Historical changes on rocky shores in the Western Cape, as revealed by repeat photography

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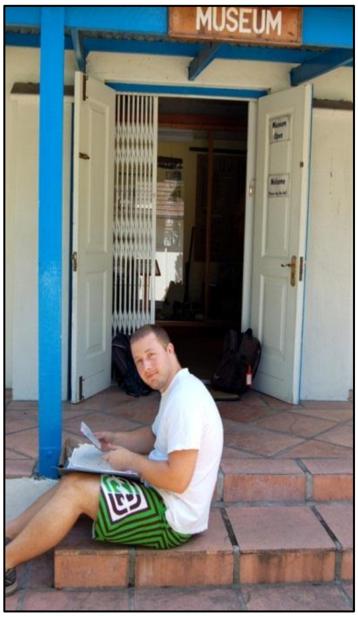

Dissertation submitted in partial fulfilment of the degree of Masters in Applied Marine Science, University of Cape Town

Date: 18 July 2012

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Acknowledgements

I would like to express my gratitude to the following people and institutions: Firstly to the Marine Research Institute of the University of Cape Town for the generous funding. Thanks are due to South African National Parks for their permission to visit Marcus Island. I am grateful to my friends and classmates, for their support and encouragement during the preparation of this dissertation. A dissertation is more than just an investigative writing exercise, it is a learning experience and often some of the lessons, little bits of knowledge, words of wisdom and experience, are not directly related to the project, but never-the-less add to one's life experience. It is for these lessons, as well as their time and enthusiasm that I would like to thank my supervisors, Professors Charles Griffiths and Timm Hoffman. Thanks are also due in large part to Phil Hockey and Otto Whitehead, who so generously agreed to share their photographs. To the museums and collections that were visited and gave their support to the project, namely the Simonstown Museum, Hout Bay Museum, the Old Harbour Museum in Hermanus and UCT's own Manuscripts and Archives Department, their support was invaluable. I would like to extend my gratitude to the Crook family for spoiling me when I could get away and providing me with a break as well as a home away from home. My family, for giving me the opportunity to always pursue my passions with the eventual goal of turning them into a (profitable) career, your words of encouragement, care and love were a constant source of strength to me while searching for the right words or the correct photographs. Lastly, but certainly not least, I would like to thank Catherine, for sticking with me, for putting up with me on my flights of fancy and various detours of self-discovery, you are my rock, your love and support have meant so much and will continue mean the world to me.

Chapter 1. Literature review

In order to effectively manage our natural environment it is necessary to understand its current condition and how this condition has changed over time. To accurately describe their condition it is often useful to examine the past, so that present conditions can be compared with those in the past. It is also important to realise the damage past oversights have had, with a view towards preventing similar occurrences from happening in the future. Ideally we need to know what the system's undisturbed baseline condition was. Trying to establish baselines is a challenging task, as it is not always possible to recall with accuracy the gradual changes over time. This has led to the term 'shifting baseline syndrome' being coined by Pauly (1995).

This illustrated review examines four main categories of change on the rocky shores of the Western Cape with the aid of repeat photography. The subjects of interest are: changes in range, climate change, intertidal invasions and direct anthropogenic effects.

1.1 Changes in range

Species can expand into new ranges naturally, i.e. without the aid of man, which is common enough after certain geological events, such as tectonic shifts. Climatic shifts may also trigger natural expansion of organisms into areas that they may have previously been unable to colonise (Grosberg and Cunningham 2001). Organisms capable of modifying habitat, which colonize new areas, can have substantive effects on community structure in the areas into which they expand (Ling 2008).

There are genetic differences between newly-colonised areas and the regions that contain the parent population. Newly-colonised areas are likely to have lower genetic diversity, as their population would contain a subset of the parent population's total alleles (Grosberg and Cunningham 2001). Range shifts have been seen in a variety of terrestrial species around the globe and have mostly been linked to changing climatic conditions (Chen *et al*. 2011). Similar observations have been made of intertidal organisms in England, leading to the suggestion that certain intertidal animals could serve as indicator species of climate change (Mieszkowska *et al*. 2006). Rouault *et al*. (2010) report a cooling trend of as much as 0.5˚C per decade in the southern Benguela, which has consequences for the organisms which inhabit the area. Long-term studies have reported that in 10 years it is possible for intertidal organisms to shift polewards by as much a 50 km (Helmuth *et al*. 2006).

Bolton and Anderson (1990) stated that the greatest contributing factor to seaweed dispersal and community composition was sea surface temperature. Interestingly Griffiths and Mead (2011) have observed increases in the range of certain cold water kelp species within False Bay, which seem to correlate with decreasing sea surface temperatures.

While the ranges of some species may expand because changing environmental conditions make conditions more favourable for them, ranges can also contract, for those organisms that are not as well adapted to the changing conditions and may lose their competitive edge. For example, cooler conditions, as well as increased competition with the invasive colder-water specialist *Mytilus galloprovincialis,* has resulted in the range contraction of the brown mussel, *Perna perna*, in the Western Cape (Griffiths and Mead 2010).

1.2 Climate change

Climate change is predicted to cause large-scale changes to ocean circulation patterns, changing biogeochemical cycles, which in turn may cause changes in species community compositions (Johnson *et al*. 2011). In Australia, a consequence of climate change was the increase in the speed of the East Australian Current, which had wide-ranging consequences. Of interest to this study is the reported loss of kelp bed habitat due to a decreased nutrient load within the current. Over-fishing of rock lobster also contributed to the reduced resilience of the habitat (Johnson *et al*. 2011). Recently climate change has become a topic of increased research and interest in biology (Simkanin *et al*. 2005). Global temperatures have risen consistently over the last century, with a 0.74˚C linear trend in those 100 years. In the last 50 years terrestrial temperatures have risen 0.13˚C per decade (Intergovernmental Panel on Climate Change [IPCC], 2007). Interestingly as mentioned earlier closer inshore along the South Coast there has been a decrease in SST, this opposed to the global average for SSTs for the period between 1985 and 2004 which showed an increase per decade of between 0.18˚C and 0.17˚C (Good *et al*. 2007).

A further consequence of climate change is sea level rise. The global average rise in sea level between 1961 and 2003 was 1.8 mm per year, with the ten year period between 1993 and 2003 averaging 3.1 mm per year; 57% of this is as a result of thermal expansion of the oceans. Ice caps and glaciers have contributed to 28% of recent sea level rise, with the remaining 15% made up from retreating polar ice sheets (IPPC 2007). Long term changes in mean sea level can also be brought about by the movement of tectonic plates (Muller *et al*. 2008).

It is predicted that climate change will result in, among other things, reduced productivity due to decreased nutrient supply, as mentioned above with regards to the East Australian Current (Barange *et al*. 2010; Wernberg *et al*. 2011). However, logic would dictate that it is also possible for the changing climatic conditions to benefit certain species, as has been seen with the range expansions of intertidal organisms in England (Mieszkowska *et al*. 2006).

The most important factor affecting the distribution of intertidal seaweed communities is water temperature in the surrounding sea (Bolton and Anderson 1990). In Australia it was thought that warming of the ocean waters, along with an increase of low level nutrient waters, has led to a range contraction of *Macrocystis pyrifera* (Wernberg *et al*. 2011). Kelps are not the only marine organisms to be affected by changing temperatures; responses to warming temperatures have been observed throughout the trophic levels, from plankton through to fish, ranging from changes in phenology, abundance and distributions (Mieszkowska *et al*. 2006). Intertidal invertebrates were resurveyed in order to determine if they had undergone any changes in range due to warming sea surface temperatures. Range extensions had occurred, but the authors cautioned that these extensions could not exclusively be linked to warming temperatures, as other influences, such as coastal development and other anthropogenic effects, could influence the signal, laboratory testing had to be carried out to determine exactly what the effects of the new temperature range would be (Mieszkowa *et al*. 2006).

1.3 Intertidal invasions

Ecological invasions have most likely been occurring in the oceans ever since man began navigating the seas (Robinson *et al*. 2005). There are two categories of introduction: deliberate and accidental introductions. Deliberate introductions involve the planned introduction of exotic species, often as ornamental or food organisms while accidental introductions are unplanned and most commonly associated with shipping. Most transfers of marine alien species have been accidental, but some species have been specifically imported for aquaculture (Griffiths *et al*. 1992). Accidental introductions occur in numerous ways i.e. via aquaculture practises with poor biological controls, as stowaways in ballast water from long distance shipping, the creation of new canals between previously geographically divided oceans and via attachment of organisms to ships hulls, known as fouling (Branch and Steffani 2004; Griffiths *et al*. 2009; Haupt *et al*. 2010). For a comprehensive discussion of the vectors of marine invasion into South Africa see Griffiths *et al*. (2009).

The first marine invasive species were introduced via shipping, these were most likely wood-boring shipworms. Their initial date of arrival is not known, but the first documented invasive species were likely to have arrived in South Africa in the late 1800s (Griffiths *et al*. 2009). Since then, there have been many other recorded invasions. In the most recent survey, 86 introduced and 39 cryptogenic marine species were recorded in South Africa (Mead *et al*. 2011). As mentioned in the previous section genetics can play an important role in elucidating the mechanisms involved in the colonization of new areas. It may be possible to use genetic studies to determine if a newly-identified colonizing organism resulted from anthropogenic influence, or via natural means i.e. a range expansion (Grosberg and Cunningham

2001). If the new colony contains unique alleles, the majority of which have been passed on from the founding population, it is strong evidence against anthropogenic influence (Grosberg and Cunningham 2001).

All deliberately introduced marine species in South Africa are from the phylum Mollusca. Examples include the oysters *Ostrea edulis, Crassostrea angulata* and *Crassostrea gigas* (Griffiths *et al*. 1992). They were initially introduced for aquaculture, under the assumption that local conditions would not support escaped populations, for instance, *C. gigas* does not spawn well in the Knysna Estuary and thus the farmed population is supported by imported spat (Robinson *et al*. 2005). Even though it suffers from poor recruitment, small self-sustaining populations have been found in a few estuarine sites, namely the Knysna, Goukou and Breede River Estuaries (Robinson *et al*. 2005). Other less-prominent intertidal invasions have occurred, such as the introduction of the periwinkle *Littorina saxatilis,* which was first recorded in 1974 in Langebaan Lagoon and Knysna Estuary (Robinson *et al*. 2005). For the most current list of marine invasive species in South Africa see Mead *et* al. (2011). The three most prolific marine invasives in South Africa are the European shore crab (*Carcinus maenas*), the Mediterranean mussel (*Mytilus galloprovincialis*) and the Pacific barnacle (*Balanus glandula*) (Griffiths *et al*. 1992; Robinson *et al*. 2005; Laird and Griffiths 2008; Simon-Blecher *et al*. 2008; Griffiths *et al*. 2009).

M. galloprovincialis covers some 2050 km of the South African coast. It was first noted in 1979 and only classified correctly in 1985, by which time it was well established (Griffiths *et al*. 1992; Robinson *et al*. 2005). It is believed that *M. galloprovincialis* arrived either in the ballast water, or on the hull of a ship in Saldanha Bay harbour on the west coast of South Africa. It moved rapidly along the coast and reached the south coast only 10 years later (Robinson *et al. 2005*). Its rapid growth and expansion were likely supported by the nutrient-rich upwelling region along the west coast, which supports large amounts of planktonic biomass. *M. galloprovincialis* competed against the local mussel *Choromytilus meridionalis* which is not as well adapted to dealing with emersion. *M. galloprovincialis* also has a greater rate of reproductive success and a greater growth rate than indigenous mussel species (Robinson *et al.* 2005; Joubert 2009).

One of the most wide-spread, though only relatively recently reported invasives in South African coastal waters is the North-east Pacific acorn barnacle *B. glandula*. It was first reported by Simon-Blecher *et al.* (2008), although Laird and Griffiths (2008) found photographic evidence that implies *B. glandula* arrived in South Africa either before or around 1992. *B. glandula* is thought to be adapted to cold and temperate waters, as evidenced by its invasion of colder water climates in Japan in the 1960's and Argentina in 1976 (Laird and Griffiths 2008; Simon-Blecher *et al*. 2008). The presence of barnacles on the west coast was limited prior to invasion of *B. glandula* and it appears that *B. glandula* has taken advantage of this relatively unoccupied area to establish itself. Since its introduction it has become the most common barnacle on the west coast of South Africa (Laird and Griffiths 2008). Its presence has influenced the zonation of the shoreline where it occurs by providing shelter for *Afrolittorina kysnaensis.* It is thought that *B. glandula* may also have excluded or outcompeted other intertidal organisms (Laird and Griffiths 2008; Simon-Blecher *et al*. 2008).

<u>Effects of invasions</u>

The effects of invasions can be measured in different ways; economically, i.e. money being spent in order to control or combat the organism or via ecological impacts, such as the displacement or fouling of other economically important species. Ecologically, the impact could be measured by the amount of disturbance the organism imparts on the biodiversity or community structure of the indigenous organisms at the affected site. Pimentel (2000) estimated that globally there are 120,000 invasive species that cost $314 billion per annum in control and damages.

It is unknown how many alien species reach foreign shores every year and fail to establish themselves, as they may not be able to compete against the local fauna and flora, or are poorly adapted to the physical demands the environment places upon them. The European shore crab *C. maenas* has been able to invade South African waters, but after three decades remains confined to sheltered bays (Hampton and Griffiths 2007; Mead *et al*. 2011). This is due to its inability to cope with South Africa's exposed shores. It does, however, pose a threat to other sheltered marine habitats and it is thought that an invasion by *C. maenas* into the sheltered waters of Saldanha Bay would be devastating for the indigenous intertidal organisms (Robinson *et al*. 2005).

In South Africa the most prolific marine invasive *M. galloprovincialis* has not proven to have had a severe negative economic impact. In fact an industry has grown up around the introduced species in the form of lucrative mussel farms (Lach *et al. 2002*). However, another invasive, *Ciona intestinalis* costs the mussel industry an estimated R100 000 each year due to the fouling of mussel ropes (Robinson *et al*. 2005). Estimates have yet to be made on how much it costs the shipping industry to remove *C. intestinalis* from ships hulls.

1.4 Direct anthropogenic effects

Anthropogenic impacts can be difficult to detect on rocky shore communities, due to various evolutionary adaptations that allow intertidal organisms to better cope in the stressful environment in which they find themselves (Crowe *et al*. 2000). The potential impacts are many, and include pollution, climate change, shipping, mineral extraction, habitat modification, disturbance and overexploitation (Griffiths *et* al. 2004; Brander *et al*. 2010).

Rocky shores have been exploited by man in what is now South Africa for some 120 000 to 100 000 years (Siegfried *et al*. 1994; Griffiths *et al*. 2004). Initially however, the impact of humans on the intertidal zone was limited by the lack of technology and small population size. Modern-day exploitation of marine resources began in the 1700s (Griffiths *et al*. 2004). There are 35 species exploited for subsistence purposes in South Africa, the main focus being on mussels and limpets, with other species such as octopus, redbait and winkles exploited to a lesser extent (Siegfried *et al.* 1994). It has only been relatively recently that commercial exploitation of intertidal resources was initiated. However, commercial exploitation has focussed on specific highly abundant species that are relatively easy to harvest. The amounts collected are subject to regulations which are aimed at providing sustainable yields; subsistence collections are much more varied in the range of species collected, with significantly lower yield per person and with much smaller areas of focus by each harvester, all of which compounds the difficulty in managing and defining fishers and enforcing regulations (Siegfried *et al*. 1994).

Human overpopulation threatens the coastal ecosystems and it is estimated that 40% of the world's population live within 100 km of the coast: in the year 2000 that was 2.3 billion people, and that number is estimated to rise to 3.1 billion by 2025 (Brander *et al*. 2010). The most recent population data for the Western Cape put the province's population at 5 287 863 (Statistic South Africa 2011). This number is almost 13% higher than the predicted population of the Western Cape forecast for the year 2026 by the Institute of Future Studies in 1996 (Haldenwang and Boshoff 1996).

1.6.1 Evidence of development disturbances

Marine biodiversity is under threat due largely to man's influence on complex environments, which has seen impacts to marine habitats across the globe (Pillay *et al*. 2010). Habitat destruction is chiefly caused by increased construction in coastal areas, overexploitation and falling levels of water quality (Pillay *et al*. 2010). The decline in *Nanozostera capensis*, renamed from *Zostera capensis* on the recommendation of Tomlinson and Posluzny (2001), was first documented by Angel *et al*. (2006). It was later investigate by Pillay *et al*. (2010) who found that between 1983 and 2009 there was a change in the distribution and density of the sea grass, *N. capensis*. Klein Oesterwal was particularly impacted resulting in a significant decrease in the species richness, specifically of those species that lived with sea grass. The loss of this habitat brought about other changes, such as an increase in the burrowing species in uncovered sands (Pillay *et al*. 2010). Two incidents leading to a loss of *N. capensis* cover in Klein Oestewal were recorded in 1976 and 2003 respectively. The first case, blasting and dredging operations in Saldanha Bay may have impacted the site. The reason for the second decline is less clear, although it is speculated that the collection of prawns for bait with the aid of pumps and associated disturbance caused by heavy foot traffic may have negatively impacted sea grass cover (Pillay *et al*. 2010). For a summary of human driven change in the South Africa marine environment see Mead *et al*. (in press).

1.5 Historical studies

There is an increasing trend among ecologists to utilize historical data as a reference tool to better understand current ecosystems and how they have changed over time, and to better improve their management (Costanza et al. 2012). This is largely seen in the management of land and water resources, where historical data can be used to assess the range and distribution of species when they were less impacted upon by man (Swetnam *et al.* 1999). Historical data can be gathered in any number of forms and from across seemingly unrelated fields. For example, sediment cores are utilized when determining trends in water quality, while weather records and satellite data all play a role in historical ecology (Swetnam *et al*. 1999). The only stipulation is that the data comprise a suitably long time period and are able to demonstrate key behaviours of the organisms or systems being studied (Swetnam *et al*. 1999).

A recent study that emphasized the importance of baseline studies globally was carried out by the Natural Geography of Shore Areas within the Census of Marine Life programme (Cruz-Motta *et al*. 2010). Its main aim was to establish baselines with regard to patterns of diversity and distribution to enable the detection of future changes and what may have caused these changes (Cruz-Motta *et al*. 2010). They found that there were six environmental drivers of intertidal species spatial distribution. An important observation was that assemblages were very closely correlated to aspects of both neutral (latitude) and environmental models (sea surface temperature (SST) and Chlorophyll-a). Cruz-Motta *et* al. (2010) noted that in this study, latitude and SST were not strongly correlated and it was put forth that SST has an important role in determining global distribution patterns of the assemblages studied. Therefore, future changes in SST that result from climate change may disrupt the functioning of these assemblages by changing their structure (Cruz-Motta *et al*. 2010).

1.6 Photographic sampling

Photography has always had a record-keeping purpose, whether to document famous people, or other news-worthy events, such as wars (Webb *et al*. 2010). Prior to photography becoming a practical tool, researchers in biology relied on drawings and precise descriptions to pass on what they observed during field work (George 1980). In truth, there are very few fields in science which have not made use of photography in some form (George 1980). Taxonomy has been a benefactor, since colour images taken in the field can be used to record distinctive characteristics of organisms that may later lose their pigmentation during preservation (George 1980). Of more relevance to this paper are the many uses of photography in marine ecology which George (1980) suggests are "…to aid site and habitat description, to record the relationship of plants and animals to the environment and to one another, to measure population numbers and the size of individual organisms, and to record the changes in community structure that take place with time." This list does not take into account the advantages of video to document behavioural studies. As the old adage explains, "a picture is worth a thousand words" and photographs taken at the shore are often far more valuable than lengthy written descriptions used to "set the scene" (George 1980).

As with all methods of sampling, however, there is a trade-off between the advantages and disadvantages involved. The disadvantages are similar to those shared with most disciplines interested in sea-shore studies. For example, time is a limiting constraint, as most photographs need to be taken at low tide and indeed the most effective photographic sampling is carried out during spring low tides, when the greatest area of intertidal habitat is exposed. Capturing images during these brief windows often complicates large-scale spatial studies, especially with limited resources (Murray *et al*. 2006). Furthermore, issues may arise with the detail that can occur due to the resolution of the photograph, as well as identifying taxa of a similar appearance, and problems with analysis may arise where the organisms are in high concentrations and/or stratified (Murray *et al*. 2006).

There are, however, many advantages to the correctly applied photographic method: the greatest perhaps being that the images may be stored and used for further analysis at a later date. These stored images can also be kept on record and used to analyse changes that may have occurred over time. Photographs also have another advantage, in that they are more easily interpreted by people without scientific backgrounds that may not easily understand graphs or complicated figures (Murray *et al*. 2006). Further examples of the application of photographic techniques in marine biology include photographic surveys and photographic tagging, which has for many years been used in identifying whales, dolphins or seals (McConkey 1999). Photography has also proved useful as a tool for conducting population counts, such as on the West Coast, where seal populations are photographed from the air to determine breeding success (Griffiths *et al*. 2004). Aerial photography has also been used in conjunction with infra-red photographic techniques to determine the extent of kelp populations along the South African coast (Bolton and Anderson 1990).

In South America, researchers used a specially encased camera to conduct short-term temporal studies on steep rocky intertidal slopes, in an attempt to increase the size and rate of sampling. They found this new method compared favourably against other methods of non-destructive sampling (Moyses *et al*. 2007). This photographic method also produced very distinct advantages over other non-destructive techniques, primarily that a greater area could be sampled faster. There were also economic benefits when compared to two other non-destructive estimation methods, namely point intersection and visual estimation, in that fewer sampling trips and a reduction in the number of people needed to carry out the survey resulted in reduced costs. Another marine repeat photography study compared fish caught off the Florida Keys over a period of 50 years, reporting changes in species composition as well as decreased sizes of the fish caught (McClenachan 2009).

1.2.1 Repeat photography

This photographic technique had its origins in the Alps, when in the late 1880s the geologist Sebastian Finsterwalder began using photography to document glaciers (Webb *et al*. 2007; Webb *et al*. 2010). However, use of repeat photography does not lie solely within the physical sciences, and has been equally useful for documenting ecological change over time (Bierman *et al.* 2005). It soon found further uses in landscape studies, specifically related to the changing populations of plants. It has also been used to document changes in vegetation (Webb *et al*., 2010). However, Webb *et al*. (2010) report that it was seldom used prior to the 1940s. After this period it enjoyed increased popularity as a technique for range management. Before the advent of photography biologists had to rely on drawings or paintings to illustrate their observations, but in 1951 Simpson used photography to match old landscape drawings made in the 1800's (George 1980; Webb *et al*. 2010). Two botanists, Shantz and Turner, used repeat photography to document large changes that had occurred in plant populations of Africa by repeating the images

Shantz had taken in the early part of the 19th century on a trip from Cape Town to Cairo (Webb *et al*. 2010). Towards the end of the 1950s MacDonald used repeat photography to measure changes in vegetation cover as a proxy for changes in climate (Webb *et al*. 2007). Some of the first studies using the technique of repeat photography were in the field of geology. Using "then-and-now" images of glaciers Gilbert documented the changes that occurred in these slow-moving masses of ice (Bierman *et al*. 2005). Hastings and Turner first conducted large-scale unrestricted repeat photography surveys, which later gave rise to the Desert Laboratory Repeat Photography Collection (Webb *et al*. 2010). Most of these repeat photography studies have taken place in North America, but studies have taken place in South America, one such example being the Alpine of the Americas Project, and in Africa studies have been conducted in Kenya and South Africa (Muriuki *et al*. 2003; Rohde and Hoffman 2008). Even Antarctica has recently been the subject of a repeat photography study (Webb *et al*. 2010).

Chapter 2. Repeat photography of rocky shores in the Western Cape

2.1 Abstract

This chapter uses repeat photography to illustrate changes that have occurred on rocky shores in the Western Cape over the past hundred years. Changes are documented under four categories; changes in range, climate change, intertidal invasion and direct anthropogenic effects. Images were sourced from books, members of the public and subject specialists and were selected based on their suitability. The sites of the images were identified and repeat photographs captured. The images depict the slow but progressive easterly spread of the kelp *Ecklonia maxima* and the range contraction of the warmer-water mussel *Perna perna.* Evidence suggests that a changing climate is the major driver of both of these changes. No change in zonation due to changing sea levels was observed. However, the range contraction of *P. perna* is complicated by the introduction of an alien mussel, *Mytilus galloprovincialis.* Repeat photography shows the changes that *M. galloprovincialis* has caused on the intertidal community in Saldanha Bay. Another invasive organism, *Balanus glandula,* is shown to have greatly altered the community structure of the Blouberg intertidal zone. Also of interest is the degree of building development that has taken place along the shore. Repeat photography proved a useful tool for documenting changes that are separated by large amounts of time, and is recommended as a useful technique for the surveillance and monitoring of rocky shores.

Keywords: *Balanus glandula*, *Ecklonia maxima*, historical study, *Mytilus galloprovincialis,* repeat photography, rocky intertidal zone, Western Cape.

2.2 Introduction

In order to effectively manage our natural environment it is necessary to understand its current condition and how this condition has changed over time. To accurately describe their condition it is often useful to examine the past, so that present conditions can be compared with those in the past. It is also important to realise the damage past oversights have had, with a view towards preventing similar occurrences from happening in the future. Ideally we need to know what the system's undisturbed baseline condition was. Trying to establish baselines is a challenging task, as it is not always possible to recall with accuracy the gradual changes over time. This has led to the term 'shifting baseline syndrome' being coined by Pauly (1995).

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These factors are all capable of bringing about changes in the state of the system and therefore altering ecological resilience (Gunderson 2000). Systems can be modified when previously neighbouring species experience a range expansion. Range expansions can occur in two ways; naturally, i.e. by the removal of certain geographical boundaries, possibly during times of geological stress due to plate tectonics, or changes in environmental conditions such as changes in climate (Grosberg and Cunningham 2001). The organism would then naturally over time colonise the newly available territory. Bolton and Anderson (1990) found that the single greatest environmental contributing factor to seaweed distribution, as well as community structure, was sea water temperature. Interestingly Griffiths and Mead (2011) show that sea surface temperatures (SSTs) along the South African south coast are decreasing due to changing weather patterns interacting with unique upwelling systems. Along with this mean cooling trend, changes in kelp distributions have been observed, the cold-water species *Ecklonia maxima* experiencing a range expansion. Conversely the warm-water *Perna perna*, the brown mussel, has experienced an eastward contraction in its range along the south-west coast of South Africa (Griffiths and Mead 2011; Bolton *et al*. 2012).

Little work has been done on the role of climate change on rocky shores within South Africa. However, internationally it has been proposed that the intertidal environment could be used as a proxy for climate change, by investigating changes in zonation, or shifts in traditional range of species (Mieszkowska *et al*. 2006).

In the marine environment species can be transported to new areas by man. This can be done unintentionally by transporting the organisms in ships' ballast water, or via attachment to ships' hulls (Robinson *et al*. 2005). Transportation can also be deliberate, as in the South African oyster industry, where spat or seed is imported in order to stock grow-out facilities. If the correct biological control protocols are not followed, this process can result in the import of more alien species (Grosberg and Cunningham 2001; Haupt *et al*. 2010). Whether the species are deliberately or accidentally introduced, alien species can have negative effects on the local biota (Ling 2008). The invasive mussel *Mytilus galloprovincialis* has outcompeted and to some degree displaced the indigenous mussels *Aulocomya ater* and *Choromytilus meridionalis* along South Africa's West Coast and is considered South Africa's most prolific intertidal invasive (Griffiths *et al*. 1992; Rius and McQuaid 2005; Robinson *et al*. 2007; Simon-Blecher *et al*. 2008). *Balanus glandula* is an invasive barnacle that occurs in large numbers and densities along much of South Africa's West Coast. It was only recently identified in 2007 (Simon-Blecher *et* al 2007; Laird and Griffiths 2008).

Perhaps the most dangerous invasive species has been modern man, through the use of increasingly advanced technology and sheer numbers, we have had an impact on nearly all of the Earth's biogeochemical cycles (Folke *et al*. 2004). Coastal development has very real effects on the neighbouring marine communities (Pillay *et al*. 2010). The institute for future studies published population statistics for the Western Cape in 1996, estimating that there were 3 659 871 people at that time residing in the Province, and forecasting that by the year 2026 that number would increase to 4 591 285 people (Haldenwang and Boshoff 1996). However, Statistics South Africa's (2011) most recent population survey already places the population of the Western Cape at 5 287 863, or nearly 700 000 more people than was anticipated for a period 12 years into the future. All of these people require housing, sanitation, water and food. People, particularly those in coastal settlements, place strain on the near-shore marine environment (Airoldi *et al*. 2008). This is demonstrated by Angel *et al*. (2006), first documented the decline of dwarf eelgrass *Nanozostera capensis* in the Langebaan lagoon and how it affected the critically endangered mollusc *Siphonaria compressa*. Following on from this research Pillay *et al*. (2010) demonstrated that the loss of *N. capensis* has been the catalyst for large-scale ecosystem change. The loss of *N. capensis,* along with habitat degradation, destruction caused by prawn pumping and trampling, has had a dramatic negative effect on the benthic invertebrate community.

Photography has, since its inception, been used for scientific research. It is an invaluable tool in conducting aerial surveys and has been used up the West Coast of South Africa and into Namibia to conduct population counts of seals (Griffiths *et al*. 2004). It has also proved to be useful in mapping and conducting abundance estimates of kelp (Bolton and Anderson 1990).

Repeat photography has for many years been used in range management in order to ascertain whether remedial actions are having the desired effect, for example reducing erosion or increasing an area's biodiversity (Bierman *et al*. 2005). One of the earliest marine investigations that used repeat photography quantified the decline in the size of trophy fish off the Florida Keys (McClenachan 2009). Moyses *et al*. (2007) also used a rephotographic technique to measure changes on steep rocky slopes, though their time period was far shorter. Repeat photography of plants has been used since the late 1950s as a proxy for climatological data (Webb, *et al*. 2007). In this chapter I attempt to use repeat photography of rocky shores to document changes in intertidal environments. By comparing these temporally separated images I illustrate, for the first time, the rate of change and impact that different natural and anthropogenic events have had on the biota of the coasts of the Western Cape.

2.3 Materials and methods

Potential photographs were sourced from several different locations. Photographs of rocky shores were requested from subject specialists, as well as the general public by means of advertisements in either specialist monthly newsletters or by placing appeals in general public newspapers. The photographs were collected and assessed on the basis of their appropriateness to the project. Parameters used in the selection of each photograph differed, but included the age of the photograph, location, distinguishable features, access to the site and the possibility of reproducing the original image. The final set of images was selected so as to reflect a spread of the conditions that characterize the western and southern coast lines of the Western Cape. The images selected were in a variety of formats including black and white and colour prints, 35 mm black and white negatives, 35 mm colour slides, and digital images. For ease of use and future editing the images were digitized by scanning the historical images on a Canon Canoscan 5600F as high resolution TIFFs with the following specifications: 300 dpi, 450 mm output size, 16 bit/channel. Once the images had been scanned and archived the original material was returned to the owners.

Prior to fieldwork, images selected to be rephotographed were printed out at A4 and placed in a file so that they remained relatively protected and were easily accessed during field trips. Repeat photography expeditions were carried out during spring low tide periods. During rephotography land-marks were identified from the past images and care was taken to align the photographs as closely as possible for accuracy of comparison. Due to difficult terrain in most locations, tripods were not utilized for this project. When the images were aligned a repeat photograph was taken at the maximum resolution possible with one of several Digital Single Lens Reflex (DSLR) cameras.

ADOBE® Photoshop® was used to adjust the images to similar dimensions (height was changed to 16 cm if in portrait, or width is changed to 16 cm if the images are landscape) and pixel density was set as 300 pixels/inch. The distance between two identical objects found in both images was measured. To yield two distances A (the largest) and B (the smaller distance). Distance A was divided by distance B and the resulting figure was multiplied by 16, to give a value of Z. The height or width of the image which contained distance B was then altered to the calculated value of Z. With images of the same size, alignment could be carried out by dragging the rephotographed image on top of the original photograph and by varying the opacity of the rephotographed image (layer) it was possible to achieve the greatest degree of overlap. Once aligned the rephotographed layer was saved. The rephotographed layer was then removed and the original image saved separately, making both images available for later examination. Percentage cover was estimated from these photographs, taking into account that not all areas within the photograph could be inhabited or built upon by the object under examination.

The study sites were located between Marcus Island in the west and De Hoop Nature Reserve in the east (33.0427 S 17.9704 E – 34.4780 S 20.5139 E). The predominant focus of the photographs was on the rocky intertidal zone and in one section, the near shore and the terrestrial environment directly bordering the coastline. The study sites span two marine regions as depicted in Mead *et al*. (2011), those being the Cool-temperate region along the West Coast, and the Warm-temperate region of the South Coast.

<u>Table 1, Site names and locations</u>

Figure	Site name	GPS co-ordinates
2.1	Oatland Point	34.2082 S 18.4611 E
2.2	Wooley's Pool	34.1326 S 18.4465 E
2.3	Dalebrook	34.1244 S 18.4527 E
2.4	St James	34.1190 S 18.4596 E
2.5	Fick's Pool	34.4222 S 19.2354 E
2.6	Old Harbour, Hermanus	34.4203 S 19.2437 E
2.7	Old Harbour, Hermanus	34.4203 S 19.2437 E
2.8	De Hoop Nature Reserve	34.4780 S 20.5139 E
4.1	Oatland Point	34.2082 S 18.4611 E
4.2	Oudekraal	33.9837 S 18.3576 E
5.1	Marcus Island	33.0427 S 17.9704 E
5.2	Marcus Island	33.0427 S 17.9704 E
5.3	Blouberg	33.8063 S 18.4648 E
6.1	Bantry Bay	33.9280 S 18.3756 E
6.2	Clifton	33.9331 S 18.3775 E
6.3	Llundudno	34.0048 S 18.3398 E
6.4	Kalk Bay	34.1290 S 18.4482 E
6.5	Muizenberg Corner	34.1090 S 18.4689 E

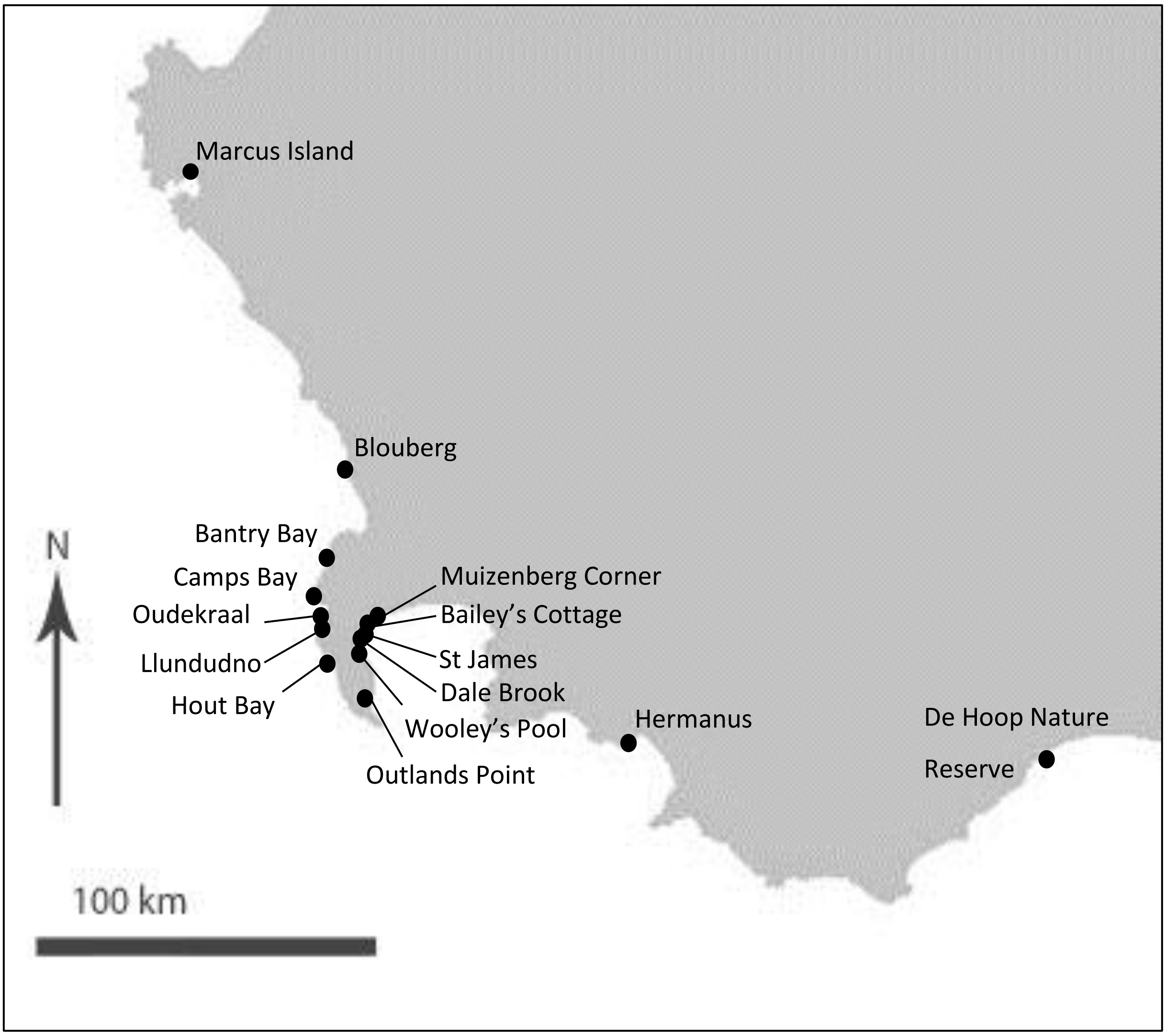

Figure 1, Map of the Western Cape showing sites

2.4 Results and discussion

Changes seen in the images can be categorised as falling under four headings 1) Changes in range, 2) Climate change, 3) Intertidal invasion and 4) Direct anthropogenic effects. Each topic is treated separately below.

2.4.1 Changes in range

The following images depict changes in range and density of cold water kelp species. The images are arranged from the western-most site of observable change, moving eastwards. Figures follow a left to right temporal order, the original images are located to the left with the more recent rephotographed image to the right.

Photo plate 1

Figure 2.1, Oatlands Point, Top: Morgans 1950; Bottom: Griffiths 2011

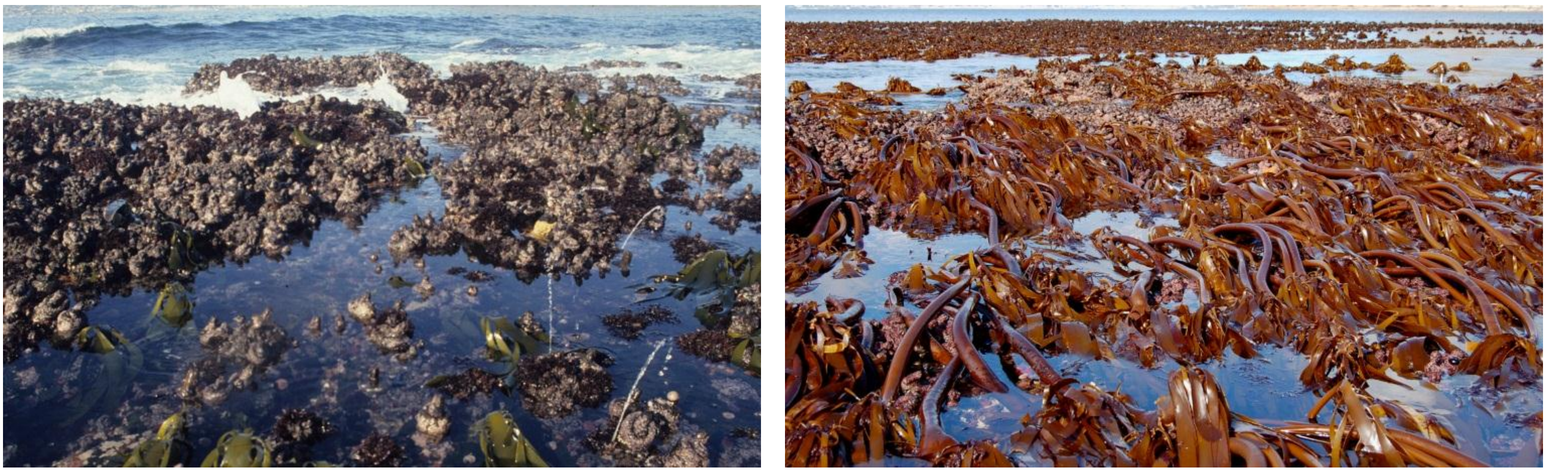

Figure 2.2, Wooley's Pool, Left: Griffiths 1992; Right: Griffiths 2011

Figure 2.3, Dalebrook, Left: Griffiths 1980; Right: Griffiths 2011

Figure 2.4, St James Top, left: Burman1950; Top right: Griffiths 1991; Bottom left: Griffiths 2011 (Circle indicates evidence of kelp heads)

Figure 2.5, Fick's Pool Hermanus, Left: Hermanus Old Harbour Museum, Approximately 1920; Right: Reimers 2012

Figure 2.6, Old Harbour, Hermanus, Left: Hermanus Old Harbour Museum, approximately 1920; Right: Reimers 2012

Figure 2.7, Hermanus Old Harbour, Left, Hermanus Old Harbour Museum, approximately 1920;

Right: Reimers 2012

Figure 2.8, De Hoop Nature Reserve Top: Griffiths 1991; Bottom: Reimers, 2012

Three general observations can be made from the previous figures, they are 1) that kelp has increased its density, 2) extended its range in recent years and 3) there are observable changes in the community structure of the kelp species.

Figure 2.1 shows kelp was present in 1950 at Oatlands Point. However, it was found exclusively in the intertidal zone, visible along the edges of large boulders, it has since become established in the subtidal zone as well. The kelp has increased from 0% cover to approximately 40% cover in the subtidal zone. These changes were recorded 60 years apart.

In Figure 2.2 the densification of kelp has increased from 5% to 80% at present. Present in the historic photograph are juvenile *E. maxima* in the intertidal zone. Kelp previously was not present in the subtidal zone; it now has a coverage of 70%. The *Pyura stolonifera* or redbait appears in both images and has maintained a high density, even though it is now heavily crowded by *E. maxima*. These images are separated by 19 years.

Species other than kelp are experiencing range shifts. In Figure 2.3 it can be seen that the mussel *P. perna* was once common in False Bay (Bright 1937). It has experienced a range contraction and is now only found in small isolated adult populations in False Bay. The 1980 image is dominated by *P. perna* with small clusters of *Mytilus galloprovincialis* around their bases. While the most recent image show effect of the cooling climate on *P. perna,* it is now only found in two locations in very small numbers at Dalebrook and Bailey's Cottage as small isolated colonies surrounded by *M. galloprovincialis*.

Figure 2.4 is a time series of three photographs of St James, spanning 60 years. This site is located 1.9 km away from Wooley's Pool (Figure 2.2). If the kelp bloom was a cyclical event it is apparent that it does not take place on a decadal cycle. Instead it the density gradient and time line of the matched photographs of the kelp seems to support that there has been be a slow shift around the Bay, originating from somewhere near Cape Point and having moved through the area of Simonstown, Fish Hoek and recently having established itself at Wooley's Pool. It would appear that it has now reached St James; the kelp being highlighted in the circle in the most recent image. Bailey's Cottage and Muizenberg remain free of kelp at the time of writing, however, given time *E. maxima* may continue to expand around False Bay in that direction.

Kelp has also spread further along the South Coast, as seen in Figure 2.5, it is a relatively recent arrival to Hermanus. In the original image from the 1920s no kelp is visible, however in the repeat image there is a 20% increase in the cover of *E. maxima*. The high density of the kelp would suggest that the kelp is not a new arrival in Hermanus. Further evidence of the range expansion of the kelp in this area is seen in Figure 2.6, which shows an increase in the coverage of kelp from none in the original photograph to 70% coverage in the near-shore environment.

Further along the South Coast in the De Hoop Nature Reserve, 116 km away (in a straight line) from the Old Harbour in Hermanus, the same kelp species *E. maxima* is also present. The images in Figure 2.8 are taken over a shorter period of time (21 y). The kelp has gone from 0% cover to 20% cover at a low density. These factors would indicate that kelp is a new arrival to this area of coastline, as has recently been documented by Bolton *et al.* (2012).

Many abiotic factors may affect the growth of kelp; these include nutrient concentrations, wave exposure, substrate and exposure to air and turbidity. These factors play a greater role in smaller scale distribution patterns (Bolton and Anderson

1990). Over larger spatial scales it has also been suggested that biological interactions may affect the composition of seaweed communities, however according to Bolton and Anderson (1990), the largest contributing factor to seaweed community composition is sea surface temperature. It is therefore most likely that this local upwelling system, as modelled by Rouault *et al*. (2010) and further described by Blamey *et al*. (2012), play a large role in the above documented changes in densification and range of *E. maxima*. The cooling trend closer to shore, most likely due to increased upwelling as well as the warming of the Agulhas further offshore due to changes in the transport of the system, are large scale shifts in oceanic climate capable of influencing biotic communities, often in conjunction with other drivers (Rouault *et al*. 2009; Rouault *et al*. 2010; Blamey *et al*. in press). The inshore cooling may also have contributed to the range contraction of the traditionally warmer-water mussel species *P. perna.* As can be seen from Figure 3 the cooling trend extends along the South Coast up until Port Elizabeth and up the West Coast of South Africa to approximately 32˚S (Rouault *et al*. 2010). This roughly coincides with what is observed in the Figures 2.1–2.7.

Other contributing factors to the increase in kelp densification could be, the recent range shift of the Cape Rock Lobster, *Jasus lalandii*, whose arrival in new areas along the South Coast has resulted in a decrease in herbivore biomass, particularly sea urchins, which graze on kelp holdfasts. The presence or lack thereof, of grazers can affect the abundance of kelp. It has been found that in general grazers decrease the abundance of algae present in their communities depending on if they are general or specific grazers. Intermediate grazing behaviour can stimulate the growth of macroalgae while high levels of grazing may decrease abundance (Zeeman *et al*. 2012).

The loss of these grazers due to predation by *J. lalandii* has contributed to a subsequent increase in overall macroagal biomass. However, it was seen that not all kelp species increased in biomass. *Laminaria* decreased in biomass, while *E. maxima* experienced an increase in biomass (Blamey *et al*. 2010). The cooling trend brought about the by the increase in upwelling may have only been a secondary factor to the increase in nutrients which would have resulted, allowing *E. maxima* to colonize areas further east along the South Coast than it was previously able to, the opposite to what has occurred in Australia (Johnson *et al*. 2011).

The presence or lack thereof, of grazers could also play a role in the abundance of kelp. It has been found that in general grazers decrease the abundance of algae present in their communities depending on if they are general or specific grazers. Intermediate grazing behaviour can stimulate the growth of macroalgae while high levels of grazing may decrease abundance (Zeeman *et al*. 2012).

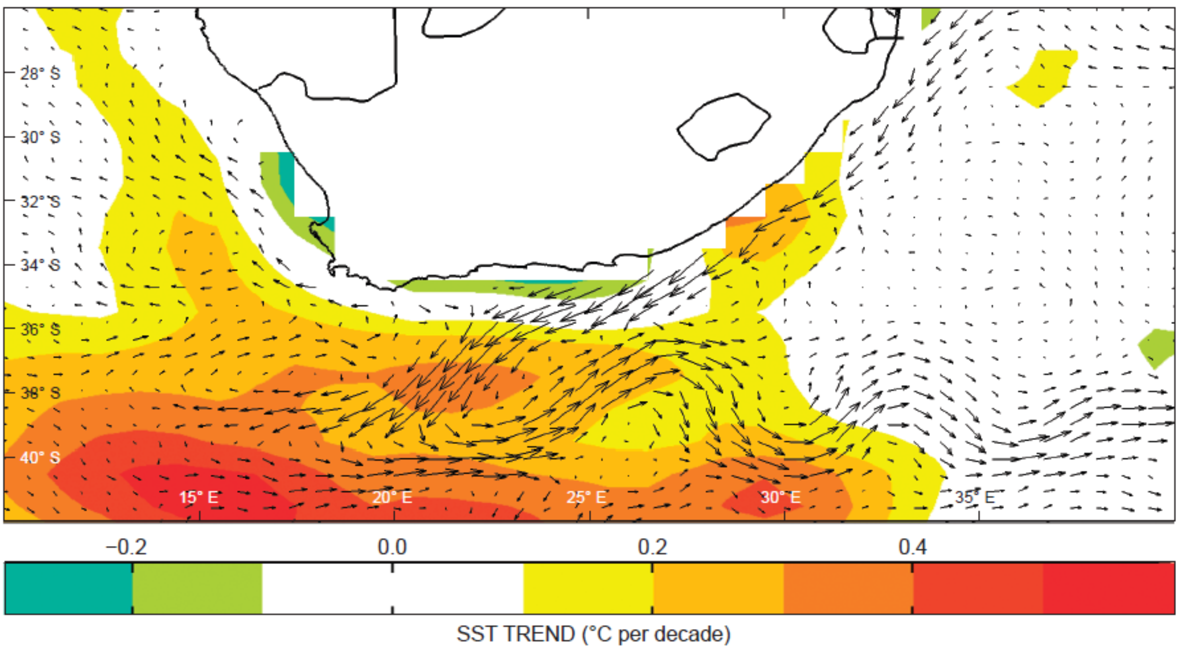

Figure 3, Average SST per decade between 1985 and 2009, after Rouault *et al*. (2010)

2.4.2 Climate Change

Photo plate 2 is comprised of images attempting to demonstrate the biological reaction to global climate change. It is linked closely to the previous section as the range shifts witnessed there are most likely a result of the cooling climate. In this section we take a closer look at another aspect of climate change. We attempt to determine the effect of sea level rise on the zonation of the intertidal zone.

Photo plate 2

Figure 4.1, Oatlands Point, Top: Morgans 1950; Bottom: Griffiths 2011

*Scale is given by the foot marker in the top image

Figure 4.2, Oudekraal, Top: Bright 1937; Bottom: Griffiths 2011

As illustrated above, the expansion in the range of the cold-water kelp species, *E. maxima*, as well as the range contraction of the brown mussel *P. perna* are most likely associated with the cooling trend off the South Coast (Rouault *et al*. 2010; Blamey *et al*. 2012). This, however, may be a change in regional climate and therefore not associated with global climate change. It is difficult to conclusively put any ecosystem changes down to changes in climate, when there may be so many other potential contributing factors (Mieszkowa *et al*. 2006). Other drivers can include climatic conditions, presence of habitat modifying organisms as well as anthropogenic effects (Grosberg and Cunningham 2001; Griffiths *et al*. 2004; Ling 2008)

Changes in vertical zonation due to sea level rise are able to be estimated using this technique. An argument for changes in zonation as a proxy measurement of sea level rise could be made, as intertidal organisms would need to re-establish themselves in the optimum competitive position. Mather *et al* (2009) reported figures of 1.48 mm y^{-1} increase in sea levels along the southern coast of South Africa between the period of 1957 and 2006. If this rate had been constant over the 74 year period between the photographs the visible change in zonation should have reflected a height difference of nearly 11 cm.

Figure 4.1, images are separated by 60 years. There have been changes in distribution and community structure of the seaweeds present. The intertidal zone has seen an increase in foliose seaweed cover from 60% to nearly 100%. However seaweed appears to have lost 20% of its upper zonation on the exposed rock face and instead clumps of *M. galloprovincialis* now exist in this area, demonstrating its ability to modify community structure. Other species of seaweed have increased in the subtidal zone, from three plants in the subtidal zone to ten kelp heads representing a 300% increase. It is difficult to compare the images in Figure 4.1 as the match is not exact. However, it does not appear that there has been an upward shift of intertidal organisms due to proposed sea level rise.

The historical image in Figure 4.2 shows a particular rock at the Oudekraal site, this image was published in 1937, decades prior to the invasion of *M. galloprovincialis*. Small Mediterranean mussels now cover 40% of the intertidal rock surface. The kelp density has also increased by 50% to 70% in the time between photographs. In Figure 4.2, if the distance between the upper limit of coralline algae (marked here by the black line) and the lowest point of the crack in the rock (indicated by the vertical arrows) are measured, no change in vertical zonation has taken place.

Observations may be complicated in this instance by the arrival of invasive intertidal organisms. The prime culprit is *M. galloprovincialis*. Due to its ability to survive greater periods of emersion, combined with other competitive advantages, it has altered community structure in the subtidal zone (Robinson *et al.* 2007). However the greatest driver of zonation change in this instance was expected to be the rising sea level.

2.4.3 Intertidal invasions

The following images all deal with the topic of intertidal invasions, the images focus on two invasives, the Mediterranean mussel, *M. galloprovincialis* and the barnacle *B. glandula*. *M. galloprovincialis* is the most visually abundant intertidal invasive organism along the South African coast. It arrived in Saldanha Bay around 1979. *B. glandula* was only documented in 2007, though there is evidence to suggest it had been present for at least a decade before that (Laird and Griffiths 2008; Simon-Blecher *et al*. 2008).

This section is divided into two parts:

Photo plate 3a contains images of *M. galloprovincialis* followed by analysis and interpretation. Photo plate 3b contains the image showing *B. glandula*.

Photo plate 3a

Figure 5.1, Marcus Island, Left: Griffiths 1989; Right: Griffiths 2012

Figure 5.2, Marcus Island, Above: Griffiths 1981; Below: Griffiths 2012

Figure 2.3 is also relevant to this section, *M. galloprovincialis* has altered the community structure of the West and Southern coasts of South Africa. It has superimposed itself on the intertidal community. There are reports of partial displacement of the indigenous mussel species along the West Coast (Robinson *et al*. 2007). It has also impacted on the zonation of the shore, due to its ability to tolerate longer emersion times and to take advantage of a higher shore line position (Robinson *et al*. 2007). For a full investigation into the consequences of the *M. galloprovincialis* invasion in South Africa, see Robinson *et al*. (2007).

Figure 5.1 shows an increase in the densification of *M. galloprovincialis* on Marcus Island between 1989 and 2012, from a patchy 30% over the slope to its current 80% sheet-like cover. Marcus Island is located in Saldanha Bay where *M. galloprovincialis* was first reported around 1979. The initial images from Marcus Island show the extent of the invasion after a single decade (Branch and Steffani 2004). It can be seen that the mussels have already established themselves by that time and it appears that they are being confined by the limpets which may have slowed their spread over the rock face somewhat.

In the repeat image of Figure 5.2 the densification of mussels on Marcus Island is very apparent. From a coverage, in the initial image, of 20% mostly confined to the trench and with isolated clusters on the flatter left rock surface and a small patches to the right. It is also noticeable that the *M. galloprovincialis* are largely surrounded by limpets. In the most recent image the coverage is now estimated at 70% of the exposed rock surface. The limpet population also appears to have declined. The spread of *M. galloprovincialis* has had other consequences. From the repeat image algae, *Ulva* and other seaweeds are seen to have taken advantage of the increased surface area and perhaps the grip that the mussel shells provide to increase their coverage on the slope. The loss of limpets has also contributed to the increase in the *Ulva sp* present on the rock surface. Other observations include an increase in the kelp present in the subtidal zone.

The second invasive species examined by repeat photography is the barnacle *B. glandula,* native to the North American Pacific. It is not known when it arrived, although there is photographic evidence of it as far back as 1992 (Laird and Griffiths 2008).

Photo plate 3b

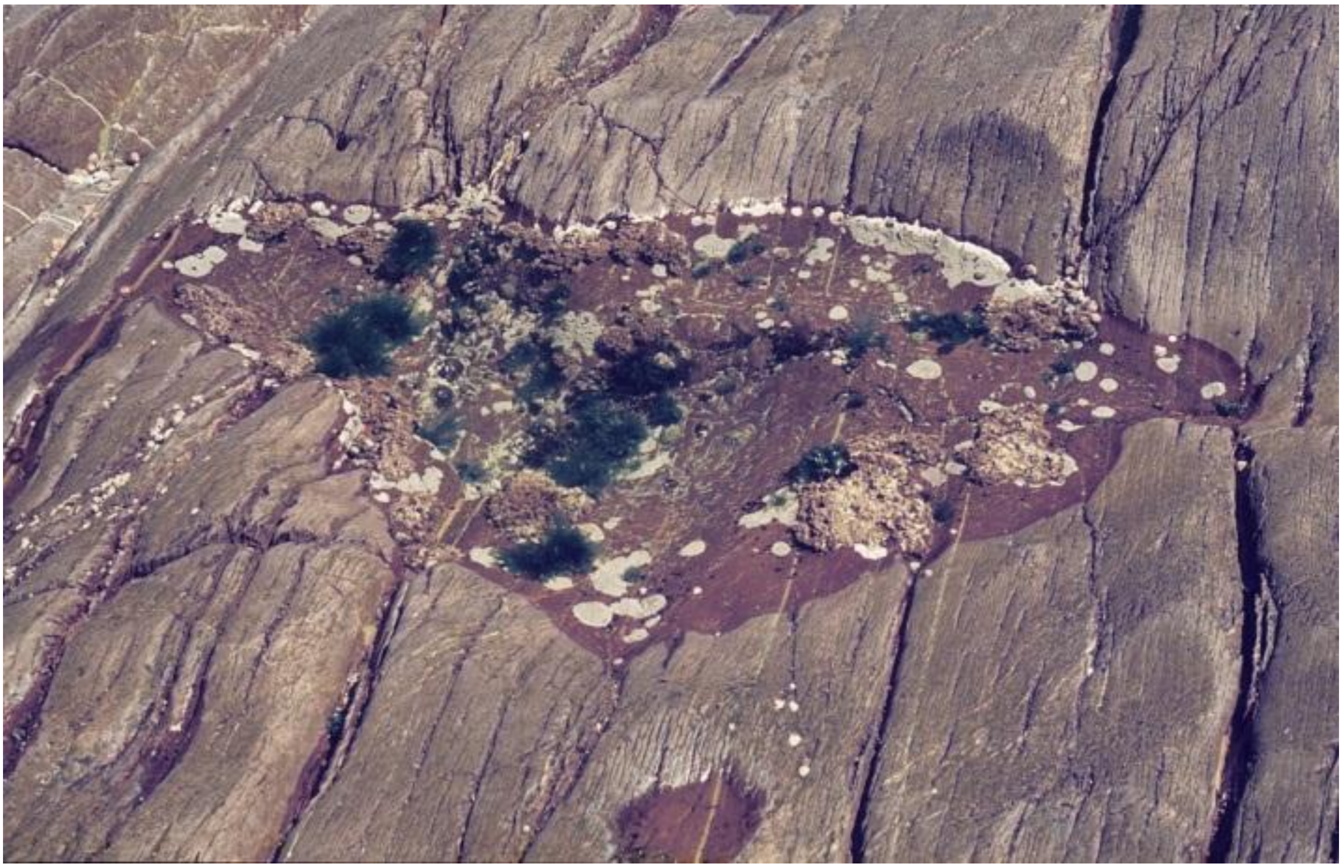

Figure 5.3, Blouberg, Left: Griffiths 1981; Right: Griffiths 2011

Figure 5.3, these images were taken at Blauberg, just north of Cape Town and show two invasive species to be abundant. First is *M. galloprovincialis* which has spread south with the prevailing ocean currents, and the second species is *B. glandula*, the North-east Pacific acorn barnacle. *B. glandula* had not yet arrived in South Africa in 1981. Three decades later it, along with another invasive, *M. galloprovincialis*, dominate the intertidal zone at Blauberg. *B. glandula* covers 50% of the exposed rock surface with *M. galloprovincialis* covering a further 10%. *B. glandula* is found at its highest densities at Moullie Point and Blauberg (Laird and Griffiths 2008). In the most recent survey *B. glandula* has been found along 400 km of South Africa's West Coast, from Elands Bay through to Misty Cliffs. For greater detail on the abundance and range of *B. glandula* see Laird and Griffiths (2008).

2.4.4 Direct anthropogenic effects

Direct anthropogenic effects refer here primarily to coastal development. These next images aim to show how rapidly coastal development has taken place in Cape Town and its surrounding coastal towns. Europeans started a permanent colony in Cape Town in 1652, prior to which man had been present in the Cape since the early Stone Age. Prehistoric evidence from shellfish middens indicates that exploitation of coastal resources was intensive and widespread, though lower population numbers and rudimentary technology likely tempered their impact on the intertidal zone (Siegfried *et al*. 1994; Griffiths *et al*. 2004). The following images capture the changes that have occurred in the development of the coast over the last century.

Photo plate 4


Figure 6.1, Bantry Bay, Left UCT archives 1890; Right: Griffiths 2011


Figure 6.2, Clifton, Left: UCT archives 1899; Right: Griffiths 2011

Figure 6.3, Llundudno, Left: date unknown ; Right: Griffiths 2012

Figure 6.4, Kalk Bay, Left: Burman 1900; Right: Griffiths 2011

Figure 6.5, Muizenberg, Top: Walker, date unknown; Bottom: Griffiths 2012

Bantry Bay in 1890 (Figure 6.1) was a sparsely-populated headland that has undergone intensive development, so much so, that from a cover of 30% comprised of low story, low density houses, all available area has been transformed into high rise, high density developments covering 90% of the non-vertical ground. This has changed the structure of the shoreline, with the erection of foundations and the deposition of soil and rocks in order to build roads. Retaining walls have been built and the natural slope has been engineered and modified. The gradual transition from the ecotone, which is the rocky shore, to the terrestrial hill side has been lost, potentially influencing the flow of nutrients between the two areas.

Clifton Bay, Figure 6.2, this stretch of coastline has been largely developed for residential and commercial purposes. The original image shows no development save for what appears to be a dirt road along the base of the mountain side. Intense development has taken place over 112 years, 40% of the visible land has been transformed with the major restricting factor being that the upper slopes belong to the Table Mountain National Park. Development along the coast can have many impacts on the near shore environment; the construction of walls where dunes were once present can result in the shortening of beaches. Reclamation of land can result in the destruction of local communities (Chapman and Underwood 2006). Also associated with increased development is the threat of increased pollution, which may affect community structure of the near-shore environment (Connell *et al*. 1999).

Figure 6.3, Llundudno, an exclusively residential suburb was founded in 1903. The development here has been less intensive upon the shoreline as the views and beaches were of prime attraction. In the first image no development is visible; however in 2012 buildings occupy 30% of the available land in the photograph.

Development can impact the run-off regime. Increasing the rate at which water would flow between the terrestrial and the coastal environment. Storm-water drains and outflows can result in point-source pollution with the adsorption and retention of contaminants from land and transport into the coastal waters (Inland and Coastal water Quality Committee Annual Report 2011).

Figure 6.4 was taken of Kalk Bay. The initial image was taken in 1900 and the repeat image in 2011. This area has been the site of human development for many years. The increase in the human population brings with it the threat of increased pollution. Kalk Bay harbour was constructed in 1918 and functions today as a port for small fishing craft. Altering the shape of the natural bay can change circulation patterns and increased boat traffic brings with it the threat of increased pollution.

Muizenberg, Figure 6.5, has undergone large scale development, similar to Camps Bay. Development below Boyes Drive has prevented an exact repeat image being taken, making estimation of the developmental changes difficult to gauge. The land behind the changing rooms in the original photograph was and still is today a wetland area. The amount of development in this area has certainly altered the water flow regime between the terrestrial and coastal environment, necessitating the channelling of water away from the foundations of the buildings now present.

The Inland and Coastal Water Quality Committee Annual Report (2011) states that 95% of bathing areas along the False Bay coast complied with the required water health standards for indicator organisms while the Atlantic coastline had a compliance of 75% of bathing areas meeting the indicator organism standards. The report further mentions storm water outlets are sources potential point sources of pollution.

Chapter 3. Conclusion

Several distinct categories of changes to the rocky intertidal zone have been observed in these repeat photographs. These include: changes in range, changes in the community structure as well as changes to the near shore terrestrial environment due to increased development. Changes in zonation were also investigated.

Increases in both the densification and extent of the range of the cold water kelp species are made visually apparent here. *E. maxima* has increased its penetration into False Bay by as much as 14 km around the Bay (10 km in a direct line), along with its range expansion it has increased in coverage in some areas by as much as 80%. While the traditional range termination of *E. maxima* kelp beds is documented as Cape Agulhas, along the south coast it has increased its cover from between 20% to 70%. It has now been photographed as far as De Hoop Nature Reserve, 61 km further than expected (Anderson *et al.* 2010; Bolton *et al.* 2012). These changes are most likely a result of changes in nutrient supply associated with increased upwelling and over all changing climatic conditions in the southern Benguela and Agulhas systems (Rouault *et al*. 2009; Rouault *et al*. 2010; Blamey *et al*. 2012). Increased competition from the invasive *M.* galloprovincialis, as well as the changing climatic conditions are likely the cause of the decrease in the abundance of the warm-water mussel *P. perna* and its range contraction eastwards away from False Bay. These observed changes in ranges are complicated by their dependence on their ecological communities and are also subject to biological interactions which can create larger community-scale disturbances. For example, the community changes brought about by the invasion of *J. lalandii* and the corresponding decrease in herbivores which have contributed to the increased biomass of *E. maxima* (Blamey 2010). A result of climate change is sea level rise which is expected to alter vertical zonation. However we observed no changes in zonation attributable to sea level rise.

Further disturbance pressure can result from the introduction of alien invasives, such as *M. galloprovincialis* and *B. glandula*. The loss of limpets and increase in seaweed cover on Marcus Island bears witness to some of these changes, *M. galloprovincialis* has increased its cover from 30% to 80% cover of the available intertidal habitat and provided a surface for seaweeds to increase its purchase further down the shore. *B. glandula* now occupies 50% of the rock surface in Blauberg. Furthermore changes in the zonation of the intertidal zone have occurred due to the introduction of these species.

The intertidal zone can also be affected by development along the shore. At all sites mentioned under anthropogenic changes at least 30% of the natural environment had been modified, and the most extreme change being documented in Bantry Bay where 90% of the available land had been developed. Changes in runoff regimes, as well as sewage and storm-water outfall can affect water quality leading to changes in benthic communities and harmful algal blooms.

The changes as recorded here visually, have been documented previously, but the impact of visual aids cannot be denied. This study adheres to the old adage that a picture is worth a thousand words. Any of the sections above could be investigated and quantified in and of themselves, and indeed do deserve to be. The use of photographic documentation is demonstrated. Many questions still remain, such as, what are the associated changes with the change in range of *E. maxima*? What environmental factors are driving these changes? Very little work has been done on the *B. glandula* threat, and what its real impact has been on the rocky shore environment. Repeat photography, such as this, allows for rapid sampling and the compilation of images over years could aid in detailing the changes rocky shores are undergoing. It is a tool that has been extensively utilized in terrestrial range management and could, with correct site selection and an annual or seasonal commitment to photographic sampling, be used to aid in the management of rocky shores.

There is also potential for developing a rapid assessment protocol, if percentage cover could be correlated with physical sampling techniques, the use of photography after pollution spills or large storms could be used to estimate the damage to the biota of the near shore environment.

Chapter 4. References

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