



Mapping and assessing ecosystem threat status of South African kelp forests

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Declaration

I hereby declare that “Mapping and assessing ecosystem threat status of South African kelp forests” is my own work and that all the sources I have used or quoted have been indicated and acknowledged by means of complete reference.

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Signature

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Abstract

At a global scale, kelp forests play a significant ecological, social and economic role through their provision of ecosystem services. South African kelp forests are no exception and recent studies have established their value. To maintain these benefits, informed management is needed. An understanding of kelp forest distribution, ecosystem functioning, pressures, and ecosystem state are key requirements for effective management. The South African National Biodiversity Assessment (NBA) synthesises research to report on the status of ecosystems to guide policy, planning and decision making. Kelp forests were excluded from two previous national assessments, as they were not represented on the National Map of Marine Ecosystem types. This thesis aimed to address this omission by producing a map of kelp forest ecosystem types and conducting the first assessment of their threat status.

This study sought to develop a modern method for mapping South African kelp forests to update previous maps developed in the mid-2000s. The novel approach extracts the Vegetation Index from kelp forests using advanced multi-resolution Sentinel-2 (A and B) satellite imagery. Using Geographic Information Systems, spectral bands 4 (RED) and 8 (NIR) (10 m resolution) were utilized to calculate Normalised Difference Vegetation Index (NDVI). An expert-guided trial and error approach was adopted to set the NDVI threshold (-0.2) at a level suitable for detecting both subtidal and surface buoyant kelp to the limits of the Sentinel-2 platform. The results showed that the high resolution and deeper water column penetration of this platform enabled the filling of previous gaps and detect both subtidal and surface protruding kelps at low cost. Additionally, the map includes for the first time, the kelp recently reported to have shifted eastwards along the south coast. A total of 1300 km of kelp forest was mapped and three biogeographical subtypes distinguished. Combining the NDVI threshold method and Supervised Classification yielded satisfactory results and an accuracy of 76%. Sentinel-2 imagery was validated using observational classification from Google Earth, field surveys, expert knowledge and previous maps. However, the Sentinel's depth penetration was affected by environmental heterogeneity along this coast. Results confirmed the complexities of retrieving spectral indices from environments with varying turbidity, depths, wave climates and the challenges associated with ground-truthing the expansive marine environment. This study advises comprehensive ground-truthing for the three kelp forest ecosystem types as a fundamental step towards long-term monitoring of South African kelp forests. The method developed advanced application of the NDVI in submerged aquatic vegetation mapping and

could be modified to support mapping of other ecosystem types such as seagrasses, other seaweed habitats and inland aquatic vegetation.

South Africa's new kelp forest map was then used to facilitate the first ecosystem threat status assessment for South African kelp forests using three criteria from the emerging International Union for Conservation of Nature Red List of Ecosystems (IUCN RLE). To assess threat status, ecosystem extent and condition were considered. Three kelp forest ecosystem types were assessed; namely Namaqua, Cape and Agulhas Kelp Forests and an additional combined single South African Kelp Forest to explore the effect of scale in assessing threat status. Literature was reviewed to develop a conceptual model to support the assessment and define ecosystem collapse. Thirteen relevant pressures were mapped to determine ecosystem degradation across the extent of each kelp forest type using a cumulative pressure mapping approach. Four categories of ecosystem condition were recognized in alignment with the IUCN thresholds for ecosystem degradation. The results of the ecosystem threat status assessment show sensitivity to the different assessment criteria, the scale of ecosystem delineation and assessment approaches. There is no reported reduction in the distribution for any of the South African kelp forest ecosystem types, therefore, the decline in extent under criterion (A) was assessed as Least Concern for all types. For criterion (B) which is related to geographic extent and threat, results were most sensitive to ecosystem delineation with results ranging between Least Concern and Critically Endangered under different sub-criteria and for different ecosystem types. Also, for the criterion (C) which is related to the extent of abiotic degradation, the results ranged from Vulnerable to Endangered under different sub-criteria and for different ecosystem types. Further work is needed to validate kelp forest ecosystem types; consider the implications of multiple scales of classification, mapping and assessment; improve pressure data, ground-truth ecosystem condition, and assess the disruption of biotic processes. In line with the protocols of the IUCN RLE, South African kelp forest ecosystem types appear threatened with plausible results ranging between Vulnerable and Endangered. The accuracy of these assessments can be strengthened by more research to refine conceptual models, calibrate assessments of degradation and better define thresholds of collapsed ecosystems.

Chapter 1: Literature Review

“I can only compare these great aquatic forests... with the terrestrial ones in the intertropical regions. Yet if in any country a forest was destroyed, I do not believe nearly so many species of animals would perish as would here, from the destruction of kelp...” Charles Darwin (cited in Steneck et al., 2002).

1.1. Introduction

There are four species of kelp in South Africa, of which two species (*Ecklonia maxima* and *Laminaria pallida*) are dominant in the large kelp forests found along the west coast of South Africa (Rothman et al., 2015). In the past, there has been a substantial focus on South African kelp forest ecology (Field et al., 1977; Velimirov et al., 1977; Velimirov and Griffiths, 1979; Field et al., 1980; Newell, Field and Griffiths 1982; Day and Branch, 2000; Blamey et al., 2010; Blamey and Branch, 2012; Blamey et al., 2012; Dyer et al., 2019 and see: review by Branch, 2008), biogeography (Bolton, 1986; 2010; Bolton and Stegenga, 2002; Anderson et al., 2007; Rothman et al., 2015; Rothman et al., 2017) and management of kelp as a resource (Levitt et al., 2002; Anderson et al., 2006; 2007; Rothman et al., 2010; Elston, Anderson and Price, 2015). Kelp forests provide important ecosystem services that can be grouped into three categories namely economic, ecological and social benefits. The economic services emanate from commercial fisheries, kelp derived products for use in aquaculture, medicine and subsistence and the social benefits include coastal protection and opportunities for tourism (Anderson et al., 2007; Cavanaugh et al., 2011; Smale et al., 2013; Blamey et al., 2013; Borrás-Chavez et al., 2016; Blamey and Bolton, 2018). The economic and ecological benefits of kelp forests are related to their role as one of the most prolific primary producers on the planet, supporting productivity per unit area that resembles that of tropical rainforests (Steneck et al., 2002; Krumhansl et al., 2016, Teagle et al., 2017). Kelp forest ecosystems provide nursery grounds, shelter and food source to many other species (Mann and Breen, 1972; Bustamante, Branch and Eekhout, 1995; Edyvane, 2003; Anderson et al., 2006). Their ecological services are not only crucial in the shallow subtidal area but also the adjacent deep-sea and beach ecosystems which are subsidised by energy provided through detritus, drift and debris (Field et al., 1977; Bustamante and Branch, 1996; Day and Branch, 2000; Branch and Branch, 2018; Filbee-Dexter and Wernberg, 2018; Dyer et al., 2019). Globally, kelp forests are instrumental in climate regulation, carbon sequestration and serve as a protective coastal barrier to storms and harsh wave action that increase sedimentation and intensify impacts from coastal squeeze

(Smale et al., 2016, Pfister et al., 2018, Wernberg and Filbee-Dexter, 2018; Hoegh-Guldberg et al., 2019). Kelp forests also play an ecosystem engineering role, as the wrack that accumulates on the drift line and supralittoral zones of beaches serve as a sand trap, which can form the basis for new foredunes to develop (Nordstrom et al., 2011) and thereby contribute further to coastal protection. Beach-cast wrack with kelp underpins the food webs of sandy beaches (Gómez et al., 2018; Rodil et al., 2018).

Despite the sound knowledge of South African kelp forest diversity and ecology, there has been little focus on mapping their distribution or understanding their ecological condition. Anderson et al. (2007) provided the first large-scale map of South African kelp forests, however, it did not cover the full extent of the South African coastline and included several gaps where kelp was known to occur. South African efforts in mapping and monitoring the distribution and extent of kelp forests lags international initiatives in this regard. Consequently, there is limited knowledge when it comes to mapping the distribution and condition trends of South African kelps (Bolton et al., 2004; Bolton, 2010), which is becoming increasingly important under global change including changing climate scenarios. Since the mid-2000s, there has not been any further development and application of high spatial and spectral remote sensing techniques to re-map South African kelp forests. As such, these kelp forests have been omitted from two National Biodiversity Assessments, due to the requirement of a comprehensive national map to inform any such assessments. As a result of the lack of the kelp forests inclusion in the national marine ecosystems map, their ecosystem condition and threat status has never been assessed.

This chapter synthesizes international and local literature on kelp forest ecosystems, the services they provide and the processes that structure and threaten them. Here, methods of mapping, specifically the concepts governing remote sensing and monitoring for coastal ecosystems and the challenges associated with remote sensing of the marine environment are introduced. Also, the concepts that govern ecosystem threat status both at international (IUCN RLE) and local (NBA) scale are explained.

1.1.1. Kelp forests

Kelp was originally used to refer to the calcined ashes that resulted from burning large brown algae (Schiel and Foster, 2015). Although it is quite common to refer to any large brown algae from different orders as “kelp”, this often results in confusion around the use of this term (Bolton, 2010). Here onwards, “kelp” will be used to refer to the Laminariales. Most kelps have a basic structure comprising a holdfast, a flexible stipe and fronds (Steneck et al., 2002). It is this basic structure that contributes to kelps being ideal habitats as each of these three structures provide distinct functional habitat types from individual kelp sporophytes (Teagle et al., 2017).

1.1.2. Kelp as a habitat

Wave motion has an enormous influence on the morphology of the holdfast, stipe and fronds. In response, kelps display varying adaptations that enable them to best cope with the various water motions in their environments (Wernberg et al., 2018). The holdfast is composed of interwoven heptera anchoring the sporophyte onto the substratum (Elston, Anderson and Price, 2015) providing a complex temporal habitat that is dominated by other seaweeds, kelp recruits as well as numerous invertebrates including mobile copepods, polychaetes, gastropods, amphipods and sessile fauna such as bryozoans, bivalves and sponges (Field et al., 1977; Steneck et al., 2002; Edyvane, 2003; Kelly, 2005; Burrows et al., 2014; Teagle et al., 2017). Of the 770 animal species that live in California kelp forests, more than 150 reside in the holdfast (Edyvane, 2003). In terms of richness and species abundance, the greatest influence results from the structure and complexity of the holdfast habitat along with external and regional factors (Burrows et al., 2014).

The stipe is relatively simple in structure and supports a diverse array of fauna and flora (Anderson et al., 2006). The stipe is a primary harbour for secondary habitat-forming species like epiphytes (Anderson et al., 2006; Burrows et al., 2014) and some fauna e.g. the limpet *Cymbula compressa* (Anderson et al., 2006; Hereward et al., 2018). Among the kelp species, stipes come in all shapes and sizes including length, rigidity, plasticity, whether they are solid or hollow, tensile strength and whether they branch (Rothman et al., 2015; Teagle et al., 2017).

The blades or fronds serve as the main photosynthetic organs and can be colonized by a range of epibionts, impacting the settlement and growth of the associated biodiversity i.e. epifauna and epiflora (Anderson et al., 2006; Teagle et al., 2017). Fronds also come in all sorts of shapes and sizes including variable thickness, rigidity, surface texture, edge formation and several

divisions. For example, some species have fronds categorized into primary blades and/or secondary lateral blades that grow from the basal meristem (Rothman et al., 2017).

Although kelp species are characterized by low taxonomic diversity, they are renowned for their functional and morphological diversity (Steneck et al., 2002; Kelly, 2005). Their physiology and combination with associated organisms play a key role in altering the state of the surrounding and adjacent environments (Steneck et al., 2002; O'Connor and Anderson, 2010; Teagle et al., 2017) earning them the term, ecosystem engineers (Steneck and Johnson, 2013; Teagle et al., 2017; Miller et al., 2018). For example, kelp forests control sediment dynamics and water flow by reducing turbulence beneath the canopy. Kelp canopies dampen the wave force, by absorbing some of the wave energy, reducing its velocity and slowing coastal erosion thus protecting the shorelines and promoting benthic productivity (Kelly, 2005; Krumhansl et al., 2016; Witman and Lamb, 2018).

1.1.3. Biogeography of kelps

Biogeography is the study of life both in space and time and its patterns of distribution (Cox and Moore, 1998; Sink, Branch and Harris, 2005). Kelp forest biogeographic patterns are driven by seascape and other environmental characteristics (Johansson et al., 2015). Typically, large brown algae have distribution patterns that coincide with the large-scale biogeographic patterns and their distribution is restricted by temperature limits along with other factors (Johansson et al., 2015; Hargrave et al., 2017). For example, the Laminariales have a geographical distribution concentrated around the equatorial range 40 S - 60 N in both hemispheres (Hargrave et al., 2017). The intermediary point of kelp diversity is in the North Pacific (Bolton, 2010) and Arctic waters of the Northern Hemisphere compared to the Southern Hemisphere, with kelps completely absent in Antarctica (Steneck and Johnson, 2013).

In temperate regions, kelp forests occur from the highest low watermark, down to depths > 40 m (Steneck et al., 2002; Bennett et al., 2016). In some regions, they occur much deeper (Bennett et al., 2016) carpeting volcanic seamounts e.g. *Laminaria rodriguezii* found at 100 m depths in the Mediterranean. In low latitudes (< 40°) the occurrence of kelp forests often coincides with deep and clear waters where light penetrates the cooler deep water. They are typically associated with the productive regions with upwelling where dense, cooler and nutrients abundant water is pushed to the surface, this replaces water that is warmer and deficient in

nutrients (Bolton and Stegenga, 2002; Steneck and Johnson, 2013; Pérez-Matus et al., 2017; Rothman et al., 2017; Dyer et al., 2019).

Principally, kelp forests are well-known for their affinity for cold, nutrient-rich water (Filbee-Dexter and Wernberg, 2018). They are prominent in upwelling regions where upwelled nutrients play a significant role in the trophic dynamics, interactions and influences the composition (i.e. benthic algal biomass and community) and productivity of kelp forest ecosystems (Pérez-Matus et al., 2017). All major ecological variables acting on coastal communities i.e. light, nutrients and temperature are subjected to the influence of upwelling (Bolton and Levitt, 1987). Along the southwest coast of Africa, southeast winds subject the region to seasonal (late spring to late summer) upwelling, creating one of the most productive marine ecosystems in the world (Bolton and Levitt, 1987; Anderson et al., 2007; Watermeyer et al., 2016). One effect of upwelling is seen in the variable seaweed environment, with distinct seaweed flora between the west, south and east coasts of southern African (Bolton and Levitt, 1987). Opposite northerly winds trigger a reverse phenomenon to upwelling, termed downwelling, which pushes warm offshore surface waters back inshore that are rich in phytoplankton and particulate matter due to initially being upwelled and pushed offshore by the south-easterly wind forces (Dyer et al., 2019).

Several authors report some kelp species that are thriving in warm temperate regions (Edyvane, 2003; Graham et al., 2007; Bolton, 2010; Steneck and Johnson, 2013; Rothman et al., 2015; Schiel and Foster, 2015; Krumhansl et al., 2016; Pereira et al., 2017). Edyvane (2003) explains that some species dominate shallow clear waters, whereas others have evolved to tolerate low-light e.g. *Laminaria hyperborea* off the coast of Spain and *L. pallida* northwestern region of South Africa and in Namibia, both of which thrive in subtidal and inshore regions with turbid water (Casal et al., 2011; Rothman et al., 2017).

For development, kelp recruits need substrate, high nutrient concentration, water motion and often cool, clear and clean water (Edyvane, 2003; Keith, 2013; Schiel and Foster, 2015). Seaweeds growth, distribution and survival also rely on the physical environment i.e. substratum type, sedimentation rate, irradiance, current flows, wave exposure, temperature, nutrient concentration, grazers, competition, disease and pollution (Bolton and Levitt, 1987; Adey and Steneck, 2001; Rothman et al., 2015; Johansson et al., 2015). Any alteration of these factors may have impacts on kelp forests ecosystems. Generally, an intrinsic and complex balance between recruitment, growth, competition and grazing governs kelp forest ecosystems.

Ecosystem mechanisms that are important for maintaining the resilience of kelp forests can be degraded by the effects of abiotic environmental factors e.g. climate change (increased storminess, extreme temperatures for prolonged periods) and biotic interactions like predations, herbivory and epiphytism (Filbee-Dexter and Wernberg, 2018). Kelp forests that are subjected to low nutrients and elevated temperatures have a high rate of erosion compared to growth (Steneck et al., 2013). The abundance of kelp forests at a local scale can be influenced by sea urchin grazing, storm frequency, pollution, invasive species and indirect top-down effects (Steneck et al., 2002; Witman and Lamb, 2018). However, many other factors influence the abundance of kelp forests and often their effects cannot be easily isolated e.g. sea surface temperature (SST), salinity and nutrients, large-scale climate changes, herbivory and microbial interactions (Pfister et al., 2018).

1.1.4. South African kelp forests

The biogeography of the South African coast has been resolved by the pioneering work of Stephenson (cited in Sink, Branch and Harris, 2005), who categorised the intertidal into three sections; the cool-temperate West Coast, the warm-temperate South Coast and the subtropical East Coast. Following this, three marine provinces were recognised and named (1) the Cool-temperate Namaqua, (2) Warm-temperate South Coast and, (3) the Subtropical East Coast were documented by Sink, Branch and Harris (2005). Further, the South African intertidal zone of the West Coast of southern Africa has been broken down into, (1) the Namaqua, (2) the Namib and (3) the Cape West Coast Sub-province, based on seaweed flora (Bolton, 1986; 2010).

The seaweeds of South African are exposed to two major upwelling systems that affect their environment (Bolton and Levitt, 1987). The Benguela Current (described above) flows along the coast of the Western Cape and Namibia. The Benguela is comprised of water masses with several components which upwell to the surface off the West Coast and has a significant influence on the biodiversity of this region (Bolton and Stegenga, 2002). Along the East Coast is the warm Agulhas Current which brings warm water towards the South Coast along the edge of the continental shelf and mixes surface water into the Benguela System. The Agulhas Bank forces this current to turn south into the Southern Hemisphere which experiences a mixing between these two major currents into the offshore part of the Benguela (Bolton and Stegenga, 2002). The dynamic of these two major currents are reflected in the seaweed communities of the West, South and the East Coast (Bolton, 1986; Bolton and Stegenga, 2002).

Field et al. (1977) described the dynamics of the Cape West Coast sub-province kelp forest community. Their work emphasised the important interaction between the primary producers (macroalgae) and environmental factors, but also the interaction with the biotic community. Day and Branch (2000) investigated the positive relationship between two key kelp forest grazer species of the Cape West Coast sub-province, the juvenile abalone *Haliotis midae* and the sea urchin *Parechinus angulosus*. Their study concurs with the work done by Tarr, Williams and Mackenzie, (1996), who first highlighted the association between juvenile abalone and urchins and suggested a link between the disappearance of juvenile abalone in kelp forest ecosystems of the South Coast and an influx of west coast rock lobster *Jasus lalandii*, which preys on the urchins in this region. These establishment periods of *J. lalandii* coincide with sharp declines in *P. angulosus* densities and subsequent declines in densities of juvenile *H. midae*, which benefit from shelter and nutrition from sea urchins (Tarr, Williams and Mackenzie, 1996; Mayfield and Branch, 2000). The shift in *J. lalandii* and resultant species interactions are associated with changes in kelp forest communities (Blamey et al., 2010; Blamey and Branch, 2012). Interestingly, the southwest coast establishment of rock lobster also corresponds to a slight increase in one of its predators, the bank cormorant in this region (Crawford et al., 2008; Pfaff et al., 2019). Another reported shift in this region is the eastward shift by kelp *Ecklonia maxima* (Bolton et al., 2012) (detailed below) and increased densities in parts of False Bay (Griffiths et al., 2010; Blamey et al., 2015). Although the reasons for the shift are not fully understood, Mead et al. (2013) suggest environmental changes as the root cause. Blamey et al. (2014) argue that if kelp forest ecosystem top predators had still been abundant e.g. reef-fish, they may have regulated the abundance of rock lobster and thus, prevented the urchin and abalone population collapse.

South African kelp forests are comprised of four species (Bolton and Levitt, 1987; Anderson et al., 2007; Steneck and Johnson, 2013; Rothman et al., 2015). The dominant *E. maxima* has a sporophyte generation that reaches 15 m in length (Bolton et al., 2012), some anecdotal data account for lengths reaching 17 m. *Ecklonia maxima* commonly termed “sea bamboo” extends southwards from just north Lüderitz, Namibia (26°38’31.34” S, 15°09’50.07” E) past the major biogeographical break on the southernmost tip of Africa Cape Agulhas (34°49’17.59” S, 20°00’ 45.85” E) to the recent location described above. Bolton et al. (2012) suggest that temperatures > 22.5° C are a limiting factor on the distribution of this species. *Ecklonia maxima* dominate shallower inshore water, forming extensive floating kelp forests to depths 5-10 m on the west and south-west coasts. *Ecklonia maxima* is characterized by a gas-filled bulb which

suspends its fronds at the surface and makes it conspicuous during low tide (Rothman et al., 2010; Bolton, 2010). This species has a distribution range that is characterized by mean inshore water temperatures ranging from 12-16° C, with the mean warmest months being 13-19° C and the coldest 11-14° C (Bolton et al., 2012).

Laminaria pallida, commonly known as “split-fan kelp”, extends southwards from Rocky Point (18°59'37.73"S, 12°28'30.23"E) in northern Namibia throughout the west coast of South Africa and beyond Cape Point to Danger Point (34°37'45.16"S, 19°17'33.59"E) along the southwest coast of South Africa (Rothman et al., 2017). In the southern part of its distribution, *L. pallida* is a subcanopy kelp to *E. maxima*. This subcanopy species is much shorter and replaces *E. maxima* at depths between 10-30 m (Elston, Anderson and Price, 2015; Rothman et al., 2017). *Laminaria pallida* has a strong seasonal growth pattern which is strongly correlated to radiation exposure (Bolton, 2010). Northwards of Cape Columbine, *L. pallida* develop a hollow stipe and dominates the shallow regions progressively replacing *E. maxima* (Anderson et al., 2006; 2007; Elston, Anderson and Price, 2015; Rothman et al., 2017). Rothman et al. (2017) attribute this transformation to increasing turbidity northwards. Split fan kelp is quite restricted along the south coast, perhaps because of its dependence on cold water temperatures. Along with the cool temperature of the west coast, the nutrients brought about by upwelling favours its expansion along the Namibian coast (Rothman et al., 2017).

Ecklonia radiata commonly referred to as “spined kelp” has a much shorter sporophyte of < 1 m in length, a solid stipe, spiny and/or smooth fronds (Bolton et al., 2012). *Ecklonia radiata* is less abundant than the other two species in South Africa and occurs in patches inshore (on the sublittoral fringe) of the south and east coasts between Koppie Alleen (34°29'02.09" S, 20°32'36.64" E), De Hoop and eastwards to Port Edward (30°48'32.26" S, 29°52'54.71" E) and sighted further east beyond Port Edward at 40 m depths (Rothman et al., 2015). Its South African distribution range is influenced by the warm Agulhas current with 17-19° C mean temperatures, warmest month mean 19-21° C and coldest 13-17° C (Bolton and Anderson, 1987; Rothman et al., 2015). The optimum temperature for net photosynthesis of this species has been estimated to be 25° C and is observed to thrive in temperatures > 18-20° C (Wernberg and Vanderklift, 2010).

Molecular evidence reveals that *Macrocystis* is monospecific and displays diverse morphologies in the different regions occupied by this (Schiel and Foster, 2015; Rothman et al., 2017). *Macrocystis pyrifera*, renowned as “giant kelp”, forms vast and iconic forests around

the world and in the Southern Hemisphere, is predominantly found in south-eastern Australia, New Zealand, many Southern Ocean Islands and along temperate South America (Rothman et al., 2015; Schiel and Foster, 2015; Guiry and Guiry, 2018). In South Africa, *M. pyrifera* is commonly known as “bladder kelp” and is not abundant, only occurring in small patches in few localities i.e. occurring within the kelp beds of *E. maxima* kelp at Kommetjie and Oudekraal (34°08'25.10" S, 18°19'45.15" E and 33°59'14.88" S, 18°20'56.96" E respectively) on the Cape Peninsula and Jacobs Bay (32°58'27.56" S, 17°54'23.20" E) near Cape Columbine. Possibly the largest bed is found along the east coast of Robben Island (33°48'27.39 S, 18°22'16.43" E) (Anderson et al., 2007; Rothman et al., 2017).

In the warm-temperate South Coast province, the dominant kelp forest forming species *Ecklonia maxima* has undergone an eastward shift of about 73 km and established itself at Koppie Alleen, in De Hoop Nature Reserve (34°28'35.76" S, 20°33'27.11" E) (Bolton et al., 2012). Although not yet well understood, this eastward expansion is attributed to gradual cooling along the south coast (Bolton et al., 2012) and perhaps increases in conditions that favour the development of this species. Rigorous studies since the 1980s have been conducted between Cape Agulhas and De Hoop Nature Reserve and before 2006, there have been no reports of sightings of *E. maxima* in this area, except for *E. radiata* (Bolton et al., 2012).

The exploitation of kelp forests for human benefits has been occurring for a long time and they have been harvested for extraction of agar for gunpowder, pharmaceuticals, plant growth stimulants and feed for aquaculture (Anderson et al., 2006; Rothman et al., 2006). The South African seaweed industry is mainly based on the harvesting of the most abundant kelp forest forming species *E. maxima* (Anderson et al., 2006; Elston, Anderson and Price, 2015). The targeting of *L. pallida* also occurs but is more pronounced in Namibia (Omorieg et al., 2010). Two methods of kelp harvesting in South Africa are recognised, namely; “lethal” and “non-lethal” (see: Rothman et al., 2006; Anderson et al., 2006). Fresh kelp is harvested along the coastline between Port Nolloth to Cape Agulhas, with the greatest quantities being harvested mainly in the Western Cape driven by the demand for feed for local abalone farms (Levitt et al., 2002). Beach cast kelp (i.e. kelp that has been washed out ashore), is also harvested. This also required management as is suggested by Dugan and Hubbard (2010) this can affect beach ecosystems by removing important food sources to macrofauna and microbes and associated organisms like amphipods with knock-on effects for foraging seabirds.

Organisms associated with the South African kelp forest ecosystems play a crucial economic role and have been valued at *ca.* ZAR 5.8 billion year⁻¹ (Blamey and Bolton, 2018). These include the two dominant kelp species and many species significant to subsistence, commercial and recreational fisheries like abalone, rock lobster, octopus (*Octopus vulgaris*), red bait (*Pyura stolonifera*), mussels, limpets and the alikreukel or giant turban snail (Blamey and Bolton, 2018).

1.2. Mapping kelp forest ecosystems

“As you read these words, you are employing remote sensing” (Lillesand, Kiefer and Chipman, 2004).

Effective management of an exploited species requires an understanding of (1) the distribution and the biomass of the resource, (2) the biology and abiotic interactions and (3) the impacts of external factors on the resource and associated species e.g. harvesting impacts (Anderson et al., 2007). Global research on kelp has largely focused on the second and the third points (Guiry and Morrison, 2013; Krumhansl et al., 2017) and this is no different for South Africa (Levitt et al., 2002; Anderson et al., 2007; Bolton, 2010; Rothman et al., 2015; 2017). In South Africa, proper management of kelp forest resources is hampered by the lack of comprehensive and up-to-date kelp forest extent maps (Anderson et al., 2007), largely influenced by the inaccessibility of the marine realm (Bennion et al., 2018) and the vast extent of the coastline (Lück-Vogel and Mbolambi, 2018; Harris et al., 2019).

1.2.1. Remote Sensing and Geographic Information Systems (GIS): Electromagnetic radiation dynamics

The most familiar forms of electromagnetic radiation to us are visible light, radio waves, ultraviolet rays, X-rays and heat (Lillesand, Kiefer and Chipman, 2004). Electromagnetic radiation is a time-varying electric and magnetic field which differs in magnitude and direction. These electric (E) and magnetic (B) components are shown respectively in Figure 1. 1 in red and blue travelling at right angles and are perpendicular to their direction of propagation (Lillesand, Kiefer and Chipman, 2004). Electromagnetic energy is characterised by wavelengths (λ), Frequencies (ν) and travels at a constant speed of light (C) which is 3×10^8 m/s (Equation 1.1) (Lillesand, Kiefer and Chipman, 2004; Bhatta, 2011).

$$C = \nu \lambda \dots\dots\dots \text{Equation 1.1}$$

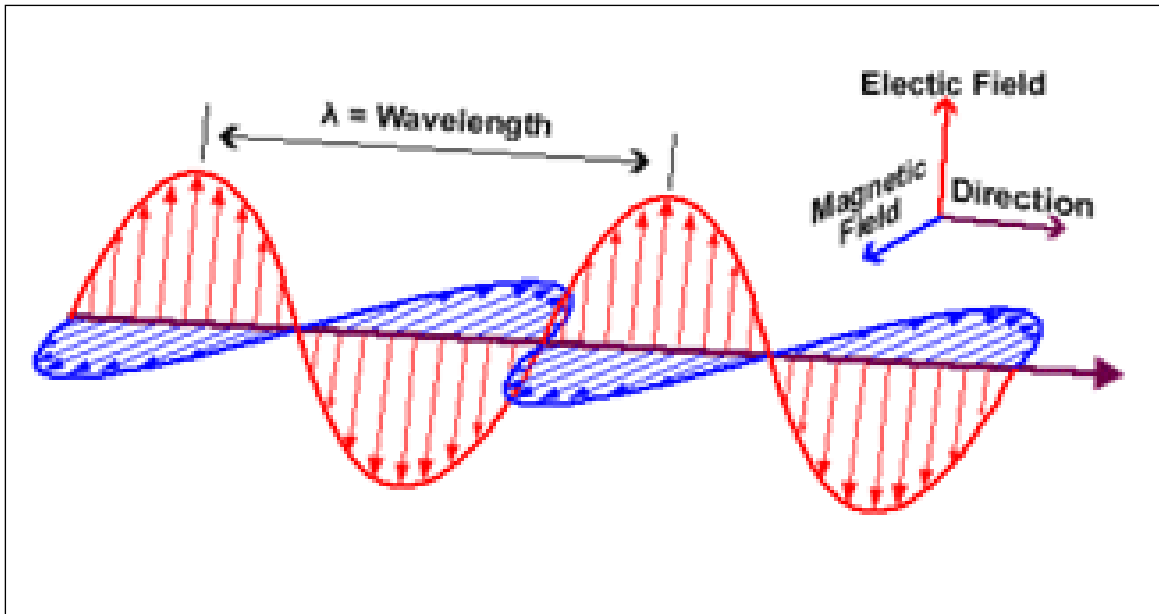


Figure 1. 1: Electric and magnetic vectors of an electromagnetic wave of a wavelength at a given instant (Bhatta, 2011).

When electromagnetic radiation encounters a surface, it interacts in one of three ways. In a variation of fractions, the energy is either reflected, absorbed and/or transmitted (Lillesand, Kiefer and Chipman, 2004). When interactive with vegetation, the presence or absence of leaf/frond pigments determines the response in the electromagnetic spectrum e.g. the visible region. The morphology of the leaf and the water content are prime factors in the signal's response in the infrared wavelength region of the spectrum (Silva et al., 2008).

The major difference in remote sensing of terrestrial and aquatic vegetation is the effect of the water column covering aquatic vegetation. Water interferes with the response of the electromagnetic spectrum as it strongly absorbs electromagnetic radiation in the optical spectral region. The rate of absorption increases with the wavelength, resulting in zero reflection in the near infra-red region (NIR). In the red spectral region, chlorophyll has an absorption maximum and the other absorption maximum in the blue spectral region of the visible part of the electromagnetic region. From the NIR (band 4) region, water reflection is less than 1%, the reason why it appears very dark in optical data whereas vegetation reflection is about 50% in the NIR (Figure 1. 2). This optical difference between water and vegetation is advantageous in making water distinguishable from macroalgae beds (Bhatta, 2011).

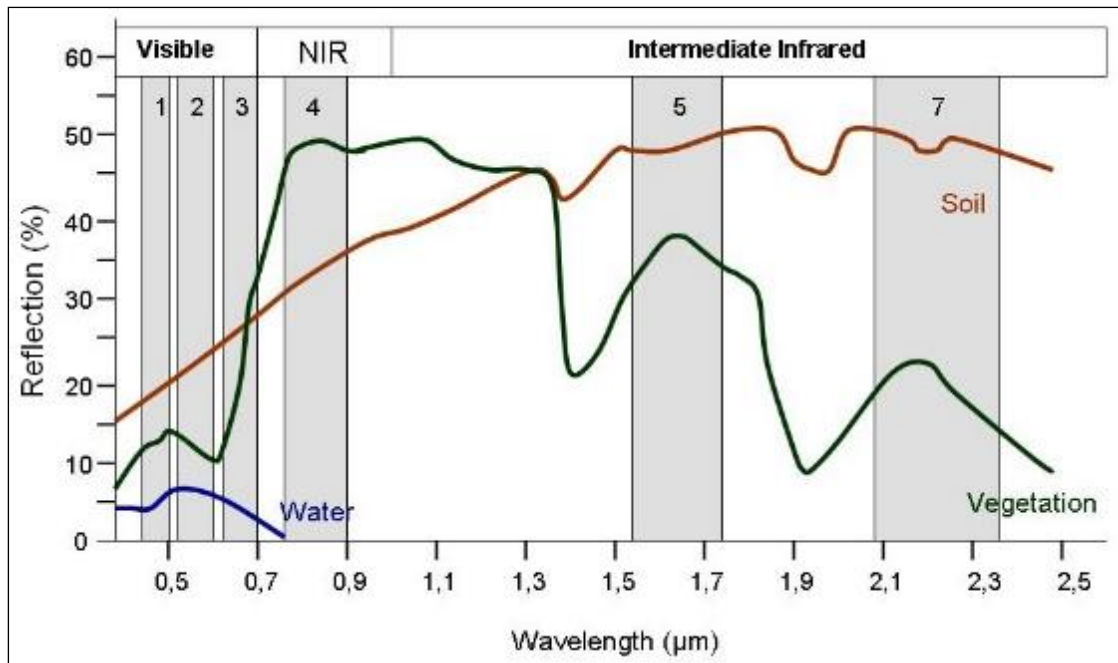


Figure 1. 2: Spectral signatures of soil, vegetation and water in the Visible (red, green and blue), NIR and Infrared spectral region of the spectrum with different spectral bands in grey (Bhatta, 2011).

1.2.2. Remote Sensing and Geographic Information Systems (GIS)

Traditionally, techniques used for mapping marine habitats depended on ground surveys, direct observation and free diving (Sagawa et al., 2008). The problems with these conventional methods include (1) detailed and continuous field surveys are costly and time-consuming attributed to by the inaccessibility of kelp habitats, (2) even when resources are available, monitoring programs provide information that is scattered spatially and temporally (inefficient for large scale mapping) and (3) the environmental conditions (weather and wave activities) limit data collection, monitoring and mapping of coastal ecosystems such as kelp forests (Sagawa et al., 2008; Hoang et al., 2016; Hedley et al., 2016). However, it is important to note, the use of traditional field-based surveys in coastal ecosystems can be very accurate at a local level (Sagawa et al., 2012; Hedley et al., 2016). Although traditional mapping methods have limitations, they are a crucial component for building current baseline knowledge and validating remotely sensed maps (Bennion et al., 2018).

In recent decades, advances in remote sensing technology and Geographic Information Systems (GIS) have enabled the acquisition of comprehensive information over a range of terrains (Casal et al., 2011; Noiraksar et al., 2014; Uhl, Bartsch and Oppelt, 2016; Hoang et al., 2016; Bennion et al., 2018). The application of remote sensing has a long history for surveying terrestrial areas compared to the aquatic environment (Noiraksar et al., 2014) and the

challenges conferred above are among the many causes for the imbalance when it comes to applying remote sensing in the marine realm. Remote sensing dates to the mid-nineteenth century when aerial photographs (balloon with a mounted camera) gained interest and their gradual transformation to the current hyperspectral remote sensing platforms (advanced satellites) (Gao, 2009; Bhatta, 2011). Multi-temporal observation allows for monitoring of dynamic coastal ecosystems such as kelp forest ecosystems (Petus et al., 2014; Bennion et al., 2018). Remote sensing has created an opportunity for developing seasonal to multi-decadal distribution time-series over larger spatial scales (Bell et al., 2018). Remote sensing is therefore gaining interest for use in mapping both marine and terrestrial environments (Casal et al., 2011; Petus et al., 2014). Many studies concur on the role of remote sensing as an increasingly effective tool for creating spatially continuous inventories and methodologically consistent databases of vegetation composition and distribution (Wabnitz et al., 2008; Casal et al., 2011; Lück-Vogel et al., 2016; Bennion et al., 2018; Terada et al., 2019). One of the benefits includes the fact that remote sensing techniques developed for one area frequently can be adopted for other environments. For example, Sagawa et al. (2012) mapped seaweed forests with IKONOS images using a method that was developed for mapping seagrass beds and coral reefs. Approaches for mapping seagrass properties in shallow water bodies and supra-tidal areas have been transforming since the availability and ease of use of spatial information (Phinn et al., 2008). Therefore, remote sensing is a viable tool for long-term monitoring of spatially extensive kelp forests (Mishra and Gould, 2016; Bell et al., 2018; Nijland, Reshitnyk and Rubidge, 2019). Therefore, this thesis tests the use of multi-spectral remote sensing to map South African kelp forests.

It is worth explaining that using satellite mapping elicits more complex analyses, i.e. remote sensing has challenges: for example, light attenuation within the water column is one of the inherent problems associated with optical remote sensing (Sagawa et al., 2008). Secondly, remote sensing imagery requires a large amount of storage and high processing computers, which might not be easily available (Wabnitz et al., 2008). Using remote sensing for mapping coastal ecosystems does not eliminate the need for ground-truthing and data collection which often requires the use of traditional methods (Silva et al., 2008; Sagawa et al., 2012). While the field surveys provide more details and accuracy for a single point, remote sensing has the potential to provide large spatial area cover (Hedley et al., 2016). Therefore, advances in remote sensing technology provide more solutions than challenges (Lück-Vogel et al., 2016). The synchronisation of ground-truth data and image acquisition dates is required for higher

accuracy of RS mapping results (Lück-Vogel et al., 2016), especially for highly dynamic environments, where a temporal mismatch between the field data and satellite data will lead to discrepancies (Lück-Vogel and Mbolambi, 2018). However, remote sensing technology played a major role in making the use of satellite images possible and relatively easy.

The marine environment is a source of misclassifications due to its dynamic conditions, which lead to discrepancies between field data and satellite data. Another source for the mismatch between spatial maps and the reality on the ground is the lack of consistent and reliable information e.g. in instances where geographic information is available it is often an interpretation of expert knowledge and may lack accurate baseline maps (Wabnitz et al., 2008). Also, the scarcity of long-term monitoring studies focusing on monitoring subtidal habitats is a hindrance to understanding the current distribution of subtidal kelp habitats, particularly in turbid waters (Uhl, Bartsch and Oppelt, 2016). However, following the launch of the many earth observation satellites since the 1960s, there has been a steady increase in remote sensing use to monitor marine ecosystems (Sagawa et al., 2008).

Remote sensing (RS) is the process of obtaining information about an object through analysing data without being in contact with the object under surveillance (Lillesand, Kiefer and Chipman, 2004; Gao, 2009; Campbell and Wynne, 2011). Fundamental to remote sensing is the fact that land and sea surface have characteristic spectral reflection properties. Remote sensing sensors are designed to detect electromagnetic radiation in specific wavelengths, the so-called spectral bands. Based on the derived specific spectral signatures information on specific vegetation and landcover can be derived (Gao, 2009; Bhatta, 2011; Lück-Vogel et al., 2016). The most important function of a remote sensing platform is the ability to discern two adjacent features (Gao, 2009). As such, there are four types of resolution in remote sensing namely spatial, spectral, radiometric and temporal resolution (Gao, 2009). Spatial resolution refers to the capacity to discern between two adjacent features and is synonymous with pixel size (Gao, 2009). Spectral resolution refers to the capability to discern subtle differences in the reflectance of the same object at various wavelengths (Gao, 2009). Radiometric resolution refers to the ability to distinguish subtle differences in the intensity of the radiant energy from object to sensor (Gao, 2009). Temporal resolution is the revisit period, which refers to the frequency of the sensor to sense the same area (Gao, 2009).

Conversely to remote sensing, GIS refers to computer-based systems that are developed to deal with any form of information that can be referenced to a geographic location (Bhatta, 2011).

Geographic Information Systems technology has made it possible to organize and integrate different remote sensing derived products with other data sources and from these to produce maps and perform scientific analysis to inform and support in management decisions (Bhatta, 2011).

Examples of remote sensors include the human eye, satellites, airborne sensors, drones, unmanned aerial systems, boat-based systems and automated underwater vehicles (Hedley et al., 2016; Bennion et al., 2018). Table 1. 1 gives a summary of applications of various optical and hyperspectral remote sensing-based methods used for mapping marine ecosystem types and the different remote sensing platforms.

Table 1. 1: Summary of selected studies that applied remote sensing to monitor and map marine ecosystem types, methods used and summary of findings and recommendations.

Authors	Satellite	Eco-types	Methods and analysis	Findings and recommendations
Casal et al., 2011	Spot 4	Kelp forest (Spain)	Direct observation: diving survey Analysis: Supervised and unsupervised classification and Confusion matrix	The confusion matrix results are greater than 70%. The use of higher spectral, spatial and radiometric resolution can improve results. Signal attenuation remained a challenge
Anderson et al., 2007	Multiple: Infrared Aerial photography, Digital multispectral imagery and Landsat	Kelp forest (South Africa)	Various (discussed in section 3.5)	Infrared and multispectral imaging yielded the most accuracy. Biomass estimation accuracy would be improved by high-resolution IR imaging. Gaps are reported on the map.
Uhl et al., 2016	Hyperspectral AisaEAGLE	Kelp forest (Germany)	Direct observation: diving Analysis: Water Anomaly Filter, Feature Detection and Supervised classification	Water Anomaly Filter removes water anomalies, Feature Detection is a promising tool that discerned Fucales and kelp habitats. The developed methods can easily be transferred to satellites. Combining FD with algae indices could improve results.
Phinn et al., 2008	Quickbird 2 + Landsat 5 + CASI 2	Seagrass (Australia)	Direct observation: Transect and photo diving Analysis: Above ground biomass, Error and accuracy assessment	Airborne Hyperspectral produces high accuracy maps. Developing more accurate methods to synchronize field and RS accuracy. Hyperspectral imagery is expensive.
Wabnitz et al., 2007	Landsat 5 and 7	Seagrass (the Gulf of Mexico and the Caribbean Sea)	Previous data: <i>In situ</i> data, IKONOS images, Analysis: Error matrix from diverse data sets	Accuracy varies for different sites (46 – 88%). Expert validation and high-resolution mapping are advised

Authors	Satellite	Eco-types	Methods and analysis	Findings and recommendations
Pasqualini et al., 2005	Spot 5	Seagrass (the Mediterranean Sea and Greece)	Visual observation: scuba diving and boat-based point field data Analysis: Supervised classification and Error matrix.	The high spatial resolutions yield varying accuracies e.g. 2.5 m resolution accuracy (73%) and 10 m (96%). Capable of discriminating coarse level habitats.
Fornes et al., 2006	IKONOS	Seaweed (Mallorca)	Reference data: Acoustic survey Analysis: Supervised classification, geometric correction and error matrix	The multispectral image accuracy is 84%. Although the method is restricted to clear and shallow water
Sagawa et al., 2012	IKONOS	Kelp forest (Japan)	Visual observation: video camera and boat-based, Analysis: Spectral reflectance profiling supervised classification, error matrix	High spatial and radiometric resolution and measuring spectral reflectance yielded an accuracy of 97.9 %.
Noiraksar et al., 2014	ALOS AVNIR 2	Kelp forest (Gulf of Thailand)	Visual observation: underwater camera, Analysis: image software, masking, geometric correction, supervised classification, Error matrix	Yielded accuracy 70% the and results can be improved by higher spatial resolution satellite data could improve results.

1.2.3. Normalized Difference Vegetation Index (NDVI)

Using the magnitude of the reflected radiation an estimation of the presence of water and moisture in and around plants can be gauged (Campbell and Wynne, 2011). Seaweeds have a strong reflection within the 350 to 950 nm wavelengths (Sagawa et al., 2012). Because the spectral signature of kelp is not different from that of land vegetation, they both have a higher spectral signature in the near-infrared region and lower in the visible region of the spectrum (Cavanaugh et al., 2011). However, submerged plants have a spectral signal that consists of both “water” and a “vegetation” signal. The degree of signal mixing depends on the density of the submerged vegetation (Sagawa et al., 2012; Uhl, Bartsch and Oppelt, 2016). Therefore, the major challenge for remote sensing is isolating the submerged plant signal from the overall water column spectral signal (Silva et al., 2008; Sagawa et al., 2012). Their photosynthetic nature and strong reflection within 350 to 950 nm wavelengths can enable differentiating interspecies around 700 to 950 nm when using hyperspectral satellite imagery (Casal et al., 2011; Sagawa et al., 2012).

Normalized Difference Vegetation Index (NDVI) is the normalized ratio between the visible red region and near-infrared bands of the electromagnetic spectrum (Gao, 2009). Normalized Difference Vegetation Index was introduced in 1973 by Rouse based on observations that actively growing vegetation strongly absorbs radiation in the visible region (red and blue) of the spectrum while strongly reflecting the radiation in the Near-Infrared region (Figure 1. 2) and is derived using the equation below (Equation 1.2) (Gao, 2009).

$$NDVI = (NIR - RED) / (NIR + RED) \dots\dots\dots \text{Equation. 1. 2}$$

The Normalized Difference Vegetation Index has also been used on floating and emergent aquatic vegetation. The normalisation of the RED-NIR ratio yields values between -1 and 1. When NDVI is greater than 0.1, this characterises presence and quantity of vegetation. Surface vegetation has a range of values between 0.1-1. Values between 0 and 0.1 usually characterize rock and empty land and values smaller than 0 indicate ice clouds, water vapour and snow. The application of NDVI for submerged aquatic vegetation (SAV) is more challenging, because of high NIR radiation absorption by water (Figure 1. 2), obscuring vegetation signals (Gatc, Gunawan and Maspiyanti, 2016).

1.2.4. Supervised Image classification

The method of supervised classification is an analysis that is commonly used in numerous mapping studies (Table 1.4). It is a process of identifying spectrally similar areas in an image by using areas of known objects for training the algorithm to assign other areas with similar spectral properties to the same object class (Gao, 2009; Noiraksar et al., 2014). In Oura Bay, Shimoda in Japan, in a study of mapping seaweed forest using remote sensing and side-scan sonar gauging, supervised classification was applied to the satellite IKONOS image to assess dominant seaweeds (Sagawa et al., 2008). Volent et al. (2007) mapped kelp forests in Kongsfjord, north-west Spitsbergen in Norway using an airborne hyperspectral imager and employed supervised classification and differential histogram equalization technique. Similarly, sargassum beds were mapped by Noiraksar et al. (2014) off the coast of Chon Buri, Thailand, using satellite supervised classification of ALOS AVNIR-2 satellite imagery. Casal et al. (2011) employed supervised and unsupervised classification techniques to map kelp beds in turbid waters on the south European Atlantic shelf. In furthering their study conducted in 2008, Sagawa et al. (2012) developed a method to overcome the difficulty of collecting ground reference data and the remote sensing complexity introduced by light attenuation in the water column, by profiling the spectral signature of kelp species and surrounding substrate using a spectroradiometer. This is an excellent method although it requires a tool that is quite expensive.

1.2.5. Satellite imagery used: Sentinel-2 Multispectral Instrument (MSI)

Sentinel-2 (S-2) is one of the European Space Agency (ESA) satellite constellations, consisting of two mirror satellites, Sentinel-2 A and 2B, designed to meet various observational needs for physical, bio-geophysical and biological spatial variables of the ocean, cryosphere and land research activities (Malenovský et al., 2012; van der Werff and van der Meer, 2016). The design objectives of Sentinel-2 are to provide systematic global acquisitions with high-resolution, multi-spectral imagery and high revisit frequency (Berger et al., 2012; Drusch et al., 2012). Sentinel-2 complements the Landsat and SPOT (Satellite Pour l'Observation de la Terre) satellite missions (Table 1. 2) (van der Werff and van der Meer, 2016) which have longer orbital periods and thus, serve as inventories for accessing historical data than Sentinel-2 which is recent. Sentinel-2A (S-2A) was launched on 23 June 2015 and started capturing imagery shortly thereafter. Whereas the identical Sentinel-2B sensor was later launched on 07 March 2017 (<http://earth.esa.int>).

Table 1. 2: Characteristics of the most common multispectral sensors used for vegetation mapping in comparison to the resolutions of Sentinel-2.

Satellite	Bands	Temporal Resolution	Principle	Spatial Resolution (m)
Sentinel-2 A&B	13	10 days (5 in combination with S-2B).	Push broom	10, 30, 60
Landsat 1-7	7	16 days	Scanner	30, 60
SPOT-5	4	26 days	Push broom	2.5, 10, 60

Sentinel-2 (A and B) is comprised of a super-spectral imager with 13 spectral bands, four in the 10 m resolution, six in the 20 m and three in the 60 m resolutions (see: Table 1. 3) and a 290 km swath width. Sentinel-2 satellite incorporates two new spectral bands in the red-edge region, significant for retrieval of chlorophyll content (Drusch et al., 2012; Malenovský et al., 2012; Berger et al., 2012; van der Werff and van der Meer, 2016). The different band combinations (Table 1. 3) are useful for optimising generalisation and use for agriculture, atmospheric penetration, healthy vegetation, land/water, atmospheric removal, shortwave infrared, vegetation analysis, snow/cloud, Normalized Difference Vegetation Index (NDVI), Normalized Difference Water Index (NDWI) and Index stacks applications (https://www.esa.int/Our_Activities/Observing_the_Earth/Copernicus/Sentinel-2). Table 1. 3 shows bands 2, 3, 4 and 8 with 10 m resolution that were used in the current study to determine true-colour and the NDVI.

Table 1. 3: Sentinel-2 MSI spectral bands. Bands used in this study are highlighted (Cited: <http://eos.com>).

Band name	Band number	Central wavelength (nm)	Resolution (m)
Coastal aerosol	1	443	60
Blue	2	490	10
Green	3	560	10
Red	4	665	10
Vegetation Red Edge	5	705	20
Vegetation Red Edge	6	740	20
Vegetation Red Edge	7	783	20
NIR	8	842	10
Narrow NIR	8a	865	20
Water vapor	9	945	60
SWIR - Cirrus	10	1380	60
SWIR	11	1610	20
SWIR	12	2190	20

1.3. Assessing kelp forest ecosystem threat status

Anthropogenic actions are the cause of much of the detriment to biodiversity and they are predicted to continue to increase (Hedley et al., 2016; Starko et al., 2019). Because of anthropogenic actions, many ecosystems including kelp forests are declining or experiencing degradation at unprecedented rates (Franco et al., 2018; Filbee-Dexter and Wernberg, 2018; Terada et al., 2019). In response, kelp forests are at risk of losing their ecosystem engineering roles as they play significant roles such as exerting control directly and indirectly to many connected communities by modifying resources including the environment available to other organisms (Pereira et al., 2017). Worldwide, a decline in kelp forests has been reported in recent decades (Connell et al., 2008; Wernberg and Filbee-Dexter, 2019). The consequences are clear in many parts of the world where continued degradation of ecosystem engineers have eroded their ability to regulate climate, sequester carbon, recycle nutrients, protect the coast and primary production *inter alia* (Krumhansl et al., 2016; Wernberg and Filbee-Dexter, 2018; Wernberg et al., 2018). Most concerning is that, at the hand of humans, the recent decades have had many marine ecosystems undergoing regime shifts, with many unable to recover to their former states, which persisted prior to the pressures/threats, even after threat removal (Halpern et al., 2012; Micheli et al., 2013; Filbee-Dexter and Wernberg, 2018; Bland et al., 2018).

Ecosystem risk assessments are fundamental tools that are developed to identify and mitigate the cascading effects of ecosystem degradation across the world (Bland et al., 2015; 2017a). The reported rates of degradation of natural ecosystems can be reduced (Keith et al., 2013). Through, developing key ecological indicators and elucidating trophic pathways that enables better measuring of the spatial extent and functional impact of human actions and guide the setting of trade-offs between ecological ecosystem services and human benefits (Crain et al., 2008; Coll et al., 2016).

1.3.1. South African National Biodiversity Assessment

The South African National Biodiversity Assessment (NBA) is a cornerstone for the fulfilment of the mandate of the South African National Biodiversity Institute (SANBI) rendered by the National Environmental Management: Biodiversity Act (Act 10 of 2004) which specifies the need to monitor and regularly report on the state of the nation's biodiversity across all environments. The NBA 2018 is a recent product published in 2019 and conducted over the previous three years. It is subsequent to the second NBA 2011 (Driver et al., 2012) which followed the first National Spatial Biodiversity Assessment (NSBA) conducted in 2004 (Driver

et al., 2005). The NBA serves as a national reporting system, devised to aid in reducing the degradation of natural ecosystems. However, the assessment of marine ecosystem is still lagging behind other realms. The NBA 2011 set the stage for fine-scale resolution ecosystem assessment, mapping and alignment with emerging global classification and ecosystem assessment efforts.

Since the NSBA 2004, marine ecosystem mapping and assessment which are foundational to this current study, has been gradually increasing and improving to facilitate the assessment of ecosystem threat status and ecosystem protection levels (Figure 1. 3) at relevant scales. The NBA team developed “headline indicators” to report on ecosystem threat status (Figure 1. 4) and ecosystem protection level (Skowno et al., 2019). The former biodiversity indicator is used to report on the rate and extent of damage/degradation to ecosystem types, whereas, the protection level status informs of the extent of ecosystem protection i.e. how much of the ecosystem is found in a Marine Protected Area (reported as Well Protected to Not Protected range). It was beyond the scope of this thesis to apply the second headline indicator which is also the topic of another study.

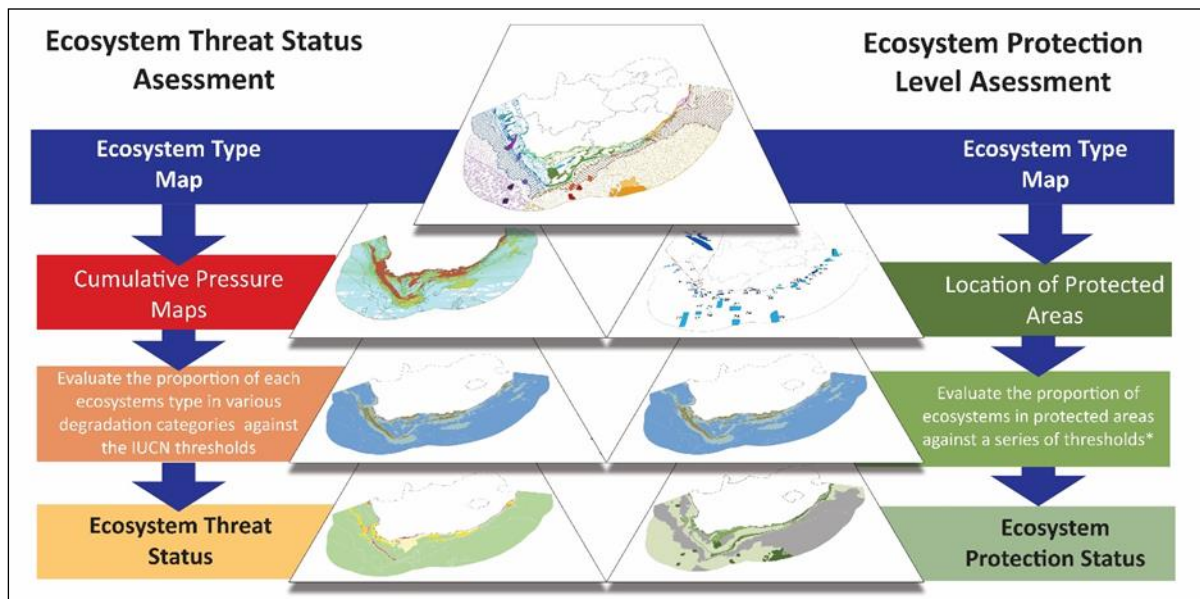


Figure 1. 3: Steps applied in South Africa to assess ecosystem threat status and ecosystem protection level (Sink et al., 2019).

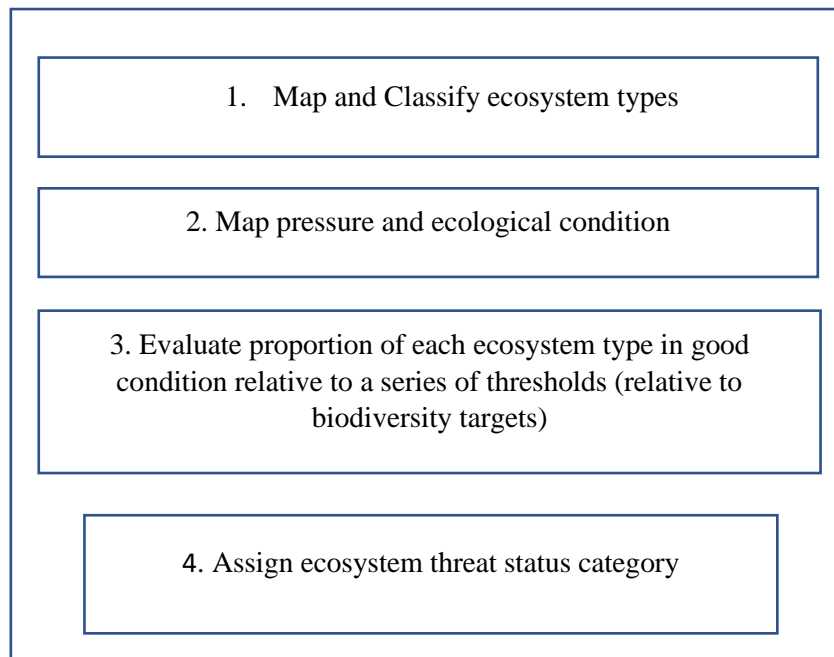


Figure 1. 4: Summary of the stages of assessing marine ecosystem threat status as developed by the NBA 2011.

Biodiversity targets are used in biodiversity assessments and planning. In South Africa, the biodiversity target is considered the minimum proportion of each ecosystem type that needs to be kept in a natural or near-natural state in the long term to maintain viable representative samples of all ecosystem types and the majority of species associated with those ecosystems (Desmet and Cowling, 2004; Reyers et al., 2007). Biodiversity targets should preferably be based on the ecological characteristics of the ecosystem concerned with terrestrial targets ranging between 16 and 36% (Desmet and Cowling, 2004). Less science has been conducted to inform marine ecosystem targets and therefore a standard biodiversity target of 20% has been used (Porter et al., 2011; Sink et al., 2012; 2019). Previous assessments of threat status and previous and current assessments of ecosystem protection level require biodiversity targets for ecosystem types. The 2011 assessment used these biodiversity targets as thresholds for the threat status categories (see: Table 3. 1 in Chapter 3 for further information) but in 2018, South Africa adopted the IUCN RLE approach (see: section below), which uses fixed thresholds irrespective of any set biodiversity targets with targets now only being used only in the protection level assessment in the NBA.

To measure ecosystem degradation, a single pressure that has high intensity and/or many pressures with high combined intensities impacting on an ecosystem will trigger a poor ecological condition (Table 1. 4). A single pressure with low intensity and/or few pressures with low intensities acting on an ecosystem will trigger a good ecological condition and the

ecological condition of an ecosystem is assessed as fair if it is impacted by a range of pressures with moderate intensities (Sink et al., 2012).

Table 1. 4: Pressure impact on overall biodiversity ecological condition (NBA 2011).

Pressure	Few pressures, low intensity	Range of pressure, moderate intensity	Many pressures, high intensity
Condition	Good	Fair	Poor
Expected biodiversity impact	Biodiversity patterns and ecological processes intact	Some ecosystem degradation	Loss of biodiversity pattern and disruption of ecological processes

1.3.2. Global Red List of Ecosystems

In 2008 the International Union for Conservation of Nature (IUCN) developed risk assessment criteria to support the global Red List of Ecosystems (RLE). Through the identification of species that are most threatened by the risk of extinction, the IUCN RLE gathers comprehensive scientific data and informs governments and society of the status of biodiversity, trends of risk to ecosystem collapse and provides data to formulate priority actions and strategic actions for ecosystem management (Rodriguez et al., 2015). This emerging model enables the assessment of the risk of ecosystem collapse globally for freshwater, marine, terrestrial and subterranean ecosystem types (Bland et al., 2015). The IUCN RLE is the product of extensive research and robust consultation and the model and criteria were from work dating back to 1994, which have since been reviewed in the IUCN 2001 and 2012 (Bland et al., 2015). The criteria are used to assess spatial and functional symptoms of ecosystem risk to collapse and were first implemented in the assessment of species Red List of threatened animals (Keith et al., 2013). There are five criteria (A-E) (discussed in chapter 3) that are used to ensure the method of assessment is 1. applied systematically across realms and geographic areas, 2. transparent and scientifically rigorous, 3. comparable and repeatable, 4 easily understood by policymakers and the general public and 5. complements the IUCN Red List of Threatened Species framework (Keith et al., 2013; Rodriguez et al., 2015; Bonansea et al., 2015; Bland et al., 2018).

While there is substantial evidence that ecosystem functions and services are linked to biodiversity (Bland et al., 2017b), the relationships between these three facets of ecosystems

can be complex. Consequently, the RLE focusses specifically on risks to biodiversity (Keith et al., 2013). On land, vegetation types, have been suggested as appropriate and consistent units that represent biodiversity and communities at an appropriate scale for use in the RLE. In the marine environment, the IUCN endorsed the use of South Africa's marine ecosystem classification. The IUCN, however, reports that a global marine ecosystem classification is underway (Bland et al., 2017a).

The concept of ecosystem collapse, an endpoint of ecosystem decline is used in the RLE framework. This is equivalent to species extinction in the Red List of Species and is defined operationally as a 'transformation of identity, loss of defining abiotic or biotic features and a state where characteristic native biota are no longer sustained' (Keith et al., 2013). The IUCN guidelines recommend using ecosystem types for national assessments and note that ecoregions are not appropriate to feed into spatial management at a national scale (Bland et al., 2017a).

1.3.3. Pressures that threaten kelp forest ecosystems

Current socio-ecological theories consider humans as part of the marine ecosystem (Borja, 2014). Humans have interacted with kelp forest ecosystems since at least 70 000 years ago (Filbee-Dexter and Wernberg, 2018). Early humans settled around rocky shores and fringing kelp forests, where they derived a variety of rich diet from the coastal ecosystems, as is evident in middens (Ainis et al., 2014; Blamey and Bolton, 2018). Anecdotal and scientific evidence indicates the role kelp forest ecosystems played in human biogeography and their benefits to billions of people across the world who live, work and engage in recreation around them (Bennett et al., 2016). Despite their importance, over-exploitation by humans is increasingly altering kelp forest ecosystems (Witman and Lamb, 2018).

When exposed to environmental conditions outside their normal physiological range, macroalgae respond in one of four possible ways: migration, adaptation, extinction or acclimatization (Biskup et al., 2014). Therefore, their limited dispersal ability makes them particularly vulnerable to climate change, which can transform their environment to unfavourable states and subsequently lead to habitat abandonment through migration and/or extinction (Merzouk and Johnson, 2011). Human impacts on these foundational marine species have increased dramatically in recent decades, resulting in a decline in about 38% of the world's kelp forests (Krumhansl et al., 2016).

A plethora of studies are reporting that kelp forest ecosystems are changing, across the entire world (Leisten, 2002; Connell et al., 2008; Ling et al., 2009; Blamey et al., 2010; Lorentsen et al., 2010; Davoult, 2011; Harley et al., 2012; Keith, 2013; Raybaud et al., 2013; Steneck and Johnson, 2013; Campbell et al., 2014; Ling et al., 2014; Seitz et al., 2013; Wernberg et al., 2015; Pereira et al., 2015; Parnell, 2015; Estes et al., 2016; Vergés et al., 2016; Krumhansl et al., 2016; Uhl et al., 2016; Smale and Moore, 2017; Witman and Lamb, 2018; Filbee-Dexter and Wernberg, 2018; Wernberg et al., 2018; Franco et al., 2018; Starko et al., 2019; Wernberg and Filbee-Dexter, 2019; Smale et al., 2019; Terada et al., 2019). In response to external pressures such as warming (Mohring et al., 2014; Vergés et al., 2016; Hargrave et al., 2017; Witman and Lamb, 2018) and ocean acidification (Poore et al., 2016; Hernández et al., 2018), some species of macroalgae are shifting their range or migrating towards the poles and are replaced by species with warmer water affinities (Johnson et al., 2011; Harley et al., 2012; Duarte et al., 2018). Johnson et al. (2011) show regional declines in the extent of *M. pyrifera* and *E. radiata* because of a recent establishment of sea urchin (*Centrostephanus rodgersii*) population that have extended their range due to warming waters along Tasmania and subsequently grazed down large areas of kelp forest with dire consequences on the ecosystems in this region. This shift has also been reported by Ling et al. (2014) and includes the range expansion of other warm-water species along the coasts of Tasmania. This example is like that of the South African west coast rock lobster, which shifted eastward and established itself in a region where it was previously rare with cascading effects that followed. In other parts of Australia, reef ecosystems have shifted from ones previously dominated by kelp forests to alternative and persistent states of seaweed turfs, as a result of ocean warming (Wernberg et al., 2015).

The strong impacts of climate-driven change have consequences including increased frequency and severity of storm events (Pereira et al., 2017; Fraser et al., 2018), intense heat-waves (Wernberg et al., 2015; Smale et al., 2019) and increased sea surface temperatures (Graham et al., 2007; Ling; 2008; Ling et al., 2009; Johnson et al., 2011; Harley et al., 2012; Lamy et al., 2018). While some populations are responding to climate change by declining e.g. seaweed communities on Japan's coast (Sagawa et al., 2012; Terada et al., 2019), others are increasing their range of distribution e.g. *E. maxima* (Bolton et al., 2012), *Sargassum horneri* (Marks et al., 2015; Pfister et al., 2018). Another example is that of *L. ochroleuca* spreading its distribution northwards and disappearing from the coast of Spain (Pereira et al., 2017).

In addition to the effects of climate change, key ecological services (Filbee-Dexter and Wernberg, 2018) and socioeconomic values derived from kelp forest ecosystems (Field et al., 1977; Bennett et al., 2016; Blamey and Bolton, 2018) are further eroded by overfishing (Steneck, 1998; Ling et al., 2009; Blamey et al., 2014), coastal development (Leisten, 2002; Mead et al., 2013), invasive species (Valentine et al., 2008), mining (Branch, 2008; Mead et al., 2013) and pollution (Liu et al., 2009; Halpern et al., 2012; Wernberg et al., 2015; Ojemaye and Petrik, 2019).

The changes that occur in the marine realm at multiple levels partly due to the pressures discussed above and also due to other interacting factors that are not yet fully understood (Bax et al., 2019). Development of global systems for observing biological oceans is becoming a technical reality (Terada et al., 2019). Studies like this current study are crucial for understanding the potential unprecedented rates at which biodiversity may be at risk and for guiding management action to reduce these risks.

1.4. Thesis Aims

In South Africa, a map of the kelp forests was last developed in the mid-2000s and gaps in the mapping of the kelp forests distribution were identified (Anderson et al., 2007). Until now, there has been no uniform mapping approach developed for monitoring the extent of kelp forests. Therefore, a gap remains in the monitoring for better management of South African kelp forest ecosystems. Secondly, the threat status of South African kelp forest ecosystems has never been assessed despite South Africa being at the forefront of developing national ecological indicators for marine and terrestrial ecosystem types (e.g. National Vegetation Mapping and Marine component of the NBA). Until now, kelp forests have been excluded from previous biodiversity assessments because of a lack of a high resolution and comprehensive kelp forest map.

The overarching goal of this study was to improve knowledge and understanding of the extent and ecological status of the kelp forest in South Africa by (1) producing an ecosystem type map of South African kelp forests using advanced remote sensing-based Sentinel-2 imagery and (2) assessing the threat status of the South African kelp forest ecosystem types using cumulative pressure mapping methodologies to facilitate the use of the International Union for Conservation of Nature Red List of Ecosystems (IUCN RLE). In addition, this study aimed to distil key lessons for global and national marine ecosystem assessments.

To address the goals of this study, current advanced remote sensing technology was used to develop a new map to update and expand the previous kelp forest map (Anderson et al., 2007). The new map serves as a fundamental step towards assessing the threat status of South African kelp forests.

The second chapter of this study focuses on the approach and outputs of research to map the South African kelp forests. The outcomes include an up-to-date baseline kelp forest map and its classification into distinct South African kelp forest ecosystem types. The third chapter uses cumulative pressure mapping to develop an ecological degradation map and assesses ecosystem threat status using three criteria of the IUCN RLE approach to evaluate the risk of ecosystem collapse for South Africa's kelp forest ecosystem types. The product of this chapter, specific to South Africa, is a threat status assessment of the different kelp forest ecosystem types and the identification of future improvements in ecosystem assessment input data and assessment. In Chapter 4, a synthesis of this research is provided including the main implications and recommended way forward.

Chapter 2: The application of contemporary remote sensing technology to map South African kelp forests

2.1. Introduction

2.1.1. Distribution of South African kelp forests

Kelp forest ecosystems fringe the rocky shores of temperate and high-latitude regions, traversing *ca.* 25% of the world's coastlines and occurring within an estimated 43% of the world's marine ecoregions (Steneck et al., 2002; Krumhansl et al., 2017; Teagle et al., 2017; Blamey and Bolton, 2018). To effectively manage the world's oceans, global scale collaborations are necessary through developing and sharing information about marine biological systems, spatial data on the distribution and intensity of anthropogenic pressures and other threats (Steneck et al., 2002; Johnson et al., 2013; Noiraksar et al., 2014). Often, global-scale studies show the distribution of South African kelp forests at a coarse-scale (e.g. Steneck et al., 2002; Krumhansl et al., 2016). Without comprehensive and up-to-date high-resolution kelp forest maps, there is little that can be done to influence proper management and protection of these key ecosystem engineers at the required local scale (Uhl et al., 2016; Bennion et al., 2018; Nijland, Reshitnyk and Rubidge, 2019). Therefore, the first fundamental step for effectively managing kelp forest ecosystems depends on understanding their distribution. As such, this study seeks to answer the question, "where are South African kelp forests?"

Advanced techniques that enable the generation of decadal monitoring data to better understand variations in the spatial scale of biodiversity need more attention (Lamy et al., 2017). The macroalgal cover, including kelp cover, is one of eight biological essential ocean variables (EOV) recently identified by the Global Ocean Observing System (GOOS, <http://www.goosocean.org>) as useful for monitoring ecosystem changes. Projects entailing long-term observations and monitoring of these EOVs can help identify how oceans are changing with human pressures and several global efforts have already begun in this regard (e.g. KEEN <http://www.kelpecosystems.org>; Bax et al., 2019; Terada et al., 2019).

Many coastal ecosystems including kelp forests are among the most valuable and at the same time most vulnerable habitats (Lück-Vogel and Mbolambi, 2018; Blamey and Bolton, 2018). This makes these ecosystems susceptible to many anthropogenic pressures that often compromise their integrity (Smale et al., 2013; Mead et al., 2013). Therefore, understanding the resilience of kelp ecosystems to the effects of anthropogenic actions is also important for

their protection (Smale et al., 2013). Their valuable ecological, social and economic services are at risk due to the multiple pressures they already face because of the unsustainable exploitation by humans (Tegner and Dayton, 2000; Halpern et al., 2012; Bennion et al., 2018).

2.1.2. Previous work on mapping SA coastal ecosystems including kelp forests

As a result of the established key ecological (Field et al., 1977; Christie et al., 2019) and economic (Vásquez et al., 2014; Blamey and Bolton, 2018) services kelp forest ecosystems deliver, research to map them and other coastal ecosystem types (i.e. corals, sea-grass beds, mangroves and estuaries) is gaining global interest (Kairo et al., 2002; Vanderstraete et al., 2005; Gullström et al., 2006; Hoang et al., 2016; Lück-Vogel et al., 2016). The use of remote sensing is attracting attention (Kairo et al., 2002; Drusch et al., 2012; Bell et al., 2018), because it allows the assessment of large and/or otherwise inaccessible areas, easier and standardised processing of data and frequent surveys that are valuable for monitoring (Silva et al., 2008; Casal et al., 2011; Bell et al., 2015; Bennion et al., 2018; Nijland, Reshitnyk and Rubidge, 2019).

In chapter 1 (Table 1. 1), several international studies using remote sensing to monitor and map coastal ecosystems including kelp forest ecosystems were outlined. In the case of South Africa, several studies focus on the mapping of coastal ecosystems (Lück-Vogel, Barwell and Theron, 2011; Lück-Vogel et al., 2016; Dayaram et al., 2017; Lück-Vogel and Mbolambi, 2018; Harris et al., 2019), although, these do not include kelp forests. The only two known studies that have mapped South African kelp forests to date include the Master of Science theses of Tarr (1993) and Rand (2006) with data summaries in Anderson et al. (2007). The current study seeks to augment these studies including addressing key gaps acknowledged by these authors. An updated map was planned, by exploiting multispectral resolution remote sensing and the interaction of submerged kelp vegetation and water with the RED and NIR spectral bands and develop a kelp forest map based on this spectral interaction.

Tarr (1993) produced hard copy maps of kelp distribution extending along the Concession Areas 5,6,7,8 and Dyer Island (Figure 2. 1) as part of his work quantifying abalone habitat. Rand (2006) digitised this work and provides a comprehensive account of how this work was refined and prepared for his analysis. Further, Rand (2006) digitised infrared aerial print photographs from 1996 and 1997, as indicated in Figure 2. 1. In collaboration with an

international organisation, Landsat 7 ETM+ from 2002 were used to estimate kelp bed planimetric areas. The latter images were compared to the infrared images from 1996 and 1997 and the map produced by Tarr (1993) (see: Rand, 2006; Anderson et al., 2007). Further, colour infrared aerial photographs (CIR) were analysed by Rand (2006). Spectral Angle Mapper (SAM), a supervised classification rule set method was applied to match pixels to reference spectra of the CIR. Landsat 5 images extending along Concession Areas 5; 7; 8; 9 (Figure 2. 1) during low tide (0.41 m), were obtained and analysed to identify kelp forests. This study also conducted unsupervised classification for comparison with the SAM following a Normalised Vegetation Index (NDVI) classification. Ski boats and GPS were used to geo-locate and map kelp beds at spring low tide.

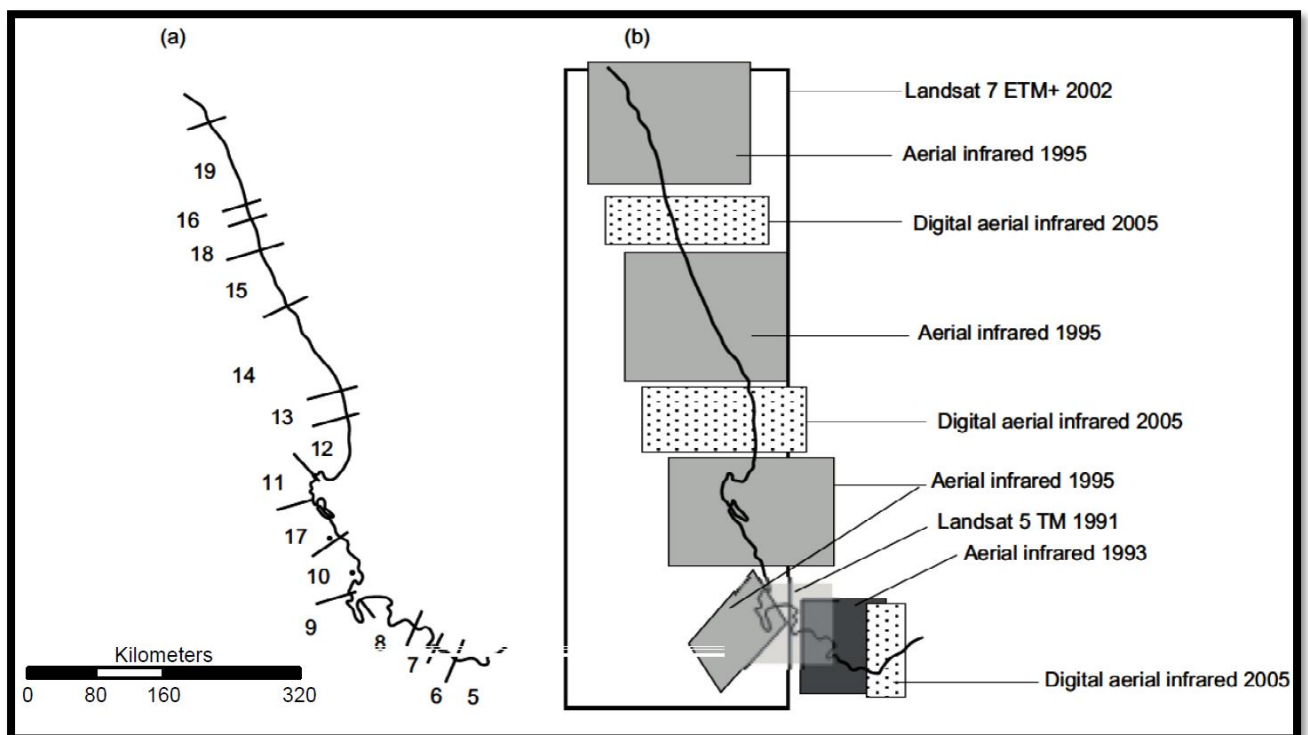


Figure 2. 1: Diagram depicting different remote sensing sources, their year of acquisition and their different coverage adapted from Anderson et al. (2007).

A study by Rothman et al. (2017) described the dominance and occurrence of the two dominant kelp forest forming species along the Southern African West Coast using benthic quadrats in shallow water (Figure 2. 2).

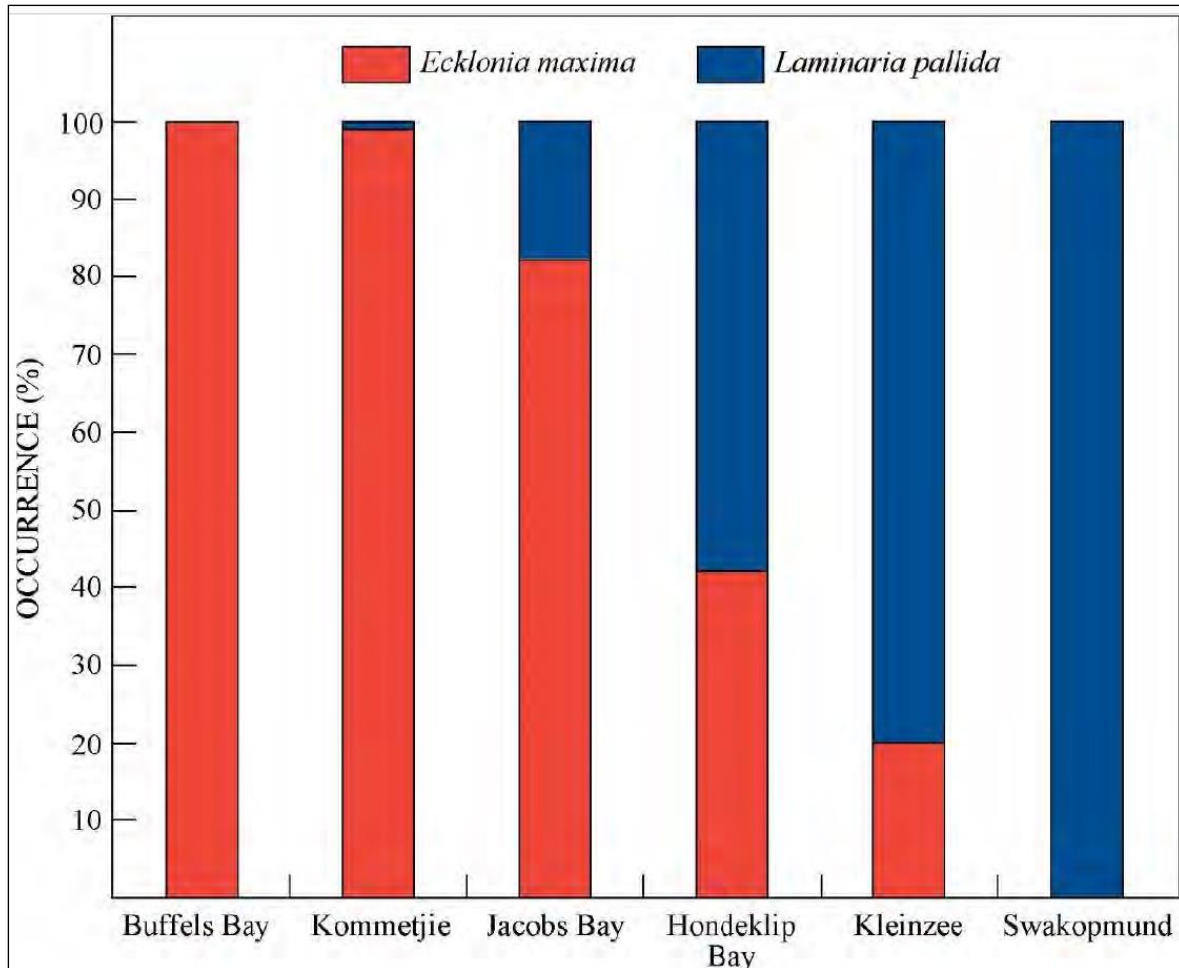


Figure 2. 2: Bargraph showing kelp species dominance and occurrence in selected shallow areas by Rothman et al. (2017).

Until now, South African kelp forests have been lacking a comprehensive high-resolution map produced by consistent methods. The previous map of the mid-2000s applied different remote sensing methods to develop the first inventory of the South African kelp forest covering 900 km of the complex coastline. However, there were still several gaps in this coarse resolution map. Canopy forming kelp forests were underestimated (due to low resolution), subtidal kelp forests were excluded (with the implications of biomass estimation) and the entire kelp forest extent was not covered. Based on the data gaps, the complexity that results from trying to align different remote sensor imagery and resolutions, results were spatially inconsistent, which inhibited its use for setting maximum yields and following concerns of over-harvesting of the kelp species (Anderson et al., 2006; 2007). Subsequently, the priority actions of the National

Biodiversity Assessment (NBA 2011) recommended the development of finer scale resolution maps of South African Kelp Forest Ecosystems. Therefore, this study addresses these.

2.1.3. Biogeography of South African kelp forests

As described in section 1.1.4 the South African coastline spans more than 3000 km (Harris et al., 2019) covering the three biogeographic regions. Biogeography of South Africa was pioneered by Stephenson (1948). The West Coast seaweed region was further evaluated by Bolton and Levitt (1987) who reported its range to extend from the Namibia/Angola border to Cape Agulhas. This historic literature, expert knowledge and Figure 2. 2 above were instrumental for the classification of the kelp forest ecosystem types of South Africa.

2.1.4. Chapter aim and objectives

This study aims to develop a map of kelp forest ecosystem types for South Africa to improve the scientific knowledge on their distribution, facilitate sustainable management and assess the ecosystem condition and threat status.

The primary objectives addressed in this chapter are:

1. to develop and apply a Sentinel-2 based remote sensing method for mapping the kelp forests,
2. assess the accuracy of the mapping results
3. and classify the ecosystem types as the first step to ecosystem assessment.

2.2. Methods

2.2.1. Study site

The two dominant South African kelp species extend from Orange River (Alexander Bay) to the De Hoop nature reserve (Figure 2. 3) covering about 1300 km of the South African coastline (Mead et al., 2013; Rothman et al., 2017). To the west of the study area is the Atlantic Ocean, to the south is the Southern Ocean and to the east is the Indian Ocean. These three oceans create one of the most diverse marine environments worldwide.

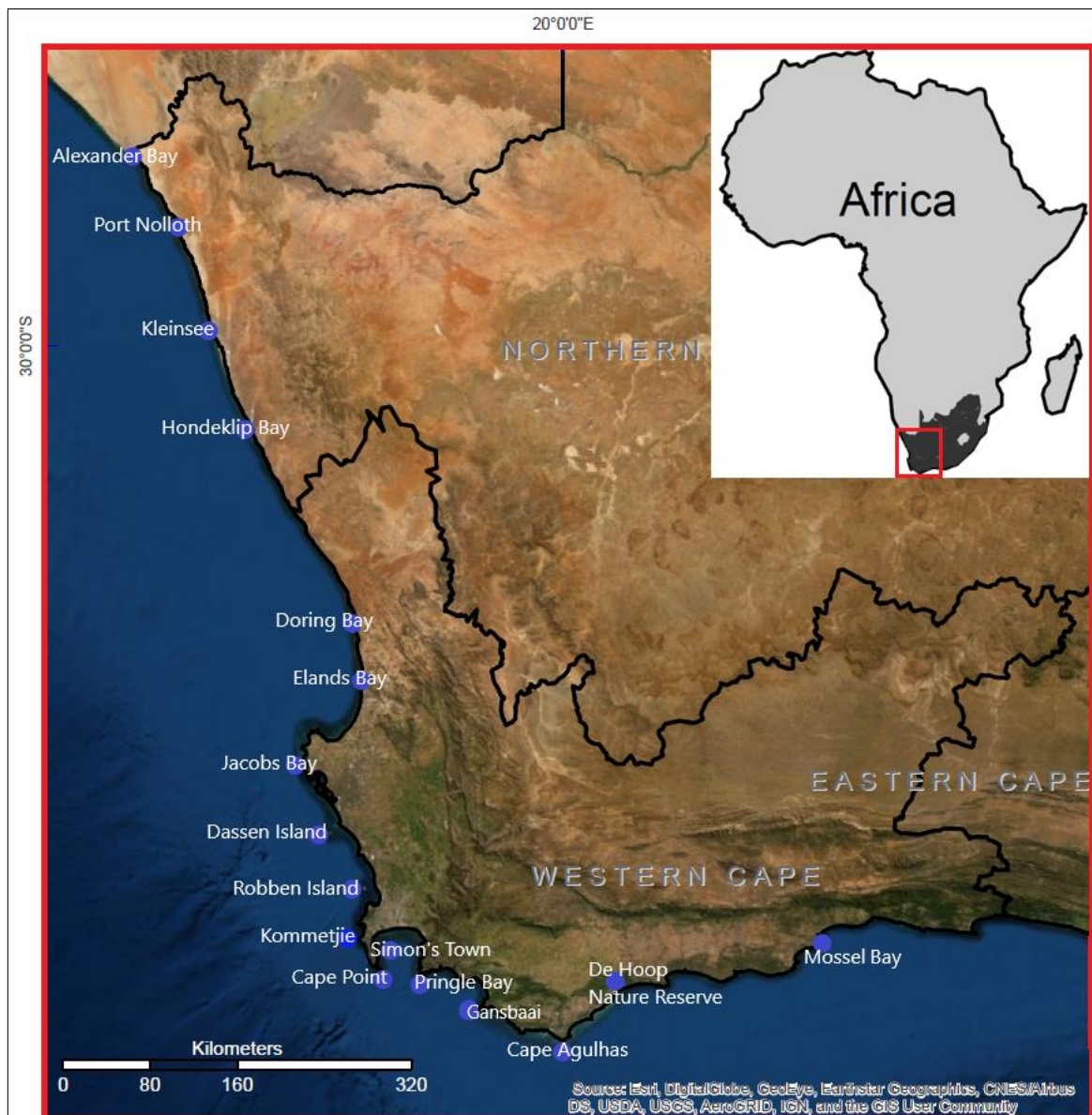


Figure 2. 3: Map of the South African West and South coast, showing locations mentioned in-text. Some of the sites were visited for *in situ* measurements and kelp extent surveys (Bolton and Anderson, 1987).

2.2.2. Model for the mapping procedure

The schematic procedure for mapping the South African kelp forest is presented in the flow chart in Figure 2. 4. It depicts the various steps conducted to produce the final kelp map. This process entailed image acquisition, processing, ground-truthing and the processing of previous maps (Figure 2. 4). The schematic model captures the following steps:

Step 1: Sentinel-2 data acquisition.

Step 2: Preparation of the data developed by Rand (2006) as reference data.

Step 3: Incorporation of ground validation data into the reference data.

Step 4: Incorporation of expert knowledge into reference data.

Step 5: Incorporation of Google Earth (GE) data into the reference data.

Step 6: Utilisation of reference data for validating S-2 image mosaic.

Step 7: Calculation of NDVI and setting thresholds.

Step 8: Comparing reference data to the NDVI map result.

Step 9: Using specific high confidence areas of interest (AOI) of reference data to develop validation points for accuracy assessment in GE.

Step 10: Exploring Maximum Likelihood Classification (MLC) (supervised classification).

Step 11: Subjecting MLC to expert review.

Step 12 Comparing MLC to NDVI and merging results for increased accuracy.

Step 13: Subclassifying the final map product to three kelp forest ecosystem types.

The major steps in this process are represented (Figure 2. 4) and discussed in more detail in the next sections.

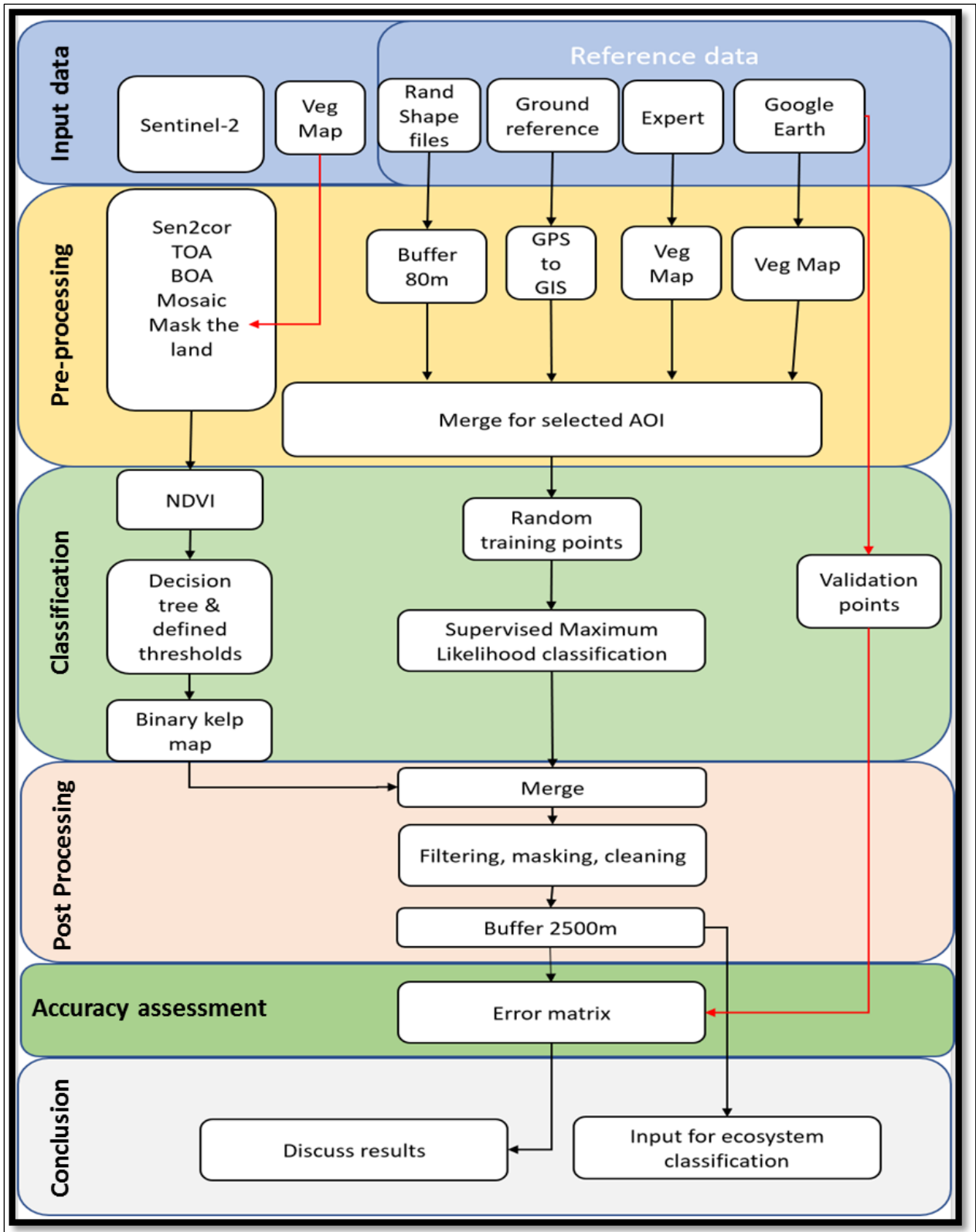


Figure 2. 4: Simplified conceptual model of the kelp mapping process conducted in this thesis. The remote sensing-based processing, collation of reference datasets for validation and supervised classification algorithm are described in the sub-sections below.

2.2.3. Input data

2.2.3.1. Ground reference data

A variety of data sources were used to develop and strengthen the maps used for reference as a source of validation and masking out areas of no concern to this study. These included field data, previous maps (Anderson et al., 2007), expert knowledge, the latest National Vegetation Map (Dayaram et al., 2017) and Google Earth.

2.2.3.2. Field data

Several locations for conducting field-based observations and for collecting ground reference data were selected based on accessibility and prior knowledge of areas along the coastline of South Africa (Figure 2. 3). The work by Rothman et al. (2017) (Figure 2. 2) was used to advise the process. The various locations in Figure 2. 3 were visited for field surveys in collaboration with the Seaweed unit (from the Department of Environment, Fisheries and Forestry). Surveys were conducted between March 2017 and June 2018 and photographs of the locations were taken where possible (Table 2. 1 and images are shown in Figure 2. 5). At some of the visited sites, canopy kelps were visible with the most conspicuous kelps species being *E. maxima* and *L. pallida*. No sightings of *E. radiata* occurred at any of the visited locations and *M. pyrifera* was sighted only in Kommetjie.

Table 2. 1: Examples of ground-truthing locations and associated GPS coordinates with photos from selected sites in (Figure 2. 5).

Description	Coordinates
Kelp forests dominated by <i>L. pallida</i> on rocky shores of Port Nolloth Bay: Figure 2. 5-A.	29°19'3" S, 16° 53'51" E
Kelp forests and other micro-algae at Kleinzee: Figure 2. 5-B.	29°42'31" S, 17°3'18" E
Dense <i>L. pallida</i> in Hondeklip Bay at low tide: Figure 2. 5-C	29°42'31" S, 17°3'17" E
Kelp forests in Jacobs Bay on the west coast: Figure 2. 5-D.	31°48'50" S, 18°14'8" E
Dense kelp forests in Buffels Bay at Cape Point. Here <i>L. pallida</i> forms a subcanopy to <i>E. maxima</i> : Figure 2. 5-E.	34°08'31.6"S, 18°26'02.6"E

Description	Coordinates
Kelp forest at Miller's Point in False Bay at low tide, <i>L. pallida</i> and <i>E. maxima</i> are both present: Figure 2. 5-F.	34°13'54.4"S, 18°28'34.8"E
Kelp forest (<i>E. maxima</i>) in the Betty's Bay Marine Protected Area on the south coast: Figure 2. 5-G.	34°21'43.7"S, 18°54'43.7"E
Two mature drift kelps ((<i>E. maxima</i> and <i>L. pallida</i>) found in the rocky intertidal of Betty's Bay: Figure 2. 5-H.	34°21'48.3"S, 18°54'12.6"E
Kelp forests breaking the surface water at Danger Point near Gansbaai on the south coast: Figure 2. 5-I.	34°13'54.4"S, 18°28'34.8"E



Figure 2. 5 Site-specific photographs taken during field observation (Table 2. 1).

2.2.3.3. Previous maps

Shapefiles of the previous kelp maps produced by Rand (2006) were available as the basis for reference information for this thesis.

2.2.3.4. Expert Knowledge

Two cohorts of specialists were selected to guide the mapping and the ecosystem threat status assessment of the kelp forests of South Africa. The first cohort of experts that were identified to play a supervisory role in this thesis was based on their experience with the concepts of this study. The second group was made up of existing experts working for and/or the South African National Biodiversity Institute's Marine Ecosystem Committee. Expert knowledge, literature review and the kelp forest map were used to subdivide the new kelp forest map into kelp forest ecosystem types as a first step to ecosystem risk assessment which is dealt with in Chapter 3. Expert consensus on the subdivision of the kelp forest map into the three ecosystem types was based on section 2.1.3 and section 1.1.3 of Chapter 1.

2.2.3.5. Google Earth

Visual observation mapping was done at a scale $< 1:3000$ using Google Earth map to assess the quality of both the previous (the offset could be seen easily by overlaying the two sets of data) and new kelp map. Google Earth cloud-free scenes and calm sea days without major oceanographic activities were used from 2006 to 2016 with different periods providing ideal images for different areas. Validation points for accuracy assessment were also generated and classified in Google Earth.

2.2.3.6. Vegetation Map 2018

The National Vegetation Map (see: Dayaram et al.,2017; Harris et al.,2019) of South Africa was analysed using ArcGIS 10.4 for masking all the landmass/vegetation to the lowest low watermark (Skowno et al., 2019). This was done to strictly remove all terrestrial vegetation that might interfere with the mapping of seaweeds and kelp forests. The assumption was that after masking the land portion of South Africa, kelp forests spectral signature will be distinguishable from that of other surrounding surfaces i.e. water. This is feasible because of their absorption patterns of red and blue bands of the visible spectrum and strong reflection of the green and the near infra-red bands (NIR) (Gao, 2009) (see: section 1.6.3 above).

2.2.3.7. Sentinel-2 satellite data

The baseline for the kelp mapping in this project were Sentinel-2 data. Sentinel-2 (A and B) imagery was selected because its orbital path covers the entire study area (Figure 2. 3). Secondly, Sentinel-2 was specifically built to bridge the observation gap existing between marine and terrestrial terrain observations (Malenovský et al., 2012). This satellite platform provides high multispectral and temporal resolutions, required for marine ecosystem types monitoring and research. Sentinel-2 imagery was acquired freely from the European Space Agency (ESA) online hub for Sentinel-2 products at (https://www.esa.int/Our_Activities/Observing_the_Earth/Copernicus/Sentinel-2 and <https://scihub.copernicus.eu/dhus>). The ESA online platform allowed for robust online interaction and enabled experimenting with different band composition and exploration of its imagery under different spectral resolution. This made it simpler to select appropriate and desired input images. The images were downloaded in Level 1C (L1C) format. Sen2cor software was used to transform the images from Level 1C Top of Atmosphere (TOA) reflectance to Level 2A (L2A) Bottom of Atmosphere (BOA) reflectance. The volume of each 100 x 100 km² image tile was 500MB for L1C and 600MB for L2A. Table 1. 3 in Chapter 1 shows the bands 2, 3, 4 and 8 with 10 m resolution utilised in this study for true-colour and vegetation index calculations.

Image acquisition dates were set for winter months (July – August 2016) and imagery with high kelp visibility prioritised. Thirty tiles were downloaded in total covering the entire study area (Figure 2. 3). To reduce the total processing effort, tiles were removed based on literature review, preliminary visual observations and review (Table 2. 2).

Table 2. 2: Locations and names of used images tiles, acquisition time and corresponding mean low water spring tide that corresponds with the image acquisition time.

Sites	Tile grid	Date of acquisition	Mean Low Water Spring (m)	Acquisition time (HHMMSS)
Northern Cape	T33 JXJ	2016/08/10	-0.2	142202
	T33 JXH	2016/07/18	0.5	140340
	T33 JYH	2016/07/18	0.5	140340
	T33 JXG	2016/07/18	0.5	140340
	T33 JYG	2016/08/17	0.6	140340
	T33 JYF	2016/07/18	0.5	140340

Sites	Tile grid	Date of acquisition	Mean Low Water Spring (m)	Acquisition time (HHMMSS)
Western Cape	T34 JBL	2016/07/18	0.5	140340
	T34 HYE	2016/08/17	0.6	140340
	T34 HBK	2016/08/27	0.1	135818
	T34 HBJ	2016/08/27	0.1	135818
	T33 HYD	2016/07/18	-0.1	075642
	T34 HBH	2016/08/04	-0.1	152804
	T34 HYC	2016/05/31	-0.2	075642
	T34 HCH	2016/05/31	-0.2	075642
	T34 HBG	2016/05/31	-0.2	075642
	T34 HCG	2016/05/31	-0.2	075642
	T34 HDH	2016/08/01	0.6	133614
	T34 HDG	2016/08/01	0.6	133614
	T34 HEG	2016/08/01	0.6	133614
	T34 HEH	2016/08/01	0.6	133614
	T34 HFH	2016/07/22	-0.1	134004

2.2.4. Pre-processing of previous data

Shapefiles developed by Rand (2006) were analysed and prepared using ArcGIS and QGIS for use as reference/validation data. The first thing that became apparent regarding these data was an offset caused by a geo-reference error. Measures of this offset were made and an average of 80 m eastward shift was identified. However, this offset could still not be corrected in this thesis. Instead, all the datasets from the previous study were merged to reduce inaccuracy as it had overlapping polygons in some locations. A buffer of 80 m was applied to further reduce the inaccuracy caused by the offset. In addition to the geo-reference error, Anderson et al. (2007) reported gaps in the map produced by Rand and his collaborators. Despite the gaps and the geo-reference error, these were the only datasets available for validation. Therefore, it can be expected that the identified weaknesses of the reference data may have led to some inherited errors in this study.

2.2.4.1. Atmospheric correction

Radiometric, geometric correction, ortho-rectification and spatial registration on a global reference system are provided in Sentinel-2 products (Drusch et al., 2012). The downloaded Level 1C images were provided as top of atmosphere (TOA) reflectance, with sub-pixel, multi-spectral and multi-date registration. Each tile consisted of 100 x 100 km ortho-images in UTM/WGS84 projection (Drusch et al., 2012). Sen2cor software and the SNAP toolbox were used to convert the Top of atmosphere (TOA) reflectance to Level 2A Bottom of Atmosphere (BOA) reflectance using the ATCOR algorithm (Atmospheric/Topographic Correction for Satellite Imagery) (Drusch et al., 2012).

2.3. Classification

2.3.1. Normalized Difference Vegetation Index

As explained in section 1.2.4 the spectral properties of water and vegetation (including kelp) in the NIR and the Red bands are fundamentally different. These differences can be extracted from the S-2 images by calculating the NDVI to assess the distribution of the kelp forests of South Africa. Arc-Calculator, a tool in ArcGIS was used for the NDVI computation for all tiles listed in Table 2. 2. Based on the examination of the spectral signatures of kelp and water respectively, the hypothesis is that vegetation, including kelp (Figure 2. 6 and Figure 2. 7), has a much higher NIR signature than water. Subsequently, the NDVI for kelp must be much higher than the NDVI of water as well and in a trial and error process, NDVI thresholds were tested to separate both classes.

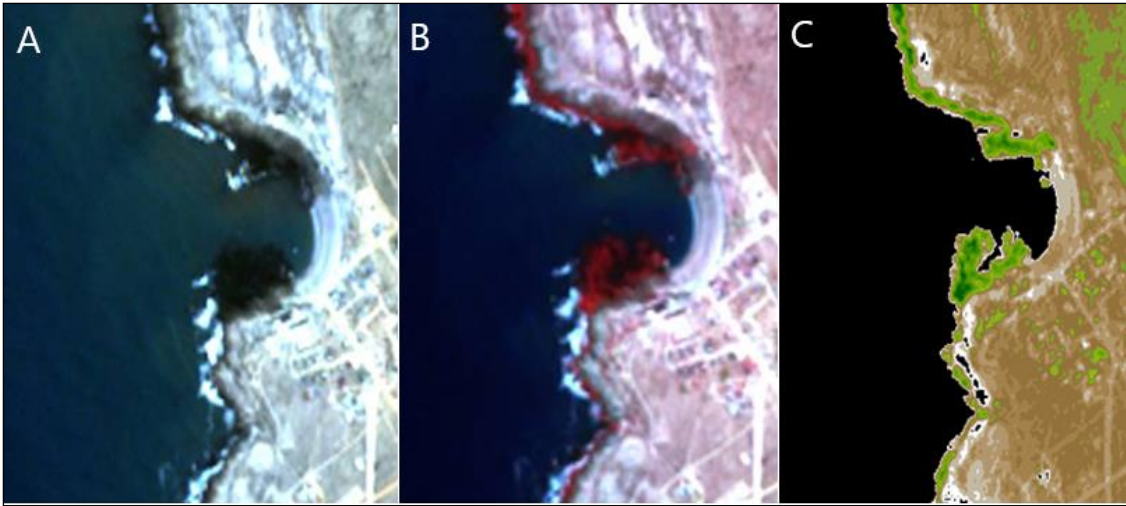


Figure 2. 6: Different S-2 band compositions of Hondeklip Bay. A: true-colour image with kelps appearing as indistinct dark patches, B: infrared bands with kelps appearing as red patches C: NDVI image with vegetation (including kelp) shown in green.

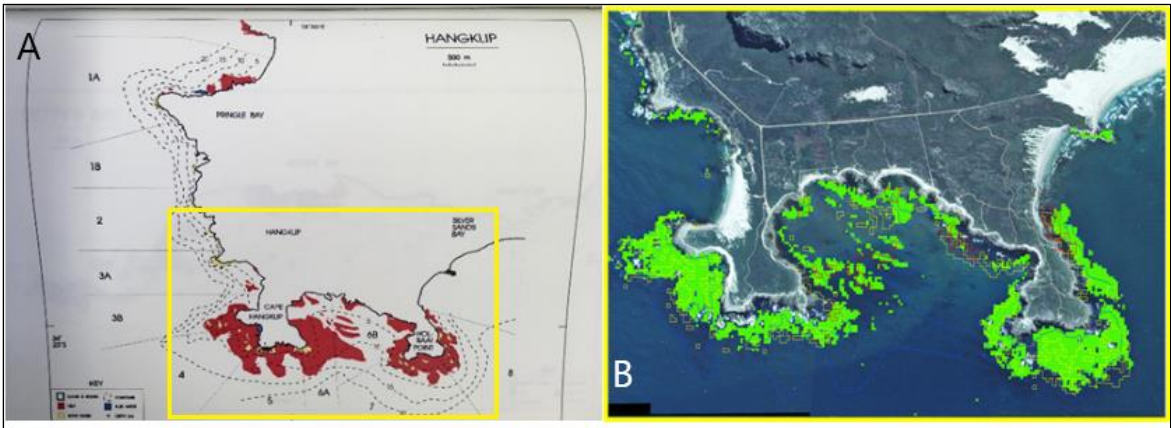


Figure 2. 7: Cape Hangklip (CHK) kelp map. Reference data according to Tarr (1993) (A) showing hand-drawn kelp distribution in red, white water in yellow and bathymetry (dotted lines). Preliminary NDVI threshold derived kelp map (green) produced from S-2 imagery in this study (B) overlaid on a Google Earth base map.

2.3.2. Supervised S-2 classification

The procedure of supervised maximum likelihood classification (MLC) is well documented (Lillesand, Kiefer and Chipman, 2004). The MLC process was done using ArcGIS 10.4. During the MLC process, the probability of pixels belonging to each of the developed training classes is calculated (Lillesand, Kiefer and Chipman, 2004). Based on expert consensus, training data were selected from reliable sources covering different environments (visual surveys, Google Earth satellite maps and expert derived reference maps) and used as training polygons. The sample locations were therefore confined to specific locations with good reference data, i.e. Dassen Island, Robben Island, Kommetjie and Cape Hangklip. The hypothesis was that

choosing training samples with a broad range of environmental conditions will be representative of the long and variable coastline of South Africa in which kelp forests occur and will thus be useful when combined with the NDVI map.

Therefore, the procedure of supervised MLC focused on developing training polygons used as training data to categorise all pixels into two classes, class 0 (kelp forest) and class 1 (non-kelp/water). The two classes were derived from 9 classes that resulted from the MLC that were selected as the most suitable classification after trial and error with a different iteration of classes. The following steps were undertaken:

1. Reference data, including visual observation, Google Earth imagery, expert knowledge and the previous kelp maps (Figure 2. 4) were collated.
2. From these data, training polygons were generated for locations with known kelp forest and non-kelp occurrence.
3. The training polygons were deliberately chosen as areas of high confidence in the previous maps and underwent expert validation for representativity of ground reference (reality on-field) kelp forest areas until satisfactory results were obtained (Lillesand, Kiefer and Chipman, 2004).
4. The maximum likelihood classification was run using the training polygons and the 4-band multispectral S-2 L2A data as input.
5. The results discerned the vegetation into several vegetation classes (Figure 2. 15). The results were re-coded into the two target classes “kelp” and “no kelp”.
6. The outcome was validated by experts to test the accuracy of the MLC results.
7. The resulting two-class map was merged with the NDVI calculated two-class map.

2.4. Post-processing

The Arc toolbox was used to post-process the kelp forest map. The map was visually processed (smoothing, buffering and union) to create continuous smooth polygons by transforming the kelp polygons from hexagonal shapes to smooth rounded shapes (Lillesand, Kiefer and Chipman, 2004). Noise and pixilated areas were manually removed using a majority filter where any polygon with a size less than 610 m² (expert derived through trial and error) was assumed to be too small to represent kelp beds and thus removed. A buffer of 2500 m was applied to the kelp polygons along with the entire kelp extent and a box whisker plot was applied to represent the NDVI threshold for AOI.

2.4.1. Accuracy assessment

2.4.1.1. Expert assessment

During an expert review session, it was decided to exclude all NDVI results east of Mossel Bay. This is because there is no baseline map, validation data and reference maps of any kelp occurring beyond this point and the fact that the NDVI threshold method does not discern between seaweed species including other coastal vegetation. Therefore, all non-applicable NDVI results east of Mossel Bay were manually deleted. Validation was conducted for Northern Cape and Western Cape provinces using visual observation, Google Earth imagery, ground-truthing data, expert knowledge, reference maps data and scientific published studies referred to as reference data above (Figure 2. 4). The validation process entailed expert review by panning and zooming (1:20 m) the entire extent covered by the new kelp polygons map to check for anomalies.

2.4.1.2. Confusion matrix

To determine the statistical proportion of agreement between the produced/new kelp map and the validation data at certain locations, selected reference data were used (Lillesand, Kiefer and Chipman, 2004; Gao, 2009). A set of random points were created for all the selected AIO using Google Earth. Random generation using ArcGIS proved to be complicated as the generated points were biased towards the no kelp class. Hence a smaller dataset of points was generated and systematically generated over Cape Hangklip, Cape Point, Robben Island and Dassen Island. The points were classified into water/no kelp and kelp classes as developed for the MLC (Gao, 2009). The points were exported as KML files to ArcGIS and used for validation. Cloud free Google Earth images were selected where only kelp forests could be visually discerned. The validation points were overlaid over the classified map results to assess the proportion of agreement between the validation data and the map accuracy determined using a confusion matrix approach.

A confusion matrix is a prominent tool for representing these calculations and is used across different fields of remote sensing (Gao, 2009). The confusion matrix provides producers and user accuracy for the classification map (Lillesand, Kiefer and Chipman, 2004) and a kappa coefficient (Gao, 2009) can also be calculated from the error or confusion matrix.

2.5. Results

2.5.1. Normalized Difference Vegetation Index thresholds for various sites

The NDVI values for the Dassen Island (DASa) kelp class are on average 0.4, with a minimum of 0.24 and a maximum of 0.49. For the Dassen Island (DASb) no kelp class the NDVI average is -0.31 (Figure 2. 9). The NDVI results for Robben Island are average (-0.12; -0.33), minimum (-0.25; -0.34) and maximum (0.13; -0.31) for kelp (ROBa) and no-kelp (ROBb) respectively and are shown in the box and whisker plot in Figure 2. 10. The NDVI results (Figure 2. 11) for Kommetjie are average (0.29; 0.15), minimum (-0.12; -0.25) and maximum (0.35; 0.14) for kelp (KOMa) and no-kelp (KOMb) respectively and are represented in a line diagram in Figure 2. 8. The NDVI results for Cape Hangklip are average (0.14; 0.06), minimum (-0.19; -0.14) and maximum (0.47; 0.13) for kelp (CHKa) and no-kelp (CHKb) respectively and are depicted in the box and whisker plot in Figure 2.9 to 2.12.

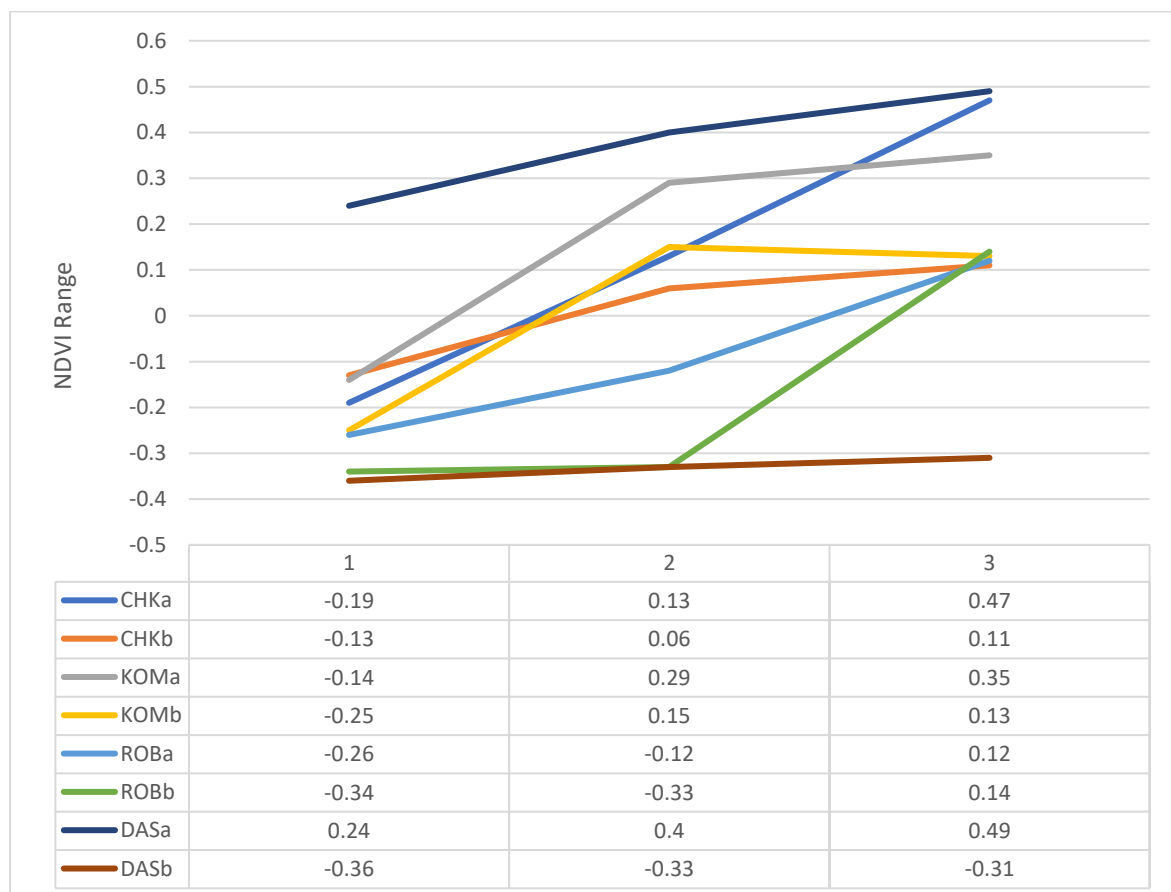


Figure 2. 8: Normalised Difference Vegetation Index (Y-axis) range for “kelp class (a)” and “no kelp class (b)” at locations Cape Hangklip (CHK), Kommetjie (KOM), Robben Island (ROB) and Dassen Island (DAS). The X-axis denotes lower quartile (1), median (2) and upper quartile (3) ranges.

The NDVI results vary across different areas for the two categories developed using maximum likelihood rules of classification. To better represent and explain the different NDVI values for the various locations, box and whisker plots were generated using R version 3.5.1 software. The results for four locations are represented in Figures 2.9 to 2.12. The upper quartile (75%), median (50%) and lower quartile (25%) are depicted as the box (in Figures 2.9 to 2.12). The maximum and the minimum are the “whiskers” and the dots are outliers. Figures 2.9 to 2.12 present the different environmental conditions using selected AOI and focus on how submerged kelp forest, surface buoyant and water turbidity affected the NDVI values.

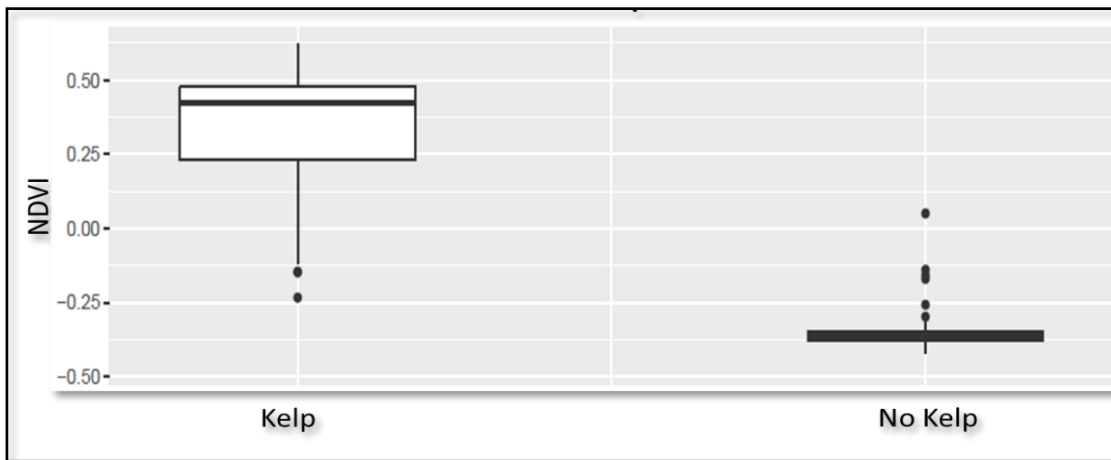


Figure 2. 9: Box plot of NDVI ranges for kelp and no kelp at Dassen Island. The medians for the two classes occur in two extremes of the NDVI range.

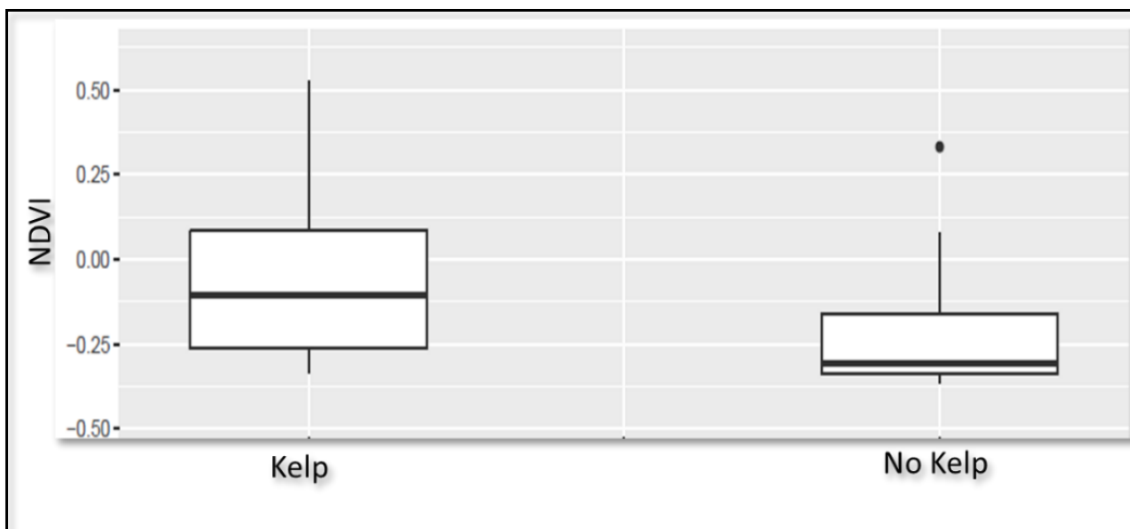


Figure 2. 10: Box plot of NDVI ranges for kelp and no kelp classes at Robben Island. The lower and the upper quartile of the two classes are overlapping.

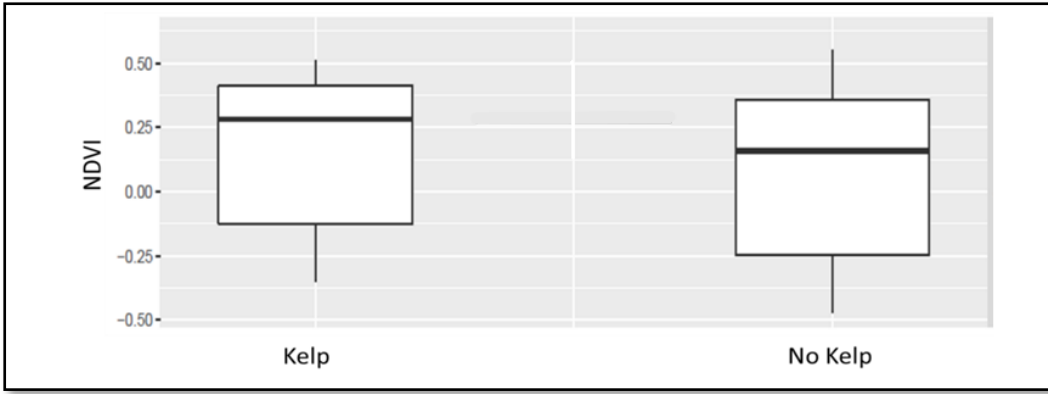


Figure 2. 11: Box plot of NDVI ranges for kelp and no kelp classes at Kommetjie. The median and upper quartile of the kelp class overlaps with the upper, median and lower quartile of the no kelp class.

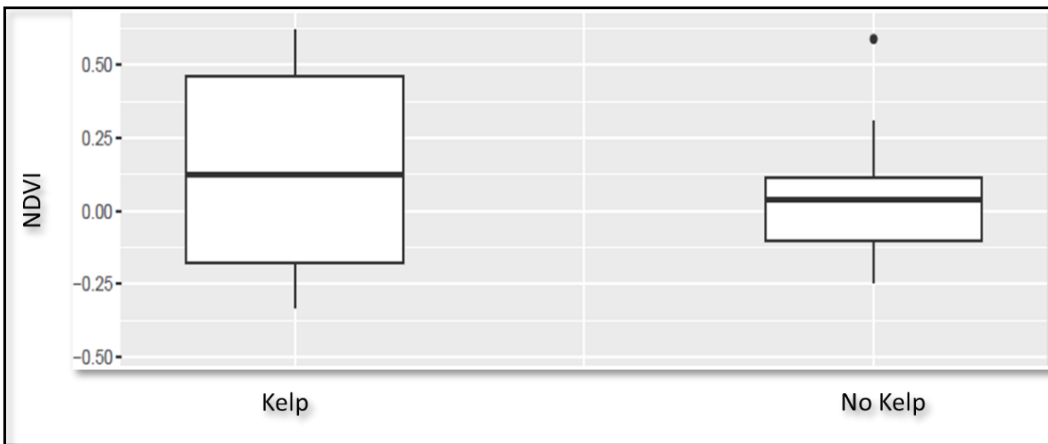


Figure 2. 12: Box plot of NDVI ranges for kelp and no kelp classes at Cape Hangklip.

The NDVI threshold of -0.2 yielded satisfactory results when reviewed by the expert panel. The expert review focused on identified areas of interest (AOI: Figure 2.9 to 2.12), namely Cape Hangklip, Cape Point, Kommetjie, Robben Island and Dassen Island. All values less than -0.2 and the value 0 were categorised as no kelp/class 1 and values greater or equals to -0.2 excluding 0 as kelp/class 0.

Figure 2. 13 shows 10 classifications of NDVI range for three AOI. The threshold -0.2 to 1 represents the kelp class. This includes both submerged and surface buoyant fronds of kelp forests. The NDVI range from -1 to -0.2 including 0 represents the non-kelp class/water. The colour NDVI range of the kelp includes red and yellow and the NDVI range of the non-kelp is represented as shades of green. White water was often visible in dark green and was concentrated near the coastline.

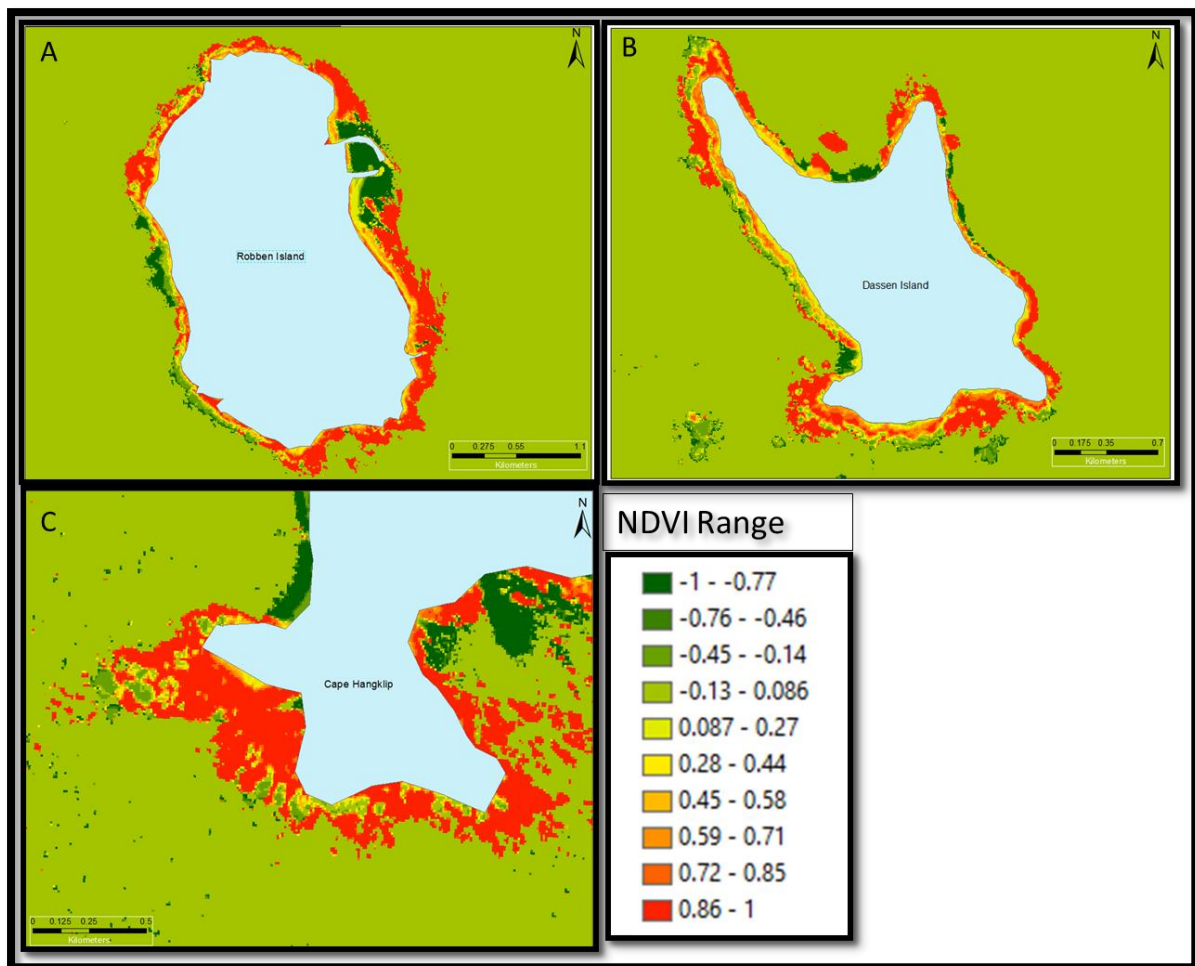


Figure 2. 13: Normalised Difference Vegetation Index (NDVI) map of South African kelp forest using thresholds (-0.2). The AOI include A (Robben Island), B (Dassen Island) and C (Cape Hangklip).

2.5.2. Supervised classification

Using the ground-truth data, expert knowledge and data from the previous maps, a supervised maximum likelihood classification (Figure 2. 14) was conducted. The assumption was that the different AOIs were representative of the coastline and the result merged with the NDVI kelp results map.

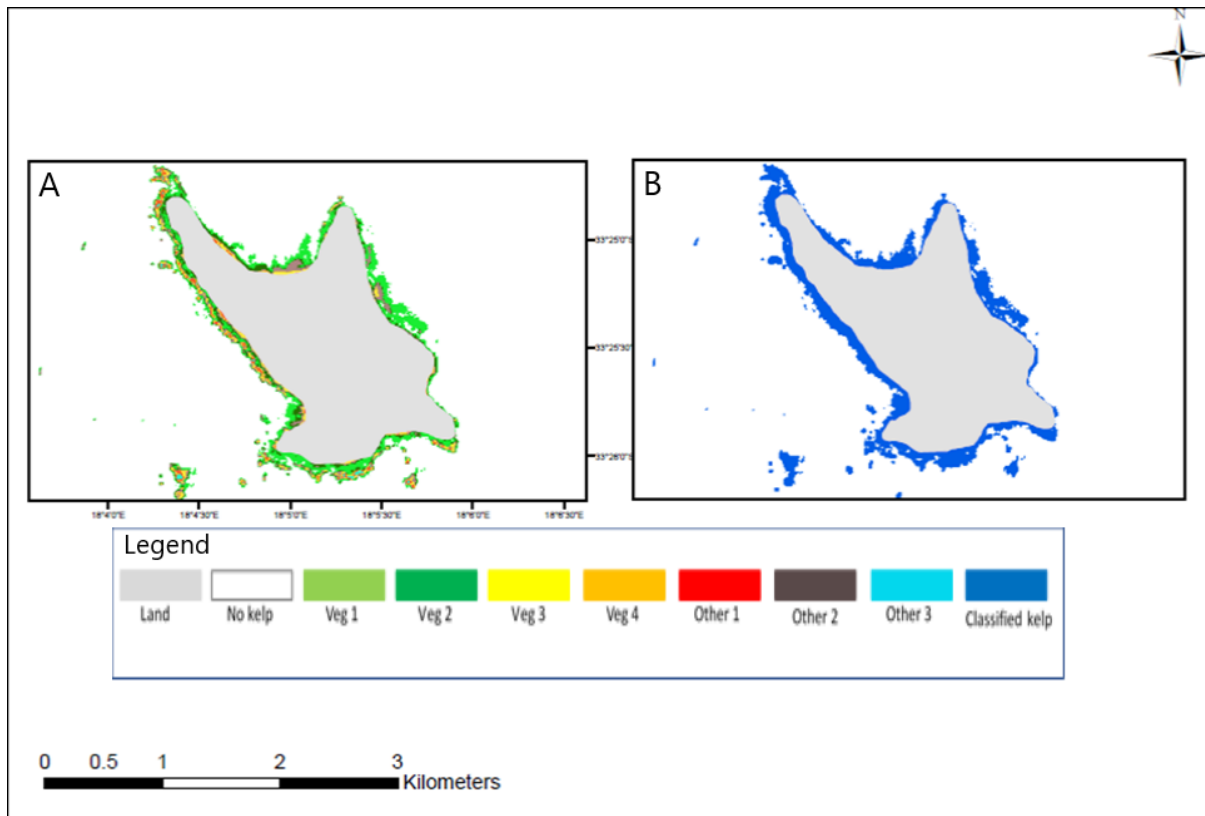


Figure 2. 14: Dassen Island: A: the original MLC produced by this study and B: the re-coded MLC product.

2.5.3. Comparison of the previous map and the new map

In addition to the mapping gaps reported by Anderson et al. (2007), a geospatial offset was identified in the previous maps as mentioned in section 2.2.3. The kelp polygons in purple overlap with the grey areas that represent the land Figure 2. 15. This limited the application of the reference data and affected the use of the previous map data as training data for the MLC and as such influenced the accuracy assessment of the mapping results. The areas with comprehensive data on the previous maps as can be seen in Figure 2. 15 are all included in the current map. Further, the new map identified many more areas with kelp forest compared to the previous map. One of the reasons for this is that the previous maps only mapped the outline

of the surface reaching shallow kelp forests. The new map, on the other hand, mapped both submerged and surface-reaching kelp forest.

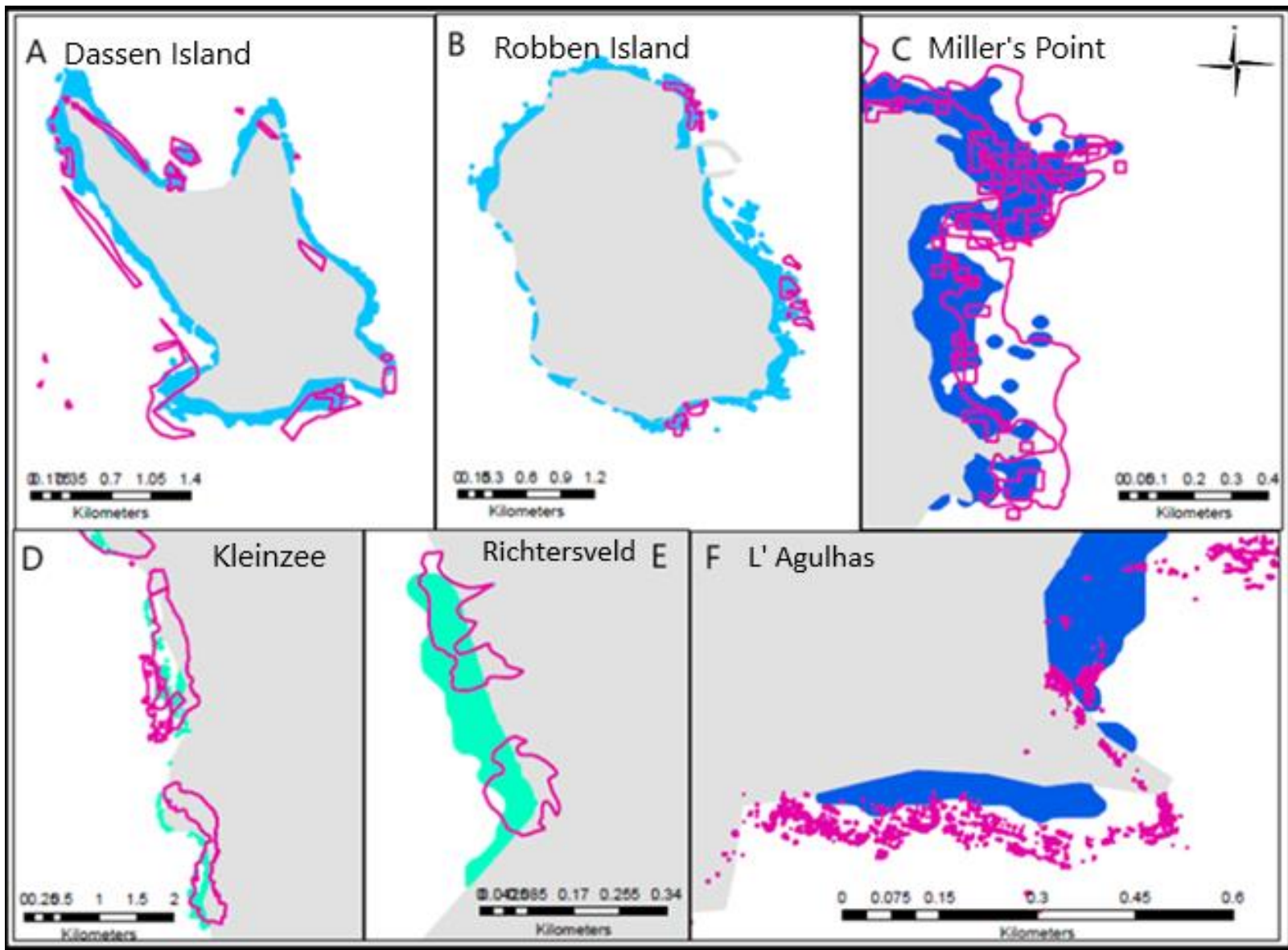


Figure 2. 15: Comparison of the previous kelp map and the new map (fusion of the NDVI and MLC). Purple polygons represent the previous kelp map (Anderson et al., 2007) and the shades of blue are the three newly mapped South African kelp forest ecosystem types. A: Dassen Island, B: Robben Island, C: Miller's Point, D: Kleinzee, E: Richtersveld and F: L'Agulhas.

2.5.4. Division of three kelp forest ecosystem maps

The ability of S-2 to penetrate up to 20 m (Johnson, 2015) made it possible to detect both shallow and deeper kelp forest clearer in areas with clear and calm waters e.g. Dassen Island (Figure 2. 15-A). This recent kelp forest map (Figure 2. 16) was categorised into three types namely, Namaqua Kelp Forest, Cape Kelp Forest and Agulhas Kelp Forest. The areas of subdivision (Donkin Bay and Cape Point) of the three kelp forest ecosystem types are also presented in the figure below.

This division was a first step for the first-ever South African kelp forest ecosystem threat status assessment (detailed in Chapter 3). This process entailed extensive expert engagements, literature review of the South African kelp forest ecosystem types biogeography and alignment with the National Map of South Africa marine ecosystems (Sink et al., 2012). The Namaqua Kelp Forest extends from Orange River to Donkin Bay, the kelp forest that falls within in the Namibian territory (Omoregie et al., 2010; Rothman et al., 2017) is acknowledged but excluded from this map. The Cape Kelp Forest occurs between Donkin Bay and Cape Point. The Agulhas Kelp Forest extends from Cape Point into the Cape Peninsula referred to as a transition zone (Blamey et al., 2010), Cape Agulhas and include the kelp recently established at Koppie Allen De Hoop (Bolton et al., 2012) along the south coast (Figure 2. 16 and Figure 3. 4).

2.5.5. Comprehensive map of South African kelp forests from Sentinel-2

The results of the merge from the NDVI thresholding and the MLC are represented in Figure 2. 16. The map depicts kelp extending ca. 1300 km along South Africa's west coast and parts of the south-west coast. It includes kelp forests in regions that were not included in previous maps. For the first time, this current map includes the recently established kelp *E. maxima* in De Hoop Nature Reserve which shifted ca. 70 km from its previous limit (Bolton *et al.*, 2012). Also, it covers about 180 km further east of the southernmost part limit of the previous 900 km kelp map by Anderson et al. (2007). This 180 km stretch of the coast includes kelp forests found in L'Agulhas, Arniston, Overberg DC (Ski haven Twp.), Cape Infanta, Still Bay, Gouritz and Vleesbaai along the south coast.

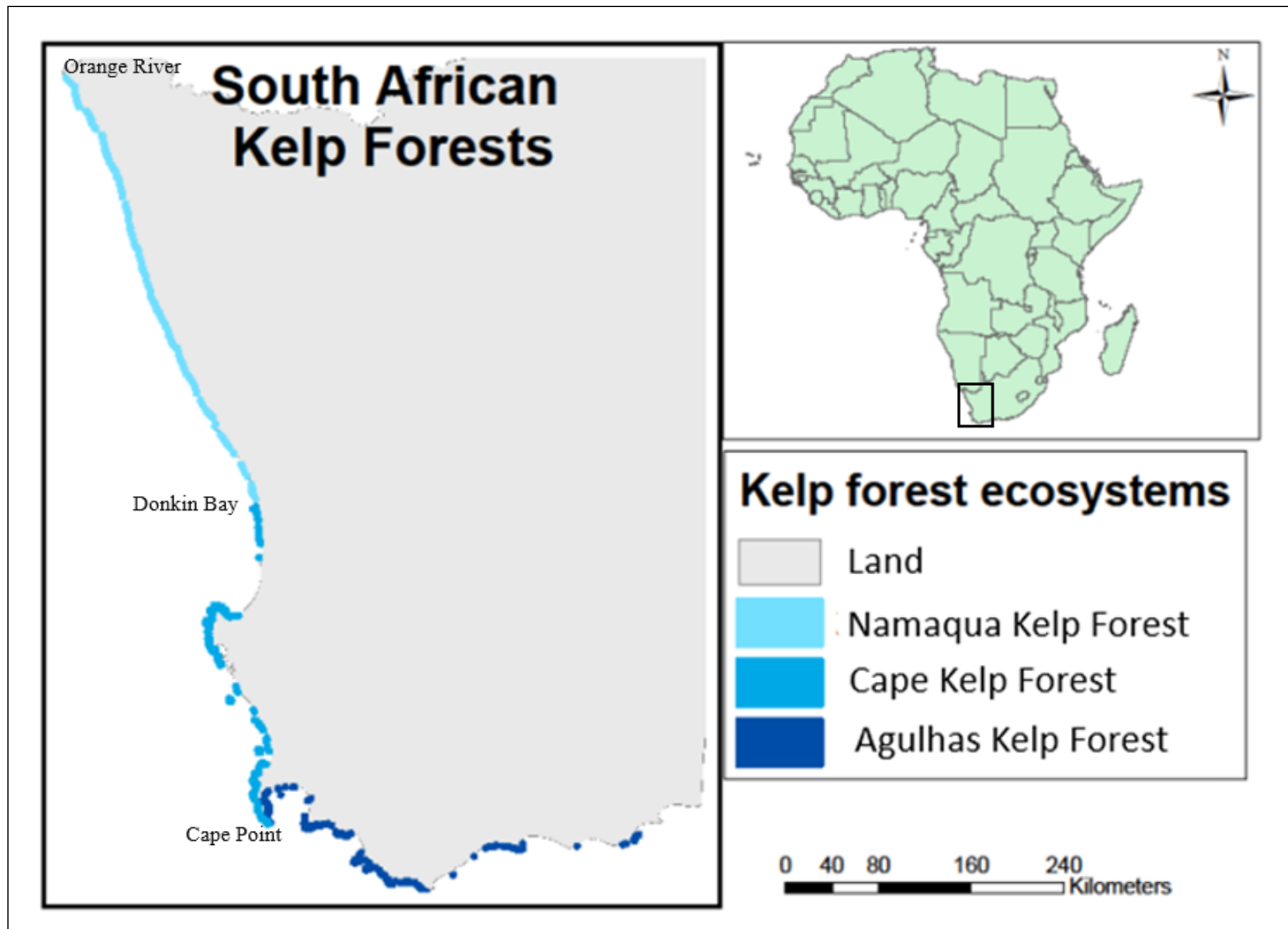


Figure 2. 16: Comprehensive South African kelp forest map, indicating distribution and bioregional subtypes.

2.5.6. Accuracy assessment

Table 2. 3 shows the results of the accuracy assessment of the kelp forest classification using NDVI thresholds. The matrix shows that 44 of the 161 validation points for kelp were classified correctly as kelp (27 %). Of the 1077 validation points for no-kelp, 895 points were classified correctly (83%). This leads to an overall accuracy of 75.85%, overall producers' accuracy of 83.10% and an overall users' accuracy of 88.44%. However, the kappa coefficient is very low (0.0089 from a possible range of -1 (no agreement) to 1 (full agreement)). The reasons for this somewhat contrasting range of results are discussed in the next section.

Table 2. 3: Confusion matrix shows the accuracy assessment of the new map (classified data) against the previous maps (validation data).

Validation data				
		Kelp	No kelp	Total
Classified data	Kelp	44	82	226
	No kelp	117	895	1012
	Total	161	1077	1238
Producers accuracy: 83.10%				
Users accuracy: 88.44%				
Overall accuracy: 75.85%				
Kappa Value: 0.0890				

2.6. Discussion

This study used two approaches for developing a South African kelp forest map, an NDVI thresholding approach and MLC classification. The result was a continuous map of kelp forest distribution and type along South Africa's coast, along a gradient of different environmental conditions, different tidal ranges and influenced by two distinct ocean currents, i.e. the Benguela and Agulhas. While the final map indicates the general success of this approach, there are several aspects of the process and the results that need further discussion. These are expanded on in the following sections.

2.6.1. Normalised Difference Vegetation Index

Kelp forests behave spectrally much like terrestrial vegetation, with low reflectance in the red and blue bands as they absorb red and blue for photosynthesis and reflect NIR (Cho et al., 2008) (see: Figure 1. 2). This study and others use the NDVI approach because it is straightforward and versatile (Ma et al., 2008; Zhang et al., 2011; Bhandari et al., 2012; Roy et al., 2016). When applied in aquatic environments, with constant vegetation density/activity, NDVI decreases with increasing water depths (Cho et al., 2008). Further aquatic environmental characteristics influence the NDVI such as turbidity (Casal et al., 2011), therefore leading to variable and overlapping NDVI results (Ma et al., 2008). The results of the varying NDVI ranges between the different regions of this thesis can be explained by the latter circumstances.

Dassen Island represents the best conditions for the application of this technique, as there was the most distinct difference between the two classes “kelp” and “no kelp” in this area. Perhaps the most challenging environment for this technique was Kommetjie where there are huge overlaps between NDVI ranges of the “kelp” and “no kelp” classes. This was attributed to by the fact that the west coast of South Africa is directly exposed to prevailing (south-westerly) swells and even small waves break as they enter the shallow edge of kelp beds causing white waters and turbidity (Anderson et al., 2007; Griffiths et al., 2010). However, this could be greatly improved by generating comprehensive reference data for all the different subareas of the coast.

Through a trial and error approach, for each pixel’s calculated NDVI, a threshold of -0.2 was set to detect both shallow and deeper kelp forest canopies. This shows that this technique has an added advantage compared to other studies that detect surface protruding seaweeds communities (e.g. (Anderson et al., 2007; Cavanaugh et al., 2011) and seagrass communities (Fornes et al., 2006; Pasqualini et al., 2005). The NDVI threshold of 0.05 set by Nijland, Reshitnyk and Rubidge, (2019) and Gatc et al. (2016) falls within the range of the detected kelp in our study. The calculated NDVI range 0.6 - 0.65 by Cho et al. (2008) and the threshold of 0.1 by Kiage and Walker (2009), overlap with the set NDVI threshold range in this current study, as shown in Figure 2. 11. The recommendation of this current study is to test and develop thresholds that are specific to the different region of the South African coastline.

The findings of this study, show the high NDVI values (vegetation), negative vegetation (water), this is similar to what Gatc et al. (2016) reported, that the NDVI range below zero is the absence of vegetation and values greater than 0.1 are typical of quality and quantity of

vegetation. The analysis of this current study uses a threshold below 0 for kelp because these pixels reflect the presence of both water and kelp.

Therefore, this conventional NDVI approach can be used for future mapping and monitoring of kelp forest ecosystems in similar environments to those found in South Africa such as Namibia. Besides, the methods can be modified to map different environments and other coastal ecosystem types with vegetation. Having successfully interpreted the NDVI values of submerged kelp forests, this study adds and perhaps changes the entire conversation around NDVI thresholds for submerged aquatic vegetation (SAV) (Cho et al., 2008). However, further experiments of different thresholds for SAV areas with better validation data are strongly recommended.

To strengthen the incomplete ground-truth data, Google Earth was crucial for visual observation and in increasing accuracy of the ground reference data. To strengthen the current map product, understanding that not all sites across the entire South African coastline can be visited, expert knowledge was leveraged and used to assess the kelp forest map and make comments on areas they are most familiar with. This process was conducted twice, and the first instance led to the expansion of the NDVI threshold to include many subtidal kelp beds i.e. off the coast of Hermanus, Cape Point, Kommetjie and Dassen Island which also allowed for a greater application of the capacity of Sentinel-2 to penetrate up to 20 m into calm and clear waters (Johnson, 2015). A recent study in the Central Coast of British Columbia. employed a similar method for mapping canopy-forming kelp species and set an NDVI threshold of 0.05 (Nijland, Reshitnyk and Rubidge, 2019) and also used a trial and error approach.

2.6.2. Accuracy assessment

Although the error matrix is quite a prominent tool, using it to assess accuracy in this study resulted in several challenges. The good Overall, Users and Producers accuracies of > 75% were in stark contrast to the very weak Kappa index of 0.0890. This statistic measure (Kappa index) serves as an indicator for the extent to which the percentage correct values are due to the true agreement or because of chance agreement. True agreement approaches kappa values of 1 and chance agreement approaches 0. Table 2.4 shows present how different Kappa values can be interpreted (Lillesand, Kiefer and Chipman, 2004). The Kappa value of 0.0890 in this thesis shows the confusion matrix results are not a result of full disagreement nor are they by chance or random agreement but of slight agreement (Table 2.4).

Table 2.4: Interpretation of kappa value range, modified from Lillesand, Kiefer and Chipman (2004) and Campbell (1996).

Kappa value	Interpretation
-1	Full disagreement
0.00	Random agreement
0.00-0.2	Slight agreement
0.21-0.4	Fair agreement
0.41-0.6	Moderate agreement
0.61-0.8	Substantial agreement
0.81-1	Almost perfect agreement

One explanation for the challenges is that the confusion matrix approach was sensitive to the size of the samples or classes. This means that the accuracy assessment results are skewed to the highly correct “no kelp” class for which about 6.7 times as many reference points were available, most of which were classified correctly. In contrast, only 161 points were available for the kelp class, most of which were classified wrongly. This impacted the Kappa index, but not the other accuracy parameters, i.e. the kappa index is low because of the high number of misclassified validation points for the kelp class (Gao, 2009). These can most likely be attributed to the shortage of validation points for kelp, which was caused by not having enough validation data i.e. limited studies that map South African kelp, and ground-truthing and the reported offset in the previous maps that were used as a reference for the S-2 classification.

The accuracy results of this study are similar to several other studies for coastal ecosystems (e.g. Pasqualini et al., 2005; Sagawa et al., 2008; Casal et al., 2011; Sagawa et al., 2012; Nijland, Reshitnyk and Rubidge, 2019). The recent study by Nijland, Reshitnyk and Rubidge (2019) on mapping canopy-forming kelp using Landsat imagery generated an overall accuracy of 80%. In that study, archives of Landsat imagery were used to calculate NDVI and extract kelp extent. Another study employed a spectroradiometer to measure the spectral signature of different seaweed species using IKONOS imagery (Sagawa et al., 2012). Their reference data constituted in in-situ data (boat-based and aquatic video camera) and the overall accuracy reached 97.6%. Although adopting some of the methods of the latter study are recommended, they are currently prohibitively expensive. The technique of this study can be applied for future maps of South African kelp forests and is suitable for application for mapping other seaweed communities, seaweed habitats and other coastal vegetation ecosystems.

2.6.3. Impacts of post-processing on mapping accuracy

The smoothing procedure used for the kelp polygons was done in ArcGIS 10.4 and polygons with an area less than 610 m² were manually deleted to correct for pixelation in the imagery. Some of the kelp polygons were deleted along with the pixels. To correct for this loss a buffer was applied, which created a high kelp probability. The accuracy assessment results are also anticipated to have been affected by this kelp forest area.

For a long time, proper management of the ecologically, socially and economically important kelp forests of South Africa has been impeded by the lack of comprehensive and contemporary kelp forests extent maps. This study has addressed this limitation. Despite the complications associated with validation data and remoteness of marine environments, this study serves as a milestone towards exploring assessments of South African kelp forest ecosystem conditions and threat status. In this study, a trial and error approach for extracting kelp NDVI thresholds from high spectral and temporal Sentinel-2 imagery was introduced. The contemporary map has provided a new perspective on the distribution of South African kelp forests. As described in the sections above, there has been a consensus on shifting boundaries of seaweed species caused by global warming. The kelp forest map along the south coast of South Africa captures the kelp that has shifted eastwards assumed to be a response to climate change. This national kelp map is important for South African kelp forest ecosystems monitoring that can be used to better the knowledge of how these ecosystem concordances respond to various aspects of global climate change that affect kelp forests. The results presented in this chapter are in line with international initiatives with goals of better understanding how kelp ecosystems react to instigated changes both from natural and anthropogenic factors e.g. of KEEN (<http://www.kelpecosystems.org>), Floating Forests (<https://off>) and Monitoring sites 1000 (Terada et al., 2019).

Chapter 3: Assessing the ecosystem threat status of South African kelp forests

“Not only is the science incomplete, but the [eco]system itself is a moving target, evolving because of the impact of management and the progressive expansion of the scale of human influences on the planet” (Holling 1995) (cited in (Cury et al., 2003).

3.1. Introduction

Across all spheres of earth, plants and animals occur as parts of spatially explicit units and never in isolation (Tait, 1981), instead they form ecosystems which are comprised of abiotic and biotic components. Marine ecosystems include a myriad of organisms from the most cryptic to the most conspicuous ranging from viruses and bacteria, phytoplankton, macroalgae and zooplankton to the invertebrates, fish, birds and mammals that characterise marine ecosystems (Branch, 2008). Kelp forests are comprised of two dimensions based on their main components, 1. the reef which provides a substrate for kelp attachment (physical) and 2. the dense forest created by the stands of kelp (biotic) (Steneck et al., 2002; Blamey and Bolton, 2018). Like many other marine ecosystems, kelp forests are under pressure from the warming of global oceans over the past century. Additionally, there is increasing evidence globally that several marine ecosystems e.g. coral reefs and seagrass habitats and kelp forests are undergoing degradation (Keith et al., 2013; Hedley et al., 2016; Topouzelis et al., 2018) from human actions. Anthropogenic pressures are threatening the diversity and suite of ecosystem services kelp forest ecosystems provide (Wernberg and Filbee-Dexter, 2019; Smale et al., 2019).

Specifically, there is a global consensus that kelp forests are generally declining and/or shifting (Raybaud et al., 2013; Wernberg et al., 2016; Pfister et al., 2017; Terada et al., 2019; Wernberg and Filbee-Dexter, 2019) although in some areas their expansion has been reported (Bolton et al., 2012; Marks et al., 2015; Franco et al., 2018). Kelp forest ecosystems exhibit rapid shifts between communities or “ecosystem states” in response to anthropogenic mediated impacts (Johnson et al., 2011; Marks et al., 2015; Filbee-Dexter and Wernberg, 2018). The most common example is the overgrazing by sea urchins, which over few years can deforest many kilometers of kelp forests, forming alternative barren states i.e. a state where areas originally dominated by kelp forests are replaced by urchin barrens (Filbee-Dexter and Scheibling, 2017). These rapid shifts have been observed in the denudation of large areas of giant kelp forests *Laminaria digitata* and *Saccharina* by urchins *Strongylocentrotus droebachiensis* in Nova Scotia and *Saccharina latissima* by urchin *Strongylocentrotus droebachiensis* in Maine

(Steneck, 1997; Steneck et al., 2002; Filbee-Dexter and Scheibling, 2014). Urchins *Centrostephanus rodgersii* and *Heliocidaris erythrogramma* were reported to denude *Ecklonia radiata* and *Phyllospora comosa* in Tasmania (Edyvane, 2003; Ling and Johnson, 2009; Johnson et al., 2011) and the urchin (*S. polycanthus*) barrens were reported as a result of denuded *Alaria fistulosa* and *Laminaria spp.* in the Aleutian Islands (Steneck et al., 2002) and in California *Macrocystis pyrifera* kelp forests transition into alternative urchin (*S. franciscanus*) barren states (Dayton et al., 1998).

In South Africa, large scale denudation of kelp forests by urchins has not been reported, but small patches (up to tens of m²) of the urchin *Paranichinus angulosus* do occur which can restrict the increase in *E. maxima* above a certain threshold in urchin density (Morris and Blamey, 2018). Regime shifts such as these are usually a result of human-mediated trophic cascades i.e. where human actions have removed top predators such as rock lobsters, sea otters and reef fish, leaving their grazing prey unchecked (Mann and Breen, 1972; Lorentsen, Sjøtun and Grémillet, 2010; Blamey and Branch, 2012; Ling et al., 2014; Wernberg et al., 2016; Pessarrodona, Foggo and Smale, 2019).

South African kelp forests are among the few small, high-value ecosystems whose planning, management and restoration efforts should be prioritised (Skowno et al., 2019). This chapter pioneers the assessment of ecological degradation and ecosystem threat status of these kelp forest ecosystems to provide insight and evidence for future assessments, sustainable use and management. A sound understanding of the hazards associated with anthropogenic pressures is instrumental for influencing decisions in conservation planning and management actions through the formulation of marine ecosystem condition assessments and assigning ecosystem threat statuses to ecosystems (Sink et al., 2012; Micheli et al., 2013; Bland et al., 2017a; 2018).

3.2. Ecosystem threat status

As a consequence of the complex multiple and interacting anthropogenic processes that can threaten marine ecosystems, the state of marine ecosystems is challenging to assess (Halpern et al., 2007). Threatening processes such as climate change, coastal development, overexploitation and pollution, can transform, deplete and redistribute key ecosystems and species (Smale et al., 2019). Additionally, understanding of these threatening processes is key (Selkoe et al., 2009) to mitigate ecosystem collapse (described below) or ecosystem shifts from one stable state to a completely different state (Blamey et al., 2010; Ling et al., 2015; Wernberg

et al., 2015; Filbee-Dexter and Wernberg, 2018; Bland et al., 2017a, 2018) while still maintaining the established ecosystem values from kelp forest ecosystems and reefs (Bennett et al., 2016; Blamey and Bolton, 2018). This is also a key step for shifting from species-based assessments towards ecosystem-based assessments that aim to preserve all elements of biodiversity and thus, avoid shifting ecosystems (Nicholson et al., 2009; Keith et al., 2013; Bland et al., 2017a; b).

Naturally, kelp forest density may fluctuate in response to the complex interactions of physical factors i.e. oscillate between different states, with short-lived stages of kelp-free or barren patches that are eventually replaced by “natural” (non-barren) kelp dominated states (Steneck et al., 2002). However, the effects of human activities can destabilise these natural fluctuations and erode the resilience and resistance of kelp forests to various pressures (Connell et al., 2008; Wernberg and Filbee-Dexter, 2019).

In 2008, the World Conservation Congress responded to the growing need of adopting ecosystem-based management by endorsing the development of standards for assessing the status of ecosystems (Nicholson, Keith and Wilcove, 2009). Some of the protocols of risk assessments used to assess ecosystems include (1) the National Biodiversity Assessment (NBA) for South Africa (Sink et al., 2012, Skowno et al., 2019), (2) the International Union for the Conservation of Nature Red List of Ecosystems (IUCN RLE) (Rodríguez et al., 2015) (www.iucnredlistofecosystems.org), (3) the Australian guidelines for assessing cumulative impacts and risk to the Great Barrier Reef (Dunstan et al., 2019) and 4. IndiSeas a project that was created to evaluate the status of exploited marine ecosystems based on established indicators (Coll et al., 2016; Lockerbie, Shannon and Jarre, 2016). These different risk assessment models use distinct symptoms such as the distribution and ecological function of ecosystems and other variables to assess the threat status of ecosystems (Rodríguez et al., 2015).

Research to assess ecosystem threat status in the marine environment is lacking compared to that done in the terrestrial realm (Sink et al., 2012; Bland et al., 2018). This is attributed to the remote nature of the ocean, prohibitive costs, technological difficulties and a focus on fisheries approach. For example, although there is reported growth in the application of the IUCN RLE, only 10% of the marine ecosystems that constitute more than 70% of the planet, are represented in the global RLE (Bland et al., 2018). The terrestrial component of the NBA of South Africa is very well developed and underpinned by more than a century of vegetation mapping whereas

the classification, mapping and assessment of marine ecosystems is still in its infancy stages (Sink et al., 2012). Traditionally, ecosystem assessment techniques were developed to suit terrestrial ecosystems which often display spatial symptoms, whereas for marine ecosystems coarse classification models were used to translate human actions into ecosystem impacts (Halpern et al., 2007; Crain, Kroeker and Halpern, 2008). For example, risk assessment protocols developed for terrestrial ecosystems are frequently based on assessing decline in spatial distribution (geographic extent), with some authors considering this basis to be less suitable for evaluating marine ecosystems (Bland et al., 2018). A second example is that the information used to inform the assessment of many red-listed species (57%) and ecosystems (38%) is either entirely dependent on spatial distribution and/or distribution in combination with other variables (Keith, Akçakaya and Murray, 2018). Whereas, marine ecosystems display symptoms of both spatial e.g. size/distribution and ecological function e.g. ecological functions to degradation (Bland et al., 2018). Nonetheless, terrestrial risk assessments have been applied for a long time (Bland et al., 2018) and many lessons can be drawn from these to strengthen the relatively novel assessment of risks to marine ecosystems.

3.2.1. Marine Ecosystem Assessment in South Africa: National Biodiversity Assessment (NBA)

As introduced in section 1.6.1 the first National Spatial Biodiversity Assessment used an expert-based approach to assess threat status at a scale of 34 biozones (a broad classification considering biogeographic and major depth patterns). The subsequent NBA 2011 assessed the condition and ecosystem threat status of 136 marine ecosystem types using cumulative pressure mapping techniques developed by Halpern et al. (2007; 2008) and Teck et al. (2010), which investigates the addition of pressures on a single point. Despite this, a national map of South African kelp forests was still lacking, hence their omission in previous assessments. However, the NBA 2011 set the stage to assess the threat status of all South African marine ecosystems and recommended the inclusion of the kelp forests in future assessments. This study has contributed three kelp forest ecosystem types into the third NBA conducted in 2018. The third NBA is the first assessment to align the South African approach with the emerging IUCN RLE in the assessment of 150 South African marine ecosystem types (Skowno et al., 2019).

3.2.2. International Union for Conservation of Nature Red List of Ecosystems (IUCN RLE)

The IUCN RLE is an emerging global tool that can be used to assess the risk to ecosystem collapse across widely contrasting ecosystems using criteria and categories (Figure 3. 1 and Table 3. 1) respectively (Bland et al., 2017a; b). The IUCN RLE requires a consistent and clear definition of the units of assessment (ecosystem types) that can be delineated spatially (Keith et al., 2013). To assess risk, the IUCN RLE recommends defining ecosystem collapse i.e. the endpoint of ecosystem decline (Bland et al., 2017a; b). The foundation of defining ecosystem collapse should be a sound understanding of the key driving processes of the ecosystems under assessment (Rodriguez et al., 2015; Murray et al., 2016; Bland et al., 2017a; b; www.iucn.org/about/union/commissions/cem). The IUCN RLE protocols recommend the evaluation of the risk of ecosystem collapse using all available data. In a case where data are not available for assessing a criterion, the Data Deficient (DD) category is assigned (see: Rodriguez et al., 2015). There are two categories for non-threatened ecosystems Least Concern (LC) and Near Threatened (NT), and three for threatened ecosystems Vulnerable (VU) Endangered (EN) and Critically Endangered (CR) and one category for Collapsed (CO) ecosystems (Keith et al., 2013; Rodriguez et al., 2015). Fundamentally, both the NBA 2011 (developed for national applications) and IUCN RLE (developed for international scale applications), have similar goals of generating scientific data to inform policies and management of biodiversity.

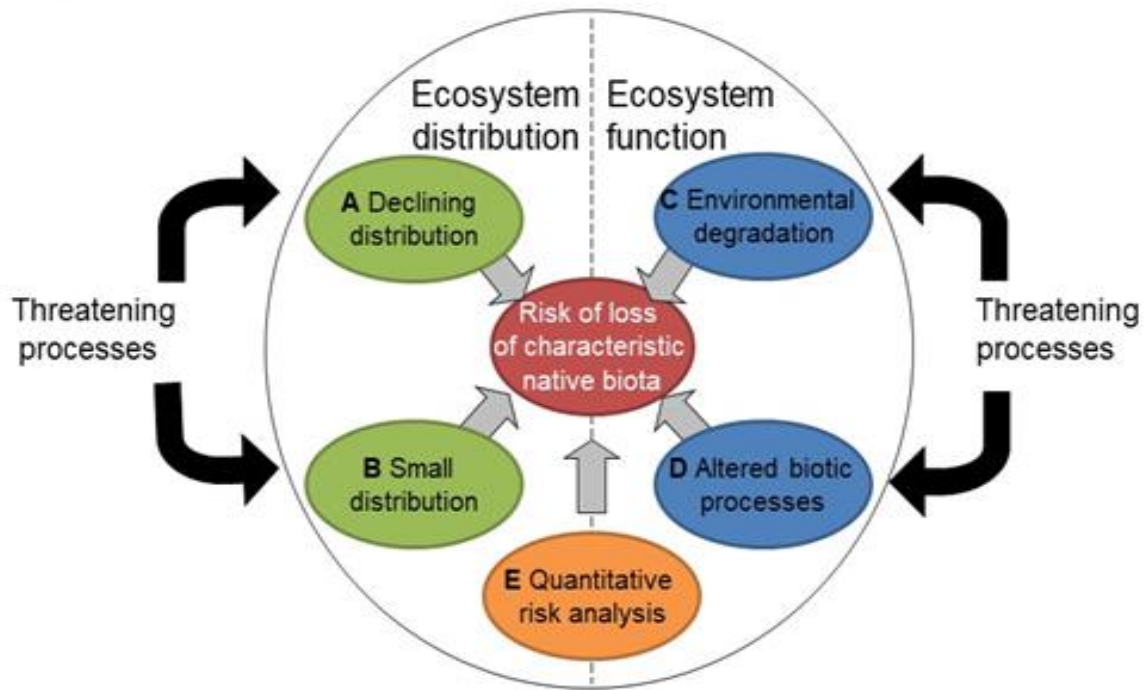


Figure 3. 1: IUCN RLE criteria for assessing the risk of ecosystem collapse. Source: (Keith et al., 2013).

Table 3. 1: Similarities and differences in categories for ecosystem threat status as used in the NBA 2011 and IUCN RLE adopted for the new NBA 2018 assessment: Collapsed (CO), Critically Endangered (CR), Endangered (EN), Vulnerable (VU), Near threatened (NT) and Least Concern (LC).

Categories	NBA 2011 (Sink et al., 2012)	IUCN RLE (Bland et al., 2017a) (as adapted for NBA 2018)
Collapsed-CO	Ecosystem collapse was not defined.	A state where all characteristic native biota and defining biotic and abiotic features are no longer sustained.
Critically Endangered-CR	Ecosystems that have less than 20% (a threshold representing the biodiversity target) of their area in good/near-natural as a result of ecosystem processes that had been extremely modified.	Ecosystems at extremely high risk of collapse according to any of the criteria A to E. Under criteria C and D, ecosystem types where more than 80% (C1, D1) or 90% (C3, D3) of the ecosystem extent is severely degraded quality.
Endangered- EN	Ecosystems that have less than 35% (a threshold of biodiversity target) of their area in good/near-natural condition because of more than 65% extent in poor ecological condition.	Ecosystems at a very high risk of collapse according to any of the criteria A to E: Under criteria C and D, ecosystem types where more than 70% (but less than 90%) of the ecosystem extent is very severely degraded or more than 90% is severely modified.

Categories	NBA 2011 (Sink et al., 2012)	IUCN RLE (Bland et al., 2017a) (as adapted for NBA 2018)
Vulnerable-VU	Ecosystems that have more than 35% of their area in good/near-natural condition but with some of the ecosystem area outside this threshold.	Ecosystems at a high risk of collapse according to any of the criteria A to E: Under criteria C3 (and D3), there are 3 ways ecosystem types qualify as Vulnerable. 1. more than 50% (but less than 70%) of the ecosystem extent is <i>very severely degraded</i> or 2. more than 70% is <i>severely degraded</i> or 3. more than 90% is <i>moderately degraded</i>
Near threatened-NT	Not applied.	An ecosystem is NT when it does not qualify for all criteria A to E, but likely to qualify for the threatened category in the near future: Any ecosystem type within 10% of the qualifying threshold for VU was considered as NT.
Least Concern- LC	An ecosystem that has more than 80% of its area in good/near-natural condition and meets associated biodiversity targets plus 15%.	An ecosystem is LC when it does not qualify for all criteria A to E, with no near-future possibility of qualifying for the threatened category: Under criteria C3 (and D3), ecosystem types where the area in good condition is more than 50% qualify.

The processes of the planet are dynamic and complex, a reason why all species and ecosystem types may eventually be replaced by others over millions of years. Therefore, all ecosystems may never be completely free of the risk of collapse (Bland et al., 2017a). Contrary to species, ecosystems do not disappear, instead, they transform into different ecosystems with transformed abiotic and biotic characteristics (Bland et al., 2017a). However, ecosystem collapse is analogous to the concept of species extinction and the IUCN RLE protocol does not consider whether or not ecosystem endpoint is reversible or irreversible (Rodríguez et al., 2015). Instead, it uses two components (X and Y: Figure 3. 2) of spatial and functional symptoms to better represent ecosystem collapse as a progression between states, the beginning of the one-state being the end of the other e.g. regime shifts (Blamey, Plaganyi and Branch, 2013; Ling et al., 2014) and trophic cascades (Estes, Burdin and Doak, 2016).

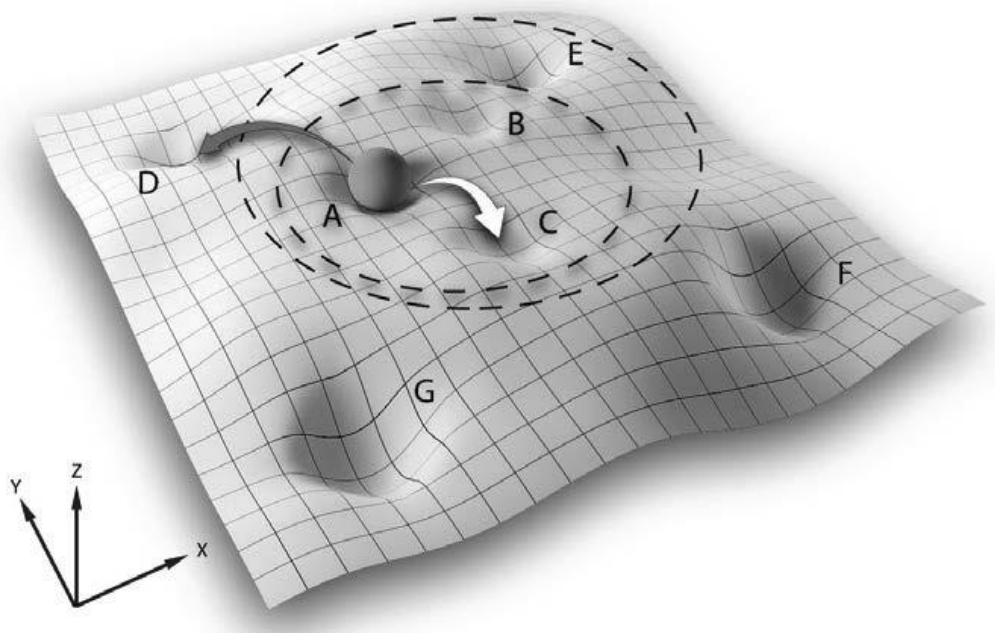


Figure 3. 2: Generalised model that represents transition and ecosystem collapse. The arrows Z represent potential for change, X and Y assess the progression along different ecosystem pathways of collapse. A, B and C (white arrow) represent the natural variability without loss of defining characteristics and the (black arrow) represents transition beyond the second circle to (D, E, F and G) collapse or alternative states. Cited from (Bland et al., 2017a).

These Red listing of ecosystem (RLE) protocols have already been successfully applied to assess the risk of collapse of the Alaskan kelp forests (Keith, 2013), Meso-American Reef (Bland et al., 2017b) and recent assessment of the shelf (30 m to 500 m) Southern Benguela ecosystem (Bland et al., 2018).

3.3. Aims and objectives

This chapter aims to assess the threat status of South Africa's kelp forests and distil key lessons to inform future ecosystem mapping and assessment. Four specific objectives were defined for this chapter:

1. To collate information on threats to South African kelp forest ecosystems.
2. to develop a conceptual model of the South African kelp forest ecosystems
3. To apply the IUCN RLE method (NBA) to assess ecosystem threat status.
4. To explore the implications of different threat assessment criteria and thresholds.

3.4. Materials and methods

To assess ecosystem threat status of kelp forest ecosystems four main components were needed. The first component was the spatial and functional delineation of the unit of assessment. The second component as per the recommendations of the IUCN RLE was the development of a conceptual ecosystem model to better understand key ecosystem interactions and pathways that can help define ecosystem collapse. The third component was the assessment of ecosystem degradation from cumulative pressure impacts and the fourth components included the systematic application of IUCN criteria to define threat status for each kelp ecosystem type. These are elucidated in depth in the following sections.

3.4.1. Units of assessment

The map of South African kelp forest ecosystems and its classification (Namaqua, Cape and Agulhas) are presented in chapter 2 (sections 2.3.4). The spatial distribution, which extends from the Orange River to De Hoop of the kelp types is presented in (Figure 2. 16), with further information of their occurrence (Figure 2. 2) and biogeography elucidated in chapter 1: (section 1.4). The extent of the developed South African kelp forest ecosystems map was used by experts to define the unit of assessment (see: Rodriguez et al., 2015; Bland et al., 2017a).

3.4.2. Conceptual model and defining ecosystem collapse

A qualitative ecosystem model that represents key interactions between the abiotic and biotic components of the South African kelp forest ecosystems was compiled. This was achieved through literature review and expert guidance to identify key defining characteristics for the three kelp forest ecosystem types and identify pressures impacting each ecosystem type. Key links between the three-dimensional structure forming kelp forests, associated biotic and abiotic components, as well as the connection of kelp forest ecosystems with surrounding

ecosystems were investigated. Firstly, when developing an ecosystem model, the IUCN RLE guidelines advise the describing of characteristic native biota, abiotic environments, key processes and interactions and spatial distribution (see: Bland et al., 2017). Secondly, for quantitative application of the IUCN RLE assessment, it is paramount that the assessor develops various reasons, mechanisms and pathways of ecosystem decline (as described in Figure 3. 2) and most significantly defines the endpoint of ecosystem decline (Bland et al., 2017a).

In Chapter 1 (section 1.3) the dynamics of the South African kelp forest ecosystems were elucidated, these include increasing pollution, the shift of the west coast rock-lobster, depleted abalone stocks, the eastward shift of *E. maxima* and decline in sea urchin densities in some regions. Using energy flow Field et al. (1977) illustrated the state of the South African kelp forest ecosystems and the roles of associated abiotic and biotic factors. The significant abiotic processes include nutrients, temperature, depth, currents, waves and wind and the biotic include primary producers and consumers that shape the kelp forest ecosystems (Field et al., 1977 and Branch and McClanahan, 2008). The above studies and other literature were synthesised and organized to form a framework that conceptualises the key processes, pathways and components governing South African kelp forest ecosystems into a model. Key intrinsic and extrinsic, biotic and abiotic processes were used to guide defining contexts that would represent/lead to ecosystem collapse.

3.4.3. Pressures and assessment of ecosystem degradation

Key pressures that threaten kelp forest ecosystems were identified through literature review and two expert workshops comprising different groups of experts but with some overlap. The first workshop included professionals in kelp ecology and biology, who identified mappable pressures that impact the South African kelp forest ecosystems. The second group comprised a broader group of marine ecosystem scientists who refined and standardised the scoring for the assessment of all ecosystem types for the NBA 2018 (see: Sink et al., 2019). To support the development of impact and recovery scores literature was reviewed and summarised for each identified pressure. The kelp experts recognized a total of thirteen pressures namely; line fishing e.g. recreational and commercial; mining; ports and harbours; abalone fishing; wastewater/pollution; West Coast Rock Lobster fishing; coastal development and kelp harvesting, which were mapped for use in a cumulative pressure assessment (see: Majiedt et al., 2019). Line fishing accounted for four types of pressures i.e. recreational, commercial,

shore-based and boat-based line fishing. This excluded climate change and extreme events which were recognised as important but could not be included at this stage due to gaps in data and a lack of expert consensus in both the magnitude and direction of changes in different regions. An ecosystem pressure matrix was used to score the impact of each pressure per broad ecosystem type, in a similar approach to that employed in the NBA 2011 (see: Sink et al., 2012) which was based on the methodology developed by Teck et al. (2010). A guided expert consensus on the functional impacts (the degree to which the natural state of any given ecosystem is affected by an individual pressure), frequency (how often the discrete pressure) of occurrence of the pressure in the ecosystem, resistance (the average tendency of the ecosystem to resist change due to this pressure) and resilience (recovery time required for the ecosystem after removal of the threat) to the different pressure impacts were assessed to develop a pressure matrix (Sink et al., 2012). The pressure matrix enabled the ranking of the different pressure intensities/severity on the different kelp forest ecosystem types.

A measure of the levels of expert confidence/uncertainty was also included. Each of the thirteen pressures was weighted and scored using the methods developed by Halpern et al. (2007; 2008) and Teck et al. (2010). Majiedt et al. (2019) mapped 31 pressures for the 150 marine ecosystem types assessed in the NBA 2018. Impact weights (W_{ij}) (Teck et al., 2010) were then calculated for each of the identified pressures, where (i) is the anthropogenic pressure or activity and (j) is the ecosystem type (Equation 3.1). Experts agreed on a functional impact score (i) for each of the pressures and a recovery potential score (j) for kelp forests. All the kelp forests were treated uniformly with some pressures being subclassified to accommodate the regional difference in impacts e.g. line fishing. The cumulative (Figure 3. 2) impact scores were measured for each ecosystem type (I_{cj}) (Equation 3.2), where β_i is the normalised pressure value scaled between 0 -1 of the intensity of pressure on a location (i).

$$W_{ij} = (i + j)/2 \dots\dots\dots \text{Equation 3.1}$$

$$I_{cj} = \sum_{i=1}^n \beta_i * W_{ij} \dots\dots\dots \text{Equation 3.2}$$

Four condition categories (“Natural”, “Moderately Degraded”, “Severely Degraded” and “Very Severely Degraded”) were defined in a similar approach to those used to assess the condition in the NBA 2011 (which were originally derived from the categories defined by the freshwater and estuarine research communities who routine measure ecosystem condition) (Sink et al., 2012, 2019). Areas of high cumulative pressures or where single pressures have very high impacts are assumed to represent poor ecological condition as per the thresholds in

Table 3.1. This was done to allow the determination of cumulative pressure impacts across each ecosystem type so that ecological degradation could be determined (steps 1- 6 below).

Four ecological degradation conditions were developed using the following steps:

Step 1: Identify pressures.

Step 2: Map the individual intensity of each pressure.

Step 3: Build the pressure matrix that reflects the impact of each pressure on kelp forests.

Step 4: Combine the pressure intensity map and the pressure matrix to develop an individual pressure impact map that represents both the intensity and impact of each pressure at the pixel level (Majiedt et al., 2019).

Step 5: Build a cumulative impact layer using (equation 3.2) as conceptualized in (Figure 3. 3).

Step 6: Translate the cumulative impact layer into a graph of ecological degradation with four categories aligned to the IUCN RLE thresholds adapted from NBA 2011: “Natural”, “Moderately Degraded”, “Severely Degraded” and “Very Severely Degraded”.

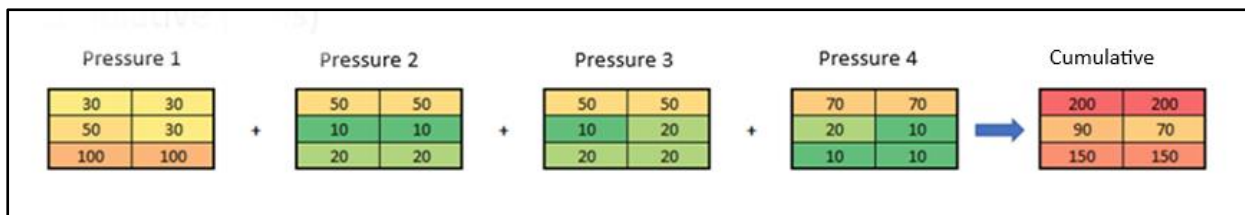


Figure 3. 3: Illustration of the approach used to assess cumulative pressures representing how individual pressures (summation of 30, 50, 50 and 70 in the top left cells for all the four pressures equal 200) acting on the same location are summed. Pressures 1, 2, 3 and 4 do not have the same spatial intensity and the intensity of each pressure is cumulative in cells that overlap.

3.4.4. Assessment of ecosystem threat status

To assess the threat status of the kelp forest ecosystem of South Africa, three criteria (Figure 3. 1) were used (explained below) and two were Not Evaluated (NE).

3.4.4.1. Spatial criteria: Criteria A: Declining ecosystem extent

Declining distribution is measured from two time series equations that consider changes in an area over a certain time frame (years) to determine the rate of decline and percentage area lost (Bland et al., 2017a). Although there was insufficient data to measure changes in ecosystem

extent, literature was reviewed for evidence of declining distribution of the three South African kelp forest ecosystem types.

3.4.4.2. Spatial criteria: Criteria B: Area of Occupancy (AOO) and Extent of Occurrence (EOO)

Criterion B evaluates the risk of ecosystem collapse by identifying ecosystems that have restricted distribution to the extent that they are at risk of collapse from the possibility of a single threatening event (Bland et al., 2017a). Firstly, an ecosystem may be assessed under criterion B, if it meets the thresholds of any of the sub-criteria B1, B2 and B3 (Table 3. 2) and secondly, if it meets any of the three conditions (i, ii and iii: Table 3. 2) of inferred continuing decline in (i) spatial extent, (ii) environmental quality and (iii) disruption of biotic interactions. To determine whether South African kelp forests meet the thresholds defined for subcriterion B1 and B2, the extent of occurrence (B1: EOO) and area of occupancy (B2: AOO) were calculated and cumulative pressure mapping was used to assess ecosystem degradation.

Per the guidelines of the IUCN RLE, the extent of occurrence (EOO) is measured by the calculation of the minimum convex polygon i.e. the smallest polygon that encompasses the entire known occurrence of the ecosystem in which no internal angle exceeds 180°. The minimum polygon must encompass all areas, such as land for marine ecosystems and ocean areas that border the evaluated ecosystem (Rodriguez et al., 2015 and Bland et al., 2017a). These features are important because they contribute to the spreading of risks across the distribution of the ecosystem by making different areas more spatially independent (see: Murray et al., 2016). Using ArcGIS 10.4, the minimum bounding geometry tool was used to calculate the extent of occurrence (EOO) (Table 3. 2) for all three types of kelp forest ecosystems Namaqua, Cape and Agulhas. Lastly, the three different kelp forest ecosystem types were merged and considered as a single type of greater extent to examine the influence of the scale of ecosystem classification in the assessment. This was named the South African Kelp Forest Ecosystem and its EOO was also calculated as described above.

The area of occupancy (AOO) is highly sensitive to the pixel resolution at which ecosystem distributions are mapped, therefore the IUCN RLE protocol recommends using a standard grid cell of (10 X 10 km). Murray et al. (2016) gave three reasons for using this conventional grid, (1) the magnitude of the grid makes it simple to determine if there is an ecosystem within the grid compared to a smaller grid, (2) to identify the existence of ecosystems that function over larger spatial scales or have characteristics that are sparse or e.g. pelagic systems and (3) larger

cells enable AOO assessment even when information on high-resolution distribution data is limited. The assumption is that a large number of small patches/polygons contribute negligible risk spreading effect compared to that of larger polygons. This may be corrected by excluding these polygons from the AOO if they account for less than (1 km²) of the grid cell area (Murray et al., 2016), this correction is referred to as the “1% rule” here onwards. To measure the AOO, grid cells (10 X 10 km) were generated using an ArcGIS convex hull geometry tool and overlaid on top of the ecosystem types polygons. The number of grids occupied by the kelp polygons for each of the four kelp forest ecosystem types (Namaqua, Cape, Agulhas and the combines South African type) was determined to support assessment against subcriterion (B2) as depicted in (Table 3. 2). The implications of applying the “1% rule” and when it is not applied were investigated on the occupied grid cells to determine the overall threat status of these two cases (Murray et al., 2016).

Table 3. 2: Criterion (B) and sub-criteria (B1, B2 and B3) thresholds and categories: Critically Endangered (CR), Endangered (EN) and Vulnerable (VU) for assessing restricted biogeographic distribution (source: Bland et al., 2017a).

Subcriterion	Measure of geographic distribution	CR	EN	VU
B1	Extent of a minimum convex polygon (km ²) enclosing all occurrences (extent of occurrence, EOO) is: AND at least one of the following (a-c): (a) An observed or inferred continuing decline in ANY OF : i. a measure of spatial extent appropriate to the ecosystem; OR ii. a measure of environmental quality appropriate to the characteristic biota of the ecosystem; OR iii. a measure of disruption to biotic interactions appropriate to the characteristic biota of the ecosystem. (b) Observed or inferred threatening processes that are likely to cause continuing declines in geographic distribution, environmental quality or biotic interactions within the next 20 years. (c) Ecosystem exists at:	≤ 2,000	≤ 20,000	≤ 50,000
B2	The number of 10×10 km grid cells occupied (area of occupancy, AOO) are: AND at least one of a-c above (same as for B1).	1 threat-defined location	≤ 5 threat-defined locations	≤ 10 threat-defined locations
B3	A very small number of threat-defined locations (generally fewer than 5) AND prone to the effects of human activities or stochastic events within a very short time period in an uncertain future, and thus capable of Collapse or becoming Critically Endangered (CR) within a very short time period (B3 can only lead to a listing as Vulnerable, VU).			VU

3.4.4.3. Functional criteria: Criteria C Environmental degradation

To evaluate the threat status of an ecosystem under criterion C, the abiotic components of the ecosystem must be experiencing reduction or environmental degradation of the capacity of the

ecosystem to sustain its characteristic biota. Using subcriteria C1 (past 50 years) and C3 (since 1750) the proportion of each ecosystem type within the respective IUCN RLE thresholds were compared (Table 3.3). This was feasible because the four categories of ecosystem degradation were deliberately aligned with the IUCN thresholds for relative severity of ecosystem change.

Table 3. 3: Criterion (C), sub-criteria, thresholds and categories of threat status for assessing relative severity of environmental degradation and ecosystem extent. The three categories of degradation as measured through cumulative impact mapping were deliberately aligned with the three relative severity thresholds ie. “Very Severely Degraded”/90, “Severely Degraded”/70 and “Moderately Degraded”/50 (modified from (Bland et al., 2017a).

Subcriterion		Extent (%)	Relative severity (%)		
			Very Severely Degraded	Severely Degraded	Moderately Degraded
C1	The past 50 years based on a change in an abiotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table:	≥ 80	CR	EN	VU
		≥ 50	EN	VU	
		≥ 30	VU		
C3	Since 1750 based on the change in an abiotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table:	≥ 90	CR	EN	VU
		≥ 70	EN	VU	
		≥ 50	VU		

3.5. Results

3.5.1. Units of assessment

The three delineated units of assessment were the Namaqua, Cape and Agulhas Kelp Forest Ecosystem types and a fourth combined South African Kelp Forest Ecosystem type was also considered. The distribution, biotic, abiotic characteristics and key pressures within each type are presented (Table 3. 4). The Namaqua Kelp Forest extends from the Orange River to Donkin Bay (Figure 3. 4) and has the following key abiotic drivers: temperature, substrate, current, upwelling and downwelling. One of the distinct biotic components of this ecosystem type is the hollow stiped form of *L. pallida* (Table 3. 4 and Table 3. 5). From the identified pressures impacting the Namaqua Kelp Forest Ecosystem, ocean mining is unique when compared to the other types. The Cape Kelp Forest extents from Donkin Bay to Cape Point with similar key abiotic drivers to that of the Namaqua Kelp Forest Ecosystem although they are spatiotemporally variable. The Cape Kelp Forest Ecosystem is dominated by *E. maxima* occurring in the shallow with *L. pallida* (solid stipe) forming a sub-canopy and large amounts of red algae on the substrate. Key identified pressures include commercial and recreational line fishing (Table 3. 4). The Agulhas Kelp Forest Ecosystem extends from Cape Point to De Hoop. A key abiotic characteristic is the warmer water in which they occur, with both the Agulhas current and Benguela current influencing its environment and biodiversity. Unique to the Agulhas Kelp Forest Ecosystem is the presence of *E. radiata* and the decreasing abundance of *L. pallida*. The Agulhas Kelp Forest Ecosystem has historically always had large herbivore biomass (abalone and urchins), as well as diverse reef fish communities. The West Coast Rock Lobster fishery in the Agulhas kelp ecosystem is relatively new compared to the other two ecosystems (Blamey et al., 2015).

Table 3. 4: Defining characteristics of Namaqua, Cape and Agulhas Kelp Forest Ecosystem types, their distribution and the various anthropogenic pressures impacting each ecosystem type.

Kelp forest ecosystem	Distribution	Abiotic drivers	Biotic drivers	Identified pressures
Namaqua (NKFE)	Orange River - Donkin Bay West Coast	Temperature Substrate Benguela current Upwelling and downwelling	<i>L. pallida</i> (hollow stipe) <i>E. maxima</i> (less dominant) Red algae (Rhodophyta) Rock lobster	West Coast rock lobster fishery Kelp harvesting Aquaculture Mining Ports and harbours Wastewater
Cape (CKFE)	Donkin Bay - Cape Point West Coast	Temperature Substrate Benguela current Upwelling and downwelling	<i>E. maxima</i> (dominant in shallows) <i>L. pallida</i> (solid stipe, sub-canopy) <i>M. pyrifera</i> (patchy) Red algae (Rhodophyta) Filter feeders (mussels and ascidians) Rock lobster	Commercial line fishery Recreational line fishery West Coast rock lobster fishery Kelp harvesting Aquaculture Ports and harbours Wastewater
Agulhas (AKFE)	Cape Point – De Hoop Nature Reserve	Temperature Substrate	<i>E. maxima</i>	Commercial line fishery Shore-based line fishery

Kelp forest ecosystem	Distribution	Abiotic drivers	Biotic drivers	Identified pressures
	South Coast	Agulhas current/Benguela current (Transition zone)	<i>E. radiata</i> (intertidal pools/fringe) <i>L. pallida</i> (solid stipe) Rock lobster Abalone Urchins Encrusting coralline algae Green algae (Phaeophyta) Other grazers Reef fish	Recreational line fishery West Coast rock lobster fishery (since the early 2000s) Abalone fishery (illegal and legal) Kelp harvesting Aquaculture Wastewater

The kelp forest ecosystem of South Africa extends from the Orange River to De Hoop and was subdivided into Namaqua, Cape and Agulhas Kelp Forest Ecosystem types in Chapter 2.

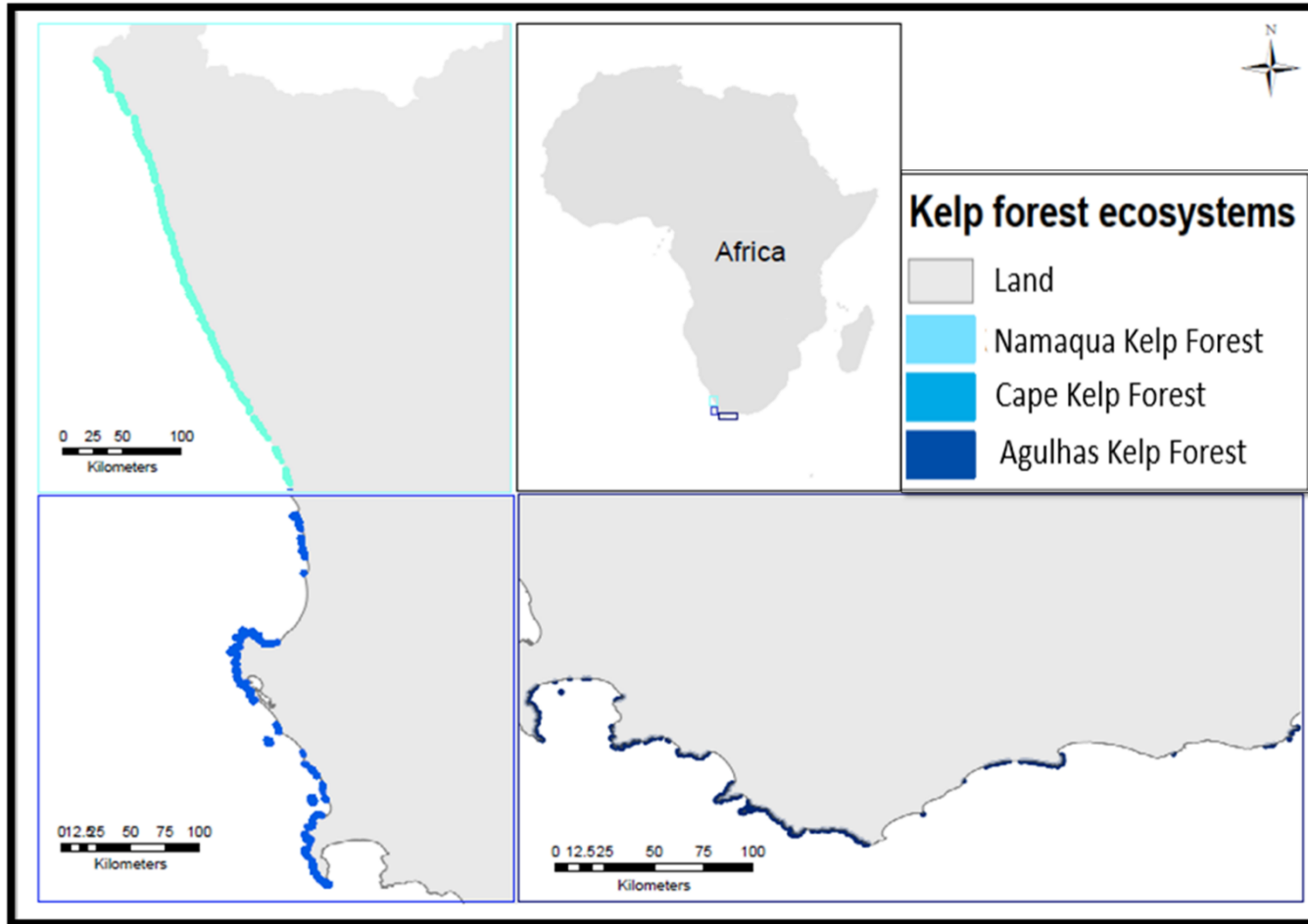


Figure 3. 4: Current distribution of South African Kelp Forest Ecosystem types. Namaqua Kelp Forest Ecosystem (aqua blue/lite blue) extends from the northernmost part of South Africa along the west coast: Orange River to Donkin Bay. From, Donkin Bay to the tip of the Cape Point, Cape Kelp Forest Ecosystem type (blue) occurs. The Agulhas Kelp Forest Ecosystem (dark blue) extends from the eastern tip of Cape Point to De Hoop Nature Reserve (Chapter 2: section 2.10.3).

3.5.2. Ecosystem model and Ecosystem collapse

The key interaction processes that govern the kelp forest ecosystem types of South Africa are represented (Figure 3. 5) in a generic ecosystem model. The ecosystem model shows how anthropogenic pressures (red), climate change (blue), key abiotic (orange) and biotic factors (green) interact within the South African kelp forest ecosystems (Table 3. 4). The arrows represent positive or increasing effects and they show how kelp detritus, drift and debris are exported into the surrounding/adjacent ecosystems and environment such as the benthos, rocky shores and offshore. The light brown curved arrow denotes the phytoplankton that is imported into the kelp forest ecosystem by the downwelling process. The abiotic components including depth, substrate, upwelling, temperature and downwelling have a positive role in the kelp forest ecosystems. The interaction of the biotic components of the kelp ecosystems is depicted from top predator to primary producer with points in the green box. Anthropogenic actions including coastal development, marine mining, harvesting and pollution have negative or decreasing effects (points) on the three types of kelp forest ecosystems and augment/increase climate change (based on global literature). The effects of climate change may be negative (points) on both the biotic and abiotic components of the kelp forest ecosystems. The Cape Kelp Forest has similarities to both the Namaqua and Agulhas Kelp Forest Ecosystem types (Table 3. 4). The difference is that monthly mean temperatures in the summer in the Agulhas ecoregion is more than 15 degrees, a key indicator of warm temperate conditions (Luening, 1990).

Instead of having a rigid definition of ecosystem collapse, a spectrum of ecosystem states is proposed in this study to suit the diversity and complexity of kelp forest ecosystems in the region. Firstly, in line with criteria A and B which concerns reducing and restricted distribution respectively, ecosystem collapse/endpoint can be reached if the current mapped the distribution of South African kelp forests (Figure 3. 4) declines to zero. Secondly, if a point where the current identity of the Namaqua, Cape and Agulhas Kelp Forest Ecosystems is to be completely transformed to a state where the ecosystems no longer sustain the current defining abiotic (criterion C) and biotic (criterion D) features (Table 3. 4 and Figure 3. 5), then ecosystem collapse may be realised. Thirdly, in line with the NBA 2018 National Map of Marine Ecosystems, if more than 90% of the ecosystem has been “Very Severely Degraded” or lost (i.e. less than 10% of the current mapped distribution for each ecosystem remains), that ecosystem is collapsed. All three states have not been realised/reached and therefore, South African kelp forest ecosystem types are not collapsed.

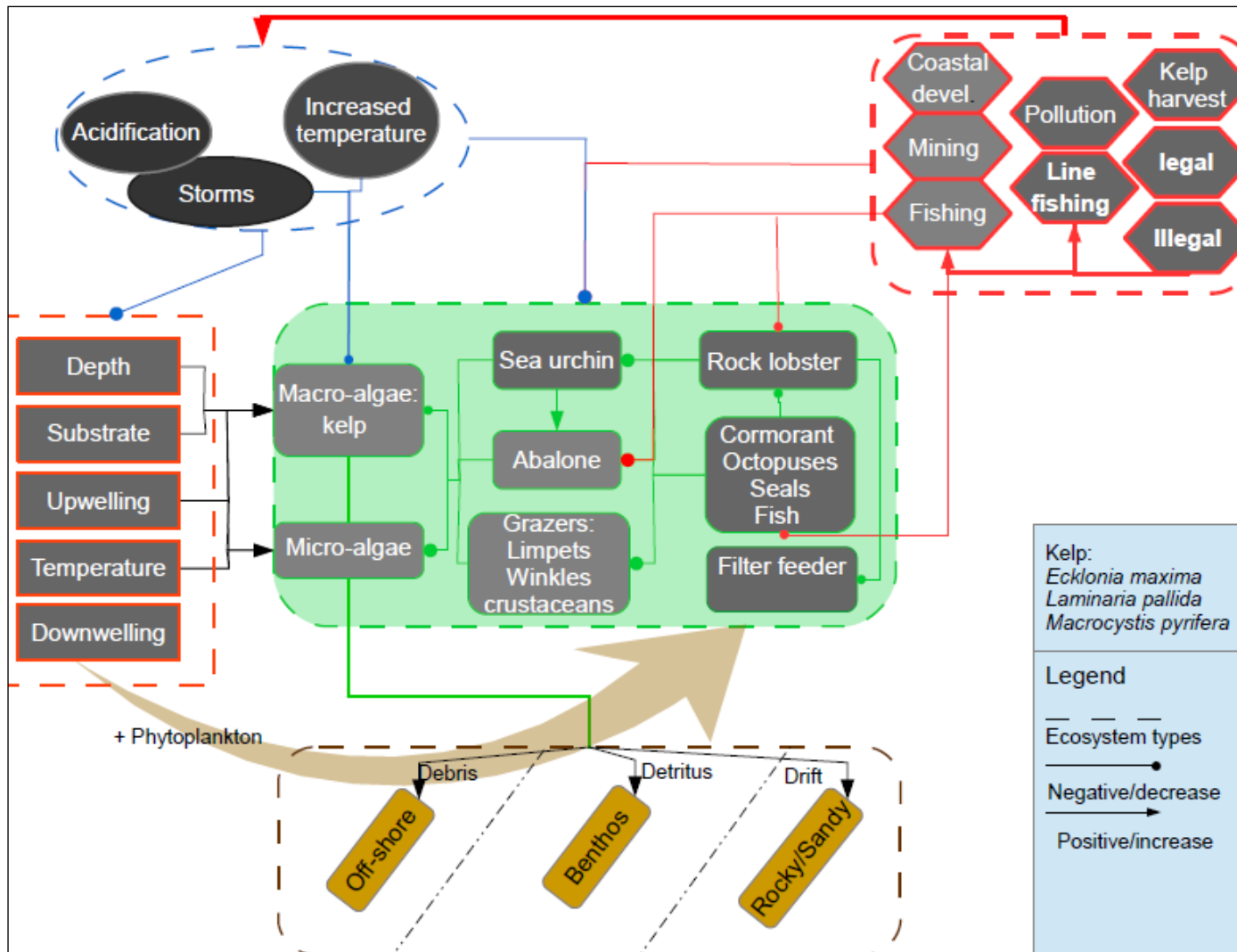


Figure 3. 5: Simplified conceptual model highlighting key ecosystem components and their interactions within kelp forest ecosystems in South Africa. Arrows depict positive impacts or increases in population sizes or effects and points represent negative impacts or population decreases. The dashed lines group different

ecosystem components together. Human pressures are enclosed in red dash lines, climate change aspects are enclosed in blue, abiotic factors are enclosed in orange, adjacent ecosystems are enclosed in brown and the biotic components of the kelp ecosystems are enclosed in green. The curved arrow represents import (+) into the kelp forest systems and the green arrow represents export to adjacent ecosystems. Key references that were used to develop this model were Field et al., 1977; Bolton and Anderson, 1987; Bolton and Levitt, 1987; Mayfield and Branch, 2000; Branch, 2008; Levitt et al., 2002; Blamey et al., 2010; Sink et al., 2012; Mead et al., 2013; Borja, 2014; Morris and Blamey, 2018; Ojemaye and Petrik, 2019; Dyer et al., 2019.

3.5.3. Pressure and assessment of ecosystem degradation

The key pressures identified for South African kelp forest ecosystems are shown in Table 3. 5 which summarises these and their impacts based on national and international knowledge. Key pressures included line fishing, mining, ports and harbours, abalone fishing, wastewater, coastal development and kelp harvesting.

Table 3. 5: Summary of numerous pressures reported to have direct and indirect impacts on kelp forest ecosystems on South African (*) kelp ecosystems and relevant international literature.

Identified pressure and summary of impacts	Relevant Literature
<p>Line fishing- Commercial and recreational fishing by hook and line removes many, predatory fish and reef-associated fish species. Species that are targeted by line fishers play crucial predatory roles in their ecological niches. Their removal has led to changes in ecosystem structure, diversity and functions, with knock-on ecosystem effects. The invasion of the rock lobsters would have been prevented if predatory line fish were still present and hence collapse of abalone and sea urchin stock in the invaded region prevented. On the south coast of South Africa line fishers targets long-lived, slow-growing, late-maturing, sex-changing vulnerable species with slow recovery.</p>	<p>Blaber et al., 2000; Griffiths, 2000; Tegner and Dayton 2000; Atkinson and Sink, 2008; Palmer et al., 2008; Götz et al., 2009; Ling et al., 2009; Attwood et al., 2011; Mann 2013; Mead et al., 2013; de Moor et al., 215; Blamey et al., 2015; Bertocci et al., 2015</p>
<p>Mining - Mining includes operations to recover diamonds, heavy minerals and phosphates. Mining types include beach mining, diver-based mining of diamonds and offshore mining by drillships, crawlers and hoppers. The mining processes disturb kelp communities by moving</p>	<p>Pulfrich et al., 2003a,b; Kelly, 2005; Halpern et al., 2007; Branch, 2008; Griffiths et al., 2010; de</p>

Identified pressure and summary of impacts	Relevant Literature
<p>boulders, cutting kelp and sucking up gravel for sorting ashore and redepositing the sediments in the intertidal. This increases sedimentation and turbidity and causes pollution (e.g. brine). Cutting of kelp leads to habitat destruction and altered sediment distribution also modifies habitats. Mining impacts also include a proliferation of fast-growing algae, a reduced cover of filter feeders and causes the disappearance of grazers. Also, microscopic, free-living gametophytes (e.g. of <i>Macrocystis</i>) are sensitive to pollution, specifically sedimentation (Edyvane, 2005). In South Africa, mining is concentrated along the northern part of the southern Benguela ecoregion.</p>	<p>Bettignies, Thomsen and Wernberg, 2012; Mead et al., 2013; Majiedt et al., 2019</p>
<p>Ports and harbours from human development of hubs for transportation, using ships primarily. Port and harbour are a source of water and sediment pollution due to ship waste. The impacts include coastal squeeze, noise pollution, the introduction of alien invasive species (Lindberg et al., 2019), altering of currents and disruption of habitats through dredge dumping and anchoring.</p>	<p>Harris et al., 2010; Sink et al., 2012; Mead et al. 2013 Bermejo et al. 2015; Majiedt et al., 2019</p>
<p>Abalone fishing- Abalone plays a key role in kelp forests include grazing of seaweeds and provision of a unique micro-habitat of other organisms growing on their shells thus increasing secondary productivity. This large marine snail is slow-growing and reaches sexual maturity at ca 9 years. Its removal reduces the grazer and/or scavenging role, prey for predators of the kelp forest ecosystems and the unique secondary micro habitat abalone provide, absence of abalone has implications of intensifying urchin barrens discussed by Tegner and Dayton, (2000). Abalone is seriously overexploited in South Africa with pressures from legal and illegal exploitation.</p>	<p>Graham et al., 2007; Blamey, Branch and Reaugh-Flower, 2010*; Plaganyi and Butterworth, 2010; Brill and Raemaekers 2013*; Zeeman et al., 2013*; Blamey et al., 2015*; DAFF 2016*; Lambrechts and Goga, 2016*; Blamey and Bolton, 2018*; Branch and Branch, 2018*; Madjiedt et al., 2019*</p>
<p>Wastewater discharge – The disposal of effluent and other wastewater products from aquaculture facilities, power stations, desalination plants and other industrial and municipal discharge causes poor water quality, increases marine pollution, including increases in chemicals</p>	<p>Kelly, 2005; Amy et al., 2017, Belkacem et al., 2017; Lior 2017, Manju et al., 2017; Ojemaye & Petrik, 2019*; Sink et al 2019*; Majiedt et al., 2019*</p>

Identified pressure and summary of impacts	Relevant Literature
<p>i.e. pharmaceuticals (Ojemaye & Petrik, 2019), eutrophication and increase of heavy metals concentration. Kelp gametophyte stages are particularly sensitive to municipal and industrial discharge (Edyvane, 2005).</p>	
<p>Rock lobster harvesting – Rock lobsters play a key predatory role in kelp forest ecosystems including keeping the populations of kelp grazers in check and indirectly promoting the abundance of seaweeds. Rock lobster fisheries have the potential of instigating regime shifts. Their removal lessens predation of grazers and other prey (urchins, winkles and mussels) which can lead to overgrazing of kelps and other seaweeds and even to low diversity urchin barrens. Rock lobster fisheries also reduce food for higher predators such as banks cormorants, line fish and sharks. Sea urchin - A major kelp grazer in many parts of the world, notorious for its capability to deforest large kelp forests. Without its primary predators can instigate regime shifts and result in “urchin barrens”, where there is little-to-no kelp forest. In South Africa, it is a dominant herbivore on reefs east of Cape Point but feeds mostly by trapping drift kelp and not actively grazing kelp forests. Instead, it plays a vital ecological role by providing shelter and food to juvenile abalone</p>	<p>Mann & Breen, 1972*; Field et al. 1977*; Tarr, Williams and Mackenzie, 1996*; Mayfield and Branch, 2000*; Day and Branch, 2000* Tegner and Dayton 2000; Lafferty, 2004; Graham et al., 2007; Blamey, Branch and Reaugh-Flower, 2010*; Griffiths et al., 2010*; Blamey and Branch, 2012*; Blamey Plaganyi and Branch, 2013*; Brill and Raemaekers, 2013* Blamey Plaganyi and Branch, 2014*; Plaganyi et al., 2014*; Blamey et al., 2015*; Blamey & Bolton, 2018*; Branch and Branch 2018*; Filbee-Dexter & Wernberg, 2018.</p>
<p>Coastal development from human settlements can degrade coastal habitats, interrupt ecological processes and increase pollution thus threatening coastal ecosystems including kelp forests. Impacts on kelp forests are related to changes in sediment dynamics, shading and coastal squeeze.</p>	<p>Mumby et al., 1995; Leisten, 2002; Sink & Attwood, 2008*; Dugan and Hubbard, 2010; O’Connor and Anderson 2010*; Jackson and McIlvenney 2011; Borja et al., 2013; Mead et al., 2013*; Bermejo et al., 2015; Nava Fuentes, Arenas Granados & Martins, 2017; Dugan et al., 2017; Luck-Vogel and Mbolambi, 2018*;</p>

Identified pressure and summary of impacts	Relevant Literature
	Witman and Lamb, 2018; Harris et al., 2019*; Thomas et al., 2019; Pfaff et al., 2019; Ojemaye & Petrik, 2019*
<p>Kelp harvesting - The collection of live, fresh, or beach-cast kelp has been a global activity for centuries. Kelp harvesting in the region is mainly to meet demands for the growing mariculture of abalone. In South Africa, fresh kelp is harvested for commercial use using two methods termed “lethal” and “non-lethal”. Non-lethal kelp harvesting has less impact on kelp ecosystems compared to the lethal method which also has slower recovery. South Africa performs precautionary kelp harvesting and the resource is well managed and currently considered under-exploited (DAFF 2016).</p>	<p>Tegner and Dayton 2000; Levitt et al., 2002*; Rothman, Anderson & Smit, 2006*; Anderson et al., 2006*; Anderson et al., 2007*; Dugan and Hubbard, 2010; O’Connor & Anderson, 2010*; Lorentsen, Sjøtun & Grémillet, 2010; Davoult et al., 2011; Guiry & Morrison, 2013; Vega, Broitman & Vásquez, 2014; Minami et al., 2014; Elston, Anderson, Price, 2015*; Frangoudes & Garineaud, 2015; Borrás-Chavez et al., 2016; Steen et al., 2016; Angus, 2017; Monagail et al., 2017; Blamey and Bolton 2018*; Bennion et al., 2018.</p>

The results of the impact scoring workshop are presented in (Table 3. 6) and their synthesised literature (Table 3. 5). The individual intensity of each pressure is colour-coded based on a scheme developed for the NBA 2018 and impact scores ranged 10 to 100. The greatest current pressure on South Africa kelp forest ecosystems is line fishing from both commercial and recreational sectors, especially in the Agulhas ecoregion. The pressure that is identified with the lowest impact intensity is kelp harvesting. Ocean mining and ports and harbours have the second-highest pressure intensity. Shore-based line fishing varies from (30 to 50) intensity depending on the region. Abalone fishery has a pressure intensity of 50 and west coast rock lobster fishery has a pressure intensity of 45.

Table 3. 6: The individual intensity of each pressure is colour-coded based on a scheme developed for the NBA 2018.

Pressures	Score
Line fishing (Commercial):(East and South Coast)	80
Line fishing (Recreational Boat-Based): (East and South Coast)	80
Mining	60
Port and harbour Impacts	60
Line fishing (Recreational Shore-Based): (East and South Coast)	50
Abalone fishing	50
Wastewater/ Pollution	50
Line fishing (West Coast): Commercial	45
Recreational boat-based line fishing (West Coast)	45
West Coast Rock Lobster	45
Coastal development	30
Recreational shore-based line Fishing (West Coast)	30
Kelp harvesting	20

3.5.4. Ecological condition

These results show the Agulhas Kelp Forest Ecosystem (AKFE) has 50% of its extent “Very Severely Degraded”, 23% “Severely Degraded”, 22% “Degraded” and 4.7% in “Natural” state. Cape Kelp Forest Ecosystem (CKFE) has 60% “Very Severely Degraded”, 19% “Severely Degraded”, 21% “Degraded” and 0.7% in “Natural” state. The Namaqua Kelp Forest Ecosystem (NKFE) has 11% “Very Severely Degraded”, 35% “Severely Degraded”, 49% “Degraded” and 5% in “Natural” state. The combined South African Kelp Forest Ecosystem (SAKFE) degradation state is 44% of the “Very Severely Degraded” extent, 25% “Severely Degraded”, 28% “Degraded” and 0.3 “Natural” (Figure 3. 6).

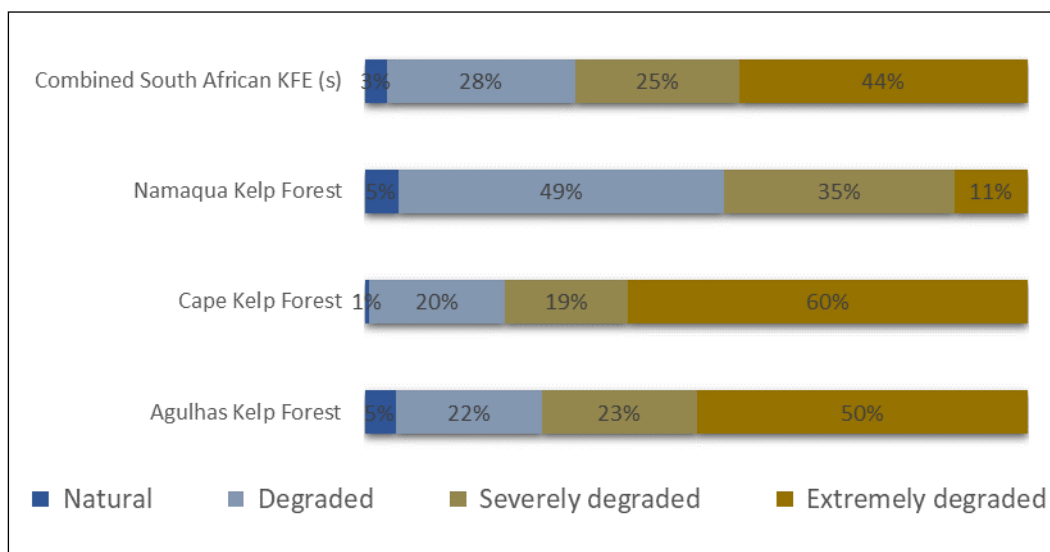


Figure 3. 6: Bargraph reflecting ecosystem degradation by showing the relative area in Natural, Moderately Degraded, Severely Degraded and Very Severely Degraded condition for each ecosystem type (see: Majiedt et al., 2019).

3.5.5. IUCN RLE Assessment

3.5.5.1. Criteria A: Reduction in geographic distribution

This criterion pertains to ecosystems that are undergoing a continued decline in the area of distribution as a result of threats causing loss and fragmentation of ecosystems. There is no past nor present studies or data that support any decline in the distribution of all the three South African kelp forest ecosystem types under subcriterion A1. Therefore all three kelp forest ecosystem types are considered as Least Concern (LC) under this sub-criteria (Table 3. 8 to 3.10).

3.5.5.2. Criteria B: Restricted geographic distribution

This criterion evaluates ecosystems with restricted or small distributions that are susceptible to spatially explicit threats and ongoing decline in environmental quality. Figure 3. 7 shows the EOO of Namaqua (4609.9 km²), Cape (8058.6 km²) and Agulhas (15029.9 km²) Kelp Forest and the relevant extent of ecosystem degradation for each kelp type is presented in Figure 3. 6. All minimum convex polygons are less than 20 000 km² but greater than 2000 km². Therefore under subcriterion B1: Namaqua, Cape and Agulhas Kelp Forest Ecosystems are assessed as Endangered (EN) (see: Table 3. 8 to 3.10). However, when the ecosystem types were combined, the South African kelp ecosystem had a measured EOO (129067 km²) (Figure 3. 7, right) and therefore qualifies as LC under subcriterion B1.

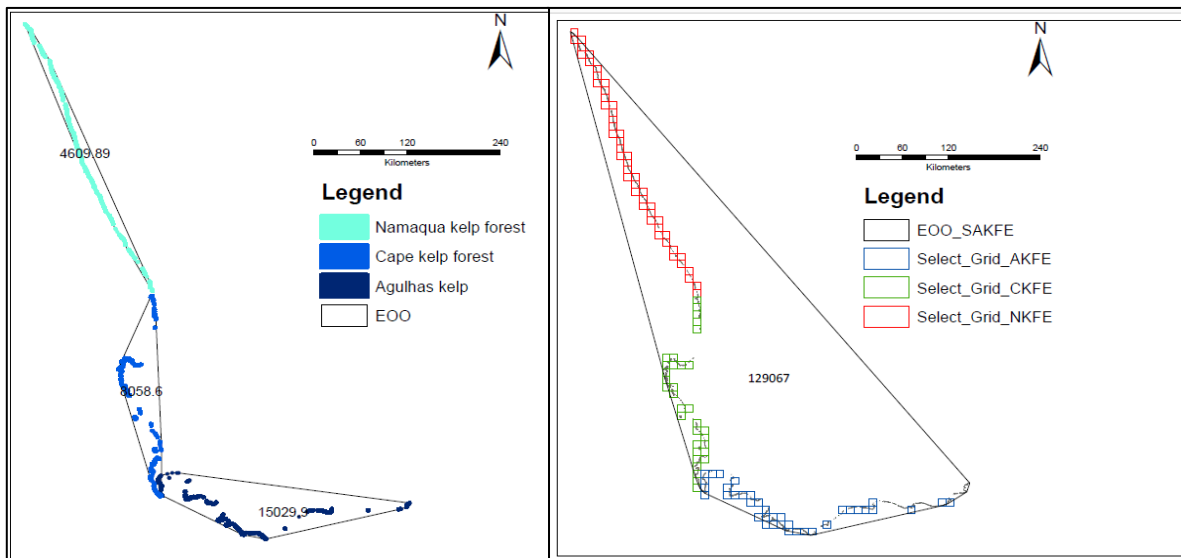


Figure 3. 7: Map of the extent of occurrence (EOO) shown using minimum bounding convex polygon for Namaqua Kelp Forest, Cape Kelp Forest and Agulhas Kelp Forests (left) and the combined South African Kelp Forest Ecosystem (right) in km². Map of the area of occupancy (AOO) depicted with a 10 X 10 grid overlaid over the Namaqua Kelp Forest Ecosystem (red grids), Cape Kelp Forest Ecosystem (green grids) and Agulhas Kelp Forest Ecosystem (blue grids) polygons (right).

Kelp forests naturally occur in patches and therefore the results of the application of the correction factor “1% rule” and when the correction was not applied are represented (Figure 3. 8) below.

Ecosystem types	AOO	1% Rule (Murray et al., 2016)	No. of occupied grids	EOO (Km ²)
Namaqua Kelp Forest →	50 →	Filter AOO grid with less than 1% of area occupied →	0	4609.9
Cape Kelp Forest →	33 →		0	8058.6
Agulhas Kelp Forest →	37 →		4	15029.9
South African Kelp Forest →	120 →		4	129067

Figure 3. 8: Schematic of the application of the EOO and the AOO measurement for the ecosystem types Namaqua Kelp Forest, Cape Kelp Forest, Agulhas Kelp Forest and South African Kelp Forest. The number of grids that had less than 1% occupied by the kelp forest polygons were excluded and only the Agulhas and South African Kelp Forest Ecosystem types meet this threshold/passed this filter.

Without the application of the “1% rule”, the three regional kelp forest types qualify for the Vulnerable (VU) threat category (see: Table 3. 2) and the uncorrected AOO for the combined South African Kelp Forest qualified for LC (Figure 3. 8). If the recommended correction is applied, the AOO (grid cells of the 1% rule) for Namaqua and Cape Kelp Forests becomes zero and becomes four for the Agulhas and the combined South African Kelp Forest type. The IUCN provides little guidance on how to proceed in the latter case, therefore in this study, a range of threat categories was reported (VU-CR). If the correction is applied and an AOO of zero which is considered to be less than 2 is attained (see: Table 3. 2 for assigning threat category), this could qualify the assessed ecosystem as Critically Endangered (CR) in contrast to VU if the correction factor is not applied.

3.5.5.3. Criteria C: Disruption of abiotic processes and interactions

Under criterion C1, the Namaqua Kelp Forest qualified as VU whereas the Cape and Agulhas and the South African Kelp Forest types qualified as EN. However, under criterion C3, the four kelp forest ecosystem types were assessed as VU. This is an indication of sensitivity to the different periods of assessment. The subcriteria (C1 and C3) results (Table 3. 7) are as a consequence of the deleterious synergic anthropogenic effects on the abiotic attributes (Table 3. 5) that have a defining role in each ecosystem ecological processes since the past 50 years and since 1750 respectively.

Table 3. 7: Sub-criteria C1 and C3 classification results for the three ecosystem types and the combined South African kelp ecosystem.

Ecosystem type	IUCN RLE Criterion C1 (last 50 years)	IUCN RLE Criterion C3 (since 1750)
Namaqua Kelp Forest	Vulnerable (VU)	Vulnerable (VU)
Cape Kelp Forest	Endangered (EN)	Vulnerable (VU)
Agulhas Kelp Forest	Endangered (EN)	Vulnerable (VU)
South African Kelp Forest	Endangered (EN)	Vulnerable (VU)

3.5.5.4. IUCN RLE summary of all assessed criteria results

The IUCN RLE threat status is summarised for four all three ecosystem types in Table (3.8-3.11). Where there was insufficient data the result was reported as Data Deficient (DD). The subcriterion A1 was used to assess all kelp forest types as LC. The different sub-criteria under criterion B were used to assess the three ecosystem types Namaqua, Cape and Agulhas Kelp Forst as EN under subcriterion B1 (Table 3. 8, Table 3. 9 and Table 3. 10), and LC for the South African Kelp Forest types (Table 3. 11). Namaqua and Cape Kelp Forest Ecosystems were assigned (VU-CR), the Agulhas and South African Kelp Forest types were assigned EN under subcriterion B2. Under subcriteria C1, Namaqua was assessed as VU and all other three as EN; and for subcriterion C3 all four kelp forest ecosystem types were assessed as VU.

It is important to note that A1, C1 and D1 consider only the past 50 years, A2, C2 and D2 also consider future likely change and A3, C3 and D3 consider change since a 1750 baseline. B1 uses a minimum convex polygon and B2 10 x 10 km grid cells. Criteria B was the most sensitive to the scale of ecosystem delineation.

Table 3. 8: Overall ecosystem threat status derived from the highest IUCN RLE category for the Namaqua Kelp Forest Ecosystem.

Criteria	A	B	C	Overall
Subcriterion 1	LC A1	EN B1a(ii)	VU C1	EN B1a(ii)
Subcriterion 2	NE A2a, b	VU to CR	NE C2	
Subcriterion 3	NE A3	NE B3	VU C3	

Table 3. 9: Overall ecosystem threat status derived from the highest IUCN RLE category for the Cape Kelp Forest Ecosystem

Criteria	A	B	C	Overall
Subcriterion 1	LC A1	EN B1a(ii)	EN C1	EN B1a(ii), C1
Subcriterion 2	NE A2a, b	VU to CR	NE C2	
Subcriterion 3	NE A2a, b	NE B3	VU C3	

Table 3. 10: Overall ecosystem threat status derived from the highest IUCN RLE category for the Agulhas Kelp Forest Ecosystem.

Criteria	A	B	C	Overall
Subcriterion 1	LC A1	EN B1a(ii)	EN C1	EN B1a(ii), C1
Subcriterion 2	NE A2a, b	EN B2a(ii)	NE C2	
Subcriterion 3	NE A3	NE B3	VU C3	

Table 3. 11: Overall ecosystem threat status derived from the highest IUCN RLE category for the combined South African Kelp Forest Ecosystem.

Criteria	A	B	C	Overall
Subcriterion 1	LC A1	LC	EN C1	EN B1a(ii) B2a(ii)
Subcriterion 2	NE A2a, b	EN B2a(ii)	NE C2	
Subcriterion 3	NE A3	NE B3	VU C3	

3.6. Discussion

This study pioneered the assessment of the threat status of South African kelp forests and serves as a guideline to improve knowledge on pressures and the threat status of these important ecosystems. Three biogeographical subtypes were distinguished (see: chapter 1: section 1.1.4) namely; Namaqua, Cape and Agulhas Kelp Forests and an additional combined broader South African Kelp Forest type included to investigate the effects of scale on the assessment (Figure 3. 4). Advancing from the previous NBA 2011, cumulative pressure impacts were mapped to assess ecosystem degradation (Figure 3. 6) and a pressure matrix (Table 3. 6) that ranks the identified pressures from the highest to lowest intensity was produced. Three IUCN RLE criteria (A, B: Table 3. 2 and C: Table 3. 3) were applied to provide the threat status of the South African kelp forest ecosystem types. Key ecosystem components, pressures and their interactions were considered within a conceptual model to support defining of ecosystem collapse (Table 3. 4 -Table 3. 5 and Figure 3. 5). The results of this study show sensitivity to the different assessment criteria, scale of ecosystem delineation and assessment approaches. In overall, the recognised threat status of the South African kelp forest ecosystem types from IUCN RLE approach is EN to risk of ecosystem collapse.

3.6.1. Ecosystem model and ecosystem collapse

The conceptual model developed for South African kelp forest ecosystems (Figure 3.5) summarises the complexity of kelp forest ecosystems into a schematic that shows the key abiotic and biotic components and interactions governing these kelp forests. It represents the intrinsic and extrinsic factors and captures the ecological role of kelp forests in adjacent linked ecosystems. This includes both nutrients imported into the kelp ecosystem from external ecosystems and nutrients exported (debris, detritus, and drift) to adjacent shallow and deep ecosystems which are driven by upwelling and downwelling (Bolton and Levitt, 1987 and Dyer et al., 2019). The model highlights pressures including anthropogenic and climate change impacts based on a synthesis of global and local literature (Table 3. 5).

It is a requirement of the IUCN RLE to develop a model to signify key processes and interactions within the ecosystem being assessed (Rodriguez et al., 2015; Bland et al., 2017a). As such, this model of kelp forests was useful for developing a better understanding of key interactions and processes governing the kelp ecosystems and valuable in terms of elucidating various pathways that may lead to ecosystem collapse (Figure 3. 2 and Figure 3. 5). Additionally, this model (Figure 3. 5) is comparable to other ecosystem models developed for

IUCN RLE ecosystem assessments e.g. the giant Alaskan kelp forest (Estes et al., 2009; Keith, 2013), ecological processes relevant to the assessment of risk for the Meso-American Reef (Bland et al., 2017b) and the southern Benguela (Bland et al., 2018) which were also developed to help define ecosystem collapse under the criteria that pertain to ecosystem functioning. Ecosystem models are most suitable for the assessments of criteria (C, D and/or E) which pertains to ecosystem functions or processes, e.g. Murray et al. (2017) did not develop a conceptual model to assess risk using criteria (A and B) which pertains to geographic distribution.

Only one generic model (Figure 3. 5) was developed for South African kelp forests i.e. the model does not separate the Namaqua, Cape and Agulhas Kelp Forest ecosystems, although differences in kelp forest ecosystems composition between these regions are recognised (Field et al., 1977; Blamey et al., 2010). It is understood that their processes at a coarse-scale can be similar but, they may differ over finer spatial scales including that of ecoregion, sub-regions and at a more local scale. An example of local-scale variations albeit in a different broad ecosystem type in the same region is the two different community types around two nearby islands (Marcus and Malgas islands) adjacent to the Cape Kelp Forest. These two islands have vastly different subtidal benthic communities due to the presence (Malgas Island) and absence (Marcus Island) of rock lobsters (Barkai, 1987). Similarly, the recent shift in rock lobsters leading to a regime shift in the Agulhas Kelp Forest Ecosystem, thereby creating contrasting benthic communities within the same ecosystem type is a further example (Blamey et al., 2010; Blamey and Branch, 2012). Other key abiotic and biotic differences between the three kelp forest ecosystem types include differences in productivity, low oxygen, environmental variability and pressures (Table 3.4) (Bolton and Levitt, 1987; Bolton and Stegenga, 2002; Griffiths et al., 2010; Bolton et al., 2012; Blamey et al., 2014). To strengthen the understanding of pathways and interactions of South African kelp ecosystem types, separate finer scale conceptual models for each of the three kelp ecosystem types may need to be developed.

To define ecosystem collapse for South African kelp forests, the concept that no ecosystem is completely free of change was recognised (Figure 3. 2). At various micro and/or macro scales, ecosystems may always be changing as a result of alterations of the different components that constitute them (Bland et al., 2017b). Therefore, it is paramount to establish thresholds of ecosystem collapse for different ecosystems. As listed below, the results of this current study offer more than a single state “spectrum” that could qualify as collapsed states for South African kelp forest ecosystems. These are described below:

- a) Firstly, if the point where the mapped distribution based on the range size matrices (the EOO and AOO; Murray et al., 2017) of the three types declines to zero i.e. if there are no more kelp forests in South Africa, this would represent ecosystem collapse.

The conceptual model shows links between external anthropogenic actions which may lead to habitat degradation and overexploitation of key species which can have negative effects on the biotic and abiotic components of the kelp forests. These anthropogenic actions may augment climate change impacts and lead to decline and/or shift of kelp forests as seen in other parts of the world (e.g. Connell et al., 2008; Krumhansl et al., 2016; Franco et al., 2016; Filbee-Dexter and Wernberg, 2018; Wernberg and Filbee-Dexter, 2019; Terada et al., 2019). This study has collated literature on regime shifts that support and show that the above point is possible, as there also have been vast areas in some parts of the world that were previously dominated by kelp forests and have been transformed to alternative states e.g. urchin barrens and algal turfs (Steneck et al., 2002; Ling and Johnson, 2009; Ling et al., 2009; Filbee-Dexter and Scheibling, 2014; Filbee-Dexter and Wernberg, 2018). Further, this reduction in geographic distribution state is similar to the definition of collapse ecosystem state for the Alaskan giant kelp forests (Keith, 2013).

- b) Secondly, if the point where key biotic components such as the engineering species which include *E. maxima* and *L. pallida*, west coast rock lobsters, key predatory fish species and kelp forest associated bacteria become functionally extinct and the abiotic (e.g. temperature, nutrients and substrate for attachment) components are degraded (Field et al., 1977; Branch, 2008). Then ecosystem collapse would be reached.

This definition is similar to that developed by Keith (2013) for the giant Alaskan kelp forests, which states that ecosystem collapse would be reached if a point where the two variables (sea otters and kelp densities) with significant trophic roles become functionally extinct (Keith, 2013; Estes et al., 2015). This has also been realised in the kelp forests of Tasmania (Connell et al., 2008; Ling et al., 2009) and this definition is parallel to that of the corals of the Meso-American Reef by Bland et al. (2017b) which are defined as a point where the main structural element of reef extent declines to zero and live coral reef cover declines to 0-5% throughout the ecosystem. Although in this study such a specific threshold was not set.

Bland et al. (2018) assessed the Southern Benguela shelf, an expensive system that is adjacent to the South African kelp forest ecosystem types. They set a threshold of collapse as a point

where the ecological role of a functional group (e.g. historically dominant small pelagic fish, predatory fish, seabirds or seals) is no longer met in a food web e.g. decline of prey and its support to predators. This definition was based on the transformation or regime shift in the 1970s from a state that was previously dominated by sardine and anchovy into an alternative state characterised by functionally low biomass of sardine, anchovy and associated predators and an explosion of jellyfish, pelagic goby and increase in the connection of benthic and pelagic energy exchange that has persisted to date (for further details of this assessment see: Bland et al., 2018).

- c) Thirdly, if all the key ecological and/or socio-economic services derived from South African kelp forest ecosystems have been drastically changed or “very severely degraded”, the ecosystem can be considered collapsed.

This above definition or indication of ecosystem collapse is directly and indirectly linked to the abiotic, biotic components and ecosystem services perspective e.g. socio-economic as well as ecological services. There is international work that has reported on the important economic services derived from kelp forest ecosystems (Vasquez et al., 2014; Bennet et al., 2016; Blamey and Bolton, 2018), hence, losing these established services could be a plausible indicator of ecosystem collapse.

3.6.2. Pressures and threats

This study has pioneered the assessment of South African kelp forest ecosystem types from various pressure impacts (Table 3. 4). Thirteen pressures and their different extent and intensities of impacts were overlaid over three kelp forest types (Table 3. 6). The assessment of ecosystem degradation (Figure 3. 6) reflects the cumulative impacts experienced to date. Of the identified and measured pressures, line fishing (commercial and recreational) has the highest impact on the South African kelp forests due to the widespread occurrence of this pressure and known long term impacts of removal of key fish species (Table 3. 6). Overfishing is of global concern in many studies because fishing is the most widespread anthropogenic impact in the marine environment (Steneck, 1998; Tegner and Dayton, 2000; Griffiths, 2000; Griffiths et al., 2010; Bellchambers, 2010; Coll et al., 2016; Perez-Matus et al., 2017). Kelp harvesting has the lowest impact score for the kelp forests of South Africa, in part due to the practice of precautionary harvesting (Levitt et al., 2002; Anderson et al., 2003; Rothman, Anderson and Smit, 2006). In other parts of the world over-harvesting of kelp forests have led to their degradation, caution and sustainable harvesting of the kelp resource has been advised because of the importance of these ecosystem engineers (Levitt et al., 2002; Frangoudes and

Garineaud, 2011; Guiry and Morrison, 2013; Elston, Anderson and Price, 2015; Borrás-Chavez et al., 2016; Angus, 2017; Monagail et al., 2017).

Also, there is other evidence noting the seriousness of the fishing pressure which include substantial declines in abalone as a result of commercial and illegal harvesting (Lambrechts and Goga, 2016). Additionally a shift in rock lobster and establishment into areas it was previously not dominant in (Tarr, Williams and Mackenzie, 1996) and as a result of this invasion a cascade of effects, including a decline in sea urchins (Blamey, Branch and Reaugh-Flower, 2010; Blamey and Branch, 2012). Establishing links between these environmental changes to specific pressures e.g. global warming has been confounded by the overexploitation of line fish, coastal development, ports and harbour development, kelp harvesting and mining, which all may lead to the transformation of ecosystem functions and diversity (Mumby et al., 1995; Levitt et al., 2002; Pulfrich et al., 2003; Atkinson and Sink, 2008; Ojemaye & Petrik, 2019).

Although global climate change was not included in the threat status assessment, it is understood to be among the key drivers of change in biodiversity, including kelp forest ecosystem degradation as has been reported in multiple international studies (Voerman, Llera and Rico, 2013; Raybaud et al., 2013; Wernberg et al., 2016; Krumhansl et al., 2016; Assis, Araújo and Serrão, 2017). The omission of climate change data from South Africa's assessments was mainly due to the knowledge gap in understanding and distinguish the causal link between South African kelp forest ecosystems perturbations and climate change noted above (Sink et al., 2012; Mead et al., 2013; Blamey et al., 2013; Sink et al., 2019). Hence, future assessments need to consider climate change data and investigate the use of different time scales in the assessment.

As illustrated in Figure 3. 5, threats associated with climate change can augment the current cumulative impact of the identified pressures (as many studies discussed in this work reveal) and possibly lead to the assignment of worse threat categories (especially under criteria C and D). This would be possible if quantitative spatial data for current and future climate were available at the resolution needed for South African kelp forests assessment. Based on the criteria that deals with geographic distribution, it is plausible that the Namaqua Kelp Forest (on the bases of having the smallest EOO) could be impacted the most by climate-driven degradation, because of its smaller extent of occurrence (EOO) (Figure 3.7) i.e. Namaqua has

a higher risk spreading factor compared to the other two types (Murray et al., 2017). However, as the Namaqua kelp forest map extends into Namibia, further mapping is needed, and a regional assessment should be conducted. An identified local knowledge gap in South Africa is the extent and direction of ecosystem alteration as a result of global climate change (Mead et al., 2013). Another challenge is that the only comprehensive and historical study of the Cape Kelp Forest dates back to the 1970s (Field et al., 1977;1980), with little replication since, except for the Agulhas Kelp Forest Ecosystem (see: Blamey et al., 2010).

3.6.3. IUCN RLE threat status

The emergent IUCN RLE offers a technical basis for local, national or subnational assessment of ecosystem status (Rodriguez et al., 2015). Of the five criteria developed for assessing the risk of ecosystem collapse, three criteria were applied, declining distribution (criterion A), restricted distribution (subcriteria B1 and B2) and degrading abiotic environment (subcriterion C1 and C3). The IUCN RLE recommends the highest status of risk to ecosystem collapse is reflected in the final results for any ecosystem assessed. Therefore, the overall plausible risk of ecosystem collapse for Namaqua, Cape and Agulhas Kelp Forest Ecosystems from this assessment, without considering the correction factor “1% rule” (Murray et al., 2016) which may be considered under subcriterion B2 (Table 3.2 and Figure 3.9), is EN (VU-EN). A key finding from this study is the range of results that vary according to the assessments approaches used and the relevant criteria applied such as the criteria assessing for risk of collapse based on declining distribution, restricted distribution and abiotic functional processes.

Sub-criterion A1 pertains to geographic decline in the last 50 years and as there is no known decline in the extent of any type of the kelp forests in South Africa, this subcriterion resulted in LC for all four considered types. There was insufficient data to assess the kelp forest ecosystems using sub-criteria A2a, A2b and A3 which pertain to geographic decline over the next 50 years, any 50 years (including past present and future) and since approximately the year 1750, respectively. However, in some parts of the world, there has been reported decline in the distribution of kelp forests owing to climate change and pollution (Connell et al., 2008; Branch, Bustamante and Robinson, 2013; Krumhansl et al., 2016; Assis, Araújo and Serrão, 2017). This may be possible for South African kelp forest ecosystems despite their recent expansion along the south coast that is generally attributed to gradual cooling (Bolton et al., 2012; Mead et al., 2013; Blamey et al., 2013). There have been recent kelp die-offs along the

south coast, possibly in association with warm water events that have been reported to SANBI's marine unit and this highlights the importance of monitoring of kelp extent.

Criterion B focuses on ecosystems of limited geographic extent and this is relevant to all South African kelp forests ecosystems which met thresholds in terms of limited extent but also the additional requirement of an observed continuing decline in environmental quality (Figure 3.6). The assessment of ecosystem degradation (Figure 3. 6) reflects the cumulative impacts and associated changes in ecosystem condition experienced to date, a finding that has been shown both locally (Bolton et al., 2012; Blamey and Bolton, 2012; Blamey et al., 2014; Ojemaye and Petrik, 2019; Lindberg et al., 2019 *in review*) and globally (Lauzon-Guay and Scheibling, 2010; Ling et al., 2011; Wernberg et al., 2016; Filbee-Dexter et al., 2017). Biodiversity features that are limited to a small distribution range are more likely to be degraded by a minimal number of threats, whereas broad or widely distributed ecosystem types are buffered from threats by spatially scattering the threats through multiple locations (Murray et al., 2017; Keith, Akçakaya and Murray, 2018).

The size of the spatial distribution of ecosystems influences how they are organised relative to the spatial extent of different pressures (Bland et al., 2017a). By measuring the EOO and AOO the spatial spread of threats and the likelihood of ecosystems being threatened by a single threatening process were assessed. The Namaqua, Cape and Agulhas Kelp Forests are all small and narrow linear systems (Figure 3. 6) that qualified as EN under criterion B1 which considers the EOO. The small range and linear shape of these ecosystems along shores make them more susceptible to risk (Murray et al., 2016; 2017). This study showed that this criterion is affected by the scale of ecosystem classification and mapping. The threat status varied substantially when the ecosystems were combined into a single type (LC) compared to when they were assessed as different types, a finding that has significance for global, regional and national RLE assessments. This finding demonstrates that different scales have an impact on the overall results when the IUCN RLE protocols are applied.

Under subcriterion B2, the Agulhas Kelp Forest was assessed as EN whereas both the Namaqua and Cape Kelp Forest were assessed as (VU to EN even CR if the correction factor is applied) to the risk of ecosystem collapse as a function of the AOO because they are made up of larger numbers of small patches which contribute a negligible risk-spreading effect (Murray et al., 2016). It is important to note that applying the correction factor (Figure 3.9) leads to the loss of two South African ecosystem types namely; the Namaqua and Cape Kelp Forests, therefore,

it is not supported to be indicative of the threat status of the South African kelp forest ecosystem types in this study. This is in part because, risk spreading of an ecosystem made up of small patches is not comprehensively understood for qualitative application, as the IUCN RLE gives little guidance on this (Murray et al., 2016; Bland et al., 2017a; Murray et al., 2017; Keith, Akçakaya and Murray, 2018). Despite its dominant role in red-list assessments, the AOO is considered the most contentious of the standard range-size metrics for predicting the risk of ecosystem collapse, because of its scale sensitivity (Keith, Akçakaya and Murray, 2018), a finding re-iterated in this study (see: Table 3.18 to 3.11). This raises concern about the suitability of this subcriterion or its currently specified thresholds to assess the risk of ecosystem collapse to any ecosystem that naturally occurs in small patches and at sub-global scales. To improve the accuracy of criterion B, there is a need to develop secondary thresholds for the subcriteria conditions, instead of only having word definitions e.g. B2a(ii) observed continuing decline (Table 3. 2).

Two sub-criteria under criterion C were applied to assess abiotic degradation of all units of assessment. The relative proportion of each ecosystem type within the four ecological degradation categories (Figure 3. 6) was used to derive ecosystem threat status. Subcriterion C1 assessed this degradation over the past 50 years, revealing a worse threat status than sub-criterion C3 which assesses abiotic degradation since 1750 (Table 3. 3). Skowno et al. (2019) also used C3 for the assessment of NBA 2018 based on pre-colonial 1750 historical reference condition, a period that corresponded approximately to the earliest onset of global industrial-scale exploitation of ecosystems. Sink et al. (2019) noted that the actual data used to assess degradation only spanned the last 50 years and results vary when these two sub-criteria are assessed. As such, criterion C1 may have been more appropriate for assessing South African marine ecosystems.

The results of subcriterion C1 (EN) were not sensitive to the scale of ecosystem classification in line with the focus of impacts on ecosystem functioning. However, criterion C was found to be sensitive to differences in pressures and extent of degradation of the ecosystem types, hence the different threat status for Namaqua (under C1 and C3) which is supported by the results in Figure 3. 6. Further, criterion C is sensitive to impact weights, inaccuracies in pressure mapping and calibration of ecosystem degradation and more work is needed to improve accuracy in this context. To better inform future assessments of ecosystem condition, there is a need for ground-truth data to validate the mapping of ecosystem degradation and

development of ecosystem indicators that are similar to those developed in Bland et al. (2018) for the southern Benguela.

The combined effects of line fishing, mining, ports and harbours, abalone harvesting, west coast rock lobster fishing, coastal development and kelp harvesting have been shown to have cumulative deleterious effects on the ecological condition of the kelp forests although they have variable intensities and spatial extents. The ecological condition results (Figure 3.6) were useful for deriving relative severity and the hierarchy of different pressures based on their intensity (in line with the four conditions previously developed in the NBA 2011; Sink et al., 2012). Some of these pressures have been reported to cause deleterious effects on kelp forest ecosystems, disturbing the ecological trophic levels, diversity, competition and predation by carnivores and grazers (Brill and Raemaekers, 2013; Pérez-Matus et al., 2017) and reduce ecosystem resilience (Ling et al., 2009).

Criterion D is based on disruption of biotic processes and it was not evaluated in this assessment because of insufficient information to comprehensively assess these risks within the scope of this thesis. Therefore, this criterion was considered Not Evaluated (NE) along with criterion E. However, because criterion D has the same thresholds as criterion C, both abiotic and biotic processes were covered under criterion C. Under this criterion, Bland et al. (2018) used biotic indicators and classified the Southern Benguela shelf ecosystem as EN under subcriterion D1 based on the fishing pressure on predatory fish, breeding pairs of sea birds and counts of Cape fur seal cubs between 1960 and 2015. The trophic cascade effects of overfishing modelled by Blamey, Plagányi & Branch, (2013) would perhaps lead to a threat status of the Agulhas Kelp Forest Ecosystem that mirrors the above results (EN under criteria D), however, their study covers only part of the Agulhas Kelp Forest Ecosystem and cannot be extrapolated for the entire extent of the ecosystem. As there is evidence of regime shift and biotic degradation in kelp forests, it is important that in future assessments this criterion is better included.

The range expansion of the west coast rock lobster (Cockcroft et al., 2008) and the associated knock-on ecosystem effects (Blamey et al. 2010), as well as the recent eastward expansion of *E. maxima* in the Agulhas Kelp Forest Ecosystem, are examples of the changes in the biotic environment. These scenarios are likely to affect the risk to ecosystem collapse differently. For example, the eastward shift in rock lobster had negative consequences for grazers such as urchins and abalone and resulted in a shift in ecosystem state (Blamey et al. 2010; Blamey and

Branch, 2012) that persists. In contrast, an expansion of kelp (e.g. Bolton et al. 2012) might reduce the risk to ecosystem collapse by expanding the ecosystem and its biotic processes.

Limitations and future research

The IUCN thresholds are more stringent than the thresholds developed by South Africa for NBA 2011 (Sink et al., 2012; 2019; Skowno et al., 2019). The NBA 2011 threat status is based on the ecosystem proportion still intact or in its natural condition. Considering that less than 5% of all three ecosystem types that were assessed were in good ecosystem condition, the Namaqua, Cape and Agulhas Kelp Forest Ecosystems would all have been assigned the threat category of CR to the risk of ecosystem degradation. Even when the three ecosystem types are viewed as one single ecosystem type, the South African Kelp Forest Ecosystem would have qualified as CR because the proportion in good ecological condition (see: Figure 3.6) remains less than the 20% biodiversity target (Sink et al., 2012). In comparison to the NBA threat status, the IUCN RLE threat status that is less pessimistic.

The NBA 2018 was largely based on subcriterion C3 but conducted a supplementary assessment to consider ecosystems with limited geographic extents that were not assessed as threatened under C3. Note that as all three kelp forest ecosystem types were assessed as VU, their extent was not further considered by Sink et al (2019). In the supplementary assessment, only the most severe thresholds under criterion B (i.e. that qualified for CR under EOO and AOO) were applied but any qualifying ecosystem types were only considered to be VU (see: Sink et al., 2019 for a comprehensive explanation of the supplementary assessment). This is the reason why the NBA 2018 assigned a different threat status to the highest level of threat (EN) deemed plausible in this study. Both the broader expert group assessing all marine ecosystem types and the kelp expert group endorsed this approach (Sink et al., 2019). This further highlights the need for additional work to refine assessment of degradation including calibration of categories and further experimentation with multiple thresholds at finer scales of ecosystem classification and mapping.

This study recommends the IUCN RLE community provide comprehensive training and guidance on the application of the RLE assessment protocols. The application of the correction factor AOO subcriterion B2 is perhaps not appropriate for application for ecosystems that are small and highly fragmented. Further international collaboration to experiment with different standard grids is recommended as it may be appropriate to apply different grids for different

scales. Nevertheless, this study acknowledges that in part the sizes of the South African kelp forest polygons are underestimated due to mapping limitations (see: Chapter 2).

In conclusion, future studies should prioritise the identified limitations to strengthen the use of this emergent IUCN RLE approach. This is paramount because many marine ecosystems including kelp forest are remote, hindering early detection of any symptoms of change. There is little to no information on kelp forest ecosystem types and associated communities for the Namaqua Kelp Forest compared to Cape and Agulhas Kelp Forests. There is intermediate knowledge regarding the Agulhas Kelp Forest. There are not enough data to determine and project back into the historic “original” state of kelp forests and in this case, a historical baseline is probably the 1970s for some regions e.g. parts of the Cape Kelp Forest Ecosystem and Agulhas Kelp Forest Ecosystem. However, it has been established (both anecdotally and scientifically) that in some regions the kelp forest ecosystems have lost some of their biotic components (Anderson et al., 2006; Bolton et al., 2012; Watermeyer et al., 2016), with further research needed to better understand the extent and implication of changes. Sentinels sites are needed for long term monitoring of kelp forests, South Africa is moving towards addressing this by putting data stations around the False Bay although this is mainly for oceanography (in comms, R.J Smit, UWC) with opportunities to include key ecosystem components as their sites overlap with the Agulhas and Cape Kelp Forest Ecosystem.

Chapter 4: Thesis Synthesis

This study had three main aims, the first was to produce a contemporary map of the South African kelp forests for inclusion in the National Map of Marine Ecosystems types. The second was to provide the first threat status assessment for South Africa's kelp forest ecosystems to inform South Africa's National Biodiversity Assessment (Skowno et al., 2019). The third goal was to distil key lessons to inform future ecosystem mapping and assessment. These goals were achieved through the application of the contemporary remote sensing techniques, the emerging IUCN RLE protocols (Rodriguez et al., 2015; Bland et al., 2017a) and comparing its implications respectively. The derived map and classification of kelp forest ecosystems was a key input into the marine ecosystem assessment (Sink et al., 2019) which provided a valuable contribution in informing current interpretations of marine ecosystem threat status and future national and international research efforts to assess marine ecosystem risk.

To produce a comprehensive map of South Africa's kelp forests, the spectral index of aquatic vegetation was extracted from a high spatial and spectral resolution Sentinel-2 (A and B) satellite platform. The new mapping procedure developed a Normalised Difference Vegetation Index (NDVI) threshold that captures both submerged and surface forming kelp forest. The contemporary kelp forest map was validated using expert knowledge, literature review, field surveys and previous maps. The second step of validation included a supervised maximum likelihood classification and error matrix which attained a mapping accuracy of 76%.

The product from this developed mapping technique was a high-resolution fine-scale map of South African kelp forests. A total of 3700 polygons were mapped with kelp forests extending along 1300 km of the coastline, a 400 km extension of the previous map developed by Anderson et al. (2007). When compared, this new map concurs with the kelp forest map developed in the early 1990s along Cape Hangklip (Tarr, 1993). However, the map also addressed gaps in previous mapping efforts (Cape Point, between the Groen and Spoeg Rivers and at Saldanha), mapped kelp forests that have never been mapped before including eastward expansion of *E. maxima* (Bolton et al., 2012) and successfully detected and included subtidal kelp forests.

Although the detection of kelp forests along the coastline of the Northern Cape was limited by turbidity, the new kelp map captured all the kelp polygons from the previous maps and improved upon the detection and mapping of additional kelp forests. Further, the limitations of the remote sensing platform in a favourable environment are reported to be a depth of 20 meters

(Johnson, 2015) but this depends on clear, calm and non-turbid water. Therefore, kelps existing below this depth have not been captured on the map, underestimating the offshore extent of these ecosystem types which have known depth limits of ca. 30 m. Improving the mapping of the outer deeper extent of kelp forests is a recognised research priority. This limitation is one mapping technicality that is likely to affect the result of the geographic distribution of kelp forests which is a key component of threat assessment under criterion B of the IUCN RLE. This challenge can perhaps be addressed by coupling use of Remotely Operated Vehicles (ROV's) along with further innovation in satellite remote sensing to develop a more three-dimensional overview of the kelp forest breadth and length along the coast.

While this study captured the spatial extent of South African kelp forests within current technological and environmental constraints, the complexities associated with applying remote sensing in the marine environment which is subject to many limitations were revealed. These limitations influence the applications of the IUCN RLE in which spatial extent is a key assessment component. Several challenges encountered in this study are similar to those experienced by other global studies that used remote sensing to map kelp forests. For example, although remote sensors have been effective in detecting kelp forests, water turbidity remains a challenge (Anderson et al., 2007; Casal et al., 2010; Bennion et al., 2018) and attenuation effects of the water column and suspended matter are common constraints (Casal et al., 2010; Cavanaugh et al., 2011).

The high spatial and spectral reflectance of Sentinel-2 (A and B) satellite has more to offer for mapping and monitoring of South African kelp forest ecosystems and a recent study (Nijland, Reshitnyk and Rubidge, 2019) recommends exploring its use. Additionally, the developed mapping approach can easily be modified to map other terrestrial and marine vegetation ecosystems within the technical limitations of the S-2 (A and B) platform. Further, with international collaboration, the NDVI mapping technique can be modified and applied to map the entire extent of kelp forest along the African continent and provide a seamless kelp forest map that includes in countries such as Namibia, Angola and Senegal and the coast of Mauritania (Wernberg and Filbee-Dexter, 2019). The ESA Sentinel-2 hub offers 13 bands at three different spectral resolutions (Drusch et al., 2012) which warrant experimental application in mapping algae although such additional exploration was beyond the scope of this study. Future, South African efforts should investigate the application of this technology for biomass estimation and more frequent monitoring to inform management actions of kelp ecosystems and resources.

To provide the first indication of the threat status of South Africa's kelp forest, this study advanced from the former South African NBA (the NBA 2011, Sink et al., 2012) approach and aligned to the emerging IUCN RLE protocols. In line with the IUCN RLE protocols, a conceptual model of the South African kelp forest ecosystems was developed to represent the key components and processes that constitute these ecosystems. The ecosystem conceptual model was used to identify various pathways of ecosystem collapse based on the key components and interactions. Cumulative pressure mapping of thirteen identified pressures was conducted to measure the intensity of impacts across ecosystem types by calculating relative patterns of ecosystem degradation as an input into criteria B and C of the assessment.

This study identified the key pressures and assessed their cumulative impacts on kelp forests in three ecoregions and at a national scale. A pressure matrix to represent the relative intensity of the impact of each pressure on kelp forests was produced drawing from a summary of the impacts based on South African and international literature. Of the identified and measured pressures, line fishing (commercial and recreational) has the highest impact on the South African kelp forests due to its widespread occurrence and the known long-term impacts of removal of key fish species (Steneck, 1998; Tegner and Dayton, 2000; Griffiths, 2000; Griffiths et al., 2010; Bellchambers, 2010; Coll et al., 2016; Perez-Matus et al., 2017). Kelp harvesting was estimated to have the least impact on kelp forests of South Africa, in part due to the practice of precautionary harvesting but also because this harvesting is not widespread at this stage (Levitt et al., 2002; Anderson et al., 2003; Rothman, Anderson and Smit, 2006, DAFF, 2016). The developed pressure matrix and pressure summary table can be a helpful tool for managers to priorities monitoring and management of anthropogenic activities to protect the services derived from kelp forest ecosystems.

Three IUCN RLE criteria (A, B and C) that consider declining distribution, restricted distribution and abiotic degradation were applied to four units of assessment in this study to assess the risk of kelp forest ecosystem collapse. The threat status of the Namaqua, Cape and Agulhas Kelp Forests were assessed. Also, the three types were combined to be considered as a single broader South African Kelp Forest Ecosystem type which may be more comparable to global ecosystem assessments that have not mapped their ecosystems in such fine resolution to examine the influence of scale. Further, the implications of the different approaches, criteria and assessment procedures were investigated to distil key lessons for current and future assessments.

The discussion of chapter 3 showed that this first assessment of kelp forests would have had a different outcome had the previous NBA 2011 approach been used in this assessment. This is related to the different focus and thresholds involved. The NBA 2011 examined the extent of good condition relative to a 20% biodiversity target whereas the more stringent global RLE assessment using criterion C focuses on the extent of degradation relative to stricter thresholds leading to a lower recognised level of ecosystem risk. Application of the NBA 2011 approach would have led to the three South African kelp ecosystems types and the combined South African Kelp forests types being classified as CR. The highest threat status recognised in this assessment from the IUCN RLE is EN to risk of ecosystem collapse.

The threat status assessment results (Table 4. 1) differed for the four ecosystem types under different subcriteria of criteria B and C and were affected by the scale of ecosystem classification. Under subcriterion B2 in which a correction factor (1% rule) may be applied to determining the AOO of an ecosystem type (Murray et al., 2016), this study found it difficult to assign an appropriate category of threat and suggests a range between (VU to CR) Table 4.1. However, as recognized by other authors (Murray et al., 2016; 2017; Keith et al., 2018) criterion B (subcriterion B2) is sensitive to the scale of ecosystem classification. Mapping and further testing at finer scales are recommended to accommodate ecosystems that are narrow in area (e.g. < 1 km²), linear and fragmented. In summary, this is an important insight into the IUCN system in comparison to the previous South African system, showing that the global criteria centered on degradation were more stringent leading to a more optimistic and probably realistic outcome than previous national approaches.

Table 4. 1: A summary of the threat status results from the IUCN RLE assessment under three criteria A, B and C. Colour coded (green: Least Concern (LC), yellow Vulnerable (Vu), orange: Endangered (EN) and light red: range between Vulnerable and Critically Endangered). An asterisk * represents results that are not considered indicative of the South African kelp forests threat status but are important in understanding the results of applying a correction procedure under B2.

Kelp ecosystem type	A1: (past 50 years)	B1: EOO (polygon km²)	B2: AOO (10*10 km grid)	C1: (past 50 years)	C3: (since 1750)
Namaqua	LC	EN	VU - CR*	VU	VU
Cape	LC	EN	VU - CR*	EN	VU
Agulhas	LC	EN	VU - EN	EN	VU
South African Kelp	LC	LC	EN	EN	VU

Notwithstanding the advantages of international comparability by aligning the threat status assessment of the South African kelp forest ecosystems, there is a need for further reference studies to compare the IUCN RLE thresholds and criteria in terms of their suitability for national objectives of conservation and management of South African ecosystems. More marine assessments covering a broader range of ecosystem types and a wider range of spatial scales is recommended. Experiments with different grid sizes are also recommended as the standardized 10 X 10 km grid may not be suitable for narrow ecosystem polygons mapped at a fine scale.

The application of the international RLE protocols has broad implications for future monitoring and management of South African kelp forests. Despite the identified data limitations and uncertainties linked to criteria and thresholds applied and the range of ecosystem threat status results, the South African kelp forest ecosystem types are threatened with plausible results ranging between Vulnerable and Endangered. In the face of global climate change (Merzouk and Johnson, 2011; Poore et al., 2016; Filbee-Dexter et al., 2019; Mabin, Johnson and Wright, 2019), current research has reported a 38% decline in the worlds kelp forests (Krumhansl et al., 2016) and which may have implications on the established role kelp forest ecosystems have and continue to play on human livelihood (Bennet et al., 2015; Blamey and Bolton, 2018; Filbee-Dexter and Wernberg, 2018). It is paramount that actions are put in place to maintain sustainable exploitation (Edyvane, 2005; Rothman et al., 2006; Borrás-Chavez et al., 2016;

Angus, 2017; Monagail et al., 2017) of these ecosystem concord and sentinel species (Reed et al., 2016).

Although the RLE assessment methodology is still new, it offers multiple criteria to assess the risk of ecosystem collapse and is useful for prioritisation even for areas with data limitations e.g. in this study, criteria D and E were not applied as a result of data and knowledge gaps. The application of different criteria led to different results that may have different uses for different planners and assessment teams. The two criteria (A and B) which deal with geographic extent may have greater relevance to a remote sensing expert who can use these criteria to monitor and report temporal variations in the spatial extent of different ecosystems. Criteria C and D would perhaps be more relevant to modelers, who can advance ecosystem conceptual models and use numerical modelling to assess the risk of ecosystem collapse using more sophisticated quantitative approaches to examine abiotic and biotic processes and changes in ecosystems such as kelp forests. The last criterion although not assessed in this work, is most suitable for marine spatial planners that conduct data extensive assessments and need to project ecosystem condition or distribution into the past or future.

The new kelp forest map of South Africa addressed the recommendation of the previous National Biodiversity Assessment (NBA) (Sink et al., 2012) to develop a high-resolution map of South African kelp forests and did so using a modern, consistent that can be repeated in monitoring efforts. As such, this map is a milestone towards addressing a national research priority and these important ecosystems were able to be included in the National Map of Marine Ecosystems and the NBA 2018 (Skowno et al., 2019). For the first time, this study has provided a reference point in the assessment of the South African kelp ecosystem types using the map developed in this work and the IUCN RLE criteria (A, B and C). However, a key contribution from this study has been the insight into the application of the IUCN criteria for national and global assessments.

This study has accomplished all the objectives that were defined at the outset. The goal of mapping kelp forest ecosystem types was achieved in meeting the first objective of applying a Sentinel-2 based remote sensing method to map kelp forests. To meet the second objective of assessing accuracy, an error matrix was compiled and overall accuracy of 76% was attained. Three different kelp forest ecosystem types were classified reflecting the three main coastal ecoregions where kelps occur to provide a foundation for ecosystem assessment. Information on the ecosystem functioning and threats to kelp forests were collated into a conceptual model

that identified different paths and implications of ecosystem collapse. Using the developed maps of ecosystem types and cumulative pressure mapping, the IUCN RLE protocols were applied to assess ecosystem threat status, and this enabled the final objective of assessing the implications of different criteria and thresholds to be examined.

In conclusion, this study serves as a baseline assessment of kelp forests in South Africa but also distilled lessons for both global and national marine ecosystem assessments and key areas for future improvement. For kelp forests, ground-truthing of ecosystem types and condition is needed and there is scope to assess biotic disruption (Criterion D). South Africa's marine assessments need more research to refine conceptual models for different ecosystem types, measure and calibrate degradation and improve definitions of ecosystem collapse. A range of quantitative approaches that consider a broader range of time scales, including future predictions, are recommended. This study has revealed the influence of the mapping and classification approach, indices of geographic extent and degradation and the scale of ecosystem assessment. International efforts covering a greater diversity of ecosystem types, particularly in the marine realm, and a broader range of mapping approaches, spatial and temporal assessment scales and indices of degradation to strengthen and guide future global, regional and national ecosystem assessments are recommended.

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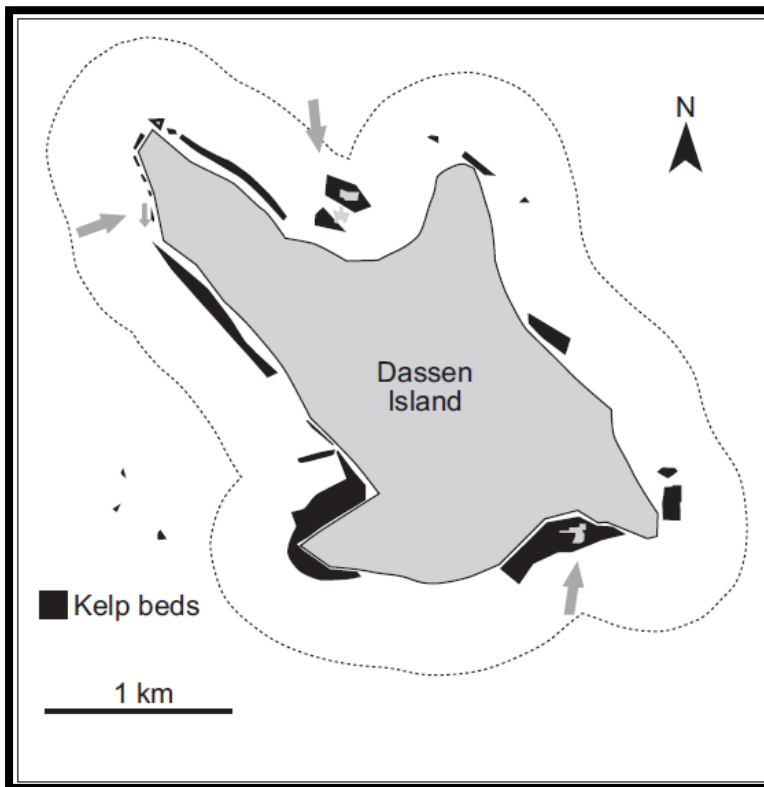
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Appendices

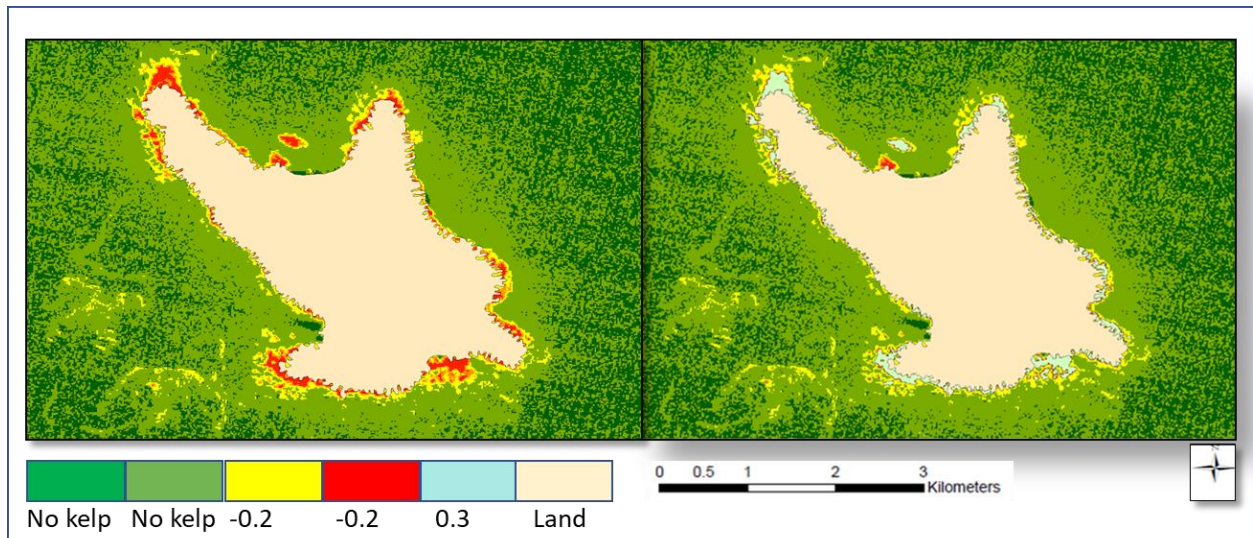
chapter 2

Appendix 2.1



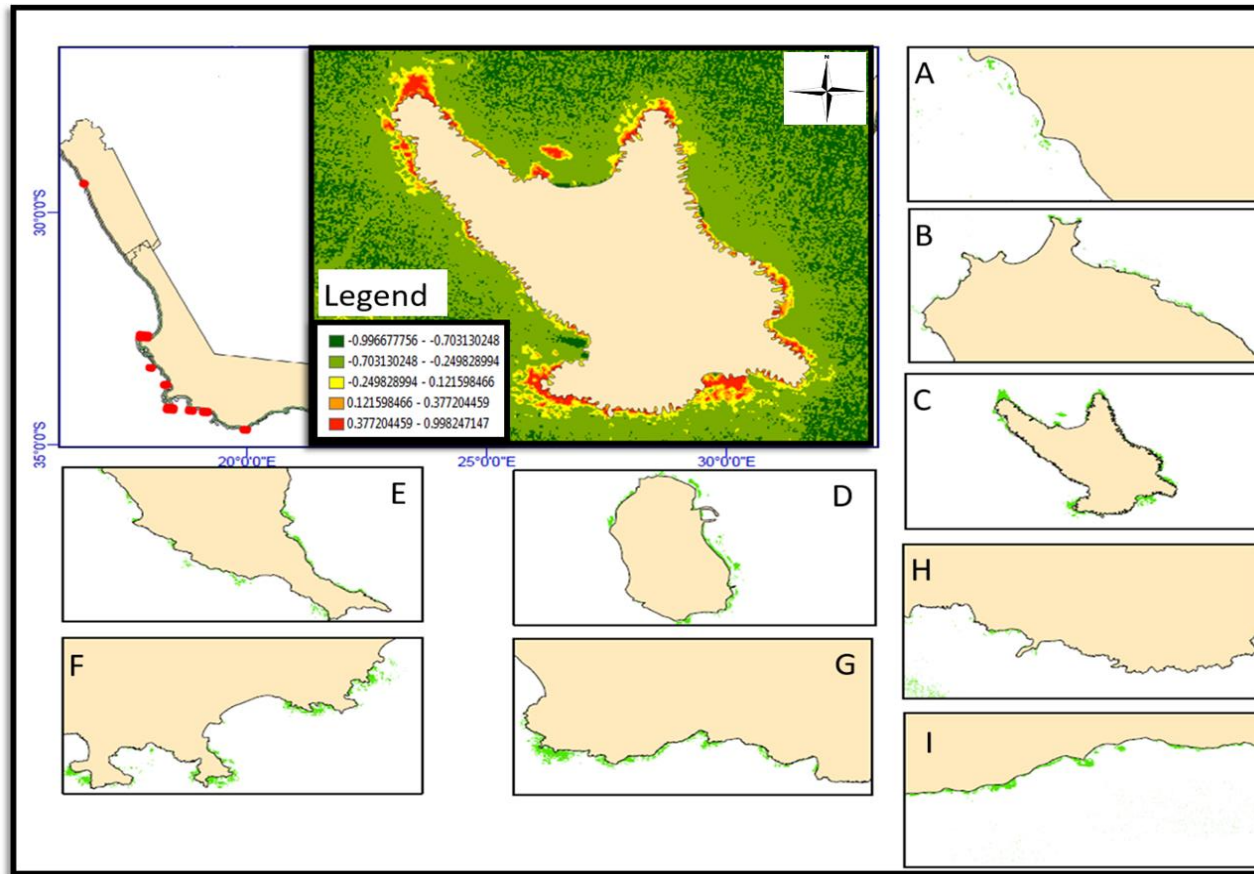
Dassen Island map created by Anderson et al (2007).

Appendix 2.2



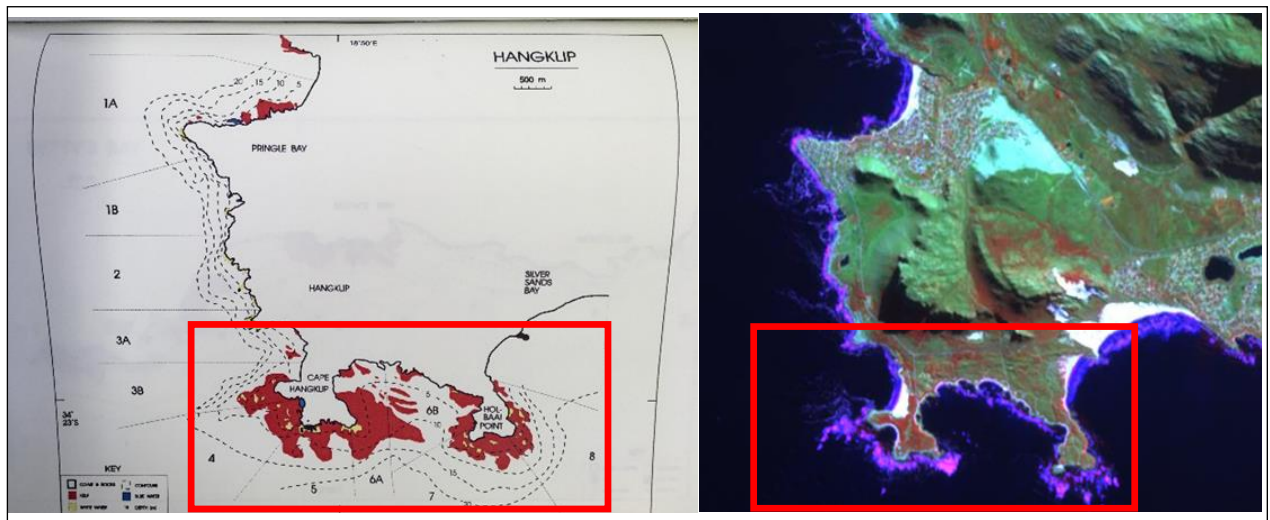
Dassen Island: the land portion is masked using the national vegetation map. Six classes are used to depict the land portion (Beige), three classes: Red, Yellow and Blue depict kelps after applying the -0.2 threshold (diagram on the left side) and the blue band (diagram on the right side) depict the result of the 0.3 threshold.

Appendix 2.3



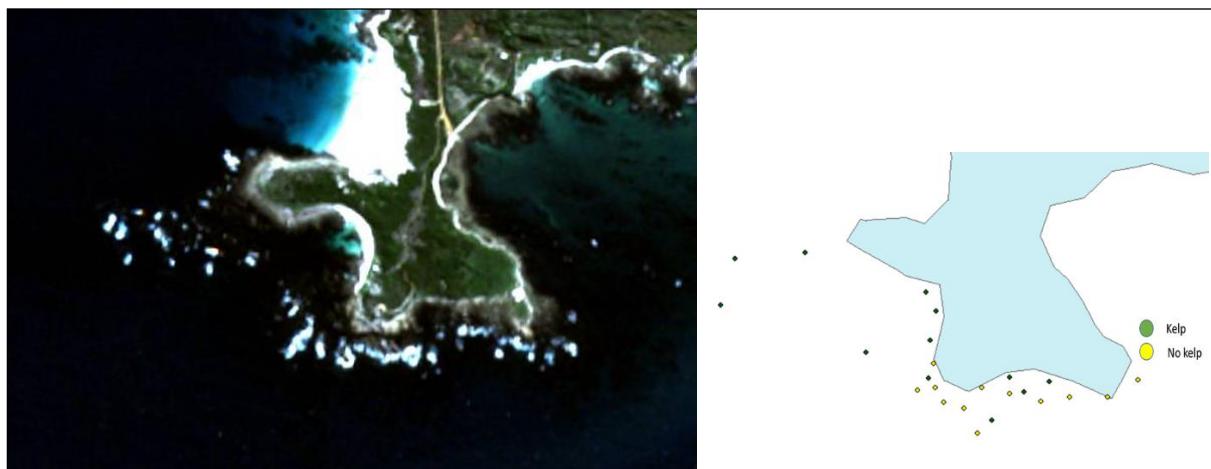
Ranges from north to south (A: Hondeklip Bay, B: Jacobs Bay, C: Dassen Island, D: Robben Island, E: Cape Point, F: Cape Hangklip, G: Betty's Bay, H: Cape Agulhas, I: De Hoop) the land portion (beige) is masked using national vegetation map to the lowest watermark. NDVI range (-0.2 to 1) (yellow, orange and red) covers both submerged kelp forests and surface protruding kelp forests.

Appendix 2.4



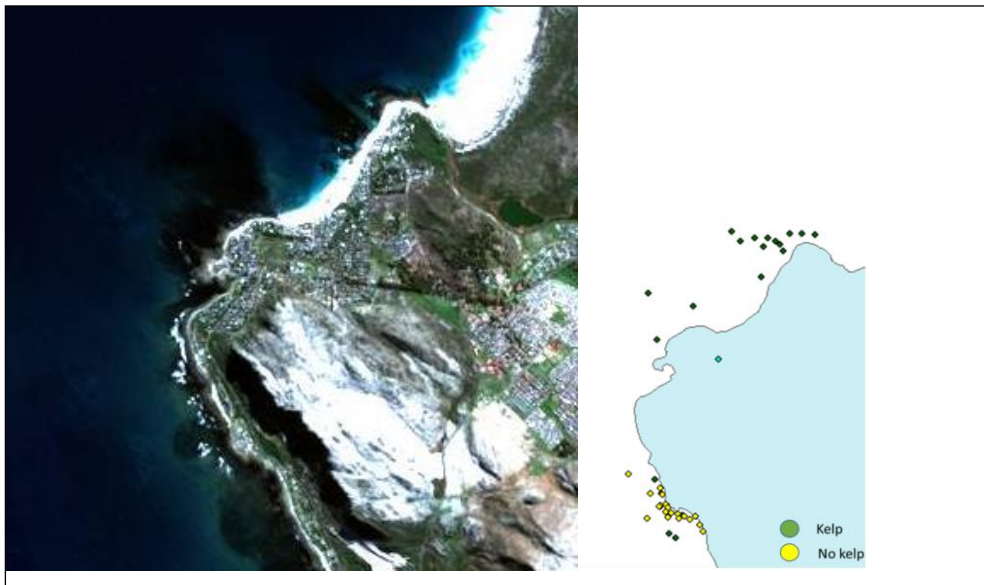
Hand-drawn kelp map by (Tarr 1993) in comparison to a band composite map (2017) from Sentinel-2.

Appendix 2.5



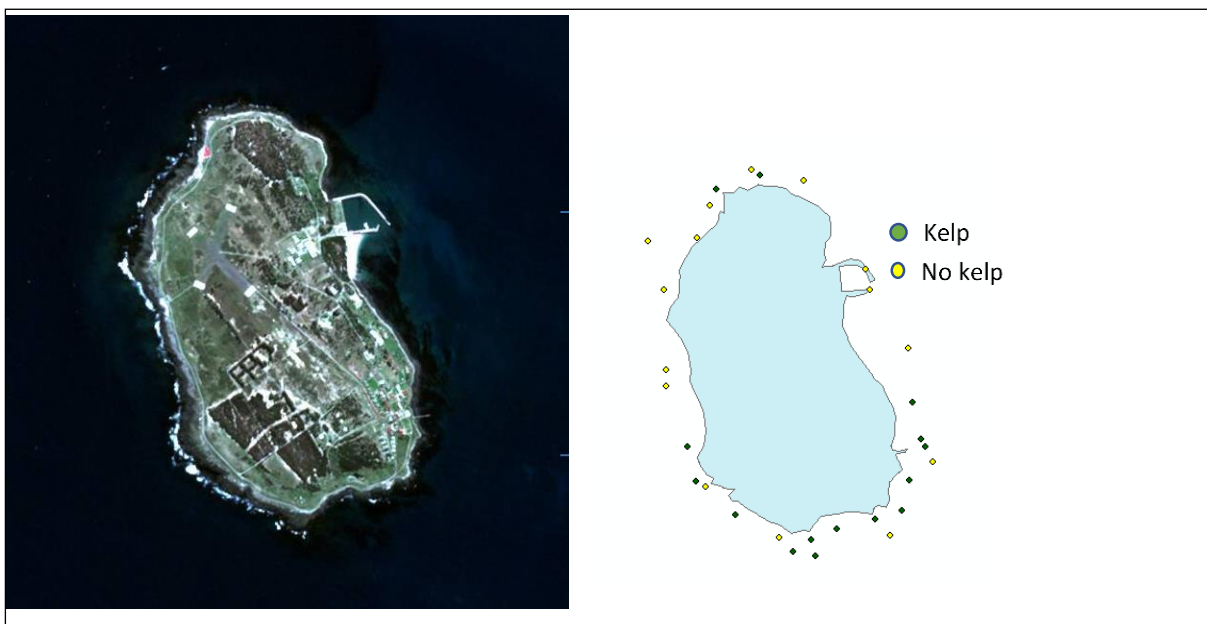
Google Earth image and Sentinel-2 map of Cape Hangklip and sample points for accuracy assessment.

Appendix 2.6



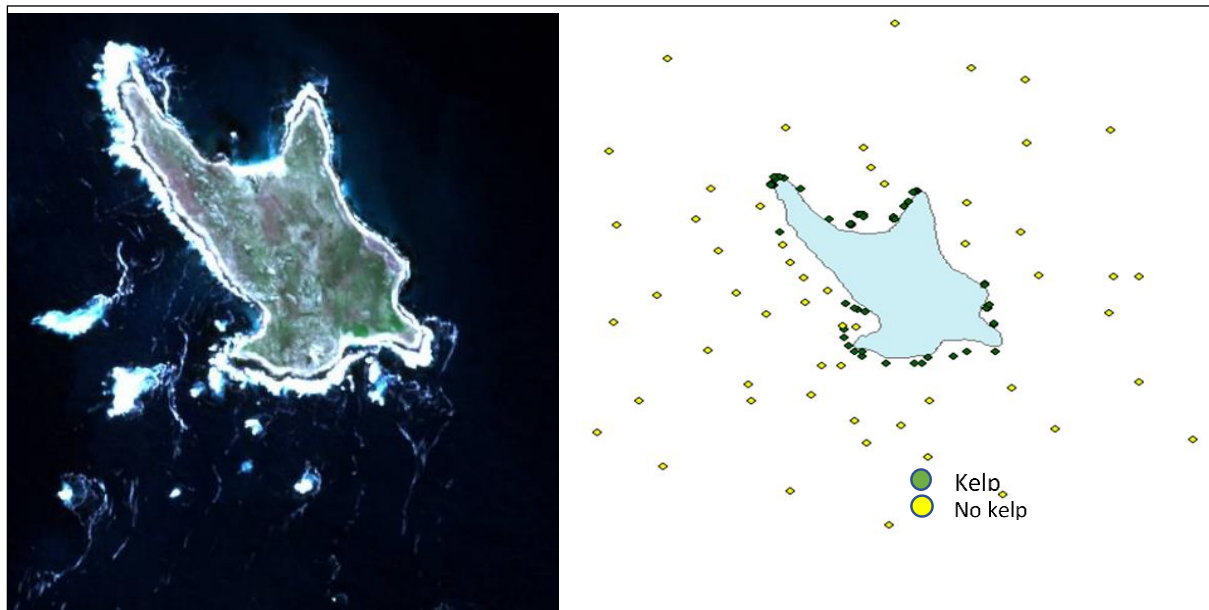
Google Earth image and map of Kommetjie with sample points used to generate accuracy assessment.

Appendix 2.7



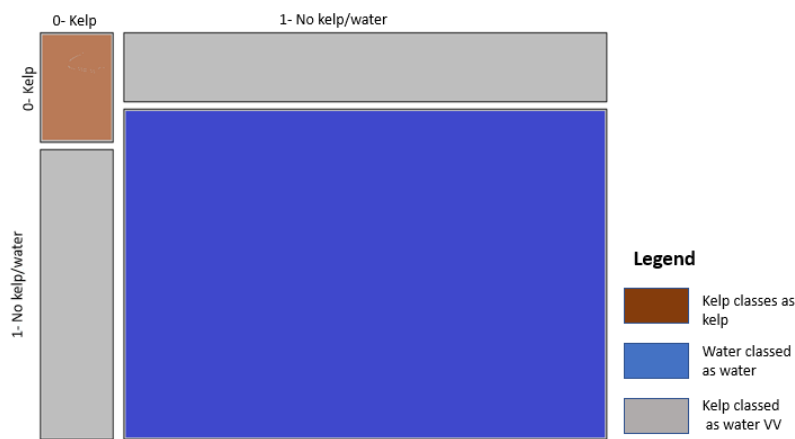
Google Earth image and map of Robben Island with sample points used to generate accuracy assessment.

Appendix 2.8



Google Earth image and map of Dassen Island with sample points used to generate accuracy assessment.

Appendix 2.9



Confusion matrix chart represents the results of a confusion matrix accuracy assessment conducted for two classes kelp (class 0) and no kelp (class 1) of classified (NDVI and supervised classification) and reference maps.

$$ARD = - (Area.t2 - Area.t1) / (year.t2 - year.t1) \quad \text{Equation 3.3}$$

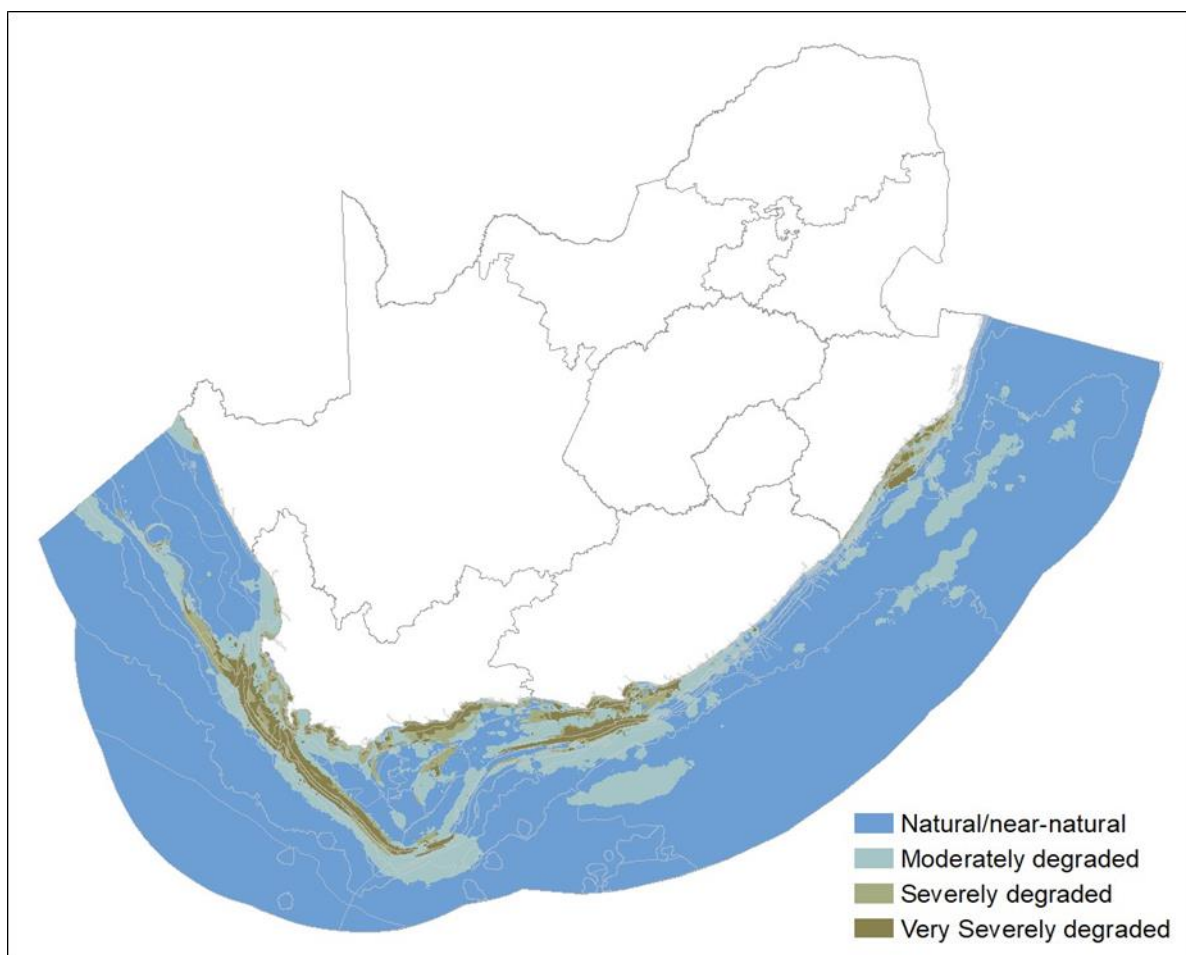
$$Area\ lost = (Area.t1 - Area.t_n) / Area.t1 \quad \text{Equation 3.4}$$

Sub-criteria and thresholds for assessing the rate of decline and therefore the risk to ecosystem collapse (Critically endangered= CR; Endangered= EN; Vulnerable= VU) (source: Bland et al 2016).

<i>Subcriterion</i>	<i>Time frame</i>	CR	EN	VU
A1	Past (over the past 50 years)	≥ 80%	≥ 50%	≥ 30%
A2a	Future (over the next 50 years)	≥ 80%	≥ 50%	≥ 30%
A2b	Any 50 year period (including the past, present and future)	≥ 80%	≥ 50%	≥ 30%
A3	Historical (since approximately 1750)	≥ 90%	≥ 70%	≥ 50%

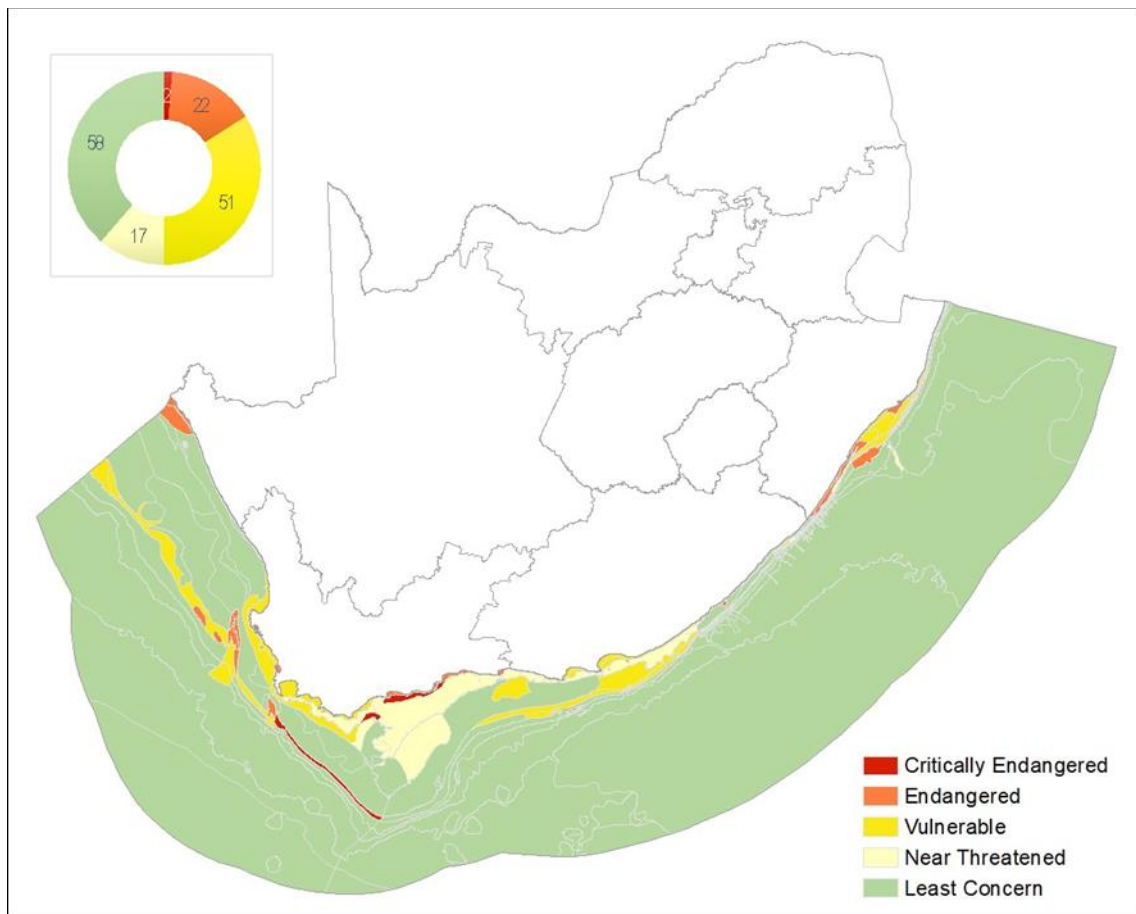
chapter 3

Appendix 3.1



Map of ecosystem degradation (Majiedt et al., 2019).

Appendix 3.2



Threat map of South African marine ecosystem types (Majiedt et al., 2019).

Appendix 3.3

Expert pressure review and impact weighting

Pressure	Ecological impact	Confidence in score	Recovery	Confidence in score	Impact + Recovery	Impact + Recovery /2	Combined Confidence Scores /2
Kelp harvesting: Lethal harvesting (Stipe and whole plant)	2	3	5	3	7	3.5	3
Kelp Harvesting: Non-lethal (Frondd cutting)	1	3	2	3	3	1.5	3
Diamond Mining	6	2	5	3	11	5.5	2.5
Nuclear power (heated effluent water)	5	1	5	1	10	5	1
Aquaculture: Land-based Abalone	1	2	1	1	2	1	1.5
Mariculture: ranching of abalone	3	1	5	1	8	4	1
Aquaculture: Land-based Fish	2	1	1	1	3	1.5	1
Aquaculture: Land-based Oysters	5	2	5	1	10	5	1.5
Coastal Development	1	2	1	2	2	1	2
Industrial Wastewater	8	2	5	2	13	6.5	2
Climate Change: average cooling	5	2	5	1	10	5	1.5

Climate change: extreme warming events	3	1	1	1	4	2	1
Climate change: extreme cooling events	3	1	1	1	4	2	1
WCRL: Hoop nets	5	2	7	2	12	6	2
WCRL: recreational	5	2	7	2	12	6	2
WCRL: poaching	5	2	7	2	12	6	2
Range expansion of WCRL	10	3	10	3	20	10	3
Line fish: commercial on the SC	8	1	8	2	16	8	1.5
Line fish: commercial on the WC	5	1	5	2	10	5	1.5
Line fish: recreational SC	8	1	8	2	16	8	1.5
Line fish: recreational WC	5	1	5	1	10	5	1
Black tides	10	3	8	2	18	9	2.5
Low oxygen events	2	3	8	2	10	5	2.5
Toxic algal blooms	2	2	8	2	10	5	2
Desalination	2	1	1	1	3	1.5	1
Climate change: increased storminess	2	1	2	2	4	2	1.5

Appendix 3.4

The individual intensity of each pressure is colour-coded based on a scheme developed for the NBA

Pressure	Score
Commercial Line fishing (East and South Coast)	8
Recreational Boat-Based Line fishing (East and South Coast)	8
Sea-Based Aquaculture	7
Mining	6
Port and Harbour Impacts	6
Recreational Shore-Based Line Fishing (East and South Coast)	5
Abalone	5
Wastewater/ Pollution	5
Commercial Line fishing (West Coast)	4.5
Recreational Boat-Based Line fishing (West Coast)	4.5
West Coast Rock Lobster	4.5
Recreational Shore-Based Line Fishing (West Coast)	3
Kelp Harvesting	20