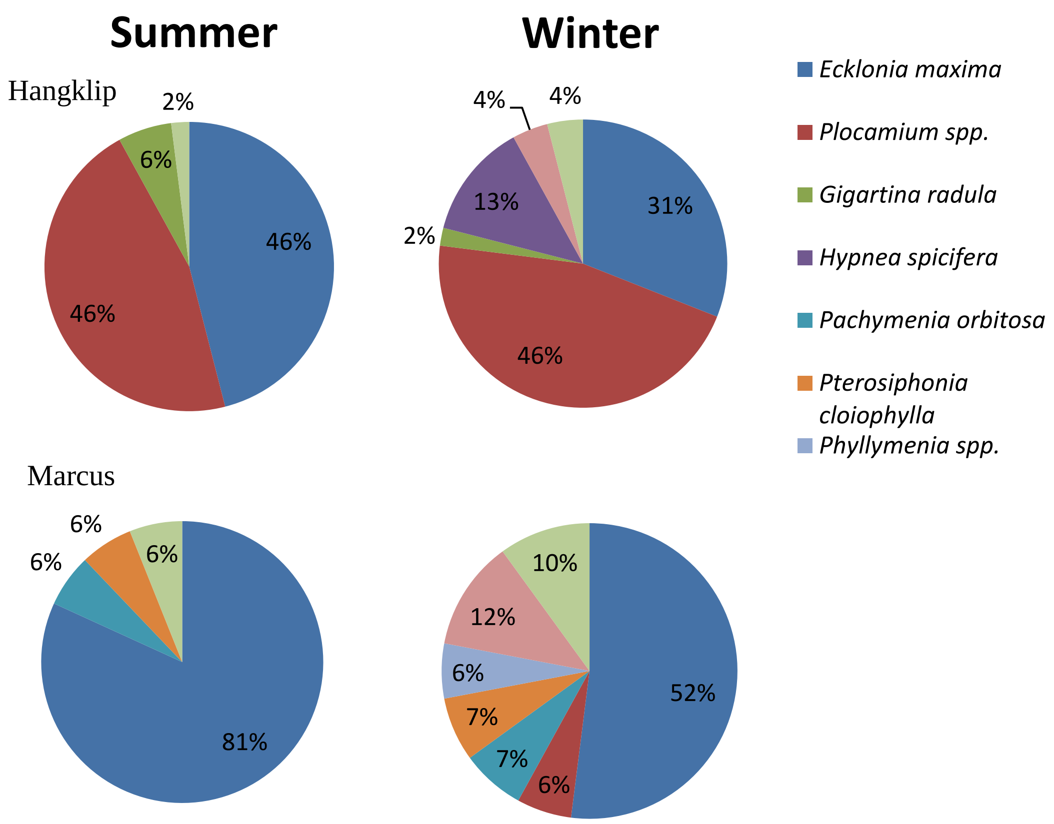

Figure 2: Seaweed species consumed by *Haliotis midae* for Cape Hangklip and Marcus Island (adapted from Barkai and Griffiths {Formatting Citation})

According to Knoop (1988) *P. corallorhiza* and *A. ephedraea* are very abundant seaweeds in Algoa Bay down to approximately 7 metres depth, with the average biomass of these seaweeds remaining relatively constant at around 284 g drymass m^{-2}. Wood (1993) identified 95 seaweed species at Great Fish Point, with *H. spicifera*, *H. rosea*, *P. corallorhiza*, and *Lithothamnion* sp. being the most predominant species and species such as *Gelidium amansii* , C. *fimbriata* and *Caulerpa filiformis* (Suhr) Hering making up the pinnacles and gully habitats. There is however no correlation between macroalgae availability and habitat selection by abalone (Wood and Buxton, 1996).

2.4.3 Distribution

Haliotids (the abalone family) have been mentioned as early as Aristotle ca. 347 B.C. and have over 90 species known worldwide, with more than 56 known extant species (Rhode et al., 2012) and more than two dozen common names across 15 languages (Wood, 1993). The genus *Haliotis* is distributed from subarctic to Antarctic waters (Wood, 1993) but the majority are in temperate, sub-tropical and tropical waters (Rhode, 2013). It is supported through fossil evidence, of which more than 35 proposed fossils have been utilized to show that the ancestral path of *Haliotis* dates back to approximately 70-80 (MYA) during the late cretaceous. However on the contrary due to the process of mineralization causing soft tissue loss and the morphological plasticity of the shells, uncertainty has arisen with regard to this accuracy (Rhode, 2013). However using the protein haemocyanin, Streit et al., (2006) found supporting evidence for the hypothesis that the Haliotidae origin was most likely in the Tethys Sea of Europe and radiated eastward to Asian and the American Pacific regions.

Today the most northern distribution record of abalone is the species *H. kamtschatkana* (Rogers-Bennett, 2007), which is found from about Sitka (Emmett and Jamieson, 1989) around the southeast of Alaska (Rogers-Bennett, 2007), down to around San Diego, CA (Emmett and Jamieson, 1989; Rogers-Bennett, 2007). This species has been well documented in terms of its farming suitability, specifically in ranching (Emmett and Jamieson, 1989; Roodt-Wilding, 2007), as considerable research has been undertaken to understanding the growth rates, feeding habits and general ecology (Emmett and Jamieson, 1989; Lloyd and Bates, 2008; Okano and Kvitek, 2009; Stierhoff et al., 2012). The southernmost distributed, and sought after abalone species are *H. austrais* (Grindley et al., 1998) and *H. iris* (commonly known as paua) (Sainsbury, 1982). These can be found down to the Snares Island group approximately 105km south-west of Stewart Island, consists of two smaller Islands called Main Island and Broughton Island. These islands have high rising granite cliffs, rising to 150m high on the west and southern sides, with very rough seas (Crawley and Cameron, 1972; Warham et al., 1982). Arguably though, the most southern abalone species has to be *H. virginea virginea* which are found around the Campbell islands (approximately 600km from the Stewart Island) but this species holds very low commercial value (Grindley et al., 1998).

In a genetic investigation performed by Streit et al., (2006) there are two major clades of abalone, namely the Northern Pacific clade and the European–Australasian clade. The Northern Pacific Clade mainly consists of *H. fulgens, H. corrugata, H. wallalensis, H. cracherodii* and *H. rufescens* from California and *H. gigantea*, and *H. discus hannai*, from Japan. The European–Australasian clade is smaller and more geographically diverse with *H. tuberculate* from Europe. *H. midae* from South Africa shows clustered groups along the coast that are largely isolated due to the current dynamics along the coast (Rhode, 2013) but show close relationship to *H. asinina* from Austrailia, *H. diversicolor* from Japan and *H. discus supertexta* from Taiwan (Streit et al., 2006). The South African species besides *H. midae* in the genus *Haliotis*, include *H. spadicea*, *H. speciosa*, *H. parva*, *H. queketti* and *H. pustulata* (Branch et al., 2010; Wood, 1993).

2.5 Geographic Information systems (GIS) for Fisheries Management

2.5.1 Classification techniques: Supervised and unsupervised

Classification techniques are used in GIS to identify objects based on their spectral or pixel characteristics from satellite imagery (Bhaskaran et al., 2010; Buhl-Mortensen, 2009; Burrough, 1986). The use of these techniques is broadly divided into supervised and unsupervised methods.

2.5.1.1 Unsupervised

Unsupervised classifications are rapid and don't require foreknowledge in the processing of classes (Baattrup-Pedersen et al., 2013; Fraisse et al., 2001). It is normally included in most GIS packages (Fraisse et al., 2001) and use an algorithm to classify image data into unimodal spectral classes (Liu, 2003) of which the user can define the number of classes desired without reclassifying the output (Baattrup-Pedersen et al., 2013). The outputs produced are full coverage images and the pixel signatures are grouped into classes based on how similar the pixel value is and the output is also not subject to any introduced error by the user (Sotheran et al., 1997). Due the rapid nature of this technique (Liu, 2003), it is also useful in terms of automation in image analysis and can assist in assessing systems where reference networks are not available or accessible (Baattrup-Pedersen et al., 2013).

2.5.1.2 Supervised

Supervised classifications are user-defined objects and signatures (Sotheran et al., 1997). Unlike unsupervised classifications, the process does not work with individual pixels rather with objects defined by the user (Walter, 2004), and signatures that are defined with foreknowledge of the area and the analyst have sufficient pixel groups in order to generate constraints for the known groups (Liu, 2003). This is a more time consuming method as the objects need to be trained prior to analysis by the analyst (Walter, 2004). Training data is advantageous because it allows the use of pre-test images, allowing the analyst to test the statistical significance of the output, before running a full analysis and "wasting time"

(Wang, 1990). This is assuming that the test image used is of similar objects and terrain (Fraisse et al., 2001; Wang, 1990). This form of classification has even shown excellent performance in video analysis (Qi et al., 2003).

2.5.1.3 Limitation of classifications

The major difficulty encountered in classification techniques is an heterogeneous appearance in datasets, meaning overlapping signatures for two or more classes' means that error can be introduced as a feature may be defined into a class which it does not belong to (Walter, 2004). This is often seen when shadows, cloud cover or other factors influence the image quality, specifically in high resolution imagery, producing a reduced statistical probability of separability between two or more classes (Johnson and Xie, 2011). Classification methods are also static meaning that once a pixel has been placed into a cluster it cannot be moved into another (Omran et al., 2005).

This can be very difficult to overcome using a technique such as object based classifications through segmentation, basically a process of merging pixels to create a single object (Dorren et al., 2003) of an image, have been suggested as a possible solution (Dorren et al., 2003; Espindola et al., 2006; Johnson and Xie, 2011).

2.5.2 Decision Support Systems (DSS)

Decision Support Systems (DSS) are simply interactive computer based systems, used to plan and predict outcome of multi-influential and dynamic processes where no or difficult to separate aspects influence an area or environment. (Sprague Jr., 1980). Aimed at being as user friendly as possible (El-Gayar and Leung, 2000), they allow authorities and local developers to plan ventures by taking into account all variables including opportunities and limitations as well as allow the assessment of management strategies with a non-biased view (Rossetto et al., 2015). This is not only suitable for testing the influence of change on conservation and ecology (Adriaenssens et al., 2004) but also for crime prevention and patrol protocols (Camacho-Collados and Liberatore, 2015) as well as for aquaculture and mariculture practices (Bolte et al., 2000; El-Gayar and Leung, 2001; Ernst et al., 2000; Nath et al., 2000).

Any good DSS model will follow the guidelines set out by Jakeman et al., (2006) and Chen and Pollino (2012) where the following questions can be answered:

(1) What is the model's purpose?
(2) What resources are available for the scope of modelling?
(3) What specific data is available and is expert knowledge required?
(4) What are the features and families?
(5) What model structure can be used?
(6) What probabilities and statistical relationships will be used?

Once these questions are answered one can then:

(a) Run the model
(b) Validate and test the output
(c) Quantify the uncertainty
(d) Evaluate the accuracy and precision of the model and update and correct.

(see Jakeman et al., (2006), and Chen and Pollino (2012)).

2.5.3 Types of models and decision support systems (DSS)

2.5.3.1 Bayesian Belief Networks (BBN)

This statistical tool is often used in ecology and conservation management as it allows a user to predict the influence that variables such as environment or habitat have on the ecological response (Marcot et al., 2006). Among the strengths of this form of modelling is that it allows the analyst to solve complex problems through the combination of robust probabilistic methods and graphic outputs, while producing a distinct model that handles the uncertainty, unpredictability, imprecision and complexity of the problem (Dlamini, 2010). It is further strengthened by the ability of the model to combine and quantify different sources of data including empirical data, statistical, mathematical, and expert knowledge to ensure superior spatial representation (Stelzenmüller et al., 2010). Another advantage of this form of modelling is that it can use data-rich or data-poor applications and is easily understood and manipulated by non-modellers if properly constructed (McCann et al., 2006). Finally this form of modelling is also advantageous as the data-driven validation tools are broad (Landuyt et al., 2013).

This type of modelling is not without its own difficulties and problems. It has been seen that with increased complexity the algorithms often fail to compute the correct probability parameters (Lam and Bacchus, 1994). In a SWOT analysis Landuyt et al., (2013) stated that this type of modelling is limited in capacity as it lacks feedback loops and data discretization, and is also threatened by limitations in data availability, public acceptance and scientific-circle acceptance.

A basic BBN model runs off of decision nodes that the user defines (Landuyt et al., 2013; Marcot et al., 2006; McCann et al., 2006) The parent node defines the starting point while summary child nodes define intermediate decisions and child nodes define outputs (McCann et al., 2006). They most often rely on influence diagrams (Marcot et al., 2006) or decision trees (McCann et al., 2006), which is another advantage as they can be used in association with other types of models (Marcot et al., 2006; McCann et al., 2006).

Landuyt et al., (2013) indicated that two-thirds of the applications from publications between 2010 and 2012 were related to aquatic ecosystems research and BBN model applications while the remainder covered services such as water regulation, genetic resources, recreation, water supply and food provision.

2.5.3.2 Ecological Niche Models

The major strength of this form of modelling is the predictive nature of presence/absence that can be inferred (Okano and Kvitek, 2009). This type of modelling uses the difference between the mean and variation of the distribution of the cells representing species observations and the global cells (Skov et al., 2008). The mathematical model which was initially developed in 1917 by J. Grinnell in his paper titled “The niche relationship of the California Thrasher” and later refined in 1957 by G.E Hutchinson, defines the species niche using environmental conditions as a function that most likely drive its distribution (Brown et al., 2011). This means that good surveys of the study locality need to be done in order to ensure accurate predictions with this form of modelling (Reiss et al., 2014). The modelling technique relies on map assumptions that at any given locality the pixel is a estimated probability that the habitat is suitable or not for the study subject (Basille et al., 2008). This type of modelling is most successful when used with tracking data and has shown great promise in habitat predictions in marine investigation and research (Skov et al., 2008). These models tend to be avoided mainly because high complexity in species dimensions and small inadequate sample sizes don't allow for correct descriptions of niches over specific environmental gradients as more often than not these niches are statistically skewed or multimodal in shape (Reiss et al., 2014).

2.5.3.3 Other Multi-criteria analysis models (MCDA)

A number of models exist and can all be incorporated into multi-criteria modelling, including and not limited to Generalized Linear Models (GLM's), Generalized dissimilarity models (GDM's), Ordination techniques, Classification and regression trees (CART), Quantile regression models, and Artificial Neural Networks (ANN's) (Reiss et al., 2014). MCDA's are aimed at reducing costs, quantifying decisions and optimizing decisions for both stakeholders and decision makers (Huang et al., 2011). The aim of this type of system is to take multidimensional datasets, defined parameters; get stakeholder input and process through the powerful GIS processing interface without bias introduced by a decision maker (Radiarta et al., 2008). The popularity in these types of approaches started from the 1990s (where only a handful of papers had been published) to 2010 seeing an increase in publications to well into the hundreds (Huang et al., 2011). Successful projects have been seen worldwide, for example in the Mediterranean MCDA's were applied successfully to demersal fisheries (Rossetto et al., 2015), wind farms in Northern Jutland (Hansen, 2005), in aquaculture in japan with *Mizuhopecten yessoensis* (Scallops) (Radiarta et al., 2008) and even in policing in Spain (Camacho-Collados and Liberatore, 2015).

Among the downfalls of these types of systems is that there are aspects and parameters such future conditions of natural systems, risk, and human subjectivity, that cannot be adequately described in the decision-making processes and therefore introduce some uncertainty (Mendoza and Martins, 2006). These systems are very subjective of the user and this means that unless the model is designed for a specific purpose, testing and optimizing is somewhat infeasible (Tammi and Kalliola, 2014).

2.5.4 Fuzzy Logic Modelling for DSS

2.5.4.1 Advantages

Novák (2005) states that fuzzy logics major advantage is the significant improvement and advancement from Zadeh's publication in 1965 (Zadeh, 1965). It has helped to creatively solve non-standard problems that "classical mathematics" were otherwise not capable of solving, and has potential to be used in many different fields including artifical intellegence, robotics, biological modelling and medicinal fields (Zadeh, 1973). The idea of Fuzzy logic modelling was aptly described as "computing with words" by Zadeh (Novák, 2005). This lingustic description of a decision, which is described by a set of rules, has a number of advantages as it is easy to include qualitative information as well as make it easy for people to understand (Novák, 2012). For example a basic rule set would include commands such as "If, And, Or, and Then" and applied to a decision, it may look something like the following:

1. "**IF** the weather is sunny **AND** the wind speed is lower than 5 knots **THEN** the beach will be nice."
2. "**IF** the seaweed avaliable is low **OR** the depth is more than 8 metres **THEN** abalone abundance will be low"

Another aspect which makes fuzzy logic unique is the use of precision. According to Zadeh (2008) this can be seperated into two forms; namely a Valued-Precision (v-precise) and the Meaning-Precision (m-precise). This has major consequences for the outcome of the memberships as while v-precision may be paradoxical when applied to incompleteness, imprecision, bipolarity, uncertainty and vagueness of a membership function (Dubois et al., 2005). M-precision on the other hand allows for granulation, meaning that a range holds some level of precision when defining a membership (Zadeh, 2008).

There is growing support for the use of fuzzy theory in modelling for ecological and habitat suitability modelling as it provides a robust model that can mathematically treat a wider range of phenomena (Barros et al., 2000). It has successfully been used in various applications for industrial, economic and environmental sectors with a variety of subjects and various terms defining the memberships (Mardani et al., 2015b). It is also the most appropriate method in describing factors where boundaries are difficult to define (Nath et al., 2000).

2.5.4.2 Disadvantages

Many of Zadeh's colleagues argued that this form of mathematics is illogical and has no use in science (Zadeh, 2008). They argue that one of the greatest downfalls of Fuzzy logic is its extreme subjectivity. This is due to the mathematical framework incorporating human influence and subjectivity (Cornelissen et al., 2001). An expert can be defined as a person with a specific degree of depth, exposure and experience in a specific area of interest (Krueger et al., 2012a). An implication of this is that there is a level of unreliability in claims that cannot be justified or statistically supported, and that may be influenced by political and social pressures (Krueger et al., 2012a).

Expert knowledge also implies a degree of learning and exposure to the environment or habitats that an individual claims to be an expert in Krueger et al., (2012b); but also is a pragmatic description of any opinion that holds interest (Krueger et al., 2012a). This can take many forms, for example scientists, or land managers (Page et al., 2012) as well as many; communities, including their traditional, cultural and historic opinion, passed down from generation to generation (Nursey-Bray et al., 2014). To use an example: while a subsistence fisherman may not have a formal education, he is an expert in his field of fishing as he knows where to look for the fish, what bait to use, and what environmental factors (pressure systems, rain, wind, temperatures, etc.) to look for in order to maximize catch success as this knowledge has been handed down to him from his forefathers.

The theory of conventional fuzzy set logic uses a degree of membership or belongingness to a set to represent and treat uncertainties and imprecisions within the set (Grzegorzewski and Mrówka, 2005; Zadeh, 1965). This is a quantifiable characterization of uncertainty phenomena or in more layman’s terms the probability (a numerical measure) that the likelihood of a particular event or occurrence of a phenomenon will take place if certain drivers or attributes are observed (Novák, 2005).

Through the expansion of fuzzy theory Atanassov in 1986 proposed the term “intuitionistic fuzzy sets” and is a generalization of natural fuzzy set theory (Dubois et al., 2005). Dubois et al., (2005) and Grzegorzewski and Mrówka (2005) both criticised Atanassov’s theories due to terminology that supposedly has brought much confusion to theoreticians and practitioners where incomplete or inaccurate sets (Grzegorzewski and Mrówka, 2005) can directly impact the handling of imprecision, irrelevance or bipolarity, uncertainty and vagueness for memberships (Dubois et al., 2005). In general fuzzy logic also has been criticized for having no clear method of approach in defining the notations and direction of research (Běhounek, 2008).

2.6 Sampling Methodologies and protocols

2.6.1 Quadrats

2.6.1.1 Size and Shape

Quadrats are a commonly used tool in ecological and biological studies, whether for benthic marine invertebrate studies, seaweed and algal community studies, and coral community surveys (Dumas et al., 2009; Leujak and Ormond, 2007; Renken and Mumby, 2009; Zakai and Chadwick-furman, 2002). Quadrats have been deployed both temporarily and permanently to assess community structure and relationships (de Waal, 2005; Won et al., 2012, 2007), monitor effects and changes to rocky and subtidal areas (Hart et al., 2008; Murray et al., 2001; Strain and Johnson, 2012; van Rein et al., 2009; Whittaker et al., 2001) as well as for random sampling across an region to develop an estimation of species richness and diversity (Griffin et al., 1999; Mccormick, 1994; Roberts et al., 2014; Toohey et al., 2007; Won et al., 2007).

One challenge in choosing a quadrats size and shape is that species composition can change seasonally and spatially (Krebs, 2014a). Quadrats are among the most commonly used method to assess benthic habitats (Pringle, 1984). They have been used to sample many different reef types, ecosystems and habitats (Table 1). Their size is selected according to the study objectives and the environment to be studied, which is often assessed by pilot studies in the region (Phillips et al., 1997). Quadrats and transects have for a long time been the main method for sampling marine benthic habitats, and every study differs in the selection of quadrat size (Table 1). However all seem to agree on rectangular or square quadrats. In the selection of quadrat size and shape careful consideration needs to be given to the sampling units of the spatial arrangement (Kenkel et al., 1989).

Table 1: Different studies that have been used to study benthic reef biota using transects and quadrats.

Reef Type	Transect Length (m)	Transect width (m)	Transect type	Quadrat size (m^2)	Placement	Shape	Author
Coral/ Sub-Tropical				0.0625	Random	Square	Anderson et al., (2005)
Antarctic	7 to 18		Line	0.017-0.32 (P)*	Selective	Rectangle	Bowden (2004)
Coral/ Tropical	50 and 100		Line	0.25	Selective	Square	Calumpong et al., (1999)
Warm Temperate	Surveyed a maximum area of 10.6 m^2		Belt	0.17 (P)	Selective	Rectangle	Celliers et al., (2007)
Cold Temperate	30	4	Belt				Coates et al., (2014)
Cold Temperate				0.0625	Random	Square	Day and Branch (2002a)
Coral/ Sub-Tropical				0.25	Random	Square	Diaz-Pulido and Garzón-Ferreira (2002)
Cold Temperate	10	2.5	Belt				Götz et al., (2009)
Temperate	30		Line	1	Selective	Square	Hart et al., (2013a)
Coral/ Sub-Tropical	50 and 100	1	Belt				Kenyon et al., (2006)
Cold Temperate				0.25	Random	Square	Leliaert et al.,(2000)
Tropical	50	.45	Belt (video transects)				Leujak and Ormond (2007)
Temperate				1	Selective	Square	Levinl and Hay (1996)
Coral/ Tropical	50		Line	9	Selective	Square	Mccormick (1994)
Temperate	30	2	Belt		Random		Raemaekers and Britz (2009)
Temperate	50	2	Belt	1	Random	Square	Ruitton et al., (2000)
Temperate	30		Line	1	Random	Square	Proudfoot et al. (2006)
Cold Temperate/ Antartic				0.25, 1, 1.56, 2.25, 2.99 and 4 m^2	Random	Square	Pringle (1984)
Coral/ Sub-Tropical	10	2	Belt	0.0625	Selective	Square	Sangil et al., (2011)
Temperate				1 and 4	Selective	Square	Won et al., (2011)
Temperate	10			0.0625	Random	Square	Wood (1993)
Temperate				0.0625	Random	Square	Wood and Buxton (1996)
Coral/ Tropical	75		Line	1	Random	Square	Zakai and Chadwick-furman (2002)
Cold Temperate				0.25	Random	Square	Zeeman et al., (2012)

*(P) stands for Photographic quadrats

Another important factor driving the selection of optimal quadrat size is the precision (Phillips et al., 1997; Pringle, 1984). While a larger quadrat size may cover a greater area, environmental heterogeneity will also increase as a result, meaning that spatial pattern interpretation will also increase in difficulty (Kenkel et al., 1989).

Nested quadrats are useful in the production of species area curves, although there is some doubt about the accuracy of the nested quadrat method in terms of precision and it remains highly controversial (Krebs, 2014a). Another potential limitation of nested quadrats is that species are usually not uniformly distributed in space (Scheiner, 2003) although the use of a nested quadrat can play an important role in selecting the optimum quadrat size to study a selected habitat or ecosystem (Krebs, 2014a).

Randomly placed quadrats however are likely to encounter a greater number of species than a nested quadrat over a larger scale (Scheiner, 2003). Few species in nature are aggregated randomly, and a general pattern seen is a clumped pattern (Krebs, 2014a) or non-random pattern (Scheiner, 2003). It has been suggested by Kenkel et al. (1989) that spatial patterns of species can be detected using plots and quadrats placed at random across a study area. Random quadrats have been well employed in many marine benthic studies including Pringle (1984); Wood (1993); Leliaert et al., (2000); Day and Branch (2002a); Day and Branch (2002b); Diaz-Pulido and Garzón-Ferreira (2002); Anderson et al., (2005); Proudfoot et al., (2006) and Raemaekers and Britz (2009).

2.6.1.2 Photographic versus Visual Sampling Methods

More recently photographic quadrats have also been favoured (Murray et al., 2001; Smale et al., 2010; van Rein et al., 2009) but with some constraints. The advantage of quadrats is that they are a function of the degree of dispersion of a species over a spatial scale (Pringle, 1984). Quadrats are regularly used for fine-scale assessments and are coupled with transects to assess a larger-scale pattern (Beenaerts and Berghe, 2005). The most optimal quadrat size is that which provides the smallest value in terms of relative cost and variability, meaning what is the cost to time ratio for a quadrat size and what is the information that the quadrat will give you per deployment (Krebs, 2014a). Pringle (1984) found that there is a curvilinear relationship between the total sampling time, sampling unit size and the sampling precision.

Among the advantages of the use of photographs is that they can be used at many scales, however as a top-down method they tend to overlook and often distort crevices and overhangs, and therefore these factors need to be accounted for in another manner (Wilding et al., 2007). Furthermore the advantage of photographic imagery and video is that it does not require trained personnel to operate the equipment (Roelfsema et al., 2006). Commercial divers can follow simple procedures and protocols developed by the researcher and then the imagery can be analysed by the trained personnel at a later stage back in the laboratory (Hart et al., 2008). Trained individuals can be used when towing a ROV but for sleds and other towing devices training can be minimal (Dumas et al., 2009; van Rein et al., 2009).

Another advantage of using imagery is that it is non-destructive and can be deployed in a number of ways including remote operated vehicles, diver operated and towed-sleds (Brown et al., 2007). Another form of non-destructive sampling that is often used with digital methods is visual census but, according to Hart et al., (1997), this methodology by itself is less cost effective when compared to the imagery techniques.

One of the major problems associated with the hands on approaches and high sampling effort of the above discussed quadrats and transects, is that this may come with increased costs and time constraints (Dumas et al., 2009). Benthic evaluation methods can be restrained further by aspects such as safety, water clarity and depth, currents, remoteness and logistics (Roelfsema et al., 2006). The use of digital sampling has provided a non-invasive and rapid method by which a diver can spend minimum time in the water and the researcher can use methods to analysis the video and images.

2.6.2 Transects

2.6.2.1 Type and Length

Line transects have been well used for reef and benthic sampling (Beenaerts and Berghe, 2005; Katsanevakis, 2007; Krebs, 2014b). Most commonly the line intercept method is used, but is limited as it is difficult to account for all species along a single transect, thus often producing underestimates (Krebs, 2014b). However variations of transects have been used that help to overcome this underestimation, including the use of belt transects. Belt transects are often difficult to sample mobile organisms (Kulbicki and Sarramégna, 1999) but have been proven successful in abalone census counts for population estimates (Hart et al., 1997).

Optimum length of transects, like quadrat size and shape is a product of the environment to be studied (Hart et al., 1997). The optimum lengths have long been debated with authors such as Mccormick (1994); Calumpong et al., (1999); and Ruitton et al., (2000) using transects of 50 metres for sub-tropical and tropical water, whereas Proudfoot et al., (2006); Raemaekers and Britz (2009); and Coates et al., (2014) who worked on temperate systems used 30 metre transects while other authors have used between 7 and 18 metre transects Wood (1993); Bowden (2004); and Götz et al., (2009). This issue of optimum length remains debated but it is agreed that, as with quadrat size, transect length needs to be dictated by the environmental conditions and research objectives associated with the specific research topic (Krebs, 2014a, 2014b).

2.6.2.2 Video

Among the greatest concern for still and video imagery is image quality (Murray et al., 2001). Visibility, turbidity (Brown et al., 2007) and scale (Bowden, 2004) all play an important role in being able to identify and quantify the species in the image. Poor visibility and high turbidity are often the major limiter for the use of imagery in marine benthos surveying (Brown et al., 2007; Celliers et al., 2007; Silvert, 1997), and sampling is limited to visibility that is greater than or equal to the focal point of the camera and recording devices used (Celliers et al., 2007). While there is no specific scale for monitoring ecological processes (Somerfield and Gage, 2000), it is important to consider the scale based on the environment that is being assessed. Often video is the preferred method of sampling, as it is efficient, accurate, can cover larger spatial extents, and is normally very simple to do (Houk and Van Woesik, 2006).

According to Leujak and Ormond (2007) video is among the more appropriate methodologies for monitoring programs that cover large scale areas, as it is fairly efficient and accurate. Many studies i.e. Brown et al., (2004); Tkachenko (2005); Houk and Woesik (2006); Leujak and Ormond (2007); Hart et al., (2008); and van Rein et al., (2009) have used video with varying success. For instance Brown et al., (2004) found that video was the most cost-effective at remote sites, whereas Hart et al., (2008) found that in the haliotid industry video techniques are sufficient for relative estimates of abundance but fail to provide accurate estimates.

2.6.3 Measurement of Rugosity

Rugosity is a measure of surface roughness for rocky substrates (Murray et al., 2001). Rugosity gives an empirical measure of the availability of holes, crevices and overhangs by comparing actual surface area to the planar surface area (Lucieer et al., 2013). However rugosity is scale dependant, as explained by Brown et al., (2007), who indicate that rocky/boulder habitats between 8 and 50 metres in length, have the greatest rugosity.

2.6.3.1 Chain and Tape

The use of the “Chain and Tape” method is often the most widely applied and cost-effective small-scale field method that can be employed by SCUBA divers (Pais et al., 2013b). The Chain and tape method is a basic method whereby a chain is laid along the bottom and a straight line measure is taken so that a factor is produced (Pais et al., 2013b).

2.6.3.2 Other Methods: Side scan sonar and Multibeam Sonar

Between the 1940s (Silva, 2003) and 1960's (Johnson and Helferty, 1990) the revolutionary development of side scan sonar (SSS) allowed for new frontiers in oceanic seafloor research to occur. Even though these early systems were low in resolution and relatively unreliable, except when searching for large objects on the ocean floor, such as shipwrecks or large oceanic features (Silva, 2003). The greatest advantage of SSS is that it is a rapid electronic and digital technique that uses the efficiency of the medium to generate and data correct image, that can be modified, processed, enhanced and analysed resulting in cheap, rapid and wide-area data on substrate type and composition (Johnson and Helferty, 1990). Another advantage of multibeam and SSS is that it produces, at small scales over large areas, mapping capabilities that can provide depth, rugosity, and backscatter data continuously.

A basic SSS consists of three main components, as shown in Figure 3, the topside display and recorder, the tow cable and the tow fish and transducer (Henriques et al., 2012; Marine SonicTechnology, 2011). The transducer is the component that is made-up of an array of acoustic units and transmits the sound (often referred to as the ping) and can be positioned either in the tow fish, which is a streamlined unit that houses and protects the transducer (Marine SonicTechnology, 2011), or can be fixed to the hull of the boat (Henriques et al., 2012). Topside is the unit that displays and records what is transmitted from the transducer via the tow cable.

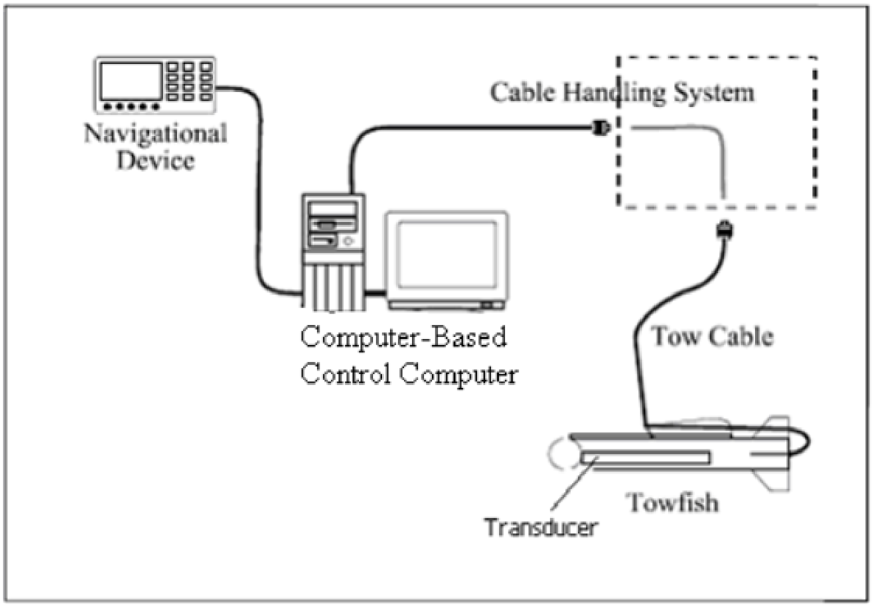

Figure 3: components for side scan sonar obtained from Marine SonicTechnology (2011)

2.6.4 Accuracy versus efficiency

A major factor that any study needs to take into consideration is the cost and time efficiency of a method. The most time-limiting factor of any subtidal and benthic sampling is the limited availability of air to the diver as well as the effect of cold and fatigue on a diver (Beenaerts and Berghe, 2005). Commercial diving has been a preferred method for many invertebrate commercial fisheries globally. For example the scallop fishery in Argentina which historically was done using dredging methods, now uses commercial divers and has developed a unique relationship between the local community, scientists and innovative fishermen to develop habitat-friendly harvesting procedure, it has also limited the rate of harvesting per individual (Orensanz et al., 2007).

Quadrats and transects require a large amounts of input energy (they can be very laborious), to obtain sufficient data. How accurate and worthwhile this data may be, is strongly influenced by the competence of the diver completing the dive, the time spent on each quadrat, the size of the quadrat or length of the transect, the environmental conditions of the day and the methodology applied to the sampling effort (Hart et al., 1997; Hart and Gorfine, 1997). It has been shown that precision for transects can be relatively high (Hart and Gorfine, 1997). The general assumption in any study is to obtain a balance between the time-spent sampling and obtaining as accurate data as possible.

Chapter 3. Development and implementation of a decision support system (DSS) for the Wildcoast abalone ranching project using GIS

3.1 Introduction

Introduced in the 1960's, Geographic Information Systems (GIS) have quickly grown in popularity for spatial analysis (Goodchild, 1993). The first case of commercial application was recorded in the early 1980's (Goodchild 1993; Pérez et al. 2003). GIS had limited influence in the sector until the early 2000's (Pérez et al., 2003). Today GIS has become important in the planning and operational phases of many aquaculture ventures (Bolte et al., 2000; El-Gayar and Leung, 2001, 2000; Longdill et al., 2008; Stewart et al., 2010). The technology has allowed for detailed and accurate spatial planning of farming areas, improving the ability of the stakeholders to make informed decisions on aspects such as site selection and likelihood of success in particular areas (Pérez et al., 2003). The power of this tool is still developing as new functionality is being developed (Falconer et al., 2016; Silva et al., 2011).

GIS software has become more user-friendly and interactive, increasingly so with the web-based systems currently in operation (Carver, 2001), and the development of software and applications for specific purposes such as environmental monitoring and management (Tsou, 2004), farming and crop management (Rao et al., 2007; Tayyebi et al., 2016), and development, with the benefit of allowing public participation processes to take place through the application (Simão et al., 2009). They can manipulate and do basic analysis on spatial datasets that can assist planners and experts in their decision making processes. A trained GIS expert can use more sophisticated software to analyse, measure and predict a large range of phenomena, which includes relationships, event probabilities and distributions (Zlatanova et al., 2002). It has helped to satisfy the need for pre-processing, modelling and analysis, and post-processing results within several disciplines, providing information that is used by decision-makers across various professions (Aswani and Lauer, 2006; Gold and Condal, 1995; Pérez et al., 2002; Stelzenmüller et al., 2013; Tobergte and Curtis, 2013). Furthermore, GIS has allowed greater interaction between researchers and stakeholders. For example in Australia, Mayfield et al. (2011) engaged with resource managers, scientists, industry partners and the general public to create a sustainable abalone fisheries management plan and develop a concession to study areas to ensure periods of stock recovery. Stelzenmüller et al. (2010) engaged with stakeholders of the general public to include their views with the marine spatial plan as well as assist in rapid updates of spatial relations. Most often multiple stakeholders, including local, industrial and research interests, are part of the management plan, and each has their own objectives, views, expectations and engagements, which need to be met or need to come to mutual agreement suitable to all participants and stakeholders (Mendoza and Martins, 2006). Engaging with stakeholders is of critical importance for aquaculture operations. However, this can be challenging as the needs and ideas can vary among stakeholders, and different mind-sets of the parties involved determine their understanding and rationalization of the systems in different ways (Wever et al., 2015).

Aquaculture has grown rapidly in the past decade (FAO, 2014), and along with this a need has arisen to utilize available and suitable space appropriately (Cheung et al., 2005; Elkan et al., 1994; Hattab et al., 2013; Navas et al., 2011). Another aspect that has developed along with the growth in aquaculture is the requirement to monitor and evaluate aquaculture activities (Adriaenssens et al., 2004; Gray et al., 2014; Lu et al., 2012). The use of modelling techniques for multi-criteria decision-making, such as fuzzy logic modelling, has grown in popularity (Adriaenssens et al., 2004; Gobi and Pedrycz, 2007; Gray et al., 2014; Lu et al., 2012; Mardani et al., 2015a; Navas et al., 2011; Takagi and Sugeno, 1985). These techniques make it possible to incorporate the element of uncertainty in site selection, environmental management, and predictive modelling processes, as well as allowing the user to weight and define the importance of certain factors in a subjective manner (Environmental Systems Resource Institute (ESRI), 2009). The modelling process is systematic, with each step constrained by a number of mathematical rules, which either define the output values as part of the set, or the degree to which they would likely be in the set or not (Olaru and Wehenkel, 2003).

While the Cape Recife ranching program was initiated in 2014, the development and preliminary testing of the fuzzy logic modelling system in this research was only implemented in mid to late-2015. Prior to this implementation, plots for seeding abalone were selected by a trained diver or biologist under the assumption made by stakeholders that the Cape Recife sector of the concession zone was the most suitable based on the fact that it was located close to the Tactical Task Force base who protect the abalone, and its accessibility as it was central between Noordhoek Skiboat club to the west and from the Port Elizabeth Harbour to the North East.

The main aim of this research was to map and prioritize concession area 2 for potential abalone ranching operations. It was further to construct a modelling support system that when implemented provided stakeholders with a tool set that assists in the decision-making of selecting sites. It was developed and implemented to alleviate the issues of extra expenses and the use of divers to search for suitable reef in the concession zone. The above approach has been used to develop a model that incorporates stakeholder requirements, including both the industry partner and the collaborating universities. It also takes into account the environmental conditions and identifies suitable seeding-plots for the abalone ranching program at Cape Recife. Data for the model was obtained through field collections, expert-knowledge, stakeholder input and remote sensing imagery. The procedures used and results obtained in this modelling process are outlined in this chapter.

3.2 Methodology

3.2.1 Imagery acquisition, correction and enhancement

3.2.1.1 Google Earth

Satellite images were obtained from Google Earth Pro V7.1.5.1557. The identification and quantification of sand and reef substrata with the aid of GIS requires images that were captured on clear sunny days, when the sea in the area of interest (Cape Recife ranching area, Port Elizabeth) was reasonably calm, with low turbidity and minimal surface turbulence. Google Earth provides access to archived historical satellite images for various dates. On many dates for which images are available the Cape Recife area had poor sea conditions, high seawater turbidity, or cloud cover. The archived images were reviewed until the most suitable images were obtained. A single group of images was selected from 08-05-2014. The zoom function was used to make each individual image a 1:54.2 cm scale. Images were then marked with place-markers for later geo-referencing and saved as a JPG file at the maximum resolution of 4800 X 2718. The Place-marker positions, which in the simplest form is an icon (default is a yellow pin) or symbol with a geographic location defined and can have a custom name given by the user (Google Developers, 2016), were saved as a KML file and later imported into ArcMap.

3.2.1.2 Colour correction and enhancement

All JPG files were colour corrected and enhanced using Adobe® Photoshop® CS5 Extended Version 12.0 64. Files were rapidly processed using the *image processor* and *actions* toolset. All files were saved in a new directory in the same format as the input file. The Image processor followed a pre-configuration of Levels adjustment for contrast, curves adjustment and exposure adjustments.

3.2.1.3 ArcMap, Georeferencing and Extent

The KML file was imported into ESRI ArcMap V10.2 and placeholders (spatially referenced point features in google earth) plotted as vector points using the KML to Layer conversion toolset. Each image was then georeferenced using the placeholder positions and the spatial referencing (the coordinate system used was WGS 84 UTM 35S was updated for all images. The combined extent of the images covered the entire experimental abalone seeding concession area (Figure 4) held by Lidomix Investments.

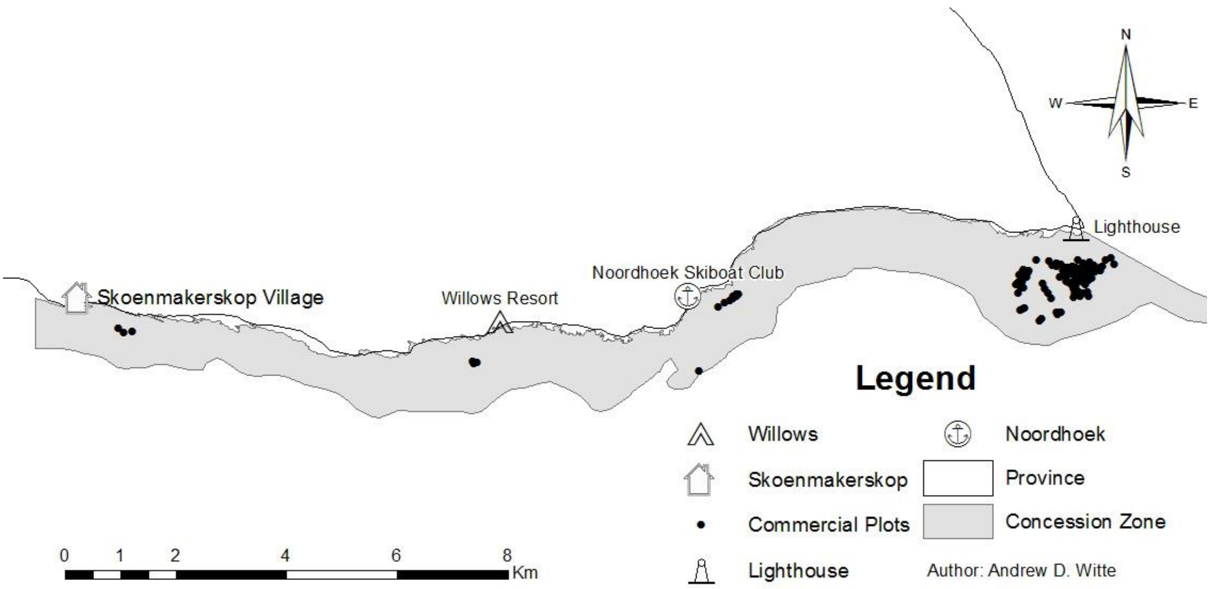

Figure 4: The concession area and nodes within.

The area is defined in Government Gazette No. 33470, and is bordered by the marine protected area westward of Beacon PECR1, Skoenmakerskop (Table 2). Each image was set to a stretch type of "*none"* in the symbology tab.

Table 2: The co-ordinates for the concession areas according to Government Gazette No. 33470

Code	Area	Latitude	Longitude
EC 1a	Skoenmakerskop MPA	34° 2' 46,05" S	25° 32' 33,39" E
EC 1b	Cape Receife	34° 2' 0,33" S	25° 42' 18,43" E

3.2.2 Mosaic dataset construction and Masking

A *Mosaic dataset* was created using the create mosaic dataset data management tool set. The georeferenced images were then added to the dataset using the add raster to mosaic dataset toolset. The dataset was then masked with a polygon feature, using the "*Extract by mask*" into an eastern and western sector covering the extent of the concession area. Each sector was analysed separately due to variation in pixel colour signatures between sectors, i.e. the western sector had colours of a lighter signature for the same features as the eastern sector, and caused confusion and pseudo-identification of features. This was particularly due to the different time and angle of the images taken when the satellite was over the specific section of coast.

3.2.3 Image Classification for Substrate

Substrate was distinguished on the basis of image colour, with darker areas representing reef, light areas sand, and mixed-substrate exhibiting spectral characteristics between these two extremes. The mosaic and individual images were processed with the "*image classification toolset"*, using an unsupervised classification. The number of classes was set at 30 classes for the classification with a minimum of 200 cells to validate a class. This was chosen due to the similarity of signatures of classes, for example deep areas give a similar signature to the shallow reef areas. This is a well recorded problem with classifying reef systems (Bouvet et al., 2003; Chauvaud et al., 1998; Congalton, 1991) and it is recommended that the use of more classes than necessary (over-estimation) is applied to capture the complexity of the system being assessed (Bouvet et al., 2003; Chauvaud et al., 1998).

A trained dataset under supervision may have rendered a more accurate result however the time constraints of producing the classification of the substrate as an automated approach such as the unsupervised classification decreased the analysis time of the approach (Chauvaud et al., 1998). This allowed for different spectral groupings, which mainly represented reef, mixed substrates (where each class represents a different degree of sand and/or rocky habitat), and sand. The captured Google Earth satellite images, of the eastern and western sectors of the concession area, showed consistent differences in spectral characteristics. In order to accommodate this, sub-sectors were created for the analysis of the eastern (Concession Area 1) and western (Concession Area 2) extent of the study area respectively.

3.2.4 Other Factors

While the modelling process relied mainly on the substrate layer, other layers were also utilized in the analysis. This included distance factors such as proximity to launch site, as well as factors of proximity to established infrastructure, and security considerations. A secondary sub-model was also created for the Cape Recife section of the concession area using physical factors of rugosity and depth, as well as a proximity factor of space between established plots. These layers were initially all captured as vector layers and were later rasterized in order that each image was maintained at pixels that represented 18 X 18 metres to employ in the modelling process.

3.2.5 Membership functions

The rasterized files from each of the above mentioned layers were used to produce membership output raster files. This was completed using the built-in ESRI ArcGIS fuzzy modelling functioning toolset. Each membership function is a set of rules that mathematically governs the output raster, and each raster file is a factor in the decision making process of the ranching program, whereby abalone are released onto the reef to grow and later be harvested according to a quota applied by the Department of Agriculture, Forestry and Fisheries. The output raster of the membership function indicates the relative importance, either negatively or positively, of the factor's influence on the concession area and influence on the decision of where to establish and release abalone onto the reefs. In order to structure the decision-making pathway a decision tree was constructed (Figure 5) and was used to indicate the importance of each factor on the decision of plot selection, as well as indicating the type of membership that was used.

The membership functions include:

I. The *Gaussian response* curve, also referred to as a simple bell-shaped response curve (Austin, 1980), is a normal distribution curve (Environmental Systems Resource Institute (ESRI), 2009). The curve is often symmetric and representative of features with smooth non-zero ranges (Zhao and Bose, 2002). This membership type is often used where mid-range values in a set hold the highest likeliness of being in the set (ESRI, 2009). The response can be represented by the following equation:

$$\mu(x) = e^{-f_1+(x-f_2)^2}$$

Where:

f_1 – the spread, where f_1 = [0.01, 1], and as $f_1 \rightarrow$L, then the steeper the distribution around the midpoint (f_2). ArcGIS defines the default spread as 0.1.

f_2 – the midpoint; a user defined point where $\mu(x)$ = 1. ArcGIS defines the default midpoint (medium) by the range of the values for the input raster.

II. The *near function* is similar to the Gaussian response in that it has a midpoint and spread that is defined in the syntax, however the equation is very different (ESRI 2009):

$$u(x) = \frac{1}{1 + f_1 \times (x - f_2)^2}$$

III. The *large function* according to ESRI (2009) would be suitable where input values, that are large in the range, are most likely to be in the set. This can be defined by the following equation:

$$\mu(x) = \frac{1}{1 + (\frac{x}{f_2})^{-f_1}}$$

IV. The *small function*, like the large function, also gives suitability for input variables to a set of small numbers in a range. The equation is similar to the large function but the spread is not negatively powered:

$$\mu(x) = \frac{1}{1 + (\frac{x}{f_2})^{f_1}}$$

V. The Fuzzy (mean and standard error large and mean and standard error small: *MSLarge* and *MSSmall*) are functions that produce sigmoidal curves where the input ranges are manipulated using the mean and standard deviation and select for values that are large for MSlarge and small for MSSmall giving them high likelihood in the set. These equations are in two parts where:

For *MSLarge*

If $x > \mu \times \sigma$

Then $u(x) = 1 - (b \times \sigma)/(x - (a \times \mu) + (b \times \sigma))$

Or $x \leq \mu \times \sigma$

Then $u(x) = 0$

And for *MSSmall*

If $x > \mu \times \sigma$

Then $u(x) = (b \times \sigma)/(x - (a \times \mu) + (b \times \sigma))$

Or $x \leq \mu \times \sigma$

Then $u(x) = 0$

VI. The *linear function* uses the straight-line equation to calculate the input variables likelihood of belonging to the set. The syntax for this function is:

FuzzyLinear (minimum, maximum)

Where, if minimum = 0 or is > maximum, then $u(x)$ will negatively correlated to the set.

and

maximum = 0 or is > minimum, then $u(x)$ will be positively correlated to the set.

The regression model used:

$$u(x) = mx + c$$

Where:

m: is the gradient of the regression line

x: is the range between and including, 0 and 1

c: Is the intercept of the y-axis where the membership will equal $u(x)$

The linear function works on a progressive scale, where the value $u(x)$approaches either the maximum or minimum values, so the suitability value will increase or decrease according to whether the correlation is positive or negative.

3.2.5.1 The Decision tree

As part of the modelling process decision trees are often constructed to assist the analysis and decision making process. This is a logical and sequenced set of choices, each with its own weighting that must be made in order to determine the most suitable decision (Nath et al., 2000). Rather than working on crisp values (any real number or integer), the use of sets can enhance a decision process, as well as allow the analyse of immense volumes of incoming data, as at each decision step criteria are met and satisfied, eliciting a particular and necessary response (Olaru and Wehenkel, 2003). An example of this in everyday life would be someone running on the street and seeing an object in the path which they must choose to either run past, stop or collide with. The individual may or may not have quantitate data, i.e. object size, speed of travel, etc., to support their decision but the use of sets apply here, as the probability of the person choosing to collide with the object is low i.e. closer to 0%, and the probability of choosing to avoid the object is high, i.e. closer to 100% and in the case of stopping the likelihood is higher than colliding but lower than avoiding the object. The membership process allows opportunity for stakeholders to comment and remark on the decision making process of the model. Figure 5 shows the process of selection and elimination, from creating the rasterized layers to the membership layer construction, the membership layers overlay to create sub models and ultimately the final overlay process to create the output model of influence.

This is a top down approach of decision making whereby at each stage of the modelling process the number of "*influencing factors*" (or nodes) decreases but the weighting of each increases (de Siqueira Campos Boclin and de Mello 2006). Each node is connected by an internode that indicates the direction and processes that take place to produce the output or candidate nodes, most commonly referred to as son-nodes (Wang et al. 2000). These son-nodes undergo more processing using fuzzy overlay methods and produce the sub-final, which is processed one step further, and a final output node (Father-node) is produced (Yuan and Shaw 1995; Wang et al. 2000; Olaru and Wehenkel 2003).

3.2.6 Vector layers

3.2.6.1 Model Layers

3.2.6.1.1 Concession area and sub-areas

The Concession area was constructed as a polygon, using the coastal outline from the high-water mark and the 10 m Isobath. The eastern concession area boundary was defined according to Permit No. 1503759 and the western most boundary as defined in the Government gazette No. 18930, Notice No. 747., is the 180° true bearing mark from beacon PECR1 near Skoenmakerskop (Figure 4).

3.2.6.1.2 Substrate

Substrate layers were obtained using the classification method described in section 3.2.3. The classification output layer is always in raster format and therefore did not require a rasterization process.

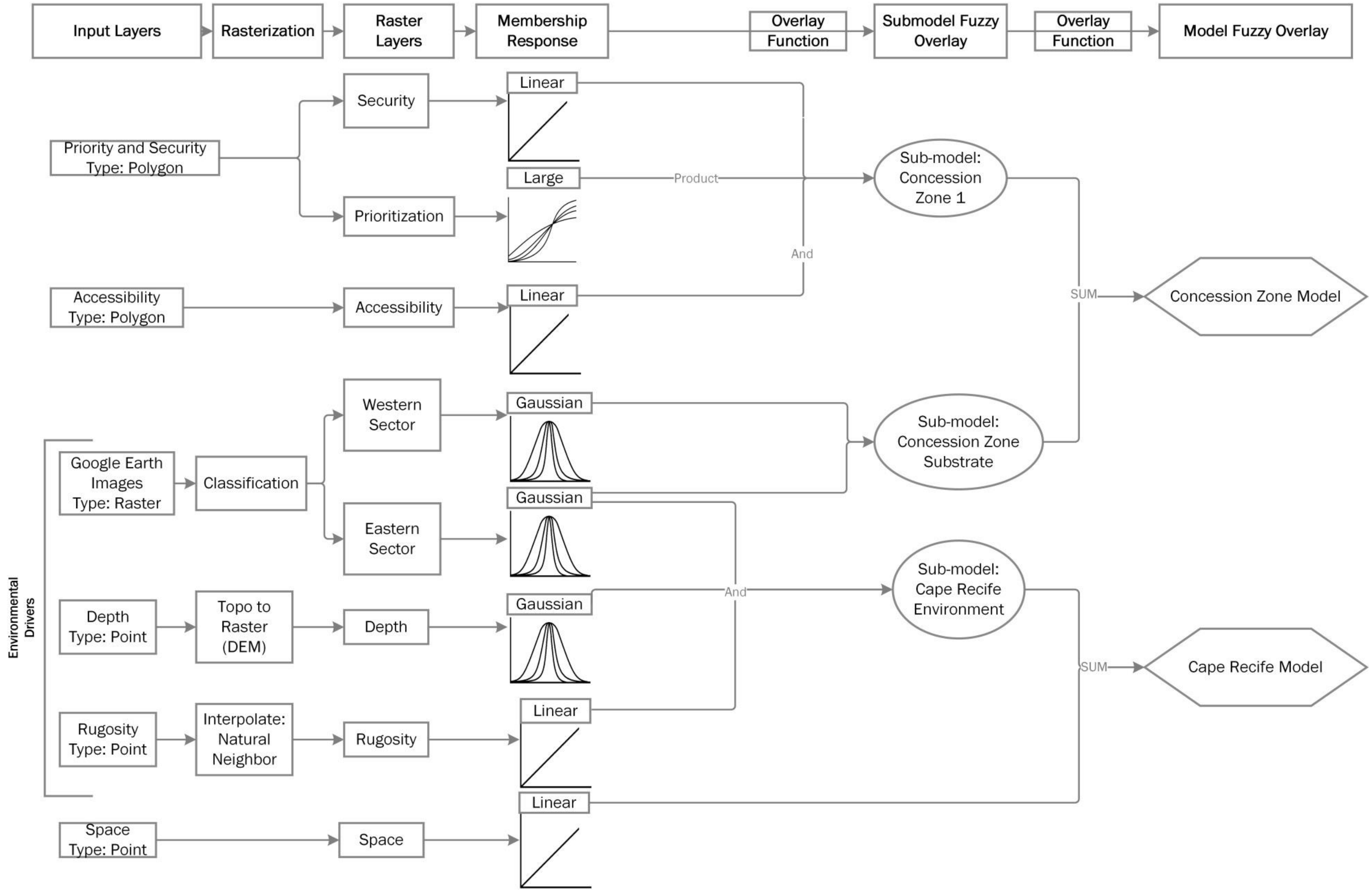

Figure 5: Decision Support Tree - The decision making model process showing the different layers and factors that influence the final output model. (based on de Siqueira Campos Boclin and de Mello (2006))

3.2.6.1.3 Priority and Security:

An important consideration for the ranching program is the accessibility of the ranching sites and degree to which the sites can be provided with security (securability) from poaching or other interference. These aspects have a strong spatial component, which includes the proximity of the ranch site to vessel launch sites, as well as line of sight and sight distance between ranched sites and surveillance observation posts. The stakeholders (Lidomix Investments and Wildcoast Abalone) identified four places along the coastline in the concession area from which they could provide surveillance and secure. These nodes were represented as point features for the Cape Recife (the stakeholders primary focus area), Noordhoek Ski-boat club, Willows holiday resort and Schoenmakerskop village.

A two kilometre buffer was placed around each of these nodes (see Table 3) under the assumption that the average day allowed one to see 2 km from the node. The resulting buffer areas were clipped so that only portions of the shoreline and shallow inshore regions visible from the sites would be selected i.e. clipped where headlands caused a break in “line of sight”, or if the shoreline created a small bay like area. This was determined under the assumption that from the nodes a view of a 180° could be obtained at the node, with the exception of Noordhoek which had a line of sight of less than 180° due to the headland due west. These nodes were then ranked from one to eight, ranging from lowest to highest importance respectively, as nodes with high elevation infrastructure, good line of sight and close proximity were considered to be higher in importance and security than sites that were further away and out of the line of sight.

Table 3: Co-ordinates of nodes identified by stakeholders for the abalone ranching project.

Node	Latitude	Longitude
Cape Recife	-34.029	025.701
Noordhoek Skiboat Club	-34.040	025.638
Skoenmakerskop village	-34.041	025.539
Willows Resort	-34.044	025.607

3.2.6.1.4 Launch Site and accessibility

A concentric buffer rings enclosed 500-metre-wide strips and extended outward to a maximum radius of 10 kilometres from the Noordhoek launch site was created to represent the accessibility of the area, i.e. a site of 500 metres from the launch site is considered more accessible than one 10 km away. The multi-ring buffer was applied to the waypoint mark of the slipway at Noordhoek Ski-boat club as this represents the only launch site on the coast in the concession area and is the most central launch site for the area. Port Elizabeth harbour was not included in the analysis due to the direct distance to Cape Recife being more than 10 km. This layer was then clipped using the concession area. The layer was then ranked with the assumptions that lower to mid-range distances between 0.5 km and 7.0 km were more suitable than the greater than 7 km distance from the launch site. It is assumed that the closer to a launch site the more accessible the area is but it is also susceptible to greater security risks as more people can access the area. However the launch site does provide a degree of security to the project as the high foot traffic and number of citizens supporting the initiative means that the TTF are receiving notifications from people at the slipway regularly about possible poaching.

3.2.6.1.5 Commercial Plot Proximity and selection for space layer:

The ranching program was established at the Cape Recife area before the establishment of the modelling focus. It became clear that the ranching program needed to model the system in order to assist with the scale of the proposed operation. It was therefore decided to establish a preliminary Cape Recife model to assist in informing decisions for ranching in this specific area.

Generally, a seeding plot is considered to be a circular site with a radius of 10metres from a centre point. New seeding plots were determined using the substrate model to find suitable ranching areas in the vicinity of Cape Recife. This was done using the existing commercially seeded plots waypoints to create proximity measures between plots. The ranching program releases abalone at defined densities of 20 to 30 individuals per m^2 to avoid over-seeding of the reef plots. Standard minimum permissible plot proximities of 40 metres between plot centre points were established as a prerequisite for a plot. This distance was identified as the maximum likely distance that an abalone would move or disperse from the origin of release (following Heasman et al. (2004)).

Furthermore the 40 m proximity was established to ensure that no plots were used in the operation twice to avoid over populating the reef areas, as doubling densities could theoretically mean up to 60 individuals per m^2. These proximity buffers were established mainly due to the unknown movement and dispersal of *H. midae* released in the Eastern Cape. It has been indicated in the literature that size and age impact the dispersal of abalone.

For example, Shepherd (1986) found that mostly smaller animals moved less than 20 m but could disperse up to 250 m in a year, while De Waal et al., (2003) found that abalone in the Western Cape moved/dispersed less than 5 m for both 14 mm and 26 mm abalone (*H. midae)*, over a period of 3 years. Coates et al., (2013) found that adult *Haliotis corrugata* could move up to 90 m per month, but the majority had very small home ranges. Abalone movement and behaviour has been found to be very area-specific and species-specific (Coates et al., 2013; de Waal et al., 2013, 2003; de Waal, 2002; Roberts et al., 2007; Shepherd, 1986).

It was later decided by Lidomix and Wildcoast Abalone that the permissible plot proximity was to be decreased to 30 m to allow for better habitat utilization due to the highly limited available habitat. Buffers with increasing radii of 2 m intervals (to a maximum radius of 500 m, were placed around each permissible plot proximity, to represent the area that had not been utilized or was not used for a plot (or the available "*space* layer"). This ensured that there was coverage of the entire area of Cape Recife and allowed the creation of buffer zones between old existing seeding plots and newly selected plots to be seeded. It also meant that the full extent of the area was included in the layer. The *Space* layer was assigned a weighting based on the importance of plot spacing for ranching, and included areas of "dead space" where plots were too closely spaced and the areas between these plots that were too small to be utilized for a new seeding plot.

3.2.6.2 Other factors used in model testing and improvement

Prior to the implementation of the modelling approach, operations at Cape Recife proceeded under the assumption of the stakeholder that it was the most suitable area to ranch abalone in. This presented a unique opportunity to collect data and model at a finer scale and test the impact that additional factors may have on the models outputs and accuracy.

3.2.6.2.1 Rugosity

Rugosity measures were collected for 33 research sites (see Chapter 4). The chain and tape method was employed with a 3 mm chain 30 m in length. The chain was laid in a manner that followed the topography of the reef and then the tape measure was used to determine the straight line distance between the start and end of the chain. The ratio between the chain length and tape length is referred to as rugosity (for a more detailed explanation, see 4.3.3)

3.2.6.2.2 Depth

Depth was determined for the centre point of each commercial plot by a diver with a UWATEC diving computer. Mean depth was also determined for each research site by averaging the depth reading for each photoquadrat assessed (see 4.3.2). A total of 172 depths were recorded between the commercial and research sites (see 5.2.2).

3.2.7 Rasterization and membership application

3.2.7.1 Rasterization of vector layers

All vector files (see 3.2.6) were rasterized in order to ensure compatibility with the membership tool functionality. The *space layer* was converted to a raster using the polygon to raster Toolset. Similarly, this process was completed for the *security* layer, *priority* layer, and *accessibility* layer. The cell size in the resultant raster layers were 18 m x 18 m for the entire concession area, while the cell size was 7 m x 7 m for the detailed Cape Recife area.

3.2.7.2 Other factors

3.2.7.2.1 Depth

The depths were used with the *Topo to Raster* toolset to create an interpolated raster of the depth for the Cape Recife area. The default cell size was used, which according to the toolset uses either the shorter width or height of the extent of input features, in the input spatial reference, and divides it by 250 ESRI (2009). This meant that the cell size was maintained at 7m X 7m for the Cape Recife node model components. This output is commonly referred to as a Digital Elevation Model (DEM).

3.2.7.2.2 Rugosity

Rugosity has been shown to play a critically important role in the distribution of abalone (Jalali et al., 2015). The rugosity values were interpolated using the *natural neighbor* interpolation technique. The default cell size was used, which maintained the cell size of 7m X 7m. This layer has a very high degree of uncertainty as rugosity is not uniform for the extent used and because relatively few values were determined for the area concerned. While this layer was included it was not weighted heavily in the analysis as it is a low accuracy layer but did indicate areas where rugosity appeared to be higher in the Cape Recife area.

3.2.7.3 Hedge effect

Hedge effects were applied to three of the memberships, including the *substrate*, *rugosity* and *Accessibility* layers (Table 4). Esri (2009) explains the "*hedge effect"* as a controlling effect that can influence the importance of the membership output. For example the "*somewhat effect"* increases the fuzzy output thus making the output less influential, while the "*very effect"* decreases the fuzzy output (meaning that value ranges in sets do not overlap), making it more influential. The use of hedges has been defined as "computing with words" (Cetisli, 2010).

3.2.7.4 Memberships

As discussed in 3.2.5, each raster layer was transformed with a specific relationship function (Table 4). Seven membership functions were created in total. Four of these membership functions applied to the entire concession area as per the permit conditions, while the other three and the substrate membership were applied to the Cape Recife area, which is the primary ranching area. Memberships are advantageous as they make all data outputs in different units of measure comparable by converting values to a range between and including 0 and 1. The application of the membership allowed for analysis of various datasets and data types (Table 4) with regards to their suitability and likely occurrence in the set of values for the output raster dataset. The ArcGIS toolset Fuzzy Membership (Spatial Analyst extension) was used.

3.2.7.4.1 Priority and Security Memberships

The two layers were created using the rasterized western and eastern concession area layers. For the Priority membership a *large response* was applied according to the stakeholder's request for the eastern areas, with the Cape Recife/Thunderbolt area rated as high priority and security due to the node being the closest to the base of operations for protection services, but further from the launch site than the Noordhoek node. The Skoenmakerskop area was rated as the lowest priority area in the concession zone as it was furthest from the base of operation (Table 4). The spread was defined as 8 (total number of classes within the set) and the midpoint was 5 which is considered as the point where 50% of the values fall within the set.

For the Security membership a *Linear function* was applied as requested by the stakeholder as this area was secured by the developer's Tactical Task Force and was assumed that the further from the operation base the less secure the team could make the area (Table 4).

3.2.7.4.2 Accessibility Memberships

An important aspect of site selection according to the stakeholders is distance from the boat launch site. The assumption that was used was that the closer the area to the launch site the easier access becomes for civilians and other commercial operators and the lower security becomes. Conversely as distance and security increases, accessibility decreases. Using the distance layer a negative *linear response* was applied with a maximum of 3 km and a minimum of 8 km. This meant that any distance that was less than 3 km from the launch site was considered highly suitable and any distance greater than 8 km was undesirable. This was applied with a hedge effect "very", to concentrate the membership output values by squaring them.

3.2.7.4.3 Substrate Membership

The *substrate membership* was applied to the classification output layers or "*substrate layers*" for both the Eastern and Western sectors. The Gaussian response was applied to the outputs from the unsupervised classification. The decision to use the Gaussian response was made through a visual assessment of the output layer over the satellite image and determining which classes mostly fell over reef and/or sand. This was completed separately for the western and eastern sectors. The membership function was squared using the hedge effect "very" to concentrate the membership output values (see 3.2.7.3).

3.2.7.4.4 Rugosity Membership

The rugosity membership layer was produced by applying the linear response function to the interpolated rugosity layer. The minimum was set as 0.1 (the lowest rugosity value obtained) while the maximum was set at 0.6 (the maximum rugosity value) (Table 4). A "somewhat" hedge effect was applied, which square roots the membership function and is known as dilution (Esri 2009).

3.2.7.4.5 Depth Membership

For the interpolated DEM a linear response membership was applied. The midpoint was set at -4 and a spread of 0.05 was applied (Table 4). 4 metres was considered as the optimal working depth for the release of abalone and the spread allowed for a wide range of depths beyond and above 4 m. No hedge effect was applied to the resulting output.

Table 4: The different membership functions applied to each layer and the user defined parameter associated with each function.

Membership Name	Layers original units	Function Applied	Midpoin t	Min	Max	Spread	Hedge Applied	Relationship
Depth	Metres	Linear	-4	N/A	N/A	0.05	None	Bell Curve
Substrate	Pixel Colour Intensity range	Gaussian	5	N/A	N/A	0.05	Very	Bell Curve
Priority	Ranked (1-8)	Large	5	N/A	N/A	8	None	Associated to Large Values only
Security	Ranked (1-8)	Linear	N/A	3	7	N/A	None	Positive Regression
Accessibility	Kilometres	Linear	N/A	8	3	N/A	Very	Negative Regression
Rugosity	Ratio	Linear	N/A	0.1	0.4	N/A	Somewhat	Positive Regression
Space	Square Metres	Linear	N/A	2	4	N/A	None	Positive Regression

3.2.8 Fuzzy overlay

The fuzzy overlay step combines different layers based on specific *overlay* methods (also called Overlay Types). Fuzzy overlay was achieved by overlaying the membership functions for both the concession zone and the Cape Recife area environmental drivers. The method of fuzzy overlay was completed using one, or more of five overlay types (Table 5). Each overlay method affects the weighting of the factor in a different manner, with focus on minimum, maximum, or sum of values.

Table 5: The Method of Fuzzy Overlay that can be applied to one or more membership output layers to create a single overlaid output layer (Environmental Systems Research Institute, 2016)

Overlay Type	Outcome
And	The **minimum values** from all of the input evidence memberships
Or	The **maximum values** from all of the input evidence memberships
Sum	Where combined evidence is **more** important than any single evidence
Product	Where combined evidence is **less** important than any single evidence
Gamma	The GAMMA type is typically used to combine more basic data. When gamma is 1, the result is the same as fuzzy sum. When it is 0, the result is the same as fuzzy PRODUCT. Values between 0 and 1 allow you to combine evidence to produce results between the two extremes established by fuzzy AND or fuzzy OR

3.2.9 Sub-models

3.2.9.1 Concession Area

3.2.9.1.1 Prioritization Sub-model

The *accessibility*, *security* and *priority* membership raster layers were overlaid to create the Prioritization Sub-model. The layers were overlaid using the "PRODUCT" overlay between the "*Priority*" and "*security*" memberships, as the use of the "PRODUCT" overlay results in output being less than the input (Esri 2009) and the "AND" overlay function was used between the "PRODUCT" overlay output and the "*accessibility"* membership which returns the minimum value of the set (Table 4).

3.2.9.1.2 Substrate

The substrate membership output layers for the eastern and western sectors were used to create this layer. No overlaying was applied but the layers were merged to create one output.

This layer was the basis on which the model was developed and was the major factor in the decision making-process (Table 4).

3.2.9.2 Environmental Sub-models

3.2.9.2.1 Substrate Only membership layer

This layer was included among the sub-models as it the main driver for abalone habitat and therefore carries 100% weighting when used by itself. The layer, which consisted of an eastern sector and a western sector, was the main layer influencing the site selection process for the abalone ranching project. It was therefore used as a minimum requirement for site selection.

3.2.9.2.2 Depth/Substrate (DS) Sub-model

While not used in the final overlay process this layer was created as an intermediate layer that could be ground-truthed by divers in order to validate the model. The layer was created using the "AND" overlay function for the *depth* and *substrate membership* raster files.

3.2.9.2.3 Depth/Substrate/Rugosity Sub-model (DSR)

An *environment sub-model* was created by overlaying the Depth, Rugosity and Substrate membership layers using the "AND" overlay function. This was the final layer for overlay with the Cape Recife Space Sub-model.

3.2.9.3 Cape Recife Space Membership Layer

The Cape Recife *space membership* layer was used as an input layer on its own. This layer was used in the decision making process to exclude areas covered by existing plots that had already been seeded and to indicate where "useable areas" remain.

3.2.10 Fuzzified Models

Two final output models were produced. The first was a model of Concession Area Usage and Priority for the ranching project. The model was constructed by overlaying the *Concession Area Prioritisation* sub-model and the *substrate only* membership. The purpose of this layer was to identify the most suitable area for ranching, with consideration of the requirements of Wild Coast Abalone and Lidomix (Pty) Ltd.

The second model was of the operational area as determined by the stakeholder. The ranching operator / permit-holder identified the Cape Recife node as the greatest priority area for the ranching operations. This decision was made for a few reasons, namely; historically the area is well known for its natural abalone population (Godfrey, 2003), secondly the security contractor for the ranching project was established at the Cape Recife lighthouse, thus good infrastructure was in place for monitoring and surveillance of the area,, and thirdly through the expert based assessment by Prof. P. Britz and Mr. W Witte (pers comm) whom identified the area as suitable through visual assessments. This provided a great opportunity to build a model for the node. This model was constructed by overlaying the *DSR* sub-model with the *space layer* using an "AND" function.

3.2.11 Site selection and ground-truthing

The model was used to select the seeding plots for the abalone ranching program. Once a plot was selected from the model these were investigated by divers who visually assessed the area and assigned a habitat ranking. The ranking scores used were based on the criteria in the Standard Operating Procedure Production model habitat ranking currently employed by Wildcoast Abalone (pers comm Mr. Warren Witte 2015). These habitat ranking values were plotted against the model output values and compared using a simple regression model.

3.2.12 Analysis

ArcGIS 10.3.0.4322 software was used for all memberships and overlays using the *Fuzzy Membership* and *Fuzzy Overlay* tool set. Furthermore predicted versus measured plots were drawn using the *geostatistical* wizard, and Kringing analysis was completed. The statistical analysis of the linear model was done using R (R Development Core Team and R Core Team, 2016) in R-Studio, Inc. Version 0.99.903 – © 2009-2016.

3.3 Results

3.3.1 Memberships

3.3.1.1.1 Concession Area Memberships

Four membership layers were produced for the concession area. The concession area is approximately 20 km^2 in size. The minimum distance, for all memberships, between the shoreline and offshore boundary line was approximately 420 m, while the maximum distance was approximately 2000 m from the shore to the boundary. The maximum distance to the 10 m isobath boundary was in the Cape Recife sector, while the minimum distance was west of Noordhoek ski-boat club.

The *Substrate* membership sub-model indicated that over 6.68 km^2 (33%) of the 20 km^2 concession area had a greater than 80% likelihood of being suitable substrate for abalone (Figure 6).

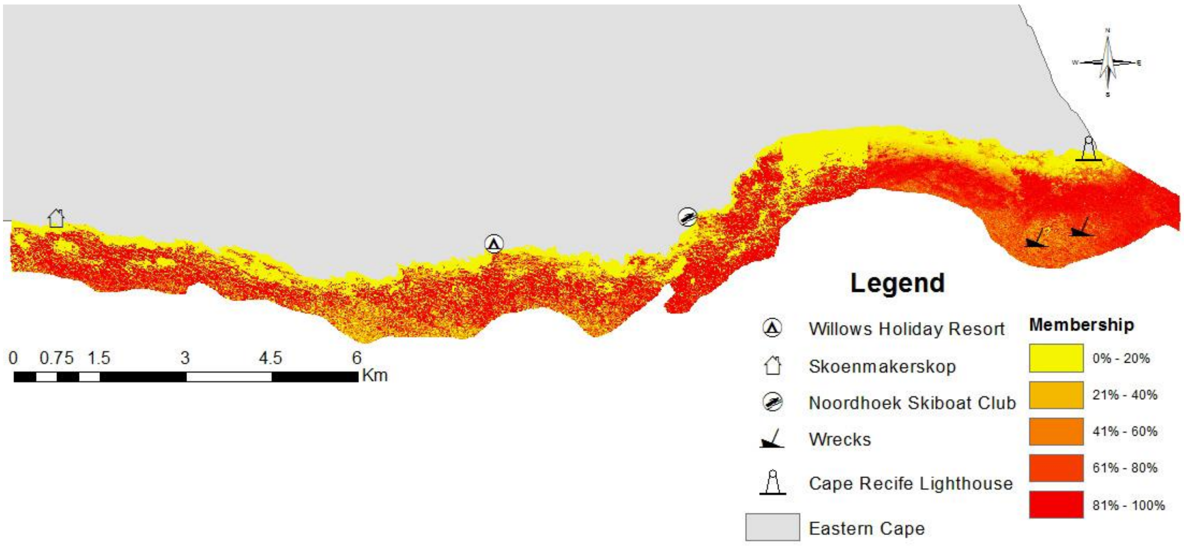

Figure 6: Substrate Membership/Sub-model for the concession area 2 with predicted versus measured scatterplot for model cross-validation.

The distance from the launch site (*Accessibility membership*) indicates that over 6.81 km^2 (34% of the concession area) satisfied the distance requirement of the stakeholder with more than 80% likelihood of falling within the set (Figure 7 (A)). The *Security membership* indicated that an area of 5.01 km^2 (over 25% of the concession area) (Figure 7 (B)) had a suitability of greater than 80%, while the *Priority membership* indicated that an area of approximately 6.97 km^2 (approx. 35% of the concession zone) would be suitable (Figure 7 (C)).

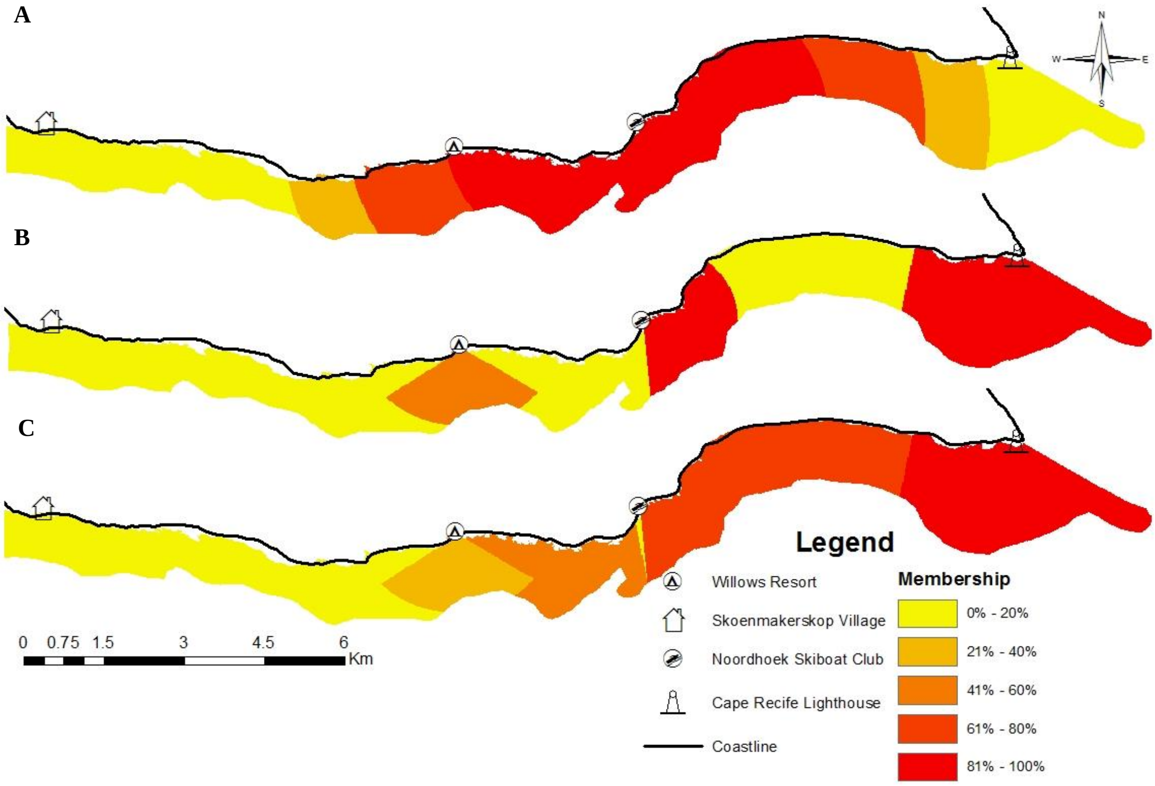

Figure 7: Accessibility, Priority and Security memberships for concession area 2 with associated crossvalidation prediction graphs for each membership.

3.3.1.1.2 Cape Recife Memberships

Through the opportunity to test the use of the modelling application with an established ranching program, the eastern substrate membership layer covered the greatest total extent (over 6.30 km^2) (Figure 8). The *depth membership* layer cover only had an extent of 5.59 km^2 for the Cape Recife area (Figure 9), which was only approximately 89% of east substrate membership's extent, meaning that these layers did not cover the same amount of area in the modelling process. Only 2.21 km^2 (35%) and 1.53 km^2 (24%) respectively of the above membership layers showed a greater than 80% likelihood to meet the criteria for suitable abalone habitat.

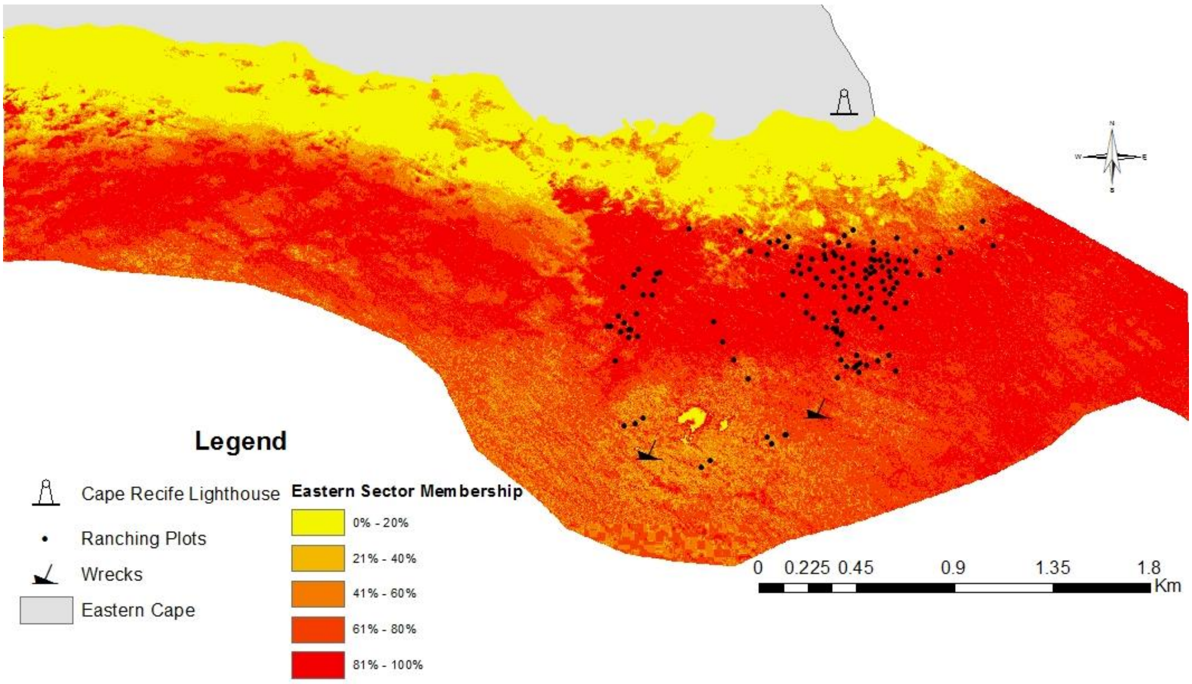

Figure 8: The substrate membership for the Cape Recife sector with predicted/measured plot

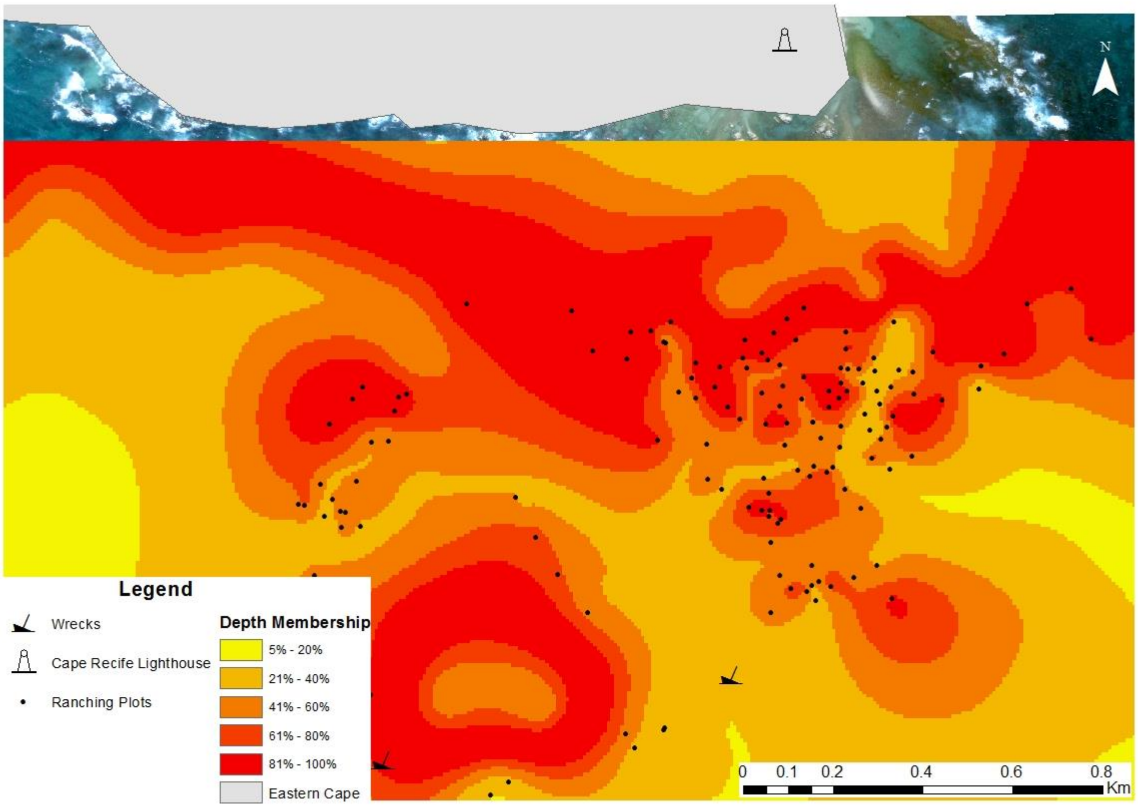

Figure 9: Depth Membership for the Cape Recife sector showing the distribution of points and the predicted /measured plot of model values.

The *Rugosity membership* layer only covered an extent of 1.34 km^2, approximately 21% of the substrate extent. The extent with a likelihood of more than 80% is 0.43 km^2, approximately 32% of the *Rugosity membership* extent (Figure 10).

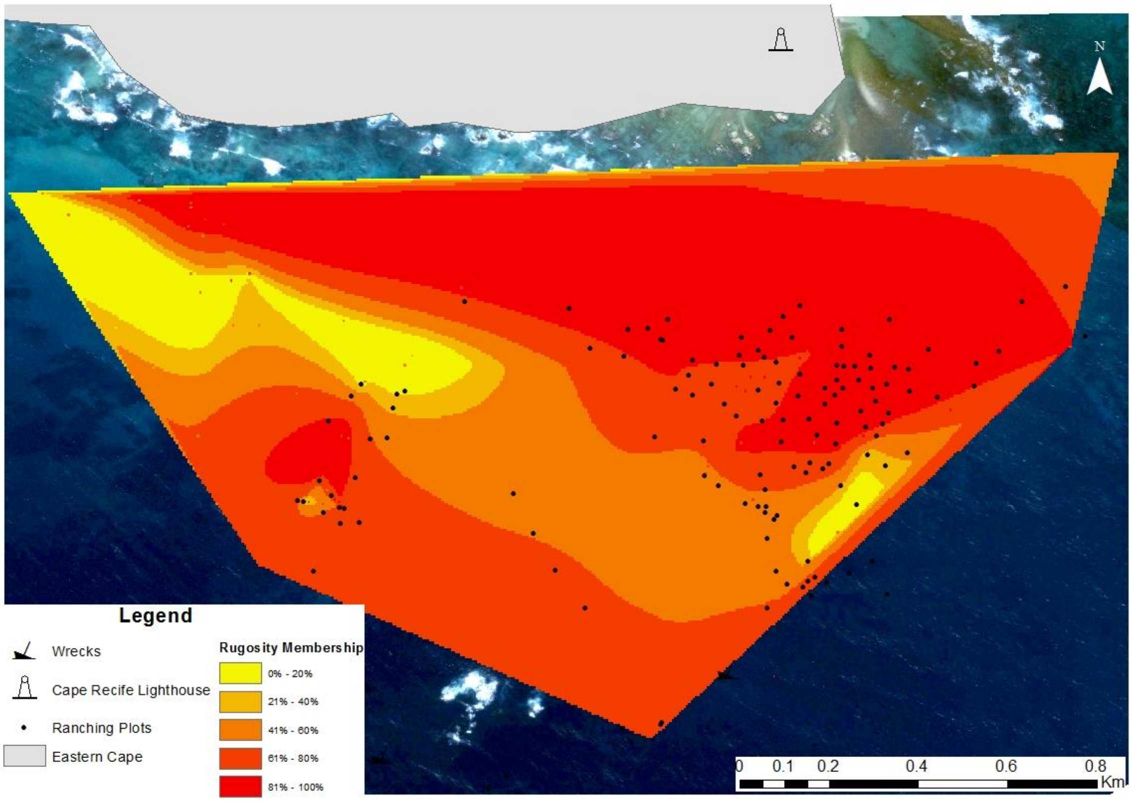

Figure 10: The Rugosity membership showing the distribution and fit of the membership as well as the distribution of the Commercial Seeding plots in the area.

The space membership layer (Figure 11) covers an extent of 3.99 km^2, which represents 63% of the available substrate extent. Subsequently 0.37 km^2 (6%) of the available suitable reef area was considered unusable due to existing seeded plots and buffer exclusions, with the remainder (94%) considered suitable.

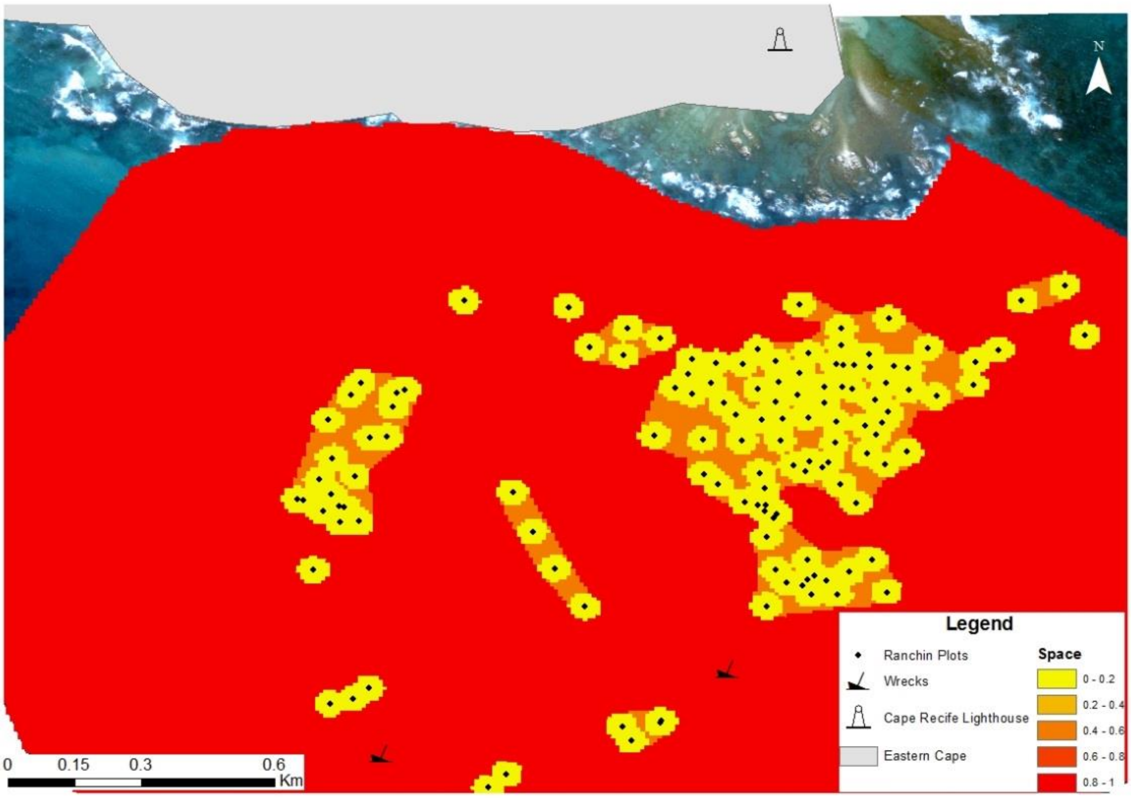

Figure 11: Available Space for the Cape Recife node according to the distance between plots required by Wildcoast Abalones Standard Operating Procedures obtained from literature.

3.3.2 Fuzzification

3.3.2.1 Concession Zone Sub-models

3.3.2.1.1 Prioritization

Of the 20 km^2 concession area the security and accessibility sub-model estimates that 3.68km^2 (≈18%) is suitable for the ranching program requirements. The most suitable sub-area, as shown in Figure 12, is the Cape Recife area with more than 77% likelihood of suitability, while the Noordhoek sub-area is between 40 to 60% likely as suitable by the ranching program requirements.

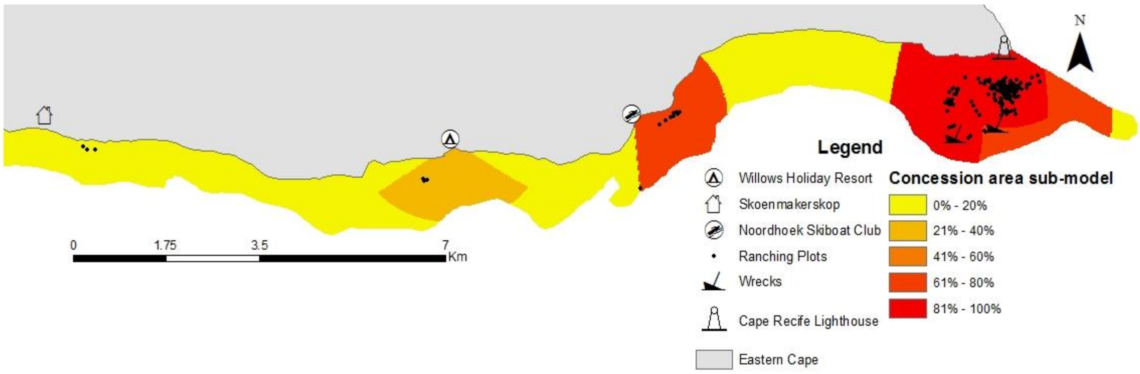

Figure 12: The Concession area prioritization sub-model showing predicted/measured plot.

Analysis of the measured compared to predicted values for the *prioritization submodel* indicated that there is a degree of uncertainty particularly in the higher measured values, with predicated ranges of between 3.3 to 9 for a measured value of approximately 8.3 with an average standard error of 0.01 (Figure 13) and the spread of measured values was across 11 data points.

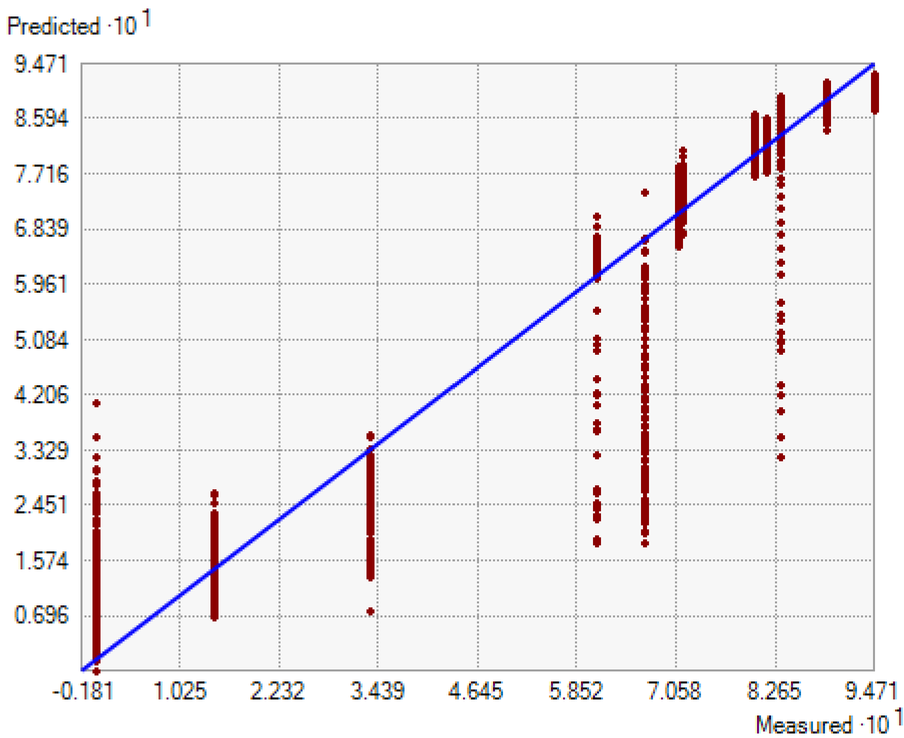

Figure 13: Predicted to measured variables for the Prioritization showing the high variability in the output sub-model (n= 61596, average standard error = 0.011)

The second sub-model is the substrate membership layer (Figure 6). This sub-model has an area of 6.68km^2 (33%) that is estimated to be over 80% likely to be suitable for abalone ranching. However, this is likely to be a slight overestimate as colour signatures in the aerial images that formed the basis for the initial substrate mapping were not homogenous across images and would have introduced some uncertainty in predictions (Figure 6).

3.3.2.2 Cape Recife Sub-models

3.3.2.2.1 Depth/Substrate/Rugosity (DSR) sub-model

The DSR sub-model (Figure 14) estimated that only about 0.10 km^2 (1.6%) has a likelihood of over 80% to be suitable abalone habitat, while the DSR prediction for habitat with a suitability of 50% to 79% is approximately 0.82 km^2 (12.7%). Commercial seeding sites fell predominantly

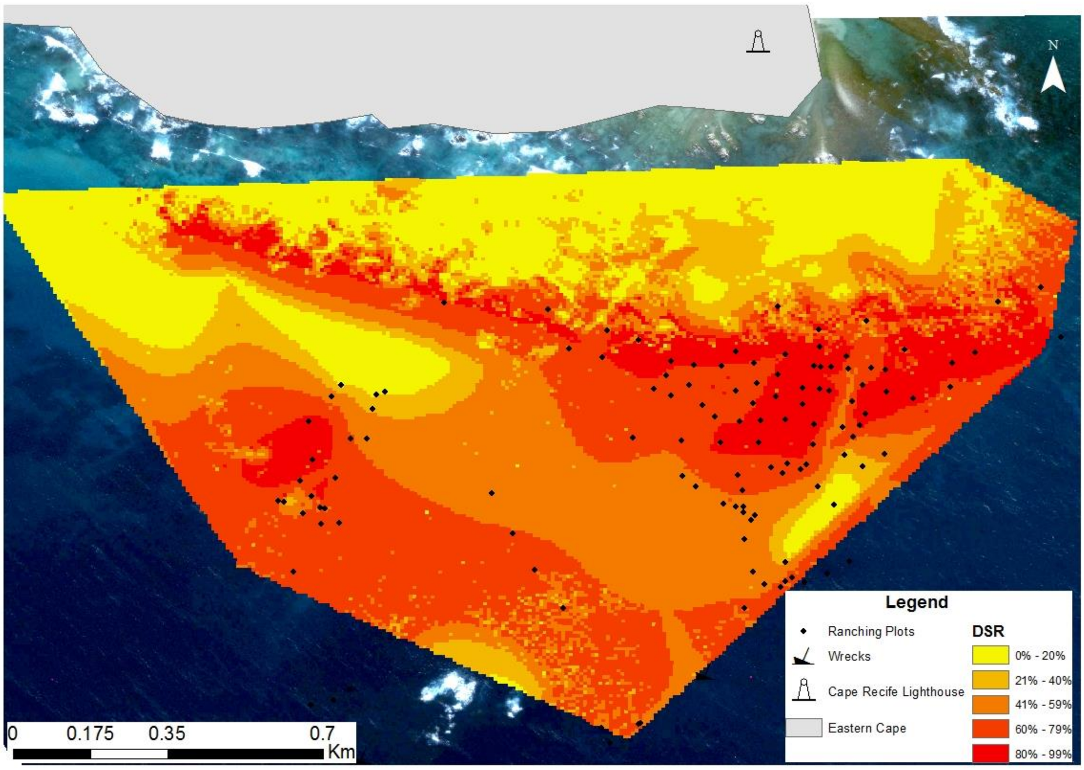

Figure 14: The Depth/Substrate/Rugostity (DSR) Sub-model for the Cape Recife area.

The DSR sub-model shows a good relationship between the predicted and measured variables but does show some variation that can introduce error in predictions (Figure 15). As can be seen in Figure 15, the spread of data points was far more varied for the measured values and the maximum range was 1.78 to more than 9.2 for a measured value of approximately 5.

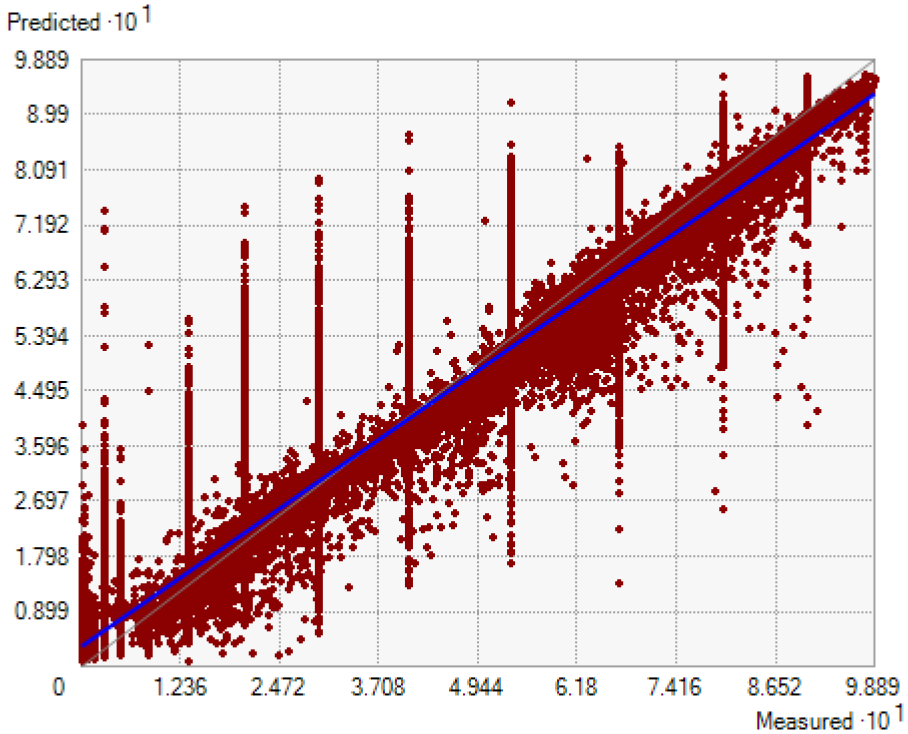

Figure 15: Predicted to measured variables for the DRS showing the high variability in the output sub-model (n= 33433, average standard error = 0.136).

As a general observation, of the Commercial plots, 21% of the plots fall on more than 80% likelihood predicted area and 50% of the plots were between 50% and 79%. Overall 71% of the plots fall on over 50% likelihood suitable area of the sub-model.

3.3.2.2.2 Space Sub-model

The space membership layer was used directly as a sub-model, which means that 0.37 km^2 of the potentially available ranching area has been considered as already utilized and therefore unavailable (Figure 11). This represents 1.8% of the entire concession area.

3.3.2.3 Fuzzy-overlay Model (fuzzified layers combined)

3.3.2.3.1 Concession area 2

The final *prioritization model* of the concession zone shows that 2.70% (0.54 km^2) of the 20 km^2 predicted area, had a predicted suitable greater than 80%, which was predominantly located in the Cape Recife node, while only 16.75% (3.35 km^2) was over 50% likely distributed between the Cape Recife node and the Noordhoek node of the Concession zone (Figure 16).

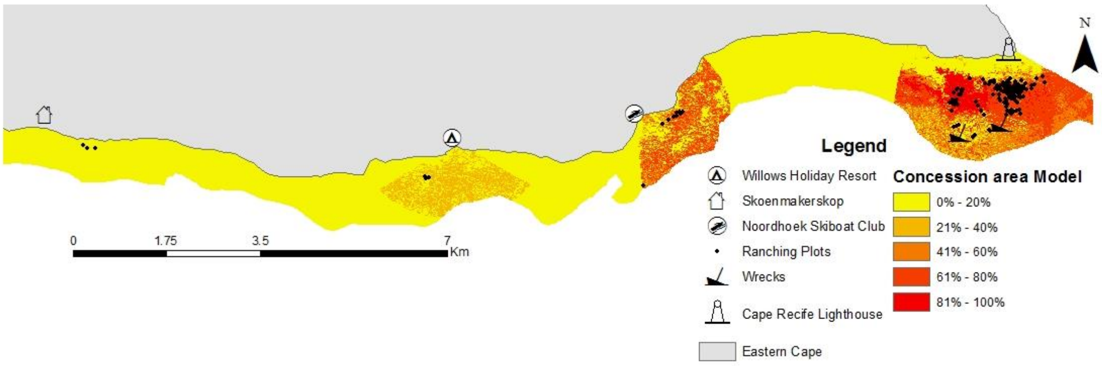

Figure 16: Concession area Suitability Model for Abalone Ranching Site Selection.

As can be seen from Figure 17, the final output has a smaller spread of values as compared to the values of Figure 18. It is also clear that for the midrange of the measured values have very high variation for the prediction values, with a range between 0 and 8.85.

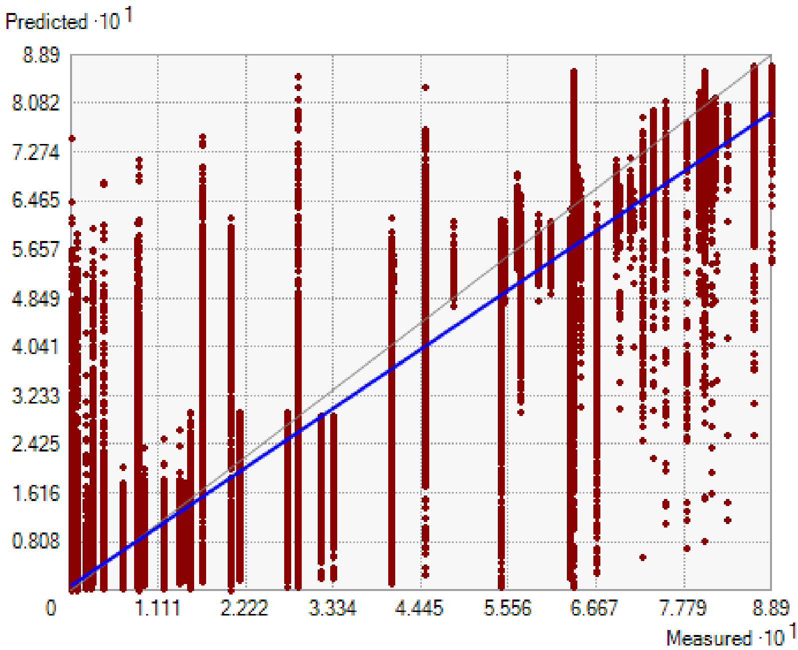

Figure 17: Predicted to measured variables for the Concession area model (n=60369 average standard error = 0.109)

Approximately 2% of the commercial seeding plots have been established in the Skoenmakerskop and Willows node, while approximately 6% were established at the Noordhoek node (Figure 4). Approximately 8% of all commercial ranching plots (n=15) are located outside the Cape Recife node. Results of the kriging (Figure 18) show that the highest accuracy of modelling was maintained at the Cape Recife node followed by the Noordhoek node.

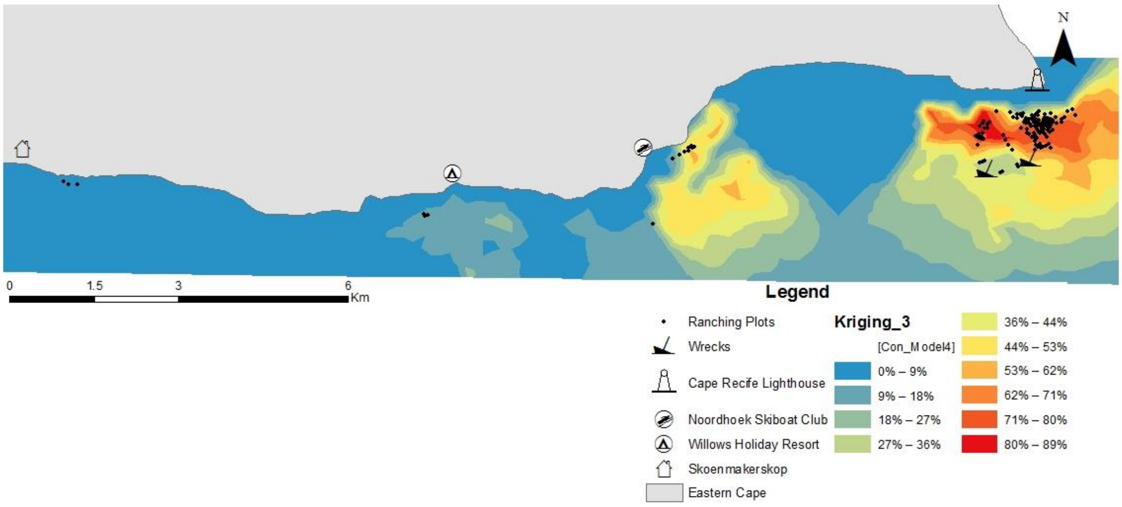

Figure 18: Concession area Kriging cross-validation for Abalone Ranching Site Selection, showing areas of highest probability

3.3.2.3.2 Cape Recife Model

The Cape Recife model represents an area of 1.57 km^2 (approximately 25 % of the Eastern Sector). The area with a "suitable habitat" likelihood of greater than 80% was 0.03 km^2 (2%). The area with over 50% likelihood was 0.71 km^2 (46%) (Figure 19).

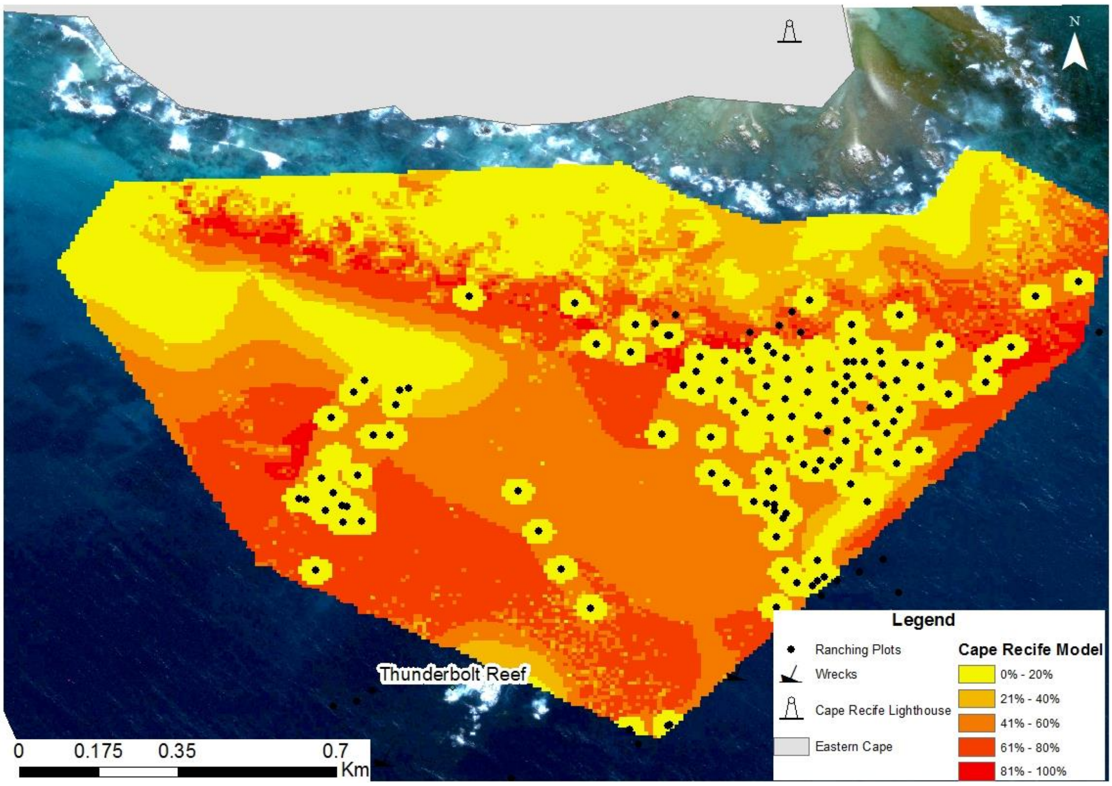

Figure 19: The Cape Recife Model for suitable site selection for the abalone ranching program.

A plot of measured vs predicted values (Figure 20) shows that the model tends to under-predict site suitability. There is also a high amount of variation within the model, which results in inaccuracy in the predictive capacity. The spread of measured values is between approximately -0.20 and 9.67, with the greatest range of predicted values in the midrange of the measured values being between 0.3 and 7.85 at a measured value of 5.15. The upper and lower ranges showed lower variation in values.

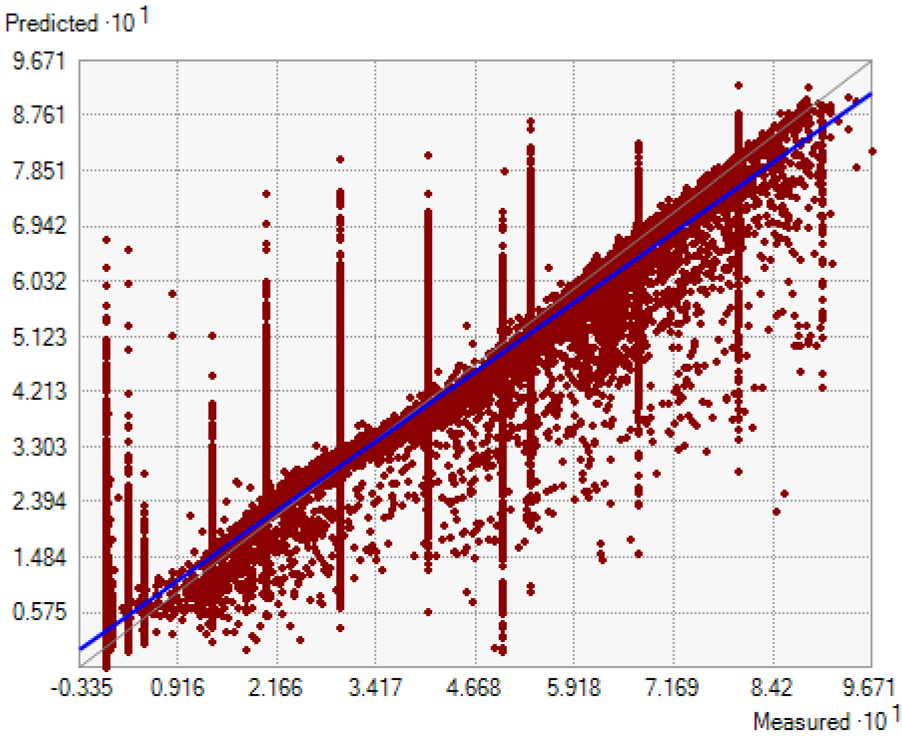

Figure 20: Predicted to measured variables for the Concession area model (n= 25072, average standard error = 0.102).

The kriging analysis shows areas where interpolated predicted values are most similar to the measured values at Cape Recife where suitability (Figure 21). As can be seen circled in red, there are three areas that showed high probability of similarity to the measured values of the model. These areas correspond to the areas where commercial plots have already been established. Another trend in the analysis is that predicted areas with the highest suitability are located on the landward side of thunderbolt reef.

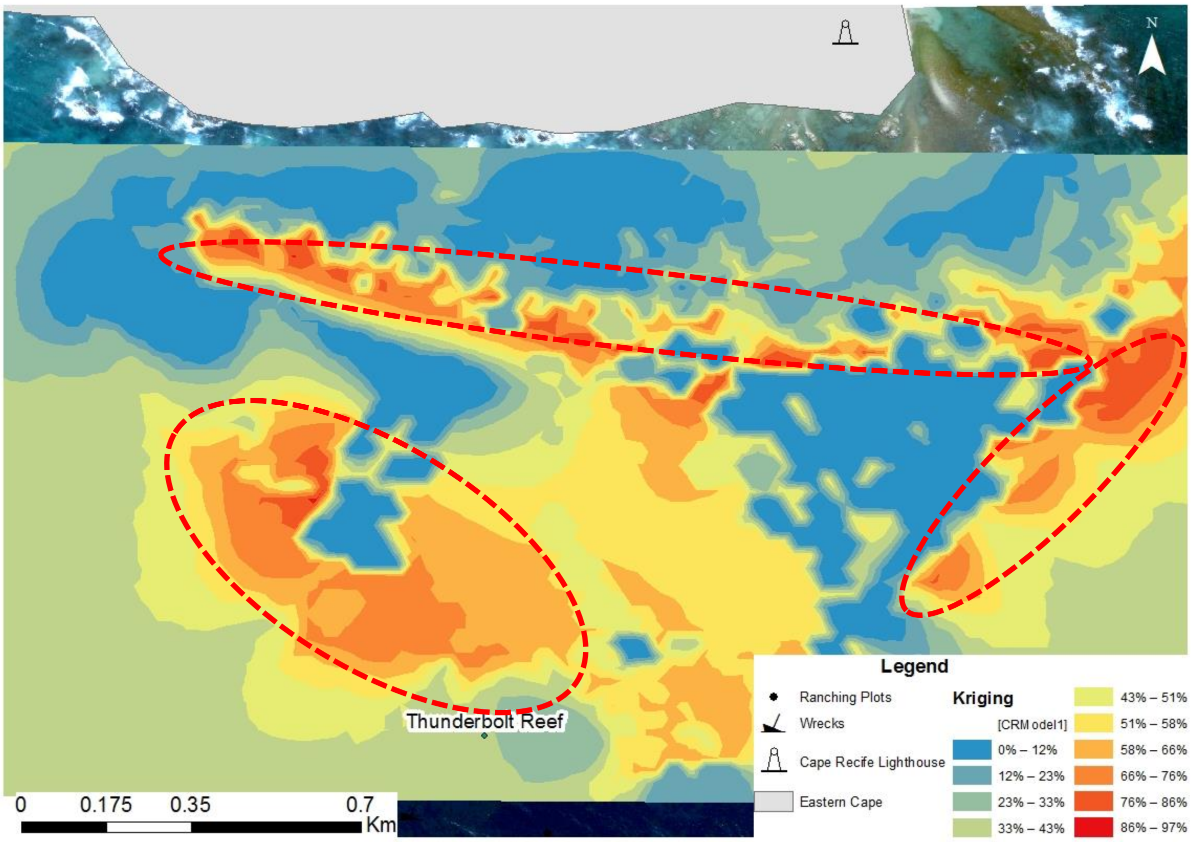

Figure 21: Kriging cross-validation of the Cape Recife sector showing areas of highest probability circled in red.

3.4 Ground Truthing

By 24 August 2016 a total of 151 commercial seeding sites were established in concession zone 2. These commercial seeding sites were selected in areas where the suitability ranking was greater than 80% likelihood of suitable abalone habitat and had abalone released onto the reef by the commercial operator Wildcoast Abalone, while this requirement was not used for a research site. The majority (91%) were located in the Cape Recife sector of the concession area. Five percent of the commercial sites were assessed prior to seeding taking place (pre-seeding samples). A total of 33 research sites were assessed during the duration of research. Forty percent of the research effort was focussed in the Noordhoek/Suicides sector, and the remaining 60% of the research sites were located in the Cape Recife sector of Concession Zone 2. In total 109 sites were used to verify the model.

Figure 22 shows the habitat suitability score predicted by the GIS modelling against the habitat ranking scores based on visual assessment by divers at each of the surveyed sites.

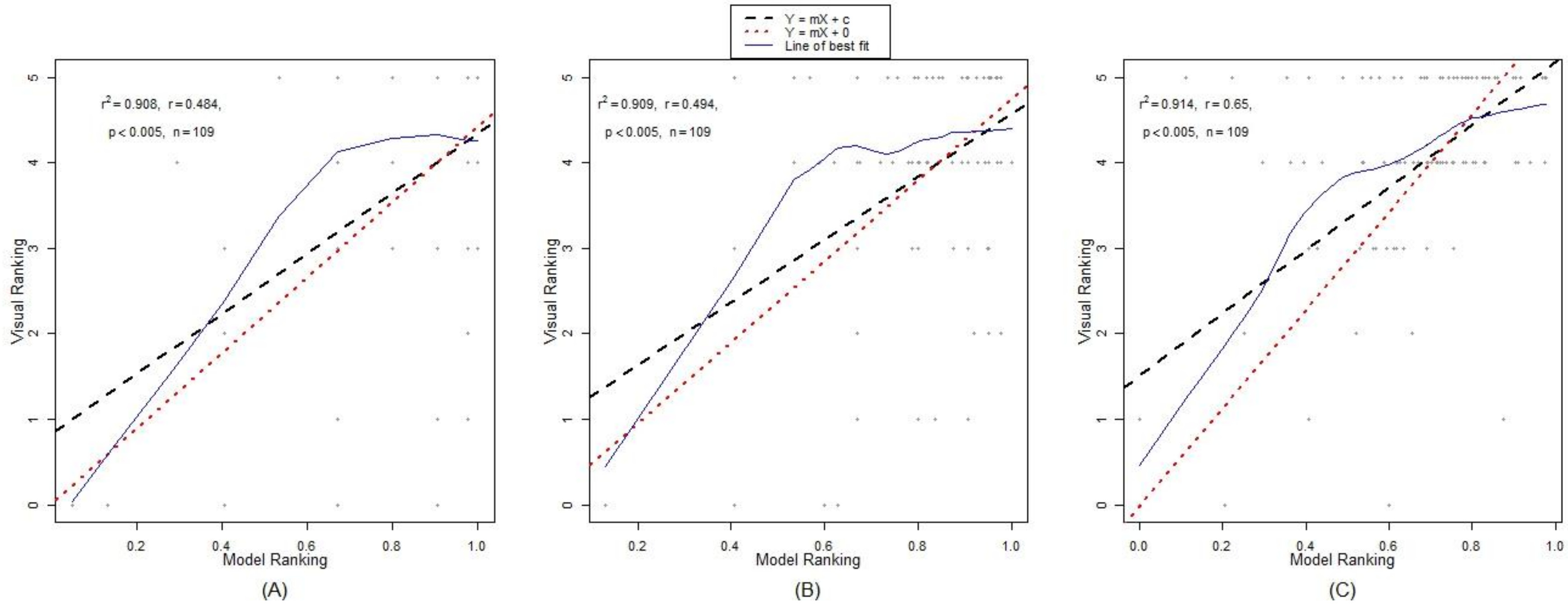

Figure 22: Regression analysis of visual rank compared to sub-model output values: (A) The Substrate Only sub-model; (B) Depth and Substrate (DS) sub-model for the Cape Recife sector; (C) DSR sub-model for Cape Recife sector.

The *Substrate Only (S) sub-model*, showed a significant but weak correlation ($r = 0.48$, $t = 5.71$, $df = 107$, $p < 0.001$) between the model output values and the *visual assessment value* (Figure 22A). The *Depth and Substrate sub-model* (*DS sub-model*) the correlation relationship remains significant but weak ($r = 0.49$, $t = 5.88$, $df = 107$, $p < 0.001$) (Figure 22B). The resultant layer from the overlaying of rugosity, with depth and substrate, the *environmental sub-model* (*DSR sub-model*) showed a stronger correlation relationship ($r = 0.65$, $t = 8.84$, $df = 107$, $p < 0.001$) (Figure 22C). The application of a linear model showed a strong relationship between the *substrate only submodel* and the visual assessment value ($r^2 = 0.908$, $t = 32.66$, $df = 108$, $p < 0.001$). The linear model showed no increase in fit for the *DS sub-model* ($r^2 = 0.908$, $t = 32.78$, $df = 108$, $p < 0.001$), but did show further increase in fit for the *DSR sub-model* ($r^2 = 0.914$, $t = 33.91$, $df = 108$, $p < 0.001$). A comparison of the output values of the different sub-models using a Wilcoxon rank sum test showed significant differences ($W = 7765.5$, $p < 0.001$) between the *S sub-model* and the *DS* sub-model. There was also significant difference between the outputs of the *DS* and the *DRS sub-model* ($W = 3268.5$, $p < 0.001$).

3.5 Discussion

3.5.1 Site Selection

The results from the GIS fuzzy model approach for multi-criteria decision making (MCDM) represent an efficient and effective approach to site selection and decision making for the abalone ranching operation. Godfrey (2003) showed that zone 2 of the concession area holds good potential for abalone ranching and stock-enhancement programs (Godfrey, 2003). Based on the specific habitat requirements of *Haliotis midae* (Wood, 1993), coupled with the stakeholder requirements and early project decisions, Cape Recife and Thunderbolt reef were quickly established as the main focus areas.

The implementation of the fuzzy model for site selection was initiated in July 2015. Site selection initially focused on the substrate only membership layer, which resulted in good likelihood of site suitability for ranching activities if selected with greater than 80% prediction from the model but with varying error in substrate type signatures, due to the difficulty in separating the influence of depth and seaweed cover changing the signature of the pixels. As other layers were incorporated the model fit improved, which can be interpreted from the prediction plots.

In Japan, scallop farming has become a priority and the need to implement MCDM systems has come to the attention of decision-makers (Radiarta et al., 2008). Countries like New Zealand, Bangladesh and India have also started adopting this approach (Booth and Cox, 2003; Cyrus and Pelot, 1998; Kluger et al., 2015; Salam et al., 2003). It has not only allowed for both biophysical parameters and socio-economic parameters to be accounted for in the suitability selection processes (Radiarta et al., 2008) but has also allowed for improved and more effective spatial planning and monitoring of the environment (Falconer et al., 2013; Silva et al., 2011). It has been shown by many authors that a robust multiple-criteria approach can significantly increase the decision-makers' ability to make informed decisions (Brigolin et al., 2015; Dapueto et al., 2015; Falconer et al., 2016; Mendoza and Martins, 2006; Radiarta et al., 2008; Silva et al., 2011). The implementation of the system in this research too has created a robust and elegant approach in order to assist the stakeholder and commercial operator in the process of site selection for the ranching of abalone.

3.5.2 Advantages and Limitations of the approach

The fuzzy MCDM approach to site selection has a number of advantages and disadvantages. While Fuzzy MCDM is a valuable and effective toolset it will always have a degree of uncertainty (Silva et al., 2011)

One of the major disadvantages in the modelling process can be attributed to the image classification process. The substrate classification was based on the colour signatures of the aerial images with the assumption that areas of darker colour (dark blue to black) would be reef and areas of light blue would be sand and sediments. Traditionally the use of classification studies have been completed in the terrestrial environment (Charaniya et al., 2004; Flügel et al., 2003; Watts et al., 2012), but the use of spectral signatures has been successfully done in the marine environment, particularly in the clean tropical waters of coral reefs (Chauvaud et al., 1998; Hochberg et al., 2003; Hochberg and Atkinson, 2003) and intertidal zones (Pech et al., 2004).

One of the major problems with the above assumptions is the influence of depth and the presence of seaweeds, which has been documented by Hochberg and Atkinson (2003). It was also assumed that as the 10 m isobath was approached that depth influenced the colour resulting in darker colour, making it very difficult to distinguish sediment from reef. Bouvet et al. (2003) found that great confusion can be introduced in unsupervised classifications where two sediment types are identified as one class, or in this case where seagrass beds in the shallow waters were confused with deeper coral rich flats.

Andréfouët et al. (2003) suggested that a depth correction factor should be applied before classification of sediments is done, which can significantly increase accuracy of classes (Wahidin et al., 2015). A technique developed by Lyzenga (1981) used different bandwidths of LandSat images to determine bottom characteristics, using an algorithm he developed. This showed that substrate could be accurately mapped to depths of 15 metres but is specific to the bandwidth and image clarity in terms of ocean water turbidity. The approach can be problematic over a range of depths causing different sediments to be classified as one group (Andréfouët et al. 2003).

Another inaccuracy that emerged while using the fuzzy approach for the abalone ranching project was that there was presence of seaweeds over both reef substrate and sand substrate, resulted in sandy areas being incorrectly classified as reef. This is difficult to derive from the dataset however, it has been suggested by many authors, that validation and site visits be made to verify classification predictions and to feed back into the classification process (Andréfouët et al., 2003; Dekker et al., 2005; Hochberg et al., 2003; Lyzenga, 1981; Villa et al., 2013; Wahidin et al., 2015). Jalali et al. (2015) proposed that if accurate spatial reef data can be obtained, the link between habitat, and abalone presence can be accurately modelled, allowing for greater control of the fishery and catch effort of abalone.

There are numerous advantages of the fuzzy model approach over other modelling techniques such as linear modelling, weighted modelling, distance-based models and Bayesian network models. According to Adriaenssens et al. (2004) the greatest advantage is the use of linguistics to govern the memberships which gives a high degree of transparency in the system under study. Fuzzy models and Bayesian networks are predominantly separated by the fact that a fuzzy approach is regarded as a soft computing strategy and therefore is easily interpretable by most users, whereas Bayesian networks, which work on similar modelling principles to the fuzzy approach but is a statistical based technique which can result in the model interpretation being much more complicated (Pradhan, 2013).

Furthermore the use of graphics, which is predominantly the output of fuzzy models, can be visually assessed, whereby the output map can be interpreted by multiple users and due to the ranking values can assist the user in the placement or selections of plots and sites. This can also be done with a good degree of accuracy and the use of graphics can limit the loss of information from numerical decisions, allowing for the effective interpretation and utilisation of the model outputs by all users regardless of the training and modelling experience (Bender and Simonovic, 2000).

The use of fuzzy approaches is flexible with respect to data input and can accommodate individual layers that include a high degree of uncertainty (Cheung et al., 2005). The level of uncertainty is reduced and precision is increased as more layers are overlaid (Zadeh 2008). It is cost effective and has rapidly become an important toolset that can be utilized by poor countries and in data-poor areas around the world (Silva et al. 2011).

Finally, it can also help decision makers in cases where site visits and accessibility are difficult, and time is limited (Adam et al. 2006). The use of aerial imagery and remotely sensed data has assisted in the establishment and management of many ocean based aquaculture ventures (Aguilar-Manjarrez and Ross 1995; Cyrus and Pelot 1998; El-Gayar and Leung 2000; Radiarta et al. 2008; Ellingsen et al. 2009; Brigolin et al. 2015; Gimpel et al. 2015; Hofherr et al. 2015) and continues to play an important role in the decision making process.

3.5.3 Cross-validation and Ground truthing

Validation techniques for predictive models are an important step in ensuring accuracy. The first of these techniques used in this study was a cross-validation technique of kriging. A scatterplot of measured vs predicted values is used to test the predictive ability of the model. The distribution of the points in the scatterplot in relation to a line with a slope of 1 provides an indication of the predictive ability of the model (Johnston et al., 2001). Individual layers, such as the *Space*, or *Security*, which were constraining layers, tend to have very high variability and often do not have a good fit, but as the layers are overlaid this results in changes in model accuracy and predictive capability with decreased uncertainty for predictions. Kohavi (1995) states that in estimating the classifier (model estimation method), it is important that low variance and bias are obtained in the method, which was found to be similar in these results, as high variance means low probability of successful predictions. Mueller et al. (2004) argued that although useful, cross-validation techniques cannot be the sole criteria of determining model suitability and that visual site checks and ground truthing are critical in any modelling process. This was confirmed in our research as comparison of visual and predicted showed significant increase in accuracy as more sites were inspected.

Abalone habitat appears to be widely determined by the substrate type, and is further influenced by depth and at a finer scale the rugosity of the reef, as can be seen between the substrate only, DS and DSR sub-models. Through field-validation techniques further patterns and responses may be observed that may be overlooked in the modelling process (Flügel et al., 2003). This was found to be the case in Australia where Lidar was used coupled with MaxEnt modelling techniques to identify suitable areas for abalone harvesting. The study also employed diver knowledge on productive areas and GPS track records which alloed for increased accuracy in predicting and mapping suitable habitat (Jalali et al., 2015). Their results indicate that there was good overlap between the data source (Lidar) and the model (MaxEnt) (Jalali et al., 2015).

Visual assessments can be very subjective, and are influenced by the divers' knowledge and experience in abalone counts. This bias is well known in this type of research and has been documented by a number of authors (Hart et al., 2008, 1997; Jalali et al., 2015; Kenyon et al., 2006; Kulbicki and Sarramégna, 1999; Raemaekers and Britz, 2009). While a strong correlation has been shown in the ground-truthing stage of the modelling process in this study, it was assumed that there was a fair degree of diver bias in the assessment due to more than one diver doing the visual assessment of suitability of the abalone habitat using the provided protocol of Witte (2014).

It has been suggested by Hart et al. (1997) that transect lines are used to increase robustness of the sampling method when doing visual assessments, that will assist in removing diver bias as it is easily mastered by divers with low experience and can introduce attributes for the diver to observe along the length of the line to indicate the visual ranking (Hart et al., 1997). Another suggestion, by Kulbicki and Sarramégna (1999) was to utilise divers in pairs with one highly experienced and one not, and to use the same divers for all sequential events. Inexperienced divers tend to significantly underestimate organism numbers (Kulbicki and Sarramégna, 1999), so by pairing them with experienced divers it is assumed that the variation will be more standardized. This however does pose the problem of diver availability and experience being a limitation is successfully completing a project. The method of sampling remains the most important factor in eliminating influence of diver experience, and environmental conditions (Hart et al., 1997; Kulbicki and Sarramégna, 1999)

In terms of the modelling there appears to be substantial change in the degree of uncertainty from substrate-only layer to the DSR sub-model as the constrained values provide a more accurate prediction. This is well documented in studies where direct environmental influences have been used in the modelling process (Cornelissen et al., 2001; Dapueto et al., 2015; Van Broekhoven et al., 2006). Cornelissen et al., (2001) suggests that ensuring the reliability, with regards to its accuracy, of a membership function produced for the modelling process, is an important process with regard to verification and validation of the fuzzy model.

3.5.4 Conclusion and Recommendations

The fuzzy logic approach to suitability modelling has proven to be successful for the abalone ranching program at Cape Recife with more than 100 commercial seeding plots being established by the end of August 2016. The model is very dynamic and easily adapted to different areas, as the use of remote sensing means that it is low in cost, and that most areas along the coast can be assessed as long as the right imagery and data are available and due to the fuzzification process being rapid and the outputs easily interpreted, results can be quickly obtained and transferred to any stakeholder. The suggestion has been made by numerous authors that modelling techniques, such as fuzzy modelling, be applied to terrestrial phenomena (Cornelissen et al., 2001), wind processes (Douvere, 2008), ocean currents (Puniwai et al., 2014), environmental monitoring, and climate change prediction (Cheung et al., 2005; Navas et al., 2011; Smale et al., 2011)

It is recommended that depth data are collected for areas where seeding and ranching operations are proposed as this coupled with substrate type and rugosity can significantly improve the suitability prediction. Furthermore, the use of imagery with greater resolution and clarity will improve predictions. One of the ways which has been suggested is to use medium altitude, long endurance (MALE) unmanned aerial vehicles (UAVs) (Watts et al., 2012), which can be used to map, at fairly low costs areas, at small to medium scales using single-lens reflex (SRL) digital cameras and GNSS/INS satellite tracking systems, as well as aid in the 3D mapping of the area (Nex and Remondino, 2013). Another potential technology that could be used in remote sensing and modelling to identifying potential abalone habitat would be the incorporation of LIDAR technology and data in the model, as suggested by Jalali et al., (2015). This will significantly and rapidly assist the assessment of an area's suitability and will also not require as specific conditions, of clean calm water with no cloud cover and small swell sizes as that of the aerial photography due to the lasers being capable of penetrating up to three times the secchi depth and not being interfered with by cloud cover (Jalali et al., 2015). This is because as the red lasers are reflected from the surface they create a reference point (almost a zero state) whereas green lasers can penetrate to the sea floor, depending on turbidity and sediment loading, reflecting back to a receiver which can then build the seascape and depth (Miller et al., 2016).

Overall the approach and method used in this study have been largely successful. The main aim of the research was to create a decision support system for the abalone ranching project to assist in the selection of suitable plots meeting both the environmental requirements of abalone and the requirements of criteria set forward by the stakeholders. The research has produced a model that is currently in the implementation stages for the site selection criteria, which has further standardized the process, as well as had added benefits of time saving and less search effort for suitable habitat.

Chapter 4. Defining the optimum sampling method for site selection in abalone ranching.

4.1 Introduction

The basis of science is built around the steps of observing a phenomenon, formulating a hypothesis, designing and executing an experiment and reporting the results. In order to efficiently and effectively describe and understand an ecosystem, experimental design and methodology needs to be carefully selected and scrutinized in order to ensure the greatest precision and accuracy is achieved (Downing, 1979). The abalone ranching project site selection was initially based on expert knowledge of a few individuals on SCUBA. This resulted in a high amount of variability and wide definition of “suitable habitat” for abalone.

The design and application of a suitable framework is believed to be, by the industrial partner, a critically important aspect in ensuring the longevity and sustainability of the ranching program. Many different methods and frameworks have been constructed and applied to survey and describe ecosystems, habitats and communities (Celliers et al., 2007; Kibele, 2016; Leujak and Ormond, 2007). The use of ecological indicators to infer system status and health, as well as assist in the monitoring of systems has been the focus of many studies globally (Ballesteros et al., 2007; Cornelissen et al., 2001; Hobday et al., 2016; Juanes et al., 2008; Naranjo et al., 1998).

Marine subtidal surveys are predominantly completed on SCUBA along transect lines, or using quadrats to quantify and describe the environment and patterns therein (Burd et al., 2008; Dumas et al., 2009; Heyns et al., 2016; Hutchings and Clarke, 2008; Leliaert et al., 2000; Parravicini et al., 2010; van Rein et al., 2009; Zeeman et al., 2012)). They are very limited by the amount of time that sampling may take place due to depth related limitations, air availability and environmental conditions (Dumas et al., 2009; Heyns et al., 2016; Parravicini et al., 2010).

This limitation drives the question: what is the minimum one can sample in order to represent the habitat of the subject organism or are there any alternative methods that can be used to sample that do not lose significant amounts of information? The research was therefore further based on a number of questions, including what is the most suitable size quadrat to represent the benthic community on the site, and how many replicates are required? How long should a transect be to give an accurate indication of the amount of suitable reef at a site? And does photographic data compare to scrape collection data?

The aim of this part of the study was to develop a simple repeatable sampling protocol that can be standardized in the industry and be utilized across habitat types and by various fieldworkers and can still be easily and objectively analysed by scientific personnel. The use of an standardize sampling protocol will also assist the ranching operation in effectively and objectively selecting suitable habitat onto which the stock can be released.

4.2 Sites

The sampling was conducted in areas selected according to the prioritized sites model (Figure 12), which included area near Cape Recife and Noordhoek (Figure 23). This study focused on the aforementioned regions which are dominated by sandy substrates (Chalmers, 2012) with high rising reefs that consist of the table mountain quartzite group (Chalmers, 2012; Rust, 1991).

The Cape Recife headland (Figure 23) is found on the southeast coast of southern Africa (Lord et al., 1987), and separates Algoa Bay from the Cape St Francis bay to the west (Steyn and du Preez, 2013) and the Woody Cape headlands to the north east forms the eastern boundary of the Algoa bay (Chalmers, 2012).

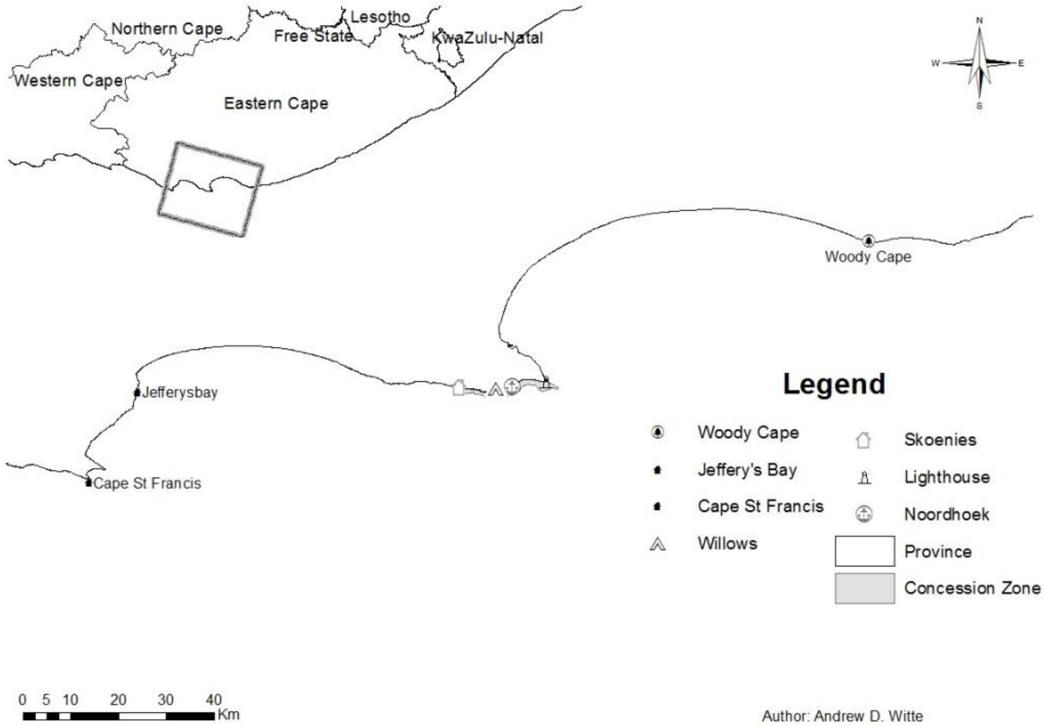

Figure 23: Site Map showing the headland of Cape Recife between Woody Cape and Cape St Francis. Showing the two half heart-shaped bays (Branch and Branch, 1981).

The substrate and environmental structuring is very diverse and varies in vulnerability to anthropogenic impacts (Lord et al., 1987). A strong upwelling cell develops on the southern shoreline of Cape Recife during westerly winds (Goschen and Schumann, 1995). These dynamics results in great variability in temperature and nutrient availability in the region (Beckley, 1983; Hutchings et al., 2013; Sauer et al., 1991; Schumann et al., 2005, 1991). Cape Recife has been described as an area with a large population of wild abalone, and although heavily poached, has been suggested to be a suitable area for the establishment of a ranching and stock enhancement program (Godfrey, 2003).

Noordhoek Ski boat club is a private launch facility and water sports club located approximately 5 kilometres to the west-south-west of Cape Recife. The area's geology remains similar to Cape Recife. The launch site is a private surf launch in a natural channel that is most often used by fishermen and spear fishermen with approximately 1200 launches per annum (per comms. Piet Botha club manager) The area is therefore subject to heavy boating activity and is a highly accessible area to the public.

4.3 Methodology

4.3.1 Functional groups

The functional group approach to benthic habitat characterisation has been used with great success on research of South African temperate reef systems. The approach has been used by a number of authors including Celliers et al., (2007), Götz et al., (2009), Díaz et al., (2011), and Heyns et al., (2016). Götz et al., (2009) refered to *"benthic taxa"* recognising seven groups in their research, whereas Celliers et al., (2007) worked in Pondoland, and recognised 23 groups referring to them as *"benthic categories"*. Heyns et al., (2016) identified taxa down to species level but incorporated the use of *"benthic taxa"* recognising 17 groups in their analysis. Díaz et al., (2011) used *"functional groups"* in an intertidal study recognising 7 different algae groups (*"Functional groups"*). With regards to this research, a total of 21 functional groups were recognised. Nine of these groups were identified to species level as they are very distinctive and easily recognisable from an image and are known to have associations to abalone. For example *Parechinus angulosus* is well known to provide protection to juvenile abalone (Day and Branch, 2002a, 2002b), and *Plocamium corallorhiza* is among the dominant food sources for abalone at Cape Recife (Wood, 1993). Due to the importance of abalone (*H. midae*) the species was classified separately from the other groups. The other groups were defined by different species and forms (see Table 6).

Table 6: Functional group description

Species/Functional Group	Description	Mean Cover (%)
Plocamium corallorhiza	The dominant Rhodophyta to 7 metres (Knoop, 1988) in Algoa Bay. It has broad and flattened fronds that, in water, tend to have an iridescent blue to purple shimmer on the fronds margins (Branch et al., 2010; Stegenga et al., 1997).	22.3
Coralline Turf	Includes species of *Jania*, *Amphiroa*, *Arthrocardia* and *Corallina*	16.1
Colonial Invertebrates	Includes a number of taxa: Porifera, Gorgonians, Zooanthids and Corals, Hydroids, Bryozoans, and Ascidiaceae excluding *Pyura stolonifera* and *Gynandrocarpa placenta*.	14.5
Foliose Algae	The dominant genera that were indicators for this group included *Codium*, *Gigartina*, *Calliblepharis*, *Portiera*, *Callophycus*, *Zonaria*, *Dictyota* and *Stypopodium*.	11.6
Encrusting Algae	Calcified algae that grow in a horizontal plain over rocky substrates	10.5
Fleshy/Foliose Turf	This group is mainly defined by *Hypnea* and *Laurencia natalensis* but also included filamentous red seaweeds such as *Polysiphonia*, Greens such as *Cladophora* and *Chaetomorpha*, and browns from the family Ectocarpaceae.	7.1
Pyura stolonifera	Commonly known as "*red bait*". It has a very dark brown skin and the siphons are slightly scalloped (Branch et al., 2010). Often found in aggregates on exposed rocky outcrops.	6.3
Laurencia flexuosa	This species has a red/purplish to brownish, flattened, complanate thallus and is branched regularly (Branch et al., 2010; Stegenga et al., 1997).	4.0
Upright Articulated Corallines	The group consisted of two species, mainly *Amphiroa ephedraea* and *Jania verrucosa*	2.0
Anthozoa	This group represented the solitary sea-anemones, including species such as *Pseudactinia* and *Actinia* sp.	1.1

Species/Functional Group	**Description**	**Mean Cover (%)**
Hailimeda cuneata	A lightly calcified green with flat disc-like segments, which appear to look like wedges (Lubke and Seagrief 1988)	0.2
Asteroidea	Starfish, featherstars and brittlestars	0.2
		Mean individuals per metre
Haliotis midae	Perlemon/Abalone	0.2
Dinoplax gigas	Saddle chitin/Giant Chiton	0.7
Parechinus angulosus	Cape Sea-urchin	1.1
Turbo sarmaticus	Alikreukel/ Giant Turbo	0.1
Opisthobranchia	Sea Slugs and Nudibranchs	0.1
Crustaceans	Crabs, Crayfish and Sea-spiders	<0.1
Prosobranchia	Sea Snails, including *Turbo cidaris* and *Oxystele*	<0.1
Gynandrocarpa placenta	Commonly referred to as Elephant's ears ascidian. Large sac-like with stalk at base. Red to orange in colour.	<0.1
Annelida	Worms	<0.1

4.3.2 Quadrats

4.3.2.1 Quadrat Size, Replicate Number and Type

Three quadrat sizes were tested in order to determine the optimum size for sampling the benthic communities in the Cape Recife area. These included square quadrats of 0.0625m^2, 0.25m^2 and 1m^2. The quadrats were constructed from 8mm steel rod. The quadrats were also painted red to increase visibility against the reef. The Camera was also fitted with a framer unit with a frame of 0.25m^2.

In order to compare the data collected from the different sized quadrats, each was used to sample reefs in the same study area. The same procedure was employed for each quadrat size: The quadrat was placed haphazardly on the reef and a still photograph of the quadrat taken for visual estimates of the abundance (% cover) of each (Plate 1). The benthic biota within the quadrat was then scraped from the reef into 1mm mesh bags for later quantification. This was completed for three (3) sites at Cape Recife. Additional quadrats of 0.25m^2 were scraped at 5 metre intervals along a transect line. The total number of 0.0625m^2 quadrats scraped was 9, while 16 of the 0.25m^2 quadrats were scraped and three 1 metre quadrats were scraped. Opportunity allowed for extra 0.25m^2 quadrats to be scrapped. The scrapes were then sorted into the functional groups, identified and weighed in the laboratory. Macroalgal specimens were kept for later identification and for the preparation of voucher specimens, while the remainder of material was discarded or fixed in 4-5% formaldehyde (Farrell et al., 1993).

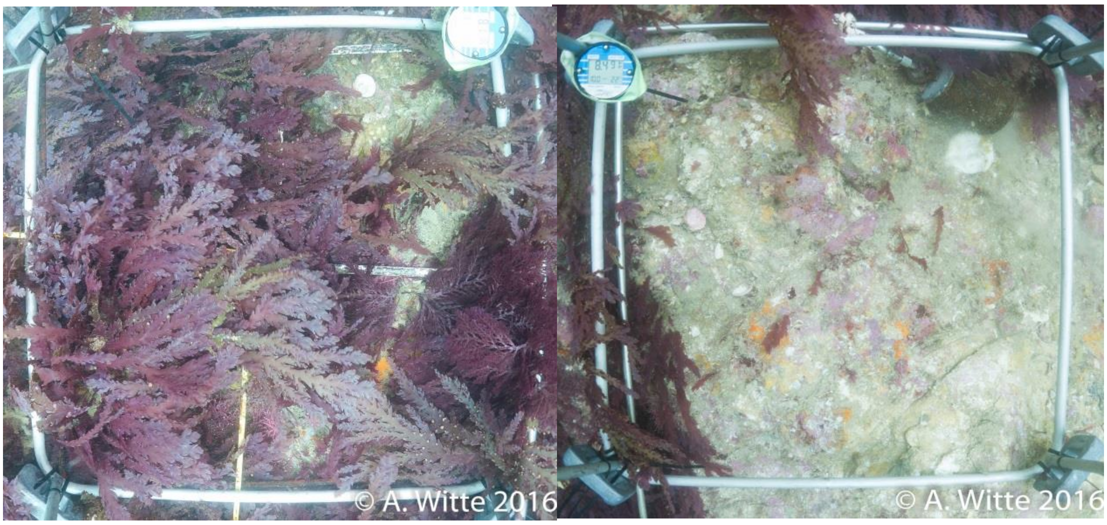

Plate 1:Before (Left) and After (Right) of a scrape site.

The quadrat size was analysed as a functional group area curve. The functional group richness was recorded for the quadrat sizes of 0.0625m^2 (n=9), 0.25m^2 (n=16) and 1m^2 (n=3). The curve was calculated using the equation (see Dengler (2009)):

$$S = cA^z$$

where S is the number of species/functional groups sampled, A is the area or quadrat size, z is the slope of the species/functional groups area relationship and c is a constants.

The total number of functional groups (see Table 6) were recorded per quadrat size and the mean number of functional groups per transect length were plotted. The resultant curve, which is similar characteristically to a species area curve, was used to determine the most suitable quadrat size following Bohnsack (1979). Due to the difficulty of identifying macrophytes to species level it was decided to use the functional group approach. However macrophytes were identified in the project and this allowed analysis of the number of macrophytes species recorded for six scrapes and associated imagery. Rare species are defined by their limited geographic distributions, habitat selection, and local density (Cao et al., 2001). Due to the restricted geographical range of this project, habitat type selectivity and the use of functional groups for the analysis, rare species were not considered in the analysis.

Photographs were enhanced by increasing the contrast, cropping and adjusting exposure (Plate 2). The images were then visually analysed to identify functional groups (Table 6). Percentage cover estimates were made for each functional group, as well as for substrate types. Emergent and visible abalone (circled in red) were also quantified in the images (Plate 2; After). As can be seen in the plate below, abalone can be very difficult to observe in an image (circled in Plate 2).

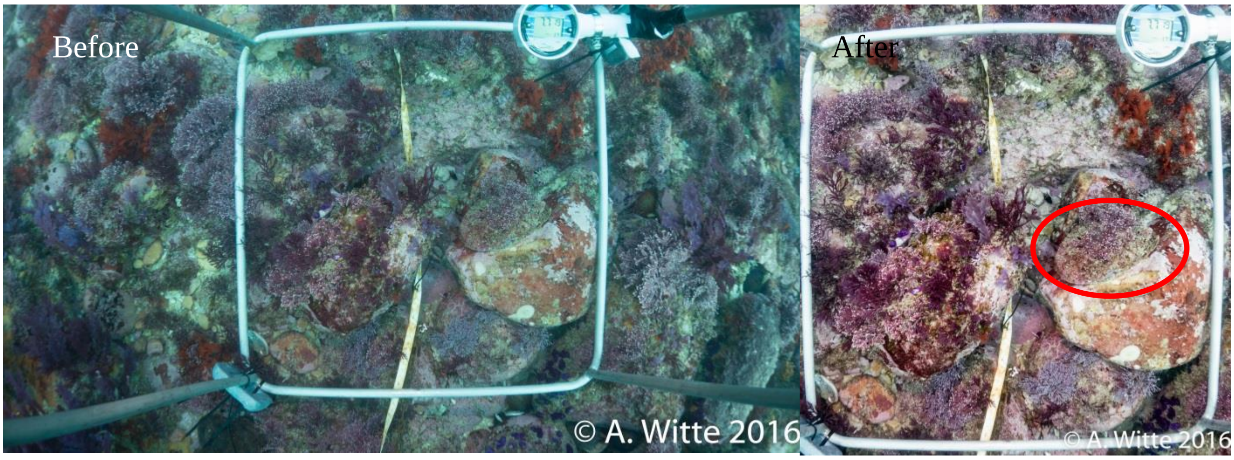

Plate 2: Picture enhancement, with abalone circled in red.

4.3.3 Transects

4.3.3.1 Functional Group Diversity

Similarly to the quadrat size, transect length was compared to see at what distance the functional group richness stabilized. Two models were used following suggestion from Dengler (2009), due to the stability of the estimated parameters, these were the:

1. The power function:

$$S = cA^z$$

2. The Logarithmic function:

$$S = c + z\log(A)$$

where S is the number of species/functional groups sampled, A is the area or quadrat size, z is the slope of the species/functional groups area relationship and c is a constants.

4.3.3.2 Habitat complexity

Habitat complexity was ranked using video footage of the transects. Each 5 metre length was ranked according to the number of crevices, holes and overhangs the area had, as well as based on the substrate type (Plate 3). Rankings were assigned according to a six point system (Gratwicke and Speight, 2005). A higher rank (5) was set for habitat that was reef with many holes and crevices and a low rank (0) was given to sandy only habitat (Table 7).

Table 7: Ranking for habitat complexity measurements for the video analysis

Rank	Description
0	Very low reef and boulders with high percentage (>91%) covered in sand
1	Low reef and boulders with medium to high (51 to 90%) sand cover
2	Low reef and boulders, few holes/crevices/overhangs, medium (21 to 50%) sand cover
3	Reef and/or boulders, few holes/crevices/overhangs, low (11-20%) sand cover.
4	Good reef profile/boulders, holes/crevices/overhangs present, very low (<10%) sand cover
5	High Reef profile, numerous holes/crevices/overhangs, No Sand.

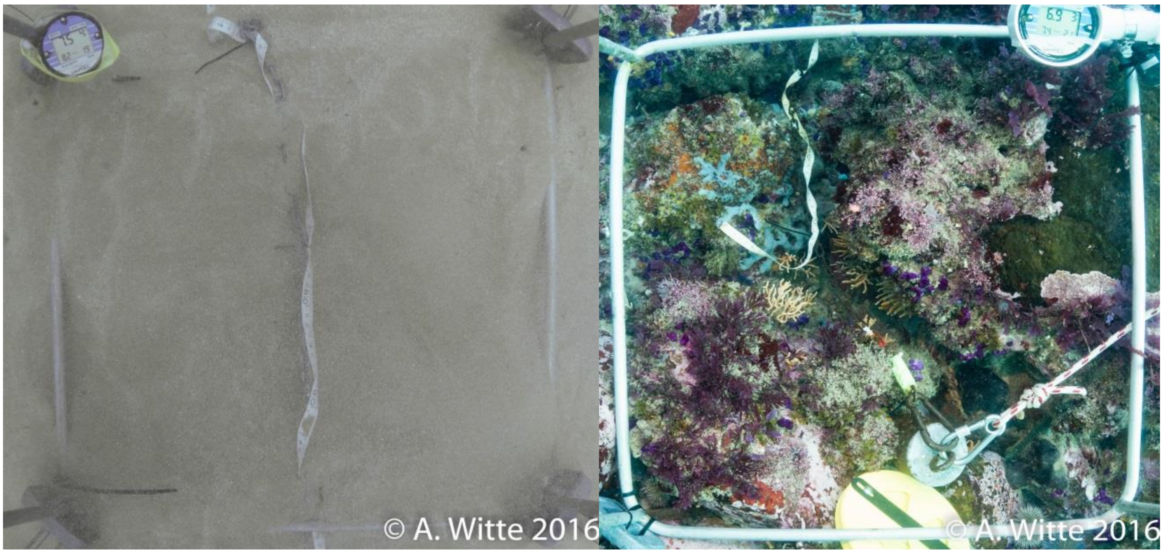

Plate 3: Quadrat Complexity; Very sandy no crevice, and low complexity (Left) and highly complex (Right)

4.3.3.3 *Abalone counts*

Counts were performed by divers using SCUBA. Emergent abalone were counted in a strip one metre left and right of the transect, for the entire length of each transect. When an abalone was encountered, divers recorded the distance (to the nearest metre) along the transect at which the abalone was observed, and if more than one was present, the total number of abalone observed at that point. No abalone were collected during these counts. Cumulative counts were compared using 5 metre intervals and the coefficient of variation was calculated for each increment. Mean counts of abalone for each transect length were recorded.

4.3.3.4 *Chain and Tape*

Rugosity was measured using the *Chain and tape* method as suggested by Mccormick (1994) and Pais et al. (2013). A 3mm link chain was rolled out over the reef ensuring the links followed the profile of the reef. The straight line distance was measured using a 50 metre metric tape measure. Maximum depth and minimum depth was recorded from the dive timer for the length of the transect.

The Surface Rugosity (SR) was calculated for each site and the maximum vertical relief (MVR) index was calculated using the following equations as per Pais et al. (2013b):

(1) $SR = \frac{Ld}{Lc}$

Where:

Ld: is the length of the straight line tape measurement along the transect

Lc: is the length of the chain along the transect

(2) $MVR = Dd - Ds$

Where:

Dd: is the deepest depth measured for the transect

Ds: is the shallowest depth measured for the transect

The coefficient of variance was used to analyse the rugosity and maximum vertical relief (MVR) measures over transects of 10, 15, 20 and 30 metres. The rugosity and MVR were also used in the correlation analyse for relationship with the total number of abalone per transect. This was done in order to determine the usefulness of the measurement for the methodology.

4.4 Analysis

Analysis was completed using r-package: *stats*, version 3.3.1 (R Core Team, 2016) and R-studio, version 0.99.903 (RStudio Team, 2015). Functions used from the package "*Vegan*" (Oksanen et al., 2015) included the *specaccum()* , was used for the quadrat size and replicates determination. *lm()* (linear model) was used in conjunction with the function *nls()* from the package "*Hmisc*" (Frank E Harrell Jr with contributions from Charles Dupont and many others, 2015) for the non-linear graphics and calculations of the power and exponential functions. The use of the *ddply()* function in package "*plyr*" (Wickham, 2011) was used to summarise and calculate means, standard deviation and standard error for datasets. Lastly the package "*MASS*" (Venables and Ripley, 2002) was used for the plotting of graphs such as barplots, line graphs and other graphics. All data was tested for normality using the *shapiro.test()* function in the r-package: *stats*, version 3.3.1 (R Core Team, 2016) as well as for *skewness()* and *kurtosis()* functions of the "moments" (Komsta and Novomestky, 2015) and "e1071" (Meyer et al., 2014) packages respectively.

4.5 Results

4.5.1 Size and Replicates

A total of 34 quadrats were scraped (9 quadrats for the 0.0625m^2, 22 quadrats for the 0.25m^2 and 3 for the 1 m^2) and associated photographs were collected per quadrat. Quadrats sizes ranged between 0.0625m^2 and 1m^2. The image data indicated an increase in the average number of functional groups from 6 functional groups in a 0.0625 m^2 quadrat to 11 functional groups in the 1 m^2 (Figure 24). Analysis of the scrape data indicated an average increase from 7 functional groups in a 0.0625 m^2 quadrat to 11 functional groups in the 1 m^2 quadrat (Figure 25).

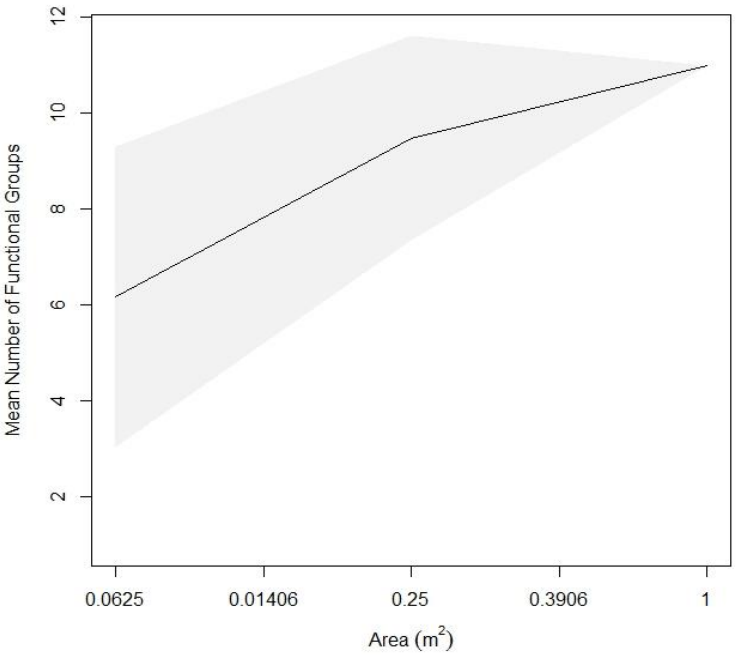

Figure 24: Mean functional group richness for different quadrat sizes determined from images (Polygon = ± SE).

A Welch two sample t-test indicated that there was a significant increase in the number of functional groups between a quadrat size of 0.0625m^2 to a 0.25m^2 quadrat for both the scrape ($t = -2.119$, $df = 16.271$, $p = 0.05$) and image data ($t = -4.679$, $df = 15.456$, $p < 0.001$), however no significance was found between 0.25m^2 and 1 m^2 quadrats of scrapes ($t = 0.405$, $df = 3.148$, $p = 0.7$) or images ($t = 1.719$, $df = 2.462$, $p = 0.2$). There was also a significant difference in the number of functional groups between scraped quadrats and photographic quadrats of 0.0625m^2 ($t = 3.244$, $df = 15.53$, $p = 0.005$, $n = 9$), but no significance was found between quadrat class of 0.25m^2 ($t = 0.889$, $df = 41.386$, $p = 0.379$, $n=22$) scrapes and images nor for the 1 m^2 class ($t = -1$, $df = 3.2$, $p = 0.387$, $n=3$) of scrapes and images.

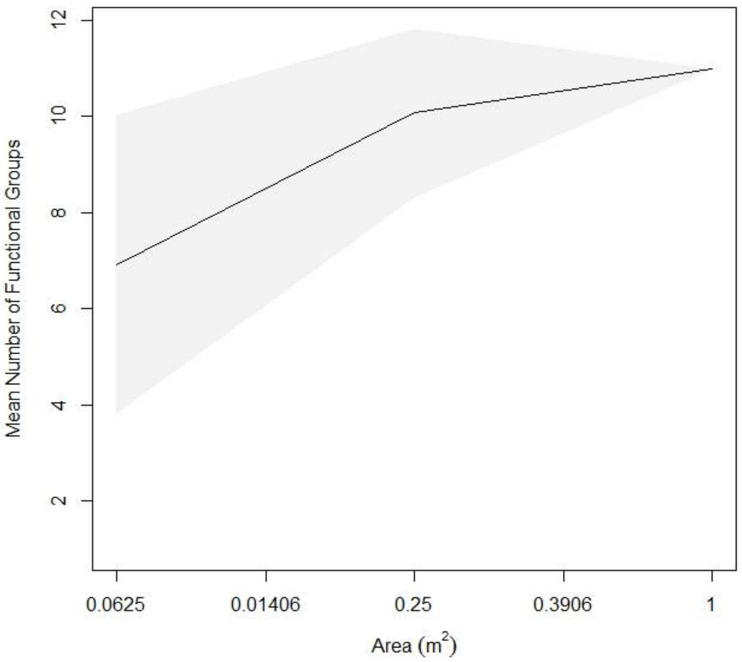

Figure 25: Mean functional group richness for different quadrat sizes determined from scrapes (Polygon = ± SE).

As shown in Figure 26 the number of replicates required to accurately represent the number of functional groups for the Cape Recife area is from 5 to 10 replicates of 0.25 m^2 quadrats (df=19; f = 3.312; p=0.08 n=22

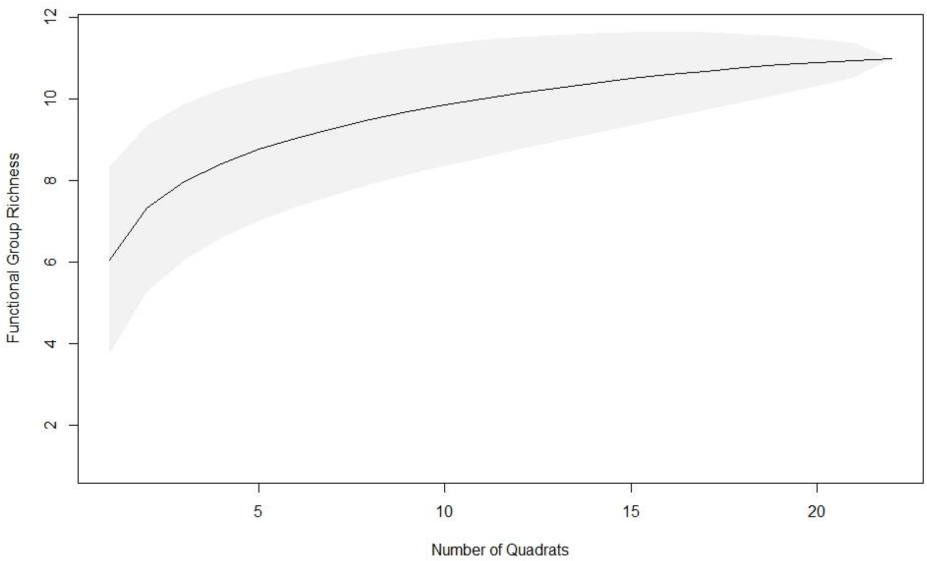

Figure 26: Mean number of functional groups recorded per quadrat (0.25m^2) (polygon = ± SE, n=22) plotted against the number of quadrats sampled.

4.5.2 Scrape data versus photographic data

The mean species richness for the macrophytes increased from 4 individuals in a quadrat of $0.0625m^2$ to 6 in a quadrat of $0.25m^2$ and remained the same for a $1m^2$ quadrat (Figure 27). There is a significant difference in richness between all images and scrapes (t = -3.691, df = 10.76, p = 0.004, n=9) however only the $0.25m^2$ showed significant difference (t = -4, df = 3.2, p = 0.03, n = 3) between images and scrapes. There is a significant difference in richness between quadrat sizes of $0.0625m^2$, and $0.25m^2$ (t = -3.503, df = 5, p = 0.02, n = 6). No significant difference between quadrats of $0.25m^2$ and $1m^2$ (t = 1.581, df = 5, p = 0.2, n = 6) was found.

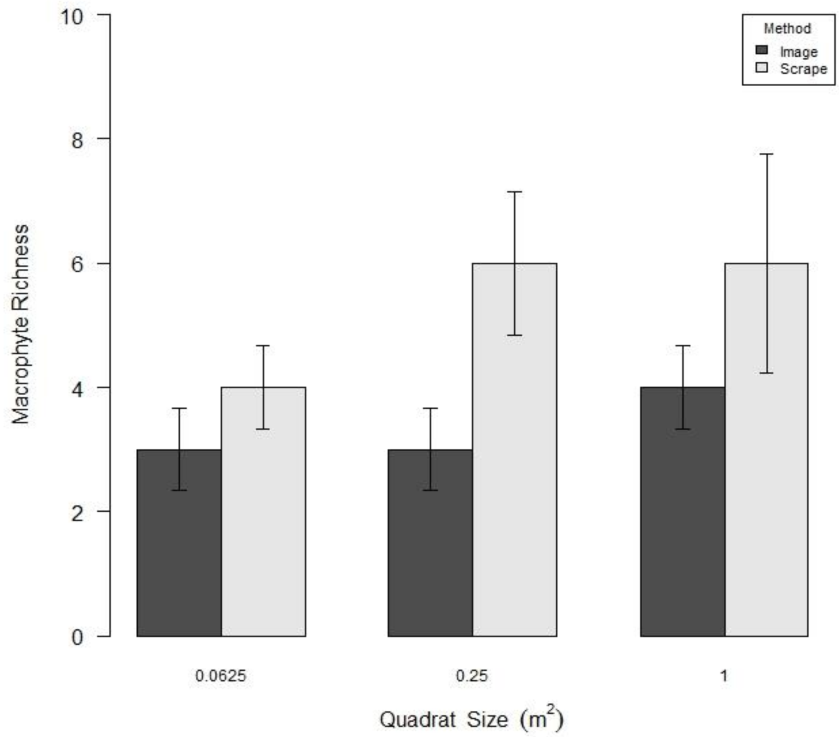

Figure 27: Mean species richness for macrophytes (±SD, n=3) for $0.25m^2$ quadrats assessed by scrapes and Images respectively

The Coefficient of Variance (CV) values for the different size quadrats and the method of quadrat (images or scrapes), show that the visual quadrats (Scrapes) as compared to the images have greater precision and that quadrat size also affects the precision of the method, however no significant difference (t = 1.660, df = 57.203, p-value = 0.103, n = 34) was found between the CV values of the Images as compared to the scrapes. The largest difference of 0.17 units was found for the CV values of the $0.0625m^2$ quadrat, while the 0.25 quadrat only had a difference of 0.04 for scrape and image data. The quadrat size of 0.0625 m^2 showed the greatest CV values, 0.2 and 0.37 for the image and scrape respectively, while the lowest CV values were seen for the 1 m^2 quadrats, 0.13 and 0.2 for the image and scrape respectively (Figure 28).

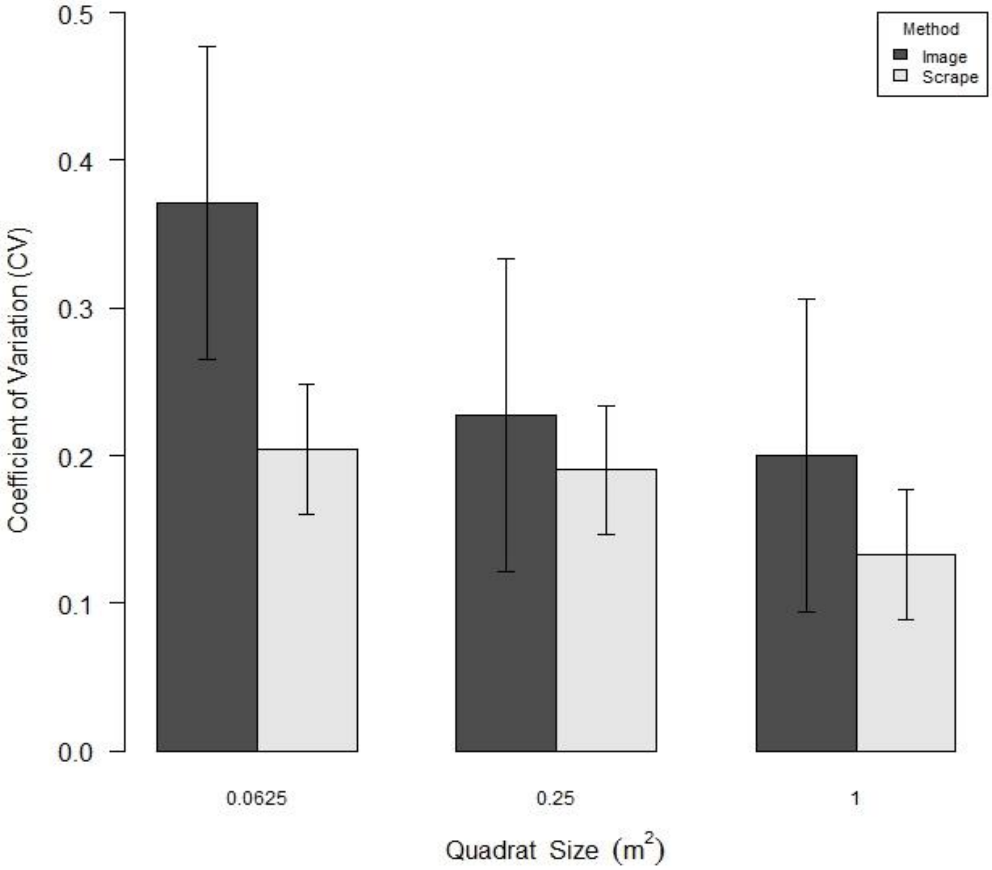

Figure 28: Mean functional group coefficient of variance (CV) (±SE, n=34) for three quadrat sizes assessed by scrapes and Images respectively

4.5.3 Transect Length

4.5.3.1 Functional Groups

The cumulative mean number of functional groups encountered along the transect was plotted against transect length (Figure 29 Left) to determine what length of transect would suitably represent the area. Both a power and exponential function were fitted to the data and in both cases good fits were obtained (r^2 = 0.97, t = 30. 39, df = 29, p <0.005) and (r^2 = 0.95, t = 20. 98, df = 29, p <0.005) respectively (Figure 29).

There is a significant difference (t = 14.755, df = 30, p<0.005, n =22) in CV value between the 1st metre of the transect and the 31 metre. The CV was lowest between 25 metres and 29 metres with a CV of 0.11 (±0.01), while the first metre had the highest CV 0.39 (±0.01) (Figure 29 Right).

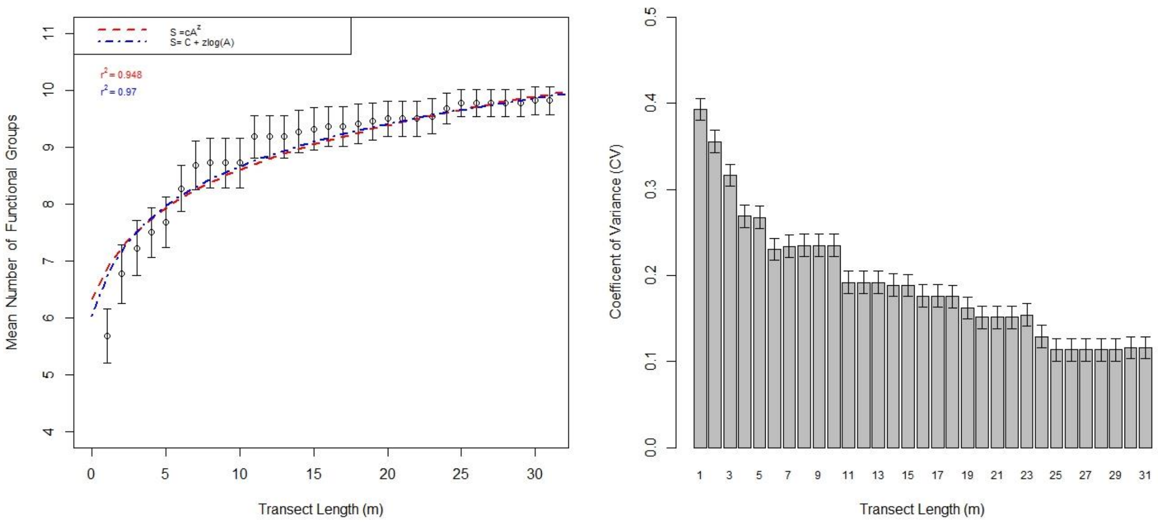

Figure 29:(Left) Mean number of functional groups (±SE, n = 22) for different transect lengths dotted lines represent power (red) and exponential (blue) response curves fitted to the data, (Right) Mean Coefficient of Variance (±SE, n = 22) for different transect lengths.

4.5.3.2 Habitat complexity

Figure 25 shows the mean habitat complexity values, based on ranking scores, along the transects. The mean habitat complexity rank was 3 with a maximum standard error of ± 0.27 at five metres and a minimum of ±0.24 for a 30 metre transect length (Figure 30).

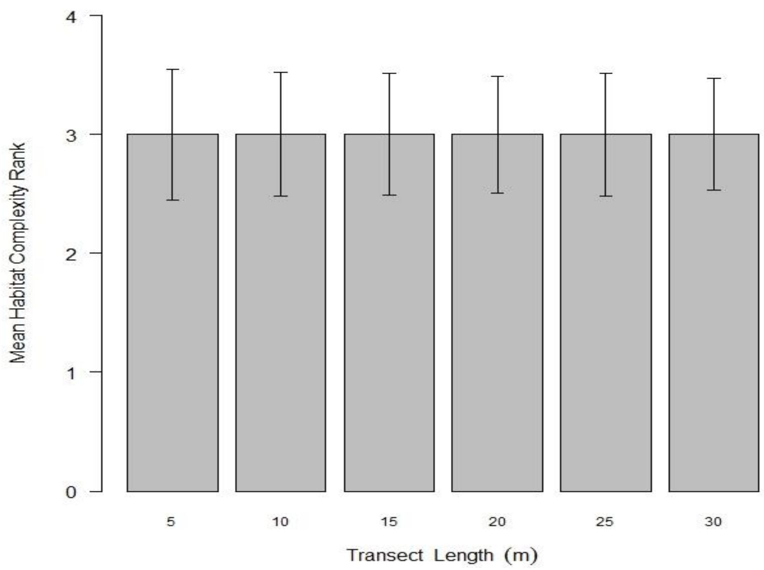

Figure 30: Mean Habitat complexity ranking (±SE, n=30) for transect lines between 5 and 30 metres.

There is no significant change in the precision (CV values) with transect length ($t = 0.503$, $df = 5$, $p = 0.636$, $n = 30$), although a decreasing trend is evident (Figure 31) for the habitat complexity ranking as transect length is increased.

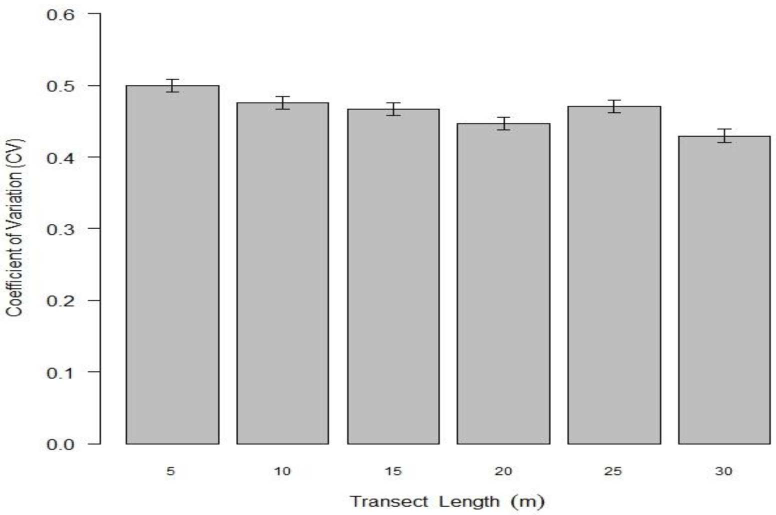

Figure 31: Habitat complexity Coefficient of Variation (CV) (±SE, n=30) for transect lines between 5 and 30 metres.

4.5.3.3 Abalone Counts

Counts took place for 45 transect lengths of which 21 transects yielded a zero count for the length, meaning only 53% of the transects intercepted abalone. A linear regression analysis (Figure 32) of the average cumulative abalone count along the transects indicates a strong linear relationship ($r^2=0.975$, $p<0.005$, $n = 45$) between the distance and the mean cumulative abalone count. As expected the further a diver swam the greater the number of abalone per transect line were counted.

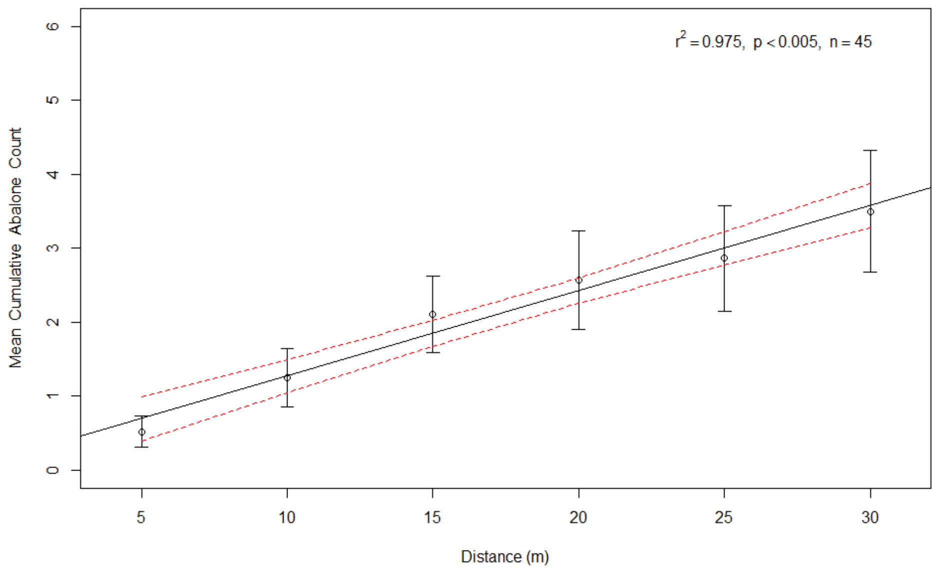

Figure 32: Mean Cumulative Abalone Counts (±SE, n= 29) for five (5) meter intervals along the transects. Solid line = best fit linear regression. Dotted line = 95% confidence interval for the regression.

In comparison the mean abalone counts per transect (Figure 33) showed an increase in the mean number of abalone between 5 metres and 15 metres. From 15 metres onward the mean abalone per unit length remained at about 0.2 abalone per line (Figure 33). There was no significant difference between the 5 metre transect length and the 30 metre transect length ($t = -1.192$, $df = 37.57$, $p = 0.241$, $n = 20$).

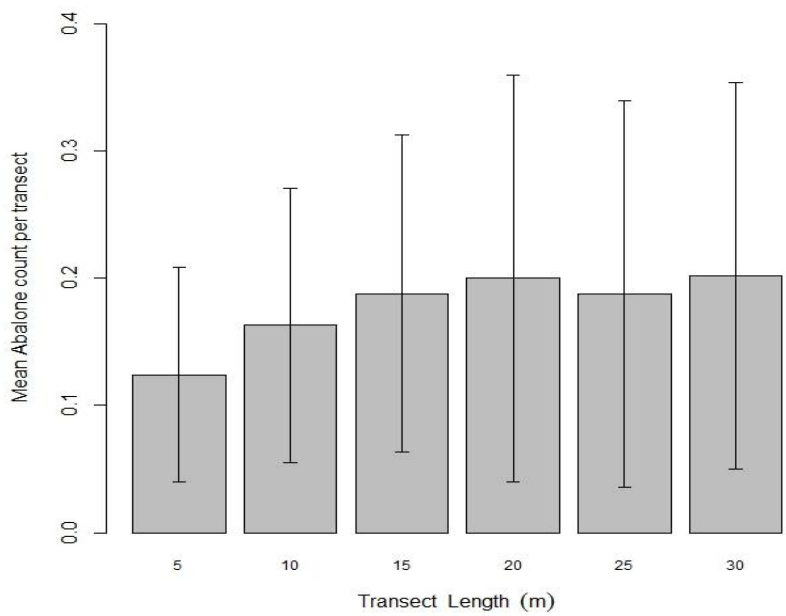

Figure 33: Mean Abalone Counts (±SE, n= 29) for transects between 5 metres and 30 metres in length.

Figure 34 shows the coefficient of variance for the abalone counts with increasing transect length. There is a significant decrease in the coefficient of variance (t = 4.1303, df = 4, p-value = 0.014, n = 29) from 5 meter interval to the 15 metre interval. The initial CV values decreases from 2.62 at 5 metres, down to 2.08 at 10 metres and then the CV values from 15 metres onwards range between the minimum of 1.55 (at 30 metres) and a maximum of 1.71 (at 20 metres).

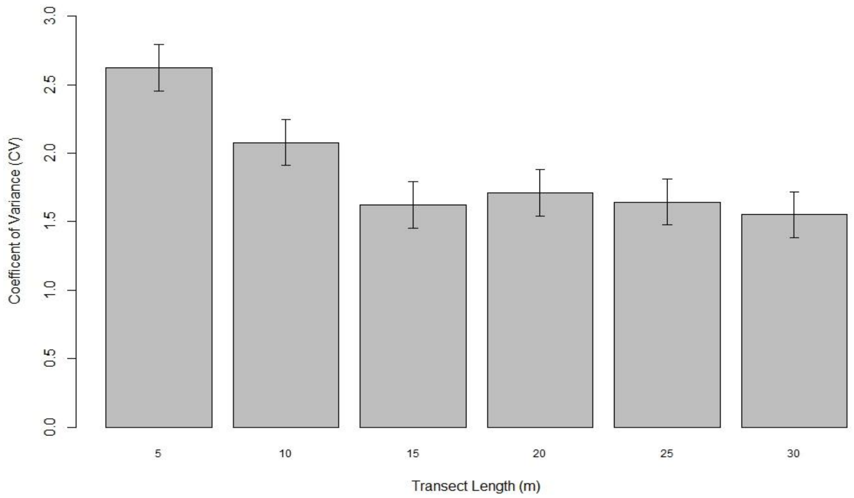

Figure 34: Coefficient of Variation (CV) (±SE, n = 29) for mean abalone counts at different transect lengths.

4.5.3.4 Rugosity and Maximum Vertical Relief (MVR)

The coefficient of variation for the rugosity values showed a greater precision for the shorter 10 and 15 metre transect lengths as compared to the 20 and 30 metre transects (Figure 35). No significant difference was found between the CV value for all compared transect lengths (t = 1.418, df = 3, p = 0.251, n=45) but the lowest CV value (0.084) was found for a 15 metre transect length.

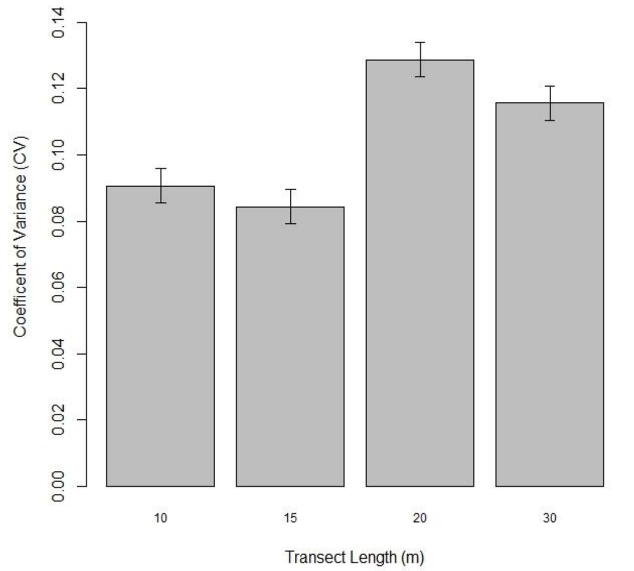

Figure 36: Coefficient of Variation (CV) (±SE, n = 45) for mean rugosity at different transect lengths.

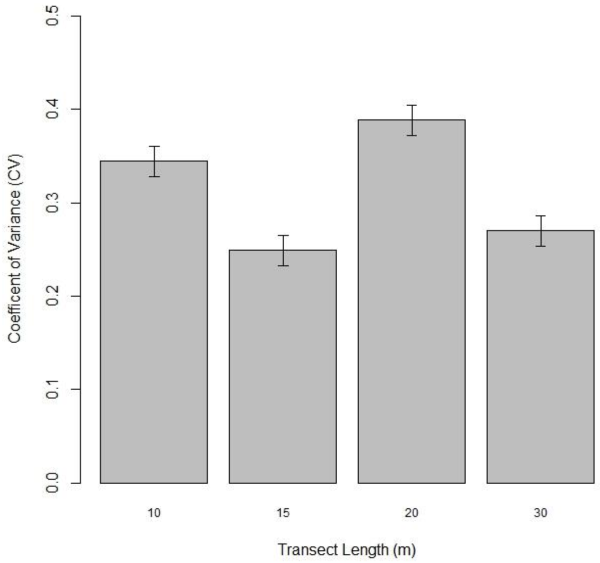

Figure 35: Coefficient of Variation (CV) (±SE, n = 45) for mean maximum vertical relief (MVR) for different transect

The coefficient of variation for the maximum vertical relief values showed no pattern to the length of transect, as the lowest CV value, and thus the highest precision was 0.25 ± 0.02 SE for a 15 metre transect and the maximum CV of 0.39 ± 0.02 SE for a 20 metre transect length. The 30 metre transect also had a very low CV of 0.27 ± 0.02 SE (Figure 36).

Correlation analysis of abalone numbers to rugosity show that there is a very weak but significant correlation between them ($t = 3.743$, $df = 43$, $p < 0.001$, $r = 0.241$, $n = 45$) (Figure 37 Left). Similarly correlation analysis between MVR and abalone numbers also showed a very weak but significant relationship ($t = 4.253$, $df = 44$, $p < 0.001$, $r = 0.291$, $n = 45$) (Figure 37 Right).

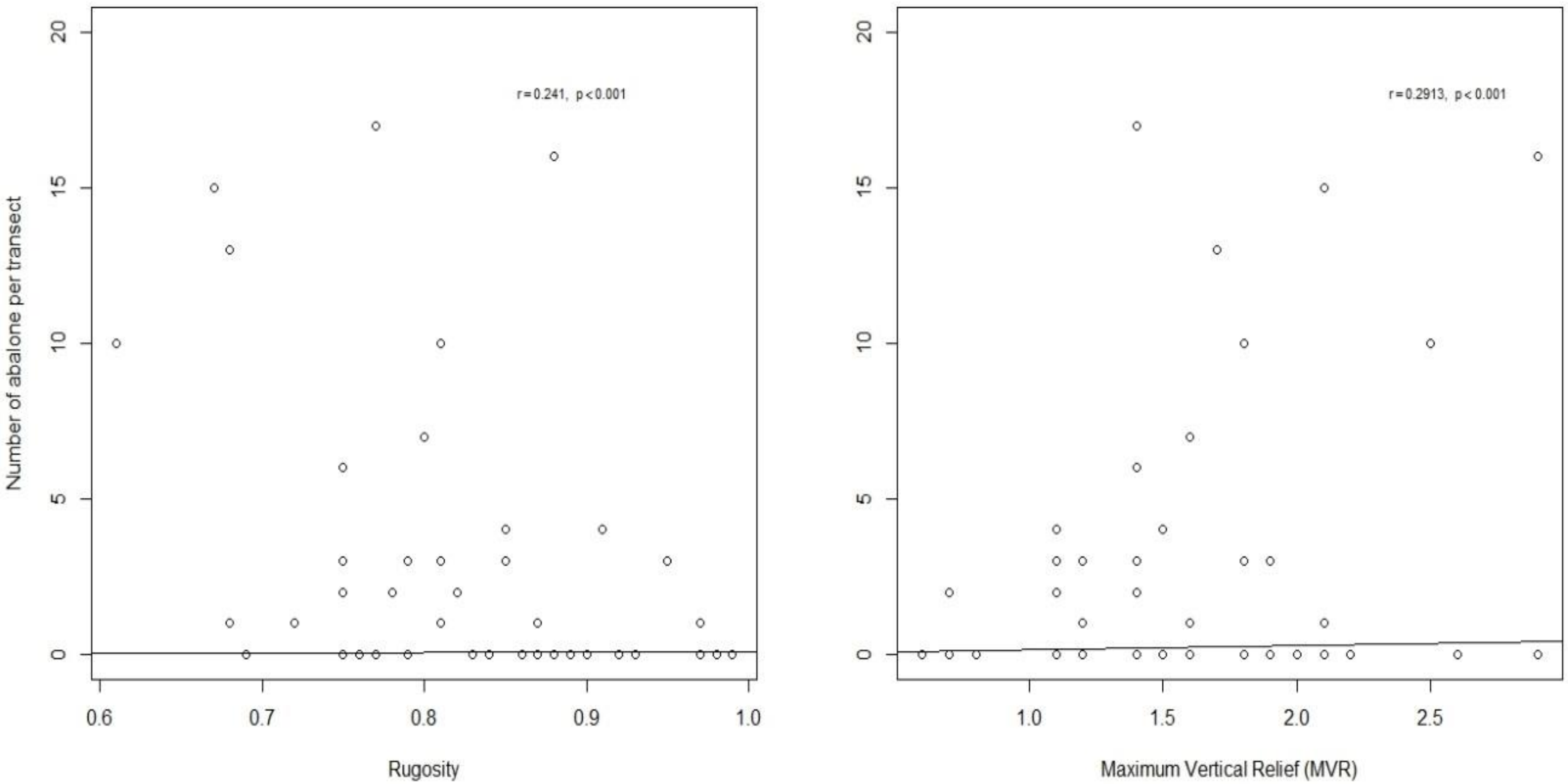

Figure 37: (Left) The mean number of abalone counted per transect length and the relationship to rugosity (n=45). (Right) The mean number of abalone counted per transect length and the relationship to the maximum vertical relief (n=45).

4.6 Discussion

The main aim of the research was to determine the optimal habitat sampling procedure for describing habitat for the abalone ranching project. Comparison was made between quadrat sizes, the number of replicates required for the optimum quadrat size and comparison of data collected between an imagery approach and a scrape (physical collection) approach for quadrats. It was also aimed to determine what length a transect should be in order to suitably analyse the reefs for the ranching operation.

4.6.1 Quadrat size and replicates required

The results revealed that quadrats greater than 0.25m^2 were optimal in describing the functional group richness for the habitat sampled, and that between five and ten replicates allows for enough representation of the number of functional groups within the habitat. There is a strong curvilinear relationship between quadrat size and sample size (replicates), but so; as the quadrat size increases, the number of samples required is less (Downing and Anderson, 1985; Parravicini et al., 2009; Pringle, 1984).

The resultant coefficient of variance of this study showed that there was an increase in the precision of the data with quadrat size for both scraped data and photographic. This was similarly found by Pringle (1984), where the precision increases with quadrat size and number but the efficiency of the sampling procedure decreased. He also noted that the sampling units are greatly affected by the size of the organism/s to be studied (Pringle, 1984). For example much of the seaweed research is done using a 0.25m^2 quadrat (Calumpong et al., 1999; Díaz et al., 2011; Engelen et al., 2005; Marinho-Soriano et al., 2001; Murray et al., 2001; Phillips et al., 1997; Sayer and Poonian, 2007; Thakur et al., 2008; Vis et al., 2003), whereas coral research has focused on larger quadrat sizes of 1m^2 and greater. (Dumas et al., 2009; Leujak and Ormond, 2007; Sato et al., 2011; Sayer and Poonian, 2007), furthermore studies on fish and other larger organisms have moved to belt transects or very large quadrat sizes (more than 10mX10m) (Buxton and Smale, 1984; Carr, 1989; Sayer and Poonian, 2007).

Various quadrat sizes have been suggested for sampling and have been utilized, depending on the focus and aims of the studies and objectives of the studies, the subject organisms, and practical considerations (Brown et al., 2004; Downing and Anderson, 1985; Leliaert et al., 2000; Parravicini et al., 2009; Pringle, 1984). Small quadrats of less than 0.09 m^2 were usually employed to target a specific species or taxon (van Rein et al., 2011), while quadrats greater than $1m^2$ have been used for more general community sampling (van Rein et al., 2011). van Rein et al. (2011) further indicated that there have been various estimates and suggestions on the number of replicates, ranging from 5 to 30 replicates per area depending on the objectives, site and resource availability of the monitoring programme.

Vance (2014) worked in various areas, including the intertidal and subtidal zones to 15 metres deep along the coast of Plymouth, UK. The area is cold temperate (Williams and Conway, 1984). Vance (2014) considered a quadrat size of $0.25m^2$ to be the best compromise and instructed divers to do 5 replicates per site. He found that quadrat size directly impacts the number of species observed in a sample. The results obtained in this study are similar to those of Vance (2014), as our results indicate quadrats of $0.25m^2$ with between 5 – 10 replicates would be suitable to adequately sample the ecosystems and provide a standardized method that can be used to monitor and assess the subtidal reefs for ecological changes associated with ranching activities in the area. Pringle (1984) also determined that, in particular for seaweeds in temperate waters, the greatest precision and efficiency is obtained with a 0.25 m^2 quadrat.

4.7 Photographic quadrats compared to scrape data

One of the largest issues with the use of imagery is the assumed loss of information. Often smaller understorey species are covered by canopy species and are missed in image analysis. Parravicini et al., (2009) found that the method of collection of data did not matter as greatly as the size of the quadrats used in the Mediterranean Sea. They suggest that 10 images with a 250 cm^2 frame sufficiently represents the equivalent of a 2500 cm^2 quadrat visually assessed (Parravicini et al., 2009). The results from this study indicate there is a significant difference in the number of macrophyta species observed in the images as compared to the scrapes. It is noted that there was no significant difference for image and scrape quadrats of 0.25 m^2 and $1m^2$. This is most likely due to a greater area of analysis (Parravicini et al., 2009). The use of imagery techniques with greater replication makes it possible to capture sufficient information with regards to the community structuring and species richness (Murray et al., 2001).

4.7.1 Advantages and disadvantages of the imagery approach

Many authors support the use of photographic sampling techniques for various reasons, with the greatest of these being the minimal destructive/no impact factor of the sampling technique (Celliers et al., 2007; Dumas et al., 2009; Leujak and Ormond, 2007; Murray et al., 2001; Parravicini et al., 2009; Sayer and Poonian, 2007; Tkachenko, 2005; van Rein et al., 2011; Vance, 2014; Wilding et al., 2007). It has been consistently compared to visual/scrape methods (Dumas et al., 2009; Kibele, 2016; Parravicini et al., 2009; Roberts et al., 1994) with the general consensus that the use of imagery is comparable to other conventional methods, with the advantage of a permanent record of the collected data (Roberts et al., 1994). Scrapes are particularly damaging to the reef as all biota is removed (Plate 1). Furthermore the use of images has been popular due to the overcoming of limitations presented in having limited bottom time on deep dives, having to train divers for in-field identifications, or to have to scrape sample from the reef and eliminates to some degree the observer subjectivity (Dumas et al., 2009; Kibele, 2016; Murray et al., 2001; Roberts et al., 1994; Sayer and Poonian, 2007; van Rein et al., 2011). The use of images and video can produce more data for a lower sampling effort (Dumas et al., 2009; Murray et al., 2001), and the files can be stored and re-analysed at later stages particularly through the use of imagery analysis software (Parravicini et al., 2009).

A number of disadvantages are well known with imagery techniques. The first is the loss or underestimation of non-canopy forming species (van Rein et al., 2011) where particularly dense communities such as seaweed beds, mussel beds and colonial invertebrate structures dominate the canopy (Murray et al., 2001). This is attributed mainly to the fact that images are only 2-dimensional (Murray et al., 2001), although this can be overcome to a degree with multiple images of the same site (Parravicini et al., 2009). Furthermore the presence of small and “difficult-to-see” species is also often compromised depending on the camera resolution, shutter speed, aperture and sensitivity settings, condition of the water and habitat complexity (Dumas et al., 2009; Murray et al., 2001; van Rein et al., 2011). The use of imagery also greatly constrains the user in the plot size and shape for analysis, as the image needs to be captured as clearly as possible thus restricting the distance the camera can be from the reef (Murray et al., 2001).

4.7.2 Transect Length

Traditionally longer transect lengths have been favoured for analysis in literature (Brown et al., 2004; Houk and Van Woesik, 2006; Leujak and Ormond, 2007; van Rein et al., 2011). Murray et al., (2001) noted that for species analysis, particularly for a clumped distribution pattern, the longer the transect length, and the more space between sampled points along the transect, the more accurate the result will be. van Rein et al., (2009) in their review made a similar statement, although they argue that this is better for biotope analysis rather than for species level analysis whereby shorter transect lengths can be utilized. Houk and van Woesik (2006) found that for analysing corals the use of a 50 metre transect videoed with 60 frames allowed for detection of up to 30% relative change with 90% confidence.

Results from this research indicated for habitat complexity ranking, that while the mean complexity rank remains fairly constant (Rank 3) between transect lengths, the precision (coefficient of variance) of the sample generally decreases as transect length is increased. Abalone, and many mobile invertebrates are known to associate with more complex habitats (Jalali et al., 2015; Zeeman et al., 2012). Probable bias was introduced as there was site selectivity of predominantly unbroken reef environment where crevices and hole indicators do not have a significant role in the assessment process. There is also a significant but weak correlation between abalone and rugosity and maximum vertical relief (MRV). According to Jalali et al. (2015) rugosity, bathymetry and habitat complexity are the three most important factors driving abalone distribution in Australia. The results confirm that rugosity and MVR are suitable measures for indicating habitat suitability but the MVR value showed very high variability in the CV values when compared to transect length. This is mainly due to changes in substrate types and their representation in the sample (Pais et al., 2013b). Rugosity, which showed a better precision for a 10 and 15 metre transect line, is well known to be insensitive to highly corrugated and tall structures (Mccormick, 1994; Pais et al., 2013a, 2013b). Thus coupling rugosity with the MVR can increase the selective power of the sampling and accuracy of describing the habitat (Pais et al., 2013a).

The results of this study indicate that for abalone, accumulative counts showed a linear relationship to the length of the transect, while the mean abalone counts showed a power-like function response to transect length where the mean number of emergent abalone per transect remained at about 0.2 abalone per metre. Tarr et al., (2000) used transects of 50 metres in their analysis of abalone abundance and found a general decrease in the abundance of abalone in all fisheries zones across South Africa. This was further confirmed by Godfrey (2003), who found that for Cape Recife, from 1998 emergent abalone numbers decreased from 125.6 to 53.8 individuals counted in 20 minutes in 2001, and the emergent abalone density declined from 1.3 to 0.8 m^{-2}over about the same period attributed to the poaching of the animals (Godfrey, 2003; Raemaekers and Britz, 2009). In Australia, Mcgarvey et al., (2008) found approximately 0.069 abalone individuals per square metre, with 21 of the 64 transect lines having a zero count. Hart et al., (2013a, 2013b) found that through stock enhancement in Australia the abundance of abalone increased to about 2.5 individuals m^{-2}, (they used 30 metre transects to do abalone counts) and found approximately an 8 individual m^{-2} carrying capacity for the reefs analysed.

The CV value for this study showed an inverse response to the mean abalone per m^2 counts, with transects greater and equal to 15 metres having greater precision than the transects less than 15 metres, although due to the low mean abalone counts the CV value was very highly indicative of a wide variance. Tarr et al. (2000) found to obtain a 23 to 26% (0.23 to 0.26) coefficient of variance a total of 124 transect of 50 metre length had to be completed in assessing the abalone abundance in fisheries zones of South Africa. Gerrodette (1987) explains that the use of the CV value reflects the measurement error and can be controlled by expanding or minimizing sampling effort. He set his CV limit at 0.2 and found a higher CV value means that the sampling effort was not great enough for the organism being sampled. Gerrodette (1987) also found that if the rate of change per unit time or distance (r) from a linear or exponential model is high then the CV value is normally also low.

For functional group numbers a similar pattern was observed in the results where the precision of the functional group counts increased in an exponential function and the CV decreased inversely. This illustrates that the longer transect lengths are more appropriate for assessing the area in terms of the functional group richness. Phillips et al., (1997) found that the use of functional group analysis for highly variable communities resulted in a high degree of information loss and did not recommend this approach for community analysis. Contrary to this van Rein et al. (2011) found that the use of groups removes large amount of community variability and recommends the functional group approach but this is dependent on the aims of the study.

Furthermore although rare species were not considered in this study due to the sampling methods focusing predominantly on functional groups, as well as the limited geographic range and selectivity bias; they are indeed important and very prolific in the marine environment and should be considered when analysing and determining diversity and community aspects (Fontana et al., 2008) and the use of rare species depends on the sensitivity of the assessment and criteria of the project (Cao et al., 2001).

4.8 Conclusion and recommendations

This research was aimed at determining the optimum sampling procedures for the abalone ranching operations at Cape Recife. The researched focused on the use of transects and quadrats, as well as the use of imagery techniques as compared to physical scrape collection. Results indicate that the optimum quadrat size among those tested is $0.25m^2$. It also indicates that 5-10 replicates of a $0.25m^2$ quadrat provide sufficient representation of the functional groups in the habitat.

The optimum transect length was 15 metres. In terms of the functional groups there is no significant increase in the number of functional groups for transects greater than 15 metres. The abalone counts showed no significant difference due to the high variability of the counts, however the mean number remained at approximately 0.2 abalone per transect length for transects greater than 15 metres. The CV was also lowest for transects greater than 15 metres. Rugosity showed poor precision (high CV values) for transects greater than 20 metres, while the MVR showed no pattern but did have the lowest CV value (greatest precision) for the 15 metre transect length. Both rugosity and MVR showed a significant but weak correlation to abalone counts. It has been well documented that abalone associate strongly to these reef characteristics (Jalali et al., 2015), and therefore they are important and integral in the methodology.

It is also recommended that measurements of rugosity using the chain and tape method are completed for shorter sections of a transect rather than recording a single measurement for the total length of the transect. Furthermore that the maximum vertical relief (MVR) should be recorded for each transect line. This allows for a greater accuracy in the reef assessment and identification of suitable abalone habitat.

No environmental variables, such as salinity, temperature, turbidity and sediment loading were measured for this study but it is recommended that further research is conducted in the effect of environmental conditions on the ranching methodology. Areas of concern include the influence of water movement and wave action on the day as if these are too high feeding does not take place (Wood, 1993), sediment loading as abalone are highly sensitive to sand as it can inhibit abalone from proper settlement in an area (Godfrey, 2003; Wood, 1993), water contaminants, nutrients and temperature which are often associated with upwelling events and link directly to water turbidity in the region (Götz et al., 2009), as well as link directly to food availability. All these variables need to be incorporated into an effective sampling protocol to give a more holistic view on the suitability of a site for ranching activities.

Chapter 5. Ecology and community aspects of shallow subtidal Agulhas reef between Cape Recife and Skoenmakerskop.

5.1 Introduction

The South African coastline is 3650 km long. It consists of 42% sandy beach, 31% mixed shore and only 27% rocky shore and hosts more than 12 914 species (Griffiths et al., 2010). Large portions of this coastline remain poorly explored, particularly in the shallow subtidal regions (Anderson et al., 2005; Anderson and Stegenga, 1989; Bolton et al., 2004). Without adequate baseline information, the selection of areas that would be suitable for abalone ranching is limited, and the monitoring of ecological impacts associated with abalone seeding is not possible.

Five different abalone habitat types were described by Wood (1993). These ranged from areas in the intertidal zone down to deeper subtidal reefs ranging up to 3.8 metres deep. Abalone distribution is mainly limited by substrate type (Godfrey, 2003), as they prefer rocky areas and tend to avoid sandy environments (Wood, 1993). Abalone are most commonly found in shallow reef areas, close to the shore, and seaweeds, as they are trap feeders (Godfrey, 2003) and feed mainly on *Hypnea spicifera, H. rosea, Plocamium corallorhiza, Ralfsia expansa* and *Calliblepharis fimbriata* while they appear to avoided algal species including *Caulerpa filiformis*, *Gelidium amansii*, *Gracilaria beckeri*, *Portieria hornemannii*, *Corallina* sp. and *Laurencia flexuosa* (Godfrey, 2003; Wood, 1993; Wood and Buxton, 1996).

Abalone habitat tends to be patchy, which results in abalone distribution being sparse and patchy, often leading to inaccurate abundance estimates (Godfrey, 2003; Wood, 1993). The subtidal reefs off Cape Recife are structurally and geologically variable, which creates very complex and diverse subtidal systems (Godfrey, 2003), which make it difficult to predict the occurrence of suitable abalone habitat on a local scale.

The broad aim of this chapter was to further evaluate areas that were predicted as potentially suitable for abalone seeding by the GIS model.

The objectives of this section of the study were to:

1. Find relationships between abalone abundance and environmental variables such as depth, rugosity and substrate.

2. Seek associations between abalone abundance and indicator species / functional groups, which could be used by ranchers to select suitable sites for seeding.

3. Determine the relationship between cover estimations and biomass in order to provide an estimate of feed availability in the area, and

4. Provide information which could serve as baseline data for future monitoring by describing species richness and diversity of the benthic algal community in the ranching area.

The following hypotheses are linked to these objectives:

i. Abalone are directly influenced by depth and substrate, and also associate with areas of greater diversity and complexity.

ii. Abalone will show a positive association with high foliose algal cover

iii. Percentage cover is correlated with biomass and is a suitable proxy to estimate food source availability.

5.2 Methods

The benthic community structure was sampled with the aid of transects and quadrats. While most of the data collection was done through photographic and video imagery of the reefs, some data was also collected from quadrat scrapes. The methods used in this portion of the research were informed by the outcome of the assessment of sampling methods, which was described in Chapter 4.

5.2.1 Transects

Transects (n=45) were deployed at various sites (Figure 38) across the demarcated concession zone of Lidomix Investments. A video recording of the benthic community along the transect was made by a diver swimming along the transect. Counts were done for abalone in a 1 meter strip on either side of the transect line. Rugosity was also recorded for each length of the transect, using the chain-and-tape method. For a more detailed description of the methodology see Chapter 4.

5.2.2 Quadrats

Based on the species area curves done for functional groups (Chapter 4, Table 6) the benthic community was sampled with 0.5 x 0.5 metre ($0.25m^2$) metal quadrats. Quadrats that were photographed and scraped of all biota, and then re-photographed after scraping (Plate 1). Scrapes were stored and cold preserved for a maximum of three days in a cold room at below 5°C. Scrapes were then sorted in the laboratory (see 5.2.2.1). Those that could not be sorted within 3 days, were preserved in 3% formaldehyde seawater and analysed later. Images were visually analysed for cover according to functional groups identified (see section 4.3.2)

5.2.2.1 Biomass and Cover Relationships

Scraped samples were separated into subsamples representing the different functional groups (Table 6). The functional groups of *Plocamium corallorhiza*, *Laurencia flexuosa*, the articulated corallines, coralline turf, were chosen due to the high presence of biomass in each scrape sample. *Plocamium corallorhiza* was also selected due to this species being a very important food source in abalone diet (Wood 1993). Foliose seaweeds biomass was very low (<0.5g) in all scrape samples and could not be observed in the images. They were therefore not included in the biomass/cover relationship analysis. Each subsample was shaken to remove any excess water and weighed using a balance accurate to 0.1 grams to obtain biomass values for the different functional group in each quadrat. Voucher specimens were retained and macroalgal vouchers lodged with the NMMU Herbarium (PEU) algal collection. Cover, as percentage of the quadrat area, was estimated from the photographic images and results were compared with biomass data using the cor.test () function and lm() function from r-package: *stats*, version 3.3.1 (R Core Team, 2016) in R-studio, version 0.99.903 (RStudio Team, 2015).

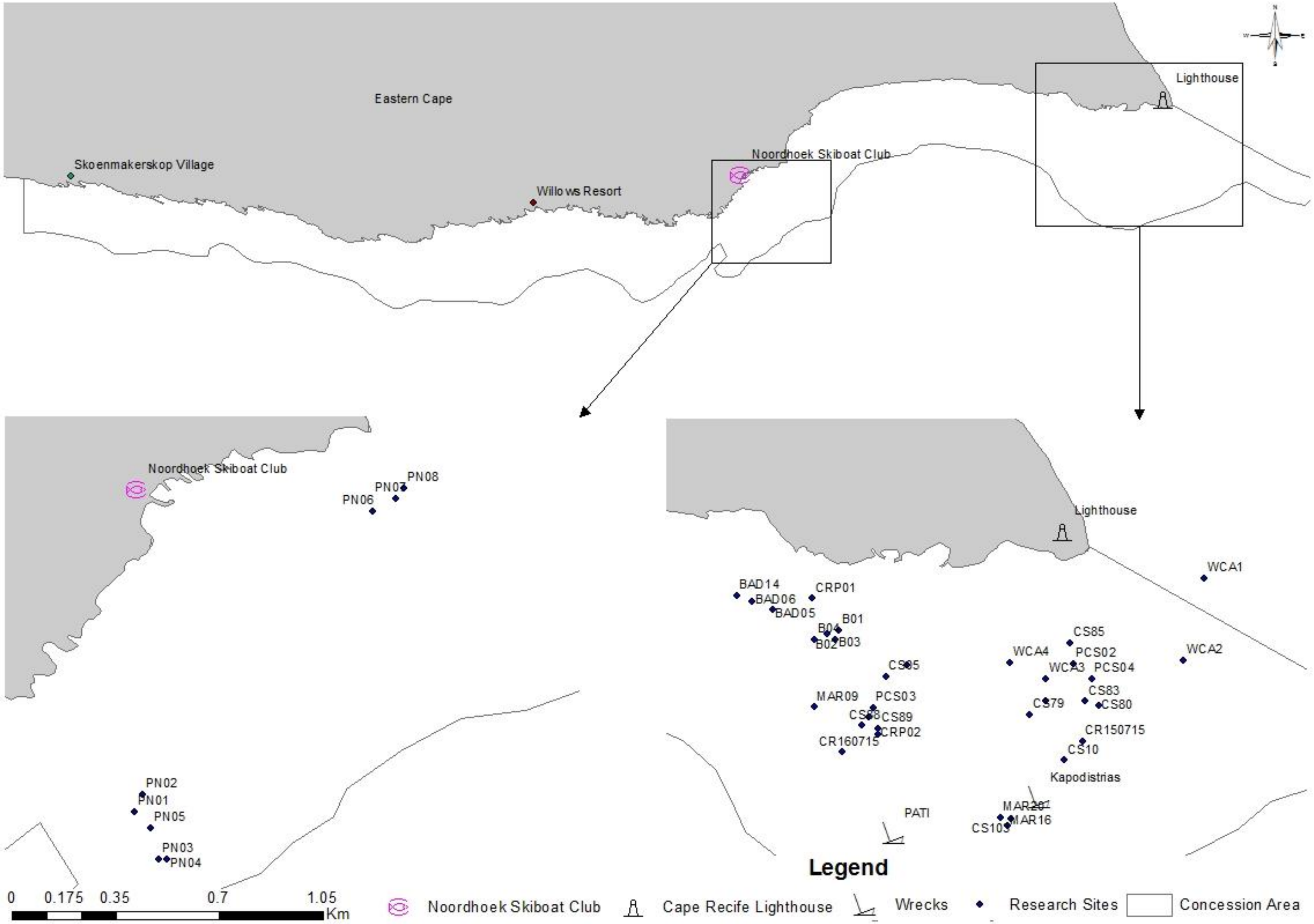

Figure 38: Site map showing Noordhoek and Cape Recife with research sites associated within each area.

5.2.2.2 Seaweed species identification and functional groups

Two methods of specimen collection were used in this study: systematic collection of specimens from quadrat scrapes, as well as haphazard collections in the general study area. A few seaweeds, such as *Plocamium corallorhiza*, *Laurencia flexuosa* and *Amphiroa ephedraea* were easily identifiable to species level and thus were identified to species level in all quadrats, however the remaining seaweeds, which could not be identified to species level or were too small and could not be detected, were grouped into functional groups. Selected seaweed specimens were then either pressed and identified in the herbarium, or were preserved 5% formaldehyde. The specimens were collected and identified in order to compile a species list for the Cape Recife area. A similar functional group approach was utilised for the invertebrates which unless easily identifiable, i.e. *Parechinus angulosus*, *Dinoplax gigas*, and *Haliotis midae*, were grouped together according to similarities in morphology and habit (for example colonial invertebrates).

5.2.3 Functional Group Diversity

Species diversity indices of Shannon-Wiener (H’) and 1/Simpson (λ) were calculated for photographic quadrat cover estimates of functional groups (Tilman et al., 1997). H’ is the most commonly used index (Beenaerts and Berghe, 2005; Gray, 2000; Parravicini et al., 2010). Due to the difficulty of identifying individual species from photographs, a functional groups approach was adopted in further sampling for the description of the benthic community .

5.2.3.1 Analysis

Percentage cover visual estimates were made from quadrat photographs from each site. Average cover than was calculated for each functional groups on a per site basis. The r-package: *stats*, version 3.3.1 (R Core Team, 2016) and R-studio, version 0.99.903 (RStudio Team, 2015) was used for analysis. Functional group diversity indices (Figure 39) were calculated using the packages “reshape2” (Wickham, 2007) and “Vegan” (Oksanen et al., 2015) and plotted using the barplot() function. The lm() function from the package “Vegan” (Oksanen et al., 2015) was used in the regression analysis of cover to biomass. While ordinations and classifications (Figure 48 to Figure 54) were completed using the package “Vegan” (Oksanen et al., 2015), package “rrcov” (Todorov and Filzmoser, 2009) and package “MASS” (Venables and Ripley, 2002).

5.3 Results

5.3.1 Biodiversity and richness of shallow subtidal reef systems of Cape Recife

H' ranged between 0.64 for the sites with the greatest sand influence, and 2.34 (areas with no sand influence / hard substrate dominated areas), whereas λ ranged between 1.46 for the sites with the greatest sand influence and 9.14 for rocky reef sites. The mean diversity for the entire area was 1.79 ± 0.06 SE for H' and 5.15 ± 0.27 SE for λ. A Welch Two Sample t-test showed no significance between Cape Recife and Noordhoek for both H' ($t = 0.88$, $df = 81.89$, $p = 0.38$) and λ ($t = 0.66$, $df = 80.14$, $p = 0.51$) (Figure 39). Both indices showed positive correlation ($r = 0.47$, and $r = 0.45$ respectively; $p < 0.005$) to rugosity.

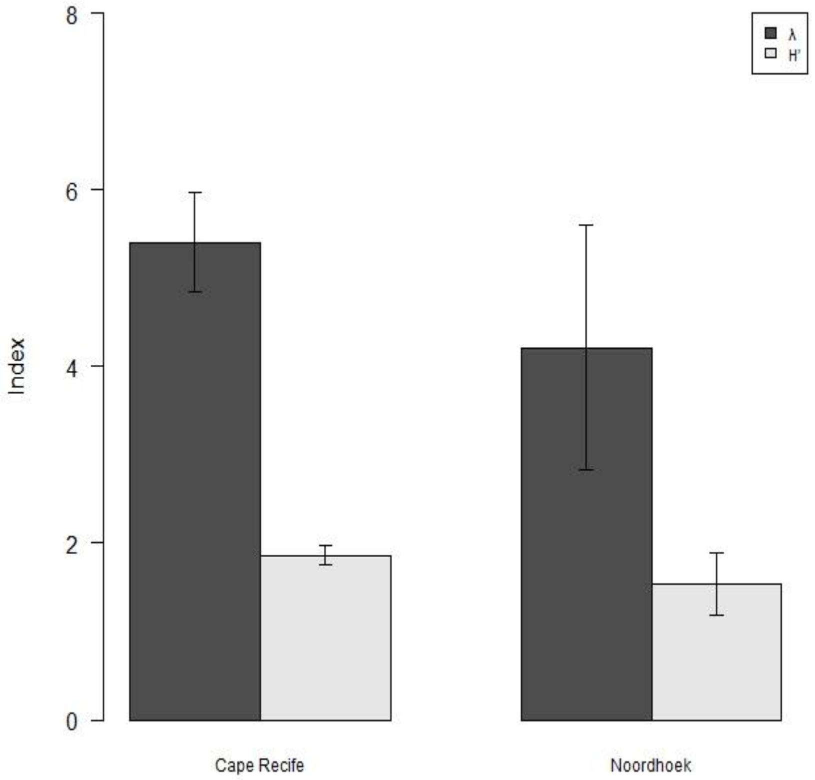

Figure 39: Mean (±SE) diversity indices for functional groups for Cape Recife (n= 37) and Noordhoek areas (n = 8).

Table 8 shows the macroalgal species collected in the study area, as well as details on the sites and depths at which these were collected. A total of 39 macrophyte species were recorded in the study area. Seventy-four percent of these species were from the phylum Rhodophyta belonging to ten families. A further 13% were from the Chlorophyta, with four families represented, and 13% were from three families in the Ochrophyta. Most of the species were collected at depths between 3 metres and 10 metres but some, namely *Zonaria* *subarticulata* (J.V.Lamouroux) Papenfuss, *Plocamium suhrii* Kützing and *Callophycus africanus* (F.Schmitz) F.E.Hewitt were collected at depths of more than 20 metres but less than 25 metres. *Plocamium corallorhiza* (Turner) J.D.Hooker & Harvey and *Laurencia flexuosa* Kützing were the two most commonly observed species in the 0 to 10 metre depth range. *P. corallorhiza* was clearly dominant, and showed a significant difference (t = 2.30, df = 64.93, p = 0.03, n = 45) for percentage cover in the 0 to 10 metre depth range. An average of 22.3% ± 2.9% SE cover across the 45 sampled sites was found for *P. corallorhiza,* while the coralline turf group (second most abundant group), consisting of mainly *Amphiroa, Arthrocardia, Jania* and *Corallina*, made up on average 15.5% ± 1.4% SE of the cover across sites (Figure 40). The most abundant animal group was the colonial invertebrates which included the sponges, bryzoans, some colonial cnidarians and sea fans (Plate 4) at 14.6% ± 1.8 SE across the study site.

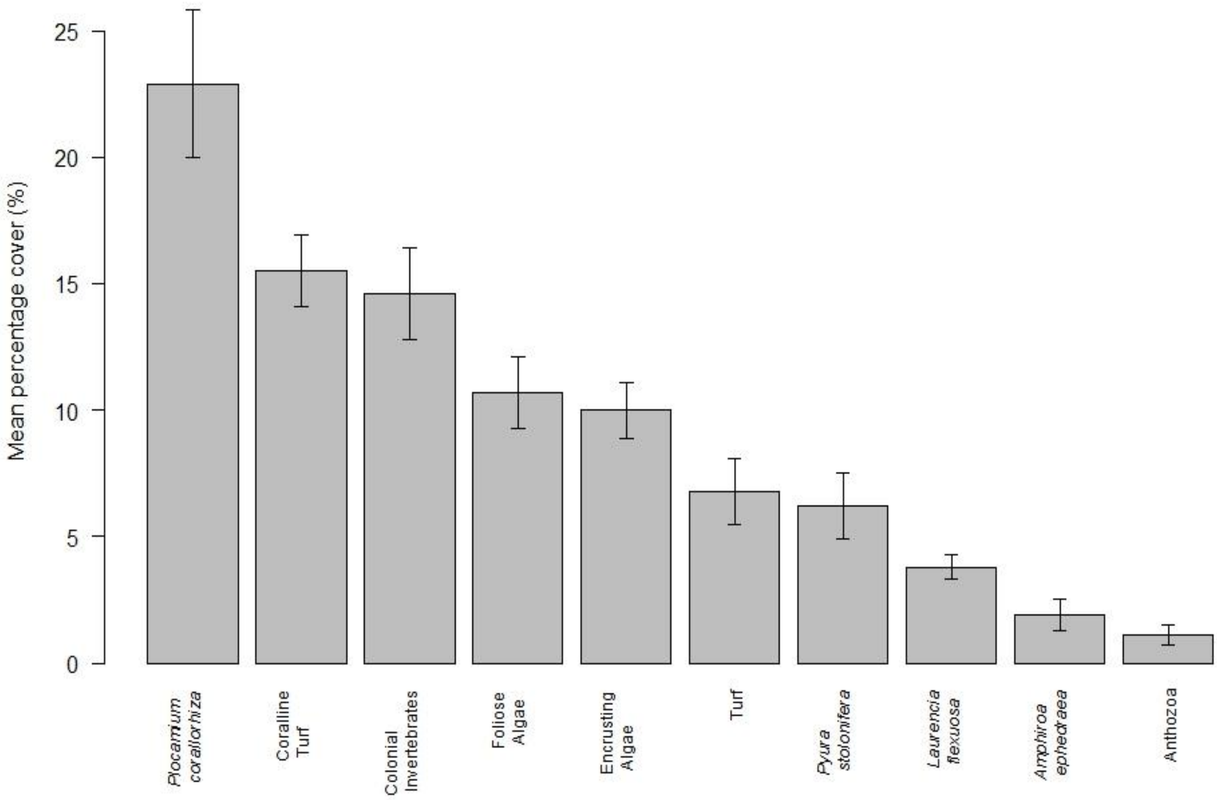

Figure 40: Mean percentage cover (+SE, n=45) of the top 10 most abundant functional groups.

It was also observed that among the samples *Plocamium corallorhiza* was the dominant seaweed with the average biomass of 332.60 g.w.w (±76.83 SE, n=34). A shift in dominance was observed between *P corallorhiza* (Plate 5) and the colonial invertebrates from 8 metres

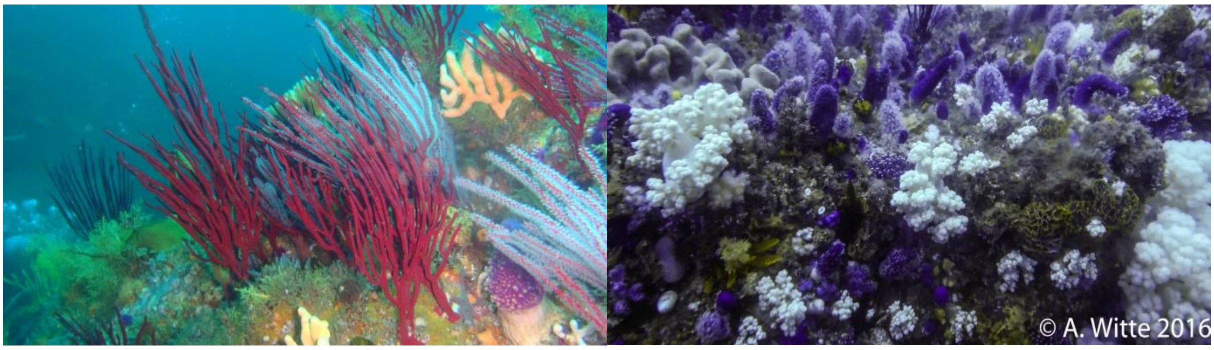

Plate 4: Examples of invertebrate dominated reef, (Right) Evans Peak Reef (25 metres deep), (Left) Gunners Reef 40 metres)

deep and deeper (max = 22 metres) (Plate 4) (Figure 41).

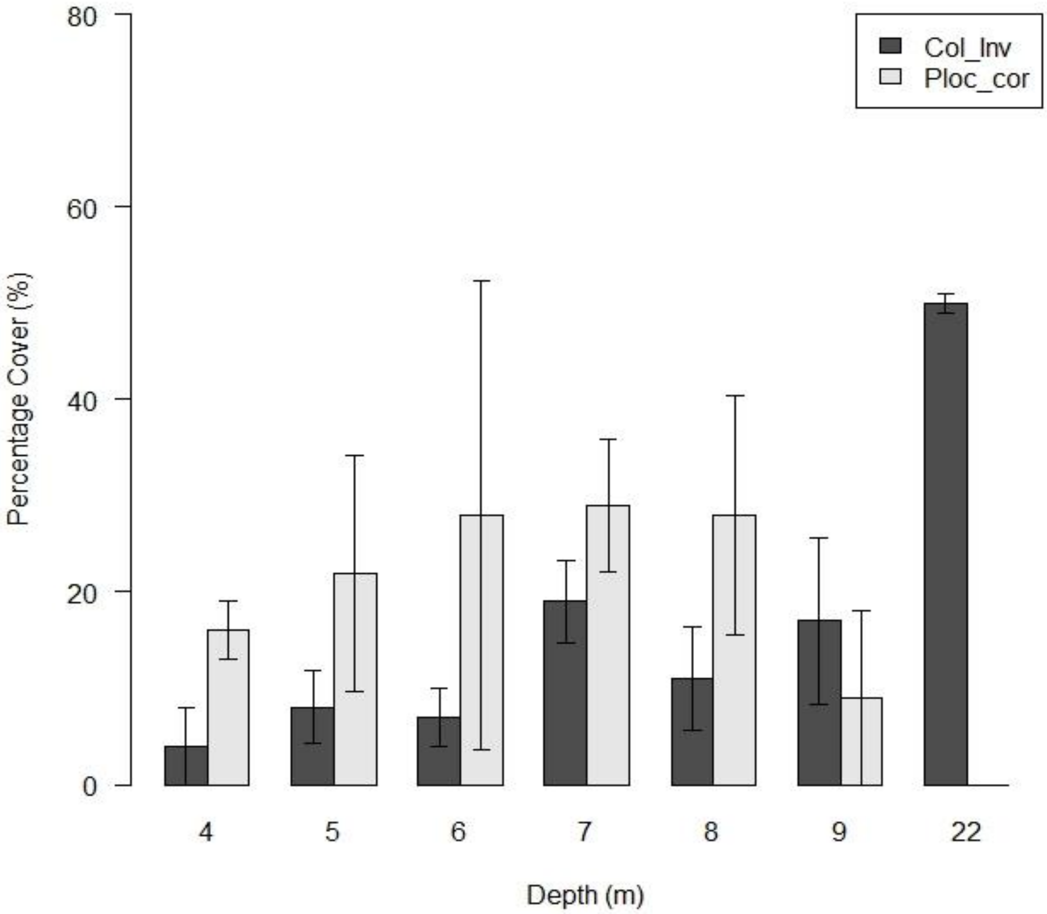

Figure 41: Change in dominance according to mean percentage cover (+SE, n=45) for Colonial invertebrates and *Plocamium corallorhiza*.

Plate 5: *Plocamium corallorhiza,* the dominant seaweed up to 10 metres deep. It can form extensive beds (best described as meadow –like).

5.3.2 The relationship between abalone and functional groups.

Analysis of the mean *Haliotis midae* counts per transects as compared to mean functional group abundance showed significant correlation with 5 of the 20 functional groups (Figure 42 and Figure 43). This included the seaweed functional groups of foliose seaweeds ($t = 3.42$, $df = 43$, $p = 0.001$, $cor = 0.46$), the coralline turf ($t = 2.13$, $df = 43$, $p = 0.04$, $cor = 0.31$), Encrusting algae ($t = 2.01$, $df = 43$, $p = 0.05$, $cor = 0.29$) (Table 10), and the invertebrate functional groups of *Dinoplax gigas* ($t = 4.45$, $df = 43$, $p < 0.001$, $cor = 0.56$) and *Parenchinus angulosus* ($t = 2.09$, $df = 43$, $p = 0.043$, $cor = 0.3$) (Table 11)

Analysis of the quadrat data found that *Haliotis midae* showed correlation to 11 of the 20 functional groups (Table 9). The seaweed groups included *Plocamium corallorhiza*, *Halimeda cuneata*, foliose algae, fleshy turfs, articulated upright corallines and the coralline turf, while invertebrates included *Dinoplax gigas*, *Parenchinus angulosus, Gynandrocarpa placenta*, prosobranchia group, anthozoa group, and the asteridea group. *Plocamium corallorhiza*, foliose algae and coralline turf were the only groups that showed a positive correlation with abalone, however the coralline turf groups correlation was extremely weak. The other groups showed negative correlations (Table 9).

Table 8: Macroalgal species collected in the study area

Genus	Species	Sub/var	Authority	Phylum	Family	Area	Depth Range
Amphiroa	*ephedraea*		(Lamarck) Decaisne	Rhodophyta	Corallinaceae	Cape Recife	1-8
Amphiroa	*beauvoisii*		J.V.Lamouroux	Rhodophyta	Corallinaceae	Cape Recife	1-8
Amphiroa	*anceps*		(Lamarck) Decaisne	Rhodophyta	Corallinaceae	Cape Recife	1-8
Arthrocardia	*flabellata*		(Kützing) Manza	Rhodophyta	Corallinaceae	Cape Recife	1-8
Arthrocardia	*carinata*		(Kützing) Johansen	Rhodophyta	Corallinaceae	Cape Recife	1-8
Arthrocardia	*duthieae*		H.W.Johansen	Rhodophyta	Corallinaceae	Cape Recife	1-8
Corallina	*officianalis*		L.	Rhodophyta	Corallinaceae	Cape Recife	1-8
Jania	*prolifera*		(J.V.Lamouroux) J.H.Kim, Guiry & H.-G.Choi	Rhodophyta	Corallinaceae	Cape Recife	1-8
Jania	*crassa*		J.V.Lamouroux	Rhodophyta	Corallinaceae	Cape Recife	1-8
Jania	*cultrata*		(Harvey) J.H.Kim, Guiry & H.-G.Choi	Rhodophyta	Corallinaceae	Cape Recife	1-8
Jania	*sagittata*		(J.V.Lamouroux) Blainville	Rhodophyta	Corallinaceae	Cape Recife	1-8
Laurencia	*glomerata*		(Kützing) Kützing	Rhodophyta	Rhodomelaceae	Cape Recife	1-8
Laurencia	*flexuosa*		Kützing	Rhodophyta	Rhodomelaceae	Cape Recife	1-8
Laurencia	*natalensis*		Kylin	Rhodophyta	Rhodomelaceae	Cape Recife	1-8
Hypnea	*rosea*		Papenfuss	Rhodophyta	Cystocloniaceae	Cape Recife	1-8
Hypnea	*tenuis*		Kylin	Rhodophyta	Cystocloniaceae	Cape Recife	1-8
Calliblepharis	*fimbriata*		(Greville) Kützing	Rhodophyta	Cystocloniaceae	Cape Recife	1-8
Gigartina	*insignis*		(Endlicher & Diesing) F.Schmitz	Rhodophyta	Gigartinaceae	Cape Recife	1-8
Callophycus	*africanus*		(F.Schmitz) F.E.Hewitt	Rhodophyta	Rhodophyta incertae sedis	Mangolds Pool and Cape Recife Reef	6-22

Plocamium	*suhrii*		Kützing	Rhodophyta	Plocamiaceae	Mangolds Pool	6-22
Plocamium	*corallorhiza*		(Turner) J.D.Hooker & Harvey	Rhodophyta	Plocamiaceae	Cape Recife	1-8
Plocamium	*beckeri*		F.Schmitz ex Simons	Rhodophyta	Plocamiaceae	Cape Recife	1-8
Gelidium	*pteridifolium*		R.E.Norris, Hommersand & Fredericq	Rhodophyta	Gelidiaceae	Cape Recife	1-8
Portieria	*hornemannii*		(Lyngbye) P.C.Silva	Rhodophyta	Rhizophyllidaceae	Cape Recife	1-8
Delisea	*flaccida*		(Suhr) Papenfuss	Rhodophyta	Bonnemaisoniaceae	Cape Recife	1-8
Zonaria	*subarticulata*		(J.V.Lamouroux) Papenfuss	Ochrophyta	Zonarieae	Mangolds Pool, and Cape Recife Reef	1-8
Dictyota	*dichotoma*		(Hudson) J.V.Lamouroux	Ochrophyta	Dictyotaceae	Cape Recife	1-8
Stypopodium	*zonale*		(J.V.Lamouroux) Papenfuss	Ochrophyta	Dictyotaceae	Cape Recife	1-8
Codium	*lucasii*	*capensis*	P.C.Silva	Chlorophyta	Codiaceae	Cape Recife	1-8
Caulerpa	*filiformis*		(Suhr) Hering	Chlorophyta	Caulerpaceae	Cape Recife	1-8
Caulerpa	*brachypus*		Harvey	Chlorophyta	Caulerpaceae	Cape Recife	1-8
Mazzaella	*capensis*		(J.Agardh) Fredericq	Rhodophyta	Gigartinaceae	Cape Recife	1-8
Ptilophora	*hildebrandtii*		(Hauck) R.E.Norris	Rhodophyta	Gelidiaceae	Noordhoek	10-12
Dictyopteris	*serrata*		(Areschoug) Hoyt	Ochrophyta	Zonarieae	Cape Recife: Kapo	10-12
Cladophora	sp.		Kützing	Chlorophyta	Cladophoraceae	Cape Recife	1-8
Sargassum	*elegans*		Suhr	Ochrophyta	Sargassaceae	Noordhoek	1-8
Dichotomaria	*diesingiana*		(Zanardini) Huisman, J.T.Harper & G.W.Saunders	Rhodophyta	Galaxauraceae	Cape Recife	1-8
Gelidium	*abborttiorum*		R.E.Norris	Rhodophyta	Gelidiaceae	Cape Recife	1-8
Halimeda	*cuneata*		Hering	Chlorophyta	Halimedaceae	Cape Recife	1-8

Table 9: Correlation coefficient and significance between *Haliotis midae* and the functional groups for quadrats.

	Correlation	Significance	df	t
Ploc_cor	0.5	<0.001	705	15.36
Dino_gig	-0.206	0.02	136	-2.460
Hali_cun	-0.219	0.006	152	-2.771
Fol_alg	0.209	0.017	126	1.109
Fle_turf	-0.168	<0.001	534	-3.942
Up_Cor	-0.190	0.001	280	-3.246
Cor_Tur	0.070	0.03	945	2.168
Ant_Grp	-0.231	<0.001	294	-4.062
Ast_Grp	-0.251	0.002	150	-3.179
Par_ang	-0.139	0.006	389	-2.768
Pros_Grp	-0.186	0.025	143	-2.264
Gyn_plac	-0.231	0.007	135	-2.754
Encr_Alg	0.008	0.825	863	0.221
Col_Inv	0.043	0.193	905	1.302
Pyu_sto	-0.042	0.387	423	-0.866
Ann_Grp	-0.139	0.121	124	-1.562
Turbo_sar	-0.100	0.271	122	-1.105
Opis_Grp	-0.090	0.323	121	-0.993
Crus_Grp	-0.093	0.307	120	-1.026
Laur_flex	-0.058	0.143	640	-1.465

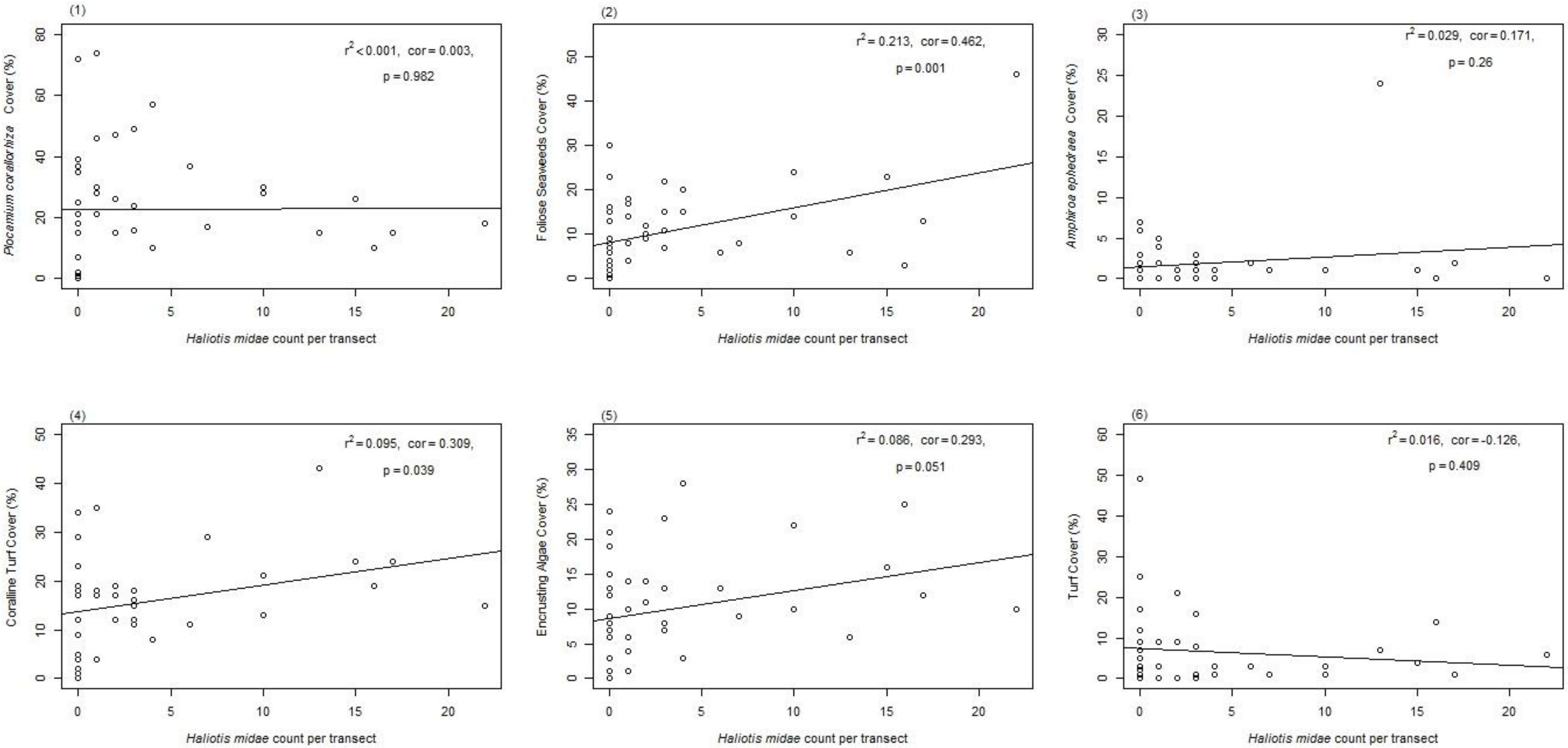

Figure 42: Relationships between macroalgal species and functional group cover to abalone counts per transect (n=45); (1) The relationship between *Plocamium corallorhiza* cover and the abalone count per transect, (2)The relationship between foliose seaweeds cover and the abalone count per transect, (3) The relationship between *Amphiroa ephedraea* cover and the abalone count per transect, (4) The relationship between coralline turf cover and the abalone counts per transect, (5) The relationship between encrusting algae cover and the abalone counts per transect, and (6) The relationship between turf cover and the abalone counts per transect.

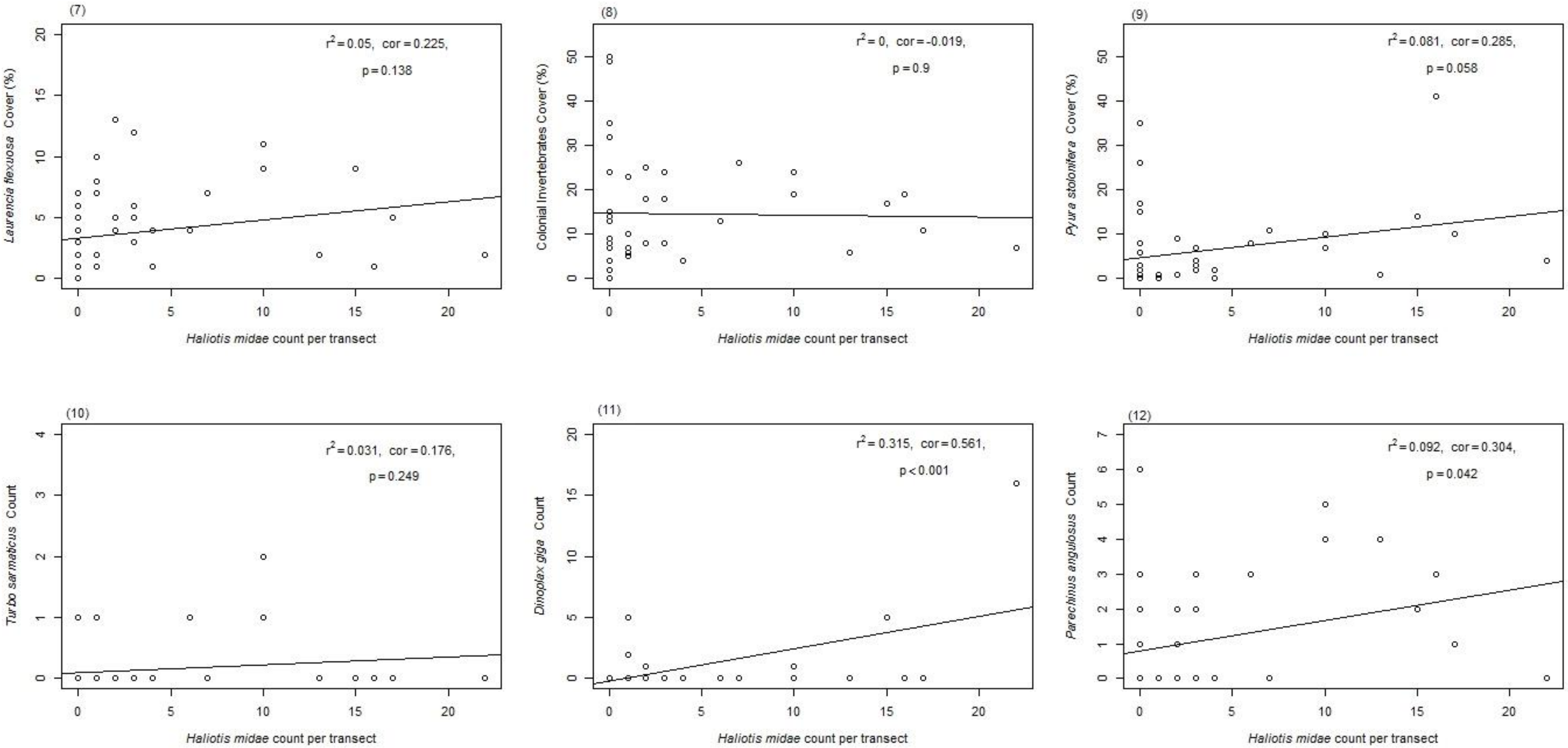

Figure 43: Relationships between functional group cover and abalone counts per transect (n=45); (7) The relationship between *Laurencia flexuosa* cover and the abalone count per transect, (8)The relationship between colonial invertebrate cover and the abalone count per transect, (9) The relationship between *Pyura stolonifera* and the abalone count per transect, (10) The relationship between *Turbo sarmaticus* counts and the abalone counts per transect, (11) The relationship between *Dinoplax gigas* counts and the abalone counts per transect, and (12) The relationship between *Parechinus angulosus* counts and the abalone counts per transect.

5.3.3 The role of depth, rugosity and vertical relief on functional group and abalone abundance.

Correlations between depth (in metres) and functional groups showed four significantly negative relationships, including three seaweed functional groups of *Plocamium corallorhiza*, foliose seaweeds and encrusting algae and *Haliotis midae* (Figure 44; Table 12). The second animal functional group, the colonial invertebrates, exhibited a significantly positive correlation to depth (cor = 0.67, t = 5.92, df = 43, $p < 0.001$) (Table 11).

Rugosity correlations were found between 6 of the functional groups. This included a strong positive relationship between *Haliotis midae* and rugosity, as well as the seaweed functional groups articulated upright corallines (*Amphiroa ephedraea), Laurencia flexuosa*, coralline turf and encrusting algae. *Parenchinus angulosus* was the only other invertebrate to show significant correlation to rugosity (cor = 0.38, t = 2.7, df = 43, $p = 0.009$) (Figure 45; Table 11).

The only functional group to show any significant relation to the maximum vertical relief (MVR) was *Pyura stolonifera* (cor = 0.35, t = 2.47, df = 43, $p = 0.017$) (Figure 46; Table 11). No relationships were found between depth, rugosity and MVR (Table 12).

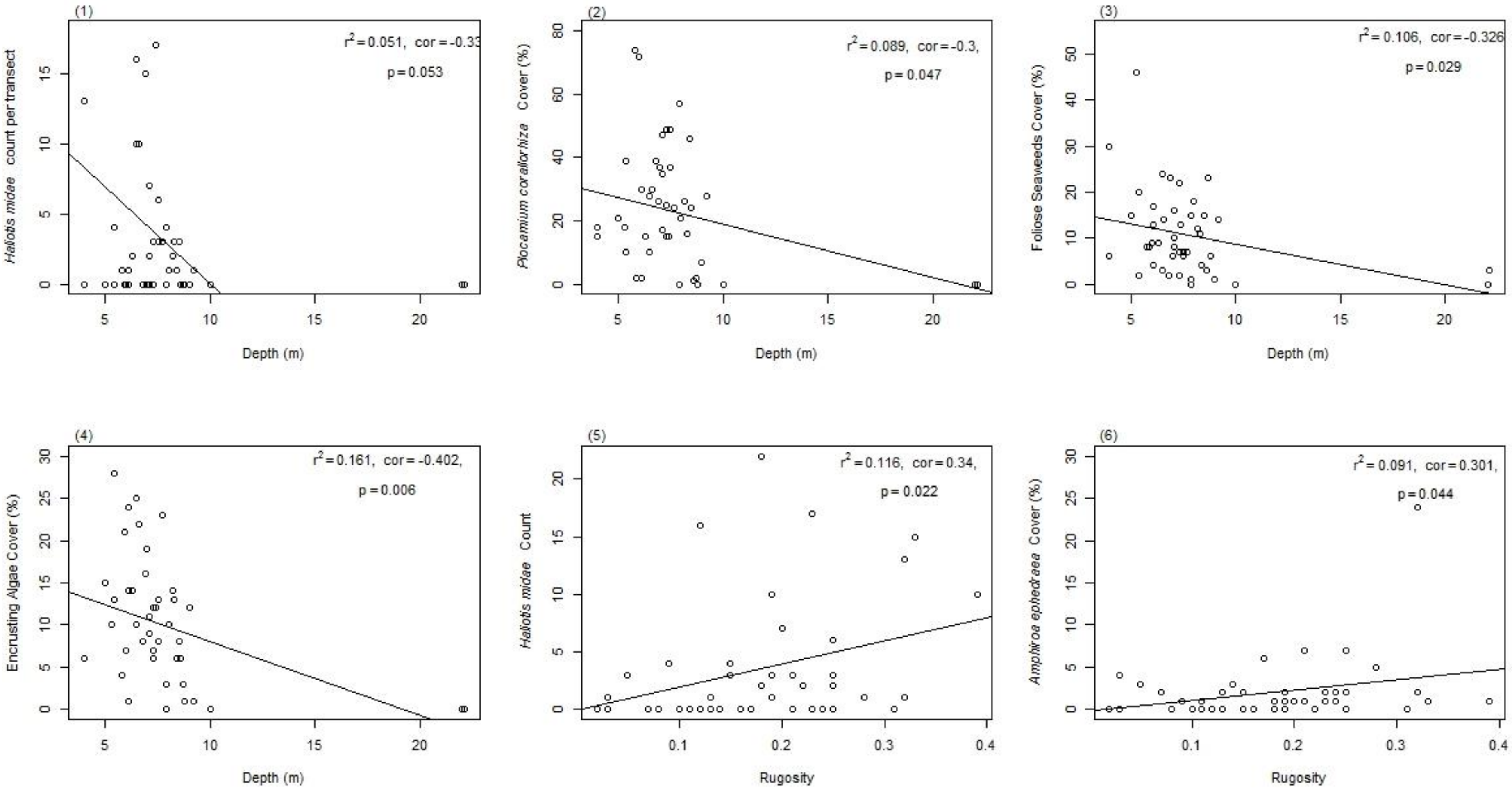

Figure 44: Relationship between functional groups, depth, and rugosity (n=45); (1) The relationship between *Haliotis midae* count per transect and depth (m), (2)The relationship between *Plocamium corallorhiza* cover and depth (m), (3) The relationship between foliose seaweeds cover and depth (m), (4) The relationship between encrusting algae cover and depth (m), (5) The relationship between *Haliotis midae* count per transect and rugosity, and (6) The relationship between *Amphiroa ephedraea* cover and rugosity.

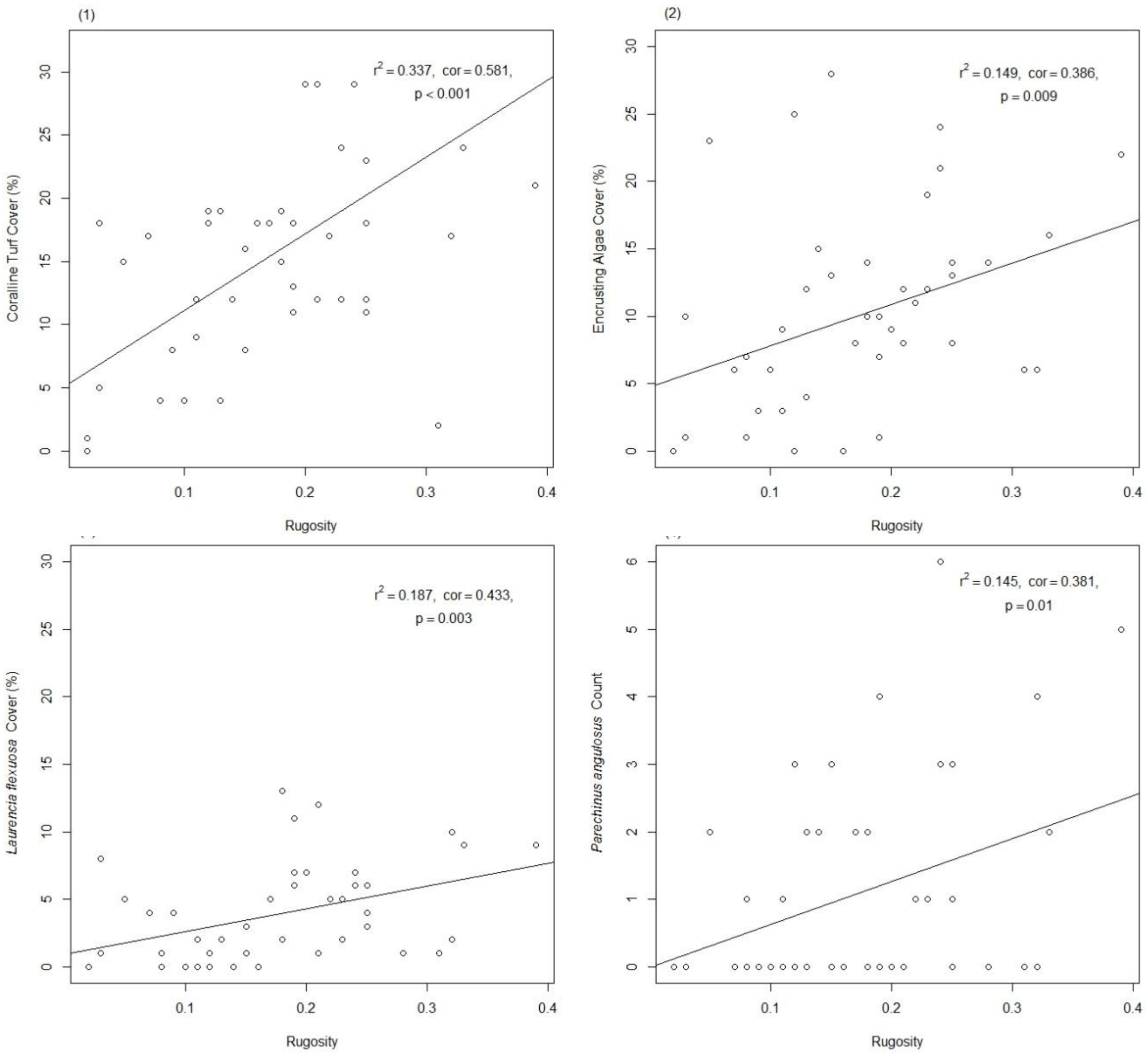

Figure 45: Relationship between functional groups, depth, and rugosity (n=45); (1) The relationship between coralline turf cover and rugosity, (2) The relationship between encrusting algae cover and rugosity, (3) The relationship between *Laurencia flexuosa* cover and rugosity, (4) The relationship between *Parechinus angulosus* counts and rugosity.

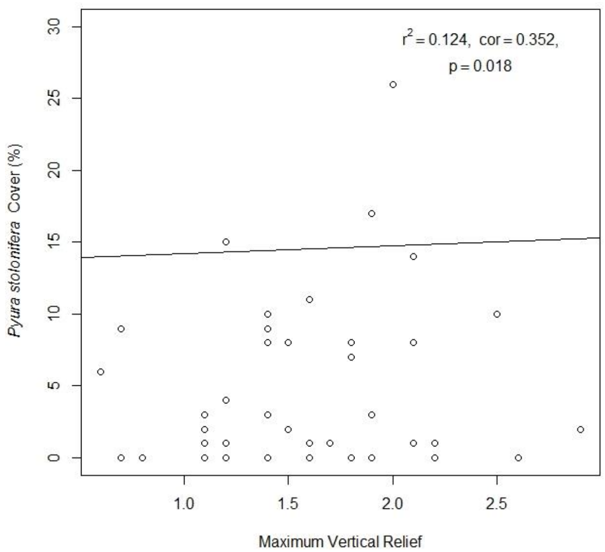

Figure 46: Relationship between the *Pyura stolonifera* functional group cover and Maximum Vertical Relief (MVR) (n=45).

Table 10: Matrix of significance for correlations between seaweed functional groups, *Haliotis midae*, depth, rugosity and maximum vertical relief

	Plocamium corallorhiza	**Foliose seaweeds**	***Amphiroa ephedraea***	**Coralline Turf**	**Encrusting Algae**	**Turf**	***Laurencia flexuosa***
Haliotis midae	0.982	0.001	0.261	0.039	0.051	0.410	0.138
Depth	0.047	0.029	0.136	0.766	0.006	0.333	0.309
Rugosity	0.134	0.091	0.044	0.00003	0.009	0.473	0.003
MRV	0.660	0.959	0.994	0.550	0.479	0.168	0.415

Table 11: Matrix of significance for correlations between invertebrate functional groups, *Haliotis midae*, depth, rugosity and maximum vertical relief

	Turbo sarmaticus	***Pyura stolonifera***	**Colonial Invertebrates**	***Dinoplax gigas***	***Parenchinus angulosus***
Haliotis midae	0.249	0.058	0.900	0.00006	0.042
Depth	0.660	0.144	0.000005	0.515	0.087
Rugosity	0.409	0.070	0.324	0.610	0.010
MRV	0.170	0.018	0.894	0.462	0.139

Table 12: Matrix of significance for correlations between *Haliotis midae*, depth, rugosity and maximum vertical relief

	Haliotis midae	**Depth**	**Rugosity**	**MRV**
Haliotis midae	1	0.053	0.022	0.327
Depth		1	0.158	0.105
Rugosity			1	0.984
MRV				1

5.3.4 Cover versus biomass relationships

Cover vs biomass relationships (Figure 47) were examined with correlation tests and linear regressions for recognised seaweed functional groups. The models showed strong relationships between cover estimates and biomass. *Laurencia flexuosa* showed the strongest correlation (cor = 0.96, df = 12, p<0.005) (Figure 47, 2.), while the articulated upright corallines showed the weakest correlation (cor = 0.87, df = 12, p<0.005) (Figure 47, 3). *Plocamium corallorhiza* showed a strong correlation for the cover biomass relationship (cor = 0.88, df = 12, p<0.005) (Figure 47, 1). The coralline turf group also showed a strong relationship (Figure 47, 4).

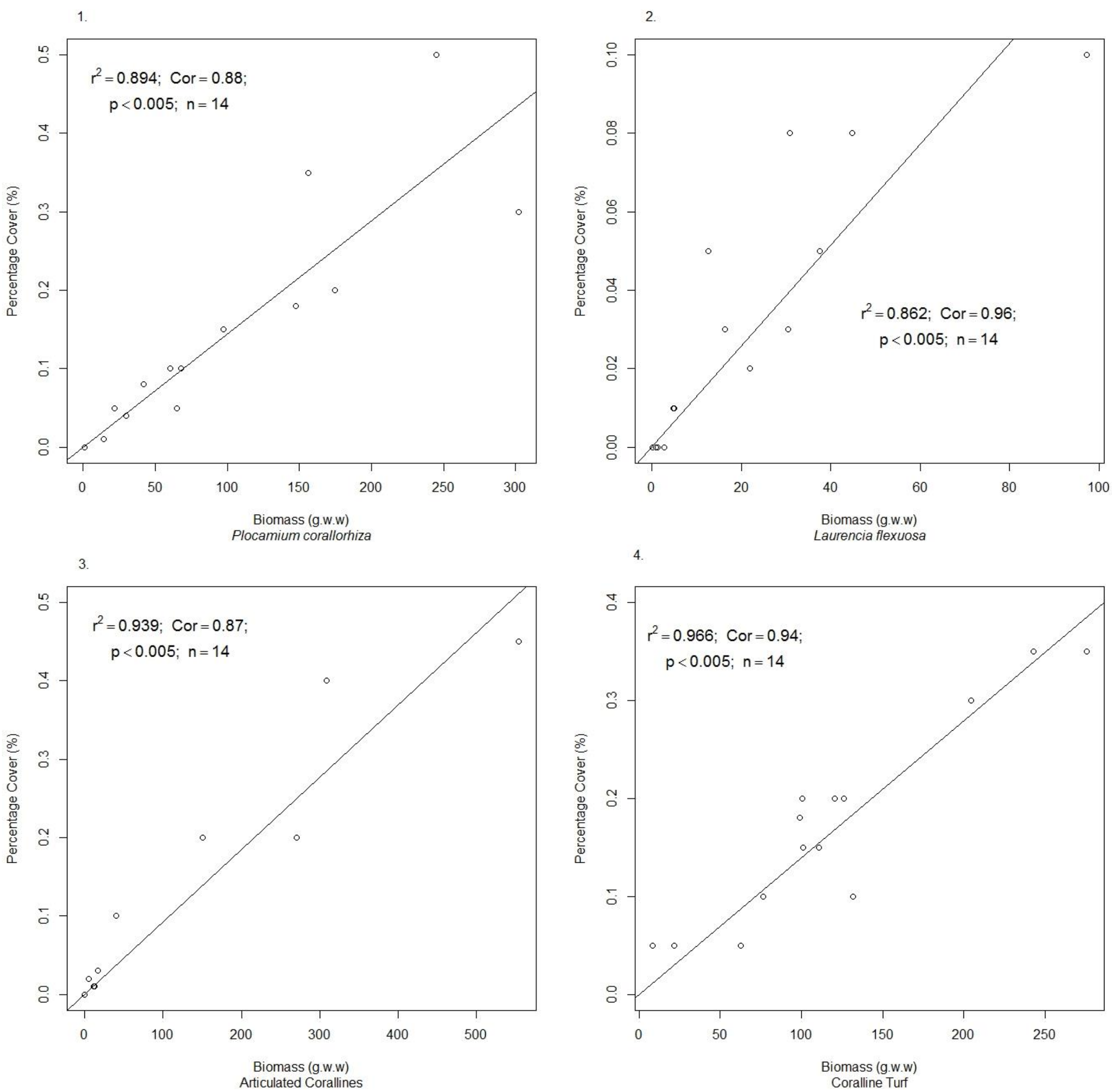

Figure 47: The correlation between biomass (g.w.w) and cover for four species: 1. *Plocamium corallorhiza*, 2. *Laurencia flexuosa*, 3. Articulated corallines, 4. Coralline Turf.

5.4 Habitat and Community structure and the influence of environmental gradients.

Ward's classification of the sites showed an estimation of 6 groups for the sites (Figure 48). Each site was influenced by a specific environmental gradient. The groups were identified as: group 1 was shallower reef (4-6 metres) that was predominantly reef with very little sand presence, while group 2 was deeper reef (between 7 and 10 metres). Group 3 was a much more profiled reef but with a fair amount of sand present, particularly in the gullies and channels between the pinnacles. Group 4 consisted of 2 sites that were dominantly sand covered with an absence of any reef profile. Group 5 was predominantly sand and low profile reef, and 6 consisted of two sites that were greater than 20 metres in depth. There was a degree of fidelity of sites to the environmental gradients, which were particularly clear in the grouping of the poor abalone habitat and the very deep habitats (groups 4, 5 and 6) (as these were the three groups to show less overlapping in the DCA compared to groups 1, 2, and 3) (Figure 50 (A)).

Clearer groupings were detected in the functional group classification and DCA (Figure 49 and Figure 50 B). The wards classification indicated that there are four groupings of functional groups. The first consisted of foliose algae, encrusting algae, coralline turf, colonial invertebrates and *Plocamium corallorhiza* and were present at all sites. Group 2 consisted of anthozoa, *Parenchinus angulosus*, articulated upright corallines, fleshy turf, *Pyura stolonifera*, *Laurencia flexuosa*, and *Haliotis midae* and were predominantly associated with reef biotopes. Group 3 consisted of Opisthobranchia, *Turbo sarmaticus*, annelids, crustaceans, *Halimeda cuneata*, and *Dinoplax gigas* and was predominantly associated with very sandy biotopes. Lastly group 4 consisted of *Gynandrocarpa placenta*, Prosobranchia and asteridea groups, and were predominantly associated with deeper sites (Figure 49). Groups 1 and 2 overlapped, whereas group 3 and 4 showed high fidelity of species and no overlap with other groups (Figure 50 (B)).

As can be seen overall four environmental gradients played a significant role in the distribution of sites and functional groups. This included the influence of the substrates including continuous rock, boulders and sand, as well as the influence of depth (Table 13).

Table 13: Environmental variable vectors and their significance in the ordination space for all sites and functional groups (Codes in Table 18).

	DCA 1	DCA 2	r^2	Pr(>r)	Significance
Reef	-0.41304	-0.91071	0.2072	0.004	**
Boul	0.93542	0.35353	0.1309	0.048	*
Peb	-0.07459	0.99721	0.0261	0.525	
Grav	-0.06237	-0.99805	0.0323	0.44	
Sand	0.25755	0.96626	0.2373	0.004	**
Dep	-0.02473	-0.99969	0.4647	0.001	***
Rug	-0.17711	0.98419	0.0063	0.868	
MVR	0.99747	0.07105	0.0614	0.229	

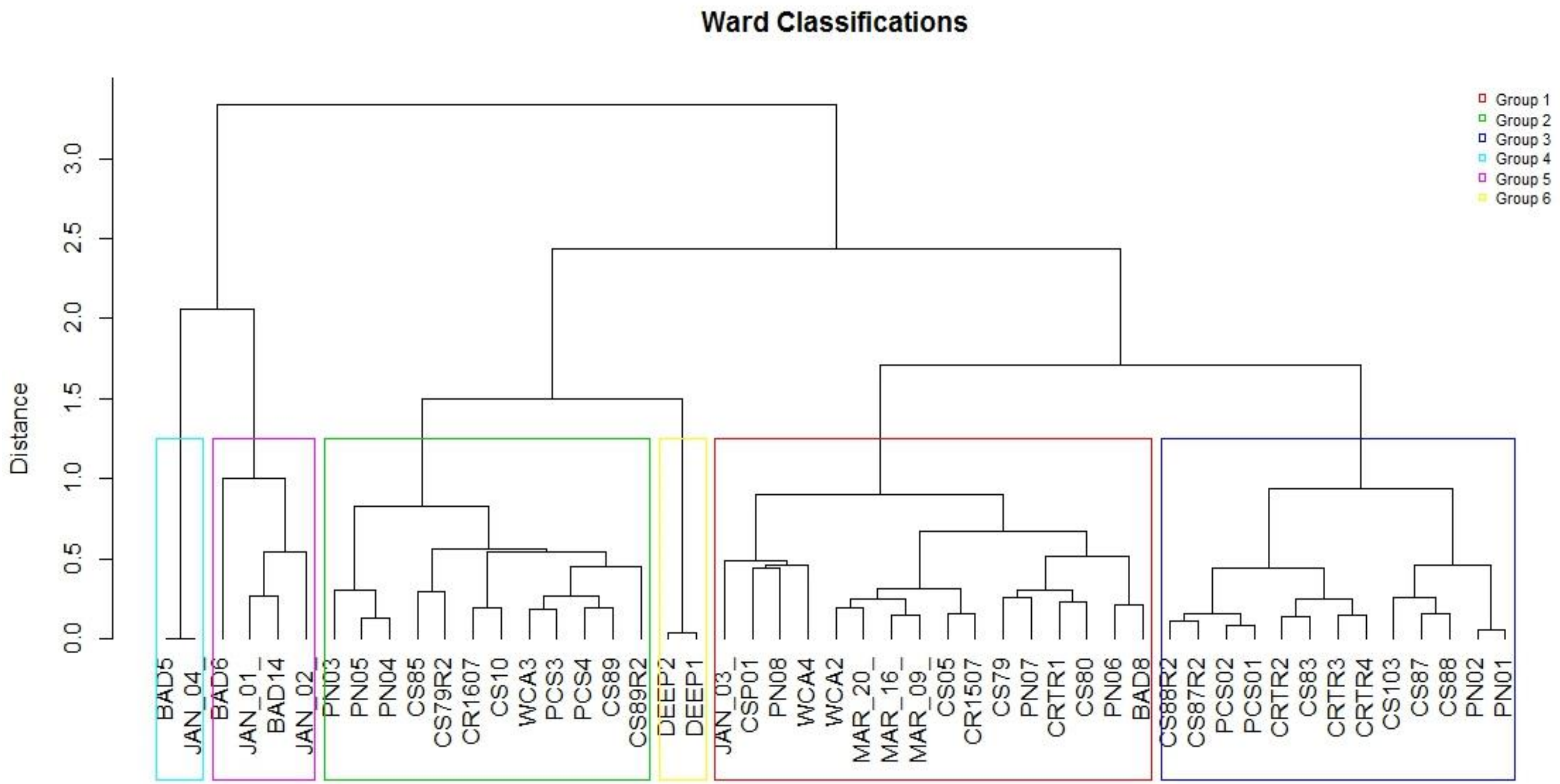

Figure 48: Ward Classification of mean site dissimilarity (index = Bray Curtis) scores for of environmental factors of different sites, showing five groups. (Codes for sites are given in Table 17).

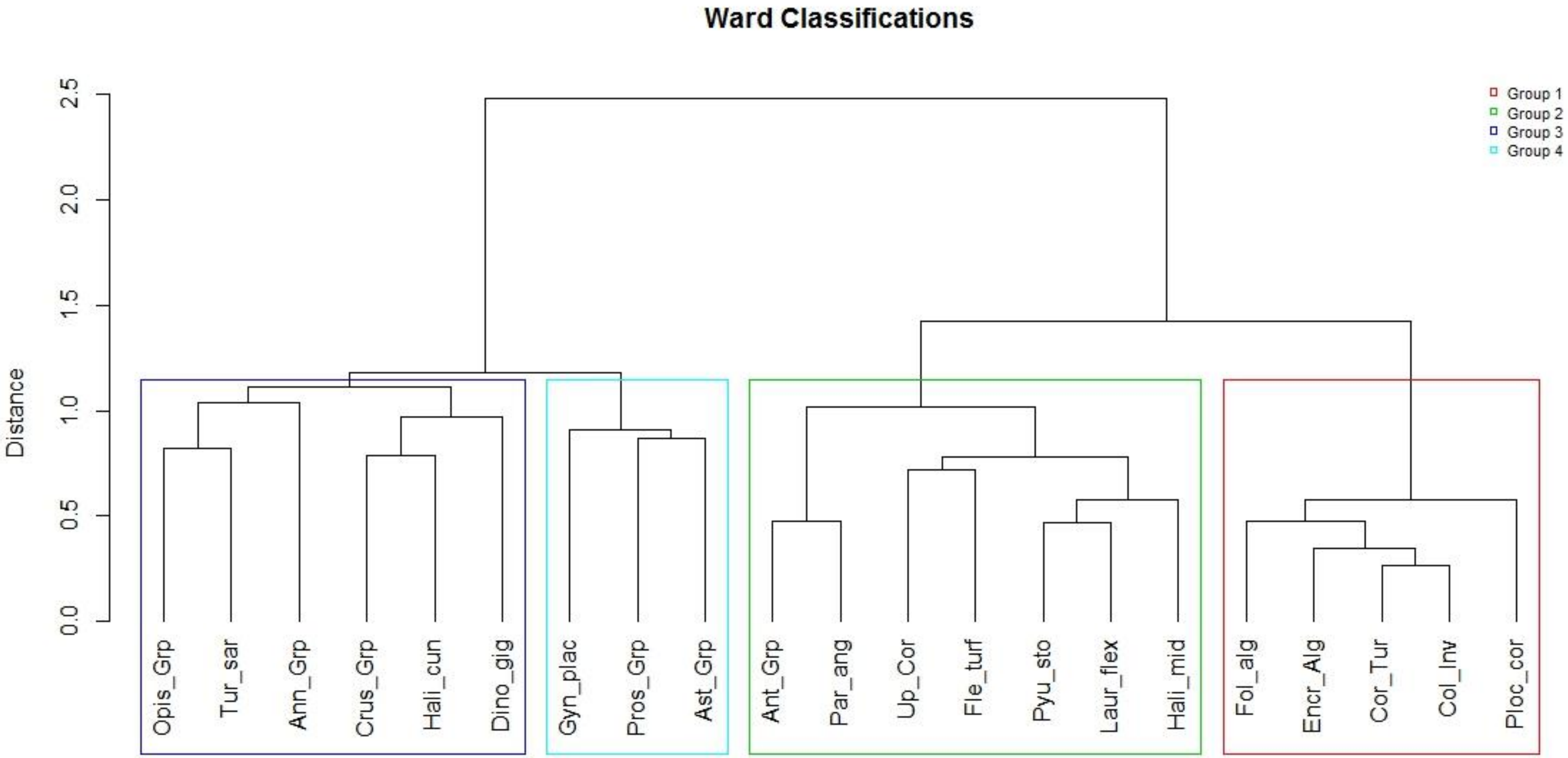

Figure 49: Ward Classification of mean functional group scores (index = Bray Curtis) for of environmental factors of different sites, showing four groups. (Codes for functional groups are given in Table 16).

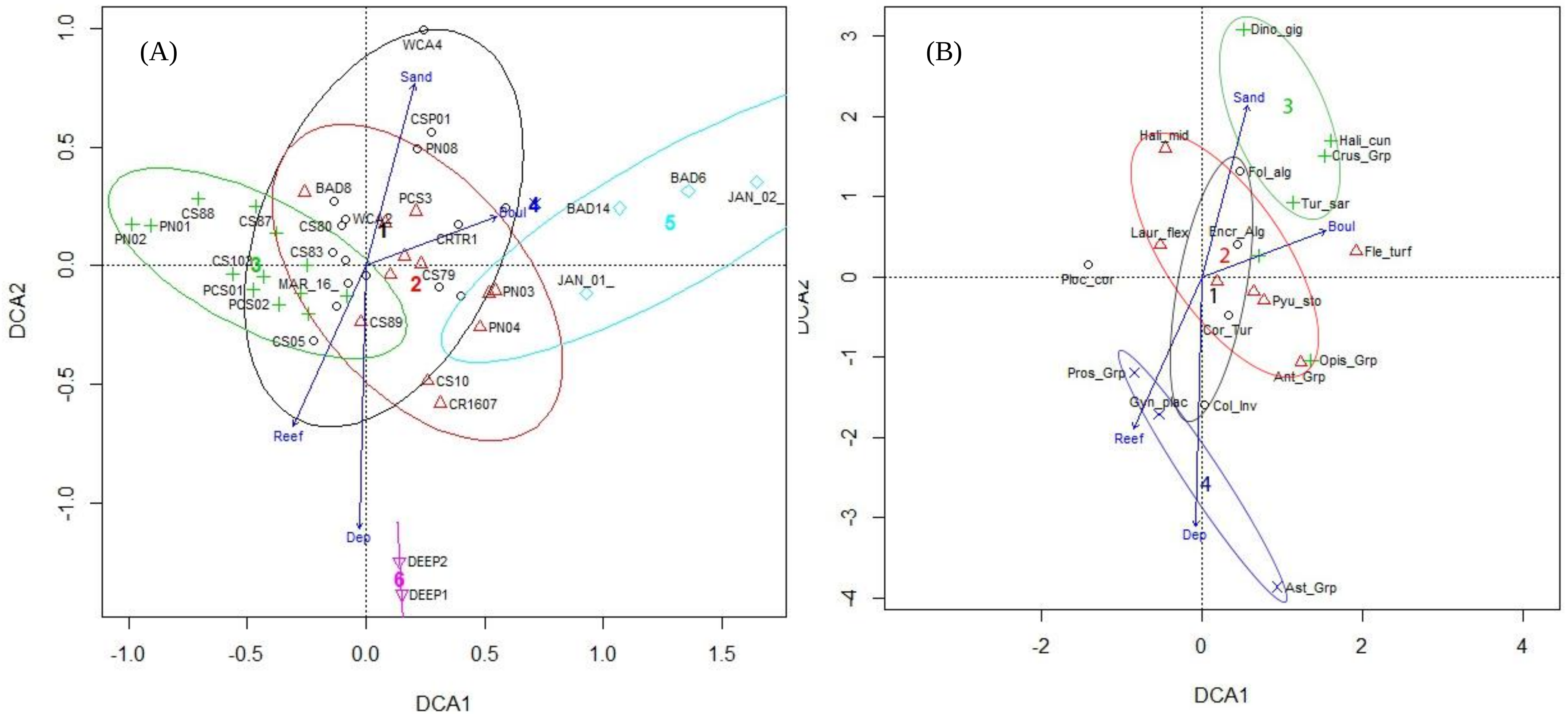

Figure 50: Detrended Correspondence Analysis (DCA); (A) similarity between sites (transects) based on functional group abundance, (B) similarity between functional group abundance based on site association (index = Bray Curtis) (Eigen values for DCA1 and DCA2 axes are 0.221 and 0.147 respectively). Effect of significant environmental gradients (p<0.05) are shown as arrows. (Codes are given for functional groups in Table 16 and for sites in Table 17).

Sites separated very clearly in terms of functional group abundances for seaweeds only. As can be seen in Figure 51 the groupings are similar to that of Figure 49, particularly the presence of the very deep sites and the sites that were almost exclusively sand dominated. As can be seen in Figure 52 (A), group 3, consisting of site BAD5 and JAN_04_, both had 100% sand respectively, and showed clear separation from the remaining groups. When focusing on the other four groups (Figure 52 (B)), it is clear that there is a degree of site fidelity with poor abalone habitat sites falling to the right (x=2), which is predominantly driven by the sand and boulder gradients, while the more suitable sites fall to the left (x = -1) which is predominantly driven by reef.

This is confirmed further in Table 14 were the gradients of reef, boulder and sand showed a significant influence on the seaweed functional groups distribution in ordination space.

Table 14:Environmental variable vectors and their significance in the ordination space for seaweed functional groups only

	DCA1	DCA2	r2	Pr(>r)	Significance
Reef	-0.30179	0.95338	0.3254	0.001	***
Boul	0.86179	-0.50726	0.1649	0.017	*
Peb	0.03533	-0.99938	0.0068	0.855	
Grav	0.18123	0.98344	0.014	0.726	
Sand	0.16527	-0.98625	0.3912	0.001	***
Dep	0.09029	0.99592	0.0575	0.243	
Rug	-0.08359	0.9965	0.0276	0.55	
MVR	0.64916	0.76065	0.0875	0.118	

Analysis of seaweed only functional groups indicated clearly three groups (Figure 53). Group 1 consisted of foliose algae, encrusting algae, coralline turf, and *Plocamium corallorhiza,* while group 2 was the articulated upright corallines, fleshy turf and *Laurencia flexuosa*, and group 3 has *Halimeda cuneata* (similar to the groupings in Figure 49). As can be seen in Figure 53, groups 1 and 2 overlap but *Halimeda cuneata* clearly separates from the others due to its high association to very sandy environments.

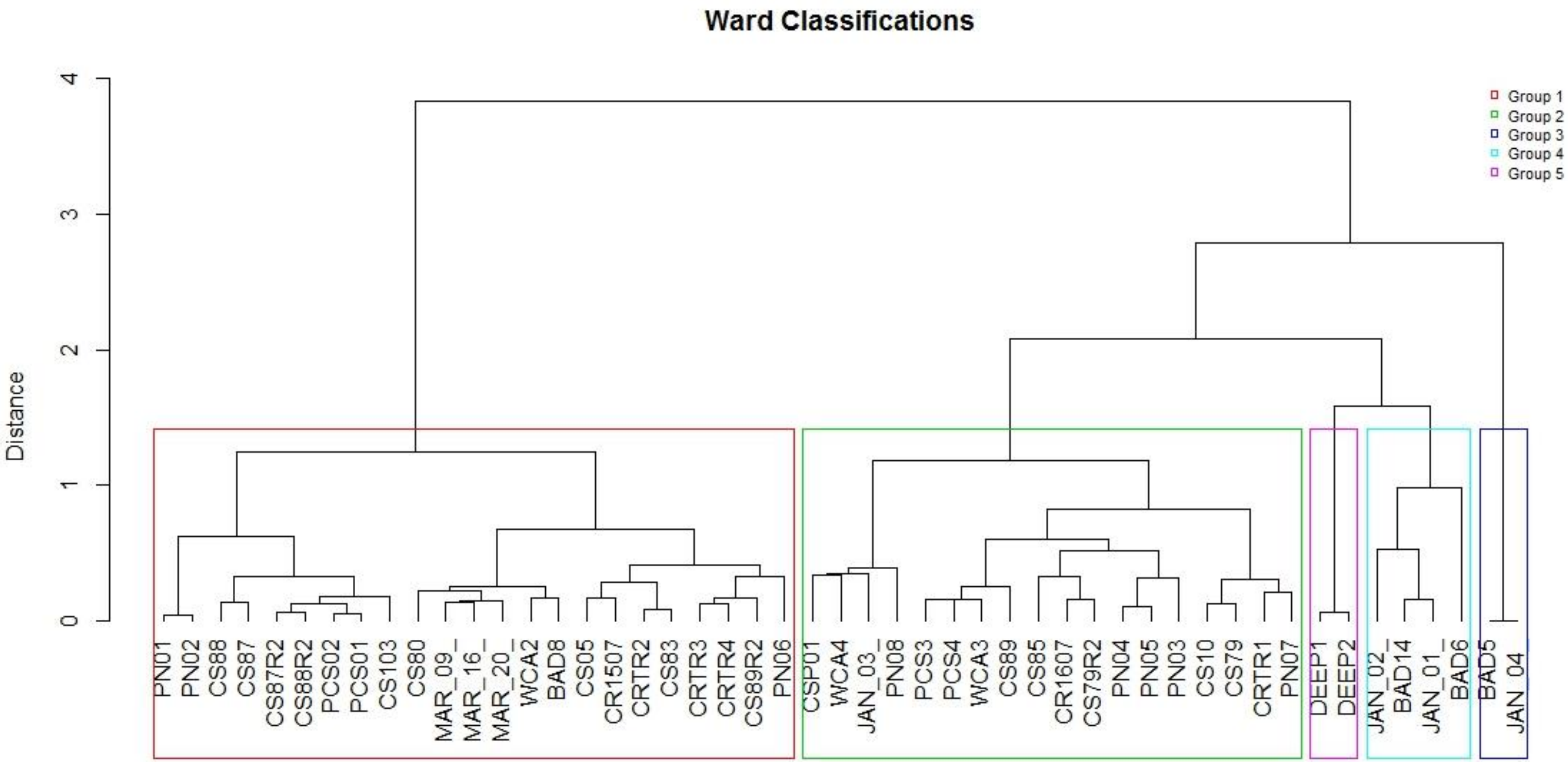

Figure 51: Ward Classification of mean site dissimilarity (index = Bray Curtis) of seaweed functional group scores for of environmental factors of different sites, showing five groups. (Codes for sites are given in Table 17).

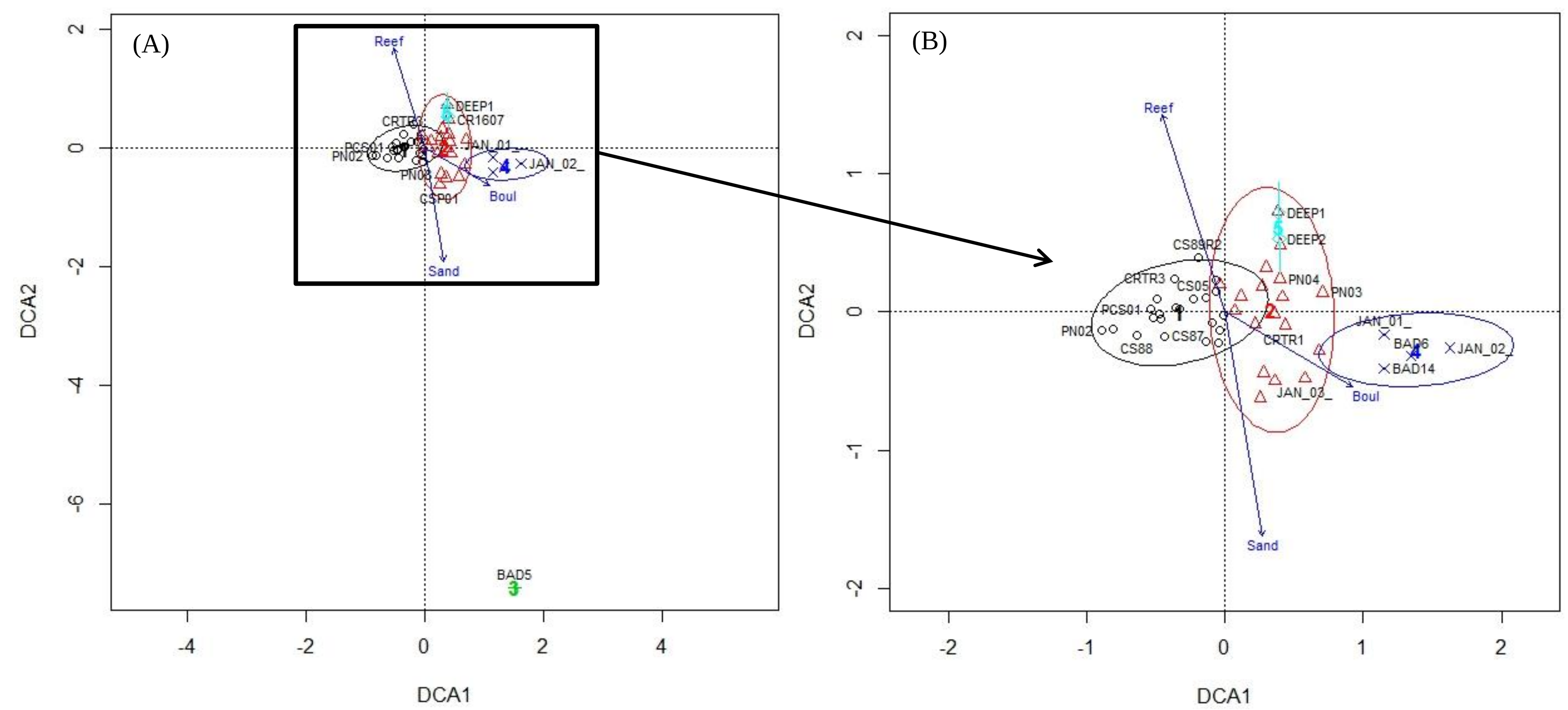

Figure 52: Detrended Correspondence Analysis (DCA); (A) similarity between sites (transects) based on seaweed only functional group abundance, (B) zoomed in view of sites (transects) based on seaweed only functional group abundance (index = Bray Curtis) (Eigen values for DCA1 and DCA2 axes are 0.264 and 0.139 respectively). Effect of significant environmental gradients ($p<0.05$) are shown as arrows. (Codes for sites in Table 17).

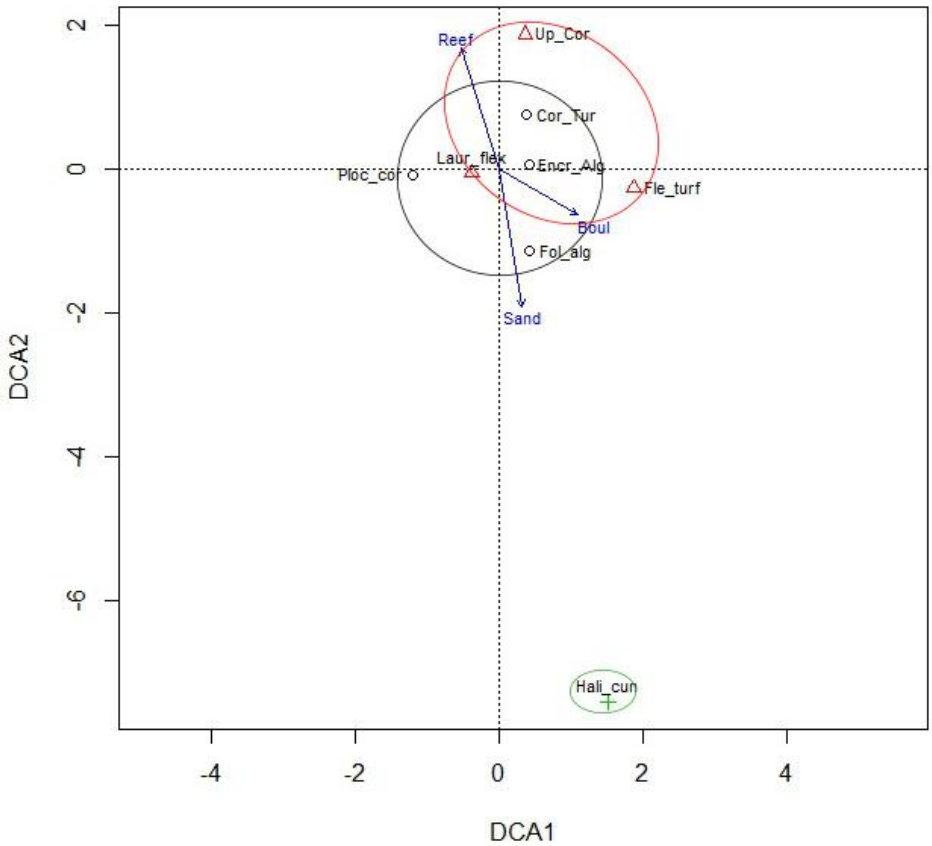

Figure 53: Detrended Correspondence Analysis (DCA) of similarity between seaweed only functional group abundance, based on site association. (index = Bray Curtis) (Eigen values for DCA1 and DCA2 axes are 0.264 and 0.139 respectively). Effect of significant environmental gradients ($p<0.05$) are shown as arrows. (Codes for functional groups in Table 16).

In terms of the invertebrate only functional groups abundance, there were no particular clear groupings in the DCA for sites (Figure 54 (A)), although the very deep sites DEEP1 and DEEP2 do to some degree show separation from the other sites. The functional groups did show a degree of fidelity to particular environmental gradient (Figure 54 (B)). The significant environmental gradients that most greatly influenced were sand and depth in ordination space ($p < 0.05$) (Table 15).

Table 15: Environmental variable vectors and their significance in the ordination space for invertebrate functional groups only (codes in Table 18).

	DCA1	DCA2	r2	Pr(>r)	Significance
Reef	-0.66063	-0.75071	0.1612	0.1	
Boul	-0.99985	-0.01741	0.0101	0.842	
Peb	0.45564	0.89016	0.0206	0.78	
Grav	0.15807	0.98743	0.1101	0.208	
Sand	0.82057	0.57155	0.1984	0.04	*
Dep	-0.46014	0.88785	0.4073	0.003	**
Rug	0.80078	-0.59896	0.0587	0.462	
MVR	0.18835	-0.9821	0.1791	0.062	

5.5 Discussion

5.5.1 Biodiversity and richness of shallow subtidal reef systems of Cape Recife

Subtidal algae continue to remain poorly studied in the Eastern Cape. Anderson and Stegenga (1989) have to date published the most extensive algae species list for Algoa bay, while an unpublished list of 35 macrophyta species was listed by Knoop (1988) for the western sector of Algoa Bay and . The findings of this research confirm 39 species identified for the Cape Recife because of the limited effort to identify the species as a detailed species list was not required for project. It was observed that many of species described were also described by Knoop (1988). Similarly to our results she found a number of calcareous reds and a dominance of *Plocamium corallorhiza* to a depth of 7 metres.

At bird Island a total of 122 species of seaweeds were identified between the intertidal zone and depth of 22 metres (Anderson and Stegenga, 1989). This was more than double the species found in this study, but this could also be resulting from the wider range of habitats and environment that Anderson and Stegenga (1989) sampled in their study. Our research was focused on habitat assumed to be most suitable for abalone and in the depth range of 4 to 9 metres, which is assumed to be a limiting factor in the number of species observed. Anderson and Stegenga (1989) found that there were three communities that appeared to be mainly influenced by water movement. Our results indicate that depth and substrate type were the predominant drivers in the community structuring of Cape Recife. Of particular intrest is the 22 metre sites showing a definitive grouping as compared to the other communities, which was similarily observed by Anderson and Stegenga (1989).

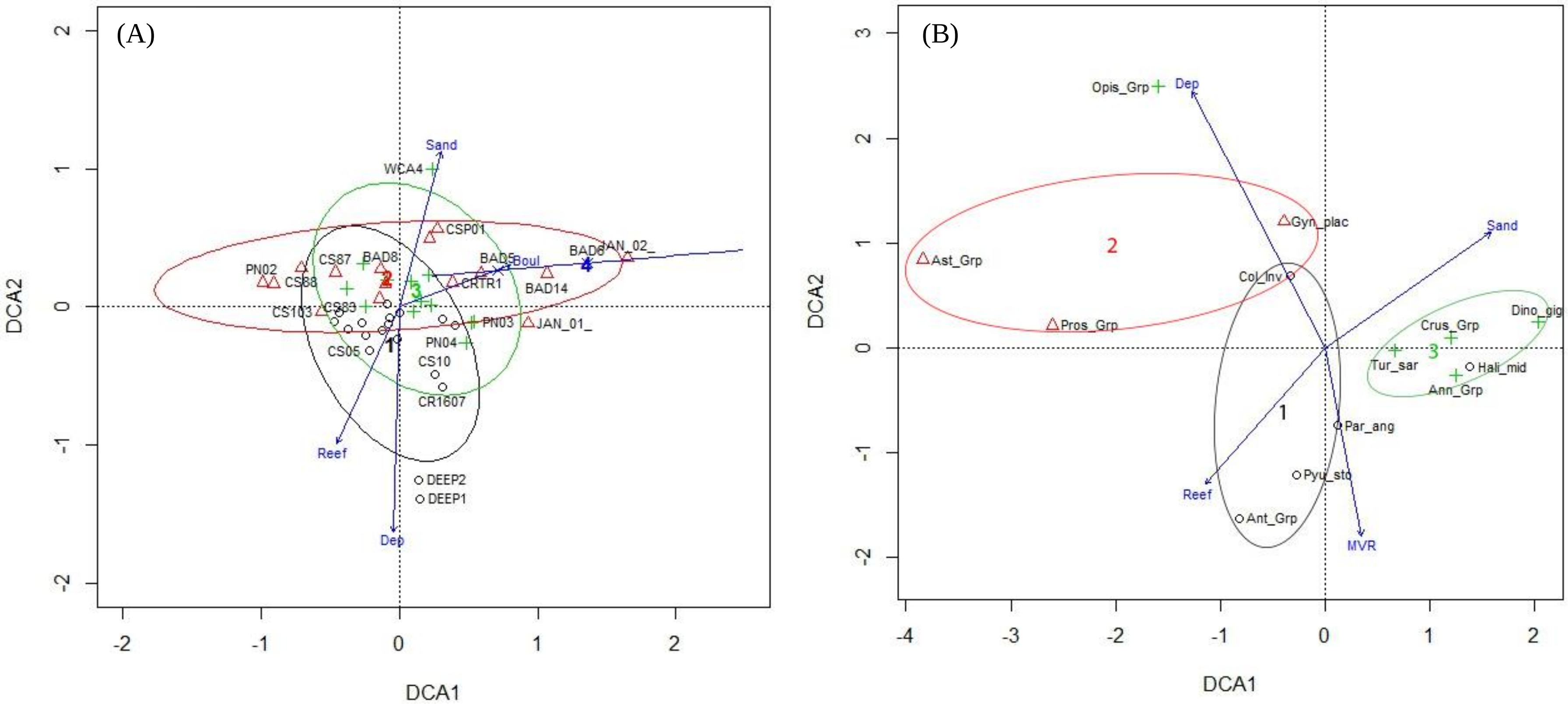

Figure 54: Detrended Correspondence Analysis (DCA); (A) similarity between sites (transects) based on invertebrate only functional group abundance, (B) similarity between invertebrate only functional group abundance based on site association (index = Bray Curtis) (Eigen values for DCA1 and DCA2 axes are 0.221 and 0.147 respectively). Effect of significant environmental gradients ($p<0.05$) are shown as arrows. (Codes for functional groups are given in Table 16 and for sites in Table 17).

Diversity indices of Shannon-Wiener and Simpson were used for functional groups observed at Cape Recife and Noordhoek sites. No significant difference was found in the diversity for the areas. It is to be noted that the Shannon-Wiener diversity index is an indicator sensitive to rare species, whereas the Simpson index is more sensitive to dominance (Mérigot et al., 2007). This research did not specifically measure for rarity or dominance, however a clearly dominant species, *Plocamium corallorhiza* has observed.

Substrate stability is a key determinant of biodiversity in a given area (Zeeman et al., 2013). It has been well recorded that the physical environment coupled with the structural complexity of reefs is directly correlated to diversity and richness (Han and Liu, 2014; Norderhaug et al., 2012), and it also directly influences community structures (Leliaert et al., 2000). It has also been shown that individual species can influence the diversity of a particular area, for example abalone can occupy more than 50% of primary rocky substrate and can collectively with the communities growing on abalone shells and local surrounding communities influence diversity greatly (Zeeman et al., 2013). Another example is *Diadema antillarum* which maintains low algal biomass in the Canarian Archipelago and significantly influences the community structure of the area (Tuya et al., 2004).

5.5.2 Indicators of abalone habitat

Many different recommendations have been made to describe suitable abalone habitat. Wood (1993) described five habitats that ranged in both depth and suitability according to substrate type, availability of cryptic hiding spaces, food availability (mainly *Plocamium*) and animal size. The first three were intertidal to subtidal but less than 1 metre deep, while the fourth was *Caulerpa* beds and finally the subtidal environment. Godfrey (2003) defined abalone habitat at Cape Recife according to substrate type, habitat relief, algal community structure, the degree of wave exposure and depth expressing the need to understand the underlying complexity at micro-scales. The results from this research indicate similar findings to Godfrey (2003), as depth and substrate show the greatest influence on distribution of the community structuring at Cape Recife. We also found a strong association to rugosity and foliose seaweeds, which indicates food availability and high amount of “hiding place” for the animals. A strong association was also found between abalone and the sea-urchin, as well as abalone and coralline species which is confirmed by Day and Branch (2000a) who indicated that coralline species play a very important role in the recruitment of abalone into an area, while urchins are important for the juvenile stages of abalone establishment as they provide protection and regulate coralline growth.

Day and Branch (2000a, 2000b) distinguished between good habitat (Clean rocky reef with encrusting algae that supported recruits) and marginal habitat (sandy, low crevices availability, with foliar algae, sponges, and colonies of ascidians often overlain by fine sediments) in the Western Cape. Day and Branch (2002b) further distinguished between four different abalone recruit habitats according to substrate, biota presence and crevice availability. The habitats included vertical and flat exposed rock, sheltered rock, unsuitable habitat (high sand, gravel and shale) and Urchin shelter. Zeeman et al., (2012) found that feeding was strongly associated to rougher seas at night, as this increases the breaking of algae from the reefs and thus the availability of food, as well as the abalones readiness to feed, although trapping rate increases with calmer conditions. The animals seem to prefer *Ecklonia maxima* and *Plocamium bekeri* in the Western Cape (Zeeman et al., 2012).

Similarly to this study results, habitat complexity has been considered as critically important in site selection for abalone and in community structure. Jalali et al., (2015) considered depth and rugosity as the two most important ant factors in habitat predictive modelling, using LiDAR datasets, for abalone fisheries. It is recommended that the use of continuous rocky reef substrate be the first indicator of suitable habitat. Furthermore the MVR and rugosity must be high to increase habitat complexity. The presence of the main food source *Plocamium corallorhiza* is a good indicator (Wood, 1993) and the presence of sea-urchins has been argued as a good indicator (Day and Branch, 2000a).

The results indicate that in terms of species similarities, the groups of foliose algae, encrusting algae, coralline turf, colonial invertebrates, *Plocamium corallorhiza*, anthozoa, *Parechinus angulosus*, articulated upright corallines, fleshy turf, *Pyura stolonifera*, and *Laurencia flexuosa*, inhabit similar habitat to *Haliotis midae*, and therefore are useful indicators in identifying suitable sites for ranching. These groups and species have been observed and described by numerous authors (Day and Branch, 2000a, 2000b, 2002b; Wood, 1993; Zeeman et al., 2012) as being indicators of suitable habitat for abalone.

Some negative indicators that are also useful include the presence of sandy substrates which is strongly supported in literature (Day and Branch, 2000a, 2000b, 2002b; Godfrey, 2003; Wood, 1993; Wood and Buxton, 1996; Zeeman et al., 2012), low rugosity and the presence of biota such as *Halimeda cuneata*, *Hypnea rosea* and *Dinoplax gigas* which are predominantly associated with very sandy environments.

5.5.3 Is cover an appropriate substitute for Biomass?

In a recent study Zeeman et al., (2014) determined that there is no significant difference in treatments analysed either by biomass or cover. This is similar to the results obtained in this study as strong correlations were found between biotic component biomass and percentage cover for two species and two functional groups. It is often desired to use percentage cover to estimate abundance rather than biomass as the method is less disturbing and destructive to the environment (Chapman and Underwood, 1996). In this study it was found that cover is appropriate and can be used to estimate available food sources. Downing and Anderson (1985) found that quadrat size does not influence the estimation of biomass for any particular area although caution is given that in areas with lower biomass more replicates are required.

5.5.4 The role of depth, rugosity and MVR in community structuring

Depth has been shown to play a very important role in the distribution of communities. Knoop (1988) found changes in the algal community structuring with depth and different productivity for each community. Her results were similar to this study, in that *Plocamium corallorhiza* tended to be dominant in the shallow waters and decrease rapidly with very few observations being made at over 7 metres. Diez et al., (2003) showed that depth along with substrate, wave exposure and other factors is significant in defining specie distribution and average cover decreased significantly with depth. Similar results were seen in Knoop's work, who stopped sampling after 15 metres as cover of seaweeds almost disappeared (Knoop, 1988). Götz et al., (2009) also found a significant decrease in algal abundance as depth increased, which is further confirmation of the results from this study.

It is interesting that colonial invertebrates do show a general increase with depth in this study, which is similar findings by Anderson and Stegenga (1989) who indicated that 40—70% of the cover at 22 metres was sponges. Celliers et al. (2007) also indicated that the deeper reefs of Pondoland tended to have a greater abundance of sponges, ascidians and bryozoans, mixed with some algae, while the shallows were dominated by seaweeds. This was also noted from our results, as well as Anderson and Stegenga (1989), particularly the coralline turf group with species such as *Arthrocardia* and *Amphiroa* being present.

5.6 Conclusion

The aim of this chapter was to further evaluate areas that were predicted as potentially suitable for abalone seeding by the GIS model. Direct relationships were found with six of the functional groups and *Haliotis midae*, while three functional groups showed a direct relationship with depth, six functional groups with rugosity, and one with maximum vertical relief. Three hypotheses that were stated are confirmed by the results as there are direct relationships between rugosity and abalone, complexity and diversity. There is also a direct relationship to foliose algae and cover is an appropriate measure for estimating standing biomass for the Cape Recife area.

The research approach has to some extent addressed many of the unknown attributes and structuring of this area. However there is still much to learn about these dynamic and productive waters. Further research is recommended into the influence of environmental and chemical conditions on the structuring of the subtidal communities. It has been suggested that for example: approaches such as that of Zeeman et al. (2012) are taken to see if there are any feeding or behavioural changes between the South coast and west coast abalone. Furthermore, research is required into the micro-structuring of the environment, including water movement, salinity, temperature fluctuations, potential impacts of extreme events such as storm events and temperature events. It is also recommended that further assessment be undertaken into the impact of ranching on the structure of subtidal communities, as well as genetic structuring of the abalone populations of the Cape Recife area.

Chapter 6. Case Study and Conclusion

6.1 Ranching in the Eastern Cape

The Ranching project at Cape Recife is the first of its kind in South Africa. A typical operation is a 3 step process. The first step is the transport of the animals to launch site, unpacking them and allowing them to recover from the travelling. While this takes place, a dive team is deployed to check and mark suitable seeding sites. These sites are selected from the fuzzy decision support system described in Chapter 3. The sites are also checked by a diver using the assessment methodology described in Chapter 4. This data also is used towards the monitoring program, which is the last step.

Once the sites are confirmed the abalone are brought out to sea, where divers are deployed to release the animals into the most suitable habitat. Up to 20 000 animals can be released on a day. The animals are monitored and survival rates and growth rates have been recorded (pers comm. Warren Witte 2016). Protective services were also provided by the Tactical Task Force to eliminate the risk of poaching on the animals. The habitat monitoring was the final sub-program to be initiated and has been completed in 2015 and 2016.

Being the first commercial operation to successfully run a ranching operation and one of two experimental projects in South Africa, has made this project very exciting and filled with various challenges. The method of seeding, duration of animal recovery, handling and transport of the animals and even the site selection process has evolved continuously throughout the program. Due to the experimental nature of this project it has provided opportunity to further understand the behaviour, adaptability and physiology of abalone, as well is providing an understanding on the fisheries management and current standing stocks in the eastern cape and the future possibilities that abalone ranching may bring.

It is estimated that between 10 to 15% recapture rates are required to obtain a modest profit (Liao et al., 2003). This is the next step in the program. Harvesting has obviously been one of the final goals for the abalone ranching project and upon approval from the relevant authorities the operation will start this stage of the operation. This obviously has a number of impacts that may occur as the process is very destructive as divers overturn rocks and remove cover species in the search for the animals. It is suggested that a monitoring be initiated in order to monitor the impacts and that succession and recovery studies be performed on selected sites to further understand the ecological response to harvesting.

6.2 Conclusion and Recommendations

GIS and spatial planning is a growing part of many industries. According to Uran and Janssen (2003) spatial decision and support systems are judged by their user-friendliness, transparency, and flexibility. The elegance of a fuzzy system is that it addresses the problem of uncertainty in deciding suitability as parameters are not fixed but rather ranges (Hattab et al., 2013). The outputs can be easily interpreted by an end user and often can provide information on boundaries and factors that are not easily measurable, such as soil and slope in landscape analysis (Nath et al., 2000). The implementation of the modelling technique was introduced late in 2015 to the project. The project now relies on the model outputs for identifying plots and has assisted the seeding process in effectively utilizing the available habitat space. It has allowed for shorter search times by divers as prior to the model implementation divers would swim hundreds of metres to identify plots of suitable habitat. GIS has also allowed effective and strategic plot planning, allowing for the identification and implementation of 30 metre buffer zones and thus utilize the limited available habitat space more effectively.

The sampling methods used in this study were intentionally robust and simple. A key aim of this study was the development of sampling methods that could be standardised for use by divers without any scientific training. However, the data collected still needed to be reliable and accurate. These methods were refined through the determination of the optimal transect length, quadrat size, as well as the use of scape and photographic data collection. It was found that the 0.25 m^2 quadrats sufficiently represented the functional groups, and is appropriate sampling units for the area. It was anticipated that the optimum transect length would be greater than the standardized 10 metre plot radius for ranching, but less than 30 metres for the different factors assessed. This was indeed found to be the case, with the results from this study indicating an optimum transect length of 15 metres. The study also illustrated that digital methods, and the use of cover estimates, are suitable as a proxy for biomass. These methods have now been used as a standard sampling protocol for future monitoring of benthic communities in the ranching area.

Haliotis midae is generally known to have very specific habitat requirements, which are known to change with age (Wood, 1993). Research on *Haliotis rubra* (Leach) has shown that factors such as depth, bathymetry, rugosity and habitat complexity are very important in habitat distribution of the species and catch rates become progressively worse as depth increases over 10 metres (Jalali et al., 2015). Results from this study also show the importance of depth and substrate on the distribution of *H. midae*. Depth has been suggested by many authors (Hart et al., 2013b; Jalali et al., 2015; Wood, 1993; Zeeman et al., 2014) to be very important in the distribution, density and growth rate of abalone. The results from this study also show that depth is a factor in distribution and abundance of abalone.

The project has shown in general how much still needs to be learnt about abalone habitat and the environmental factors that influence the success of abalone ranching. While some environmental factors were considered in this study, there was very little focus on the effect of chemical and environmental conditions on habitat suitability for abalone. It is recommended that samples are taken and loggers are deployed in order to monitor chemical factors, such as composition of seawater, salinity, and organic content, and environmental conditions such as pH, temperature, turbidity, wave height and water movement. Another area of research that still needs attention is the feeding patterns and selectivity of abalone to their environment. It is well known that abalone are trap feeders (Hart et al., 2013a; Zeeman et al., 2014, 2012) and the movement and availability of seaweed fragments in the Cape Recife area has not yet been studied.

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Appendix A: Codes for environmental gradients, sites and functional groups

Table 16: Codes for functional Groups

Code	Description
Ploc_cor	*Plocamium corallorhiza*
Laur_flex	*Laurencia flexuosa*
Hali_cun	*Halimeda cuneata*
Fol_alg	Foliose Algae
Fle_turf	Fleshy Turf
Up_Cor	Articulated Upright Corallines
Cor_Tur	Coralline Turf
Encr_Alg	Encrusting Algae
Col_Inv	Colonial Invertebrates
Ant_Grp	Anthozoa
Pyu_sto	*Pyura stolonifera*
Par_ang	*Parenchinus angulosus*
Pros_Grp	Prosobranchia
Gyn_plac	*Gynandrocarpa placenta*
Ast_Grp	Asteridea
Hali_mid	*Haliotis midae*
Tur_sar	*Turbo sarmaticus*
Opis_Grp	Opisthobranchia
Crus_Grp	Crustaceans
Dino_gig	*Dinoplax gigas*
Ann_Grp	Annelida

Table 17: Site codes and descriptions

Code	Depth	Area	GIS Model Rating	Ranching activity	Date visited
CR1507	7.7	Cape Recife	4	None	15-07-2015
CR1607	9.0	Cape Recife	2	None	16-07-2015
CRTR1	5.0	Cape Recife	5	Monitoring	08-04-2015
CRTR2	7.0	Cape Recife	2	Monitoring	08-04-2015
CRTR3	6.8	Cape Recife	5	Monitoring	08-04-2015
CRTR4	5.4	Cape Recife	1	Monitoring	08-04-2015
CS10	7.3	Cape Recife	3	Seeding	12-05-2015
PN01	6.0	Noordhoek	5	None	07-10-2015
PN02	5.8	Noordhoek	5	None	07-10-2015
CS79	8.3	Cape Recife	4	Seeding/Monitoring	30-10-2015
CS80	7.1	Cape Recife	5	Seeding/Monitoring	30-10-2015
CS83	7.5	Cape Recife	4	Seeding/Monitoring	06-11-2015
PN03	6.5	Noordhoek	5	None	06-11-2015
CS05	7.3	Cape Recife	4	Seeding	12-05-2015
PN04	5.9	Noordhoek	5	None	08-11-2015
PN05	6.1	Noordhoek	5	None	08-11-2015
CS85	7.3	Cape Recife	5	Seeding/Monitoring	08-11-2015
CS87	7.9	Cape Recife	5	Seeding/Monitoring	09-11-2015
CS88	7.1	Cape Recife	4	Seeding/Monitoring	09-11-2015
CS89	7.9	Cape Recife	4	Seeding/Monitoring	09-11-2015
BAD5	7.9	Cape Recife	0	None	08-12-2015
BAD6	8.0	Cape Recife	0	None	08-12-2015
BAD8	6.1	Cape Recife	0	None	08-12-2015
BAD14	6.1	Cape Recife	0	None	08-12-2015
PN06	6.3	Noordhoek	3	None	15-12-2015
PN07	5.4	Noordhoek	3	None	15-12-2015
PN08	4.0	Noordhoek	3	None	15-12-2015
CSP01	8.6	Cape Recife	0	None	15-12-2015
JAN_01_	8.8	Cape Recife	0	None	22-01-2016
JAN_02_	8.7	Cape Recife	0	None	22-01-2016
JAN_03_	9.0	Cape Recife	0	None	22-01-2016
JAN_04_	9.2	Cape Recife	0	None	22-01-2016
MAR_09_	8.4	Cape Recife	4	None	09-03-2016
CS103	8.5	Cape Recife	3	Seeding	09-03-2016
MAR_16_	8.2	Cape Recife	3	None	09-03-2016
MAR_20_	22.0	Cape Recife	4	None	09-03-2016
DEEP1	22.1	Noordhoek	0	None	12-05-2016
DEEP2	7.1	Noordhoek	0	None	12-05-2016
PCS02	7.5	Cape Recife	4	Monitoring	09-06-2016
PCS01	7.4	Cape Recife	4	Monitoring	13-05-2016
WCA3	5.3	Cape Recife	5	Monitoring	08-06-2016

Table 7: Site codes and descriptions (Cont.)					
WCA4	6.5	Cape Recife	1	Monitoring	08-06-2016
WCA2	6.6	Cape Recife	2	Monitoring	09-06-2016
PCS4	6.9	Cape Recife	5	Monitoring	09-06-2016
PCS3	7.7	Cape Recife	4	Monitoring	12-05-2016

Table 18: Codes for environmental gradients

Code	**Description**
Reef	Unbroken rock; mostly bedrock
Boul	Large loose rocks >30cm diameter
Peb	Small loose, rounded and smoothed rocks less than 30cm but greater than 5cm
Grav	Small particles less than 5cm but greater than 1 cm (Including shale and small shells).
Sand	Particles less than 1cm
Dep	Depth measurement
Rug	Rugosity
MVR	Maximum Vertical Relief.

Appendix B: Glossary of terms

Geographic Information System (GIS): A computer system for capturing, storing, analysing and displaying spatial and temporal phenomenon.

Multi-criteria Decision Support System (MCDC): A toolset that assesses conflicting criteria and assists user in making decisions that most optimally suit criteria overlap.

Extent: The geographic boundaries of the area being used in analysis

Georeferencing: A process of spatially orientating an image, or giving spatial reference to data that previously did not have any.

Supervised Classification: User defined land use classes chosen by the user, for a particular study area based on pixel signatures

Unsupervised Classification: Machine defined land use classes for a particular study area based on pixel signatures, where the user does not choose the classes

Membership Functions: Mathematical rule sets applied to a particular layer

Memberships: The output raster of a layer that has been processed through the use of a membership function

Rasterization: The process of converting a layer from a vector layer into a raster layer

Fuzzification: The process of overlaying memberships to create a single output raster.

Ground Truthing: The vigorous testing and processing, whereby the models predicted outputs are tested in the field.

Crossvalidation: The process of interpolating values by removing the measured model values and predicting the value for the point and comparing their relationship.

Rugosity: The ratio describing how smooth or rugged the surface of the substrate is.

Maximum Vertical Relief (MVR): The measure in the maximum change of depth for a particular area along a transect line with regards to the peak of the highest pinnacle and the trough of the lowest gully along the line.

Appendix C: R Scripts 2015-2017

```
#Model truthing and validation for GIS Fuzzy Approach
#Load Library
library(moments)
library(e1071)
#Load data
Mydata <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Substrate.csv", sep = ",",
header = T)
Mydata<- na.omit(Mydata)
X <- Mydata$S
A <- Mydata$Visual
Y <- Mydata$DS
Z <- Mydata$DSR

#T.test
t.test(X,Z)

#Model fitting
fit<- lm(A ~ X, data=Mydata)
fit1<- lm(A ~ X + 0, data=Mydata)
fit2<- lm(A ~ Y, data=Mydata)
fit3<- lm(A ~ Y + 0, data=Mydata)
fit4<- lm(A ~ Z, data=Mydata)
fit5<- lm(A ~ Z + 0, data=Mydata)

#summary
summary (fit)
summary (fit1)
summary (fit2)
summary (fit3)
summary (fit4)
summary (fit5)

#Correlation
A1<-round(cor(X,A),3)
A1
A2<-round(cor(Y,A),3)
A2
A3 <- round(cor(Z,A),3)
A3

t.test(X,Y)
t.test(Z,Y)
t.test(X,Z)
#Create a new window.
windows(48, 23)
par(mfrow=c(1,3))
```

```
##Plot models
plot(X,A, ann = FALSE, pch=20, col = "grey60", cex.axis = 1)
mtext(side = 1, text = "Model Ranking", cex = 0.8, line = 2)

mtext(side = 2, text = "Visual Ranking", cex = 0.8, line = 2)

mtext(side = 1, text = "(A)", line = 4)

abline(fit, lwd = 2, lty=2)
abline(fit1, lwd = 2, lty=3, col = "red")
lines(lowess(X,A), col="blue")

p1 <- summary(fit1)$coefficients[,4]
p1
r2a <- round(summary(fit1)$r.squared, 3)
eqn <- bquote(atop(paste (r^2 == .(r2a) * "," ~~ r == .(A1) * ","),
        paste(p < 0.005 * "," ~~ n == 109)))
text(0.1, 4.5, eqn, pos = 4,cex = 1)

##Fit1
plot(Y,A, ann = FALSE, pch=20, col = "grey60", cex.axis = 1)
mtext(side = 1, text = "Model Ranking", cex = 0.8, line = 2)

mtext(side = 2, text = "Visual Ranking", cex = 0.8, line = 2)

mtext(side = 1, text = "(B)", line = 4)

abline(fit2, lwd = 2, lty=2)
abline(fit3, lwd = 2, lty=3, col = "red")
lines(lowess(Y,A), col="blue")


p1 <- summary(fit3)$coefficients[,4]
p1
r2b <- round(summary(fit3)$r.squared, 3)
eqn <- bquote(atop(paste(r^2 == .(r2b) * "," ~~ r == .(A2)
          * ","), paste(p < 0.005 * "," ~~ n == 109)))
text(0.1, 4.5, eqn, pos = 4,cex = 1)

par(xpd=T)

legend(0.4,5.85, c("Y = mX + c", "Y = mX + 0", "Line of best fit"),
    lty=c(2,3,1), lwd = c(2,2,1), col = c("black" , "red" , "blue"),cex = 1)
par(xpd=F)

##Fit2
plot(Z,A, ann = FALSE, pch=20, col = "grey60", cex.axis = 1)
mtext(side = 1, text = "Model Ranking", cex = 0.8, line = 2)

mtext(side = 2, text = "Visual Ranking", cex = 0.8, line = 2)
```

```
mtext(side = 1, text = "(C)", line = 4)

abline(fit4, lwd = 2, lty=2)
abline(fit5, lwd = 2, lty=3, col = "red")
lines(lowess(Z,A), col="blue")

p1 <- summary(fit5)$coefficients[,4]
p1
r2c <- round(summary(fit5)$r.squared, 3)
eqn <- bquote(atop(paste(r^2 == .(r2c) * "," ~~ r == .(A3)
                * ","), paste(p < 0.005 * "," ~~ n == 109)))
text(-0.02, 4.5, eqn, pos = 4,cex = 1)

### Chapter 4 and Chapter 5 graphics

###Species Area and Diversity
library(moments)
library(e1071)
library(vegan)
Sites <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Biomass50.csv", sep = ",", header = T)
Sites1<-t(Sites)
Mydata <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Biomass_Sum.csv", sep = ",", row.names = 1, header = T)
Mydata1<- t(Mydata)
Mydata2 <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Mean_Cover.csv", sep = ",", row.names = 1, header = T)
Mydata3<- t(Mydata2)

TLA<-read.table("C:/Users/Andrew/Documents/Masters/Analysis/TLA.csv", sep = ",", row.names = 1, header = T)
TLA1<- (TLA)
SP <- diversity(Mydata)
SP

plt<-diversity(Sites)
plt
CurT <- specaccum(TLA1, method = "random")
windows(15,10)
plot(CurT, xlab = expression(Area~(m^2)) , ylab = "Functional Group Richness", ci=2, ci.type = "polygon", ci.col = "grey95", ci.lty = 0, xaxt = "n")
axis(1, at = c(0, 0.5,1,1.5,2,2.5,3,3.5), labels = c("0", "0.0156","0.0625","0.01406","0.25", "0.3906","1","2"))

windows()
par(mfrow=c(1,2))

Cur <- specaccum(Mydata1, method = "coleman")
```

```
plot(Cur, xlab = expression(Area~(m^2)) , ylab = "Mean Number of Functional Groups", ci=2, ci.type = "polygon", ci.col = "grey95", ci.lty = 0, xaxt = "n", cex=0.8)
axis(1, at = c(0, 0.5,1,1.5,2,2.5,3,3.5), labels = c("0", "0.0156","0.0625","0.01406","0.25", "0.3906","1","2"))
Cur1 <- specaccum(Mydata3, method = "coleman")
plot(Cur1, xlab = expression(Area~(m^2)) , ylab = "Mean Number of Functional Groups", ci=2, ci.type = "polygon", ci.col = "grey95", ci.lty = 0, xaxt = "n", cex=0.8)
axis(1, at = c(0, 0.5,1,1.5,2,2.5,3,3.5), labels = c("0", "0.0156","0.0625","0.01406","0.25", "0.3906","1","2"))

Cur <- specaccum(Sites1, method = "coleman")
plot(Cur, xlab = "Number of Quadrats" , ylab = "Mean Number of Functional Groups", ci=2, ci.type = "polygon", ci.col = "grey95", ci.lty = 0)

mod1<-fitspecaccum(Cur, "lomolino")
mod1
coef(mod1)
fitted(mod1)
Cur1 <- specaccum(Sites1, method = "random", permutations = 100)
Cur1
summary(Cur1)
boxplot(Cur1, add = TRUE)

specpool(Mydata)
pool <- poolaccum(Mydata, minsize = 11, permutations = 100)
summary(pool)
windows(15,10)
plot(pool)

pool <- specpool(Mydata1)
pool

sp1 <- specaccum(Mydata1)
sp2 <- specaccum(Mydata1, "random")
sp2
summary(sp2)
plot(sp1, ci.type="poly", lwd=2, ci.lty=0, ci.col="Grey95")
boxplot(sp2, col="grey65", add=TRUE, pch="+")
sp1$sites

library (vegan)
library(plyr)
library(lattice)
library(reshape2)
data(BCI)

H <- diversity(BCI)
simp <- diversity(BCI, "simpson")
invsimp <- diversity(BCI, "inv")
r.2 <- rarefy(BCI, 2)
```

```
Sp   <-   read.table("C:/Users/Andrew/Documents/Masters/Analysis/Mon/Div3.csv",   sep   =   ",",
row.names = 1, header = T, fill = T)
Sp1   <-   read.table("C:/Users/Andrew/Documents/Masters/Analysis/Mon/Div4.csv",   sep   =   ",",
row.names = 1, header = T, fill = T)
Sp2   <-   read.table("C:/Users/Andrew/Documents/Masters/Analysis/Mon/Div5.csv",   sep   =   ",",
row.names = 1, header = T, fill = T)
T.sp<- as.data.frame(t(Sp))

####DATA SUMMARIZE

#####NH AND CR#####
Frm1<-as.data.frame(diversity(Sp2, index = "shannon", base = exp(1)))
colnames(Frm1) <- ("SH")
Frm1
Frm2<-as.data.frame(diversity(Sp2, index = "inv"))
colnames(Frm2) <- "Sim"
Frm2
is.na(Frm2) <- sapply(Frm2, is.infinite)
df2 <- merge(Frm1,Frm2, by = 0)
df2<-na.omit(df2)
df2$ID<-"Cape Recife"
df2
df <- merge(Frm1,Frm2, by = 0)
Frm1<-as.data.frame(diversity(Sp1, index = "shannon", base = exp(1)))
colnames(Frm1) <- ("H'")
Frm1
Frm2<-as.data.frame(diversity(Sp1, index = "inv"))
colnames(Frm2) <- "??"
Frm2
df1 <- merge(Frm1,Frm2, by = 0)
df1$ID<-"Noordhoek"
df1
df3 <- rbind(df1,df2)
df3


data <- ddply(df3, c("ID"), summarise,
            n = length(SH),
            Mean = mean(SH),
            SD = sd(SH),
            SE = SD / sqrt(n))
data$IDX<-"H'"
data
data1 <- ddply(df3, c("ID"), summarise,
             n = length(Sim),
             Mean = mean(Sim),
             SD = sd(Sim),
             SE = SD / sqrt(n))
data1$IDX<- "??"
```

```
data1
df <- rbind(data,data1)
df
t.test(df2$Sim,df3$Sim)
#####ALL SITES########### Not important
Frm1<-as.data.frame(diversity(Sp, index = "shannon", base = exp(1)))
Frm1
colnames(Frm1) <- ("SH")
Frm2<-as.data.frame(diversity(Sp, index = "inv"))
Frm2
colnames(Frm2) <- "Sim"
is.na(Frm2) <- sapply(Frm2, is.infinite)

df <- merge(Frm1,Frm2, by = 0)
df<-na.omit(df)
df$ID<- "All"
df
data <- ddply(df, c("ID"), summarise,
              n = length(SH),
              Mean = mean(SH),
              SD = sd(SH),
              SE = SD / sqrt(n))
data$IDX<-"SH"
data
data1 <- ddply(df, c("ID"), summarise,
               n = length(Sim),
               Mean = mean(Sim),
               SD = sd(Sim),
               SE = SD / sqrt(n))
data1$IDX<- "Sim"
data1
df5 <- rbind(data,data1)
df5

df<-rbind(df4,df5)
df

###Draw Plot

tapply(df$Mean, list(df$IDX, df$ID),
       function(x) c(x = x))

Bar <- tapply(df$Mean, list(df$IDX, df$ID),
              function(x) c(x = x))
SE <- tapply(df$SE, list(df$IDX, df$ID),
             function(x) c(x = x))
windows()
barCenters <- barplot(height = Bar,
                      beside = TRUE, las = 1, ylim = c(0,8),
                      cex.names = 0.75, names.arg = c("Cape Recife", "Noordhoek"),
```

```
                    ylab = "Index",
                    xlab = "",
                    border = "black", axes = TRUE,
                    legend.text = TRUE,
                    args.legend = list(x = "topright",
                                       cex = .7))

segments(barCenters, Bar - SE * 2, barCenters,
         Bar + SE * 2, lwd = 1.5)

arrows(barCenters, Bar - SE * 2, barCenters,
       Bar + SE * 2, lwd = 1.5, angle = 90,
       code = 3, length = 0.05)

###Multibarplot with Error
Mydata <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Quad_Size.csv", sep = ",",
header = T)

library(Hmisc)
library(vegan)
library(plyr)
##Scape

data <- ddply(Mydata, c("Size"), summarise,
              n = length(Scrape),
              Mean = mean(Scrape),
              SD = sd(Scrape),
              SE = SD / sqrt(n),
              CV = SD/Mean)
data
sd(data1$CV)/sqrt(3)
data$ID <- "Scrape"
data
##Images
data1 <- ddply(Mydata, c("Size"), summarise,
               n = length(Image),
               Mean = mean(Image),
               SD = sd(Image),
               SE = SD / sqrt(n),
               CV = SD/Mean)
data1
data1$ID <- "Image"
#merge by row
df <- rbind(data,data1)

#output
df
head(df)
df$CVse <- c(0.022,0.022,0.022,0.053,0.053,0.053)
tapply(df$CV, list(df$ID, df$Size),
```

```
      function(x) c(x = x))

tabbedCV <- tapply(df$CV, list(df$ID, df$Size),
            function(x) c(x = x))
tabbedSE <- tapply(df$CVse, list(df$ID, df$Size),
            function(x) c(x = x))
windows()
barCenters <- barplot(height = tabbedCV,
              beside = TRUE, las = 1,
              ylim = c(0, 0.5),
              cex.names = 0.75, names.arg = c(0.0625, 0.25, 1),
              ylab = "Coefficient of Variation (CV)",
              xlab = expression(Quadrat~~Size~~(m^2)),
              border = "black", axes = TRUE,
              legend.text = TRUE,
              args.legend = list(x = "topright",
                          cex = .7))

segments(barCenters, tabbedCV - tabbedSE * 2, barCenters,
       tabbedCV + tabbedSE * 2, lwd = 1.5)

arrows(barCenters, tabbedCV - tabbedSE * 2, barCenters,
      tabbedCV + tabbedSE * 2, lwd = 1.5, angle = 90,
      code = 3, length = 0.05)

###Chapter 5
#Relationships and correlations Ecology of the habitat.
##Load Library
library(vegan)
library(MASS)
library(effects)
library(lme4)
library(lmtest)
library(nlme)
library(Hmisc)
##Read in data
SP <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/Relate.csv", sep = ",",
row.names = 1, header = T)
names(SP)

x1 <- SP$Hali_mid
x2 <- SP$Depth
x3 <- SP$Rugosity
x4 <- SP$MRV

y1 <- SP$Ploc_cor
y2 <- SP$Foliose
y3 <- SP$Amph_eph
y4 <- SP$Cor_Tur
y5 <- SP$Encrust_Alg
```

```
y6 <- SP$Turf
y7 <- SP$Laur_flex
y8 <- SP$Turbo_sar
y9 <- SP$Pyura
y10 <- SP$Col_Inverts
y11 <- SP$Dinoplax_gigas
y12 <- SP$Par_ang

###Correlation
#Abs
cor.test(x1,y1)
cor.test(x1,y2)
cor.test(x1,y3)
cor.test(x1,y4)
cor.test(x1,y5)
cor.test(x1,y6)
cor.test(x1,y7)
cor.test(x1,y8)
cor.test(x1,y9)
cor.test(x1,y10)
cor.test(x1,y11)
cor.test(x1,y12)
cor.test(x1,x2)
cor.test(x1,x3)
cor.test(x1,x4)
#Depth
cor.test(x2,y1)
cor.test(x2,y2)
cor.test(x2,y3)
cor.test(x2,y4)
cor.test(x2,y5)
cor.test(x2,y6)
cor.test(x2,y7)
cor.test(x2,y8)
cor.test(x2,y9)
cor.test(x2,y10)
cor.test(x2,y11)
cor.test(x2,y12)
cor.test(x2,x3)
cor.test(x3,x4)
#Rug
cor.test(x3,y1)
cor.test(x3,y2)
cor.test(x3,y3)
cor.test(x3,y4)
cor.test(x3,y5)
cor.test(x3,y6)
cor.test(x3,y7)
cor.test(x3,y8)
cor.test(x3,y9)
```

```
cor.test(x3,y10)
cor.test(x3,y11)
cor.test(x3,y12)

#MVR
cor.test(x4,y1)
cor.test(x4,y2)
cor.test(x4,y3)
cor.test(x4,y4)
cor.test(x4,y5)
cor.test(x4,y6)
cor.test(x4,y7)
cor.test(x4,y8)
cor.test(x4,y9)
cor.test(x4,y10)
cor.test(x4,y11)
cor.test(x4,y12)

#Models
windows()
par(mfrow=c(2,3))

#1
plot(y1 ~ x1, pch=1, xlab=expression( italic ("Haliotis midae") ~ "count per transect"), ylim = c(0,80), ylab=expression( italic ("Plocamium corallorhiza")~~"Cover (%)"))
mod1 <- lm(y1~x1, data = SP)
summary(mod1)
abline(mod1)
cor.test(x1,y1)
p1 <- round(summary(mod1)$coefficients[2,4], 3)
p1
r1<-round(cor(x1,y1),3)
r2a <- summary(mod1)$r.squared
eqn1 <- bquote(atop(paste(r^2 < 0.001
                    * "," ~~ cor == .(r1)
                    * ","), paste(p == .(p1))))
text(14, 70, eqn1, pos = 4,cex = 1)

par(xpd=TRUE)
text(0, 88, "(1)",cex = 1)
par(xpd=FALSE)


#2
plot(y2~x1, pch=1, xlab=expression( italic ("Haliotis midae") ~ "count per transect"), ylim = c(0,55), ylab="Foliose Seaweeds Cover (%)")
mod2 <- lm(y2~x1, data = SP)
summary(mod2)
abline(mod2)
cor.test(x1,y2)
```

```
p2 <- round(summary(mod2)$coefficients[2,4], 3)
p2
r2<-round(cor(x1,y2),3)
r2b <- round(summary(mod2)$r.squared, 3)
eqn2 <- bquote(atop(paste(r^2 == .(r2b)
                    * "," ~~ cor == .(r2)
                    * ","), paste(p == .(p2))))
text(13, 50, eqn2, pos = 4,cex = 1)

par(xpd=TRUE)
text(0, 60, "(2)",cex = 1)
par(xpd=FALSE)

#3
plot(y3~x1, pch=1, xlab=expression( italic ("Haliotis midae") ~ "count per transect"), ylim = c(0,30), ylab=expression( italic ("Amphiroa ephedraea")~~"Cover (%)"))
mod3 <- lm(y3~x1, data = SP)
summary(mod3)
abline(mod3)
cor.test(x1,y3)
p3 <- round(summary(mod3)$coefficients[2,4], 3)
p3
r3<-round(cor(x1,y3),3)
r2c <- round(summary(mod3)$r.squared, 3)
eqn3 <- bquote(atop(paste(r^2 == .(r2c)
                    * "," ~~ cor == .(r3)
                    * ","),paste (p == .(p3))))
text(14, 28, eqn3, pos = 4,cex = 1)

par(xpd=TRUE)
text(0, 32.5, "(3)",cex = 1)
par(xpd=FALSE)

#4
plot(y4~x1, pch=1, xlab=expression( italic ("Haliotis midae") ~ "count per transect"), ylim = c(0,50), ylab="Coralline Turf Cover (%)")
mod4 <- lm(y4~x1, data = SP)
summary(mod4)
abline(mod4)
cor.test(x1,y4)
p4 <- round(summary(mod4)$coefficients[2,4], 3)
p4
r4<-round(cor(x1,y4),3)
r2d <- round(summary(mod4)$r.squared, 3)
eqn4 <- bquote(atop(paste(r^2 == .(r2d)
                    * "," ~~ cor == .(r4)
                    * ","), paste(p == .(p4))))
text(14, 45, eqn4, pos = 4,cex = 1)

par(xpd=TRUE)
```

```
text(0, 54, "(4)",cex = 1)
par(xpd=FALSE)

#5
plot(y5~x1, pch=1, xlab=expression( italic ("Haliotis midae") ~ "count per transect"), ylim = c(0,35), ylab="Encrusting Algae Cover (%)")
mod5 <- lm(y5~x1, data = SP)
summary(mod5)
abline(mod5)
cor.test(x1,y5)
p5 <- round(summary(mod5)$coefficients[2,4], 3)
p5
r5<-round(cor(x1,y5),3)
r2e <- round(summary(mod5)$r.squared, 3)
eqn5 <- bquote(atop(paste(r^2 == .(r2e)
                 * "," ~~ cor == .(r5)
                 * ","),paste (p == .(p5))))
text(13, 32, eqn5, pos = 4,cex = 1)

par(xpd=TRUE)
text(0, 37.75, "(5)",cex = 1)
par(xpd=FALSE)


#6
plot(y6~x1, pch=1, xlab=expression( italic ("Haliotis midae") ~ "count per transect"), ylim = c(0,60), ylab="Turf Cover (%)")
mod6 <- lm(y6~x1, data = SP)
summary(mod6)
abline(mod6)
cor.test(x1,y6)
p6 <- round(summary(mod6)$coefficients[2,4], 3)
p6
r6<-round(cor(x1,y6),3)
r2f <- round(summary(mod6)$r.squared, 3)
eqn6 <- bquote(atop(paste(r^2 == .(r2f)
                 * "," ~~ cor == .(r6)
                 * ","), paste(p == .(p6))))
text(13, 55, eqn6, pos = 4,cex = 1)

par(xpd=TRUE)
text(0, 65, "(6)",cex = 1)
par(xpd=FALSE)

##New Window
windows()
par(mfrow=c(2,3))

#7
```

```
plot(y7~x1, pch=1, xlab=expression( italic ("Haliotis midae") ~ "count per transect"), ylim = c(0,20), ylab=expression( italic ("Laurencia flexuosa")~~"Cover (%)"))
mod7 <- lm(y7~x1, data = SP)
summary(mod7)
abline(mod7)
cor.test(x1,y7)
p7 <- round(summary(mod7)$coefficients[2,4], 3)
p7
r7<-round(cor(x1,y7),3)
r2g <- round(summary(mod7)$r.squared, 3)
eqn7 <- bquote(atop(paste(r^2 == .(r2g)
                   * "," ~~ cor == .(r7)
                   * ","), paste(p == .(p7))))
text(13, 18, eqn7, pos = 4,cex = 1)

par(xpd=TRUE)
text(0, 22, "(7)",cex = 1)
par(xpd=FALSE)


#10
plot(y10~x1, pch=1, xlab=expression( italic ("Haliotis midae") ~ "count per transect"), ylim = c(0,55),  ylab= ("Colonial Invertebrates Cover (%)"))
mod10 <- lm(y10~x1, data = SP)
summary(mod10)
abline(mod10)
cor.test(x1,y10)
p10 <- round(summary(mod10)$coefficients[2,4], 3)
p10
r10<-round(cor(x1,y10),3)
r2j <- round(summary(mod10)$r.squared, 3)
eqn10 <- bquote(atop(paste(r^2 == .(r2j)
                    * "," ~~ cor == .(r10)
                    * ","), paste(p == .(p10))))
text(13, 50, eqn10, pos = 4,cex = 1)

par(xpd=TRUE)
text(0, 60, "(8)",cex = 1)
par(xpd=FALSE)

#9
plot(y9~x1, pch=1, xlab=expression( italic ("Haliotis midae") ~ "count per transect"), ylim = c(0,55),  ylab=expression( italic ("Pyura stolonifera")~~"Cover (%)"))
mod9 <- lm(y9~x1, data = SP)
summary(mod9)
abline(mod9)
cor.test(x1,y9)
p9 <- round(summary(mod9)$coefficients[2,4], 3)
p9
r9<-round(cor(x1,y9),3)
```

```
r2i <- round(summary(mod9)$r.squared, 3)
eqn9 <- bquote(atop(paste(r^2 == .(r2i)
                      * "," ~~ cor == .(r9)
                      * ","), paste(p == .(p9))))
text(13, 50, eqn9, pos = 4,cex = 1)

par(xpd=TRUE)
text(0, 59.5, "(9)",cex = 1)
par(xpd=FALSE)

#8
plot(y8~x1, pch=1, xlab=expression( italic ("Haliotis midae") ~ "count per transect"), ylim = c(0,4),
ylab=expression( italic ("Turbo sarmaticus")~~"Count"))
mod8 <- lm(y8~x1, data = SP)
summary(mod8)
abline(mod8)
cor.test(x1,y8)
p8 <- round(summary(mod8)$coefficients[2,4], 3)
p8
r8<-round(cor(x1,y8),3)
r2h <- round(summary(mod8)$r.squared, 3)
eqn8 <- bquote(atop(paste(r^2 == .(r2h)
                      * "," ~~ cor == .(r8)
                      * ",") ,paste(~~ p == .(p8))))
text(13, 3.5, eqn8, pos = 4,cex = 1)

par(xpd=TRUE)
text(0, 4.35, "(10)",cex = 1)
par(xpd=FALSE)

#11
plot(y11~x1, pch=1, xlab=expression( italic ("Haliotis midae") ~ "count per transect"), ylim =
c(0,20), ylab=expression( italic ("Dinoplax giga")~~"Count"))
mod11 <- lm(y11~x1, data = SP)
summary(mod11)
abline(mod11)
cor.test(x1,y11)
p11 <- round(summary(mod11)$coefficients[2,4], 3)
p11
r11<-round(cor(x1,y11),3)
r2k <- round(summary(mod11)$r.squared, 3)
eqn11 <- bquote(atop(paste(r^2 == .(r2k)
                      * "," ~~ cor == .(r11)
                      * ","), paste( ~~ p < 0.001)))
text(13, 18, eqn11, pos = 4,cex = 1)

par(xpd=TRUE)
text(0, 22, "(11)",cex = 1)
par(xpd=FALSE)
```

```
#12
plot(y12~x1, pch=1, xlab=expression( italic ("Haliotis midae") ~ "count per transect"), ylim =
c(0,7), ylab=expression( italic ("Parechinus angulosus")~~"Count"))
mod12 <- lm(y12~x1, data = SP)
summary(mod12)
abline(mod12)
cor.test(x1,y12)
p12 <- round(summary(mod12)$coefficients[2,4], 3)
p12
r12<-round(cor(x1,y12),3)
r2l <- round(summary(mod12)$r.squared, 3)
eqn12 <- bquote(atop(paste(r^2 == .(r2l)
                   * "," ~~ cor == .(r12)
                   * ","),paste( p == .(p12))))
text(13, 6.2, eqn12, pos = 4,cex = 1)

par(xpd=TRUE)
text(0, 7.55, "(12)",cex = 1)
par(xpd=FALSE)

####################################################################################
############################################################
##################################Abs
Relate###############################################################################
#####################
#Models
windows()
par(mfrow=c(2,3))

#1 Ab_Depth
###Plot graphic
plot(x1 ~ x2, pch=1, xlab="Depth (m)", ylim = c(0,18), ylab=expression( italic ("Haliotis
midae")~~"count per transect"))

mod1 <- lm(x1~x2, data = SP)
summary(mod1)
abline(mod1)
cor.test(x1,x2)
p1 <- round(summary(mod1)$coefficients[2,4], 3)
p1
r1<-round(cor(x1,x2),2)
r2a <- round(summary(mod1)$r.squared,3)
eqn1 <- bquote(atop(paste(r^2 == .(r2a)
                  * "," ~~ cor == .(r1)
                  * ","),paste( p == .(p1))))
text(16, 16, eqn1, pos = 4,cex = 1)

par(xpd=TRUE)
text(4, 19.5, "(1)",cex = 1)
par(xpd=FALSE)
```

```
#2 Ploc_Depth
plot(y1 ~ x2, pch=1, xlab="Depth (m)", ylim = c(0,80), ylab=expression( italic ("Plocamium corallorhiza")~~"Cover (%)"))
mod1 <- lm(y1~x2, data = SP)
summary(mod1)
abline(mod1)
cor.test(x2,y1)
p1 <- round(summary(mod1)$coefficients[2,4], 3)
p1
r<-round(cor(x2,y1),2)
r2a <- round(summary(mod1)$r.squared,3)
eqn1 <- bquote(atop(paste(r^2 == .(r2a)
                     * "," ~~ cor == .(r)
                     * ","), paste( p == .(p1))))
text(15.5, 70, eqn1, pos = 4,cex = 1)

par(xpd=TRUE)
text(4, 86, "(2)",cex = 1)
par(xpd=FALSE)

#3 Foliose_Depth
plot(y2~x2, pch=1, xlab="Depth (m)", ylim = c(0,55), ylab="Foliose Seaweeds Cover (%)")
mod2 <- lm(y2~x2, data = SP)
summary(mod2)
abline(mod2)
cor.test(x2,y2)
p2 <- round(summary(mod2)$coefficients[2,4], 3)
p2
r<-round(cor(x2,y2),3)
r2b <- round(summary(mod2)$r.squared, 3)
eqn2 <- bquote(atop(paste(r^2 == .(r2b)
                     * "," ~~ cor == .(r)
                     * ","), paste(p == .(p2))))
text(16, 50, eqn2, pos = 4,cex =1)

par(xpd=TRUE)
text(4, 59.5, "(3)",cex = 1)
par(xpd=FALSE)

#4 Encrust_Depth
plot(y5~x2, pch=1, xlab="Depth (m)", ylim = c(0,30), ylab="Encrusting Algae Cover (%)")
mod2 <- lm(y5~x2, data = SP)
summary(mod2)
abline(mod2)
cor.test(x2,y5)
p2 <- round(summary(mod2)$coefficients[2,4], 3)
p2
r<-round(cor(x2,y5),3)
r2b <- round(summary(mod2)$r.squared, 3)
```

```
eqn2 <- bquote(atop(paste(r^2 == .(r2b)
                    * "," ~~ cor == .(r)
                    * ","), paste(p == .(p2))))
text(14.5, 27.5, eqn2, pos = 4,cex = 1)

par(xpd=TRUE)
text(4, 32.5, "(4)",cex = 1)
par(xpd=FALSE)

#1 Ab_Rug
plot(x1 ~ x3, pch=1, xlab= "Rugosity", ylim = c(0,22), ylab=expression( italic ("Haliotis midae")~~"Count"))
mod1 <- lm(x1~x3, data = SP)
summary(mod1)
abline(mod1)
cor.test(x3,x1)
p1 <- round(summary(mod1)$coefficients[2,4], 3)
p1
r1<-round(cor(x3,x1),2)
r2a <- round(summary(mod1)$r.squared,3)
eqn1 <- bquote(atop(paste(r^2 == .(r2a)
                    * "," ~~ cor == .(r1)
                    * ","),paste( p == .(p1))))
text(0.25, 20, eqn1, pos = 4,cex = 1)

par(xpd=TRUE)
text(0.02, 23.8, "(5)",cex = 1)
par(xpd=FALSE)

#2 Amp_rug
plot(y3~x3, pch=1, xlab="Rugosity", ylim = c(0,30), ylab=expression( italic ("Amphiroa ephedraea")~~"Cover (%)"))
mod3 <- lm(y3~x3, data = SP)
summary(mod3)
abline(mod3)
cor.test(x3,y3)
p3 <- round(summary(mod3)$coefficients[2,4], 3)
p3
r3<-round(cor(x3,y3),3)
r2c <- round(summary(mod3)$r.squared, 3)
eqn3 <- bquote(atop(paster^2 == .(r2c)
                * "," ~~ cor == .(r3)
                * ","),vpaste(p == .(p3))))
text(0.25, 27, eqn3, pos = 4,cex = 1)

par(xpd=TRUE)
text(0.02, 32.5, "(6)",cex = 1)
par(xpd=FALSE)

windows()
```

```
par(mfrow=c(1,2))
#3 Coral_rug
plot(y4~x3, pch=1, xlab="Rugosity", ylim = c(0,32), ylab="Coralline Turf Cover (%)")
mod3 <- lm(y4~x3, data = SP)
summary(mod3)
abline(mod3)
cor.test(x3,y4)
p3 <- round(summary(mod3)$coefficients[2,4], 3)
p3
r3<-round(cor(x3,y4),3)
r2c <- round(summary(mod3)$r.squared, 3)
eqn3 <- bquote(atop(paste(r^2 == .(r2c)
                    * "," ~~ cor == .(r3)
                    * ","),paste(p <0.001)))
text(0.25, 29, eqn3, pos = 4,cex = 1)

par(xpd=TRUE)
text(0.02, 34.5, "(1)",cex = 1)
par(xpd=FALSE)


#4 Encrust_rug
plot(y5~x3, pch=1, xlab="Rugosity", ylim = c(0,30), ylab="Encrusting Algae Cover (%)")
mod3 <- lm(y5~x3, data = SP)
summary(mod3)
abline(mod3)
cor.test(x3,y5)
p3 <- round(summary(mod3)$coefficients[2,4], 3)
p3
r3<-round(cor(x3,y5),3)
r2c <- round(summary(mod3)$r.squared, 3)
eqn3 <- bquote(atop(paste(r^2 == .(r2c)
                    * "," ~~ cor == .(r3)
                    * "," ), paste(p == .(p3))))
text(0.25, 27, eqn3, pos = 4,cex = 1)

par(xpd=TRUE)
text(0.02, 32.5, "(2)",cex = 1)
par(xpd=FALSE)

windows()
par(mfrow=c(1,2))
#5 Laurencia_rug
plot(y7~x3, pch=1, xlab="Rugosity", ylim = c(0,30), ylab=expression( italic ("Laurencia
flexuosa")~~"Cover (%)"))
mod3 <- lm(y7~x3, data = SP)
summary(mod3)
abline(mod3)
cor.test(x3,y7)
p3 <- round(summary(mod3)$coefficients[2,4], 3)
```

```
p3
r3<-round(cor(x3,y7),3)
r2c <- round(summary(mod3)$r.squared, 3)
eqn3 <- bquote(atop(paste(r^2 == .(r2c)
                    * "," ~~ cor == .(r3)
                    * ","), paste(p == .(p3))))
text(0.25, 26.5, eqn3, pos = 4,cex = 1)

par(xpd=TRUE)
text(0.02, 32.3, "(3)",cex = 1)
par(xpd=FALSE)

#6 Laurencia_rug
plot(y12~x3, pch=1, xlab="Rugosity", ylim = c(0,6), ylab=expression( italic ("Parechinus angulosus")~~"Count"))
mod3 <- lm(y12~x3, data = SP)
summary(mod3)
abline(mod3)
cor.test(x3,y12)
p3 <- round(summary(mod3)$coefficients[2,4], 3)
p3
r3<-round(cor(x3,y12),3)
r2c <- round(summary(mod3)$r.squared, 3)
eqn3 <- bquote(atop(paste(r^2 == .(r2c)
                    * "," ~~ cor == .(r3)
                    * ","),paste(p == .(p3))))
text(0.25, 5.5, eqn3, pos = 4,cex = 1)

par(xpd=TRUE)
text(0.02, 6.45, "(4)",cex = 1)
par(xpd=FALSE)

windows()
#2 Pyura_MVR
plot(y9~x4, pch=1, xlab="Maximum Vertical Relief (m)", ylim = c(0,30), ylab=expression( italic ("Pyura stolonifera")~~"Cover (%)"))
mod3 <- lm(y9~x4, data = SP)
summary(mod4)
abline(mod4)
cor.test(x4,y9)
p3 <- round(summary(mod3)$coefficients[2,4], 3)
p3
r3<-round(cor(x4,y9),3)
r2c <- round(summary(mod3)$r.squared, 3)
eqn3 <- bquote(atop(paste(r^2 == .(r2c)
                    * "," ~~ cor == .(r3)
                    * ","),paste(p == .(p3))))
text(2, 28, eqn3, pos = 4,cex = 1)
```

```
#################################################################################
################

d  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/AAmp.csv", sep = ",", header = T)
d1  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/AAnn.csv", sep = ",", header = T)
d2  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/AAnt.csv", sep = ",", header = T)
d3  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/AAst.csv", sep = ",", header = T)
d4  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/ACol.csv", sep = ",", header = T)
d5  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/ACor.csv", sep = ",", header = T)
d6  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/ACr.csv", sep = ",", header = T)
d7  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/ADi.csv", sep = ",", header = T)
d8  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/AEnc.csv", sep = ",", header = T)
d9  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/AFol.csv", sep = ",", header = T)
d10  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/AGyn.csv", sep = ",", header = T)
d11 <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/AHal.csv", sep = ",", header = T)
d12  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/ALau.csv", sep = ",", header = T)
d13  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/AOp.csv", sep = ",", header = T)
d14  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/APar.csv", sep = ",", header = T)
d15  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/APloc.csv", sep = ",", header = T)
d16  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/APor.csv", sep = ",", header = T)
d17  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/APyu.csv", sep = ",", header = T)
d18  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/ATur.csv", sep = ",", header = T)
d19  <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/Rel/ATurf.csv", sep = ",", header = T)

cor.test(d$Hali_mid,d$Amph_eph)
cor.test(d1$Hali_mid,d1$Annelida)
cor.test(d2$Hali_mid,d2$Anthozoa)
cor.test(d3$Hali_mid,d3$Asteridea)
cor.test(d4$Hali_mid,d4$Col_Inverts)
cor.test(d5$Hali_mid,d5$Cor_Tur)
```

```
cor.test(d6$Hali_mid,d6$Crustacean)
cor.test(d7$Hali_mid, d7$Dinoplax_gigas)
cor.test(d8$Hali_mid,d8$Encrust_Alg)
cor.test(d9$Hali_mid,d9$Foliose)
cor.test(d10$Hali_mid,d10$Gyn_plac)
cor.test(d11$Hali_mid,d11$Halimeda)
cor.test(d12$Hali_mid,d12$Laur_flex)
cor.test(d13$Hali_mid, d13$Opisthobranchia)
cor.test(d14$Hali_mid, d14$Par_ang)
cor.test(d15$Hali_mid, d15$Ploc_cor)
cor.test(d16$Hali_mid,d16$Prosobranchia)
cor.test(d17$Hali_mid,d17$Pyura_stolon)
cor.test(d18$Hali_mid,d18$Turbo_sar)
cor.test(d19$Hali_mid,d19$Turf)

#DCA and Cluster analysis
### Load required library
library (vegan)
library (rrcov)
library (MASS)

### Import Data
sw <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/DCA/SW.csv", sep = ",", row.names = 1, header = T, fill = T)
csw <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/DCA/CSW.csv", sep = ",", row.names = 1, header = T, fill = T)
swSt <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/DCA/SWGsite.csv", sep = ",", header = T, fill = T)
swSp <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/DCA/SWGSp.csv", sep = ",", header = T, fill = T)

In <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/DCA/Inv.csv", sep = ",", row.names = 1, header = T, fill = T)
cin <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/DCA/CInv.csv", sep = ",", row.names = 1, header = T, fill = T)
InSt <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/DCA/InvGsite.csv", sep = ",", header = T, fill = T)
InSp <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/DCA/InvGSp.csv", sep = ",", header = T, fill = T)

Env <- read.table("C:/Users/Andrew/Documents/Masters/Analysis/DCA_GLM/MENV.csv", sep = ",", row.names = 1, header = T, fill = T)

### Create hcplot function to draw dendrograms
"hcoplot" <- function(tree, diss, k,
                 title=paste(deparse(tree$call), sep="\n"))
{
  require(gclus)
  gr <- cutree(tree, k=k)
  tor <- reorder.hclust(tree, diss)
```

```
  plot(tor, hang= -1, xlab="", sub = "", main = "Ward Classifications", ylab = "Distance")
  so <- gr[tor$order]
  gro <- numeric(k)
  for (i in 1:k)
  {
    gro[i] <- so[1]
    if (i<k) so <- so[so!=gro[i]]
  }
  rect.hclust(tor, k=k, border=gro+1, cluster=gr)
  par(xpd=T)
  legend("topright", paste("Group",1:k), pch=22, col=2:(k+1), bty="n", cex = 0.7)
}

## Calculate distance matrices
dist.mat<- vegdist(sw) ###Methods = "manhattan", "euclidean", "canberra", "bray", "kulczynski", "jaccard", "gower", "morisita", "horn", "mountford", "raup" , "binomial" or "chao". Gower, Bray-Curtis, Jaccard and Kulczynski indices are good in detecting underlying ecological gradients (Faith et al. 1987). Morisita, Horn-Morisita, Binomial and Chao indices should be able to handle different sample sizes (Wolda 1981, Krebs 1999, Anderson & Millar 2004), and Mountford (1962) and Raup-Crick indices for presence-absence data should be able to handle unknown (and variable) sample sizes.
dist.mat1<- vegdist(csw)
## Agglomerate cluster
Clust.res <- hclust(dist.mat, method="ward.D") ###"ward.D", "ward.D2", "single", "complete", "average" (= UPGMA), "mcquitty" (= WPGMA), "median" (= WPGMC) or "centroid" (= UPGMC).
Clust.res1 <- hclust(dist.mat1 , method = "ward.D") ###"ward.D", "ward.D2", "single", "complete", "average" (= UPGMA), "mcquitty" (= WPGMA), "median" (= WPGMC) or "centroid" (= UPGMC).

##Plot Clust
plot(Clust.res)
plot(Clust.res1)

###Draw Dendrograms
windows(35,20)
hcoplot(Clust.res, dist.mat, k=4)
windows(35,20)
hcoplot(Clust.res1, dist.mat1, k=3)

###Cutrees
grp <- cutree(Clust.res,4)
grp1 <- cutree(Clust.res1,3)

###Seaweeds

# Detrended Correspondence Analysis (DCA)
# ****************************************************************
dca <- decorana(sw)
dca
```

```
dca1<- decorana(csw)
dca1
# To fit environmental vectors
ef <- envfit(dca,Env, permu = 999)
ef

###Sites
###################################################################
# A Note
## Use the dataframe below and in excel change Numbers to Letters
##and save as csv, which you later import as a dataframe!
###################################################################
###Create a data frame with groups from cluster
df<-as.data.frame(grp)
###Rename Coloum
names(df)[names(df)=="grp"] <- "Clust.Grp"
df$Sites<-rownames(df)
df

###Draw DCA
windows()
spe.plt <- plot(dca, display="site", type = "n")
points(dca, display="site", pch=as.numeric(swSt$Grp),
      col=as.numeric(swSt$Grp))
groupz <- sort(unique(grp))
for(i in seq(groupz)) {
  ordiellipse(dca, grp, kind="sd", conf=0.95, label=T,
            font=2, cex=1, col=i, show.groups=groupz[i])
}
plot(ef, p.max = 0.05, col="blue") #the higher you make p.max, the more vectors will show
identify(spe.plt,'sites', cex=0.7)

###Species
###Create a data frame with groups from cluster
df1<-as.data.frame(grp1)
###Rename Coloum
names(df1)[names(df)=="grp"] <- "Clust.Grp"
df1$Sp<-rownames(df)
df1

###Draw DCA
windows()
spe.plt <- plot(dca, type = "n")
points(dca, display="species", pch=as.numeric(swSp$Group),
      col=as.numeric(swSp$Group))
groupz <- sort(unique(grp1))
for(i in seq(groupz)) {
  ordiellipse(dca1, grp1, kind="sd", conf=0.95, label=T,
            font=2, cex=1, col=i, show.groups=groupz[i])
```

```
}
plot(ef, p.max = 0.05, col="blue") #the higher you make p.max, the more vectors will show
identify(spe.plt,'species', cex=0.7)

###Inverts

# Detrended Correspondence Analysis (DCA)
# ***************************************************************
dca <- decorana(In)
dca

anova(dca)

dca1<- decorana(cin)
dca1
# To fit environmental vectors
ef <- envfit(dca,Env, permu = 999)
ef

###Sites
###Create a data frame with groups from cluster
df<-as.data.frame(grp)
###Rename Coloum
names(df)[names(df)=="grp"] <- "Clust.Grp"
df$Sites<-rownames(df)
df
###Draw DCA
windows()
spe.plt <- plot(dca, type = "n", ylim = c(-2,2), xlim = c(-2,2))
points(dca, display="site", pch=as.numeric(InSt$Group),
      col=as.numeric(InSt$Group))
groupz <- sort(unique(grp))
for(i in seq(groupz)) {
  ordiellipse(dca, grp, kind="sd", conf=0.95, label=T,
            font=2, cex=1, col=i, show.groups=groupz[i])
}
plot(ef, p.max = 0.05, col="blue", cex=0.7) #the higher you make p.max, the more vectors will show
identify(spe.plt,'sites', cex=0.7)

####Labels (automatic)
ordilabel(dca, display="site")
###Species
###Create a data frame with groups from cluster
df1<-as.data.frame(grp1)
###Rename Coloum
names(df)[names(df)=="grp"] <- "Clust.Grp"
df1$Sp<-rownames(df)
df1
```

```
###Draw DCA
windows()
spe.plt <- plot(dca, type = "n")
points(dca, display="species", pch=as.numeric(InSp$grp1),
       col=as.numeric(InSp$grp1))
groupz <- sort(unique(grp1))
for(i in seq(groupz)) {
  ordiellipse(dca1, grp1, kind="sd", conf=0.95, label=T,
              font=2, cex=1, col=i, show.groups=groupz[i])
}
plot(ef, p.max = 0.05, col="blue") #the higher you make p.max, the more vectors will show
identify(spe.plt,'species', cex=0.7)
```

Appendix D: Framer Unit, Cameras and Costs; a comparative analysis

A framer unit was constructed for the Sony camera as well as for the Go-Pro hero 3 (GP) unit. The Sony Unit was constructed of stainless steel tube 15mm in diameter (Plate 6). The base was a square quadrat of 0.025 m^2, connected to a mark III camera handset and with a maximum height of 650mm. The GP unit was constructed from 20mm PVC electrical piping. Both framer units were also equipped with a UWATEC Dive timer attached with an electrical PVC inspection connection joined to an arm of the framer unit. The unit had a maximum height of 6...mm and a small GP shoe held the GP parallel to the substrate. Both units were weighted with 3 to 5 kilograms of lead weight. Due to the upgrading of the GP, prices were unavailable for the GP3 so pricing for the GP4 unit were used. The main differences in these units are the upgraded specifications. Available units were compared by total cost, megapixel rating, and focal length.

Plate 6: Sony Camera framer unit

The Sony Mark III camera rig overall had the best specifications in terms of the camera megapixels, depths rating (but only if in the housing) and focal length. A significant difference ($p<0.05$) in price was evident for the GP and Sony (Figure 55). The mean price for the Sony including the housing unit was ZAR 44 891.67 (± 339.06 ZAR SD) while the GP costs on average ZAR 9443 (± 586.58 ZAR SD). The GP is significantly smaller ($p=0.03$) in dimension than the Sony (Table 19) and has only half the megapixel rating of the Sony. The Sony unit is rated to a depth of 100m while the GP is only rated to 40 m.

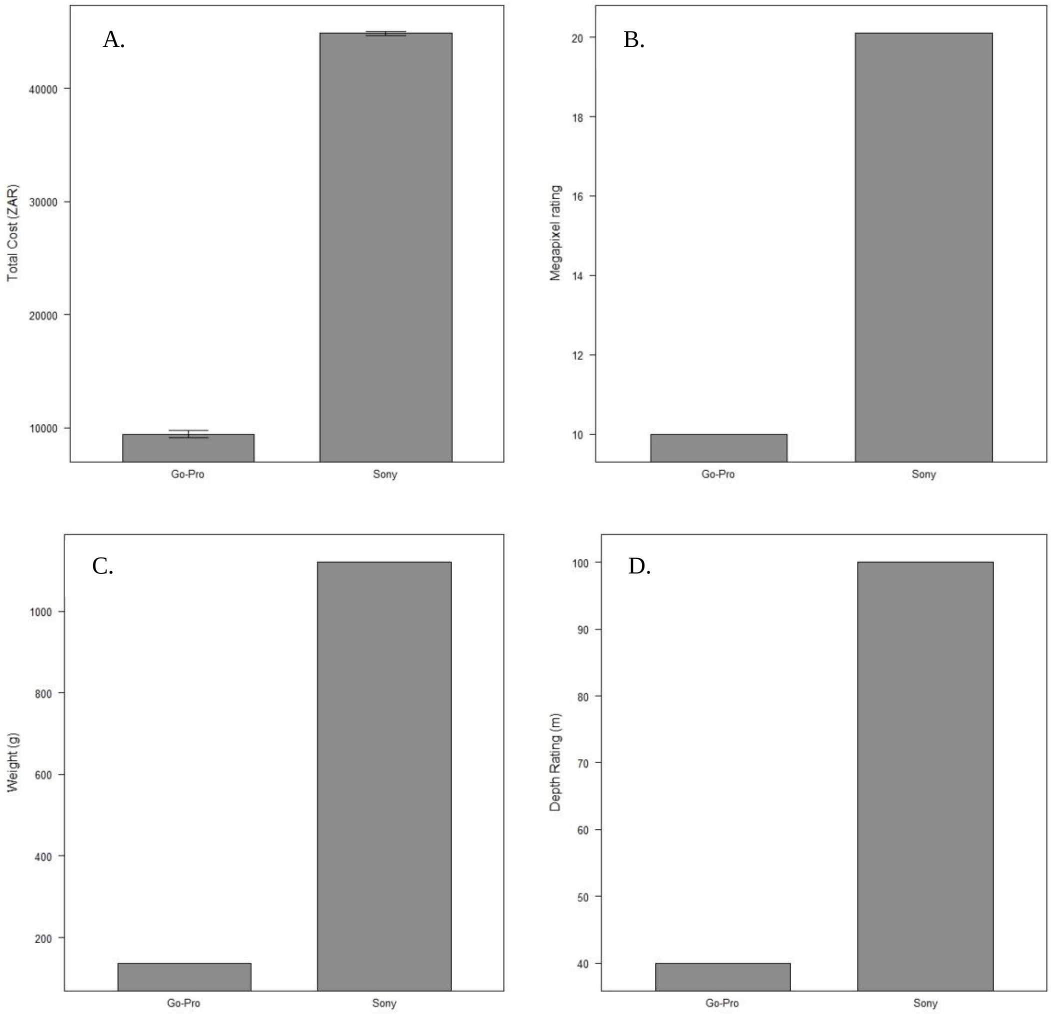

Figure 55: Camera Statistics and Specifications, (A.) Camera and Housing Prices (SE), (B.) Megapixel rating for perspective units, (C.) Weight per unit, (D.) The Depth rating for each unit.

The Sony Framer was constructed from stainless-steel hollow tubing, while the go-pro unit was constructed from PVC electrical piping. This influenced the cost of constructing the framer.

Table 19: Camera statistics and costs for the Sony Mark III and Gopro Hero 3 plus as according to the rand/dollar exchange rate of R13.91 to a USD.

Camera Type	**Average Price (ZAR)**	**Housing Price (Approx.)**	**Framer Price**	**Focal Distance (cm)**	**Megapixels**	**Weight (g)**	**Dimensions (cm)**			Depth Rate (m)
							Width	Height	Depth	
Sony	21000	24000	500	400	20.1	1120	15	10.1	10.7	100
Go-Pro	9440	700	300	500	10	137	6.5	6.7	3.5	40

There is a clear differentiation between the camera set-ups in the project. There is a trade-off between specifications and cost. As expressed by Brown et al. (2004) the use of a technique must be efficient enough to meet the requirements of the research objectives but also robust and cost effect.

Prince (2013) ran cost-benefit analysis of ranching in Australia and found for the success of such programs, it is of crucial importance that cheap seed sources and/or brood stock are used and where possible discount rates are obtained, which directly determines the cost of capital.

Recording Time

Average recording times for the transects showed a strong linear relationship, where time was directly proportional to length (r^2 = 0.98, t = 13. 36, df = 3, p <0.005)(Figure 56 Left). The CV values also showed little relative variation between distance and time (t = -0.231, df = 1.8118, p = 0.8). The 30 metre length transects showed the highest CV values of 0.56 while the lowest cv value was 0.41 for the 20 m length (Figure 56 Right).

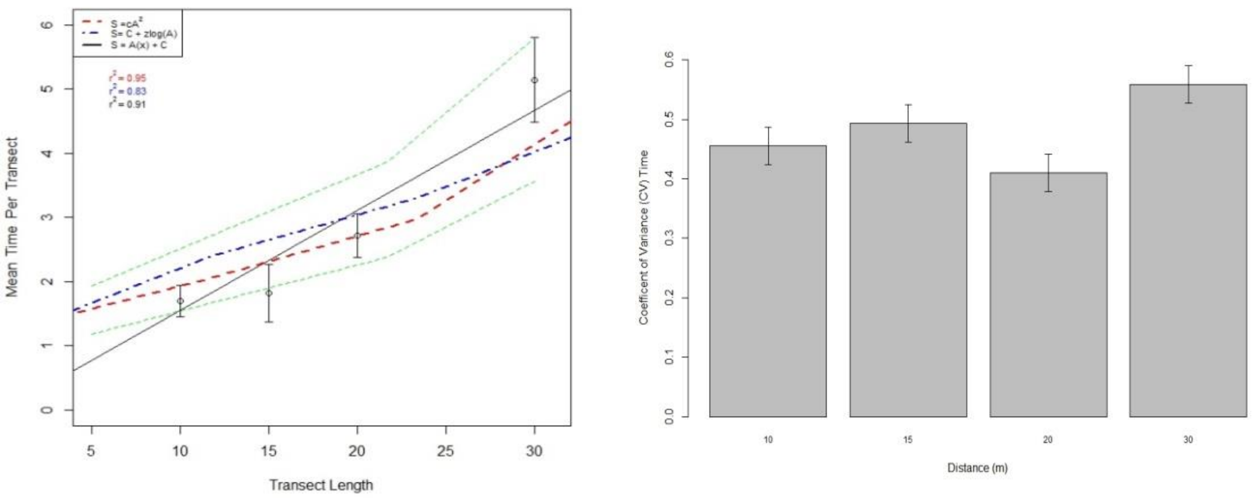

Figure 56: (Left) Regression of recording time (SE) compared to transect length (m) with 95% confidence intervals, (Right) Coefficient of Variance for times (SE) compared to distance.