Assessment of the alien marine intertidal invertebrates in the Kogelberg Biosphere Reserve, South Africa

by
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ABSTRACT

The privilege of utilising resources from the globe's marine ecosystems is threatened. Adequate regional and global conservation efforts are vital. Understanding species' distributions and the environmental variables influencing community structures are important for instigating optimum conservation plans. A major threat facing marine ecosystems is the introduction and proliferation of alien invasive species. The distribution and geographical expansion of marine alien invasive species, along with environmental variables driving their increase is also critical for drawing up conservation plans. In response to this point, this study addresses gaps in our knowledge regarding intertidal marine invertebrate species distributions particularly that of marine alien species in the Kogelberg Biosphere Reserve (KBR) and adjacent Betty's Bay Marine Protected Area (MPA) in the Western Cape of South Africa, a prime area for south-coast marine conservation. Species identifications were done in September 2010. Two marine alien invertebrate species were identified along the coastline. These were the invasive mytilid Mediterranean mussel *Mytilus galloprovincialis* and the bryozoan *Watersipora subtorquata*. A quantitative systematic survey of the intertidal region, consisting of seven rocky shore sites was then undertaken from October 2010 to December 2010. The abundance of all focal marine invertebrate species, within six selected habitat types was assessed. Environmental variables were also recorded. Statistical tests were done to investigate for significant interaction in mean abundance and mean species richness among intertidal zones and protection status (MPA and non-MPA), and among intertidal sampling units (SU's) and protection status. There was also investigated for significant difference in mean abundance and mean species richness across intertidal zones and habitat types where protection status was not taken into account, and to test for significant differences in mean abundance and mean species richness between the intertidal zones and habitat types. Correspondence analyses were used to illustrate relationships between species composition and the defined environmental variables. Tests were done to investigate for significant differences in mean abundance of the identified marine alien species between the MPA and non-MPA, study sites, SU's, and wave exposures. These results indicated that vertical and horizontal stress gradients along the shore mainly influence abundance and species richness. The significantly higher abundance of *M. galloprovincialis* outside the MPA is due to its preference for wave-exposed areas. The reason for the difference in the total abundance of *W. subtorquata* in the different habitats is not clear, although it appears that the higher abundance of this species in the MPA is because the sheltered nature of the MPA. *M. galloprovincilais* displaces indigenous species. *W. subtorquata* does not cause any significant harm along the coast of South Africa. This study indicates that the abundance of *M. galloprovincialis* and *W. subtorquata* in the MPA is relatively low. The MPA is therefore in good condition with regards to intertidal alien species. It is important that the MPA and adjacent KBR's coastline maintain its current pristine status.

OPSOMMING

Die voorreg om mariene hulpbronne te benut, word bedreig. Voldoende nasionale en globale mariene bewaring is krities. ‘n Deeglike kennis van spesies verspreiding en omgewingsveranderlikes wat gemeenskapstrukture beïnvloed is noodsaaklik vir optimale bewaringspogings. Mariene indringerspesies en verspreiding van indringerspesies bedreig mariene ekosisteme. Dit is belangrik om basiese kennis in te win oor die geografiese verspreiding van indringerspesies saam met die omgewingsveranderlikes wat hierdie verspreiding beïnvloed. Hierdie studie fokus dus daarop om die kennis met betrekking tot intergety ongewerwelde spesiesverspreiding, veral dié wat uitheems is langs die kus van die Kogelberg Biosfeer Reservaat (KBR) en aangrensende Bettys Baai Mariene Beskermde Gebied (MBG) in die Wes-Kaap in Suid-Afrika uit te brei. Hierdie streek word beskou as gesog vir mariene bewaring in Suid-Afrika. Identifikasie van spesies is in September 2010 gedoen. Twee intergety uitheemse ongewerwelde spesies is langs die kus geïdentifiseer; die indringer ‘mytilid’ Mediterreense mossel - *Mytilus galloprovincialis* en die ‘bryozoa’ *Watersipora subtorquata*. ‘n Kwantitatiewe sistematiese oorsig van die intergety rotsagtige streek, bestaande uit sewe lokaliteite, is gedoen. Steekproefneming is vanaf Oktober 2010 tot Desember 2010 uitgevoer. Alle fokale intergety ongewerwelde spesies, in ses geselekteerde habitattipes is getel. Omgewingsveranderlikes is aangeteken. Daar is statisties getoets vir verskille in die talrykheid en spesiesrykheid tussen die intergety zones en beskermingsstatus (MBG en nie-MBG), en tussen die intergety habitattipes en beskermingsstatus. Daar is ook getoets vir verskille in talrykheid en spesiesrykheid tussen die intergety zones en habitattipes waar beskermingsstatus nie in ag geneem is nie. Ooreenstemmende analises is gedoen om die verhoudings tussen die spesiesamestelling en die gedefinieerde omgewingsveranderlikes te illustreer. Daar is statisties getoets vir verskille in die talrykheid van die geïdentifiseerde uitheemse spesies tussen die MBG en nie-MBG, die verskillende lokaliteite, die verskillende habitattipes en tussen die verskillende golf blootstellings. Hierdie

studie dui aan dat vertikale en horisontale stresgradiënte die hoofoorsaak is wat talrykheid en spesiesrykheid beïnvloed. Die aansienlik hoër talrykheid van *M. galloprovincialis* buite die MBG is van die gebied se hoër golfblootstelling. Die beduidende verskil in talrykheid van *W. subtorquata* is onduidelik. Dit kan egter afgelei word dat die beskutte aard van die MBG 'n rol speel by die aansienlik hoër talrykheid van hierdie spesie binne die MBG. *M. galloprovicialis* verplaas inheemse spesies. Geen beskikbare bewys dui daarop dat *W. subtorquata* enige beduidende skade langs die kus van Suid-Afrika aanrig nie. Die lae talrykheid van beide hierdie spesies binne die MBG dui daarop dat hierdie unieke area steeds in goeie toestand ten opsigte van intergety uitheemse spesies is. Dit is noodsaaklik dat die huidige status van die MBG en aangrensende kuslyn gehandhaaf word.

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Assessment of the alien marine intertidal invertebrates in the Kogelberg Biosphere Reserve, South Africa

1. Introduction

1.1. Marine protected areas

The vastness of the ocean functions as a complex ecosystem (Hyrenbach *et al.* 2000; Agardy 1994) of which little is known compared to its terrestrial counterpart (Lam 1998; Sobel 1993). Global knowledge of marine biodiversity, abundance and distribution is limited (Edgar *et al.* 2005; Edgar & Samson 2004; Turpie *et al.* 2000). As a result, marine ecosystems are behind in conservation efforts (Blamey & Branch 2009; Jones 2001; Tuck & Possingham 2000). The ocean is also considered to be the most productive and critical ecosystem with regards to its ecological processes (Hyrenbach *et al.* 2000). It has been only recently acknowledged that ocean biodiversity and ecology, ecosystem functioning, and resources are highly threatened (Agardy 1994). This is the result of human activities (Levin *et al.* 2002; Hyrenbach *et al.* 2000; Tuck & Possingham 2000), which include overexploitation, development and industry (Tunley 2009). Negative impacts are visible across the globe and may be irreversible (Levin *et al.* 2002; Hoegland *et al.* 2001; Hyrenbach *et al.* 2000; Agardy 1994). Coastal regions only notice degradation once it has reached accumulative damage (Agardy 1997).

Marine protected areas (MPA's) were first defined and implemented in the late nineteenth century (Hoegland *et al.* 2001). It is recognized as the only major approach for conserving marine ecosystems. MPA's have now been implemented in nearly every coastal country (Agardy *et al.* 2003). The selection, design and implementation of MPA's have mainly focussed on conserving exploited fish stock and other economical viable species, threatened species, biodiversity and ecosystems, and world heritage sites. Regions are also set aside to conduct valuable research and education (Boudouresque & Verlaque 2005).

Major global marine conservation shortcomings include confusion that exists with regards to MPA design and management (Agardy *et al.* 2003; McNeill 1994; Sobel 1993), and insufficient scientific knowledge (Jones 2001). The use of different terminology, aimed at defining the objectives for a specific MPA, is an obstacle that needs to be addressed (Levin *et al.* 2002; McNeill 1994). Furthermore, insufficient assessment of the success of MPA's has been done on a global scale (Mora *et al.* 2006).

MPA's are defined according to the objectives to be met for a specific MPA. The latter gave rise to a magnitude of terms describing MPA's (Agardy *et al.* 2003). The following definition provides clarity in the broader context: the rationale of MPA's is to manage the use of space and exploitation of resources, protecting threatened and endangered species, their habitats (Tunly 2009; Hoegland *et al.* 2001; Tuck & Possingham 2000; Agardy 1997) and the ecosystem that supports them (Hoegland *et al.* 2001; Agardy 1997), and to protect features of historical or cultural importance by regulating or prohibiting specific or all human activities (Agardy *et al.* 2003). It plays a key role in education, and presents sites to conduct important research (Tunley 2009). The world's most recognised definition is that of the International Union for Conservation of Nature (IUCN) defining MPA's as "any area of intertidal or subtidal terrain, together with its overlying waters and associated flora, fauna, historical and cultural features, which has been reserved by legislation to protect part or all of the enclosed environment" (Agardy *et al.* 2003; Levin *et al.* 2002; McNeill 1994).

Holistic management methods should be developed to protect and maintain the ecological processes of the marine environment. It is not viable to only focus on isolated regions of this highly integrated and vast open system. Management networking (Mora *et al.* 2006; McNeill 1994) and process-orientated conservation (Jennings *et al.* 1996) is therefore critical. An understanding of how these processes and their functioning link with one another is a challenge. This must be addressed to ensure successful conservation of the marine environment

(Agardy 1997). This imposes a major challenge, as the migrations of many species within corridors are not understood extensively (Hyrenbach *et al.* 2000). It is in human communities' interests that social and economical implications are addressed with the development of management strategies for a region (Agardy 1994).

MPA design has mainly focussed on near shore habitats. This is due to insufficient available information of pelagic ecosystems. Managing large multi-purpose MPA's is of great interest as it provides opportunities to conserve highly mobile species. It also protects demarcated regions from pollution and unwanted negative impacts from human activities (Hyrenbach *et al.* 2000; McNeill 1994).

It is essential that small, demarcated, protected areas should function as no-take zones. New Zealand's Leigh Marine Reserve, proclaimed in 1977, is one of the first successful small no-take MPA's (Sobel 1993). Sobel (1993) proposes that small no-take MPA's should be interspersed within large multi-purpose marine reserves. Australia's Great Barrier Reef Marine Reserve is the world's largest marine reserve (Hoegland *et al.* 2001). The Great Barrier Reef Marine Reserve functions as a multi-purpose reserve (Jones 2001) integrated with no-take areas (Sobel 1993). Mora *et al.* (2006) found that 1% of all coral reefs are protected by no-take and no poaching MPA's. Forty percent of protected coral reefs are conserved in small MPA's of 1 to 2 km^2 in surface area. This is inadequate, as propagule dispersal of coral reef species covers areas of more than 10 km^2 (Mora *et al.* 2006).

1.2. Marine protected areas of South Africa

The coastline of South Africa is influenced by the cold Benguela Current's upwelling system along the west coast and by the warm Mozambique Current along the east coast. This gives rise to a rich diversity of marine species (Tunley 2009). The South African marine community is considered relatively pristine (Branch *et al.* 2010).

There is a need to expand the presence of well managed marine reserves in South Africa (Awad *et al.* 2002). Marine communities along the coast have experienced much pressure from human intervention. Marine regions are being overexploited and degraded (Tunley 2009). Development, pollution from industry, climate change, marine alien species introductions and recreational use of the marine environment threatens the coastal regions (Tunley 2009). In 2007, an astonishing 302 of South Africa's marine species were listed on the IUCN's Red List of Threatened Species.

The benefits that MPA's provide for the management of fisheries gave rise to South Africa's commitment to protect 20% of the coastline by 2012. This has been exceeded as a total of 23% of the coastline was protected by 2010 (Griffiths *et al.* 2010). The Department of Environmental Affairs' Marine and Coastal Management branch regulates all marine and coastal activities (Tunley 2009), which is managed by designated conservation institutions.

Based exclusively on marine invertebrate distribution, it was found by Awad *et al.* (2002) that priority regions for the establishment of MPA's are in Durban and along the coast of Port Elizabeth and East London. In 2009, South Africa proclaimed 22 Government Gazetted MPA's. The conservation objectives may include protecting threatened and endemic species, world heritage sites, associated ecosystems such as wetlands, estuaries and lagoons, providing breeding grounds, and preventing overexploitation and other negative human impacts (Tunley 2009).

It is of concern that many MPA's in South Africa function as isolated units. In 2010, only 9% of all the MPA's were declared as no-take MPA's. Protected regions also exhibit a spatial imbalance. This needs to be addressed to ensure that a greater proportion of South Africa's marine biodiversity is protected (Griffiths *et al.* 2010).

The Tsitsikamma MPA, established in 1964, is the first and largest MPA in South Africa (Tunley 2009; Turpie *et al.* 2006). The West Coast National Park is accompanied by a key marine reserve (Robinson *et al.* 2004). Langebaan Lagoon, situated within the reserve, is proclaimed as a Ramsar Site. Ramsar Sites are defined as internationally important wetlands (Tunley 2009; Robinson *et al.* 2007b; Robinson *et al.* 2004). Furthermore, the establishment of the De Hoop Marine Reserve in 1985 significantly contributed to the recovery of exploited fish stocks (Bennett & Attwood 1991).

Tunley (2009) conducted an extensive survey in 2009 outlining shortcomings and improvements of implemented MPA's. Supplementary improvements to be implemented were identified. At the time of this survey the Kogelberg Biosphere Reserve's Betty's Bay MPA did not have skilled staff employed (Tunley 2009). Nevertheless, currently this MPA is being patrolled, and educational programmes are in place. CapeNature employed a MPA manager with an office situated in Betty's Bay.

1.3. Marine alien and invasive species

The introduction of marine alien species within a new home range is often aided through human activities (Floerl & Inglis 2005; Streftaris *et al.* 2005; Robinson *et al.* 2004; Levin *et al.* 2002; Orensanz *et al.* 2002; Stachowicz *et al.* 2002ab; Stachowics *et al.* 1999; Grosholz & Ruiz 1996). Biotic invaders could be detrimental to the environment causing extinctions, a significant decline in native populations and severe habitat alterations (Rius *et al.* 2009; Mack *et al.* 2000; Stachowics *et al.* 1999)

Marine alien species have been introduced by accident or intentionally (Robinson *et al.* 2005; Streftaris *et al.* 2005; Robinson *et al.* 2004; Bax *et al.* 2003; Grosholz 2002; Levin *et al.* 2002; Stachowicz *et al.* 2002ab; Kolar & Lodge 2001; Mack *et al.* 2000; Stachowics *et al.* 1999) through various vectors (Streftaris *et al.* 2005; Bax *et al.* 2003; Everett 2000). Marine alien species are generally only given attention once they have established and have altered their new environment (Occhipinti-Ambrogi *et al.* 2011).

Introductions of marine alien species began with the establishment of transoceanic travel (Rius *et al.* 2009; Robinson *et al.* 2005; Bax *et al.* 2003; Occhipinti-Ambrogi & Savini 2003), considered to be the most prevalent method

(Molnar *et al.* 2008; Orensanz *et al.* 2002; Lewis *et al.* 2003; Mack *et al.* 2000). There is generally a statistically positive correlation in the rate of introductions and the rate of increase in transport and trade (Robinson *et al.* 2005; Robinson *et al.* 2004; Mack *et al.* 2000). Methods of transport include larvae in ballast water and fouling of ship hulls (Cohen *et al.* 2005; Floerl & Inglis 2005; Streftaris *et al.* 2005; Floerl *et al.* 2004; Robinson *et al.* 2004; Bax *et al.* 2003; Lewis *et al.* 2003; Grosholz 2002; Mack *et al.* 2000). The shipworm *Teredo navalis* bores into wooden structures. This species has been introduced globally as a fouling organism (Bax *et al.* 2003). Marine alien species are also introduced through recreational use of the ocean, industries such as gas, oil and construction, and the trade of species for mariculture, aquaculture and aquariums (Occhipinti-Ambrogi & Savini 2003; Grosholz 2002). The Pacific oyster *Crassostrea gigas* has been intentionally introduced for culturing in 45 regions (Molnar *et al.* 2008).

An incalculable number of immigrants do not establish in their new environment due to unfavourable conditions (Molnar *et al.* 2008; Floerl & Inglis 2005; Levin *et al.* 2002; Mack *et al.* 2000). However, once a species becomes invasive, it may have adverse implications to the indigenous marine biodiversity and marine communities (Occhipinti-Ambrogi *et al.* 2011; Lynch & Rochette 2009; Rius *et al.* 2009; Molnar *et al.* 2008; Robinson *et al.* 2007ab; Stachowicz & Byrnes 2006; Streftaris *et al.* 2005; Bax *et al.* 2003; Byers & Noonburg 2003; Grosholz & Ruiz 2003; Occhipinti-Ambrogi & Savini 2003; Levin *et al.* 2002; Stachowicz *et al.* 2002ab; Kolar & Lodge 2001; Everett 2000; Mack *et al.* 2000; Ruiz *et al.* 1999; Stachowics *et al.* 1999), economy and human health (Occhipinti-Ambrogi *et al.* 2011; Molnar *et al.* 2008; Streftaris *et al.* 2005; Stachowicz *et al.* 2002ab; Kolar & Lodge 2001; Mack *et al.* 2000; Stachowics *et al.* 1999).

Marine introductions and invasions are considered irreversible (Bax *et al.* 2003). Genovesi (2005) and Branch & Steffani (2004) note two localised successful eradication efforts of marine alien species. These are the removal of the Asian mussel *Mytilopsis sallei* in Cullen Bay, Australia and the bristleworm *Terebrasabella heterounicata* from a mariculture operation in California.

Prevention of introductions is therefore essential for effective conservation (Bax *et al.* 2003).

Predicting whether introduced species will become invasive is challenging, yet critical. Coastal invaders spread quickly over extensive geographical ranges (Grosholz 2002). The invasive history of a species and the native community dynamics must be understood. This is essential for predicting whether a new home range of a species is favourable for its proliferation (Branch & Steffani 2004). Environmental stress provides favourable conditions for invasion. It plays a significant role in the increasing rate of introductions. Environmental stress is mainly from human mediated activities and pollution (Occhipinti-Ambrogi *et al.* 2011; Occhipinti-Ambrogi & Savini 2003; Ruiz *et al.* 1999).

Invasion and impacts of the zebra mussel *Dreissena polymorpha* in the North American Great Lakes stimulated the need to gain knowledge of coastal invasive species (Grosholz 2002).

The mytilid Mediterranean mussel *Mytilus galloprovincialis* is a successful invader across the globe (Branch & Steffani 2004). The global distribution of this species is visible in both hemispheres of temperate climates (Bownes & McQuaid 2006). Intense grazing of the periwinkle *Littorina littorea* along the coast of Rhode Island, United States of America (USA) dramatically changed the soft sediment environment to a hard surface (Grosholz 2002). Phytoplankton blooms are a food source for its associated pelagic ecosystems. The halt of the production of San Francisco Bay's, USA phytoplankton is related to the invasion of the Asian clam *Potamocorbula amurensis* (Grosholz 2002). Subtidal invasion of New England, USA by the green alga *Codium fragile* and the bryozoan *Membranipora membranacea* resulted in the decline of the growth of native kelp forests (Levin *et al.* 2002). The collapse of the bivalve fisheries industry of eastern North America resulted from the introduction of the European shore crab *Carcinus maenas* (Bax *et al.* 2003).

Streftaris *et al.* (2005) and Occhipinti-Ambrogi *et al.* (2011) reviewed marine alien species introductions and invasions of the European seas and the

Italian coastline respectively. The literature provides valuable information on the status of these regions. Spread of alien species from the Red Sea to the Mediterranean Sea in North Africa occurred rapidly with the opening of the Suez Canal in 1969 (Streftaris *et al.* 2005; Occhipinti-Ambrogi & Savini 2003). Lessepsian migration is the movement of marine alien species from the Red Sea to the Mediterranean Sea. This migration has caused a dramatic increase in the marine alien fauna of the Mediterranean Sea (Turon *et al.* 2007; Streftaris *et al.* 2005; Occhipinti-Ambrogi & Savini 2003). Occhipinti-Ambrogi & Savini (2003) state that marine alien fauna species comprise 5% of the total 13% fauna species present in the Mediterranean Sea. Streftaris *et al.* (2005) state that 12% of the benthic community along the coast of Israel originates from the Red Sea.

Waste water pollution from aquaculture, trawling activities and pollution from agriculture and development resulted in a decrease in the abundance of the native sea grass *Posidonia oceanica* along the Mediterranean coast. The decline of *P. oceanica* decreased its competitive nature with the green algae *Caulerpa taxifolia* and *Caulerpa racemosa. C. taxifolia* and *C. racemosa* continue to proliferate in the region (Occhipinti-Ambrogi & Savini 2003) and is considered as one of the most severe modifiers of critical habitats along the Italian coast (Occhipinti-Ambrogi *et al.* 2011). The Italian coastline currently has 165 recorded marine alien species. The most favourable region for introductions is the Venice Lagoon. Rapid environmental changes significantly increased the number of introductions within the lagoon (Occhipinti-Ambrogi *et al.* 2011).

The Black Sea has been highly invaded, causing significant changes in the natural functioning of its ecosystem (Paavalo *et al.* 2005; Occhipinti-Ambrogi & Savini 2003). Introduction of the comb jelly fish *Mnemiopsis leidyi* consumed zooplankton blooms. This resulted in the collapse of various fish stocks within the Black Sea (Streftaris *et al.* 2005; Occhipinti-Ambrogi & Savini 2003; Bax *et al.* 2003). The zooplankton blooms have recovered with the introduction of the invasive comb jelly fish *Beroe ovata* feeding on *M. leidyi* (Occhipinti-Ambrogi & Savini 2003).

Orenzans *et al.* (2002) reviewed marine alien species introductions and invasions of the south-western Atlantic. The literature provides valuable information on the status of the south-western coast of Argentina, Uruguay and Brazil. Invasion by the tube worm *Ficopomatus enigmaticus*, a reef building species, significantly changed the sedimentation dynamics of the Mar Chiquita Lagoon. This considerably hindered navigation of the lagoon. The mussel *Limnoperna fortunei* has spread vigorously in the La Plata Basin. This invasion causes the blocking of filters of water treating, industrial and power-generating plants. Mortality of this species also causes obstruction of pipes, which results in contamination of the water efflux into the basin. Other invasions of the south-western Atlantic include the Pacific barnacle *Balanus glandula*, the geographic expansion of the Pasific barnacle *Cassostrea gigas*, and the alteration of benthic marine communities from the invading kelp *Undaria pinnatifida* (Orensanz *et al.* 2002).

It is of concern that the introduction marine alien species in the Southern Ocean may increase from increased tourist and research activities (Lewis *et al.* 2003). Climate change has put much stress on natural environments. An increase in temperature of the Southern Ocean may also lead to an increase in introductions (Lewis *et al.* 2003).

The introduction of pathogens and parasites along coastal regions is of great concern (Streftaris *et al.* 2005; Bax *et al.* 2003). The bacterium *Vibrio cholerae* has been introduced in major harbours of the USA (Ruiz *et al.* 1999). The global toxic red tide phenomena is partly the result of the introduction of dinoflagellates (Streftaris *et al.* 2005; Bax *et al.* 2003).

1.4. Marine alien and invasive species of South Africa

South Africa’s marine environment is diverse (Branch *et al.* 2010), and more diverse ecosystems decrease invasive success (Stachowicz & Byrnes 2006; Byers & Noonburg 2003; Stachowicz *et al.* 2002a). This is from a decrease in available space (Stachowicz *et al.* 2002a) and resources (Stachowicz & Byrnes 2006).

Marine alien species in South Africa are less prevalent than in other regions of the world. However, many regions along the coast of South Africa have not been surveyed extensively (Griffiths *et al.* 2009ab; Robinson *et al.* 2005). Incomplete taxonomic knowledge of South Africa's marine invertebrate species makes it difficult to detect new introduced species (Griffiths *et al.* 2009ab).

Most research of marine alien and invasive species has focussed on the invasive nature of a single species: The mytilid Mediterranean mussel *Mytilus galloprovincialis*. Research on this species has mainly been done along the south and south-east coast by Bownes & McQuaid (2009), Rius & McQuaid (2009), Hanekom (2008), Nicastro *et al.* (2008), von der Meden *et al.* (2008), Bownes & McQuaid (2006) and Zardi *et al.* (2006), and to the west of Cape Peninsula by Robinson *et al.* (2007ab), Steffani & Branch (2005) and Steffani & Branch (2003). It appears that the south-west coast to the east of Cape Peninsula has not received similar research attention.

Saldanha Bay Harbour, one of South Africa's larger ports, is known to have a high risk of new introductions. Parts of Saldanha Bay and Langebaan Lagoon are included within the West Coast National Park Marine Reserve (Robinson *et al.* 2004). The marine reserve has been surveyed extensively for marine alien species. Nine of the 17 marine alien species recorded in 2000 were found to be present in Saldanha Bay Harbour in 2004 (Robinson *et al.* 2004). The invasive clam *Tapes philippinarum* and the abalone *Haliotis rufescenens* have since disappeared. *M. galloprovincialis*, the European shore crab *Carcinus maenas* and the periwinkle *Littorina saxatilis* were found to have naturalised outside the boundaries of the harbour (Robinson *et al.* 2004). Further recorded species in the harbour include the ascidians *Diplosoma listerianum*, *Ciona intestinalis* and *Ascidiella aspersa*. Another species recorded in 2002 outside the harbour boundaries is the anemone *Sagartia ornata* (Robinson *et al.* 2004).

Robinson *et al.* (2004) recorded *M. galloprovincialis* at 79% of the surveyed rocky shore sites along the West Coast National Park Marine Reserve. This is the only research focussing on multiple marine alien species abundance and distribution in South Africa. Branch *et al.* (2010) and Griffiths *et al.* (2009b) document the past distribution of marine alien species.

First evidence of *C. maenas* was noted in 1983 at Table Bay Harbour (Robinson *et al.* 2005; Grosholz & Ruiz 1996). It was found to be present at seven intertidal sites by 1990, with no further expansion along the west coast (Robinson *et al.* 2005). This could be from their limited ability to get a hold along the wave-exposed west coast (Hampton & Griffiths 2007). This species is therefore confined to sheltered sites (Hampton & Griffiths 2007; Grosholz & Ruiz 1996). Hampton & Griffiths (2007) found that *C. maenas* has shorter and lighter legs than the native crab *Plagusia chabrus*, giving it less stability and poor grip. The diet of *C. maenas* compared to native crabs was found to be dissimilar, limiting competition between these species (Grosholz & Ruiz 1996).

Significant populations of the Pacific barnacle *Balanus glandula* have recently being detected along the west coast (Griffiths *et al.* 2009ab) from Misty Cliffs to Elands Bay (Laird & Griffiths 2008). This invasive barnacle had been misidentified and it is recognised that the South African west coast has been invaded for as long as 15 years. *B. glandula* occupies the Balanoid zone covering up to 90% of the rock face in some regions of Saldanha Bay (Griffiths *et al.* 2009b).

M. galloprovincialis and *B. glandula* have proliferated along the South African coast (Griffiths *et al.* 2009ab). Griffiths *et al.* (2009b) lists 21 marine alien species along the South African coastline. Branch *et al.* (2010) lists 24 marine alien species. A combination of the two records gives a total of 31 introduced marine alien species in South Africa (Appendix A). Appendix A also lists the distribution range and impacts of the introduced marine alien species. Branch *et al.* (2010) note that the alien bryozoan *Membranipora membranacea* has been misidentified for the bryozoans *Membranipora rustica*, *M. membranacea* now being excluded from the alien species list.

1.5. *Watersipora subtorquata* introductions and impacts

The red-rust bryozoan *Watersipora subtorquata* is encrusting in nature, forming colonies up to a few centimetres in diameter (Abdel-Salam & Ramadan 2008). Colonies have a red to brownish appearance (Abdel-Salam & Ramadan 2008) outlined with black dots (Branch *et al.* 2010).

W. subtorquata has been recorded in various regions of the world (Abdel-Salam & Ramadan 2008; Floerl *et al.* 2004). It is now considered to have a cosmopolitan distribution (Clark & Johnston 2009). This species is tolerant to antifouling biocides used on ship hulls (Floerl & Inglis 2005; Floerl *et al.* 2004). Experiments conducted by Floerl *et al.* (2004) indicate that *W. subtorquata* facilitates and enhances the presence of various sessile marine species during transoceanic travel. Species aggregating on *W. subtorquata* that otherwise would not foul treated ship hulls may therefore be transferred to new locales (Floerl *et al.* 2004). *W. subtorquata* provides refuge for juvenile fish and mobile invertebrate species (Nydam & Stachowicz 2007). Experiments conducted by Clark & Johnston (2009) show that *W. subtorquata* has a fast growth rate on available open space. This is an important finding contributing to its invasive success.

A case-study conducted in Australia by Marshall & Keough (2008) found that larger species with large offspring size have a higher rate of survival success. The offspring size of the species was found to be larger in the colder temperate region of Melbourne. Offspring size may however decrease below specific temperatures (Marshall & Keough 2008). This could explain its current distribution as a marine alien species along the south-west coast of South Africa (Branch *et al.* 2010). It has not been recorded that this species causes significant harm to the biodiversity of the coast of South Africa.

1.6. *Mytilis galloprovincialis* invasions in South Africa

The invasive mytilid Mediterranean mussel *Mytilus galloprovincialis* is black, sometimes bluish in colour. It may appear slightly brown towards the lower part of the shell. Tiny holes are present in the resilial ridges of the inner surface (Branch *et al.* 2010).

Globally, *M. galloprovincialis* is considered a successful invasive species. This species has high fecundity, re-colonisation rate (Nicastro *et al.* 2008; Robinson *et al.* 2007b; Bownes & McQuaid 2006) and resistance to desiccation (Robinson *et al.* 2005; Robinson *et al.* 2004). *M. galloprovincialis* was first recorded in South Africa along the west coast at Saldanha Bay Harbour in 1979 (Branch & Steffani 2004). It is now the most invasive marine invertebrate species in South Africa (Nicastro *et al.* 2008; Robinson *et al.* 2007a; Robinson *et al.* 2005; Branch & Steffani 2004; Robinson *et al.* 2004). It has the ability to proliferate at a range of temperatures (Nicastro *et al.* 2008) invading the west, south-west coast and the east coast of South Africa. A survey conducted by Robinson *et al.* (2005) indicates that *M. galloprovincialis* is present along the west coast and the east coast, almost reaching East London.

M. galloprovincialis is characteristic at sites with medium to high wave energy and decline in numbers or is absent at sites with low or extreme wave energy (Blamey & Branch 2009; Steffani & Branch 2003). Water flow is essential for suspension feeders to draw up free floating food particles present in the water. Sites with extreme wave energy display lower growth rates of *M. galloprovincialis*. Individuals may dislodge from the substratum or close their valves limiting their food consumption capacity (Hammond & Griffiths 2004). The larval stage of *M. galloprovincialis* is planktotrophic, allowing the species to travel with currents for weeks. This pattern of dispersal was clearly observed along the east coast of South Africa. It was observed that this species expands its range over a short period of time (McQuaid & Phillips 2000).

It has substantially altered the mussel beds of South Africa (Robinson *et al.* 2007a; Robinson *et al.* 2005) increasing mussel biomass along the coastline (Robinson *et al.* 2005). It displaces indigenous species (Bownes & McQuaid 2006) and affects intertidal community structures (Robinson *et al.* 2007ab; Robinson *et al.* 2005). By 1990, *M. galloprovincialis* became the dominant mussel along the west coast, displacing the indigenous mussels *Aulacomya ater*

(Robinson *et al.* 2007a; Bownes & McQuaid 2006; Robinson *et al.* 2004) and *Choromytilus meridionalis* (Robinson *et al.* 2007a; Robinson *et al.* 2004). Robinson *et al.* (2004) recorded *M. galloprovincialis* at 79% of the sampled rocky shore areas in the West Coast National Park, with *C. meridionalis* and *A. ater* found at only three and four of the sites respectively. *M. galloprovincialis* has not completely displaced the indigenous mussel *Perna perna* (Bownes & McQuaid 2009; Robinson *et al.* 2005). *P. perna* flourishes in the lower mussel zone, while *M. galloprovincialis* flourishes in the upper mussel zone (Erlandsson *et al.* 2006). This could be from the stronger basal attachment of *P. perna* (Erlandsson *et al.* 2006) and the higher tolerance of *M. galloprovincialis* to desiccation (Rius & McQuaid 2009).

M. galloprovincialis outcompetes the adult limpet species *Scutellastra granularis* from open rock surfaces (Steffani & Branch 2005). The total density of *S. granularis* increases as *M. galloprovincialis* provides refuge for juveniles (Robinson *et al.* 2005). This poses a problem, as juvenile *S. granularis* settling on the surface of mussels do not reach their appropriate body size for maximum reproduction (Branch & Steffani 2004). *M. galloprovincialis* displaces the limpet *Scutellastra argenvillei* in areas that are subjected to fairly strong wave action (Robinson *et al.* 2007a; Robinson *et al.* 2005; Steffani & Branch 2005; Branch & Steffani 2004).

Robinson *et al.* (2007b) found that the once invaded sites of Langebaan Lagoon supported a higher biomass of rocky shore specific species than at the non-invaded sites. This was from the favourable refuge provided by *M. galloprovincialis* (Robinson *et al.* 2007ab). Sulphide accumulation within the sediment from the presence of the mussel beds excluded sand-flat species from the invaded sites. After *M. galloprovincialis* died off from an unknown reason, it was found that the removal of the mussel shells helped with the recovery of the sandbank communities (Robinson *et al.* 2007b).

The endemic oystercatcher *Haematopus moquini* adapted its diet to mainly feed on *M. galloprovincialis*. This significantly recovered the numbers of *H. moquini* (Robinson *et al.* 2005; Branch & Steffani 2004). Local human communities along the west coast have also reaped benefits from the introduction of *M. galloprovincialis*. A small-scale mussel culturing industry is in operation along the west coast (Robinson *et al.* 2005; Robinson *et al.* 2004).

1.7. Study objectives

The focus of this study is to specifically fill gaps in our knowledge of the current status of intertidal rocky shore marine alien species and marine biodiversity. The aim is to focus on a small geographical region which has not received much research attention. The selected study region is the coastline of the Kogelberg Biosphere Reserve (KBR), accompanied by a significant MPA, the Betty's Bay MPA. Only the marine alien invertebrate species *W. subtorquata* and *M. galloprovincialis* found along the rocky shores of this study are discussed in the introduction. However, many other alien species has been recorded in South Africa (Appendix A). Knowledge gained may assist management authorities to make sound management decisions for the MPA and its bordering coastline.

The following specific objectives are addressed:

1) Are intertidal marine alien invertebrate species present along the rocky shores of the MPA and its adjacent coastline?
2) Is there a significant difference in the total intertidal marine invertebrate abundance and species richness between the MPA and the adjacent non-MPA (NMPA) areas, and if so, how is it influenced by environmental variables?
3) Is there a significant difference in the intertidal marine alien invertebrate species abundance between the MPA and the adjacent NMPA, and if so, how is it influenced by environmental variables?

2. Methods

2.1. Study area

The Kogelberg Biosphere Reserve (KBR), a near pristine mountainous and coastal conservation area, is situated 90 km south-east of Cape Town in the Western Cape Province, South Africa. The KBR includes terrestrial and marine ecosystems along the south-eastern boundary of the False Bay coastline. It stretches along the coast from Gordons Bay to the Bot River mouth, and to the north in the region of Grabouw. It is protected from metropolitan development to the west and rural development to the north-east. The KBR covers a core region of 103 629 ha. International principles of biosphere reserve conservation state that the core region must be left untouched (Stanvliet *et al.* 2004). The management authority for the KBR, which includes the Betty's Bay MPA, is CapeNature (Tunley 2009). The region has a Mediterranean climate subjected to cold winds coming from the Atlantic Ocean.

The KBR coastline is diverse, outlined with sandy beaches and rocky shores, estuary runoffs, dense subtidal kelp forests and reefs. The Betty's Bay MPA was first declared in 1981 as the H.F. Verwoerd Marine Reserve (Tunley 2009). Under the Marine Living Resource Act 18 of 1998, the marine reserve was proclaimed the Betty's Bay MPA in 2000. It stretches approximately 8 km along the coastline. The western boundary (34°22'45'' S; 18°53'46'' E) is located at beacon B1 in Stoney Point and the eastern boundary (34°21'21'' S; 18°56'14” E) is located to beacon B4 to the east of Jock's Bay. The southern boundary is located at two nautical miles south (34°24'45'' S) from the high tide mark (WWF 2009). The MPA is relatively small with a surface area of 21.14 km^2 (Tunley 2009; WWF 2009)

The African penguin *Spheniscus demersus* colony, a Red Listed species and other species, particularly the abalone *Haliotis midae*, line fish and the west-coast rock lobster *Jasus lalandii* are protected within the MPA (Tunley 2009; WWF 2009).

The protected coastal region experiences much pressure from overexploitation (Tunley 2009), abalone poaching (Stanvliet *et al.* 2004) and development (Tunley 2009). Shore angling is the only resource utilization activity allowed. It is considered that the Betty's Bay MPA will function adequately if declared as a no-take MPA (Tunley 2009).

2.2. Study sites

The study was done along the False Bay coastline of the Kogelberg Biosphere Reserve (KBR), accompanied with a significant MPA, the Betty's Bay MPA, in the Western Cape Province, South Africa. Rocky shore sites were identified from Kogel Bay to Kleinmond over a linear distance of ±35 km. Distances between the sites varied considerably. The longest distance between sites was 7 km and the shortest distance was 800 m. The sites were selected based on the comparability between distinct habitat types.

Seven sites were selected referred to as Kogel Bay (KB) (34°14'40" S; 18°51'11" E), Rooi Els (RE) (34°17'57" S; 18°48'59" E), Pringle Bay (PR) (34°20'52" S; 18°49'29" E), Stoney Point (SP) (34°22'14" S; 18°53'41" E), Betty's Bay Main Beach (MB) (34°22'01" S; 18°53'58" E), Jock's Bay (JB) (34°21'26" S; 18°55'59" E) and Kleinmond (KM) (34°20'44" S; 18°59'34" E). KB, RE and PB are located to the west of the MPA, and SP, MB and JB are located within the MPA, and KM is located to the east of the MPA (Fig. 1).

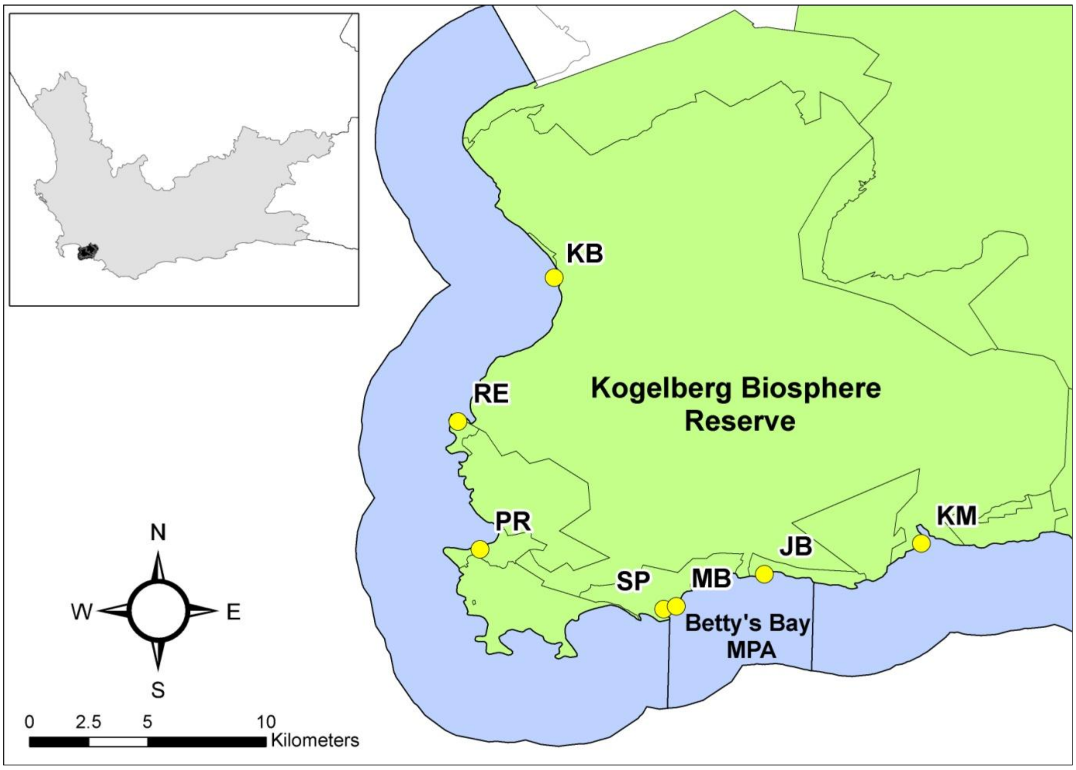

Figure 1. Study sites along the Kogelberg Biosphere Reserve (KBR) coastline, Western Cape Province, South Africa. (MPA=marine protected area, KB=Kogel Bay, RE=Rooi Els, PR=Pringle Bay, SP=Stoney Point, MB=Betty’s Bay Main Beach, JB=Jock’s Bay, KM=Kleinmond)

2.3. Study site descriptions

Rocky shores are subjected to wave action which varies considerably along the coastline. Hydrodynamic force influences the community structure of intertidal rocky shores (von der Meden *et al.* 2008; Denny 2006; Zardi *et al.* 2006; Hammond & Griffiths 2004). It affects the distribution of species, the degree of disturbance to species, and their feeding patterns. Species typically found to occur in a region have been successfully predicted according to the extent of wave exposure (Denny 2006).

The Class Bivalvia consists of filter feeding species. A number of mussel species grow on exposed rock face low on the shore (Branch *et al.* 2010). These species require a sufficient flow of water for feeding. A variety of mussel species typically prefer wave-exposed regions. The barnacle, *Octomeris angulosa* is characteristic of areas exposed to high wave energy (Branch *et al.* 2010). The study sites were classified as sheltered, partially exposed and exposed *a priory* based on characteristic species.

Kogel Bay (KB) is adjacent to the main beach west of the MPA. It stretches ±90 m along the shore and ±50 m offshore. It is classified as exposed. The structural geology is primarily flat rock. Crevice formation of the rock face is confined to a small area of the site. Dense bands of mussel grow on the rock face low on the shore. Kelp beds of *Ecklonia maxima* and *Laminaria pallida* are not present offshore (Fig. 2a).

Rooi Els (RE) is adjacent to the main beach west of the MPA. It stretches ±90 m along the shore and ±65 m offshore. It is classified as exposed. The structural geology of the rock face is fragmented. Large boulders are present high on the shore. Dense bands of mussel grow on the rock face low on the shore. Kelp beds of *E. maxima* and *L. pallida* grow offshore (Fig. 2b).

Pringle Bay is adjacent to the main beach west of the MPA. It stretches ±225 m along the shore and ±55 m offshore. It is classified as sheltered. The structural geology of the rock face is fragmented. Large boulders are present high on the shore. Kelp beds of *E. maxima* and *L. pallida* grow offshore (Fig. 2c).

Stoney Point is within the MPA adjacent to beacon B1. It stretches ±390 m along the shore and ±40 m offshore. It is classified as sheltered. The structural geology of the rock face is fragmented. Kelp beds of *E. maxima* and *L. pallida* grow offshore (Fig. 2d).

Betty’s Bay Main Beach (MB) is within the MPA. It stretches ±265 m along the shore and ±65 m offshore. It is classified as sheltered. The structural geology of the rock face is fragmented. Kelp beds of *E. maxima* and *L. pallida* grow offshore (Fig. 2e).

Jock’s Bay (JB) is within the MPA adjacent to beacon B4. It stretches ±210 m along the shore and ±60 m offshore. It is classified as partially exposed. The structural geology of the rock face is fragmented. Patches of mussel grow on the rock face low on the shore. Kelp beds of *E. maxima* and *L. pallida* grow offshore (Fig. 2f).

Kleinmond (KM) is adjacent to the Palmiet River east of the MPA. It stretches ±90 m along the shore and ±55 m offshore. It is classified as partially exposed. The structural geology of the rock face is fragmented. Patches of mussel grow on the rock face low on the shore. Kelp beds of *E. maxima* and *L. pallida* grow offshore (Fig. 2g).

Figure 2a. Kogel Bay rocky shore

Figure 2b. Rooi Els rocky shore

Figure 2c. Pringle Bay rocky shore

Figure 2d. Stoney Point rocky shore

Figure 2e. Betty's Bay Main Beach rocky shore

Figure 2f. Jock's Bay rocky shore

Figure 2g. Kleinmond rocky shore

Figure 2. The rocky shore study sites included in this study

2.4. Zonation of rocky shores

Rocky shores mainly display three primary zones; the Littoral zone (supralittoral fringe), Balanoid zone (midlittoral zone) and sublittoral fringe (infralittoral fringe) (Branch *et al.* 2010; Stephenson & Stephenson 1949). The Littoral zone is the dry region which forms the transition towards the adjacent terrestrial region. This region is generally only submerged during spring tide. Depending on the environmental conditions, the exposed rock face may be subjected to spray. The Balanoid zone is completely intertidal. Two subzones are identified, the upper Balanoid zone and the lower Balanoid zone. The sublittoral fringe forms the lowest band along rocky shores. It has the shortest exposure time during low tide (Stephenson & Stephenson 1949).

Zonation of rocky shores is subjective, as distinct conditions influence zonation patterns. However, the primary zones are considered universal. Additional zones and variation of subzones are widely recognised. A combination of specific conditions resulted in an additional zone, the limpet *Patella cochlear* zone, along specific regions of the South African coast.

It is recommended that the characteristic species should define distinct zones and subzones (Stephenson & Stephenson 1949). Presence of the class Bivalvia and the class Cirripedia is indicative of the sublittoral fringe and the lower Balanoid zone. A reduction in numbers or omission of the class Cirripedia represents the upper Balanoid zone (Stephenson & Stephenson 1949). Dominance of Littorina species is indicative of the Littoral zone (Branch *et al.* 2010).

2.5. Intertidal zones and sampling unit descriptions

A pilot study was done in September 2010 to determine characteristic species distinguishing the intertidal zones and habitat types, referred to as sampling units (SU’s). Intertidal zones and SU’s were selected *a priori* based on characteristic species.

Isopod, amphipod and insect species amongst washed up seaweed, and the periwinkle *Littorina knysnaensis* are the only marine invertebrate species present along the study area's Littoral zone (Branch *et al.* 2010). Marine alien isopod and amphipod species of South Africa are confined to major harbours and lagoons (Branch *et al.* 2010; Griffiths *et al.* 2009b). The Littoral zone was therefore omitted from the scope of this study.

Three main intertidal zones were identified. The intertidal zones were identified to be low, in the middle and high along the shore referred to as the low tide zone (LT), mid tide zone (MT) and high tide (HT) zone. The transition zone of the sublittoral fringe and lower Balanoid zone is subjective and difficult to distinguish. The low tide zone was therefore identified to form part of the sublittoral fringe and lower Balanoid zone. The mid tide zone and the high tide zone were identified to form part of the upper Balanoid zone.

Transects were conducted from the above defined low tide zone to the high tide zone at the seven selected sites. The three sites, namely SP, MB and JB, were located inside the MPA, and the four sites, namely KB, RE, PB and KM, were located outside the MPA. The number of transects per site varied depending on the size of the site. The greatest number of transects at a specific site was nine, and the least was three. A total number of 42 transects were used for this study, of which 20 were inside the MPA, and 22 were outside the MPA. Six intertidal SU's were identified along each transect from the low tide zone to the high tide zone.

SU's of the low tide zone were referred to as low tide pool exposed (LPEx), low tide pool sheltered (LPSh) and low tide rock face (LF). LPEx is continuously exposed to an influx and efflux of water during low tide. It is not demarcated by exposed rock (Fig. 3a). LPSh is sheltered from the influx and efflux of water at low tide. It is demarcated by exposed rock forming isolated rock pools (Fig. 3b). LF is outlined with crevices (Fig. 3c). SU's of the mid tide zone were referred to as mid tide pool (MP) and mid tide rock face (MF). MP is sheltered from the influx and efflux of water at low tide. It is demarcated by exposed rock forming isolated rock pools (Fig. 3d). MF is outlined with crevices (Fig. 3c). The SU of the high tide zone was referred to as high tide pool (HP). HP is sheltered from the influx and efflux of water at low tide. It is demarcated by exposed rock forming isolated rock pools (Fig. 3e).

A single LPEx, LPSh, LF, MP, MF and HP was sampled along each transect. No SU was re-sampled between adjacent transects. Distances between transects were selected so to avoid pseudoreplication of SU's (Fig. 4).

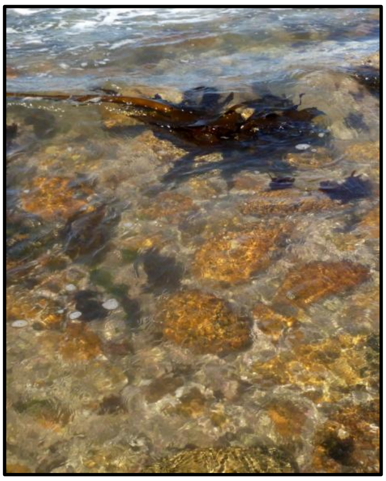

Figure 3a. Low tide pool exposed

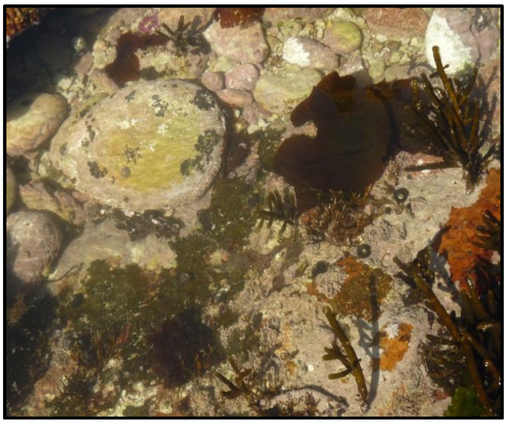

Figure 3b. Low tide pool sheltered

Figure 3c. Rock formation as seen at both low tide rock face and mid tide rock face

Figure 3d. Mid tide pool

Figure 3e. High tide pool

Figure 3. Sampling units included in this study

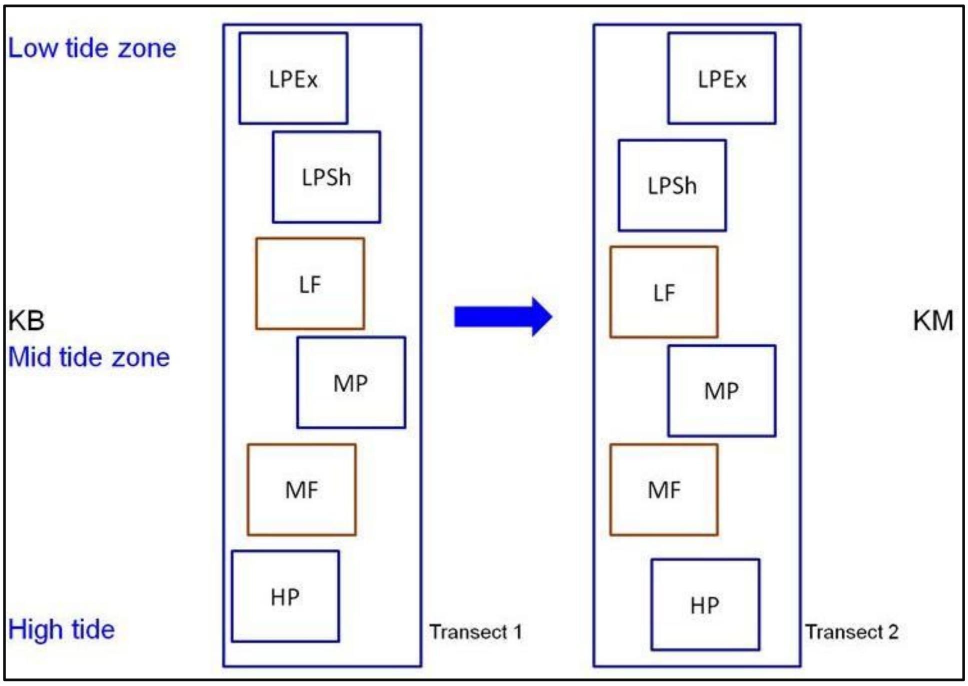

Figure 4. Diagram of two typical transects, with identified sampling units, as conducted from the low tide zone to the high tide zone. (KB → KM: transects conducted along the coastline from the study site namely KB (Kogel Bay) along the coastline to the study site namely KM (Kleinmond). Sampling units: LPEx = Low tide pool exposed, LPSh = Low tide pool sheltered, LF = Low tide rock face, MP = Mid tide pool, MF = Mid tide rock face, HP = High tide pool)

2.6. Species identification

A total of 64 identified focal marine invertebrate species were included in the study (Appendix B). Appendix C gives the identification features of selected species.

Species identification was done during the pilot study in September 2010. Species included in the quantitative study were restricted to the intertidal zone. Species were identified in the field using Branch *et al*. (2010). Where there was uncertainty, one or two individuals of a species were removed and preserved in 75% ethanol for further investigation. Marine alien species present along the coastline of the KBR, accompanied with the Betty's Bay MPA were identified.

2.7. Data collection

Quantitative sampling was done from October 2010 to December 2010. An *in situ* visual count of the focal marine invertebrate species was done at each SU. Three medium sized rocks in each of the selected LPEx, LPSh, MP and HP were randomly overturned to count individuals living under rocks. The counting method for each species was consistent.

2.8. Data analyses

In community ecology, the objective is to understand how species relate to their environment. The approach is to sample the abundance of a number of focal species within an adequate amount of defined sampling units (SU's) (Ter Braak 1986).

Multivariate nested analysis of variance (MANOVA) is used to test whether there are significant differences between populations dependent on multiple characteristics. In the case where only two characteristics are incorporated within the test, two-way nested analysis of variance (ANOVA) may be used. If only one characteristic is incorporated in the test, one-way ANOVA may be used (Townend 2002).

A typical data set as assembled by community ecologists consists of the abundance of focal species at a series of sites, and data of the environmental variables taken at these sites. Sites may be referred to as any defined habitat type such as sampling units, transects and study sites (Ter Braak 1986). Ordination techniques graphically represent the species composition related to different sites. The ecological distances between sites are known as the chi-square distances. These ecological distances are the differences in species composition between sites. Sites close to one another are interpreted to have similar species compositions. Sites further apart from one and other are interpreted to have dissimilar species compositions (Ludwig & Reynolds (1988). Correspondence analysis (CA) is an ordination technique aimed to quantify categorically defined data (Greenacre 1984). Applying CA pose some difficulties. Species which occur rarely in a data set should be removed, as these may disproportionately influence the analysis (Legendre & Gallagher 2001).

Count data with a large number of zeros cannot be transformed. If transformation is possible, statistical assumptions may be violated when applying classical statistics. Nonparametric tests are also not recommended, as these tests make assumptions such as assuming homogeneity of variance. Generalized Linear Models (GLMs) use link functions and are considered the best method to analyse nonnormal data (Bolker *et al.* 2009).

For this study, two-way ANOVA fixed effect test was used to test for significant interaction in mean abundance and mean species richness of marine invertebrates among intertidal zones and protection status (MPA and NMPA). One-way ANOVA was used to test for significant difference in mean abundance and mean species richness of marine invertebrates across intertidal zones, where protection status was not taken into account. Two-way ANOVA fixed effect test was also used to test for significant interaction in mean abundance and mean species richness of marine invertebrates among SU’s and protection status. One-way ANOVA was also used to test for significant difference in mean abundance and mean species richness of marine invertebrates across SU's, where protection status was not taken into account.

The Fisher Least significant (LSD) post hoc test was used to test for significant differences in mean abundance and mean species richness of marine invertebrates between intertidal zones. The Fisher LSD post hoc test was also used to test for significant differences in mean abundance and mean species richness of marine invertebrates between intertidal SU's.

CA was done to investigate the species composition relation to the MPA and NMPA, the study sites, the SU's, and the wave exposures. CA was selected as the variables were classified categorically (Greenacre 2002).

GLMs were used to test for significant difference in mean abundance of *M. galloprovincialis* across the MPA and NMPA, study sites, SU's, and wave exposures. GLMs were also used to test for significant differences in mean abundance of *M. galloprovincialis* between the MPA and NMPA, study sites, SU's, and wave exposures. GLMs were used to test for significant difference in mean abundance of *W. subtorquata* across the MPA and NMPA, study sites, SU's, and wave exposures. GLMs were also used to test for significant differences in mean abundance of *W. subtorquata* between the MPA and NMPA, study sites, SU's, and wave exposures (Zuur *et al.* 2010; O'Hara 2009). These GLMs were calculated with normal distribution (with an identity link functions) for all data, as means were >5, and the minimum number of successes and failures were <5 (Bolker *et al.* 2009). As these analyses showed no overdispersion of the variances compared to the models, Wald χ^2 (Z), statistics were calculated using the penalised quasilikelihood technique (Bolker *et al.* 2009).

2.8.1. Mean abundance and mean species richness of marine invertebrates

2.8.1.1. Intertidal zones and protection status interaction

The normal probability plot of the raw residuals indicated that the data was normally distributed. No transformation of the raw data was required. The abundance and species richness of marine invertebrates for the intertidal zones, referred to as LT, MT and HT were grouped together. Two-way ANOVA fixed effect test was used to test for significant interaction in mean abundance and mean species richness of marine invertebrates among intertidal zones and protection status (MPA and NMPA). One-way ANOVA was used to test for significant difference in mean abundance and mean species richness of marine invertebrates across intertidal zones, where protection status was not taken into account. The Fisher LSD post hoc test was used to test for significant differences in mean abundance and mean species richness of marine invertebrates between intertidal zones. STATISTICA version 10 software package (Statsoft Inc. 2011) was used to run the normal probability plot and the two-way (ANOVA) with the Fisher LSD post hoc test.

2.8.1.2. Sampling units and protection status interaction

The normal probability plot of the raw residuals indicated that the data was normally distributed. No transformation of the raw data was required. The abundance and species richness of marine invertebrates for the SU's referred to as LPEx, LPSh, LF, MP, MF and HP were grouped together. Two-way ANOVA fixed effect test was used to test for significant interaction in mean abundance and mean species richness of marine invertebrates among SU's and protection status (MPA and NMPA). One-way ANOVA was used to test for significant difference in mean abundance and mean species richness of marine invertebrates across SU's, where protection status was not taken into account. The Fisher LSD post hoc test was used to test for significant differences in mean abundance and mean species richness of marine invertebrates between SU's. STATISTICA version 10 software package (Statsoft Inc. 2011) was used to run the normal probability plot and the two-way (ANOVA) with the Fisher LSD post hoc test.

2.8.2. Relationship between species composition and environmental variables

Rarely encountered species were removed from the raw data as these species may disproportionally influence the species composition in relation to the categorised environmental variables (Legendre & Gallagher 2001). Marine invertebrate species with a total count greater than 500 were included in the CA analyses.

CA was done to investigate the species composition relation to the MPA and NMPA, the study sites, the SU's, and the wave exposures. STATISTICA software package (Statsoft Inc. 2011) version 10 was used to run the CA analyses.

2.8.3. Mean abundance of the alien marine invertebrates

Generalized Non-Linear Models (GLMs) were used to test for significant difference in mean abundance of *M. galloprovincialis* across the MPA and NMPA, study sites, SU's, and wave exposures. GLMs were also used to test for significant differences in mean abundance of *M. galloprovincialis* between the MPA and NMPA, study sites, SU's, and wave exposures. GLMs were used to test for significant difference in mean abundance of *W. subtorquata* across the MPA and NMPA, study sites, SU's, and wave exposures. GLMs were also used to test for significant differences in mean abundance of *W. subtorquata* between the MPA and NMPA, study sites, SU's, and wave exposures (Zuur *et al.* 2010; O'Hara 2009). These GLMs were calculated with normal distribution (with an identity link functions) for all data, as means were >5, and the minimum number of successes and failures were <5 (Bolker *et al.* 2009). As these analyses showed no overdispersion of the variances compared to the models, Wald χ^2 (Z) statistics were calculated using the penalised quasilikelihood technique (Bolker *et al.* 2009). SAS software package (SAS Institute Inc. 2006) version 9.1.3 was used to run the analyses.

3. Results

A total of 64 identified focal marine invertebrate species were included in the study (Appendix B). Appendix C gives the identification features of selected species.

The mytilid Mediterranean mussel *Mytilus galloprovincialis* (Fig. 21a) and the bryozoan *Watersipora subtorquata* (Fig. 21b) were the only alien species sampled. Pits visible within the resilial ridges of *M. galloprovincialis* (Fig. 21a) (Branch *et al.* 2010) distinguish this species from the indigenous black mussel *Choromytilus meridionalis.*

3.1. Indicative species

Information of indicative species of the different SU's are as follows:

3.1.1. Low tide pool exposed

The flat rock surfaces, along with loose rocks, were mostly covered with encrusting algae. The winkle *Oxystele sinensis* was present of which a significant proportion was adults. The whelk *Burnupena cincta* was present. The following seaweed species were present: wrack brown algae *Bifurcariopsis capensis*, tongue-like red algae *Gigartina polycarpa*, the green algae *Ulva* spp. and small plants of the kelp *Ecklonia maxima.*

3.1.2. Low tide pool sheltered

The flat rock surfaces, along with loose rocks, were mostly covered with encrusting algae. *O. sinensis* was present with a reduced number of adult species. *B. cincta* was present. The following seaweed species were present: predominantly *B. capensis, G. polycarpa* and *Ulva* spp.

3.1.3. Low tide rock face

The barnacle *Tetraclita serrata* was present. *M. galloprovincialis* was present at wave-exposed sites. The following seaweed species were present: predominantly bladder brown algae *Splachnidium rugosum*, wrack brown algae *Bifurcaria brassicaeformis*, *G. polycarpa* and *Ulva* spp.

3.1.4. Mid tide pool

A significant decrease in encrusting coralline algae covered the flat rock surfaces and loose rocks. *O. sinensis* was reduced or completely absent. The winkle *Oxystele tigrina* was present. The whelk *Burnupena lagenaria* was present and a reduced number of *B. cincta* was present. The following seaweed species were present: predominantly stringy brown algae *Chordariopsis capensis* and *Ulva* spp.

3.1.5. Mid tide rock face

T. serrata was reduced or completely absent. The limpets *Siphonaria serrata* and *Siphonaria concinna* were present. The following seaweed species were present: predominantly flat red algae *Porphyra capensis.*

3.1.6. High tide pool

Encrusting coralline algae was absent. *B. cincta* and *B. lagenaria* were generally absent, but the whelk *Burnupena catarrhacta* was present. The following seaweed species were present: predominantly *Ulva* spp.

3.2. Mean abundance and mean species richness of marine invertebrates

3.2.1. Intertidal zones and protection status interaction

3.2.1.1. Mean abundance

Two-way ANOVA indicated that there was no significant interaction in mean abundance of marine invertebrates among intertidal zones and protection status (F=3.00, p>0.05). However, there was a significant difference in mean abundance of marine invertebrates across intertidal zones, when protection status was not taken into account (F=12.50, p≤0.001).

The Fisher LSD post hoc test showed that the mean abundance of marine invertebrates of LT was significantly higher than MT (p≤0.01), and HT (p≤0.001). The test also showed that the mean abundance of marine invertebrates of MT was significantly higher than HT (p≤0.01) (Fig. 5).

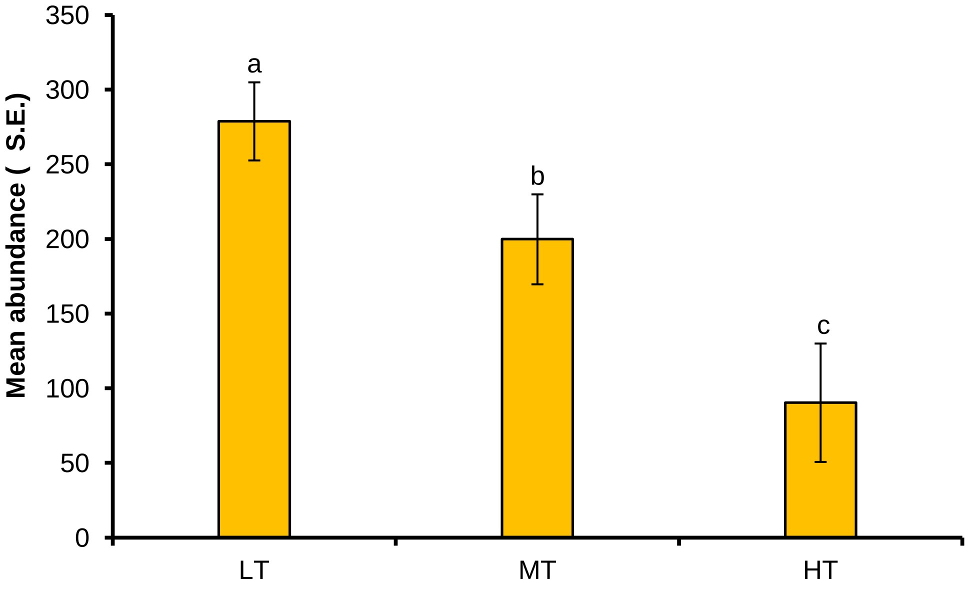

Figure 5. Mean abundance of marine invertebrates of the intertidal zones. (LT=low tide zone, MT=mid tide zone, HT=high tide zone)

3.2.1.2. Mean species richness

Two-way ANOVA indicated that there was no significant interaction in mean species richness of marine invertebrates among intertidal zones and protection status (F=0.05, $p>0.05$). However, there was a significant difference in mean species richness of marine invertebrates across intertidal zones, when protection status was not taken into account (F=14.50, $p\leq0.01$).

The Fisher LSD post hoc test showed that the mean species richness of marine invertebrates of LT was significantly higher than MT ($p\leq0.01$), and HT ($p\leq0.001$). The test also showed that the mean species richness of marine invertebrates of MT was significantly higher than HT ($p\leq0.01$) (Fig. 6).

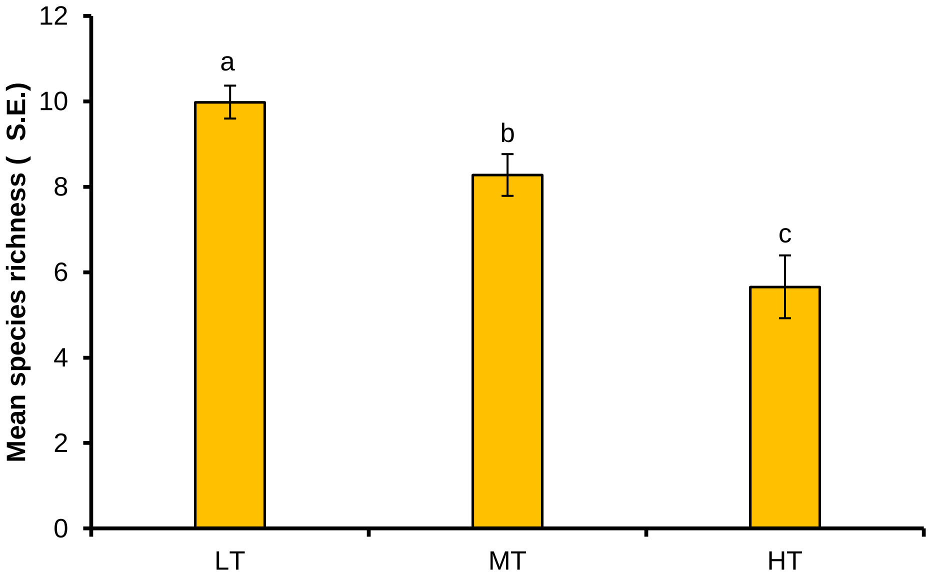

Figure 6. Mean species richness of marine invertebrates of the intertidal zones. (LT=low tide zone, MT=mid tide zone, HT=high tide zone)

3.2.2. Sampling units and protection status interaction

3.2.2.1. Mean abundance

Two-way ANOVA indicated that there was a significant interaction in mean abundance of marine invertebrates among SU's and protection status ($F=9.84$, $p \leq 0.001$). There was also a significant difference in mean abundance of marine invertebrates across SU's, when protection status was not taken into account ($F=22.14$, $p \leq 0.001$).

The Fisher LSD post hoc test showed that the mean abundance of marine invertebrates of NMPA LF was significantly higher than MPA LPEx ($p \leq 0.001$), LPSh ($p \leq 0.001$), LF ($p \leq 0.001$), MP ($p \leq 0.001$), MF ($p \leq 0.001$), HP ($p \leq 0.001$), and NMPA LPEx ($p \leq 0.001$), LPSh ($p \leq 0.001$), MP ($p \leq 0.001$), MF ($p \leq 0.001$) and HP ($p \leq 0.001$). The test also showed that the mean abundance of marine invertebrates of MPA HP was significantly lower than MPA LPEx ($p \leq 0.05$), MPA LF ($p \leq 0.001$), MPA MP ($p \leq 0.01$) and MPA MF ($p \leq 0.05$) (Fig. 7).

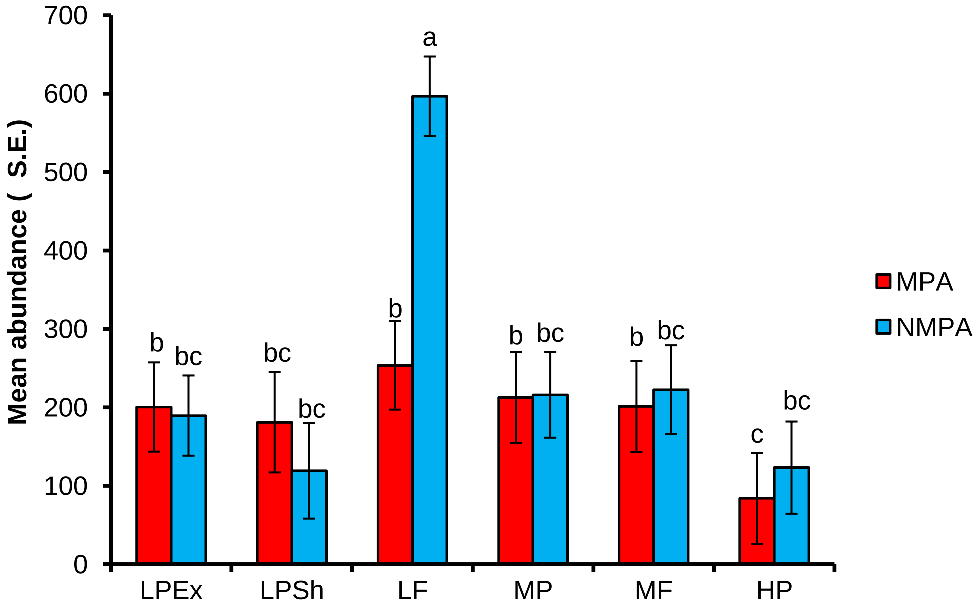

Figure 7. Mean abundance of marine invertebrates of the sampling units (SU's) (MPA=marine protected area, NMPA=non-MPA, LPEx=low tide pool exposed, LPSh=low tide pool sheltered, LF=low tide rock face, MP=mid tide pool, MF=mid tide rock face, HP=high tide pool)

3.2.2.2. Mean species richness

Two-way ANOVA indicated that there was no significant interaction in mean species richness of marine invertebrates among SU's and protection status (F=0.61, p>0.05). However, there was a significant difference in mean species richness of marine invertebrates across SU's, when protection status was not taken into account (F=94.3, p≤0.05).

The Fisher LSD post hoc test showed that the mean species richness of marine invertebrates of LPEx was significantly higher than LF (p≤0.001), MF (p≤0.001), and HP (p≤0.001), and that the mean species richness of marine invertebrates of LPSh was significantly higher than LF (p≤0.001), MF (p≤0.001), and HP (p≤0.001), and also that the mean species richness of marine invertebrates of MP was significantly higher than LF (p≤0.001), MP (p≤0.001), and HP (p≤0.001). The test also showed that the mean species richness of marine invertebrates of LF is significantly higher than MF (p≤0.001), and that the mean species richness of marine invertebrates of HP is significantly higher than MF (p≤0.05) (Fig. 8).

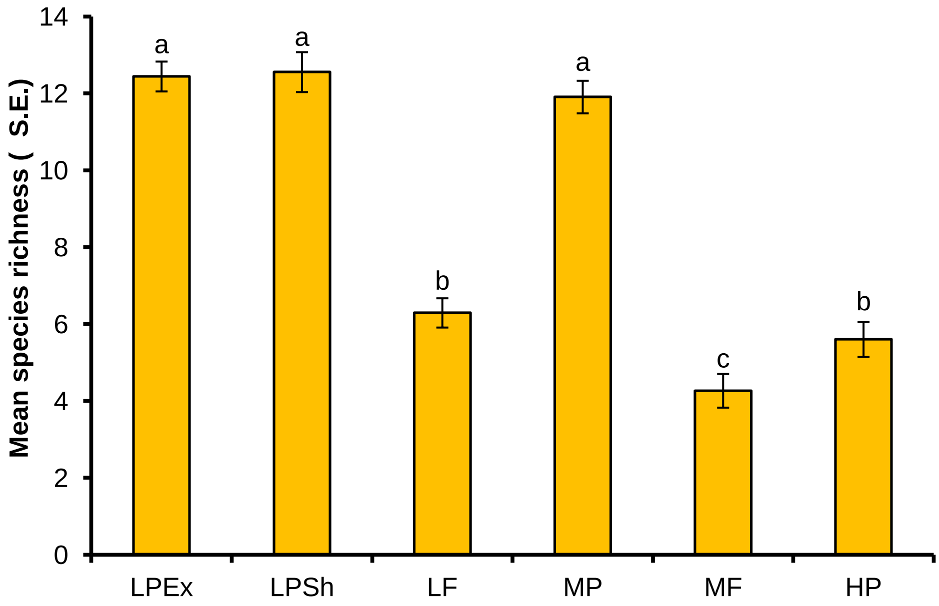

Figure 8. Mean species richness of marine invertebrates of the sampling units (SU's). (LPEx=low tide pool exposed, LPSh=low tide pool sheltered, LF=low tide rock face, MP=mid tide pool, MF=mid tide rock face, HP=high tide pool)

3.3. Relationship between species composition and environmental variables

Species that were included in the CA ordination and their abbreviations are given in Table 1.

Table 1: Species included in the CA ordination, and their abbreviations.

Phylum	Class	Species	Species abbreviation
Athropoda	**Cirripedia**	*Tetraclita serrata*	*T.se*
		Octomeris angulosa	*O.an*
Bryozoa		*Watersipora subtorquata*	*W.su*
Mollusca	**Bivalvia**	*Mytilus galloprovincialis*	*M.ga*
	Gastropoda	*Scutellastra longicosta*	*S.lo*
		Cymbula oculus	*C.oc*
		Scutellastra granularis	*S.gr*
		Helcion pectunculus	*H.pe*
		Helcion pruinosus	*H.pr*
		Siphonaria serrata	*S.se*
		Oxystele variegata	*O.va*
		Oxystele impervia	*O.im*
		Oxystele sinensis	*O.si*
		Oxystele tigrina	*O.ti*
Echinoderms	**Asteroidea**	*Parvulastra exigua*	*P.ex*
	Echinoidea	*Parechinus angulosus*	*P.an*

3.3.1. Marine protected area and non-marine protected area

CA ordination indicated that the alien species *M. galloprovincialis* was strongly represented in the NMPA, while the alien species *W. subtorquata* was strongly represented in the MPA. *T. serrata*, *O. angulosa* and *P. angulosus* were represented within the NMPA. *O. variegata* and *O. impervia* were represented within the MPA. *S. longicosta*, *C. oculus*, *S. granularis*, *H. pectunculus*, *H. pruinosus*, *S. serrata*, *O. sinensis*, *O. tigrina* and *P. exigua* were represented in both the MPA and NMPA (Fig. 9).

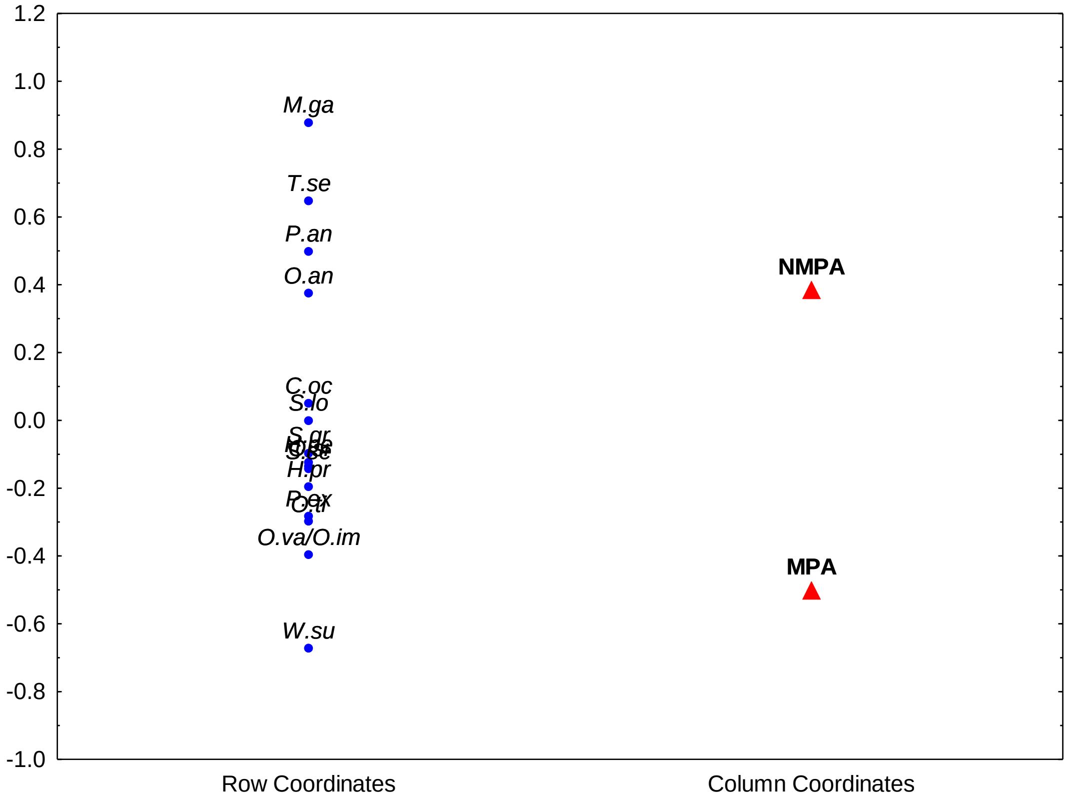

Figure 9. One dimensional plot of the row coordinates (species coordinates) and the column coordinates (MPA and NMPA). (MPA=marine protected area, NMPA=non-MPA, *T.se=Tetraclita serrata, O.an=Octomeris angulosa, W.su=Watersipora subtorquata, M.ga=Mytilus galloprovincialis, S.lo=Scutellastra longicosta, C.oc=Cymbula oculus, S.gr=Scutellastra granularis, H.pe=Helcion pectunculus, H.pr=Helcion pruinosus, S.se=Siphonaria serrata, O.va/O.im=Oxystele variegate/Oxystele impervia, O.si=Oxystele sinensis, O.ti=Oxystele tigrina, P.ex=Parvulastra exigua, P.an=Parechinus angulosus*)

3.3.2. Study sites

CA ordination indicated that the alien species *M. galloprovincialis* was strongly represented at KB and at KM. The alien species *W. subtorquata* was strongly represented at SP and MB, and to a lesser extent, at JB. *T. serrata* was represented at RE, and to a lesser extent, at KB. *O. angulosa* was represented at KB and *P. angulosus* was represented at KM. *O. variegata*, *O. impervia*, *H. pectunculus*, *O. sinensis* and *S. longicosta* were represented at SP, MB and JB. *H. pruinosus* and *O. tigrina* were represented at SP, and to a lesser extent at PR and RE. *C. oculus*, *S. granularis* and *P. exigua* were represented at JB and KM. *S. serrata* was represented at PR (Fig. 10).

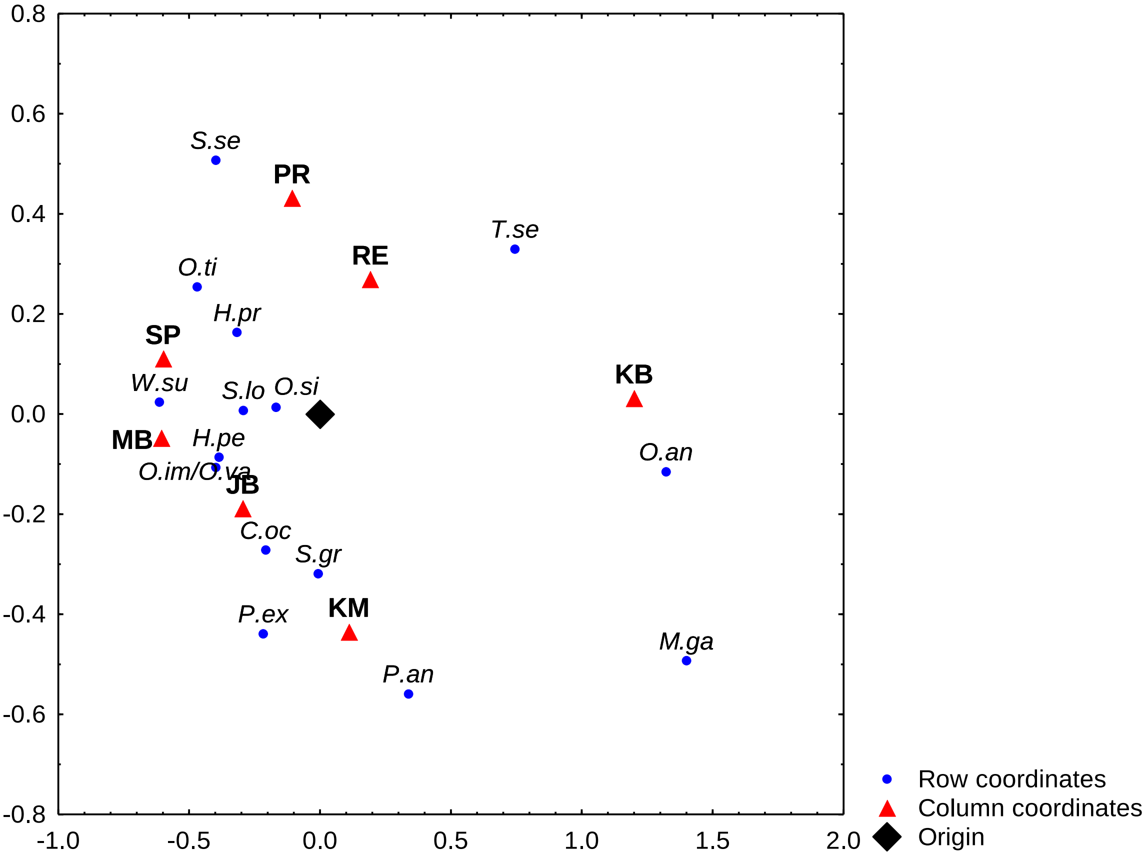

Figure 10. Two dimensional plot of the row coordinates (species coordinates) and the column coordinates (study site coordinates). (KB=Kogel Bay, RE=Rooi Els, PR=Pringle Bay, SP=Stoney Point, MB=Betty's Bay Main Beach, JB=Jock's Bay, KM=Kleinmond, *T.se=Tetraclita serrata, O.an=Octomeris angulosa, W.su=Watersipora subtorquata, M.ga=Mytilus galloprovincialis, S.lo=Scutellastra longicosta, C.oc=Cymbula oculus, S.gr=Scutellastra granularis, H.pe=Helcion pectunculus, H.pr=Helcion pruinosus, S.se=Siphonaria serrata, O.va/O.im=Oxystele variegate/Oxystele impervia, O.si=Oxystele sinensis, O.ti=Oxystele tigrina, P.ex=Parvulastra exigua, P.an=Parechinus angulosus*)

3.3.3. Sampling units

CA ordination indicated that the alien species *M. galloprovincialis* was strongly represented at LF and MF. The alien species *W. subtorquata* was strongly represented at LPEx and LPSh. *T. serrata*, *O. angulosa*, *S. granularis*, *H. pectunculus*, *S. serrata*, *O. variegata* and *O. impervia* were represented at LF and MF. *C. oculus* was represented at MF and LF, and to lesser extent at LPEx and LPSh. *S. longicosta* and *O. sinensis* were represented at LPEx and LPSh. *P. angulosus* was represented at LPSh and *P. exigua* was represented at MP. *H. pruinosus* was represented at both LPSh and MP. *O. tigrina* was represented at both MP and HP (Fig. 11).

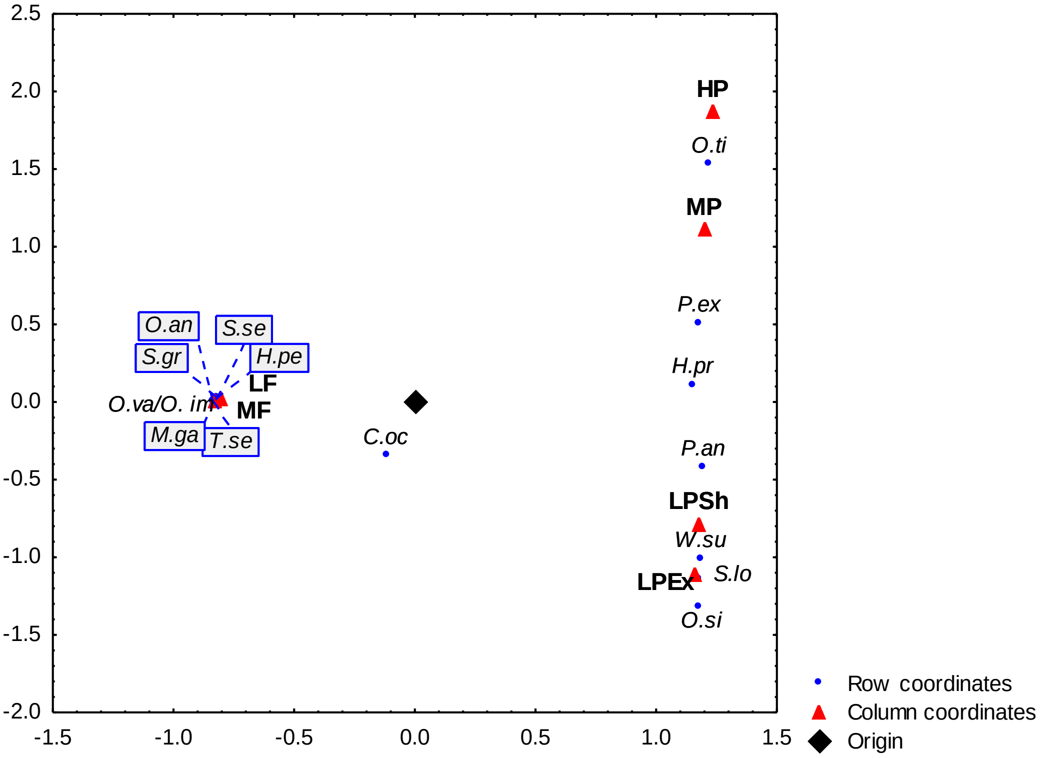

Figure 11. Two dimensional plot of the row coordinates (species coordinates) and the column coordinates (sampling unit coordinates). (LPEx=low tide pool exposed, LPSh=low tide pool sheltered, LF=low tide rock face, MP=mid tide pool, MF=mid tide rock face, HP=high tide pool, *T.se=Tetraclita serrata, O.an=Octomeris angulosa, W.su=Watersipora subtorquata, M.ga=Mytilus galloprovincialis, S.lo=Scutellastra longicosta, C.oc=Cymbula oculus, S.gr=Scutellastra granularis, H.pe=Helcion pectunculus, H.pr=Helcion pruinosus, S.se=Siphonaria serrata, O.va/O.im=Oxystele variegate/Oxystele impervia, O.si=Oxystele sinensis, O.ti=Oxystele tigrina, P.ex=Parvulastra exigua, P.an=Parechinus angulosus*)

3.3.4. Wave exposure

CA ordination indicated that the alien species *M. galloprovincialis* was strongly represented at exposed sites, while the alien species *W. subtorquata* was strongly represented at sheltered sites. *T. serrata* and *O. angulosa* were represented at exposed sites. *P. angulosus*, *O. variegata*, *O. impervia*, *C.oculus*, *S. granularis* and *P.exigua* were represented at partially exposed sites. *H. pruinosus* and *O. sinensis* were represented at sheltered sites, and to a much lesser extent at partially exposed sites. *S. longicosta*, *H. pectunculus* were represented at partially exposed and sheltered sites. *S. serrata* and *O. tigrina* were represented at sheltered sites (Fig. 12).

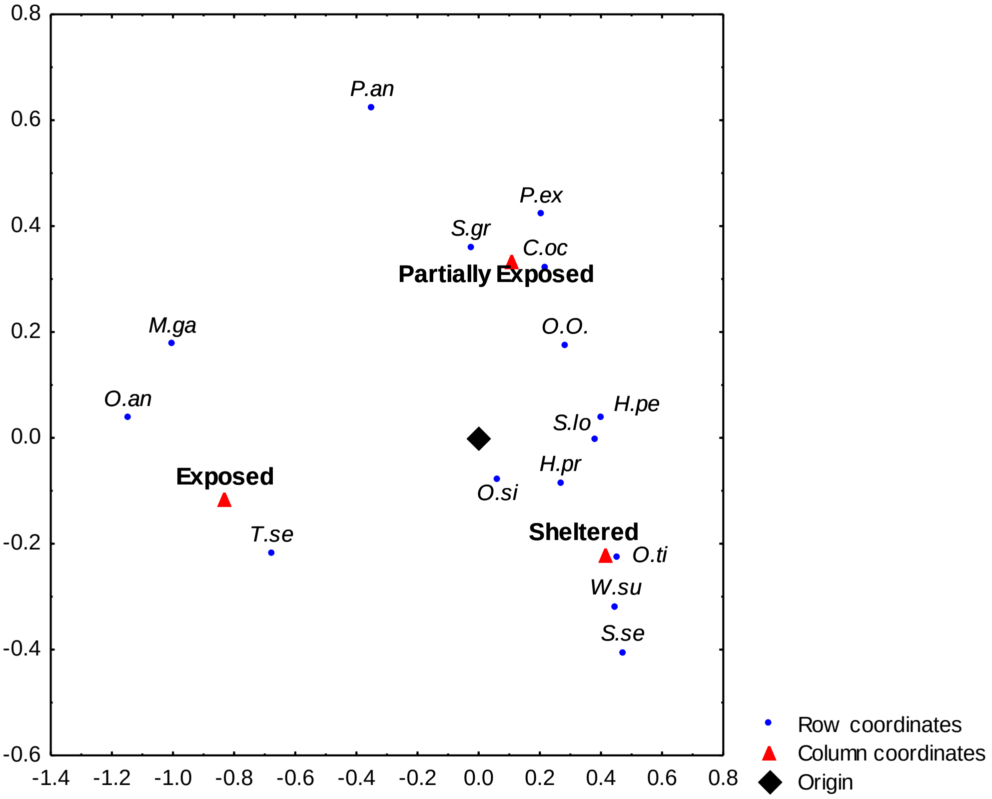

Figure 12. Two dimensional plot of the row coordinates (species coordinates) and the column coordinates (wave exposure coordinates). (*T.se=Tetraclita serrata, O.an=Octomeris angulosa, W.su=Watersipora subtorquata, M.ga=Mytilus galloprovincialis, S.lo=Scutellastra longicosta, C.oc=Cymbula oculus, S.gr=Scutellastra granularis, H.pe=Helcion pectunculus, H.pr=Helcion pruinosus, S.se=Siphonaria serrata, O.va/O.im=Oxystele variegate/Oxystele impervia, O.si=Oxystele sinensis, O.ti=Oxystele tigrina, P.ex=Parvulastra exigua, P.an=Parechinus angulosus*)

3.4. Mean abundance of the alien marine invertebrates

3.4.1. Mean abundance of Mytilus galloprovincialis

3.4.1.1. Marine protected area and non-marine protected area

Wald χ^2 test indicated that there was a significant difference in mean abundance of *M. galloprovincialis* between the MPA and NMPA. Mean abundance of *M. galloprovincialis* was significantly higher in the NMPA (df=1, Z=4.38, $p \leq 0.05$) (Fig. 13).

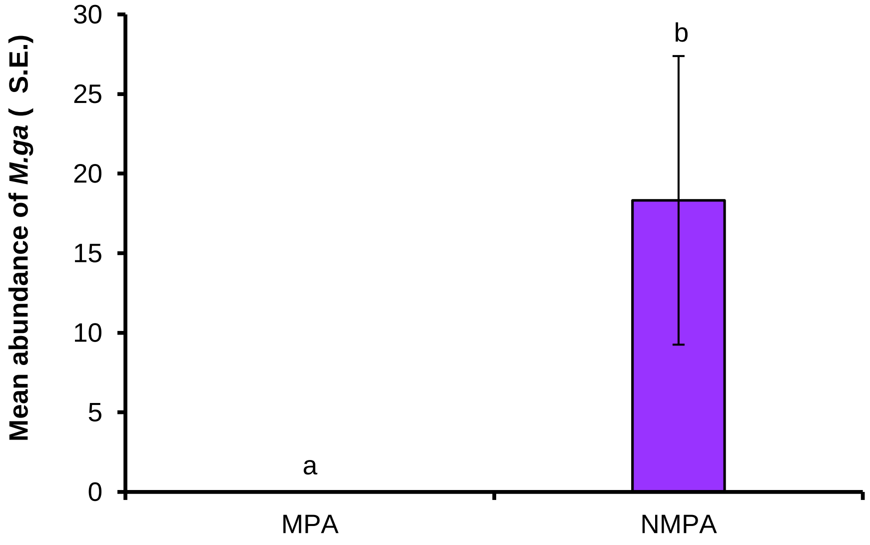

Figure 13. Mean abundance of *Mytilus galloprovincialis* (*M.ga*) between the MPA and NMPA. (MPA=marine protected area, NMPA=non-MPA)

3.4.1.2. Study sites

Wald χ^2 test indicated that there was a significant difference in mean abundance of *M. galloprovincialis* across the study sites (df=6, Z=25.13, p≤0.001). Mean abundance of *M. galloprovincialis* was significantly higher at KB than RE (df=1, Z=12.62, p≤0.001), PR (df=1, Z=18.72, p≤0.001), SP (df=1, Z=19.15, p≤0.001), MB (df=1, Z=16.55, p≤0.001), JB (df=1, Z=19.28, p≤0.001), and KM (df=1, Z=10.71, p≤0.01) (Fig. 14).

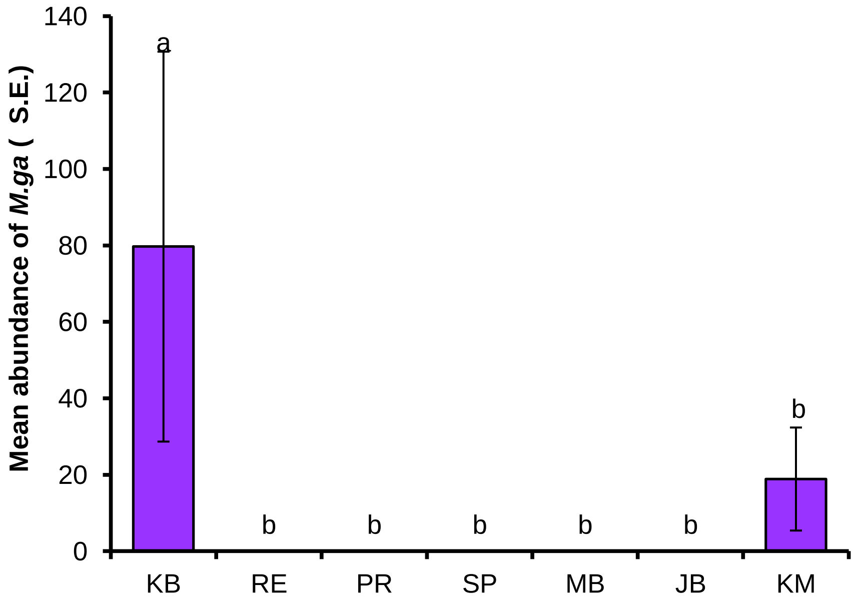

Figure 14. Mean abundance of *Mytilus galloprovincialis* (*M.ga*) between the study sites. (KB=Kogel Bay, RE=Rooi Els, PR=Pringle Bay, SP=Stoney Point, MB=Betty's Bay Main Beach, JB=Jock's Bay, KM=Kleinmond)

3.4.1.3. Sampling units

Wald χ^2 test indicated that there was a significant difference in mean abundance of *M. galloprovincialis* across the SU's (df=5, Z=16, $p \leq 0.001$). Mean abundance of *M. galloprovincialis* was significantly higher at LF than LPEx (df=1, Z=9.93, $p \leq 0.01$), LPSh (df=1, Z=6.79, $p \leq 0.01$), MP (df=1, Z=8.96, $p \leq 0.01$), MF (df=1, Z=8.65, $p \leq 0.01$), and HP (df=1, Z=8.32, $p \leq 0.01$) (Fig. 15).

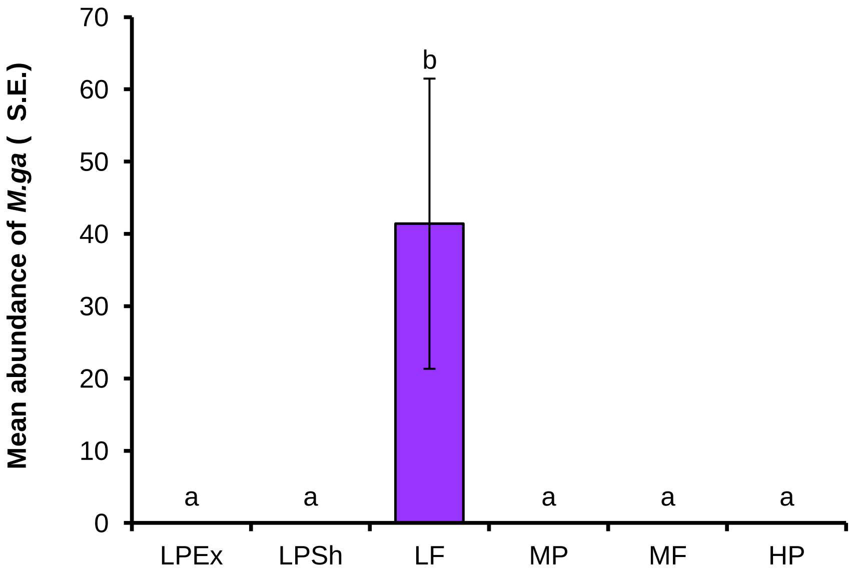

Figure 15. Mean abundance of *Mytilus galloprovincialis* (*M.ga*) between the sampling units (SU's). (LPEx=low tide pool exposed, LPSh=low tide pool sheltered, LF=low tide rock face, MP=mid tide pool, MF=mid tide rock face, HP=high tide pool)

3.4.1.4. Wave exposure

Wald χ^2 test indicated that there was a significant difference in mean abundance of *M. galloprovincialis* across the different wave exposures (df=2, Z=9.90, $p \leq 0.001$). Mean abundance of *M. galloprovincialis* was significantly higher at the exposed sites than the partially exposed sites (df=1, Z=5.75, $p \leq 0.05$), and the sheltered sites (df=1, Z=9.90, $p \leq 0.01$) (Fig. 16).

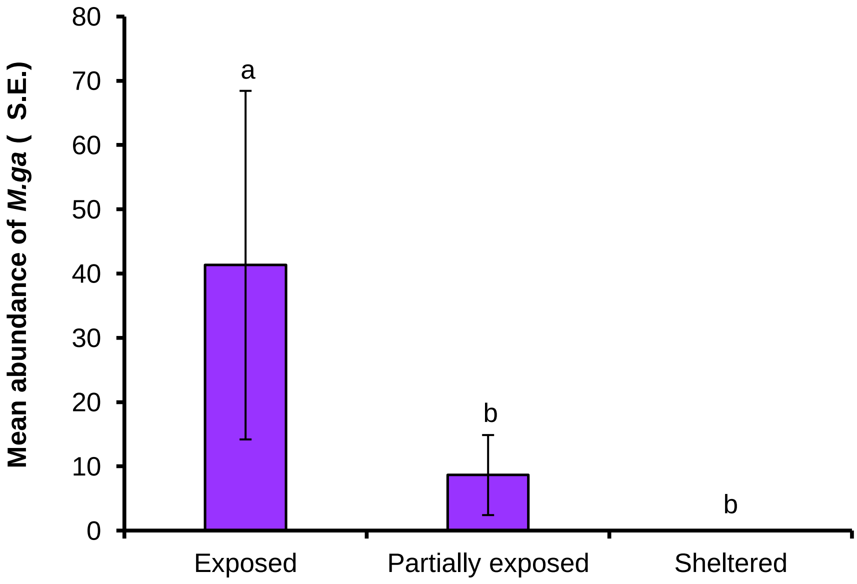

Figure 16. Mean abundance of *Mytilus galloprovincialis* (*M.ga*) between the different wave exposures.

3.4.2. Mean abundance of Watersipora subtorquata

3.4.2.1. Marine protected area and non-marine protected area

Wald χ^2 test indicated that there was a significant difference in mean abundance of *W. subtorquata* between the MPA and NMPA. Mean abundance of *W. subtorquata* was significantly higher in the MPA (df=1, Z= 5.97, $p \leq 0.05$) (Fig. 17).

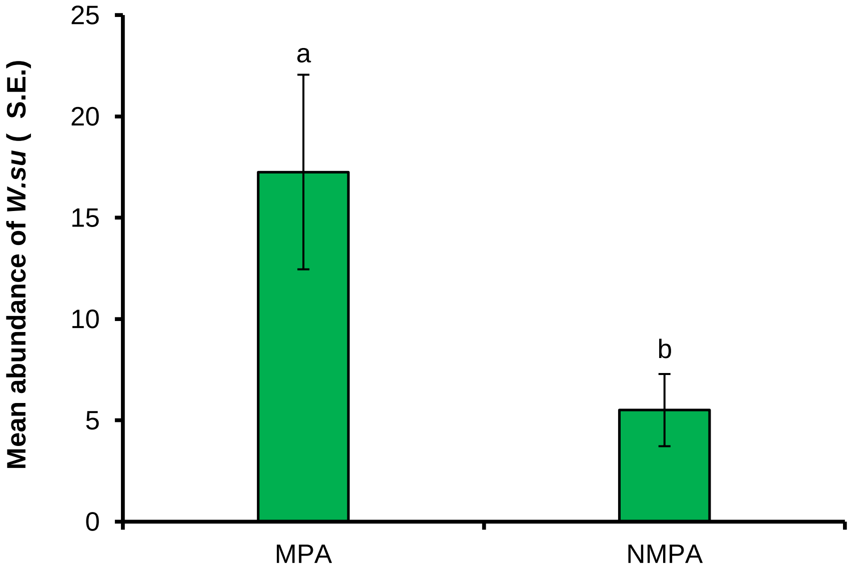

Figure 17. Mean abundance of *Watersipora subtorquata* (*W.su*) between the MPA and NMPA. (MPA=marine protected area, NMPA=non-MPA)

3.4.2.2. Study sites

Wald χ^2 test indicated that there was no significant difference in mean abundance of *W. subtorquata* across the study sites (df=6, Z=8.70, $p>0.05$). However, the mean abundance of *W. subtorquata* was significantly higher at MB than PR (df=1, Z=5.41, $p\leq0.05$), and KM (df=1, Z=5.23, $p\leq0.05$) (Fig. 18).

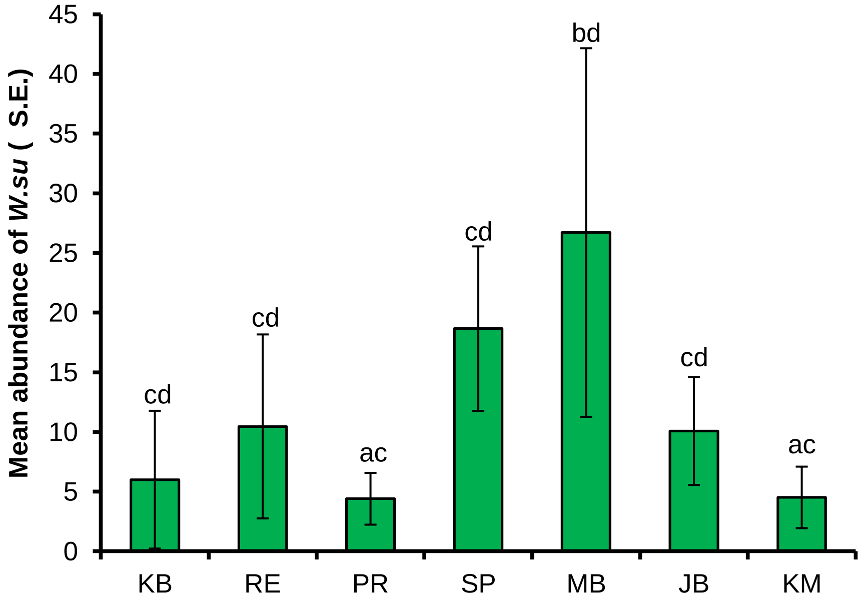

Figure 18. Mean abundance of *Watersipora subtorquata* (*W.su*) between the study sites. (KB=Kogel Bay, RE=Rooi Els, PR=Pringle Bay, SP=Stoney Point, MB=Betty's Bay Main Beach, JB=Jocks's Bay, KM=Kleinmond)

3.4.2.3. Sampling units

Wald χ^2 test indicated that there was a significant difference in mean abundance of *W. subtorquata* across the SU's (df=5, Z=39.33, p≤0.001). Mean abundance of *W. subtorquata* was significantly higher at LPEx than LPSh (df=1, Z=4.30, p≤0.05), LF (df=1, Z=26.97, p≤0.001), MP (df=1, Z=13.94, p≤0.001), MF (df=1, Z=22.9, p≤0.001), and HP (df=1, Z=22.13, p≤0.001). Mean abundance of *W. subtorquata* was also significantly higher at LPSh than LF (df=1, Z=4.85, p≤0.05), MF (df=1, Z=4.33, p≤0.05), and HP (df=1, Z=4.22, p≤0.05) (Fig. 19).

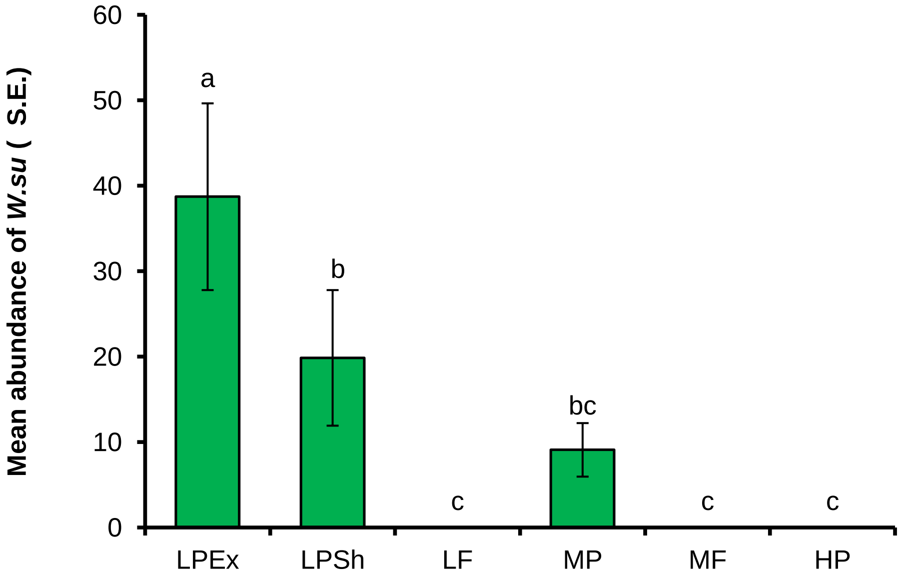

Figure 19. Mean abundance of *Watersipora subtorquata* (*W.su*) between the sampling units (SU's). (LPEx=low tide pool exposed, LPSh=low tide pool sheltered, LF=low tide rock face, MP=mid tide pool, MF=mid tide rock face, HP=high tide pool)

3.4.2.4. Wave exposure

Wald χ^2 test indicated that there was no significant difference in mean abundance of *W. subtorquata* across the different wave exposures (df=2, Z=2.21, $p>0.05$) (Fig. 20).

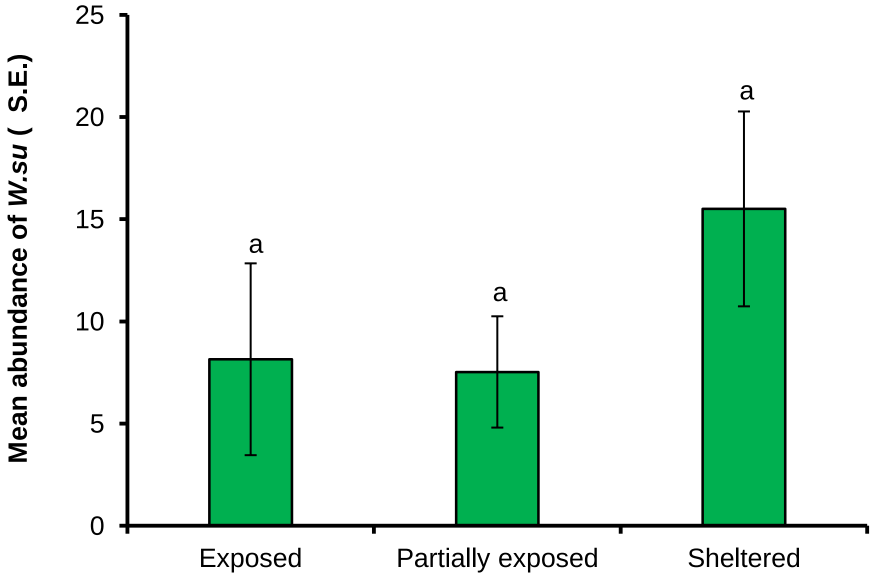

Figure 20. Mean abundance of *Watersipora subtorquata* (*W.su*) between the different wave exposures.

4. Discussion

4.1. Community structure and gradient stress

Abundance and species richness along rocky shores reflect that of biotic and abiotic disturbances (Garrity 1984; McQuaid & Branch 1984; Dayton 1971). Rocky shore community structuring is a result of many contributing factors. These include thermal stress, desiccation, light intensities, wave exposure, inter-specific competition for space and food, and predation (Harley 2003; Harley & Helmuth 2003; Dahlhoff *et al.* 2001; Garrity 1984; Branch 1981; Dayton 1971; Evans 1948). Cold water from the ocean flushes into tidal pools as the tide rises. An increase in water temperature from low tide pools across the vertical gradient to high tide pools is a result of an increase in air exposure time (Stephenson & Stephenson 1949). Water temperature and salinity is important for determining the structure of local tidal pool communities (Harley 2003; Harley & Helmuth 2003; Evans 1948). Competition for space is an important limiting factor for community structuring of marine sessile organisms (Dayton 1971). Wave exposure results in horizontal and vertical gradients of physical stress (Harley & Helmuth 2003; McQuaid & Branch 1984; Dayton 1971). Pounding from high wave exposure may be stressful for some species. High water flow may however, prevent desiccation of species inhabiting exposed rock. McQuaid & Branch (1984) found that the horizontal gradient between sheltered and wave-exposed sites had the most profound effect on species biomass.

It was evident that a hierarchy exists along the vertical and horizontal gradients of the Kogelberg Biosphere Reserve’s rocky shores. The significant decrease in species richness from the low tide to the high tide zones most likely reflected an increase in exposure time and water temperature (Fig. 6). The significant decrease in species richness from the low tide and mid tide pools to the high tide pools most likely reflected an increase in water temperature (Fig. 8). Long exposure may result in desiccation and an increase in water temperature results in thermal stress (Harley 2003; Harley & Helmuth 2003; Evans 1948).

Few species are adapted to the extreme stress of inhabiting exposed rock (Garrity 1984). Water spray may also protect vulnerable species from desiccation. This most likely reflected the significant decrease in species richness along the mid and low tide rock face. The significantly higher species richness at the low tide rock face compared to the mid tide rock face most likely reflected the shorter exposure time low on the shore (Fig. 8).

The significant decrease in abundance from the low tide to the high tide zones most likely reflected an increase in thermal stress and desiccation (Fig. 5). In turn, the significant decrease in abundance from the low tide to the high tide pools most likely reflected increasing thermal stress (Fig. 7).

The significant difference in abundance between the low tide rock face within the marine protected area (MPA) and the low tide rock face outside the MPA was an important finding (Fig. 7), arising from the significantly higher abundance of the mytilid mussel *Mytilus galloprovincialis* along the low tide rock face (Fig. 15). *M. galloprovincialis* was also well-represented along the mid tide rock face (Fig. 11). However this finding may be rejected as there was no records of this species along the mid tide region (Fig. 15). The abundance of *M. galloprovincialis* was also significantly higher outside the MPA (Fig. 13).

Lethal temperature studies showed that the winkle *Oxystele sinensis* scarcely reaches mid tide levels (Evans 1948). This was reflected from its presence in the low tide pools and its general absence in the mid tide pools. This species was well-represented in low tide pools (Fig. 11).

The whelk *Burnupena cincta* scavenges low on the shore and in subtidal areas. The whelk *Burnupena lagenaria* scavenges intertidally and in shallow subtidal regions (Branch *et al.* 2010). This was reflected from the substantial reduction in numbers of *B. cincta* from the low tide pools to the mid tide pools. It was also reflected from the presence of *B. lagenaria* in the mid tide pools.

The starfish *Parvulastra exigua* inhabits tidal pools so as to avoid desiccation (Branch 1981). Low tide pools are subjected to a higher wave disturbance, which was reflected here by preference of this species for mid tide pools and sheltered low tide pools (Fig. 11).

H. pectunculus was well-represented along the low tide rock face and mid tide rock face. This may be from this species' increased heat loss ability as it is outlined with ridges. The limpets *Scutellastra granularis* and *Siphonaria serrata* species are also outlined with ridges. The latter most likely reflected the distribution *S. granularis* and *S. serrata* along the mid tide rock face and low tide rock face (Fig. 11).

The barnacles *Octomeris angulosa* and *Tetraclita serrata* are filter feeders, preferring higher wave energy (Branch 1981). This reflected the distribution of these species at the wave-exposed sites (Fig. 12). *O. angulosa* was well-represented at the wave-exposed site Kogel Bay. *T. serrata* was well-represented at both the wave-exposed sites Kogel Bay and Rooi Els (Fig. 10). Kogel Bay and Rooi Els are situated outside the MPA. Both these species were well-represented outside the MPA (Fig. 9).

Seaweed species plays an intrinsic role in the food web. Seaweed species photosynthesise and are primary producers. These species are the basis of the food web (Branch 1981). Seaweed species are adapted to different environments. They accordingly structure themselves along gradients of physical stress.

Small plants of the kelp *Ecklonia maxima* inhabit shallow waters. Wrack brown algae *Bifurcariopsis capensis* mainly inhabit low tide pools (Branch *et al.* 2010). *E. maxima* and *B. capensis* release heptera. It ensures that these species firmly attach themselves to their substratum (Branch 1981). This prevents dislodging, particularly in areas with strong wave exposure. The latter reflected the noticeable presence of these species in low tide pools.

Tongue-like red algae *Gigartina polycarpa*, bladder brown algae *Splachnidium rugosum*, and wrack brown algae *Bifurcaria brassicaeformis* grow low on the shore (Branch *et al.* 2010). This reflected the noticeable presence of *G. polycarpa* in low tide pools and along the low tide rock face. It also reflected the noticeable presence of *S. rugosum* and *B. brassicaeformis* along the low tide rock face.

Stringy brown algae *Chordariopsis capensis* prefer mid tide pools. Flat red algae *Porphyra capensis* are tolerant to desiccation. *P. capensis* grows along mid shore and high shore regions (Branch *et al.* 2010). This reflected the noticeable presence of *P. capensis* along the mid tide rock face.

Different green algae *Ulva* spp. grow along different zones of the shore. *U. rigida* grows low on the shore in both tidal pools and along the rock face. *U. intestinalis* grows in mid tide and high tide pools. *U. fasciata* grows in high tide pools (Branch *et al.* 2010). This reflected the distribution of *Ulva* spp. inhabiting the low, mid and high tide zones. It also reflected the presence of *Ulva* spp. in nearly all the habitat types. *Ulva* spp. have large surface areas for photosynthesis. Photosynthesis cannot occur when seaweed species are desiccated. This most likely reflected their preference of inhabiting tidal pools. *Ulva* spp. were more prevalent in mid tide pools and high tide pools as these species have a tolerance to salinity fluctuations and increased water temperatures (Branch 1981).

4.2. Marine alien species distribution and abundance

4.2.1. *Mytilus galloprovincialis*

Wave exposure has a profound effect on the distribution and abundance of *M. galloprovincialis*. This was found to be the case along the coastline of the Kogelberg Biosphere Reserve. *M. galloprovincialis* is a filter feeder preferring wave-exposed sites (Steffani & Branch 2003). *M. galloprovincialis* was strongly represented at the wave-exposed sites (Fig. 12). This was complemented by the significantly higher abundance of this species at the wave-exposed sites (Fig. 16).

M. galloprovincialis flourishes low along the shoreline (Branch *et al.* 2010). Low on the shore, *M. galloprovincialis* is subjected to a higher water flow necessary for feeding. Its resistance to desiccation (Robinson *et al.* 2005; Robinson *et al.* 2004) reflects its prevalence along exposed rock face. In this study, it was evident that *M. galloprovincialis* was well-represented along exposed rock face low on the shore (Fig. 11). This was complemented by the significantly higher abundance of this species along the low tide rock face (Fig. 15). *M. galloprovincialis* was also well-represented along the mid tide rock face

(Fig. 11). However this finding may be rejected as there was no records of this species along the mid tide region (Fig. 15)

Classification of the study sites according to wave exposure indicated that the sites outside the MPA are mostly exposed. Pringle Bay, outside the MPA, was the only site classified as sheltered. The study site Kogel Bay, situated outside the MPA, was classified as wave-exposed. This study indicated that *M. galloprovincialis* was well-represented at Kogel Bay and Kleinmond (Fig. 10). It also indicated that this species was well-represented outside the MPA (Fig. 9). The latter is complemented by the finding that the abundance of *M. galloprovincialis* was significantly higher at Kogel Bay (Fig. 14). The abundance of this species was also significantly higher outside the MPA (Fig. 13). It is evident that the Kogelberg Biosphere Reserve coastline outside the MPA was found here to be significantly more invaded with *M. galloprovincialis*. These findings may also reflect the definite peak in abundance along the low tide rock face outside the MPA (Fig. 7).

The distribution and abundance of *M. galloprovicialis* along the coastline of the Kogelberg Biosphere Reserve may be ascribed to the horizontal stress gradient that exists from wave exposures (Harley & Helmuth 2003; McQuaid & Branch 1984; Dayton 1971). Although this species is resistant to desiccation (Robinson et al. 2005; Robinson et al. 2004), sufficient water flow is required for feeding (Steffani & Branch 2003). The sheltered nature of the MPA was evidently not beneficial for abundant growth of *M. galloprovincialis*. The rocky shore sites outside the MPA were subjected to higher wave exposure, which created favourable conditions for the recruitment and flourishing of *M. galloprovincialis.*

4.2.2. *Watersipora subtorquata*

The red-rust bryozoan species *Watersipora subtorquata* is alien to South African waters (Branch et al. 2010), and is considered to have a cosmopolitan distribution (Clark & Johnston 2009). Its tolerance to antifouling biocides creates a suitable substratum for the aggregation, facilitation and possible introduction of species within a new region (Floerl & Inglis 2005; Floerl *et al.* 2004). *W. subtorquata* has not been found to be harmful to the South African coast.

W. subtorquata is a colony forming species. Environments of low physical stress favour the survival of such species. Sites with high wave exposure may constantly overturn and bury loose rocks and other substratum, which decreases available space for colony forming species to recruit and survive (Jackson 1977). Experiments conducted by (Clark & Johnston 2009) conclude that *W. subtorquata* has a fast growth rate on available open space. It appears that the sheltered nature of the MPA may be beneficial for the settlement and growth of this species.

Classification of the study sites according to wave exposure indicated that rocky shores inside the MPA were mostly sheltered. Jock's Bay, inside the MPA, was the only site classified as partially exposed, with no sites classified as exposed. It was evident that the presence of *W. subtorquata* was higher inside the MPA. This study indicated that *W. subtorquata* was well-represented inside the MPA (Fig. 9). This was complemented by the finding that the abundance of *W. subtorquata* was significantly higher inside the MPA (Fig. 17). The study sites outside the MPA were mostly exposed. The decreased available space for the flourishing of *W. subtorquata* may have reflected the significantly lower abundance of *W. subtorquata* outside the MPA (Fig. 17). The study sites, Betty's Bay Main Beach and Stoney Point were both inside the MPA and classified as sheltered. This study indicated that *W. subtorquata* was well-represented at sheltered sites (Fig. 12), particularly at Betty's Bay Main Beach and Stoney Point (Fig. 10). The abundance of *W. subtorquata* was however not found to be significantly higher at the sheltered sites (Fig. 20). The abundance of this species was found to be significantly higher at the sheltered site Betty's Bay Main Beach compare to the sheltered site Pringle Bay, and the partially exposed site Kleinmond. There was also no significant difference in the abundance of this species between Betty's Bay Main Beach and the wave-exposed sites Kogel Bay and Rooi Els, and between the sheltered site Stoney Point and the wave-exposed sites Kogel Bay and Rooi Els (Fig. 18). This creates speculation for the preference of this species at sheltered regions.

W. subtorquata was well-represented at the low tide pools (Fig 11). The abundance of *W. subtorquata* was significantly higher at the exposed low tide pools compare to the sheltered low tide pools, mid tide pools and high tide pools (Fig. 19). This was an interesting finding as space for recruitment decreases in areas that are subjected to a higher wave exposure. A physiologically thermo-tolerance study by Zerebecki & Sorte (2011) in California indicated that *W. subtorquata* is one of the least heat tolerant invasive species. The decrease in abundance high up the shore most likely reflected an increase in thermal stress.

Predation dynamics may also play an important role in the distribution of *W. subtorquata*. Colony forming species are sensitive to predation, particularly by sea urchins, fish and crab species, chitons, and snails (Nydam & Stachowicz 2007; Jackson 1977). Large predator species impact the assemblages of colony forming species. Small predator species may impact assemblages by feeding on larvae immediately following settlement (Nydam & Stachowicz 2007).

The sea urchin *Parechinus angulosus* was well-represented at the sheltered low tide pools (Fig. 11). Feeding of *P. angulosus* on *W. subtorquata* may also have reflected the significant decrease in abundance of *W. subtorquata* at the sheltered low tide pools (Fig. 19). *P. angulosus* was common outside the MPA (Fig. 9). This may have also reflected the significant decrease in abundance of *W. subtorquata* outside the MPA (Fig. 17).

Biotic and abiotic factors significantly impact the assemblages of colony forming species (Nydam & Stachowicz 2007). No information is available on the predation of *W. subtorquata* in South Africa. There is also no information on the effects of wave exposure on *W. subtorquata* in South Africa. Difference in the abundance of *W. subtorquata* along the Kogelberg Biosphere Reserve coastline was most likely from the integrated functioning of various factors. The effects of wave exposure and predation by *P. angulosus* are speculative and far from conclusive. Various other predator species may also influence the abundance of *W. subtorquata*. To have a comprehensive understanding on the distribution of *W. subtorquata* in South Africa, much more research on the predation of these species needs to be done. It is also important to understand how environmental variables influence the distribution of predator species feeding on *W. subtorquata*.

4.3. Conservation management

Management of marine ecosystems has gained more attention over the past two decades (Agardy 1994) as human activities have increased (Hoegland *et al.* 2001). Oceans are considered the most productive ecosystems overall (Hyrenbach *et al.* 2000). Protection of this vast and dynamic system is critical. The selection, design and implementation of MPA’s focus on the objectives that needs to be met for a specific region (Agardy *et al.* 2003). MPA’s have primarily been implemented to conserve exploited fish stock, other economical viable species, threatened species, biodiversity and ecosystems, and world heritage sites (Boudouresque & Verlaque 2005; Hoegland *et al.* 2001). It has also been implemented to protect regions from various polluting activities (Hoegland *et al.* 2001). It is considered that the management of fisheries has been the main focus for implementing MPA’s (Hoegland *et al.* 2001).

The global lack of systematic marine species sampling is problematic, and species decline and extinction may therefore go unnoticed (Edgar *et al.* 2005). South African marine fauna taxonomic knowledge is fairly good (Griffiths *et al.* 2010; Awad *et al.* 2002). Most attention has been given to coastal regions. This is primarily from the high cost involved gathering information at great depths (Griffiths *et al.* 2010). However, incomplete taxonomic knowledge of South Africa’s marine invertebrate species makes it hard to detect new introduced species (Griffiths *et al.* 2009ab). Sufficient knowledge of the distribution of marine invertebrate species is vital to ensure that a greater variety of species are conserved. Much more research needs to be done to gather ample taxonomic knowledge and information on species richness and abundance in South Africa. Information acquired along the rocky shores of the Kogelberg Biosphere Reserve coastline may be helpful for filling gaps in our knowledge. Sufficient comparative information requires quantitative sampling over large geographic regions (Edgar *et al.* 2005). Edgar *et al.* (2005) recommend a global quantitative survey to assess and address threats to the marine environment. This may only be realistic with sufficient funding for such a large scale, yet important project.

No information is available on designing MPA's to protect regions from possible marine alien species introductions. This may be from the knowledge that once a species establishes along the coast, it is virtually impossible to eradicate (Molnar *et al.* 2008; Branch & Steffani 2004). Marine alien species will potentially invade any favourable region along the coast. Implementing MPA's to decrease the geographical extent of marine alien species is in all likelihood not practical. Identifying and eliminating pathways are considered as the only practical method in preventing possible introductions. Prevention is therefore critical for effective conservation (Bax *et al.* 2003; Orensanz *et al.* 2002). Control of already invaded regions is more costly than prevention (Bax *et al.* 2003).

Early detection may provide opportunity to remove introduced species. Australia and the United States of America reduce possible introductions by employing a preventative approach. Management authorities are provided with a list of species that may potentially invade a region. Additional information on the marine alien and invasive species in regions with similar environments is given (Robinson *et al.* 2004). The invasive history of a species and the native community dynamics must be understood. This is critical for predicting whether its new home range is favourable for its proliferation (Branch & Steffani 2004). It is suggested that regions should undergo regular surveys. Surveys along with a list of potential alien and invasive species offer a sound basis to detect new introductions. This has been the case in developed countries employing the preventative approach (Robinson *et al.* 2004).

Routine surveys require resources and much financial assistance. This may be a potential set-back for developing countries such as South Africa. The first step in employing preventative measures for South Africa is to draw up a list of species that may potentially be invasive. Harbours are major regions of introductions. Routine harbour surveys give opportunity for early detection and possible eradication (Bax *et al.* 2003). Routine surveys of major South African harbours should be done. If viable routine surveys of marine reserves, marine protected areas and coastal regions should be done. The training of staff is also vital. This will also require resources and financial assistance.

Inconsistent information of existing datasets on marine alien species makes comparability difficult. Many of these datasets do not provide information on the negative impacts of marine alien species. This makes risk assessments not viable (Molnar *et al.* 2008). It is of interest that datasets are reviewed and that a middle ground is established on an international level. Molnar *et al.* (2008) reviewed existing datasets. The newly compiled database provides essential information in understanding invasions. These include the characteristics of specific alien species and their native environment. It may therefore be successfully used to manage and prevent further invasions (Molnar *et al.* 2008).

Management of no single vector will successfully stop invasions (Bax *et al.* 2003). Occhipinti-Ambrogi *et al.* (2011) state that a sound understanding of pathways is required to prevent invasions. Global management is as important as regional management. Marine alien species are able to proliferate across borders. The rate of inoculation in major trade harbours are increasing. Good international trade conduct requires sound management initiatives between countries.

Progress in implementing an international regulatory system has been slow. National regulations have therefore been set up by a number of countries. The International Convention for the Control and Management of Ships' Ballast Water and Sediments was put into practice in 2003. The Convention recommends the exchange of ballast water at sea. This is not viable for high risk vessels. Alternative treatments are also recommended by the Convention. It is of concern that alternative treatments may not be beneficial for the environment (Bax *et al.* 2003). The Convention will only be enforced once it is endorsed by the required amount of states and percentage of the world's tonnage (Gollasch 2006). It is anticipated that the Convention will come into force at the end of 2012 (DNV Managing Risk 2011). Saldanha Bay Harbour was selected to represent Africa in the Global Ballast Water Management Programme (GloBallast) (Robinson *et al.* 2004; Bax *et al.* 2003). GloBallast aimed to assist developing countries to effectively manage and minimise the introduction of marine alien species. Further research on ballast water management is critical.

Antifouling biocides are commonly used to prevent fouling of ship hulls. Copper is the most commonly used biocide. Fouling species have been found to be tolerant to copper biocides. These include the bryozoans *W. subtorquata* and *Bugula neritina* (Floerl & Inglis 2005). Many biocide containing antifouling paints have been prohibited in several regions. These biocides have negative effects on the environment and native communities. Environmentally friendly antifouling paints are available. These paints have also proven to have toxic effects on test species (Karlsson & Eklund 2004).

The International Convention on the Control of Harmful Antifouling Systems on Ships was put into practice in 2001. The Convention prohibits antifouling paints harmful to the environment. Vessels entering any harbour of any nation registered under the Convention may not use organotin based paints. Strict procedures for certification are in place. It is however required that by 2008, all vessels must be free from organotin paints or coated to prevent leakage (Savarese 2005). In October 2008, the Convention was endorsed by the required number of States and percentage of the world's tonnage (NOAA Office of the General Council 2011).

This study indicates that the Kogelberg Biosphere Reserve coastline in the Western Cape Province, South Africa is relatively pristine. Only two intertidal rocky shore invasive alien species were found along the intertidal zone. These are the mytilid Mediterranean mussel *Mytilus galloprovincialis* and the bryozoan *Watersipora subtorquata*. Eradication of *M. galloprovincialis* and *W. subtorquata* within the Kogelberg Biosphere Reserve coastline is not viable. This is from the understanding that once a species establishes along the coast, it is virtually impossible to eradicate (Molnar *et al.* 2008; Branch & Steffani 2004). A list of potential invaders from regions with similar environments should be made available to the managing authority. Routine surveys of the Kogelberg Biosphere Reserve coastline, while consulting such a list are recommended. This will require much education and training of all relevant people. It will also require much financial assistance. Harvesting of *M. galloprovincialis* should be considered. *M. galloprovincialis* has a high re-colonisation rate (Nicastro *et al.* 2008; Robinson *et al.* 2007b; Bownes & McQuaid 2006) and harvesting will most likely not greatly reduce the presence of these species.

It is concerning that most of South Africa's MPA's function as isolated units. Oceans are integrated open systems and the protection of isolated units may not be viable. A holistic management approach (Mora *et al.* 2006; Agardy 1994; McNeill 1994) is vital. Sobel (1993) proposes that small no-take MPA's should be interspersed within large multi-purpose marine reserves. The Betty's Bay MPA covers a small geographical region. Exploiting resources by shore angling is permitted. The MPA appears to function as an isolated unit. It is currently under discussion that the protected geographical region should be expanded. It has also been suggested that the MPA should extend from the Steenbras River mouth to the Bot River mouth incorporating distinct zones. It is proposed that each zone is defined and managed accordingly (WWF 2009). This will be beneficial, as the region will then function as a multi-purpose marine reserve. This relatively pristine coastline will therefore be conserved holistically rather than as an isolated unit. It should also be expected that greater species richness will be conserved.

5. Conclusion

The world's marine environment is experiencing much pressure, particularly from human activities. Too much time with inadequate marine conservation efforts has elapsed. Without further knowledge, marine ecosystems, which provide many resources, may ultimately deteriorate. It has only recently being acknowledged that the ocean is highly threatened. However, action is now being taken across the globe. Conserving this vast and integrated ecosystem is challenging and urgently requires much more research to fill gaps in our knowledge. Introductions of marine alien and invasive species are one of many threats experienced by marine ecosystems. This study, along with a wealth of available research findings, clearly indicates that environmental variables play a significant role in the geographic distribution of marine alien and invasive species. Designing suitable MPA's and marine reserves are not viable to protect regions from the proliferation of marine alien and invasive species. It is imperative that both regional and global preventative approaches are considered. The Kogelberg Biosphere Reserve coastline in the Western Cape Province, South Africa is still regarded as a relatively pristine region, with only two invasive alien species being recorded in this study. The proliferation of the rocky shore invasive mytilid Mediterranean mussel *Mytilus galloprovincialis* is related to wave exposure. From this study, it was found that this species are prevalent at wave-exposed regions. Environmental variables determining the geographical range of the invasive bryozoan *Watersipora subtorquata* is speculative. Fortunately, this species does not appear to be causing any significant harm to the coastal biodiversity of South Africa. Eradication of these species within the Kogelberg Biosphere Reserve coastline is not viable. It is important that routine surveys of this pristine region are done to detect early introductions. Many resources, including financial assistance, must be provided to ensure adequate conservation. Knowledge of possible species which may invade this region must be understood. Implementing MPA's and marine reserves are vital for the conservation of marine ecosystems. The possible expansion of the Betty’s Bay Marine Protected Area into a marine reserve will be suitable for conserving this region holistically.

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Appendix A: List of marine alien species along the coast of South Africa.

Species	Common name	Distribution	Impacts	Source
Sagartia ornata	Anemone	Langebaan Lagoon		Branch *et al.* (2010); Griffiths *et al.* (2009b)
Metridium senile	Anemone	Cape Town Harbour; Table Bay Harbour; Agulhas bank		Branch *et al.* (2010); Griffiths *et al.* (2009b)
Pennaria disticha	Hydroid	East-coast to Mozambique		Branch *et al.* (2010)
Obelia dichotoma	Hydroid	Namibia to Mozambique		Branch *et al.* (2010)
Obelia geniculata	Hydroid	Luderitz to Cape Agulhas		Branch *et al.* (2010)
Ficopomatus enigmaticus	Worm	Estuaries and low saline areas from the south-west coast to Durban	Harmless	Branch *et al.* (2010)
Balanus glandula	Barnacle	Southwest coast	Displacing indigenous species	Branch *et al.* (2010); Griffiths *et al.* (2009b)
Monocorophium acherusicum	Amphipod	Estuaries and lagoons from Namibia to mid Kwazulu Natal		Branch *et al.* (2010); Griffiths *et al.* (2009b)
Jassa marmorata	Amphipod	Saldanha Bay & Durban Harbours	Fouls ship hulls, piers and buoys	Griffiths *et al.* (2009)
Jassa morinoi	Amphipod	False Bay, Port Elizabeth & Durban Harbours	Fouls ship hulls, piers and buoys	Griffiths *et al.* (2009b)
Jassa slatteryi	Amphipod	Saldanha Bay, Flase Bay & Knysna Harbours	Fouls hip hulls, piers and buoys	Griffiths *et al.* (2009b)
Paracerceis sculpta	Isopod	Port Elizabeth Harbour		Griffiths *et al.* (2009b)
Carcinus maenas	Crab	Table Bay to Hout Bay	Negative economic impact on the shellfish industry	Branch *et al.* (2010); Griffiths *et al.* (2009b)
Watersipora subtorquata	Lace animal	Southwest coast		Branch *et al.* (2010)
Bugula dentate	Lace animal	Cape Point to Mozambique		Branch *et al.* (2010)
Bugula neritina	Lace animal	Western Cape to Durban	Fouls ship hulls	Branch *et al.* (2010)
Bugula flabellata	Lace animal	Namibia to Port Alfred	Fouls ship hulls	Branch *et al.* (2010)
Mytilus galloprovincialis	Mussel	Namibia to Port Alfred	Mussel culturing industry; Oystercather rehabilitation; displacement of indigenous species	Branch *et al.* (2010); Griffiths *et al.* (2009b)
Crassostrea gigas	Oyster	Southwest coast to Port Alfred (oyster farms); Breede, Goukou & Knysna Estuaries	Oyster culturing industry	Branch *et al.* (2010); Griffiths *et al.* (2009b)
Ostrea edulis		Alexander Bay		Griffiths *et al.* (2009b)
Littorina saxatilis	Periwinkle	Langebaan and		Branch *et al.*

		Knysna Lagoons		(2010); Griffiths *et al.* (2009b)
Tritonia nilsodhneri	Nudibranch	Cape Peninsula		Branch *et al.* (2010)
Tetrapygus niger	Urchin	Namaqualand oyster farms	Consumes algae and forms ‘urchin barrens’	Branch *et al.* (2010); Griffiths *et al.* (2009b)
Microcosmos squamiger	Ascidian	East coast south of Kwazulu Natal	Abundant on ropes in harbours	Branch *et al.* (2010); Griffiths *et al.* (2009b)
Ciona intestinalis	Ascidian	Namibia to Mozambique	Negative for mussel culturing as it aggregates on ropes	Branch *et al.* (2010); Griffiths *et al.* (2009b)
Clavelina lepadiformis	Ascidian	Harbours from Saldanha Bay to East London		Branch *et al.* (2010); Griffiths *et al.* (2009b)
Botryllus schlosseri	Ascidian	Harbours from Saldanha Bay to Durban		Branch *et al.* (2010); Griffiths *et al.* (2009b)
Diplosoma listerianum	Ascidian	Harbours from Saldanha Bay to Mozambique		Branch *et al.* (2010); Griffiths *et al.* (2009b)
Cladophora prolifera	Green algae	West coast from False Bay to Namibia		Branch *et al.* (2010)
Schimmelmannia elegans	Red algae	Table Bay Harbour		Griffiths *et al.* (2009b)

Appendix B: List of marine invertebrate phylums, classes and species included in the study.

Phylum	Class	Order	Infraorder	Species
Cnidaria	**Anthozoa**	**Actiniaria**	**Actiniaria**	*Anthothoe chilensis*
				Actinia mandelae
				Bunodosoma capensis
				Aulactinia reynaudi
				Anthopleura michaelseni
Arthropoda	**Pycnogonida**			*Tanystylum brevipes*
	Cirripedia			*Tetraclita serrata*
				Octomeris angulosa
	Malacostraca	**Decapoda**	**Anomura**	*Paguristes gamianus*
				Diogenes brevirostris
			Brachyura	*Plagusia chabrus*
				Cyclograpsus punctatus
				Hymenosoma orbiculare
Bryozoa				*Watersipora subtorquata*
Mollusca	**Bivalvia**			*Aulacomya ater*
				Mytilus galloprovincialis
	Polyplacophora			*Ischnochiton textilis*
				Ischnochiton oniscus
				Chiton tulipa
				Chiton nigrovirescens
				Acanthochitona garnoti
	Gastropoda			*Haliotis* spp.
				Scutellastra barbara
				Scutellastra cochlear
				Scutellastra longicosta
				Cymbula miniata
				Cymbula oculus
				Cymbula granatina
				Scutellastra granularis
				Helcion pectunculus
				Helcion pruinosus
				Siphonaria serrata
				Siphonaria concinna
				Dendrofissurella scutellum
				Fissurella mutabilis
				Gibbula multicolor
				Gibbula capensis
				Gibbula cicer
				Gibbula zonata
				Oxystele variegata
				Oxystele impervia
				Oxystele sinensis
				Oxystele tigrina
				Turbo sarmaticus
				Turbo cidaris
				Clionella sinuata
				Nucella dubia
				Afrocominella elongata
				Burnupena cincta
				Burnupena lagenaria
				Burnupena catarrhacta
				Faciolaria lugubris lugubris
				Volvarina zonata
				Conus algoensis
				Conus mozambicus
				Berthellina granulata
				Anteaeolidiella indica
Echinodermata	**Asteroidea**			*Parvulastra exigua*
				Marthasterias glacialis
	Ophiuroidea			*Ophiarachnella capensis*
				Amphiura capensis
				Amphipholis squamata
				Ophioderma wahlbergii
	Echinoidea			*Parechinus angulosus*

Appendix C: Identification features of selected species as described by Branch *et al.* (2010).

Figure 21a. Pitted resilial ridge of *Mytilus galloprovincialis*

Figure 21b. Orange to brown flat crust of *Watersipora subtorquata* with pitted zooecia

Figure 21c. Radial shields of *Amphipholis squamata* touching one and other

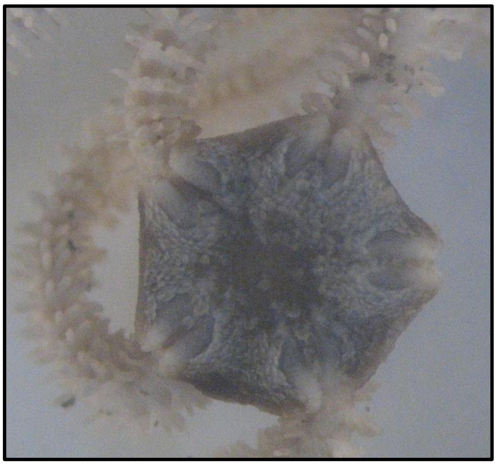

Figure 21d. Radial shields of *Amphiura capensis* touching one and other along the edge of the disk forming “V” shapes

Figure 21e. Pink inner lip of *Oxystele sinensis*

Figure 21f. White inner lip of Oxystele tigrina

Figure 21g. Non-adhesive knobs of *Bunodosoma capensis*

Figure 21h. Radiating dark stripes across the oral disk of *Anthopleura michaelseni*

Figure 21i. Poorly developed arms of *Parvulastra exigua*

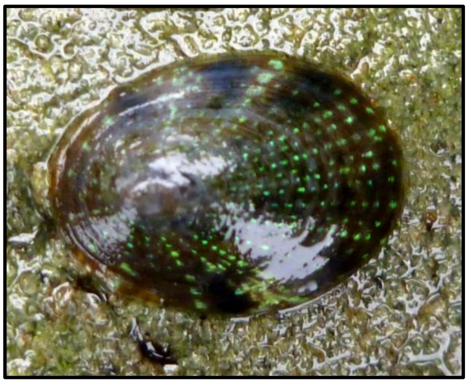

Figure 21j. Iridescent green rays and off centred apex of *Helcion pruinosus*

Figure 21k. White granules on the ribs of *Scutellastra granularis*

Figure 21l. Prickled ribs and anterior positioned apex of *Helcion pectunculus*