Ecological Engineering: an assessment of the ecological impact of Reno mattress structures used in erosion control in the Keurbooms Estuary, South Africa

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Abstract

Global climate changes have been associated with ocean warming and sea-level rise. Armouring of coastlines has become common practice with the increasing threat of coastal erosion. The transformation of soft sediment habitats to hard, artificial habitats because of coastline armouring can lead to changes in species diversity, composition and distribution. It is, therefore, essential to assess changes to habitats from coastal development as well as the ecological impact erosion control structures have within coastal systems. Ecological engineering attempts to combine engineering principals and ecological processes to reduce environmental impacts from coastal development and the implementation of artificial structures. Estuaries are particularly vulnerable to anthropogenic impacts through development, and are extremely important systems offering nursery and foraging grounds for many species. These systems are, however, particularly vulnerable to anthropogenic impacts from urbanisation and development. Within South Africa many estuaries are being transformed by the addition of artificial structures to combat erosion, one such structure is the Reno mattress (a flattened wire box filled with rocks). This study compared the fish diversity and abundances of existing Reno mattress structures and natural eelgrass (*Zostera capensis*) habitat in the Keurbooms Estuary, South Africa. Benthic invertebrates were sampled using standard core sampling and an adapted suction sampling approach within the two habitats. The non-destructive method of mini Baited Remote Underwater Video Systems (BRUVs) was used to sample fish. Seasonal benthic invertebrate and fish abundances and assemblages were assessed from winter 2018 to spring 2019 with greater abundances of both recorded in summer. Significantly greater abundances, diversity, and richness of fish were found in the Reno mattress habitat compared to *Z. capensis*. Invertebrate taxa displayed some overlap between habitats, however, three higher taxonomic groups were only recorded within Reno mattress habitat and one only within the eelgrass habitat. Fish assemblages differed significantly between the two established habitats. A Before-After-Control-Impact (BACI) investigation was used to assess the ecological impact of newly installed Reno mattresses in the Keurbooms Estuary. *Zostera capensis* extent was sampled by determining the percentage cover of 0.5 m X 0.5 m quadrats and measurements of eelgrass blades. Percentage cover and blade length decreased during the installation of Reno mattress, but then recovered shortly after completion of the installation. Abundances, richness and diversity of invertebrates and fish were found to be similar before and after the installation which suggests that the installation had no net negative impact on the site. The Reno mattresses were found to attract fauna typical of rocky shore environments as well as a few invasive alien invertebrate species. This study noted that a hybrid habitat of Reno mattress and eelgrass was created and may in fact provide the positives of both to a system. In any coastal development it will be important to balance the demands of a growing population and the protection of natural habitats. The results of this study suggest that complex artificial structures such as Reno mattresses do provide habitat for fish and invertebrates. However, the use of these structures should be in combination with natural vegetation (e.g. as a hybrid habitat) and not one that replaces intertidal and subtidal natural habitat especially eelgrass. There is limited information regarding the ecological impacts of using Reno mattresses in estuaries and this study provides new information on their ecological efficacy that should be valuable for future coastal erosion control practices.

Keywords: ocean sprawl, artificial structures, estuary, coastal erosion control

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Declaration

This thesis is the result of the author's own work, except where acknowledged or specifically stated in the text. It has not been submitted for any other degree or examination at any other university or academic institution.

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“Always the edge of the sea remains an elusive and indefinable boundary. The shore has a dual nature, changing with the swing of the tides, belonging now to the land, now to the sea.”

Rachel Carson

Chapter 1

General Introduction

1.1 Global climate change and coastal erosion

Changes in global climate have been associated with ocean warming (Nicholls & Cazenave, 2010) and since 1955 global temperatures have increased steadily with projections that will continue rising during the 21st century (Intergovernmental Panel on Climate Change (IPCC), 2007; 2014). The greatest warming of the oceans is in the surface waters, and from 1971 to 2010 the upper 75 m have increased by an average 0.11 (ranging from 0.09 to 0.13) °C per decade (IPCC, 2014). Over the next 100 years global sea surface temperatures may rise by as much as 6.4°C (IPCC, 2007). Sea-level rise has been directly linked to global warming (IPCC, 2007; Allen *et al*., 2018). Between 1901 and 2010 the average global sea-level rose by 0.19 m (IPCC, 2014), and it is predicted that sea levels will rise at a greater rate this century (Holgate & Woodworth, 2004). Two major causes for this are thermal expansion of seawater, and water input from both land reservoirs and ice melt (Titus *et al*., 1991; Twilley *et al*., 2001; IPPC, 2007; Nicholls & Cazenave, 2010).

Much of the world has experienced climatic variability such as an increase in severe storms, changes in precipitation, and flooding events since 1950 (Titus *et al*., 1991; Twilley *et al.,* 2001; IPPA, 2007; 2014). As a result, coastal erosion is an ever-growing global problem driven by climatic change and increasing sea levels (Bruun, 1988; Feagin *et al*., 2005a; Nicholls *et al*., 2007; Cartwright *et al.*, 2008; Firth *et al.* 2013a). For example, estimates found that 15% of Europe’s coastline are actively eroding (European Commission (EC) 2004), and 36% of the USA coastline is vulnerable to erosion (NOAA, 2012). Estimates of the total cost of coastal erosion in Europe between 1990 and 2020 were over €161 000 million, averaging €5400 million per annum (European Commission, 2004). An estimated 60% of the world's largest cities are located within 100 km of the coast (Tibbets, 2002; Bulleri & Chapman, 2010; IPCC, 2014; Firth *et al*., 2016b). Therefore 30-40% of the world's population currently live close to the sea (Small and Nicholls, 2003) with about 10% living in the low elevation coastal zone, which is less than 10 metres above sea level (McGranahan *et al*., 2007). This close proximity to the coast means a greater risk to the effects of coastal erosion including loss of property, reduction in area available for animal and plant communities (Feagin *et al*., 2005b), as well as property damage (Midgely *et al*., 2005).

Of the 48 African countries, 33 have coastlines and are extremely vulnerable to sea-level rise, coastal erosion and flooding (Brown *et al*., 2011). Detailed impacts of coastal erosion on populations are not well studied in many African countries as most are poverty stricken (Brown *et al*., 2011) as many focus on recovery from damage and disasters rather than creation of an adaptive capacity (Mirza, 2003). Many have no coastal baseline data making shoreline modelling and coastal erosion predications difficult (Umvoto, 2010; Department of Environment Affairs and Development Planning (DEA&DP), 2011). It may, however, be possible to use modelling of known values in the prediction of areas with lacking data (ESRI, 2019).

South Africa has a coastal length of 3079 km (Brown *et al*., 2011), most of which is rugged with the shore exposed to high wave energy (Theron *et al*., 2010; Wigley, 2011). Within a South African context coastal erosion has been linked to the increased frequency and severity of storms and flooding (Cartwright *et al*., 2008; Theron & Rossouw, 2008; Sink *et al*., 2012). The east coast of South Africa has experienced a high number of extreme storm events since 2005 causing flooding (Theron *et al*., 2010; Corbella and Stretch, 2012), and an estimated 23% of the coastline is considered to be at high risk of erosion (Wigely, 2011). Infrastructure damages following the storm on the eastern coast of March 2007 were R400 million and possibly as much as R2 billion (Theron *et al*., 2010). Approximately 30% (193 800 000 people) of South Africa's population lives within a 100 km of the coast (Theron & Rossouw, 2008; Umvoto, 2010; DEA&DP, 2011) with several major towns and cities situated at sea level (Griffiths *et al*., 2005). Erosion of coastlines in South Africa is a growing concern and will be exacerbated by an increasing number of people (Wigley, 2011). Within the Western Cape rapid urbanisation of the coastline has been recorded with 90% of people living in urban and residential areas (Midgely *et al*., 2005; Cartwright *et al*., 2008). Although assessments into the potential risk of erosion of coastlines have been carried out in some locations (Midgely *et al*., 2005; Umvoto, 2010; Theron *et al*., 2010; Corbella & Stretch, 2012), a coherent database for the vulnerability of the South African coastline does not exist (Wigley, 2011). Theron & Rossouw (2008) concluded that the southern coast of South Africa was particularly vulnerable to erosion, as most severe wave conditions occur here, especially within the region of Mossel Bay to Natures Valley (Fig. 1.1) which has a number of sandy areas that have a high potential for erosion (Midgely *et al*., 2005).

As a response to sea-level rise and increased coastal erosion, coastlines are often protected with artificial structures (Midgely *et al*., 2005; Firth *et al*., 2013a; Dafforn *et al*., 2015a; b; Evans *et al*., 2016; Heery *et al.*, 2017). Coastal erosion has contributed greatly to a phenomenon called ocean sprawl, which is due to human interventions in coastal processes (Airoldi *et al*., 2005).

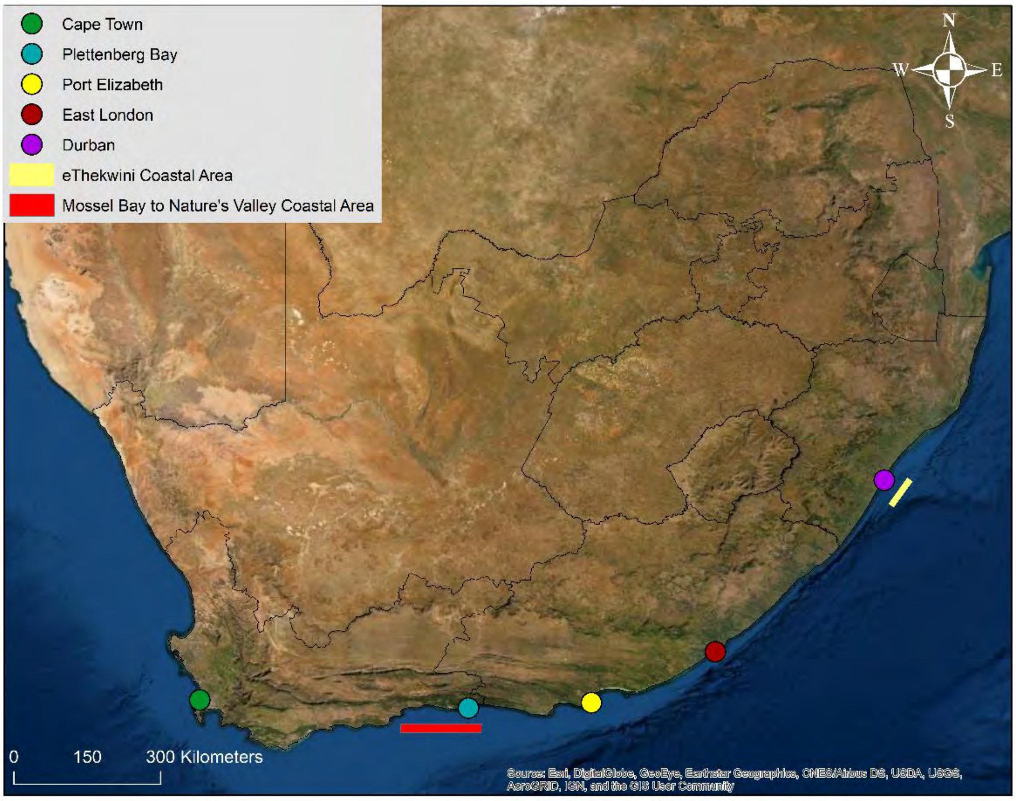

Figure 1.1. Map of South Africa showing: major coastal cities (Cape Town, Port Elizabeth, East London and Durban), in relation to Plettenberg Bay where this study takes place. The section of coastline (red line) vulnerable to erosion. The eThekwini coastal area (yellow line) which is a hot spot for artificial structures as referred to in the text.

1.2 Ocean sprawl

Ocean sprawl is defined as: the proliferation and expansion of artificial structures associated with protection and coastal industries in marine habitats (Duarte *et al*., 2013; Firth *et al*., 2016a; b). Globally, up to 70% of coastlines adjacent coastal cities are modified (Dugan *et al*., 2011; Dafforn *et al*., 2015a; b; Dafforn 2017) and in Australia, Asia, Europe, and North America large sections of urbanised coastline are modified through the addition of artificial structures (Bulleri and Chapman, 2010; Dugan *et al*., 2011; Firth *et al*., 2013a; Dafforn *et al*., 2015b; Dafforn 2017; Table 1.1). This 'armouring' of coastlines aims to protect property, infrastructure and assets (Airoldi *et al*., 2005; Firth *et al*., 2016b).With growing populations and urbanisation, coastal ecosystems are vulnerable to change (Airoldi & Beck, 2007; Dugan *et al*., 2011; Loke *et al*., 2014; 2016).

Historically, communities have been armouring coastlines with hard structures for hundreds of years to stop erosion (EC, 2004; Airoldi *et al*., 2005). The implementation of defence structures has largely been dependant on the value of the locality as well as the nature of the coastline (Corbella and Stretch, 2012). As early as 175 BC low lying coastal areas of the Netherlands were protected by earthen mounds or dams (Charlier *et al*., 2005; Dugan *et al*., 2011). Breakwaters and other structures were used to stabilise harbours in the Mediterranean around 2 BC (Charlier *et al*., 2005) and seawalls were commonly used in Europe between the 5th and 15th century and likely used in the Middle and Far East Asia not only along coasts but also within estuaries (Charlier *et al*., 2005; Dugan *et al*., 2011). The use of groynes and dikes to control erosion has been recorded from at least the 14th century in Northern Europe (Charlier *et al*., 2005) and Denmark began protecting sections of the North Sea with groynes in the 1840s (Charlier *et al*., 2005). Defence structures were predominately used to protect infrastructure and soft-sediment areas against erosion resulting from wave action or flooding (Chapman & Underwood, 2011).

Table 1.1: A summary of the use and extent of artificial structures along coastlines around the world.

Place	Extent	Reference
Australia	32-49% of foreshore areas are modified with seawalls in some Sydney estuaries. Break walls, pontoons and jetties contribute to 10% of length of coastline of the Great Barrier Reef	Creese *et al*., 2009; Waltham & Sheaves, 2015
England and Wales	44% of coastline is defended with hard engineered structures.	Society, 2001; Dafforn *et al*., 2015b; Firth *et al*., 2016b
Europe	22,000 km^2 of coastal zone are armoured.	EC, 2004; Airoldi & Beck, 2007
Mediterranean coasts (France, Spain, and Italy)	An estimated 1500 km of artificial structures can be found along the coast.	Airoldi & Beck, 2007
United States	More than 50% of some estuaries and bays are modified and about 5 to 30% coastlines are armoured. Over 22 000 km of the shoreline is armoured.	Dugan *et al*., 2011; Gittman *et al*., 2015
Japan- Okinawa Island	63% of the 476 km coastline is severely altered through armouring.	Musucci & Reimer, 2019
Singapore	319 km of seawalls constituting 63.3% of the coastline.	Lai *et al*., 2015; Loke *et al*. 2016
South Africa	17% of the coastline has some kind of development within the 100 m of shoreline. Erosion control structures such as groynes; gabion baskets; Loffelstein walls and rock revetments have been implemented along the east coast since the 1980s most are still present. 11% of the 100km eThekwini coastline (Fig. 1.1) is armoured, consisting of 90% hard rock.	Sink *et al*., 2012; Corbella & Stretch, 2012

There are several types of coastal artificial structures, varying in design and materials of which they are made (Figure 1.2 Table 1.2; Firth *et al*., 2016b). Most are resistant to wave action and are constructed out of stone, steel, wood, geotextiles or concrete (Dugan *et al*., 2011). Structures including seawalls, revetments, and bulkheads, are built parallel to the shoreline, usually to protect developments and infrastructure from erosion, or shorelines from movement (European Commission, 2004). In more sheltered areas, artificial structures are used to protect the edges of reclaimed land (Dugan *et al*., 2011). Some structures located offshore are used to reduce erosion and shoreline changes (Dugan *et al*., 2011). Such as, commercial developments, e.g. harbours, have extended along coastlines around the world (Bishop *et al*., 2017) and residential marina estates are becoming increasingly prominent in shallow sheltered estuarine habitats around the world (Waltham and Connolly, 2007; Dugan *et al*., 2011).

Figure 1.2. Different types of artificial structures commonly used in erosion control along coastlines. A: Sea wall of Leisure isle within the Knysna Estuary, B: wooden retaining wall of the Knysna Harbour, C: Loffelstein walls make up part of a promenade in Umhlanga (KZN), D: Gabions form the walls and canals of Thesen Island Marina in Knysna.

Table 1.2: Descriptions of the various artificial structures implemented in coastal areas to combat erosion.

Type of artificial structure	Description	Reference
Bulkheads and seawalls	Vertical or steeply curved structures separating land from water. Can reduce soft-sediment intertidal habitat but do create intertidal and subtidal hard substratum.	EC, 2004; Dugan *et al*., 2011
Loffelstein walls	A coastal retention structure constructed at the backshore and not intended to withstand direct wave action.	Corbella and Stretch 2012
Groynes	Placed perpendicular to the shoreline on beaches and are used to control the movement of sand by littoral drift, or retain sediment.	Dong, 2004; EC, 2004.
Revetments	Large rocks (rip-rap), articulated concrete blocks or tetrapods and have more gradual slopes used to absorb the energy of waves.	EC, 2004.
Gabion baskets	Wired box filled with rocks used in environmental engineering applications such as storm water management, beach stabilisation e.g. Caponga Beach, Brazil, harbour and marina developments e.g. Thesen Islands Marina in Knysna, South Africa.	Maccaferri, 1915; Morais *et al*., 2006; Claassens, 2016.
Dolosse	Developed in South Africa, which are uniquely anchor shaped, used globally to dissipate wave energy and protect coastlines.	Merrifield & Zwamborn, 1967; Cartwright *et al*., 2008.
Jetties and breakwaters	Extend from the shoreline in harbours and inlets controlling flow of sediment and water as well as giving access to boats.	EC, 2004; Dugan *et al*., 2011.
Residential marina estates	Artificial waterway developments. May contain several types of artificial structures.	Waltham and Connolly, 2007; Dugan *et al*., 2011

Coastal developments are generally associated with habitat loss and can be considered a major threat to marine ecosystems (Airoldi and Beck, 2007; Firth *et al.* 2013b; Dafforn *et al.*, 2015a), often replacing soft vegetated shores with hard artificial structures. Artificial erosion control structures are typical in soft sediment environments (Firth *et al*., 2013a) which fundamentally changes the nature of these habitats (Bulleri & Chapman, 2010; Chapman & Underwood, 2011; Lai *et al*., 2015) either directly through displacing organisms or by altering physical and biotic parameters influencing the sediment (Heery *et al*., 2017; 2018). As many artificial structures lack the complexity of natural habitats, they often support lower biodiversity compared to natural habitats (Firth *et al.* 2013a, b; Firth *et al.* 2014a, b;

Firth *et al.* 2016a; Loke *et al*., 2014; 2016). The implementation of artificial structures, especially those with smooth surfaces, leads to physically homologous surfaces (Firth *et al*., 2016b), which are colonised by very few organisms (Firth *et al*., 2013a; Loke *et al*., 2016). Heterogeneity leads to biodiversity of organisms, which promotes the coexistence of competitors (Firth *et al*., 2014a; Porter *et al*., 2018). Habitat complexity and spatial heterogeneity are important factors in the structure and functioning of coastal communities (Firth *et al*., 2014a) and so there is a drive to increase complexity when implementing artificial structures. Hard artificial structures may not function in the same way as the natural substrata they replace (Airoldi & Beck, 2007; Sanabria-Fernandez *et al*., 2018) as they differ physically, in terms of substratum composition, complexity, surface area, and age (Chapman & Underwood, 2011; Bulleri and Chapman 2010, Bishop *et al.* 2017). This can potentially change the assemblages of organisms that inhabit such structures (Chapman, 2003; Airoldi *et al*., 2005; Chapman & Blockley, 2009; Firth *et al.* 2013a, Heery *et al.* 2017), favouring different species, including invasive species (Duarte *et al*., 2013), compared to natural habitats, therefore resulting in significant community differences. However, these differences may decrease over time and result in suitable habitat for animals (Sanabria-Fernandez *et al*., 2018). Changes in diversity results in changes to the composition of animals thus altering ecological processes (Hellyer *et al.* 2011, Heery *et al.* 2017). Another ecological implication of artificial structures is that they increase connectivity, through the introduction of new hard substrata that were previously not available, thus allowing for spread of invasive species (Glasby *et al*., 2007; Dugan *et al*., 2011; Duarte *et al*., 2013). Consideration of these impacts is therefore essential to investigating the ecological efficacy of artificial structures used in erosion control.

1.3 Ecological Engineering in a changing world

There is a need to ensure that ecological consideration is taken in coastal development and sprawl (Morris *et al*., 2018). The proliferation of artificial structures along coastlines around the world is inevitable and expected to increase (Firth *et al*., 2016a; Bishop *et al*. 2017). Ecological engineering combines ecological, societal and economic needs in the design of new artificial structures thus aiming to make them more complex and therefore ecologically beneficial (Schulze, 1996; Bergen *et al*., 2001; Chapman & Blockley, 2009; Chapman & Underwood, 2011; Firth *et al.* 2013a; 2014a; 2016a). Ecological engineering attempts to combine engineering principals and ecological processes to reduce environmental impacts from the implementation of artificial structures (Chapman & Underwood, 2011).

Two types of ecological engineering approaches are commonly used: “hard” and “soft” (Chapman & Underwood, 2011). The soft approach includes the partial or complete removal or rearrangement of the infrastructure, where possible, and reverting the area to a natural state (Chapman & Underwood, 2011). For example, Davis *et al*. (2006) removed a wall from San Diego Bay and replaced it with marsh plants to reduce erosion. Hard approaches not only include innovative designs for structures, designed with ecosystem functioning in mind (Chapman & Underwood, 2011), but is also achieved by altering already installed hard structures to improve their ecological functioning (retrofitting) (Chapman & Underwood, 2011). In most cases, already existing structures have modifications made to them. A “hybrid” approach also can be taken by combining natural habitats into already present shoreline structures or combining hard and soft approaches (Chapman *et al*., 2018). Most artificial structures lack the microhabitats found in natural habitats (Clynick *et al*., 2008) and incorporating surface roughness, crevices, pits, and ledges can facilitate utilization of species that would not normally be able to live on very smooth, featureless surfaces (Chapman & Blockley, 2009; Firth *et al.* 2014b). Even small-scale engineering interventions can result in a positive effect on biodiversity of artificial structures (Firth *et al.* 2016b). Chapman & Blockley (2009) demonstrated that in Sydney Harbour creating artificial “rock pools” along a vertical wall significantly increased the diversity of species, and Firth *et al.* (2016a) also found that concrete “rock pools” could be applied to armoured shorelines in Galway (Ireland).

Ecological engineering can be used to achieve many purposes such as provide hard substrata for colonising invertebrates, alter currents, impede fishing, and attract and increase the abundance of target species through providing shelter and protection (Pickering & Whitmarsh, 1997; Spieler *et al*., 2001; Firth *et al.* 2016a). Structural complexity has been positively correlated to fish abundance and diversity (Spieler *et al*., 2001; Sherman *et al*., 2002) and so manipulating complexity may allow for better mimicry of natural substrata (Hellyer *et al*., 2011). The use of artificial structures, especially in coral reef restoration to increase fish abundance and diversity, has been increasing around the world (Sherman *et al*., 2002; Perkol-Finkel *et al*., 2012). There is the possibility that artificial structures could be used as a means of conserving threatened species. For example, Claassens & Hodgson (2018) found that the endangered Knysna seahorse (*Hippocampus capensis*, Boulenger 1900) seeks refuge and utilises artificial structures in a residential marina in the Knysna Estuary in South Africa, and in Nelson Bay (Port Stephens, New South Wales) *Hippocampus whitei*, Bleeker 1855 (White’s seahorse) has been recorded inhabiting artificial structures (Simpson *et al*., 2020).

One of the coastal ecosystems where artificial structures are often employed are estuaries, and the same is true for South Africa. Despite its prevalence, there seems to be very little research on ecological engineering and the efficacy of these structures as novel habitats for estuarine fauna (Baird *et al*., 1981; Elliott *et al*., 2016). Even with the increasing implementation of artificial erosion control structures globally, there remains a lack of studies on their impacts and how patterns of species assemblages can be altered (Clynick *et al*., 2008; Dugan *et al*., 2011).

Gabions and Reno mattresses™ (a flattened type of gabion) are increasingly used as erosion control structures in sheltered areas (Firth *et al*., 2014b; 2016b) especially in South Africa. These structures consist of wire cages filled with rocks (Maccaferri, 1915) used in environmental engineering applications. To promote sediment retention and erosion control, geotextile material is used as an extra lining. Preliminary work found that careful selection of rock sizes can enhance diversity and abundance of epibiota and can provide protection against predation (Firth *et al*., 2014b).

In recent years artificial structures in the form of Gabions and Reno-mattresses have been introduced into estuaries (Claassens, 2016) in areas normally colonized by natural vegetation such as seagrass. However, there is a dearth of knowledge of the impacts of these erosion control structures, especially within South Africa. With limited information on the ecological impact and without any evidence based decisions being made this raises the question of whether the two habitat (eelgrass vs Reno mattress) types are faunistically similar.

1.4 South African Estuaries

The region where rivers and the ocean meet are referred to as estuaries and are considered to be transitional waters (Day *et al*., 1989; Whitfield, 1998; Whitfield & Elliott, 2011). According to Whitfield & Elliott (2011) they are defined as "a semi-enclosed body of water that is connected to the sea either permanently or periodically, has a salinity that is different from that of the adjacent ocean due to freshwater inputs, and includes a characteristic biota". Estuaries play a critical role in linking freshwater, marine and terrestrial environments and therefore, are vulnerable to pressures taking place at multiple levels (van Niekerk & Turpie, 2012).

Estuaries provide a number of functions and ecosystem services that coastal communities depend upon including, carbon storage; significant buffering against floods and storms; and natural resources (van Niekerk & Turpie, 2012). Estuaries are important systems as they are known areas of high biodiversity and productivity (Whitfield, 1999; Turpie *et al.,* 2002; Vasconcelos *et al.,* 2007; van Niekerk & Turpie, 2012) largely due to the richness of nutrients derived from both fresh and marine water sources (de Villiers *et al*., 1999). Providing a number of fish and invertebrates species transits between the ocean and rivers, they act as refuge for spawning, as well as provide nursery areas and feeding grounds (Whitfield, 1994b; 1999; Turpie *et al.,* 2002; Vasconcelos *et al.,* 2007; James & Harrison, 2008; Sutherland *et al.,* 2012). Many estuaries are critical habitats that support endemic species (Turpie *et al.,* 2002; Sutherland *et al.,* 2012) and therefore their management is vital.

There are 290 estuaries and 42 micro-estuaries in South Africa (Whitfield, 2000; van Niekerk & Turpie, 2012; Skowno *et al*., 2019), which make up approximately 70000 ha of productive habitat (Turpie *et al*., 2002; van Niekerk & Turpie, 2012; van Niekerk *et al*., 2013). In southern Africa, estuaries have been categorised, using a multidisciplinary approach, as permanently open estuaries; temporarily open/closed estuaries (which refers to the state of the mouth); estuarine bays; estuarine lakes; and river mouths (Whitfield, 1992; 1998). Larger estuarine systems, coinciding with urban areas, tend to be more valuable with subsistence harvesting in estuaries and coastal habitats estimated at R35.7 million per year (Turpie *et al*., 2017). Estuaries in South Africa are characterised by large numbers of juvenile marine fishes that return to sea after maturing (Whitfield, 1983; 1990; 1994b). An estimated R803 million per annum was calculated for the nursery value of South African estuaries with the highest value the south Western Cape and Eastern Cape. This value, however, is only 58% of what it would be if all estuaries were in their natural state (Turpie *et al*., 2017). Despite the importance of estuaries, many are threatened through increasing urbanisation and utilisation by humans (Turpie *et al*., 2002) and the impacts of climate change (Fijii, 2012).

Intense urban development along South Africa's coastline is focused in estuaries (Morant & Quinn, 1999; Midgely *et al*., 2005; Cartwright *et al*., 2008; Skowno *et al*., 2019), as they capture the benefits of coastal living, making them threatened and vulnerable to change. According to Skowno *et al*., (2019), estuaries are the most threatened realm in South Africa with 29% under significant pressure from development (an increase of 16% since the 2012 van Niekerk & Turpie study) less than 10% having no development at all. All the large estuarine systems are heavily overexploited in terms of their living resources, in particular fish and invertebrate species, which results in changes in population size, biomass and community composition of the systems (van Niekerk & Turpie, 2012; Skowno *et al*., 2019).

The key pressures on estuaries within South Africa are flow modification, pollution, habitat destruction, climate change, and exploitation of living resources (van Niekerk & Turpie, 2012; Skowno *et al*., 2019). A variety of habitat types (sand or mudflats, mangroves, salt marshes, macroalgae, seagrasses, and submerged macrophytes) are found within South African estuaries some of which can be overlooked during species diversity assessments (Whitfield, 1983; Turpie *et al*., 2002; van Niekerk & Turpie, 2012). Estuaries with a range of substrata and habitats have greater diversity of species and thus should be considered as more important and have greater conservational efforts (Turpie *et al*., 2002). Habitat modification is driven by urban development, riparian, and transport infrastructure (Skowno *et al*., 2019). Within estuaries one such habitat that is most threatened by human development and recreational activities is seagrass beds (Adams *et al*., 1999; Duarte, 2001; Barnes, 2017).

Seagrasses are angiosperms (flowering plants) that are rooted in soft subtidal and low intertidal substrata. As they are almost always covered by sea water they are classified as submerged macrophytes (Adams *et al*., 1999; Short *et al*., 2006). Although most grow subtidally there are a few species capable of living in the intertidal zone, e.g. *Zostera* spp., *Phyllospadix* spp., and *Halophila* spp. (den Hartog, 1970; Hemminga & Duarte, 2000).

Seagrass beds are found in shallow estuarine and coastal waters on all continents, except Antarctica (Green & Short, 2003; Whitfield & Elliot, 2011; Bertelli & Unsworth, 2014) and contribute greatly to ecosystem functioning (Duarte, 2001; Corlett & Jones, 2007; Browne, 2012). Considered to be permanent complex habitats for both fish and invertebrate species, they act as nursery and foraging grounds, and provide a refuge for smaller individuals and protection from predators (Whitfield, 1989; Jackson *et al.* 2001; Short *et al*., 2006). Seagrass beds provide food sources for a number of species that either feed directly on them or the epifaunal growth on their leaves (Jernakoff & Nielsen, 1998; Van Elven *et al*., 2004; Browne, 2012). They also provide the regulation of nutrient cycles, provision of oxygen to sediments and water as well as exporting organic carbon to adjacent ecosystems (Adams *et al*., 1999; Duarte, 2001; Short *et al*., 2006).

Seagrasses are important foraging and refuge habitats for many commercially and recreationally important fish species in South African estuaries (Sheppard *et al*., 2011; Whitfield, 2019) and harbour large numbers and biomass of invertebrates (Barnes, 2010). Eelgrass (*Zostera capensis* Setchell, 1933) is the most dominant seagrass species in South Africa, extending from the southeast coast to Kenya (Adams, 2016), and is abundant in permanently open estuaries where euhaline salinities are prevalent (Adams *et al*., 1999; Browne, 2012; Adams, 2016; Whitfield, 2019).

Seagrasses are vulnerable to degradation by human influenced stresses such as, bait digging, boating damage, nutrient enrichment, reclamation of land, or dredging (Adams *et al*., 1999; Duarte, 2001; Adams, 2016; Human *et al*., 2016). There have been significant losses of seagrass meadows around the world, which is expected to accelerate, due to anthropogenic pressures and therefore, numerous monitoring and conservation efforts have been implemented (Duarte, 2001; Orth *et al*., 2006).

1.5 Study area: The Keurbooms Estuary

The Keurbooms Estuary (34°02’17’’S, 23°23’12’’E), situated near the town of Plettenberg Bay on the southern coast of South Africa (Fig. 1.2), is one of the few permanently open estuaries in South Africa (Whitfield, 1992; Bornman & Adams, 2005). At approximately 295 ha (Midgely *et al*., 2005), with a river catchment of 859 km^2 (Schumann, 2015), the Keurbooms Estuary has been ranked 18th in South Africa in terms of conservational importance (Turpie *et al*., 2004) with an overall importance score of 88.3 (Turpie & Clark, 2007). Water depth ranges from 0.5 m to 2.7 m (James & Harrison, 2011), and water temperatures range between 12°C and 28°C (Day, 1981). It is characterised by regular fluctuations in salinity and water levels (James & Harrison, 2011) where the lower reaches have an active tidal exchange in which the entire water column is flushed (Huizinga & Slinger, 1999). The turbidity in the system is generally low, with an average Secchi depth reading of 1.5 m in water depths averaging at 1.4 m(Duvenage & Morant, 1984) and less than 1 NTU (James & Harrison, 2011). The system is oligotrophic with low nutrient concentrations (Duvenage & Morant, 1984). The last estimated value for subsistence fisheries in the Keurbooms Estuary was R 379006 per annum (Turpie & Clark, 2007). Eelgrass occupies intertidal mudflats of many Cape estuaries (Adams *et al*., 1999), and is the dominant submerged macrophyte in the Keurbooms Estuary with 64 ha (Adams, 2016) occurring both intertidally and subtidally (Duvenage & Morant 1984; Bornman & Adams, 2005).

The largest proportion of invertebrates found within the Keurbooms Estuary are benthic or associated with the aquatic vegetation (Bornman & Adams, 2005). A total of 29 species of marine and estuarine fish have been recorded in the estuary (Whitfield, 1994a; Bornman &

Adams, 2005) with a number recognised as over-exploited such as Spotted Grunter (*Pomadasys commersonnii* Lacepède, 1810), Leervis (*Lichia amia* Linnaeus, 1758), and White Steenbras (*Lithognathus lithognathus* Curvier, 1829). This estuary requires protection as it provides essential habitat for the endangered Knysna seahorse (*Hippocampus capensis*) (Whitfield, 1995; Lockyear *et al*., 2006).

The Keurbooms Estuary is very susceptible to flooding events and most erosion damage occurs below the N2 road bridge, where most of the urban development is found (Enviro-Fish Africa, 2010). Bank restoration has been implemented along the eastern bank (Royal HaskoningDHV, 2018).

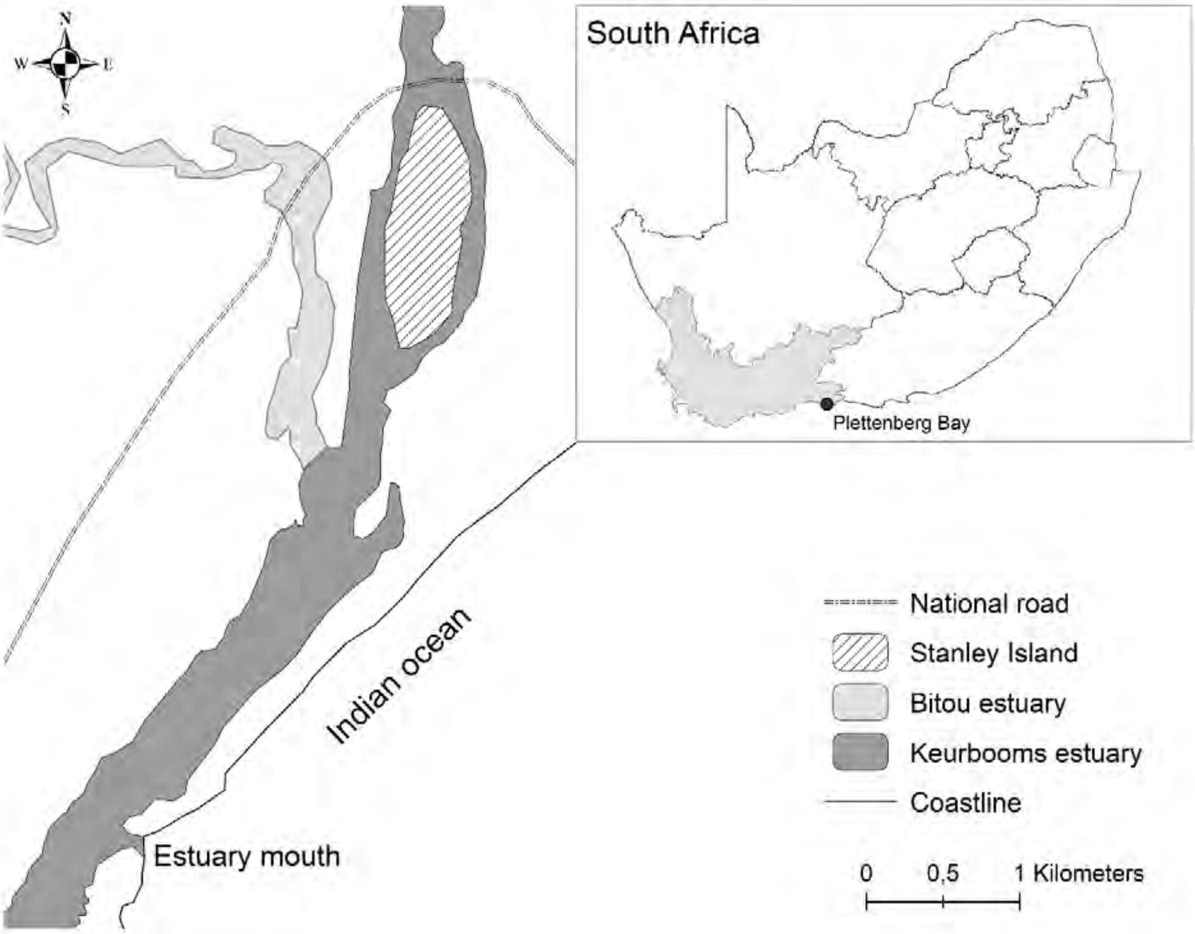

Figure 1.3. Map showing the Keurbooms Estuary situated near the town Plettenberg Bay on the southern coast of South Africa.

1.6 Rationale and Aims

There are no ecological data for the impact of erosion control structures in South African estuaries and if artificial structures are to be increasingly used in South African estuaries, it is important for managers to know whether these structures can have positive or negative impacts. This study will generate data to inform management. The aim of this study was to assess the ecological efficacy of the erosion control structure Reno mattress and understand the ecological impact of bank stabilisation by comparing the faunistic differences between natural and artificial habitats in the lower reaches of the Keurbooms Estuary. This was achieved by firstly comparing natural eelgrass to artificial habitat to determine if faunistic differences were present and secondly, the ecological impact from construction of an artificial habitat was assessed.

Chapter 2

A comparison of faunistic assemblages in natural and artificial habitats

2.1 Introduction

Growing human populations have put great pressure on coastal areas especially around estuaries (Waltham & Connolly, 2011; Heery *et al*., 2018; Henderson *et al*., 2019). These important systems are especially vulnerable to development as they are influenced by changes in both the marine and terrestrial environments (Airoldi *et al*., 2009; James *et al*., 2013; Rivero *et al*., 2013). Climate change is y having an impact on the marine environment (Allen *et al*., 2018). For example, changes to the marine environment include sea-level rise and the increased frequency and intensity of storms, which will have significant impacts on estuarine systems (Fujii, 2012; James *et al*., 2013; Leuven *et al*., 2019). The growing threat of erosion and flooding, resulting from sea-level rise, has led to the need to protect coastal infrastructure, thus armouring of the banks of estuaries is expected to increase (Perkol-Finkel *et al*., 2012; Firth *et al*., 2013a).

Changes from a natural system to one altered by artificial structures, through the development of harbours and erosion control structures, is a threat that requires investigation (Clynick, 2007; Chapman and Underwood, 2011; Dafforn *et al*., 2015a). The implementation of artificial structures within estuaries, however, is often done without comprehensive knowledge of their ecological consequences (Connell and Glasby, 1999; Clynick *et al*., 2008; Rivero *et al*., 2013). Artificial structures can change the nature of natural habitats through the displacement of animals by allowing different types of species to colonise an area, thus altering species abundances, composition, and the ecosystem functions they support (Heery *et al*., 2017; Franzitta & Airoldi, 2019; Mayer-Pinto *et al*., 2018; Henderson *et al*., 2019). For example Heery *et al*. (2017) noted that artificial structures may affect sedimentary ecosystem functioning not only through direct displacement but also in provision of ecosystem services. And Mayer-Pinto *et al*., (2018) found large differences in the community structure and function between natural and artificial habitats in the one of largest urbanised estuaries in the world, Sydney Harbour.

Estuaries are characterised by soft sediments and the lower reaches of some have seagrass beds, a habitat which is particularly vulnerable to anthropogenic disturbance (Adams *et al*., 1999; Duarte, 2001). As seagrasses are found at the margin between land and sea, and face a number of anthropogenic impacts such as eutrophication, land reclamation (Short *et al*., 2006; Airoldi & Beck, 2007) and siltation (Adams, 2016). The loss of seagrass may result in a shift in the dominance of different primary producers, the loss of sediment protection, as well as significant loss and alteration of biodiversity, therefore altering food webs (Hemminga & Duarte, 2000; Duarte, 2001). One such estuary in the Western Cape of South Africa (Fig. 1.1; 1.2) in which seagrass habitat has been lost through the application of erosion control is the Keurbooms Estuary. This estuary is susceptible to flooding (Duvenage & Morant 1984; Schumann, 2015) with the eastern bank in the lower reaches having experienced marked erosion, caused by flooding, over the past 13 years. To remediate erosion damage of the eastern bank in the eastern channel within the lower reaches of the Keurbooms Estuary, it has been extensively armoured with Reno mattresses (see Chapter 1 for description of these structures). This presents the question as to whether the artificial Reno mattress habitats created within the Keurbooms Estuary are faunistically similar when compared to natural eelgrass (*Zostera capensis*) habitats. This study, therefore, aimed to assess the ecological efficacy, in terms of providing habitat to species while preforming an erosion control function, of Reno mattresses by comparing the fish and invertebrate species diversity, and abundance within this artificial habitat with those of adjacent natural eelgrass habitat. It was expected that the Reno mattress and natural eelgrass habitat would have similar species of invertebrates and fish, however, abundances would differ.

2.2 Methods and Materials

2.2.1 Study site

Urban sprawl along the lower eastern bank of the eastern channel of the Keurbooms Estuary has resulted in armouring of the shoreline as well as construction of wooden jetties. To control the erosion a number of structures have been introduced, including: retaining walls and rip-rap (pers. obs, Fig. 2.1). More recently, Reno mattresses have been placed in some sections of the lower reaches of the Keurbooms Estuary and extend from the intertidal zone subtidally onto eelgrass (*Zostera capensis*) beds. Eelgrass makes up the majority of natural subtidal habitat in this region of the estuary (Duvenage & Morant, 1984; Bornman & Adams, 2005).

Figure 2.1. Artificial erosion control structures found along the eastern channel in the lower reaches of the Keurbooms Estuary. A: rip-rap rocks, B: wooden retaining walls.

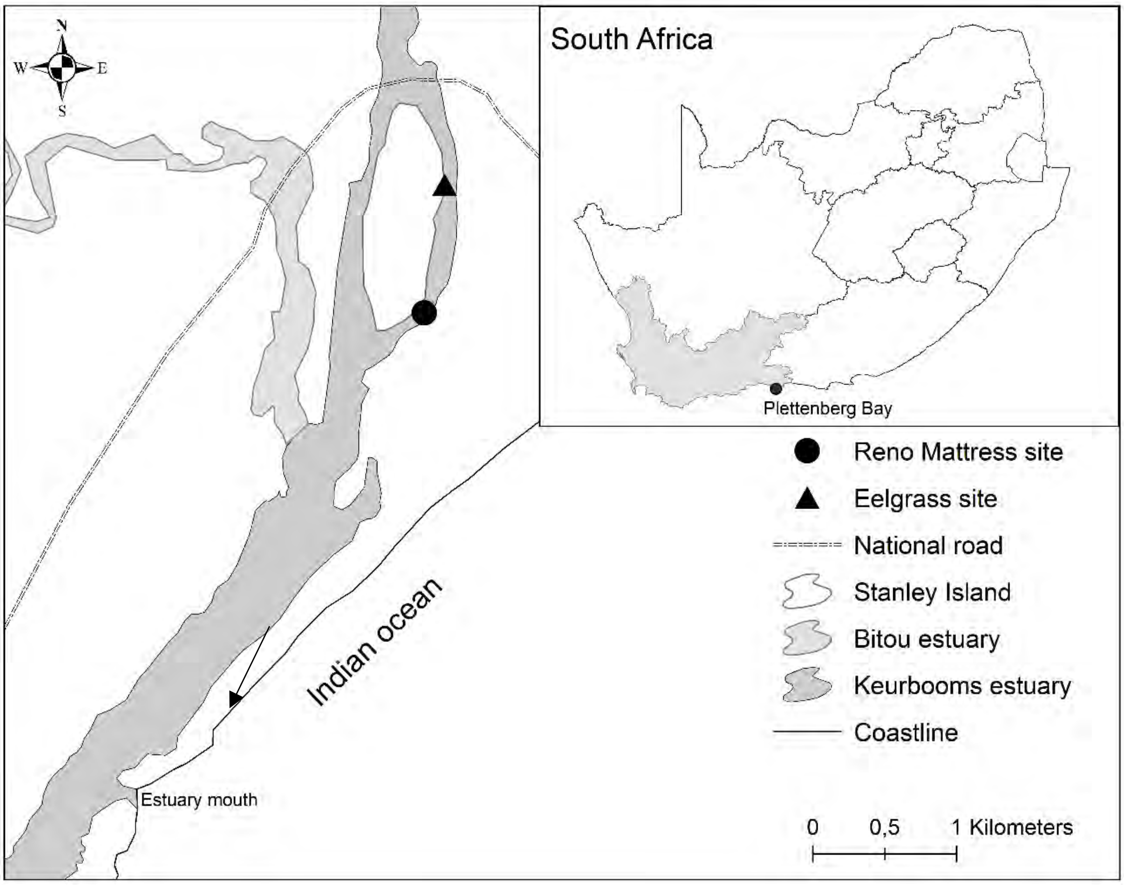

Figure 2.2. The Keurbooms Estuary with the position of each habitat: triangle: Eelgrass (*Zostera capensis*), circle: Reno-mattress.

To investigate the macrofaunal differences between natural and artificial habitats, two sample sites were chosen based on habitat type (Figs. 2.2 and 2.3):

I. Eelgrass (*Zostera capensis*). Dense subtidal eelgrass beds adjacent to the western bank of the eastern channel of the estuary.

II. Reno mattress (> 3 years old). An existing 120 m Reno mattress structure (consisting of several mattresses laid end-to-end parallel to the shoreline) was identified 1500 m downstream from the eelgrass site. This Reno mattress was constructed to remediate erosion damage caused by flooding. The upper section of the structure is located intertidally and is exposed during low tide. The lower section extends subtidally. In some areas of the subtidal Reno mattress, the mattress is covered by sediment and eelgrass has established. A strip of eelgrass is located along the bottom edge of the structure and this habitat was regarded as a hybrid which included hard Reno mattress structures with adjacent eelgrass patches (Fig. 2.4). This was the only location in the estuary where established (> 3 years) subtidal Reno mattress was present which limited by the availability of suitable habitat type therefore sampling took place at this site only as it was also suitably positioned in terms of tidal height (sub-tidal).

Figure 2.3. Two habitats types found within the eastern channel of the Keurbooms estuary: A: Eelgrass (*Zostera capensis*); B: Hybrid Reno mattress and eelgrass habitat (with mussels and barnacles growing on wire).

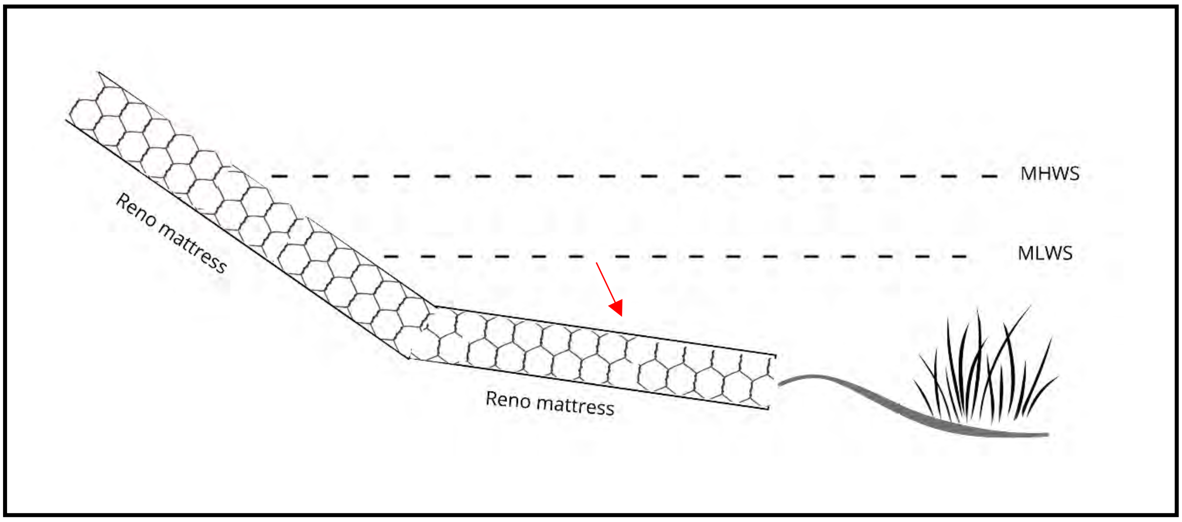

Figure 2.4. Schematic of the layout of Reno mattress at Habitat II within the Keurbooms Estuary. MHWS- mean high water spring; MLWS- mean low water spring. Red arrow indicates where samples were taken.

To determine the macrofaunistic differences between the natural vegetation and artificial habitats, Habitat I (eelgrass) was compared with Habitat II (Reno mattress) over four seasons (winter- July 2018, spring- October 2018, summer- February 2019, autumn- April 2019). To ensure comparability between habitats, all samples were collected subtidally (about 2 m below low tide depth at Habitat I and <1 m low tide depth at Habitat II). Owing to the limited extent of the Reno mattress structure and to ensure independence of sampling (minimum of 30 m distance between sites), the number of sites surveyed was four. Both invertebrate and fish taxa were monitored in the comparison between habitats.

2.2.2 Invertebrate sampling approach

Owing to the inherent physical structural differences between the two habitat types (wire boxes filled with rocks vs eelgrass), different sampling approaches were used in each habitat. Therefore a qualitative comparison detailing the types of invertebrates found in the two different habitats was the focus for this chapter. Benthic macrofauna found in the eelgrass habitat (Habitat I) was sampled seasonally using methods adapted from Barnes (2010, 2013) and, Barnes & Ellwood (2012). Within Habitat I, at the four sample sites, 10 replicate core sediment samples, spaced 1 m apart, were taken haphazardly within a 10 by 10 m area. Thus, a total of 40 samples were collected from the eelgrass habitat for each season. Each core had a diameter of 50 mm and a depth of 50 mm (Fig. 2.5). The 50 mm depth was selected as the majority of animals are found to dwell within this depth in seagrass beds (Klumpp & Kwak 2005). Samples were collected in a polythene bag during spring low tides by snorkelling.

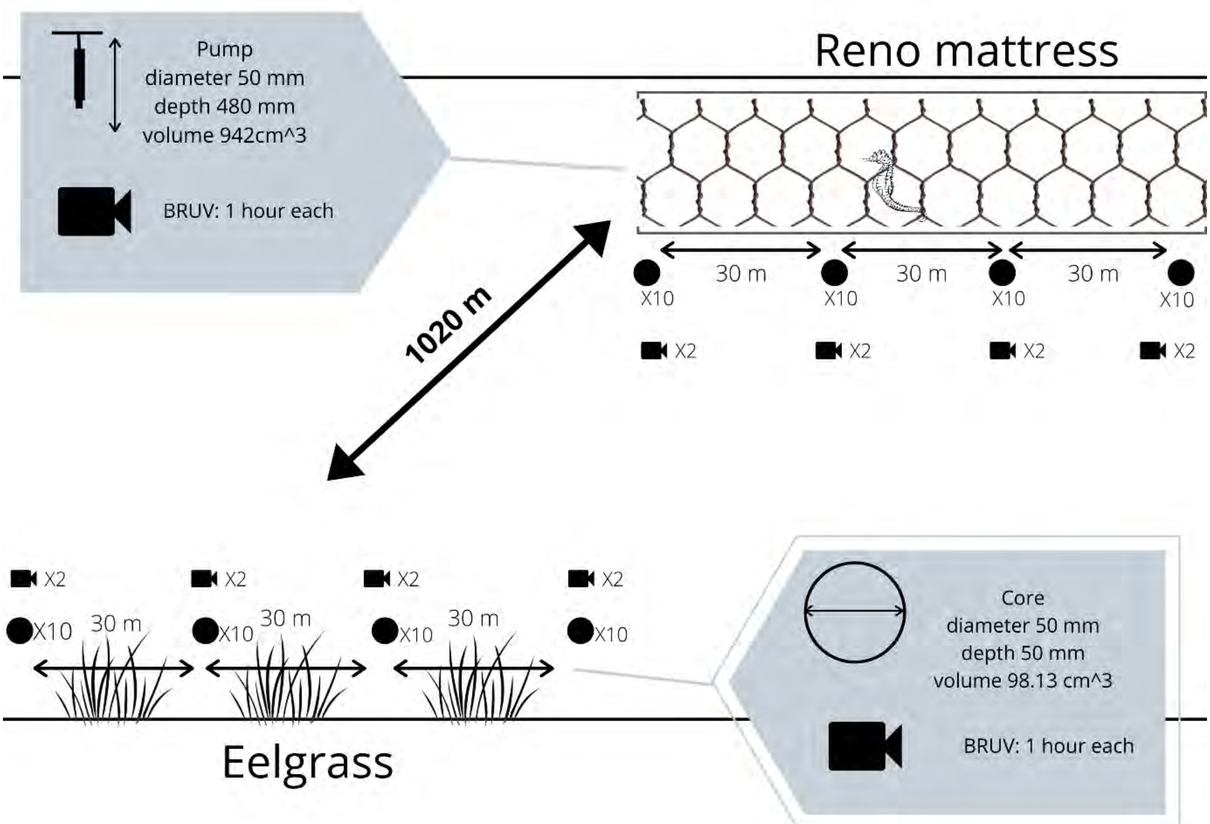

Figure 2.5. Sampling procedure at Habitat I (Eelgrass) and Habitat II (Reno mattress) for invertebrates and fish taxa and percentage cover of *Z. capensis*.

The sample approach described above could not be used within the Reno mattress habitat owing to the hard nature of the artificial structure. A suction device (Fig. 2.6), adapted from the design of Gulliksen & Derås (1975), was used to collect all epibenthic macrofauna found within the crevices and on top of the Reno mattress structures. Suction devices have been found to be complementary to other non-destructive methods of sampling (Rostron, 2001; Hellyer *et al.*, 2011) and provide a number of advantages to sampling including ease of collection within areas of hard substratum (Rostron, 2001). The simple handheld suction device allowed for a more refined collection of animals and sediment. Within each of the four sites identified 10 samples were sucked, one metre from one another. Thus, a total of 40 samples were collected from the Reno mattress habitat each season.

All collected samples from Habitat I and II were sieved through a 710 µm mesh sieve and retained material was placed into a large white tray from which living macrofauna were extracted by eye and through the use of a microscope, extraction continuing until no further animals could be seen after a 3 minute search. Faunal individuals were identified to lowest taxonomic level possible using field guides: Day (1969) and Branch *et* al., (2010) (names and taxonomy were verified using World Register of Marine Species (WoRMS) (http://www.marinespecies.org/aphia.php?p=taxlist (2019)) and through assistance from Professor Richard Barnes (Cambridge and Rhodes Universities)), and counted. All adult invertebrates found were then categorised into functional feeding groups. An identifier code was created, adapted from Macdonald *et al*. (2010), using the diet (herbivorous, omnivorous, and carnivorous) and feeding mode (browser, deposit feeder, detritus feeder, filter feeder, grazer, lignivorous, predator, parasite, and scavenger) of each identified animal (Table 2.1).

Figure 2.6. Modified manual suction pump system adapted from the design of Gulliksen & Derås (1975) used to collect all epibenthic macrofauna found within the crevices and on top of the Reno mattress structures.

2.2.3 Fish sampling approach

Baited Remote Underwater Video systems (BRUVs) are an established method to investigate fish assemblages (Malcolm *et al*., 2007; Bernard & Götz, 2012; Roberson *et al.,* 2015). However, in small tidal systems such as rock pools and estuaries large video systems are not ideal owing to poor visibility, and therefore compact miniature BRUVs (mini BRUVs) are used (Harasti *et al.*, 2014; Davis *et al*., 2018).

The use of mini-BRUV systems was in accordance to the methods outlined by Harasti *et al*. (2014). Sampling was carried out seasonally (winter- July 2018, spring- October 2018, summer- February 2019, autumn- April 2019), and cameras were deployed at the same four sites where invertebrate sampling took place (30 m apart to minimise any interaction between cameras). Eight high definition Go-Pro underwater video cameras (one Hero, two Hero 3, two Hero 3+, one Hero 4, one Hero 6, and one Hero 7) were attached to a PVC pipe frame that consisted of a 30 cm bait arm with a small mesh bait bag incorporated into it. All the cameras were set to the same resolution and field of view to ensure consistency in frame size among cameras. The Go-Pro cameras were set to wide-angle video with a resolution of 720 and 60 frames per second. Two 1 kg lead weights were used to anchor the system (Fig. 2.7). Bait consisted of one crushed pilchard (*Sardinops neopilchardus*) per camera system. To increase replication, the four sites within each habitat were sampled twice over a two-day period. Each deployment lasted an hour and was conducted an hour before an incoming neap high tide. This resulted in the collection of eight one-hour samples per habitat type per season. It has been shown that generally a deployment time of one hour is suitable to record 90-95% of fish species (Bernard & Götz, 2012; Harasti *et al.*, 2015).

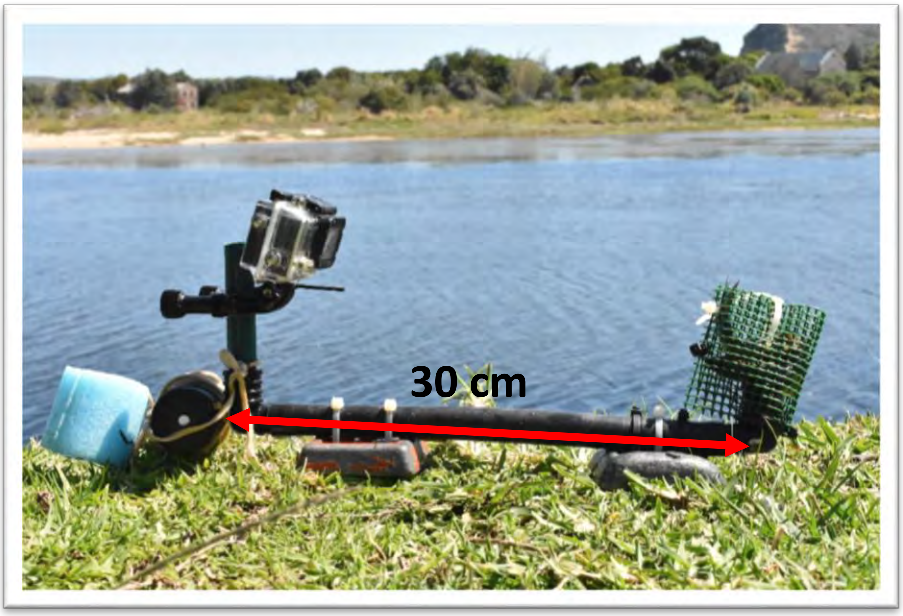

Figure 2.7. Miniature Baited Underwater Remote Video system (mini-BRUV) used to sample fish species within the Keurbooms estuary.

The one-hour videos were analysed using SeaGIS EventMeasure 5.25 software (www.seagis.com.au) following standard procedure (Harasti *et al*., 2015). Fish were identified to species level where possible according to FishBase (https://www.fishbase.se/search.php). It was not possible to identify the families Mugilidae and Clinidae to species level from camera footage and therefore all species are referred to family level respectively. The MaxN value of each video was determined, where MaxN is the maximum number of any individual fish (of a species) in any one frame during the duration of the video (Cappo *et al*., 2003). Species were sorted into feeding guilds (piscivorous, carnivorous, herbivorous, zoobenthic predator, detritivorous) as stated in Smith's Sea Fishes (Smith & Heemstra, 1988), and their dependence on estuaries was noted by scoring species from I-V as described by Wallace *et al*. (1984) (Table 2.4).

2.2.4 Statistical analysis

A direct statistical comparison of invertebrate abundances between Habitat I and II was not possible owing to the differences in sampling approach used for each habitat type. Therefore, qualitative descriptions of invertebrate species diversity were given as an indication of similarities or differences between habitats. Invertebrates were grouped into 14 higher taxonomic levels (Caenogastropoda, Euthyneura, Heterobranchia, Pteriomorphia, Heterodonta, Sedentaria, Errantia, Nematoda, Polycladida, Decapoda, Amphipoda, Eumalacostraca, Cumacea, and Thecostraca). Using the statistical program R studio (R Development Core Team, 2014) within each habitat higher taxonomic levels and functional group abundances were compared between seasons using a Kruskal-Wallis and significant differences were assessed using a Least Significance Difference (LSD) post hoc test.

Fish species that had only a single recording (i.e. one specimen only) during the entire study were removed from the analysis as they do not accurately represent the composition of species. Species richness was calculated using counts of the number of fish species in a sample. Using the statistical program R studio (R Development Core Team, 2014) a Shannon Diversity index (H') was calculated for each sample. The normality of species richness and diversity was tested using Shapiro-Wilk tests and differences between habitats were assessed using Wilcoxon rank sum tests and differences among seasons were compared using a Kruskal-Wallis. All non-normal data were analysed using non-parametric statistical tests. Using the statistical program Primer-E 6 & PERMANOVA, abundances, species assemblages and functional feeding groups of fish were compared between habitat types and seasons. Multivariate analyses were carried out after abundance data were square-root transformed to balance out the importance of those species found in large abundances and those in much smaller abundances (Anderson *et al*., 2008). A resemblance matrix using a Bray-Curtis similarity index was created and PERMANOVA and PERMANOVA pairwise tests were performed to determine where variation was found. Principal Coordinate Analysis (PCO) and Canonical Analysis of Principal coordinates (CAP) plots were done to visualise the variation with Pearson correlation vectors imposed on the visualisations.

2.3 Results

2.3.1 Descriptive comparison of invertebrate assemblages

A total of 36 macroinvertebrate taxa were recorded in both habitats across all seasons, of which four were only found within Habitat I (Eelgrass) and 16 only found within II (Reno mattress) (Table 2.1). Of the 14 higher taxonomic groups identified 71.43% were found in both habitats. Three higher taxonomic groups were only recorded in Habitat II (Thecostraca, Heterobranchia, and Cumacea) and one group (Nematoda) was only recorded in Habitat I (Table 2.1). A total of 21 macroinvertebrate taxa were recorded in Habitat I. Within Habitat I the abundances of Amphipoda differed significantly among seasons (H_3= 37.22, $p < 0.01$, Kruskal-Wallis) with the greatest abundances in spring ($p < 0.01$, LSD, Fig. 2.8). Abundances of Caenogastropoda differed significantly among seasons (H_3= 39.01, p <0.01, Kruskal-Wallis) with greatest abundances in summer (p < 0.01, LSD, Fig. 2.8). Abundances of Decapoda differed significantly among seasons (H_3= 7.65, p < 0.05, Kruskal-Wallis) and greater abundances were found in autumn compared to winter (p < 0.05, LSD, Fig. 2.8). Abundances of Errantia differed significantly amongst seasons (H_3= 8.01, p < 0.05, Kruskal-Wallis) with greater abundances found in winter compared to summer (p < 0.01, LSD, Fig. 2.8). The abundances of Heterodonta differed significantly among seasons (H_3= 14.15, p < 0.01, Kruskal-Wallis) with the greatest abundances recorded in winter (p < 0.05, LSD, Fig. 2.8). The abundances of Polycaladida differed significantly among seasons (H_3= 9.11, p < 0.05, Kruskal-Wallis). The abundances of Sedentaria showed a significant increasing trend from winter to autumn (H_3= 9.1139, p < 0.05, Kruskal-Wallis, Fig. 2.8).

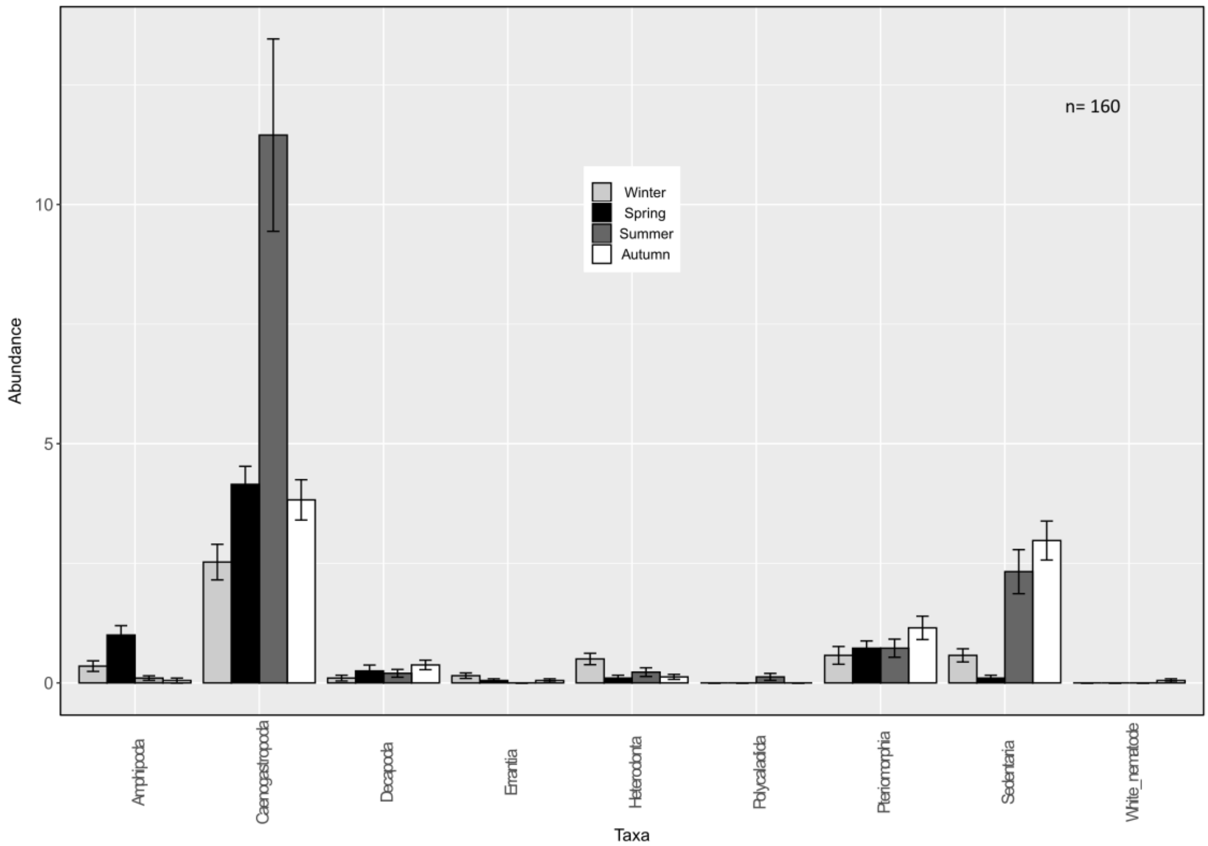

Figure 2.8. Mean (±se) abundances of invertebrates across seasons at Habitat I (Eelgrass) recorded in the Keurbooms Estuary.

A total of 33 macroinvertebrate taxa were recorded in Habitat II. Within Habitat II the abundances of Amphipoda differed significantly among seasons (H_3= 78.29, $p < 0.01$, Kruskal-Wallis) with the greatest abundances in spring ($p < 0.01$, LSD, Fig. 2. 9). Abundances of Errantia differed significantly among seasons (H_3= 8.67, $p <0.05$, Kruskal-Wallis) with greater abundances found in autumn and spring ($p < 0.01$, LSD, Fig. 2.9). Abundances of Sedentaria differed significantly among seasons (H_3= 66.39, $p <0.01$, Kruskal-Wallis) with greater abundances found in autumn and summer ($p < 0.05$, LSD, Fig. 2.9).

Table 2.1: Invertebrate taxa found in the Keurbooms Estuary. Season: W= winter, Sp= spring, Su= summer, A= autumn; Habitat: I= Eelgrass, II= Reno mattress; Functional feeding group: C_S= carnivore scavenger, C_ P= carnivore predator, C_Pa= parasite, H_B= herbivore browser, H_G= herbivore grazer, O_D= omnivore detritus feeder, O_De= omnivore deposit feeder, O_F= omnivore filter feeder, O_L= omnivore lignivorous, O_P= omnivore predator, O_S= omnivore scavenger. Highlighted rows represent all taxa only found in a single habitat and season or only a single sighting throughout the sampling period.

Taxa	Season	Habitat	Functional group
<u>PLATYHELMINTHES</u>			
Polycladida			
Planocera gilchristi	W, Su	I, II	C_P
<u>NEMERTEA</u>			
White nemertean	A	I	O_D
<u>ANNELIDA</u>			
<u>Polychaeta</u>			
Sedentaria			
Prionospio sp.	W, Sp, Su, A	I, II	O_De
Capitella sp.	W, Sp, Su, A	I, II	O_De
Desdemona sp.	A	II	O_F
Ficopomatus enigmaticus	W	II	O_F
Errantia			
Glycera sp.	W, Sp, Su, A	I, II	C_P
Simplisetia sp.	W, Su, A	I, II	O_De
<u>Clitellata</u>			
Hirudinea			
Pontobdella	W	II	C_Pa
<u>ARTHROPODS</u>			
<u>Insecta</u>			
Chironomid	Su, A	II	O_D
<u>Crustacea</u>			
Thecostraca			
Amphibalanus amphitrite	W, Sp, Su	II	O_F
Eumalacostraca			

Isopoda			
Paridotea ungulata	Sp, A	I, II	H_B
Cyathura sp.	W, Sp	II	C_P
Sphaeromatoids	W, Sp	I, II	O_L
Cumacea	W, A, Sp	II	O_F
Amphipoda			
Melita sp.	W, Sp, Su, A	I, II	O_D
Victoriopisa sp.	W	I	O_De
Monocorophium acherusicum	W, Su	I	O_F
Decapoda			
Upogebia africana	W, Su, A	I, II	O_F
Hymenosoma orbiculare	W, Sp, Su, A	I, II	C_P
Diogenes brevirostris	Su	II	O_S
MOLLUSCA			
Bivalvia			
Pteriomorphia			
Arcuatula capensis	W, Sp, Su, A	I, II	O_F
Mytilus galloprovincialis	W, Su, Sp	II	O_F
Heterodonta			
Macoma litoralis	W, Sp, Su, A	I, II	O_F
Lasaea adansoni	W, Su, Sp	I, II	O_F
Gastropoda			
Heterobranchia			
Siphonaria sp.	Sp	II	H_G
Elysia sp.	W	II	H_G
Euthyneura			
Haminoea alfredensis	W, Su, A	I, II	H_G
Bursatella leachii	W	II	H_G
Godiva quadricolor	Sp	I	C_P
Favorinus ghanensis	Sp	II	C_P
Philine aperta	W	II	C_P
Caenogastropoda			
Thiaridae	W, Sp	I, II	O_S
Nassarius kraussianus	W, Sp, Su, A	I, II	C_S

Hydrobia knysnaensis	W, Sp, Su, A	I, II	H_G
ECHINODERMS			
Asteroidea			
Parvulastra exigua	W	II	C_P
Echinoidea			
Euechinoidea			
Parechinus angulosus	W	II	H_G

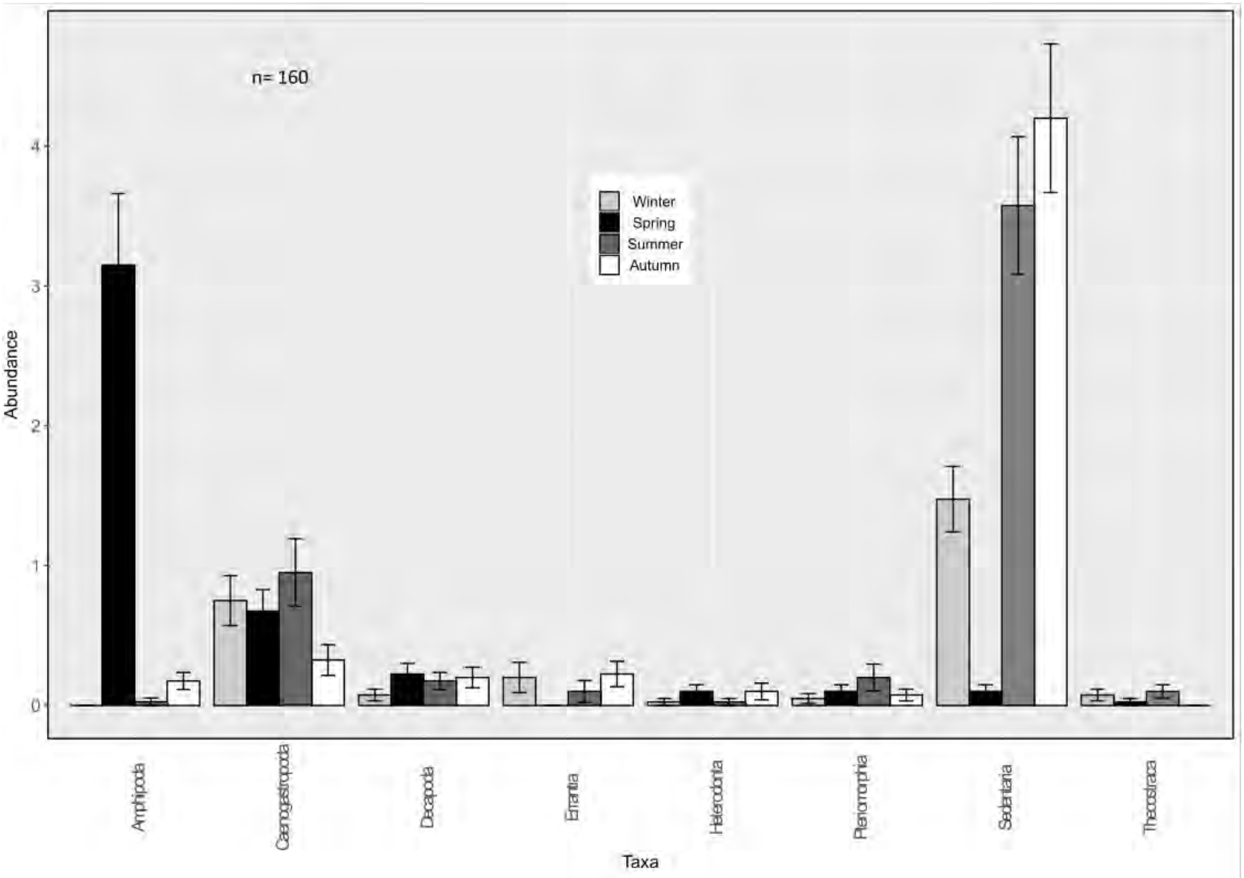

Figure 2.9. Mean (±se) abundances of invertebrates across seasons at Habitat II (Reno mattress) recorded in the Keurbooms Estuary.

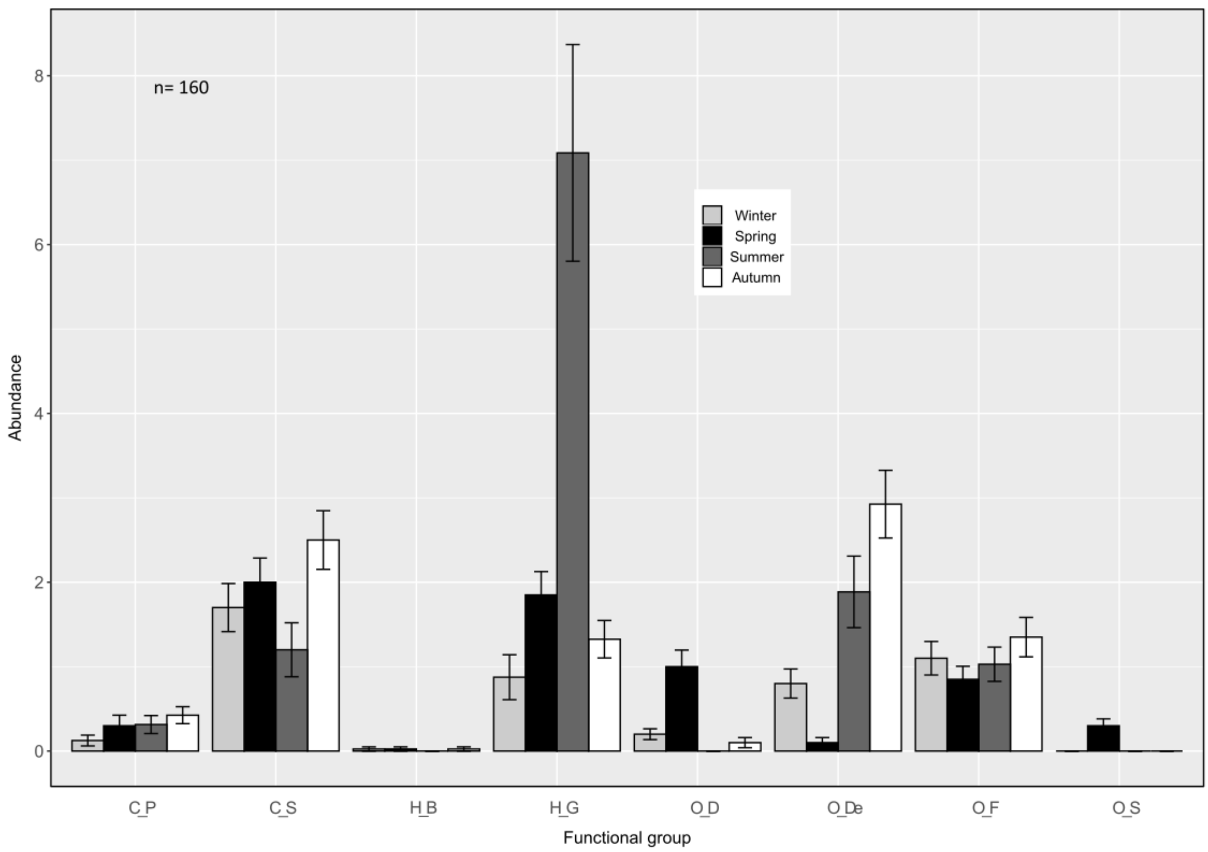

Figure 2.10. Mean (±se) abundances of invertebrate Functional groups (C_P: carnivore predator, C_S: carnivore scavenger, H_B: herbivore browser, H_G: herbivore grazer, O_D: omnivore detritivore, O_De: omnivore deposit feeder, O_F: omnivore filter feeder, and O_S: omnivore scavenger) across seasons at Habitat I (Eelgrass) recorded in the Keurbooms Estuary.

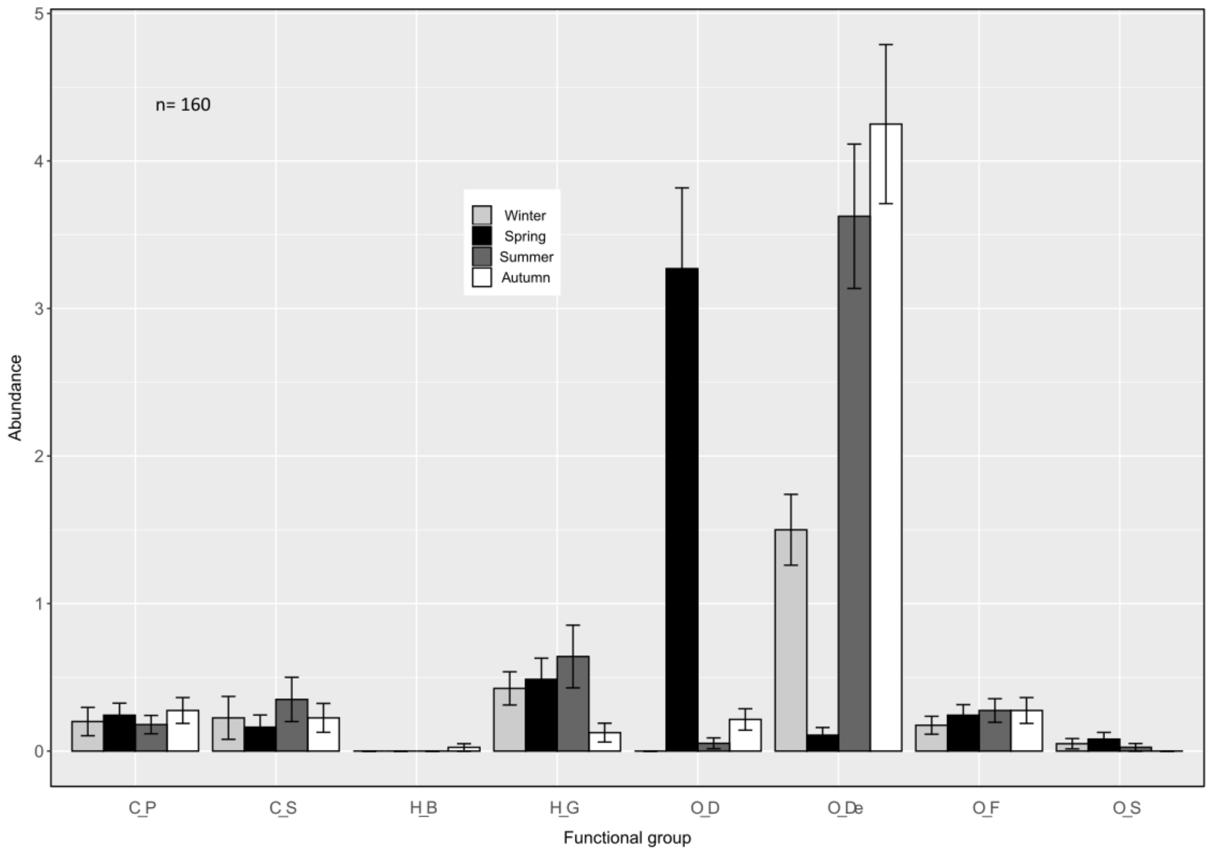

Figure 2.11. Mean (±se) abundances of invertebrate Functional groups (C_P: carnivore predator, C_S: carnivore scavenger, H_B: herbivore browser, H_G: herbivore grazer, O_D: omnivore detritivore, O_De: omnivore deposit feeder, O_F: omnivore filter feeder, and O_S: omnivore scavenger) across seasons at Habitat II (Reno mattress) recorded in the Keurbooms Estuary.

Of the nine invertebrate functional feeding groups identified, eight were found to overlap between the two habitats (Table 2.1). Within Habitat I abundances of the functional feeding group carnivore scavenger (C_S) differed significantly among seasons (H_3= 10.08, $p < 0.05$, Kruskal-Wallis) with greater abundances found in summer compared to autumn ($p < 0.01$, LSD, Fig. 2.10). Abundances of the functional feeding group herbivore grazer (H_G) differed significantly among seasons (H_3= 56.25, $p < 0.01$, Kruskal-Wallis) with the greatest abundances recorded in summer ($p < 0.01$, LSD, Fig. 2.10). Abundances of the functional feeding group omnivore detritus feeder (O_D) differed significantly among seasons

(H_3=42.70, $p < 0.01$, Kruskal-Wallis) with greatest abundances found in spring ($p < 0.01$, LSD, Fig. 2.10). Abundances of the functional feeding group omnivore deposit feeder (O_De) differed significantly among seasons (H_3= 57.72, $p < 0.01$, Kruskal-Wallis) with the fewest number of animals found in spring ($p < 0.05$, LSD, Fig. 2.10). The abundances of the functional feeding group omnivore scavenger (O_S) differed significantly among seasons (H_3= 33.81, $p < 0.01$, Kruskal-Wallis) with the greatest abundances recorded in spring ($p <$ 0.01, LSD, Fig. 2.10).

Within Habitat II the abundances of the functional feeding groups omnivore detritus feeder (O_D) and differed significantly among seasons (H_3= 69.54, $p < 0.01$, Kruskal-Wallis) with the greatest abundances found in spring ($p < 0.01$, LSD, Fig. 2.11). Abundances of the functional feeding group omnivore deposit feeder (O_De) differed significantly among seasons (H_3= 63.04, $p < 0.01$, Kruskal-Wallis) with greatest abundances found in autumn and summer and the fewest animals found in spring ($p < 0.01$, LSD, Fig. 2.11).

2.3.2 Comparison of fish assemblages

In total, eighteen taxa of fish were identified, with a 72.22% overlap between habitats. Three species were only recorded in Habitat II (*Caffrogobius caffer*, *Chaetodon marleyi*, and *Lutjanus fulviflamma*) and one species of Ophichthidae was only recorded in Habitat I (Table 2.4). Species richness (W= 287.5, $p < 0.01$, Wilcox, Fig. 2.12) and diversity (W= 209, $p < 0.01$, Wilcox, Table 2.5) were significantly greater within Habitat II. There was a significant increase in the species richness from winter through summer and a decrease in autumn (H_3= 32.14, $p < 0.01$, Kruskal-Wallis; Fig. 2.12). Eight species were found to dominate the abundances of both habitats although greater abundances for most of these species were found in Habitat II (Fig. 2.13).

Table 2.2. Fish taxa recorded in the Keurbooms estuary by the mini BRUVs within Habitat I (Eelgrass) and Habitat II (Reno mattress) for each season (W= winter, Sp= spring, Su= summer, A= autumn). Estuary dependency category according to Wallace *et al.* 1984 (Category I- species completely estuarine dependent for their entire life cycle, Category II- species dependent on estuaries during only their juvenile stage, Category III- species whose juveniles occur mainly in estuaries but are also found at sea, Category IV- species whose juveniles mainly occur at sea but are abundant in estuaries, Category V- species whose juveniles occur at sea but sometimes are found into estuaries). Functional feeding group (C= carnivorous, O= omnivorous, P= piscivorous, H= herbivorous, Z= zoobenthic predator, D= detritivorous). Highlighted rows represent all taxa only found in a single habitat and season or only a single sighting.

Taxa	Season	Habitat	Estuary dependence category	Functional group
ACTINOPTERYGII				
TELEOSTEI				
Ophichthidae				
species 1	Su	I	III	C
Ariidae				
***Galeichthys feliceps* (Valenciennes 1840)**	W, Sp, Su, A	I, II	IV	C
Syngnathidae				
***Syngnathus temminckii* (Kaup 1856)**	W, Sp	I, II	IV	C
Haemulidae				
***Pomadasys commersonnii* (Lacepède 1810)**	Sp, Su, A	I, II	II	Z
Lutjanidae				
***Lutjanus fulviflamma* (Forsskal 1775)**	A	II	IV	C
Sparidae				
Diplodus capensis	Su, A	I, II	IV	O

(Smith 1844)				
Diplodus hottentotus	Su, A	I, II	V	C
(Smith 1844)				
Lithognathus lithognathus	Sp, Su, A	I, II	II	Z
(Curvier 1829)				
Rhabdosargus sp.	W, Sp, Su, A	I, II	II	H (juvenile)
(Steindachner 1881)				C (adult)
Sarpa salpa	Su, A	I, II	IV	C (juvenile)
(Linnaeus 1758)				H (adult)
Monodactylidae				
Monodactylus falciformis	Sp, Su, A	I, II	II	H
(Lacepède 1800)				
Chaetodontidae				
Chaetodon marleyi	Su	II	V	O
(Regan 1921)				
Carangidae				
Lichia amia	Su, A	I, II	II	P
(Linnaeus 1758)				
Mugilidae				
species	W, Sp, Su, A	I, II	II-IV	D
Clinidae				
species	W, Sp, Su, A	I, II	I-V	C
Gobiidae				
Psammogobius knysnaensis	W, Sp, Su, A	I, II	I	C
(Smith 1936)				
Caffrogobius nudiceps	W, Sp, Su	I, II	IV	C
(Valenciennes 1837)				
Caffrogobius caffer	W, Sp, Su, A	II		C
(Günther 1874)				

704

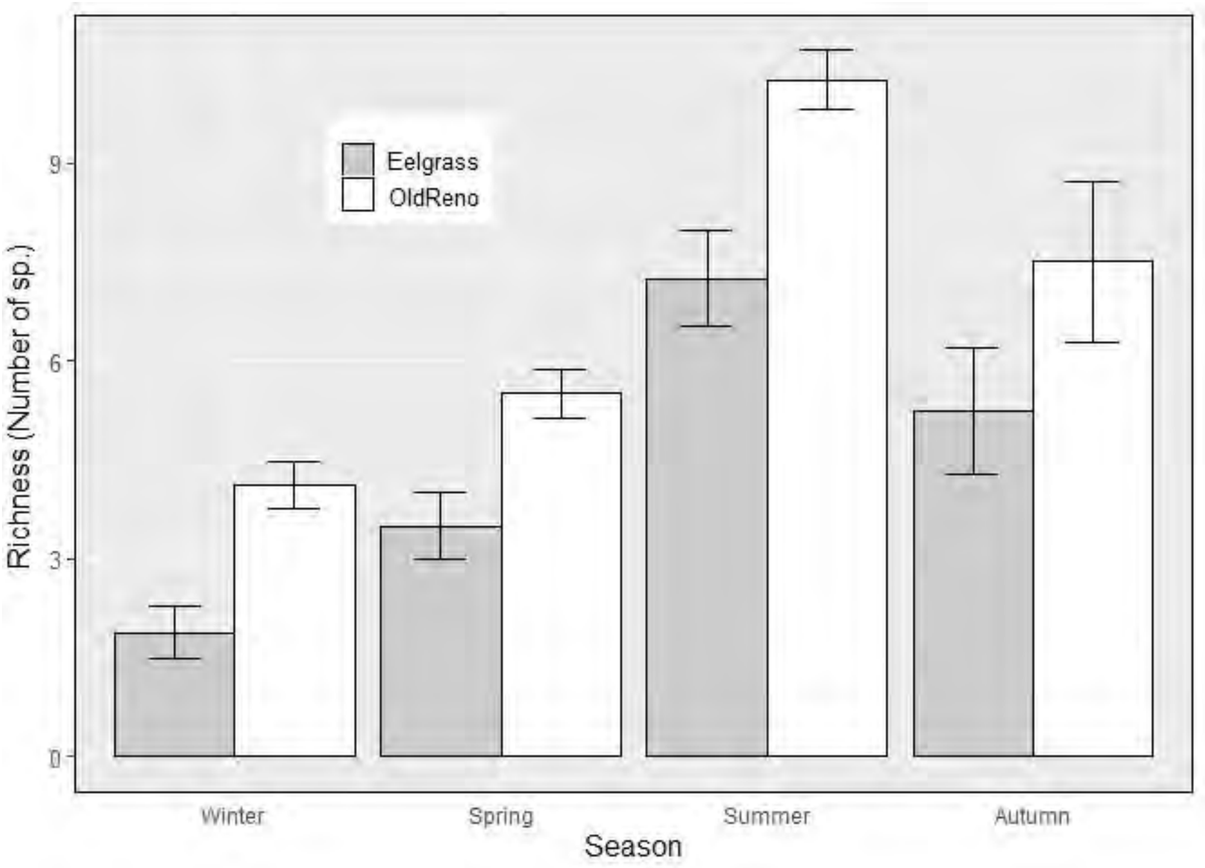

Figure 2.12. Mean (± se) species richness of fish across seasons within Habitat I (Eelgrass) and II (Reno mattress) recorded in the Keurbooms Estuary.

Table 2.3. Mean (± se) Shannon-Weiner diversity index (H') of fishes in Habitat I (Eelgrass) and II (Reno mattress).

Habitat	**Mean (± se) Shannon-Weiner diversity index (H')**
Eelgrass	0.95 (± 0.09)
Reno mattress	1.43 (±0.07)

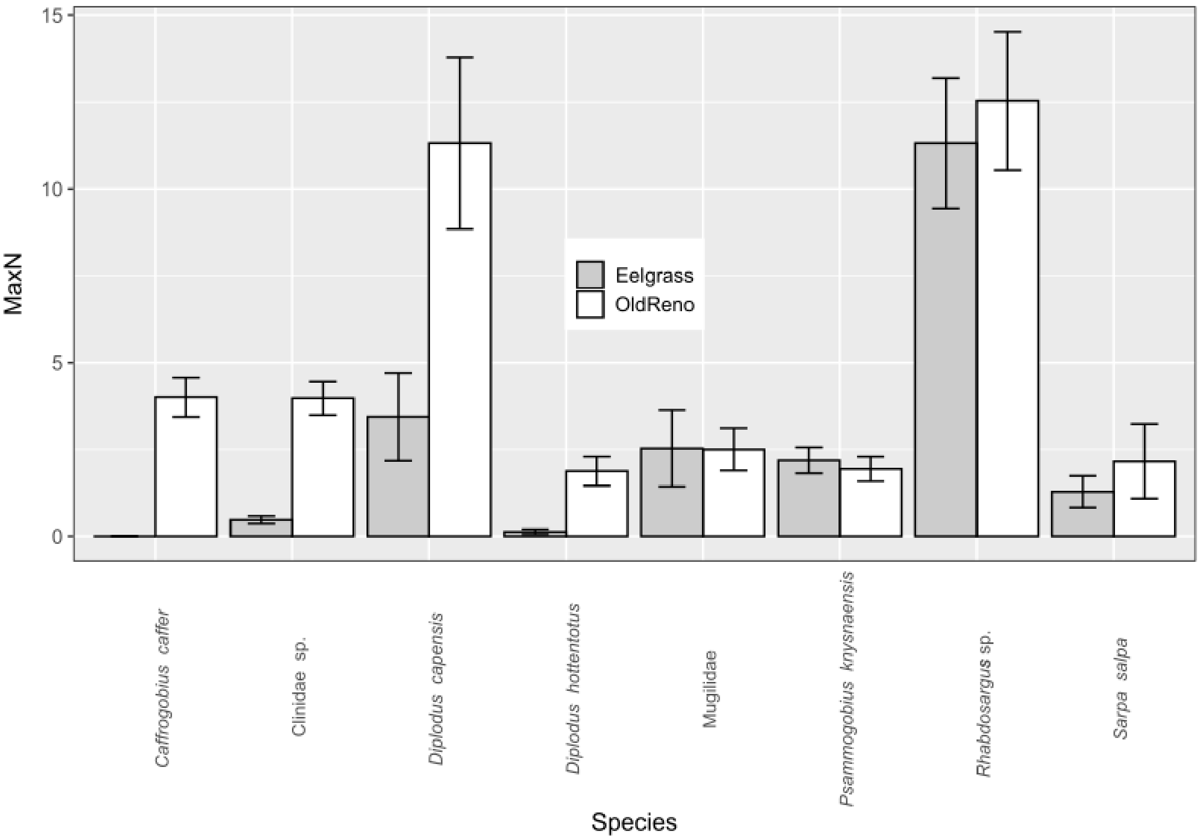

Figure 2.13. Mean (± se) MaxN (maximum number of any individual fish species in any one frame) of dominant fish species across seasons within Habitat I (Eelgrass) and II (Reno mattress) recorded in the Keurbooms Estuary.

Fish species assemblages were significantly different between habitats and across seasons ($p < 0.01$, PERMANOVA, Table 2.6) with no significant effect from the interaction between season and habitat. The factors habitat and season explained 54.5% of the variation observed and the separation between habitats was strongly correlated to the species *Caffrogobius caffer* and Clinidae spp. (Fig. 2.14).

Table 2.4. Fish and functional feeding group PERMANOVA results showing differences between habitats, seasons, and the interaction of both.

Variables	**df**	**SS**	**MS**	**Pseudo-F**	**P (perm)**	**Unique perms**
Animals						
Habitat	1	17218	17218	19	0.0001****	9950
Season	3	52882	17627	19.45	0.0001****	9921
SeasonXHabitat	3	4344.6	1448.2	1.60	0.0731	9927
Residual	53	48027	906.18			
Total	60	1.22e5				
Functional Groups						
Habitat	1	4331.3	4331.3	8.78	0.0003****	9956
Season	3	38599	12866	26.09	0.0001****	9946
SeasonXHabitat	3	6120.2	2040.1	4.14	0.001***	9938
Residual	53	26134	493.1			
Total	60	74679				

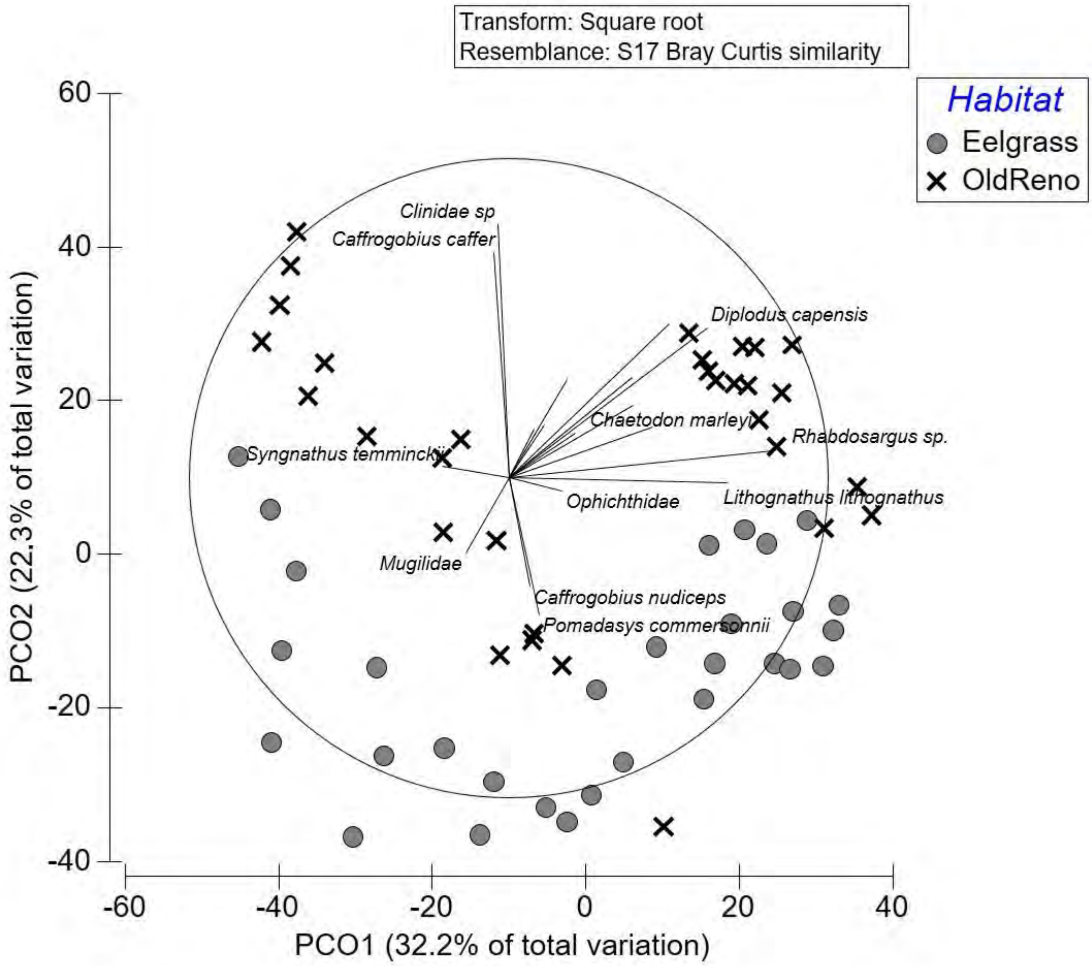

Figure 2.14. Principal coordinate analysis to visualise patterns of fish assemblages between Habitats I (●) and II (X) with species overlay.

Pairwise tests revealed that significant differences in fish assemblages and abundances were found between Habitat I and II ($t = 4.359$, $p < 0.01$, PERMANOVA). All seasons were found to differ significantly within each habitat type ($p < 0.01$, PERMANOVA). The CAP analysis showed that differences between habitats showed a strong correlation to *Caffrogobius caffer* and Clinidae spp. (Fig. 2.15).

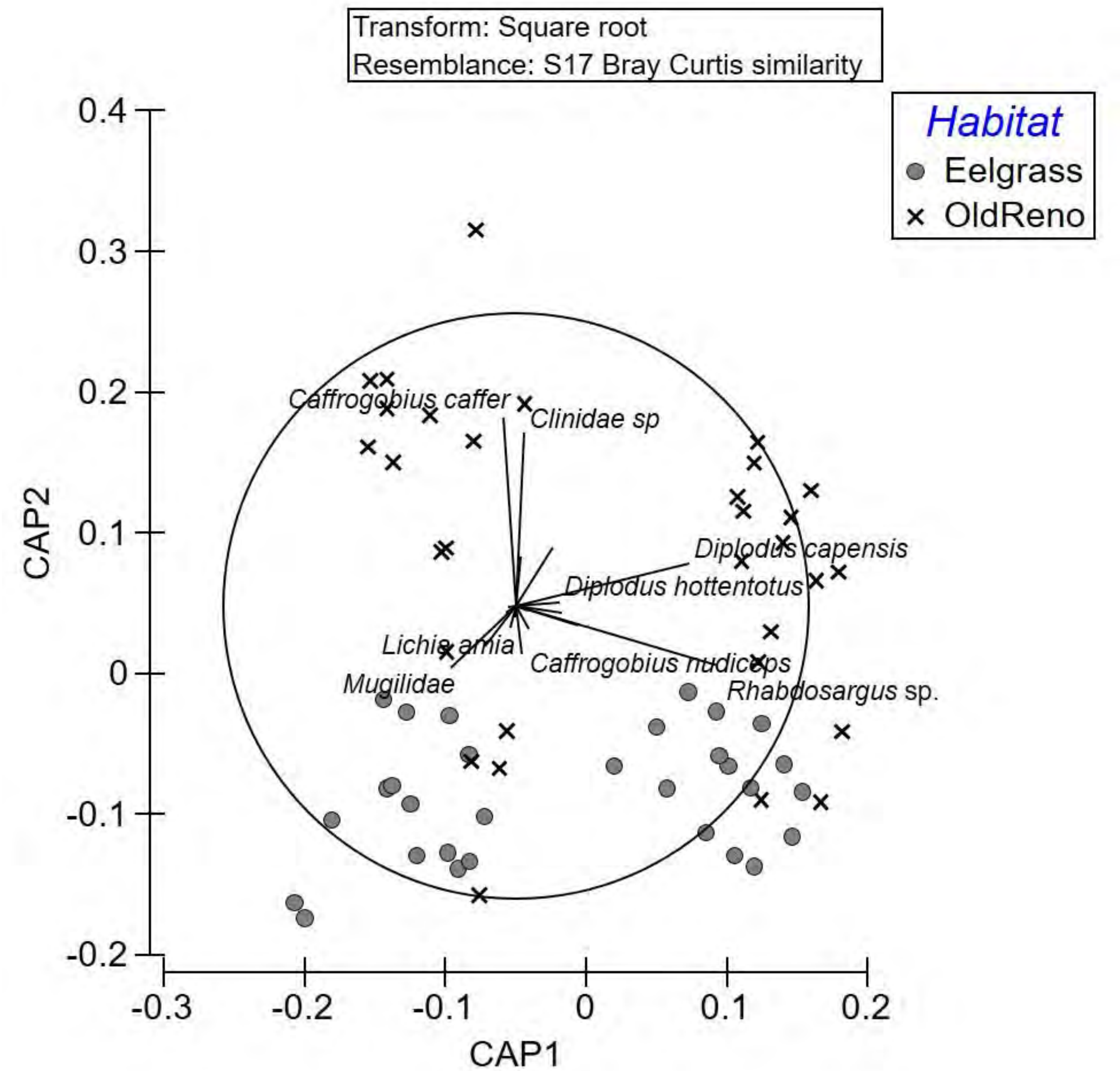

Figure 2.15. Canonical analysis of principal coordinates analysis to visualise taxa (Clinidae sp., and *Caffrogobius caffer*) that drive the differences between Habitats I (●) and II (X).

Six functional fish groups were identified, and 100% overlap was found between Habitat I and II. Four of these functional feeding groups dominated the abundances of fish in both habitats (Fig. 2.16).

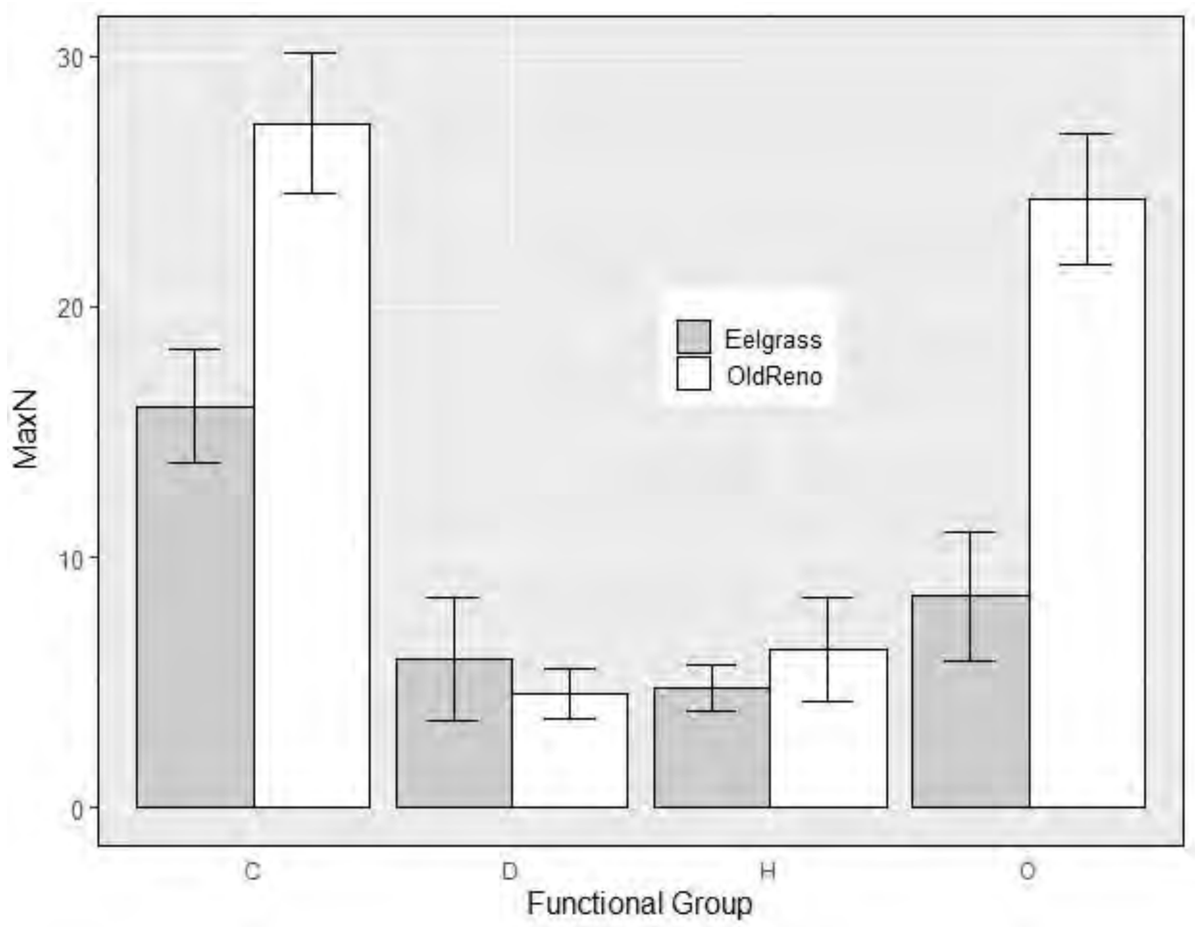

Figure 2.16. Mean (±se) MaxN of dominant functional feeding groups found within Habitat I and II. Functional feeding groups: C= carnivorous, D= detritivorous, H= herbivorous, O= omnivorous recorded in the Keurbooms Estuary.

The assemblages of fish functional feeding groups were significantly different between habitats and across seasons ($p < 0.01$, PERMANOVA, Table 2.6). The factors habitat and season explained 76.7% of the variation observed and the divide between habitats was driven by the strong correlation between PCO axes 1 and the functional group detritivores (Fig. 2.17), where the functional feeding group detritivores was the only group to have greater abundances within Habitat I.

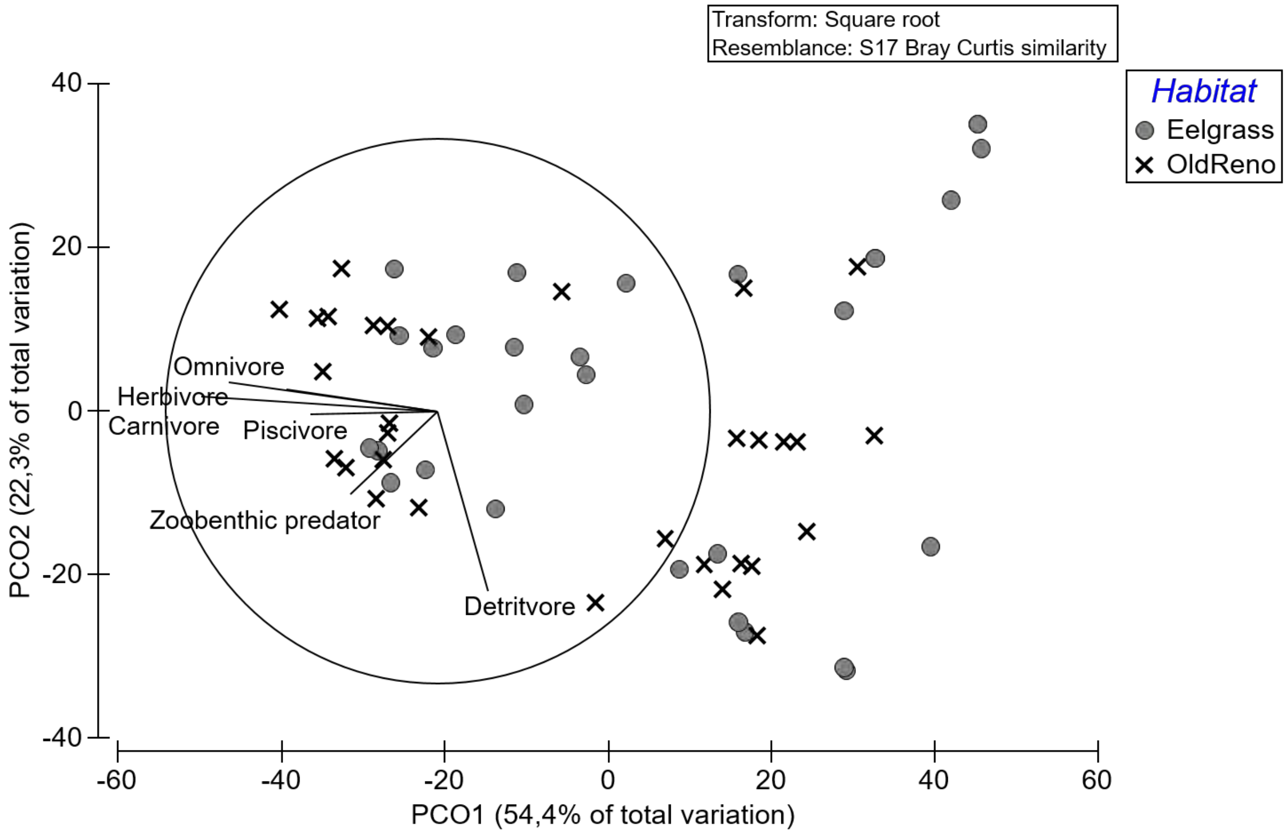

Figure 2.17. Principal coordinate analysis visualisation of functional feeding group assemblages between Habitat I (●) and II (X).

Pairwise tests revealed that functional feeding groups differed significantly between Habitat I and II (t= 2.96, $p < 0.01$, PERMANOVA). All seasons differed significantly ($p < 0.01$, PERMANOVA).The CAP analysis, through minimising the residual variation, showed the differences between habitats had a strong positive correlation to the functional feeding group detritivorous (Fig. 2.18).

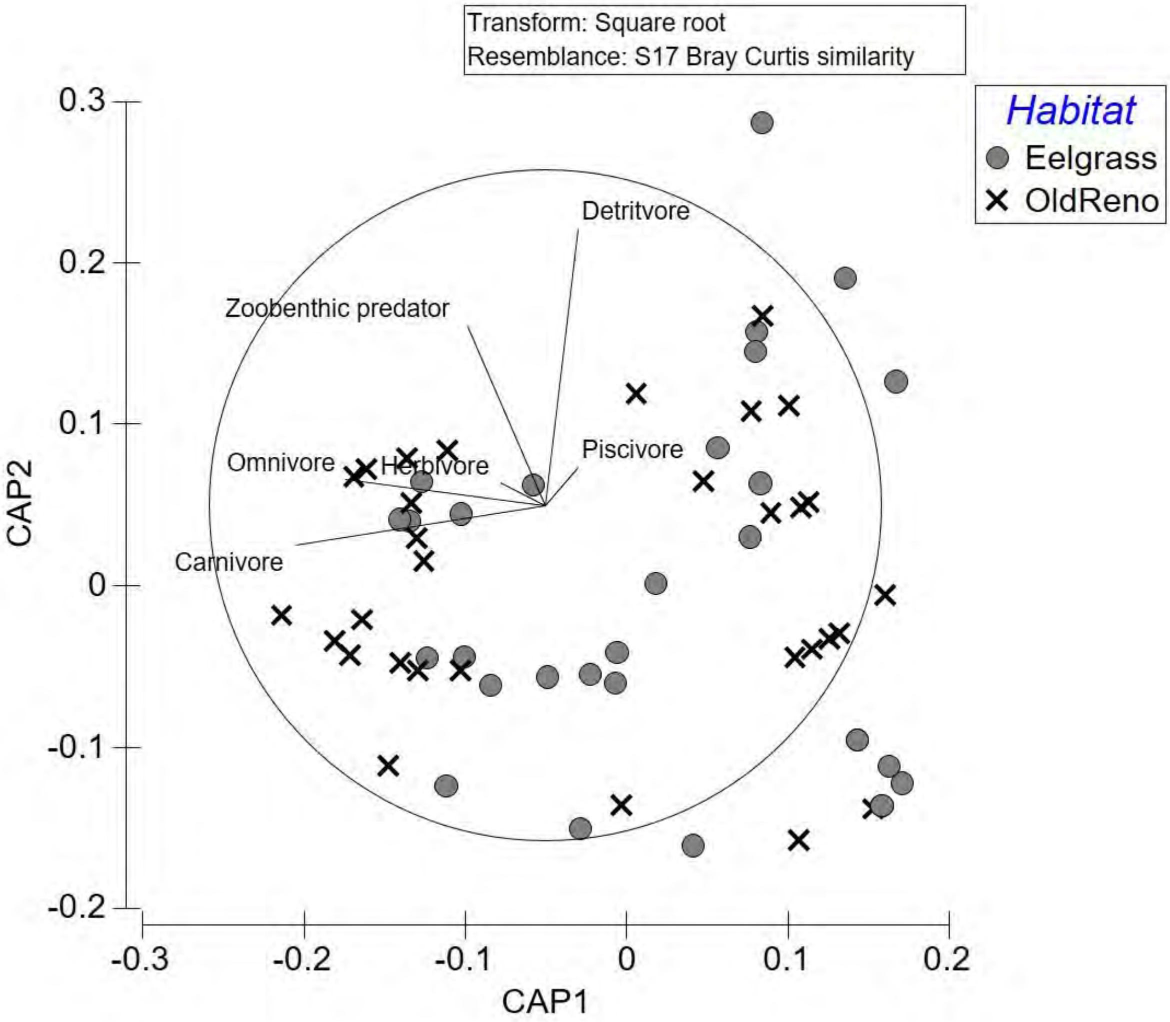

Figure 2.18. Canonical analysis of principal coordinates visualisation of functional feeding group assemblages between Habitats I (●) and II (X).

2.4 Discussion

Understanding how species assemblages change owing to habitat alteration is essential in predicting how the implementation of artificial structures in coastal systems can influence species diversity (Airoldi *et al*., 2005). The present study compared the faunal assemblages between neighbouring natural (eelgrass) and artificial (Reno mattress) habitats in the Keurbooms Estuary and found clear differences in the abundance and diversity of fish. Although, invertebrate diversity and abundances could not be statistically compared some differences in the types of taxa were noted between habitats. There was some overlap of invertebrate taxa between habitats, however, this study noted the presence of taxa only within either habitat. For example the presence of Desdemona and *Ficopomatus enigmaticus* only in Habitat II.

The Keurbooms Estuary is characterised as having a high macroinvertebrate diversity, especially within vegetated areas (Duvenage & Morant, 1984; Bornman & Adam, 2005; Enviro-Fish Africa, 2010). This was confirmed by this study as a total of 36 invertebrate taxa were found within the natural eelgrass habitat and the Reno mattress. Of the 21 benthic taxa found within eelgrass beds, 76.2% were the same as those recorded in the nearby Knysna Estuary (Barnes, 2010; 2013) and are typical of southern Cape estuaries. Of the 42 species recorded in the previous survey carried out by Day (unpublished data) in Keurbooms Estuary eight species were found in this study.

Abundances of invertebrates in both eelgrass and Reno mattress habitats were significantly different among seasons, with greater numbers recorded in spring and summer. This may be a result of the presence of new recruits of some species, as summer is when many South African warm temperate marine invertebrates reproduce (Hodgson, 2010). The variability of estuarine fauna is often thought to be due to the variability of estuaries themselves with little focus on seasonal changes (Akin *et al*., 2003; Hirst & Kilpatrick, 2007; Xu *et al*., 2016). Few studies have focused on the seasonal changes in invertebrate numbers recorded in South African estuaries (Puttick, 1977; Kalejta & Hockey, 1991; Hodgson, 2010), and the Keurbooms Estuary has little information regarding seasonality of invertebrate abundances. The study carried out by Kalejta & Hockey (1991) in the Berg River Estuary found that the peak in the biomass of invertebrates occurred during austral winter, spring and autumn was attributed to the presence of juveniles, indicating recruitment increases the abundance of invertebrates. These seasonal differences may be influenced by seasonal fluctuations in temperature and salinity that many South African estuaries experience (de Villiers *et al*., 1999). Declines in salinity are evident within the Keurbooms Estuary during rainfall peaks in austral autumn and spring (Bornman & Adams, 2005) as this leads to increased freshwater input. Seasonal fluctuations in invertebrates have also been linked to the growing season of *Z. capensis* beds whereby a greater abundance invertebrates may utilize this habitat as seen in the Berg River Estuary (Kalejta & Hockey, 1991) and in *Palaemon pacificus* within three Eastern Cape estuaries (Emmerson, 1986).

The high abundance of the functional feeding group herbivorous grazers in the eelgrass habitat suggest that this habitat is dominated by animals that perform strong top-down influences on plant assemblages and provide links to species in higher trophic levels (Orth & van Montfrans, 1984), as well as those scavenging animals that perform nutrient cycling. By contrast, the high abundance of omnivorous detritus and deposit feeders within Reno mattress indicates that a greater number of animals perform the function of detritus and deposit cycling in this habitat. Detritus represents a primary food resource that can attract diverse assemblages of detritivores, especially in soft sediment habitats (Krumhansl &

Scheibling, 2012), and is usually less evident in habitats with artificial structures (Heery *et al*., 2017). However, unlike many artificial structures, Reno mattress within the Keurbooms Estuary has become silted up again allowing for the development of a hybrid type of habitat. Heery & Sebens (2018) showed that artificial structures can become sinks for detritus which may explain the dominating detritus feeding group within Reno mattress. As the Keurbooms Estuary is susceptible to flooding (Bornman & Adams, 2005) it is possible that any sediment accumulated within the Reno mattress may be removed or more deposited further down on the Reno mattress potentially altering the habitat.

The introduction of artificial structures can lead to the loss of soft sediments, thus altering community structure (Martin *et al*., 2005). Hard artificial structures can support similar assemblages of invertebrates, however, they are usually less diverse compared to natural habitats(Connell & Glasby, 1999; Firth *et al*., 2013a), which is confirmed by the results of the present study. Historically, within the Keurbooms Estuary there were no extensive rocky areas and, therefore, invertebrates typically associated with rocks were scarce (Duvenage & Morant, 1984). With an increase in bank stabilisation using hard materials, species adapted to such substrata would increase. The Reno mattress has enabled a number of species, usually those associated with rocky substrata, to appear (Table 2.1) which includes: *Mytilus galloprovincialis* (Lamark, 1819), *Amphibalanus amphitrite* (Darwin, 1854), *Siphonaria* sp., *Parvulastra exigua* (Lamark, 1816), *Parechinus angulosus* (Leske, 1778), and *Ficopomatus enigmaticus* (Fauvel, 1923). Artificial structures are also a means for invasive species to spread within a system (Bulleri & Airoldi, 2005; Airoldi & Bulleri, 2011; Dafforn, 2017) as seen by the presence of the invasive species *M. galloprovincialis* and *F. enigmaticus* within the Reno mattress habitat. The invasive Mediterranean mussel (*M. galloprovincialis)* was likely introduced into South Africa around the 1970s and has spread along the temperate coastlines of South Africa including the intertidal shorelines of the southern coast (Griffiths *et al*., 1992; Robinson *et al*., 2005; Picker & Griffiths, 2011). The 1950 accidental introduction of *F. enigmaticus* into South African waters has led to this species being found in many estuaries along the entire South African coastline (Picker & Griffiths, 2011).

Unfortunately direct statistical comparisons of invertebrate abundances between the two habitats could not be made owing to the different sampling methods used between the habitat types (suction vs core). Despite this descriptive comparisons of species assemblages within each habitat were made and therefore differences in the types of invertebrates between habitats could be seen. Within Habitat II (Reno mattress) the restriction to sample benthic invertebrates found within crevices of the structure means that some animals may have been missed during sampling. However, despite this restriction, a number of taxa were found to be represented within the Reno mattress habitat indicating this method can still be useful. In order to directly compare these two habitats, suction sampling should be carried out in both the eelgrass and the Reno mattress. Photo quadrat sampling of surface invertebrates may also provide a better method of comparing the two habitats and thus differences in types of animals present may be clearer. The structure of this habitat differed completely from the natural habitat as it had limited soft sediment and therefore the types of species would be expected to differ.

Fish taxa found in the lower reaches during this study were similar to findings from previous studies within the Keurbooms Estuary (Duvenage & Morant, 1984; Whitfield, 1994a; James & Harrison, 2011) as well as other permanently open estuaries along the southern Cape of

South Africa (Whitfield, 1994b; Whitfield 1998; Harrison & Whitfield, 2006). Of the 18 fish taxa identified in this study eight are endemic to South Africa (Whitfield, 1994a) indicating the importance of this estuary for endemic fish. A score system was developed by Wallace *et al*. (1984) which classified and scored fish based on their dependence on estuaries. In the present study most of the species of fish fell into category IV (those whose juveniles occur at sea but are abundant within estuaries). These species benefit from estuaries but do not really depend on them for their life histories, however, their presence contributes to adult stocks (Wallace *et al*., 1984). Five species recorded (*Pomadasys commersonnii*; *Lithognathus lithognathus*; *Rhabdosargus* sp.; *Monodactylidae falciformis*; *Lichia amia*) are those that are dependent on estuaries during their juvenile phase of life, indicating that the Keurbooms Estuary is an important nursery habitat for these fish as has been found for other southern coast estuaries (Wallace *et al*., 1984; Whitfield, 1994a). Of these species *P. commersonnii, L. lithognathus*, *Rhabdosargus* sp. and *L. amia* are important angling species (Wallace *et al*., 1984; Whitfield, 1994a; Mann, 2013).

The present study found that whilst the species recorded on underwater cameras within estuaries were similar to those found by Becker *et al*. (2010; 2012), a greater MaxN was recorded. The studies carried out be Becker *et al*. (2010; 2012) used small camera systems mounted to metal tripods within two Eastern Cape estuaries which may be the reason for the greater numbers of fish recorded in the present study as the sampling procedure made use of baited cameras which attracts more fish to the vicinity of the camera. The use of mini-BRUVs have been found to be more effective in documenting fish within small rock pools than visual censuses and observer operated videos (Davis *et al*., 2018).

A greater species richness, diversity, and abundance of fish were present at this habitat compared to eelgrass. High species richness, diversity and abundance of fish is a common occurrence where there are coastal artificial structures such as wharfs and breakwaters (Rilov & Benayahu, 2000; Bulleri & Chapman, 2010) as well as within marinas (Clynick, 2007). The most abundant functional feeding groups within Habitat II were the carnivorous and omnivorous fish indicating that there must be adequate prey for fish to feed on or in the Reno mattress habitat. Fish may feed off sessile animals or algae on artificial structures (Rilov & Benayahu, 2000) which provide a suitable substratum for invertebrates to live on (Connell & Glasby, 1999; Spieler *et al*., 2001; Bulleri, 2005).

Reno mattress differs from most commonly used erosion control structures as it has crevices between rocks allowing for animals to live within it. The installation of this three-dimensional artificial structure may provide new shelter and protection for specific fish species (Spieler *et al*., 2001; Clynick *et al*., 2008), especially for crypto-benthic species present within a system (Ushiama *et al*., 2019). Differences of fish assemblages between sandy bottoms and artificial habitats have been found to be generally driven by benthic taxa (Franzitta, 2013) such as the Gobiidae and Clinidae species found in this study. Benthic and cryptic fish species aggregating at artificial habitats are often found in crevices using them for shelter and resources (Pérez-Ruzafa *et al*., 2006). Artificial structures have also been likened to natural rocky shore habitats in terms of attraction by fish (Rilov & Benayahu, 2000; Bulleri, 2005; Clynick *et al*., 2008). The present study found fish species usually associated with rocky shores were attracted to the artificial structure e.g. *Diplodus hottentotus* (Zebra fish), *Diplodus capensis* (Blacktail), and *Sarpa salpa* (Strepie). Although there was an overlap of fish species between the two habitats types, one species in particular was consistently only found with the artificial habitat- *Caffrogobius caffer*

(Gobiidae). Normally abundant within intertidal rock pools (Butler, 1982) between False Bay to Umgazi along the South African coast, the endemic goby *C. caffer* has been noted in a few estuaries for example the Kariega Estuary (Paterson & Whitfield, 2000) and previously within the Keurbooms Estuary (Duvenage & Morant, 1984). These fish are primarily restricted to the intertidal zone and therefore it is interesting that this a species from high shore rock pools is found utilising the subtidal Reno mattress of this habitat.

Single sightings of two other species (*Chaetodon marleyi* doublesash butterflyfish, and *Lutjanus fulviflamma* dory snapper) were noted within Habitat II. *Chaetodon marleyi* is often associated with rocky or coral reefs, however, juveniles have been noted to enter estuaries (van der Elst, 1988). This species was also noted in the Reno mattress habitat of Thesen Islands Marina in the Knysna Estuary (Taylor, 2018). *Lutjanus fulviflamma* is widely distributed throughout the Indo-Pacific and occurs mainly in reefs and rocky substrata (van der Elst, 1988; Whitfield, 1998). Along the South African coastline juveniles of this species have been noted in estuaries as south as Swartkops (Whitfield, 1998). Artificial reefs have been suggested as a potential tool for restoration and rehabilitation as they can attract fish and invertebrate assemblages and increase diversity (Spieler *et al*., 2001; Sherman *et al*., 2002; Perkol-Finkel *et al*., 2006). The presence of these two reef fish within the Reno mattress habitat therefore means that this structure may act as an artificial reef and their presence may be viewed as enhancement of local species richness. Whether this is a positive result of Reno mattress depends on the context of a system i.e. the local characteristics of system.

There is a novelty of using mini-BRUVs within an estuarine environment as few studies have made use of this technique (Gladstone *et al*., 2012). Some studies within South Africa have used underwater cameras within estuaries for sampling fish (Becker *et al*., 2010; 2012; Pollard *et al*., 2018), however, the use of BRUVs is infrequent. The use of bait is useful in mitigating constraints of sampling in water with low visibility (Gladstone *et al*., 2012) which is the case in most estuaries. In Australia, BRUVs have been effectively used within estuaries to compare fish assemblages between habitats (Gladstone *et al*., 2012; Morton & Gladstone, 2014) and differing protection zones (McKinley *et al*., 2011). Although BRUVs can provide sampling without disturbances to fish, they have their own limitations and restrictions. Species identification can be difficult and sometimes impossible (Becker *et al*., 2012), particularly for the Mugilidae species in this study. The field of view can be obstructed by vegetation, as noted in the study carried out by Becker *et al*. (2012) in the Bushmans Estuary (Eastern Cape of South Africa). Baiting may attract more fish to the area and thus may be considered a bias towards carnivores and scavenger fish, however, Harvey *et al*. (2007) found that while this was true it did not decrease the numbers of herbivores or omnivores. Watson *et al*. (2005) found baiting increased the ability to discriminate between assemblages.

It is important to note that adjacent to the artificial habitat were beds of seagrass and, therefore, Reno mattress can be considered additional habitat within the Keurbooms Estuary. This combination of both natural eelgrass and hard artificial Reno mattress is an important feature of this system. The demonstration that there are significant differences of abundances and assemblages of fish between artificial and natural habitats raises the question: as development on coastlines is only expected to increase (Morris *et al*., 2018) what is the impact of installing more of these structures in an estuary? Chapter 3 aims to investigate the ecological impact a newly installed section of Reno mattress has on the faunal assemblages and diversity on the eastern bank of the eastern channel in the Keurbooms estuary.

Chapter 3

Determining the potential impact of Reno-mattress construction on fauna using a Before, After, Control, Impact (BACI) approach

3.1 Introduction

Assessing the impacts from disturbance humans have on the environment is of increasing concern (Underwood, 1991; Chapman & Underwood, 2011; Chevalier *et al*., 2019), and assessments of estuarine and coastal ecological health are crucial in guiding management (Teichert *et al*., 2018; Chevalier *et al*., 2019). Estuaries in particular are increasingly threatened by anthropogenic activities (Blaber *et al*., 2000; Turpie *et al*., 2002; Waltham & Connolly, 2007; Gittman *et al*., 2016a), and as a means to assess these disturbances benthic invertebrates and fish communities are often used as indicators of ecological status (Teichert *et al*., 2018). Restoration of heavily eroded and degraded estuarine environments often involve using hard bank stabilisation methods such as artificial structures (Dugan *et al*., 2011; Elliott *et al*., 2016; Heery *et al*., 2017).

Prior to the authorisation of any development it is usual, in some countries but not all, to undertake some kind of environmental impact assessment (EIA). However, the effectiveness of these assessments can vary from country to country, especially in developing countries (Wood, 2003). There is currently no accepted standardised criteria for evaluating the ecological impact of implemented artificial structures (Perkins *et al*., 2015; Chapman *et al*., 2018). Therefore, proper evaluation of impacts are required to best manage and protect coastal environments (Chevalier *et al*., 2019). Very few studies are carried out before the development of coastal structures and urbanisation (Elliott *et al*., 2016), and even rarer is an assessment of the impact on coastal areas particularly within a South African context. Before-After-Control-Impact (BACI) designs are recognized as a robust approach for documenting environmental impacts (Underwood, 1991; 1992; Smith *et al*., 1993).

There is still limited knowledge on the effects of coastal defence structures on fish communities (Morris *et al*., 2018; Franzitta & Airoldi, 2019). Effective monitoring to assess what impact a specific development has had is lacking, and therefore there is little data to inform future coastal development decisions and policies. The Beyond BACI approach (Underwood, 1991; 1992) acknowledges the importance of temporal and spatial aspects into the design of an assessment to limit confounding factors (Roberts *et al*., 1998). Therefore, an impacted site needs to be compared to essential reference control sites, where the impact does not occur (Green, 1979; Underwood, 1991; Heery *et al*., 2018; Morris *et al*., 2019). These sites need to be studied both prior to, during, and after the specific impact being assessed as they can provide helpful insights into the expected natural variability (Bernstein & Zalinski, 1983; Stewart-Oaten *et al*., 1986; Morris *et al*., 2019).

The present study is the first to investigate the impact of the deployment of Reno mattresses to combat erosion within the Keurbooms Estuary using a BACI approach. The aim was to determine the ecological impact of newly installed Reno mattresses within this estuary by investigating the impact on fish and invertebrate diversity, and abundance. It is hypothesized that the area, where new Reno mattress is placed, would become more similar in species composition to the habitat where Reno mattress has been present for a period of time, compared to natural eelgrass habitat. This study aims to provide impact data to assist future management of these installations and thus help in future erosion control decisions.

3.2 Methods and Materials

3.2.1 Study site

Seventy metres of Reno mattress structures were installed on the eastern bank of the eastern channel of the Keurbooms Estuary in November 2018, with the installation completed in April 2019 by a homeowner of a residential estate. The Reno mattresses were positioned on bare sediment, landward and adjacent to subtidal *Zostera capensis* beds. This meant that the Reno mattresses were intertidal. Prior to the installation process photographs of the erosion damage and state of the eastern estuarine bank were taken (Appendix A). The process of installation included the deployment of back-fill (rocks, rubble, and sediment) onto the eroded bank and geotextile after which the Reno mattress wire box structures were placed on top and filled with rocks. To limit the impact of increased turbidity on adjacent eelgrass, vertical shade cloth barriers were placed at the edge of the Reno mattress during the installation period. Photographs were taken once the installation was complete (Appendix A). This section of newly developed artificial structure was categorised as a third habitat type (Impacted) within the larger study site. Two control sites, one of subtidal eelgrass (Control_Eel) and one of Reno mattress older than 3 years (Control_Reno) (Habitats I and II as discussed in Chapter 2) were chosen in the same region of Keurbooms Estuary (Fig. 3.1). Within each habitat type, four sample locations were selected at least 30 m apart to allow for independence of sampling.

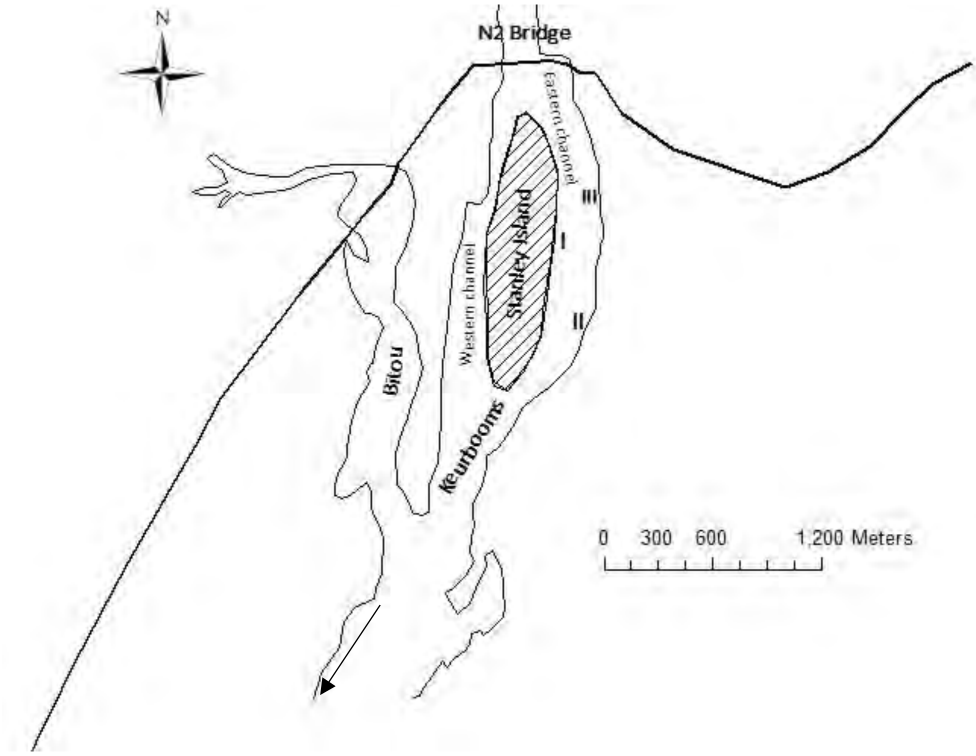

Figure 3.1. The Keurbooms estuary showing the location of the three different habitats studied: I-Control_Eel, II- Control_Reno (older Reno mattress), and III- Impacted (newly installed Reno mattress). Black line indicates N2 road. Black arrow indicates direction to the ocean.

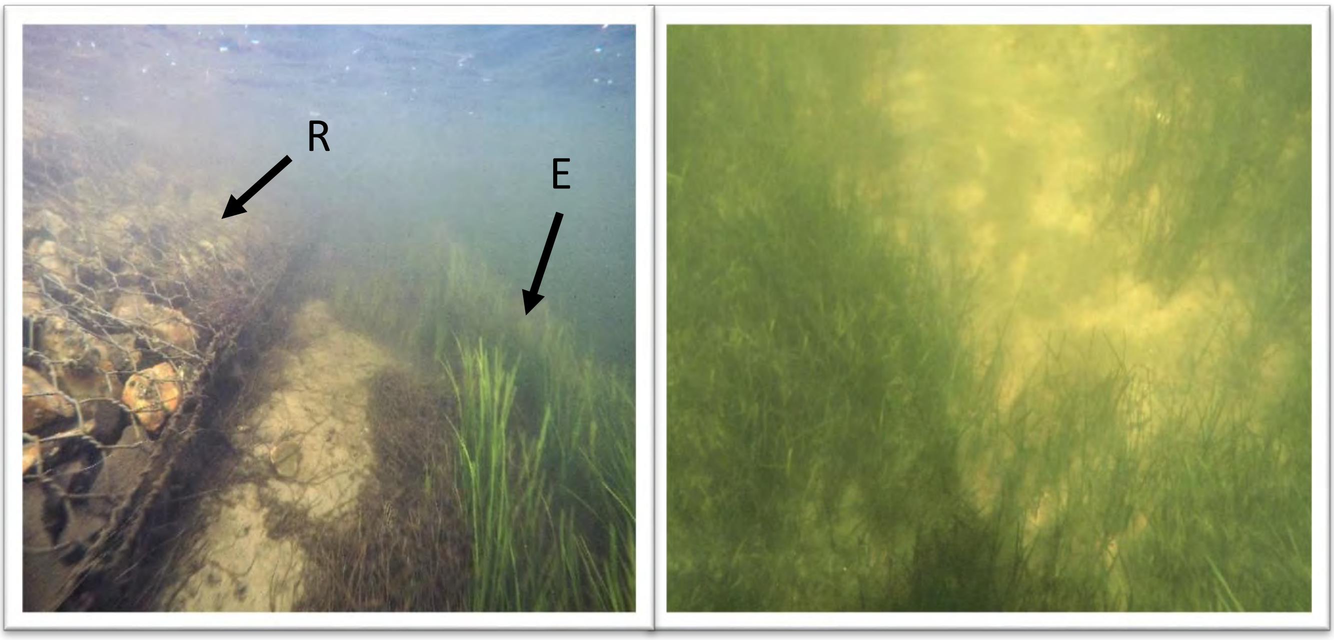

Figure 3.2. Newly installed Reno mattress (Impacted Habitat- image on left) completed in July 2019 located on bare sediment on the eastern bank of the Keurbooms Estuary below the N2 road bridge. Prior to installation eelgrass (*Z. capensis*- image on right) was predominant throughout the habitat.

After the installation of the Reno mattress (R), subtidal eelgrass (E) could be seen along the bottom edges of the mattress.

3.2.2 Eelgrass assessment

To ascertain if the construction process of the Reno mattress affected the extent and area of adjacent subtidal eelgrass at the Impacted habitat, monthly eelgrass extent was mapped by swimming along the outline of the eelgrass using a handheld GPS during three phases of installation of the Reno mattress installation: Before- winter and spring 2018, During- summer and autumn 2019, and After- winter and spring 2019. Coordinates were mapped on Google Earth and polygons were created to measure the area of eelgrass and any changes to the area of eelgrass. To monitor any additional impacts on the eelgrass, length was determined by measuring 10 subtidal eelgrass blades to the nearest mm from the rostrum to the end of the blade from each of four sample areas within each habitat. In addition, the percentage cover of eelgrass was assessed by haphazardly placing five 0.5 m X 0.5 m quadrats within each site, at each habitat (therefore making a sample size of 20 quadrats per habitat), and taking a photograph using an underwater camera (Sealife DC1400). The quadrat images were then compared to the seagrass percentage cover guide as recommended by Short *et al.* (2006). Measures of eelgrass length and percentage cover were done twice for each installation phase (Before- winter and spring 2018, During- autumn 2018 and summer 2019, and After- winter and spring 2019).

3.2.3 Before-After-Control-Impact (BACI)

In many instances, studies need to work with what is present at the sample location and in the case of this study the newly constructed Reno-mattress was limited to the extent of 70 m along the eastern bank. The BACI approach (as suggested by Underwood (1992)) used in this study included two control habitats (eelgrass (Control_Eel) and older Reno mattress (Control_Reno)), and an impacted habitat where newly constructed Reno mattress was placed. Prior to the installation of Reno mattress the focus area (Impacted Habitat) was dominated by eelgrass and therefore was compared to an eelgrass control habitat (Habitat I as described in Chapter 2). After the installation of Reno mattresses, the Impacted Habitat was compared to a Reno mattress control habitat (Habitat II as described in Chapter 2).

A BACI design approach (Bernstein & Zalinski, 1983; Stewart-Oaten *et al*., 1986; Smith, 2002) was used to investigate if the installation of a Reno mattress had an impact on the diversity and abundance of invertebrates and fish. The methods used to determine faunistic differences of invertebrate and fish species assemblages and abundances were carried out as described in Chapter 2 (summarized in Fig. 3.3). As the newly constructed Reno mattresses were positioned intertidally, core samples were taken (as in Habitat I (eelgrass)) subtidally at the bottom edge of the newly installed mattress instead of pumping as in Habitat II (older Reno mattress)) to investigate invertebrate assemblages and abundances. To establish the baseline fish and invertebrate abundance and diversity of the focus site (future Impacted Habitat), two surveys were completed prior to the commencement of the Reno mattress installation (July (winter) 2018 and October (spring) 2018). To determine whether the baseline fish and invertebrate abundance and diversity differed after the installation was finished, two surveys were completed (July 2019 (winter) and September 2019 (spring)). The two identified control habitats (Control_Eel and Control_Reno) were also surveyed at the same time.

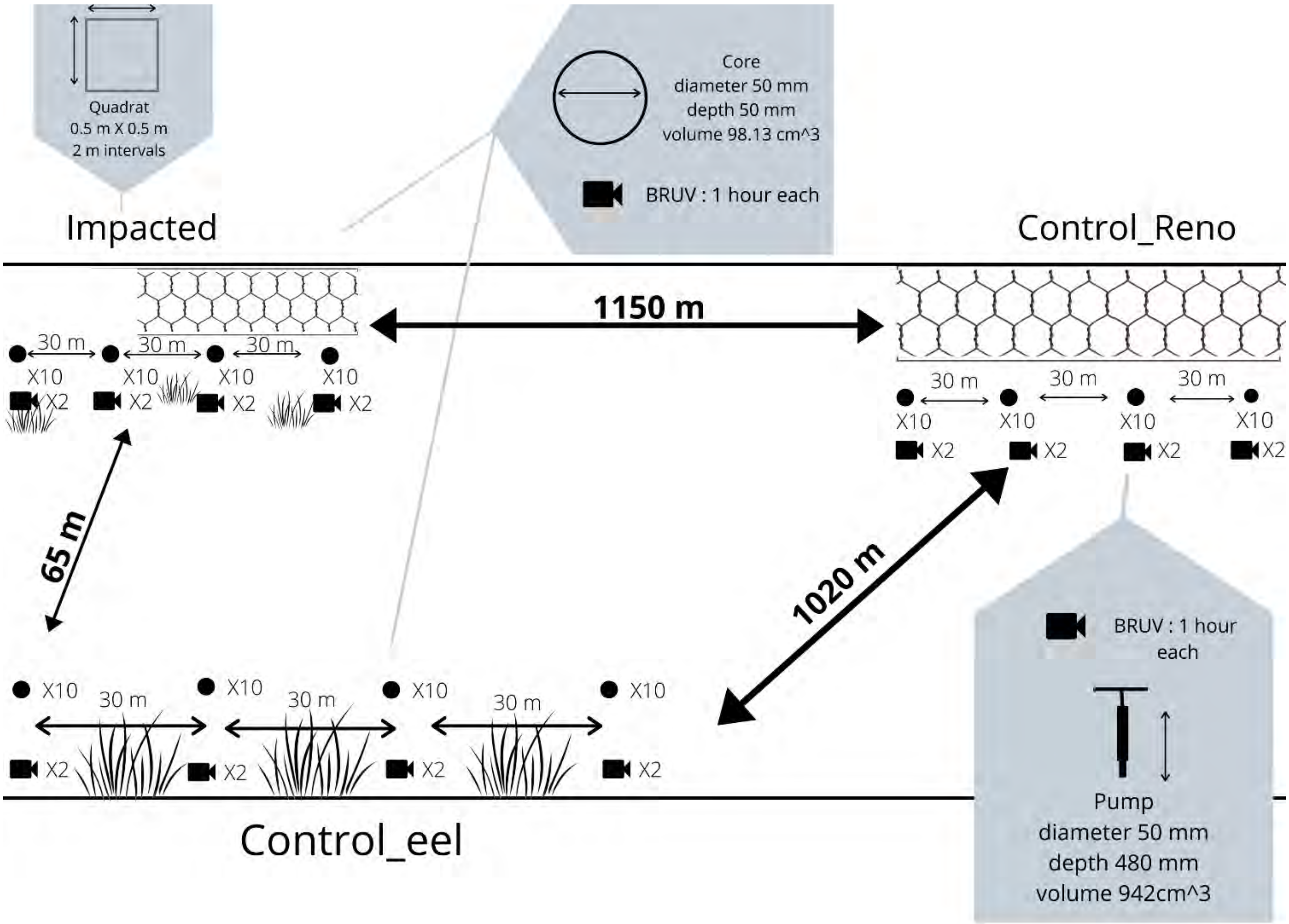

Figure 3.3. Schematic of the sampling procedure used for invertebrates and fish for three habitats (Impacted, Control_Eel, and Control_Reno) and for colonisation of sessile invertebrates on the new Reno mattress within the Impacted Habitat in the Keurbooms Estuary.

3.2.4 Colonisation of Renomattress

Older sections of Reno mattress are colonised by various animals, particularly sessile invertebrates e.g. mussels (Fig. 3.4), and barnacles. To determine the colonization of newly installed Reno mattress, intertidal invertebrates were assessed along a transect at 29 positions using a 0.5 x 0.5 m quadrat and following a standard photo quadrat approach (Bohnsack, 1979). Along the length of the newly constructed Reno mattress, every 2 m, quadrats were photographed on a monthly basis, from April 2019 to January 2020, to ensure the same section was photographed each month (Fig. 3.3). Quadrats were placed along each position at the same tidal height (half a metre above MLWS), and a Go-Pro camera, mounted to a frame to ensure that the height (55 cm) at which each photograph was taken was constant, was used to take photos of the quadrats. The percentage cover of barnacles and counts of gastropods present in the photographs was then assessed using PhotoQuad software (https://www.mar.aegean.gr/sonarlab/photoquad/index.php). Invertebrate animals found on the Reno mattress were identified to the lowest taxonomic level where possible (taxonomy was verified using World Register of Marine Species (WORMS- (http://www.marinespecies.org/aphia.php?p=taxlist)).

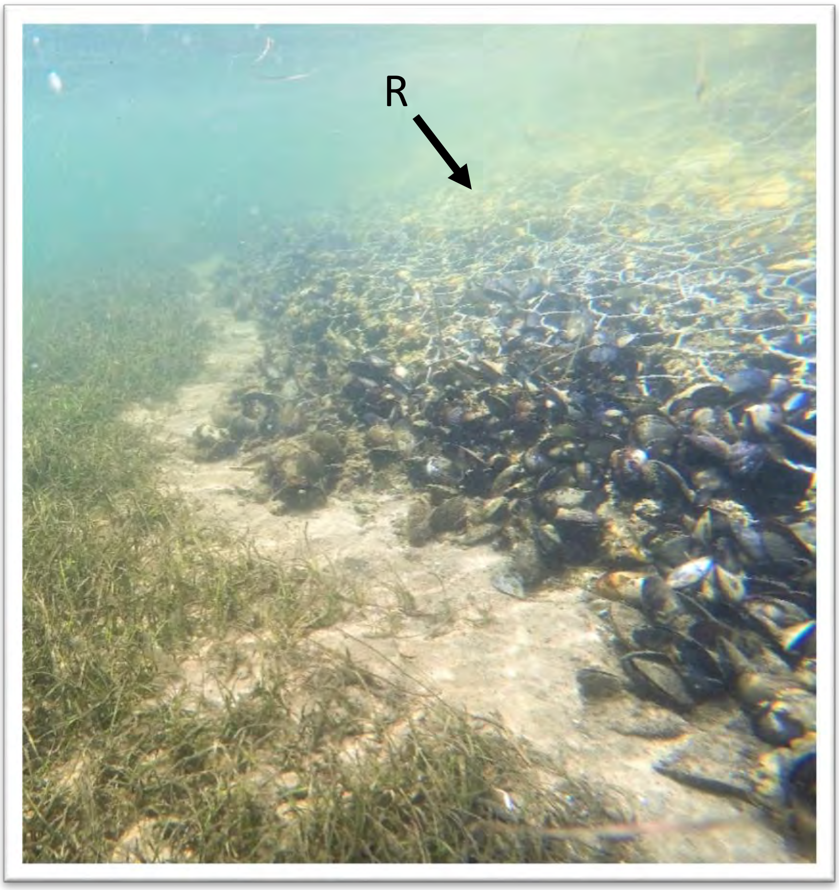

Figure 3.4. In the Control_Reno Habitat sessile invertebrates such as mussels colonise older sections of Reno mattress (R).

3.2.5 Statistical analysis

The statistical program R (R Development Core Team, 2014) was used in statistical analyses. Normality of all data was determined using Shapiro-Wilk tests. Differences of the area of eelgrass within Habitat III between installation phases were assessed using ANOVA. Percentage cover and length of eelgrass data were found to be not normally distributed ($p < 0.05$) and were assessed using Kruskal-Wallis test and significant differences were assessed using a Least Significance Difference (LSD) post hoc test.

Before and after data were compared, each installation phase had the same seasons (winter and spring). Seasonal variability of the abundances of invertebrates and fish within the system were noted, but seeing that it was not the focus, detailed results were not presented. Species richness was calculated as the counts of species per sample. A Shannon Diversity index (H') was calculated for each sample at each site. Overall significance of diversity and species richness was carried out using a Fit linear mixed-effects model (lme4 package (Bates & Maechler, 2010)). Significant differences were further assessed using post-hoc comparisons (Estimated marginal means (Least-squares means) using the emmeans package (Lenth *et al*., 2019)). Invertebrates from two phyla (Annelida and Mollusca) and one subphylum (Crustacea) that were the most abundant taxa of the data were chosen for further analysis. In addition to the above, overall significance of abundances of each group was tested using a Fit linear mixed-effects model. Invertebrates were grouped into five functional feeding groups (Carnivore Scavenger, Herbivorous Grazer, Omnivorous Detritus feeder, Omnivorous Deposit feeder, and Omnivorous Filter feeder) as these were the most abundant. Using a Fit linear mixed-effects model, abundances of each group were compared. Significant differences were further assessed using post-hoc comparisons. Similar statistical analyses were conducted for fish data. Fish were grouped into the four most abundant families (Clinidae, Gobiidae, Sparidae, and Mugilidae) in the data. The fish were also grouped into three functional feeding groups (Carnivorous, Omnivorous, and Zoobenthic predators) representing the most abundant groups.

Monthly differences of the percentage cover of colonising barnacles and counts of gastropods on the newly constructed Reno mattress were assessed using a Kruskal-Wallis test and significant differences were further assessed using post-hoc comparisons (Tukey test).

3.3 Results

3.3.1 Eelgrass assessment

Seasonal differences in the area, percentage cover, and eelgrass length were found, however, the primary aim of monitoring eelgrass was to assess the impact of the newly installed Reno mattress. Therefore this confounding effect was accounted for through the use of control sites. The data for the eelgrass area at the Impacted Habitat was normally distributed ($p > 0.05$, Shapiro-Wilk) and did not differ significantly across installation phases (before, during, and after installation) ($F_2 = 0.52$, $p > 0.05$, ANOVA, Fig. 3.5). Area of eelgrass ranged from 1374 m^2 to 2078 m^2 during the study.

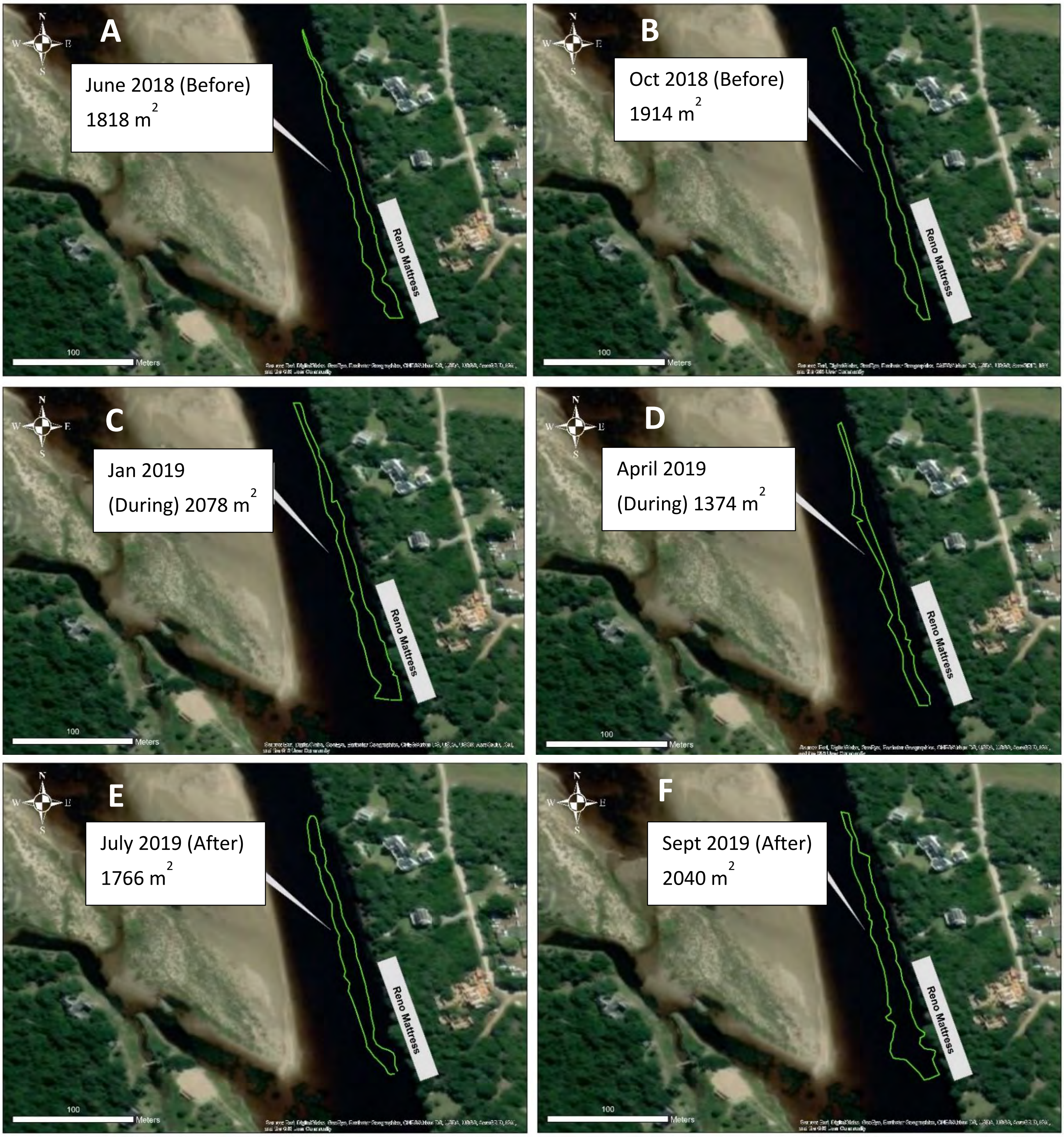

Figure 3.5. Distribution and area of eelgrass found at the Impacted Habitat. A) winter 2018, B) spring 2018, C) summer 2019, D) autumn 2019, E) winter 2019, and F) spring 2019.

Percentage cover data of eelgrass were not normally distributed ($p < 0.05$, Shapiro-Wilk), and differed significantly between all habitats ($H = 168.93$, $p < 0.01$, Kruskal-Wallis, Fig. 3.6).

Within the Control_Eel, the percentage cover was 95% throughout the study duration. Within the Impacted Habitat percentage cover of eelgrass decreased from 75% to 30% during the installation of Reno mattress, but increased to 73% after the installation (Fig. 3.6). At this habitat there was no significant difference in percentage cover before and after installation (W=623.5, $p > 0.05$, Wilcoxon test). Within the Control_Reno habitat, percentage cover of eelgrass increased significantly during the installation (H_2= 28.90, $p < 0.05$, Kruskal-Wallis, Fig. 3.6) and further increased after the installation ($p < 0.05$, LSD test).

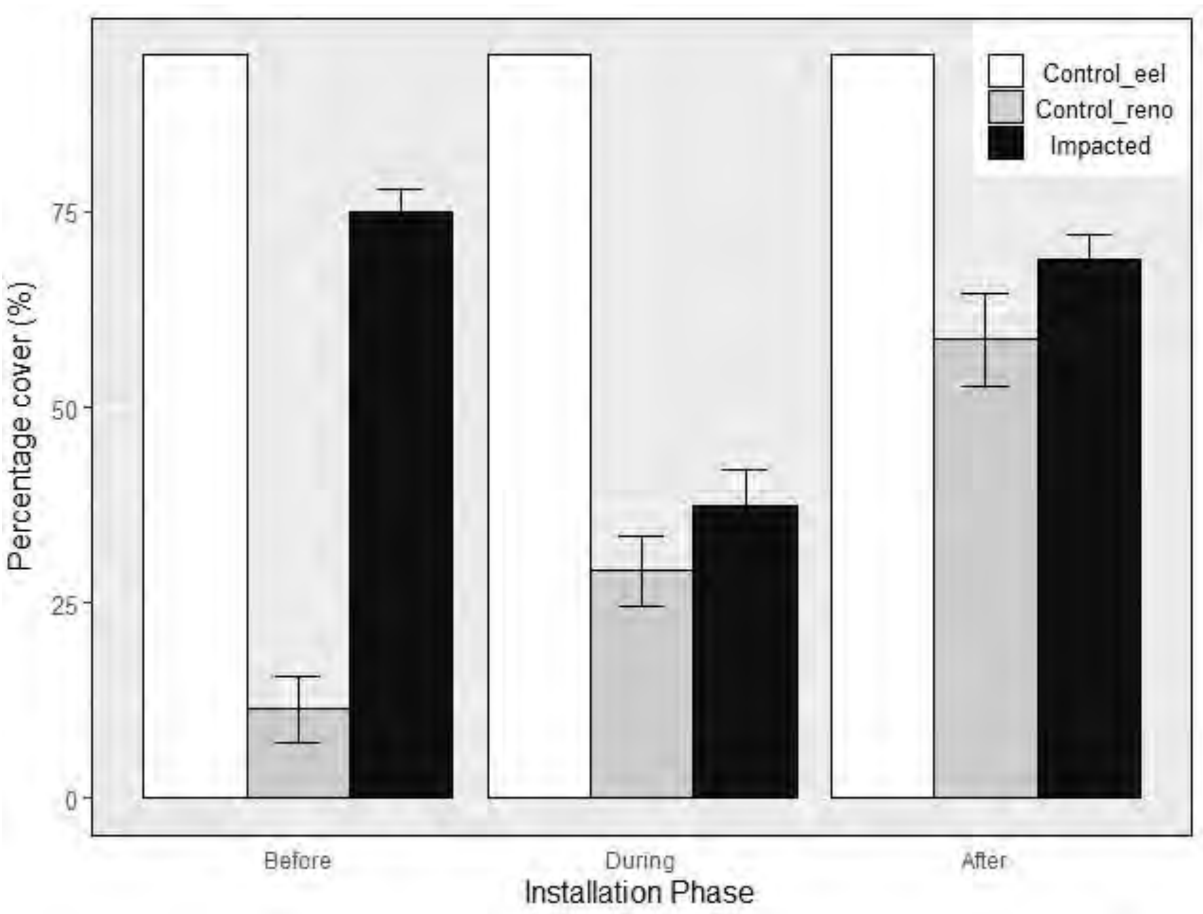

Figure 3.6. Mean variation of percentage cover (mean ±se) of eelgrass within habitats (Control_Eel, Control_Reno, and Impacted) before, during, and after the installation of Reno mattress structure at the Impacted Habitat.

The length of eelgrass blades was not normally distributed ($p < 0.05$, Shapiro-Wilk). Lengths differed significantly between all habitats (H_2= 231.82, $p < 0.01$, Kruskal-Wallis) with differences specifically found between Control_Eel and Control_Reno Habitats ($p < 0.05$, LSD test), and Control_Eel and Impacted Habitats ($p < 0.05$, LSD test, Fig. 3.7). The greatest average lengths (487.34- 579.61 mm) of eelgrass were found at Control_Eel Habitat. Average length of eelgrass blades found within the Impacted Habitat were significantly shorter after the installation of Reno mattress compared to before ($W = 31337$, $p < 0.01$, Wilcoxon test). Within the Control_Reno Habitat the average blade length decreased significantly during installation ($H_2 = 48.65$, $p < 0.01$, Kruskal Wallis), but then increased after the installation and did not differ significantly between the before and after installation phases ($p > 0.05$, LSD, Fig. 3.7).

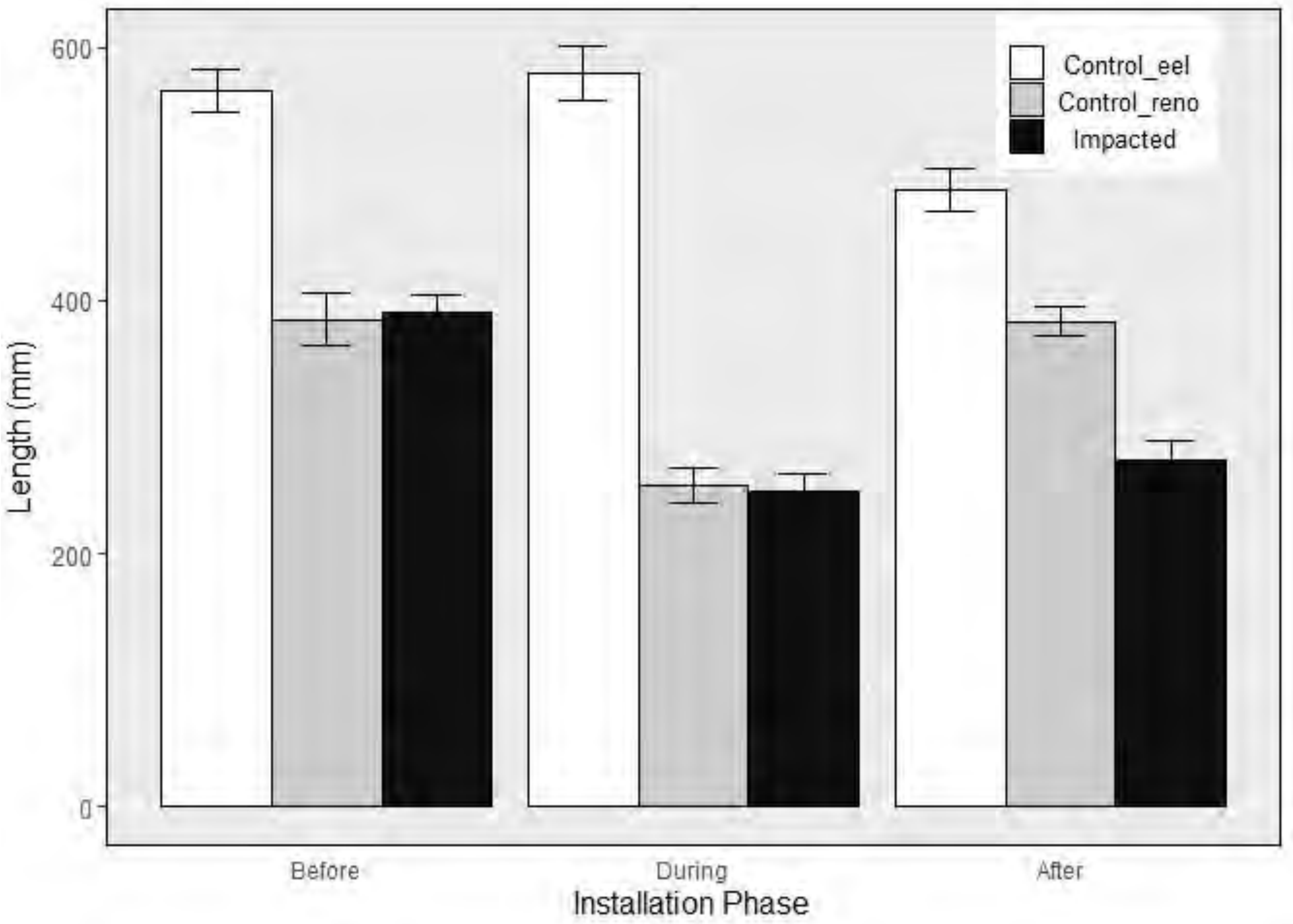

Figure 3.7. Mean change of length (mean ±se) of eelgrass blades found within three habitats (Control_Eel, Control_Reno, and Impacted) in 2018 and 2019 during the installation of Reno mattress at the Impacted Habitat.

3.3.2 BACI

3.3.2.1 Invertebrates

Of the 20 taxa recorded within the Impacted Habitat, 14 were found both before and after the installation of Reno mattress, and a further six taxa were newly recorded after installation was complete (Table 3.1). Species richness within the Impacted Habitat increased significantly after the installation of Reno mattress (t_{120609}= 1.98, $p < 0.05$, Tukey, Fig. 3.8). Within both control habitats, species richness increased during the course of the study, however, this increase was not significantly different between the before and after installation phases (Control_Eel: t_{120609}= 0.97, $p > 0.05$, Control_Reno: t_{120609}= 1.44, $p > 0.05$, Tukey). Although Fig. 3.9 indicates that diversity increased during the course of the study at all habitats, no significant difference in invertebrate diversity between the before and after installation phases was found within the Impacted Habitat (t_{87418}= 1.70, $p > 0.05$, Tukey), and control habitats (Control_Eel, t_{87418}= 0.85, $p > 0.05$, Tukey; Control_Reno, t_{87418}= 1.68, $p > 0.05$, Tukey).

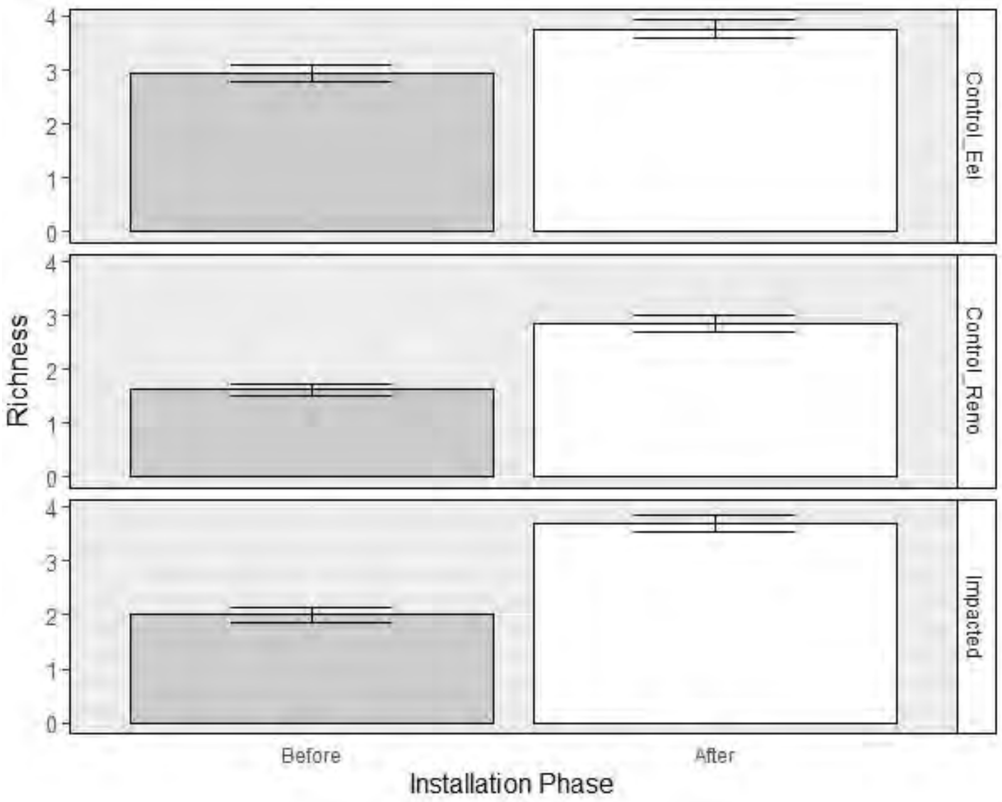

Figure 3.8. Mean (±se) number of invertebrate species within three habitats (Control_Eel, Control_Reno and Impacted) during before and after installation phases of Reno mattress at the Impacted Habitat.

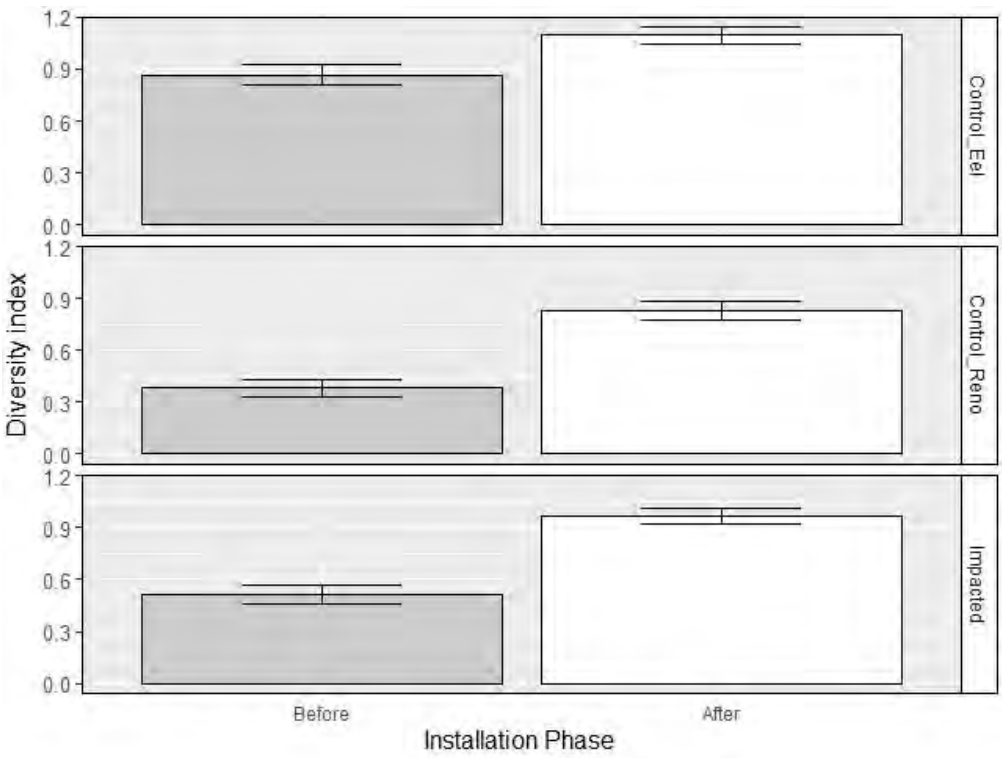

Figure 3.9. Mean (±se) Shannon diversity (H') of invertebrates within three habitats (Control_Eel, Control_Reno and Impacted) during before and after installation phases of Reno mattress at the Impacted Habitat.

Table 3.1: Invertebrate taxa found in Keurbooms Estuary. Habitat: I= Control_Eel, II= Control_Reno, III= Impacted; Functional feeding group: C_S= carnivore scavenger, C_ P= carnivore predator, C_Pa= parasite, H_B= herbivore browser, H_G= herbivore grazer, O_D= omnivore detritus feeder, O_De= omnivore deposit feeder, O_F= omnivore filter feeder, O_L= omnivore lignivorous, O_P= omnivore predator, O_S= omnivore scavenger. Highlighted rows represent all taxa only found in a single habitat and sample time or only a single sighting.

Taxa	Installation phase	Habitat	Functional group
PLATYHELMINTHES			
Polycladida			
Planocera gilchristi	After	I, III	C_P
NEMERTEA			
White nemertean	After	I, II, III	O_D
ANNELIDA			
Polychaeta			
Sedentaria			
Prionospio sp.	Before, After	I, II, III	O_De
Capitella sp.	Before, After	I, II, III	O_De
Desdemona sp.	After	II, III	O_F
Ficopomatus enigmaticus	After	II	O_F
Errantia			
Glycera sp.	Before, After	I, II, III	C_P
Simplisetia sp.	Before, After	I, II, III	O_De
Clitellata			
Hirudinea			
Pontobdella sp.	After	II	C_Pa
ARTHROPODS			
Insecta			
Chironomid	After	II, III	O_D
Crustacea			
Thecostraca			
Amphibalanus amphitrite	Before	II	O_F
Emalacostraca			
Isopoda			
Paridotea ungulata	Before, After	I, II	H_B
Cyathura sp.	After	II, III	C_P
Sphaeromatoids	After	I, II	O_L
Cumacea	After	II, III	O_F
Amphipoda			
Melita sp.	Before, After	I, II, III	O_D
Victoriopisa sp.	Before	I	O_De
Monocorophium acherusicum	Before, After	I	O_F
Decapoda			
Upogebia africana	Before, After	I, II, III	O_F
Hymenosoma orbiculare	Before, After	I, II, III	C_P
MOLLUSCA			
Bivalvia			
Pteriomorphia			
Arcuatula capensis	Before, After	I, II, III	O_F

Mytilus galloprovincialis	Before, After	II	O_F
Heterodonta			
Macoma litoralis	Before, After	I, II, III	O_F
Lasaea adansoni	Before, After	I, II, III	O_F
Gastropoda			
Heterobranchia			
Siphonaria sp.	After	II	H_G
Elysia sp.	After	II	H_G
Euthyneura			
Haminoea alfredensis	Before, After	I, II	H_G
Bursatella leachii	After	II	H_G
Godiva quadricolor	Before, After	I, III	C_P
Favorinus ghanensis	After	II	C_P
Philine aperta	After	II	C_P
Caenogastropoda			
Thiaridae	Before, After	I, II, III	O_S
Nassarius kraussianus	Before, After	I, II, III	C_S
Hydrobia knysnaensis	Before, After	I, II, III	H_G
ECHINODERMS			
Asteroidea			
Parvulastra exigua	Before	II	C_P
Echinoidea			
Euechinoidea			
Parechinus angulosus	Before	II	H_G

Within the Impacted Habitat abundances of Annelida were significantly greater after installation compared to before the installation (t_{260012}= 3.82, $p < 0.01$, Tukey, Fig. 3.10), but did not differ significantly within control habitats Control_Eel (t_{260012}= 1.53, $p > 0.05$, Tukey) or Control_Reno (t_{260012}= 1.49, $p > 0.05$, Tukey). Abundances of Crustacea (t_{412866} = -0.16, $p > 0.05$, Tukey, Fig. 3.10) and Mollusca (t_{823}= -1.040, $p > 0.05$, Tukey, Fig. 3.10) within the Impacted Habitat did not differ significantly between installation phases (before and after). Within Control_Eel Habitat abundances of Crustacea (t_{412866}= -0.16, $p > 0.05$, Tukey, Fig. 3.10) and Mollusca (t_{827}= 1.46, $p > 0.05$, Tukey, Fig. 3.10) did not differ significantly between before and after installation phases. Within Control_Reno abundances of Crustacea significantly decreased (t_{412866}= -2.54, $p < 0.05$, Tukey) and Mollusca increased (t_{825}= 2.23, $p < 0.05$, Tukey, Fig. 3.10) between before and after installation phases.

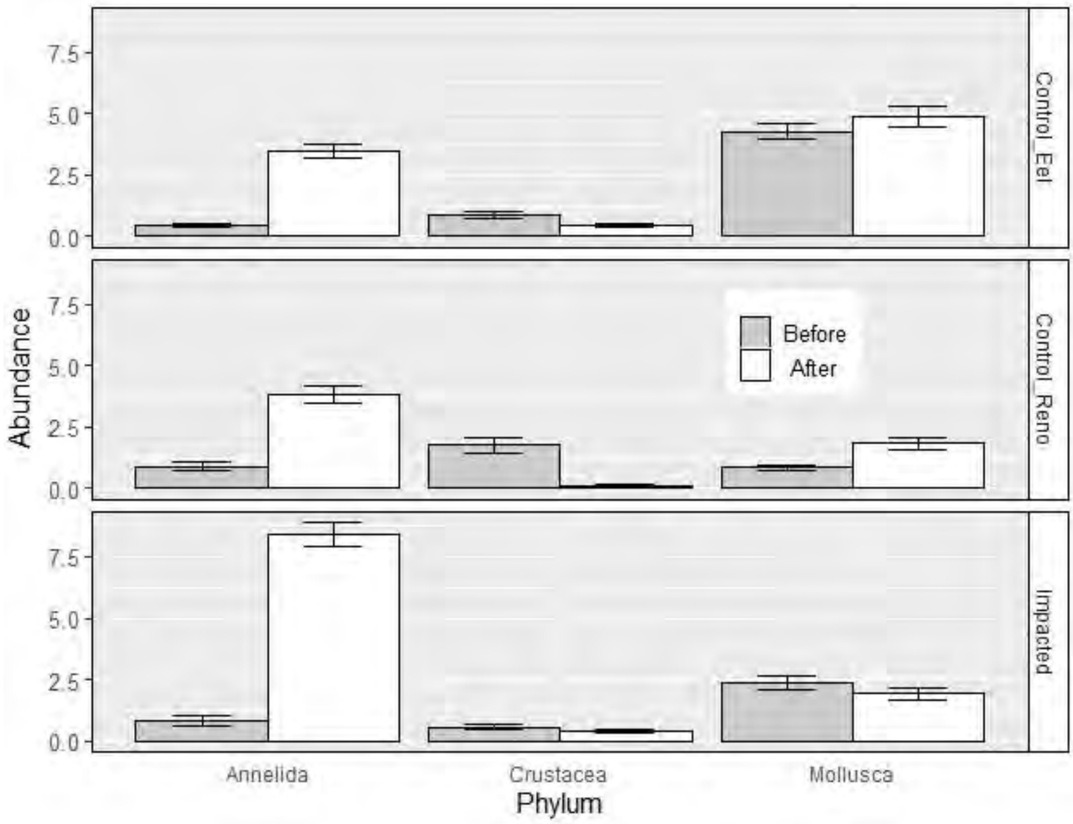

Figure 3.10. Mean (±se) abundance of two major invertebrate phyla (Annelida and Mollusca) and one subphylum (Crustacea) at three Habitats (Control_Eel, Control_Reno, and Impacted) during different installation phases (before and after) of Reno mattress at the Impacted Habitat.

No significant differences within the Impacted Habitat between the before and after installation phases were found in the abundances of the functional feeding groups Carnivore_scavenger ($t_{268} = -2.030$, $p > 0.05$, Tukey, Fig. 3.11), Herbivore_grazer ($t_{630} = 0.26$, $p > 0.05$, Tukey, Fig. 3.11), Omnivorous_detritus feeder ($t_{9695} = -0.54$, $p > 0.05$, Fig. 3.11), and Omnivorous_deposit feeder ($t_{8228189} = 1.71$, $p > 0.05$, Tukey, Fig. 3.11). Abundances of Omnivorous_filter feeder were significantly greater after installation of Reno mattress at Impacted Habitat ($t_{1535} = 2.05$, $p < 0.05$, Tukey, Fig. 3.11) and Control_Eel Habitat ($t_{1526} = 2.33$, $p < 0.05$, Tukey, Fig. 3.11). Within the Control_Eel Habitat, there were no significant differences of abundances between installation phases of the functional feeding groups Carnivore_scavenger ($t_{268} = -2.00$, $p > 0.05$, Tukey, Fig. 3.11), Omnivorous_detritus feeder ($t_{9695} = -0.96$ $p > 0.05$, Fig. 3.11), and Omnivorous_deposit feeder ($t_{8228189} = 0.47$, $p > 0.05$, Tukey, Fig. 3.11). The abundances of the functional group Herbivore_grazer were significantly greater after compared to before at Control_Eel Habitat ($t_{631} = 3.94$, $p < 0.01$, Tukey, Fig. 3.11) and Control_Reno Habitat ($t_{627} = 2.6$, $p < 0.01$, Tukey, Fig. 3.11). No significant differences within Control_Reno between installation phases were found in the abundances of the functional feeding groups Carnivore_scavenger ($t_{268} = 0.39$, $p > 0.05$, Tukey, Fig. 3.11), Omnivorous_deposit feeder ($t_{7822402} = 0.36$, $p > 0.05$, Tukey, Fig. 3.11), and Omnivorous_filter feeder ($t_{1492} = 0.20$, $p > 0.05$, Tukey, Fig. 3.11). The abundances of the functional feeding group Omnivorous_detritus feeder were significantly smaller after the installation ($t_{9319} = -3.28$, $p < 0.01$, Fig. 3.11).

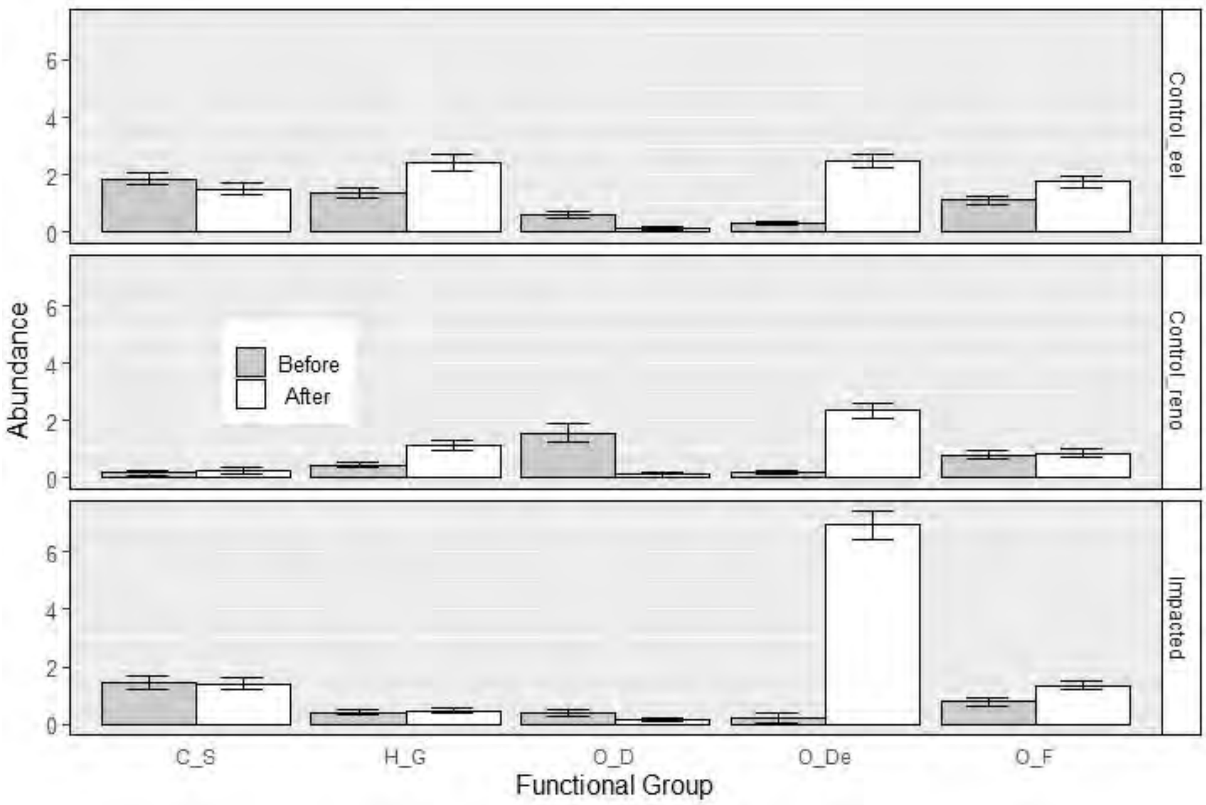

Figure 3.11. Mean (±se) abundance of five major invertebrate functional feeding groups (C_S: carnivore scavenger, H_G: herbivore grazer, O_D: omnivore detritus feeder, O_De: omnivore deposit feeder, and O_F: omnivore filter feeder) within three habitats (Control_Eel, Control_Reno, and Impacted) during before and after installation phases of Reno mattresses at the Impacted Habitat.

3.3.2.2 Fish

Of the 16 fish taxa that were noted in the Impacted Habitat, one species (*Hippocampus capensis*) was only recorded before the installation of Reno mattress, three species (*Diplodus capensis*, *Sarpa salpa*, *Lichia amia*) were present only after the installation, and 12 were found before and after the installation of Reno mattress (Table 3.2). Two species (*Lutjanus fulviflamma*, and *Chaetodon marleyi*) were only found in Control_Reno and one family (Ophichthidae) was found in Control_Eel only. Although Fig. 3.12 indicates there was an increase in the richness of fish during the study there was no significant difference between before and after installation phases at all habitats (Impacted Habitat, t_{323} = 1.52, p

> 0.05, Tukey; Control_Eel Habitat, t_{323} = 1.18, $p > 0.05$, Tukey; Control_Reno Habitat, t_{323} =0.59, $p > 0.05$, Tukey).

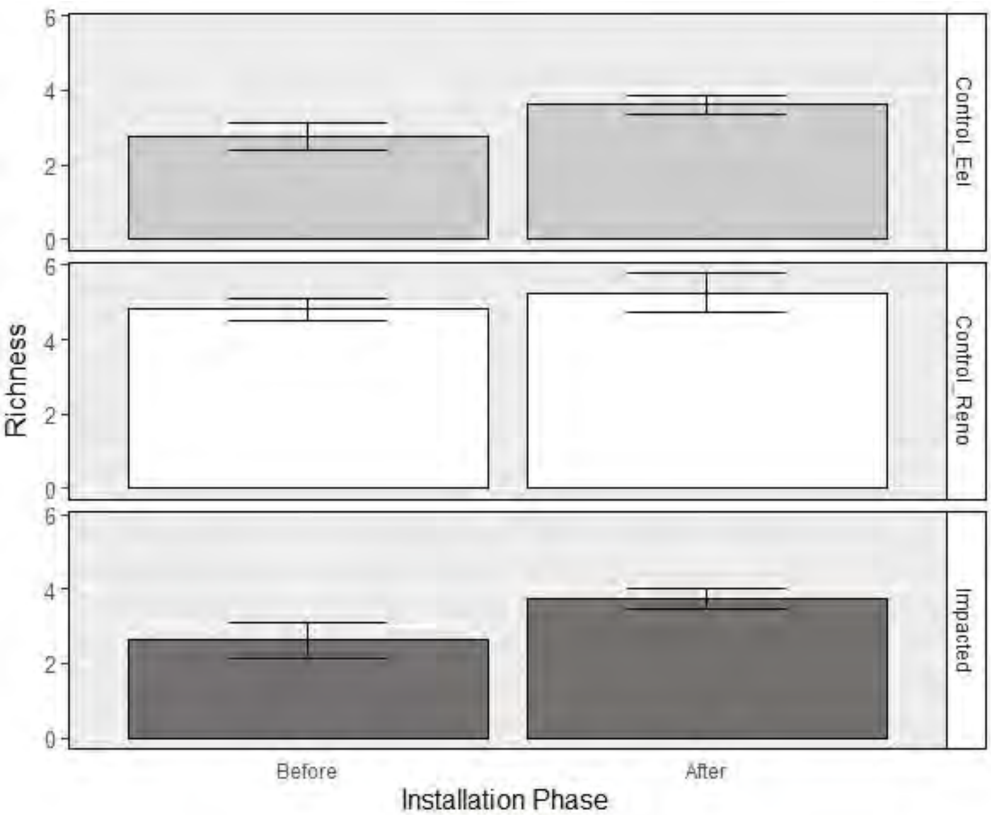

Figure 3.12. Mean (±se) number of fish species within three habitats (Control_Eel, Control_Reno and Impacted) during before and after installation phases of Reno mattress at the Impacted Habitat.

Although fish diversity appeared to increase during the study at the Impacted and both Control Habitats (Fig. 3.13) there was no significant differences between the before and after installation phases (Impacted Habitat, t_{3734} = 1.503, $p > 0.05$, Tukey; Control_Eel Habitat, t_{3734} =0.97, $p > 0.05$, Tukey; Control_Reno Habitat, t_{3734} =0.2, $p > 0.05$, Tukey).

Table 3.2. Fish taxa recorded in the Keurbooms estuary on mini BRUVs within each habitat (I- Control_Eel, II- Control_Reno, III- Impacted) for each installation phase (Before, During, After). Estuary dependency category according to Wallace *et al.* 1984 (Category I- species completely estuarine dependent for their entire life cycle, Category II- species dependent on estuaries during only their juvenile stage, Category III- species whose juveniles occur mainly in estuaries but are also found at sea, Category IV- species whose juveniles mainly occur at sea but are abundant in estuaries, Category V- species whose juveniles occur at sea but sometimes are found into estuaries. Functional feeding group (C= carnivorous, O= omnivorous, P= piscivorous, H= herbivorous, Z= zoobenthic predator, D= detritivorous). Highlighted rows represent all taxa only found in a single habitat and sample period or only a single sighting.

Taxa	Installation phase	Habitat	Estuary dependence category **(Wallace *et al.* 1984)**	Functional group
ACTINOPTERYGII				
TELEOSTEI				
Ariidae				
***Galeichthys feliceps* (Valenciennes 1840)**	Before, After	I, II, III	IV	C
Syngnathidae				
***Hippocampus capensis* (Boulenger 1900)**	Before	III	I	C
***Syngnathus temminckii* (Kaup 1856)**	Before, After	I, II, III	IV	C
Haemulidae				
***Pomadasys commersonnii* (Lacepède 1810)**	Before, After	I, II, III	II	Z
Sparidae				
***Diplodus capensis* (Smith 1844)**	After	I, II, III	IV	O
***Diplodus hottentotus* (Smith 1844)**	Before, After	I, II, III	V	C

Lithognathus lithognathus **(Curvier 1829)**	Before, After	I, II, III	II	Z
***Rhabdosargus* sp.** **(Steindachner 1881)**	Before, After	I, II, III	II	H (juvenile) C (adult)
Sarpa salpa **(Linnaeus 1758)**	After	I, II, III	IV	C (juvenile) H (adult)
Monodactylidae				
Monodactylus falciformis **(Lacepède 1800)**	Before, After	I, II, III	II	H
Carangidae				
Lichia amia **(Linnaeus 1758)**	After	I, II, III	II	P
Mugilidae*	Before, After	I, II, III	II-IV	D
Clinidae*	Before, After	I, II, III	I-V	C
Gobiidae				
Psammogobius knysnaensis **(Smith 1936)**	Before, After	I, II, III	I	C
Caffrogobius nudiceps **(Valenciennes 1837)**	Before, After	I, II, III	IV	C
Caffrogobius caffer **(Günther 1874)**	Before, After	II, III		C

(*) It was not possible to identify fish in these families to species level from camera footage and were identified to family level.

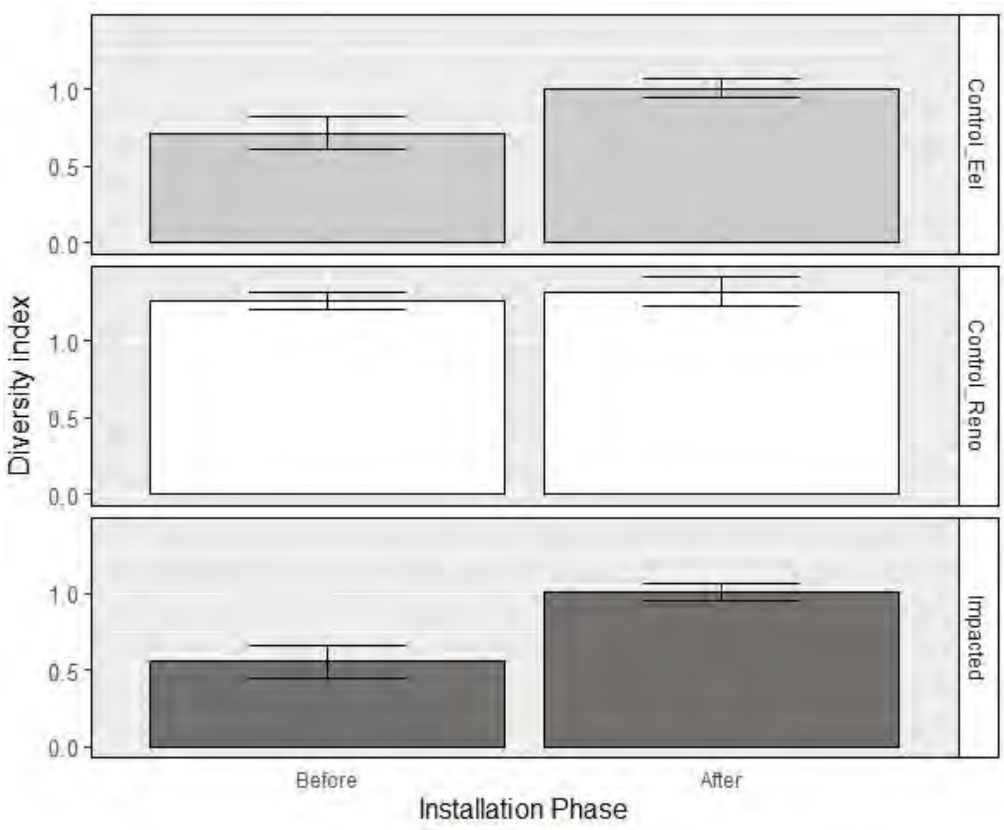

Figure 3.13. Mean (±se) Shannon diversity (H') of fishes within three habitats (Control_Eel, Control_Reno and Impacted) during before and after installation phases of Reno mattress at the Impacted Habitat.

Table 3.3. Results of the post hoc Tukey tests for differences in MaxN of each family of fish between before and after installation phases within each habitat (Impacted, Control_Eel, and Control_Reno).

Family	**df**	**t value**	**p value**
Impacted Habitat			
Clinidae	11972	-0.04	$p > 0.05$
Gobiidae	4579	-0.37	$p > 0.05$
Mugilidae	231	-0.39	$p > 0.05$
Sparidae	981	1.23	$p > 0.05$
Control_Eel Habitat			
Clinidae	11972	0.11	$p > 0.05$
Gobiidae	4579	1.18	$p > 0.05$
Mugilidae	243	-0.57	$p > 0.05$
Sparidae	986	-0.08	$p > 0.05$
Control_Reno Habitat			
Clinidae	12713	-0.44	$p > 0.05$
Gobiidae	4579	-0.37	$p > 0.05$
Mugilidae	231	-0.78	$p > 0.05$
Sparidae	981	0.48	$p > 0.05$

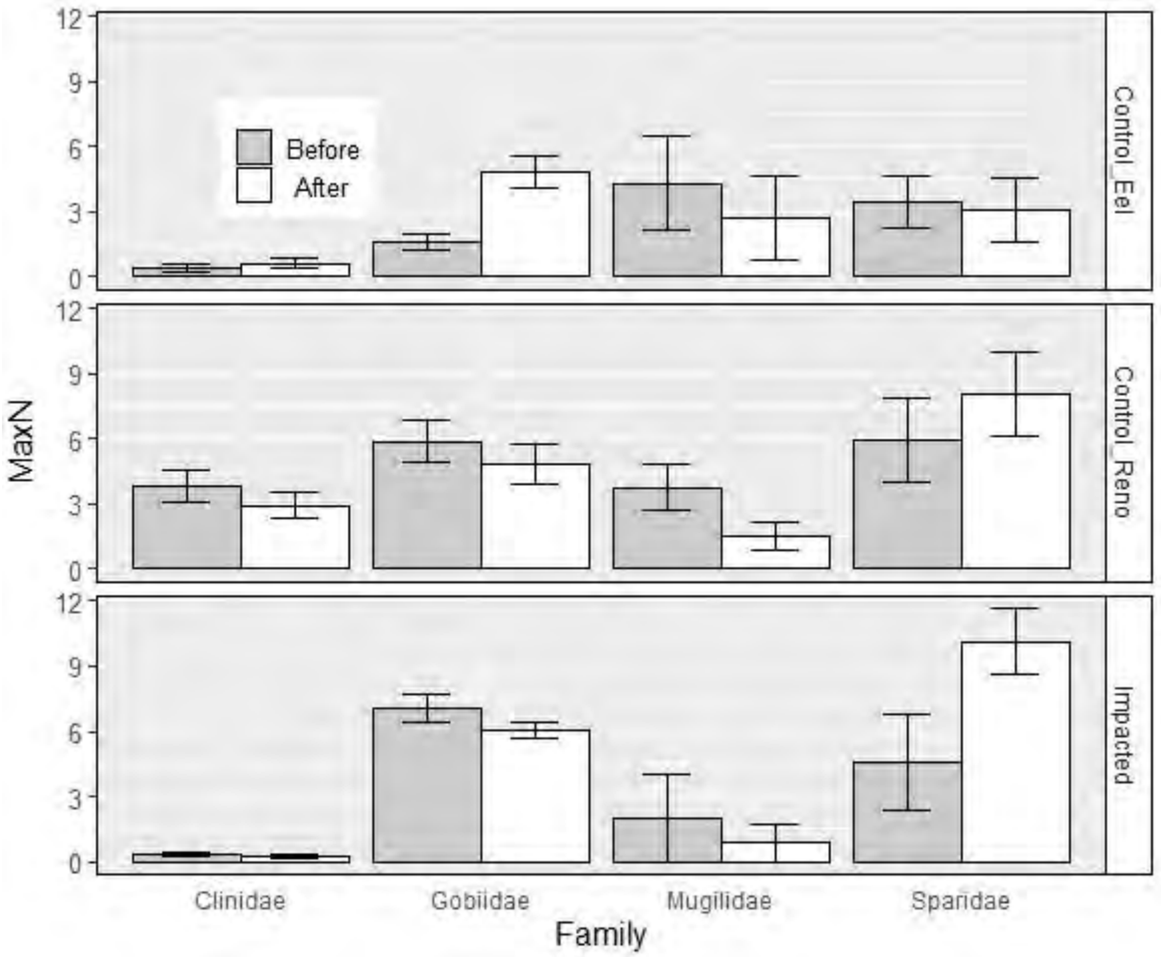

Figure 3.14. Mean (±se) MaxN (maximum number of any individual fish group in any one frame) of four major fish families (Clinidae, Gobiidae, Mugilidae, and Sparidae) within three habitats (Control_Eel, Control_Reno, and Impacted) during before and after installation phases of Reno mattress at the Impacted Habitat.

At all habitats the MaxN of the families Clinidae, Gobiidae, Mugilidae and Sparidae did not differ significantly between the before and after installation phases (Fig. 3.14, Table 3.3). At all sites the MaxN of the functional feeding groups Carnivore, Detritivore, and Zoobenthic predator did not differ significantly between the before and after installation phases (Fig. 3.15, Table 3.4).

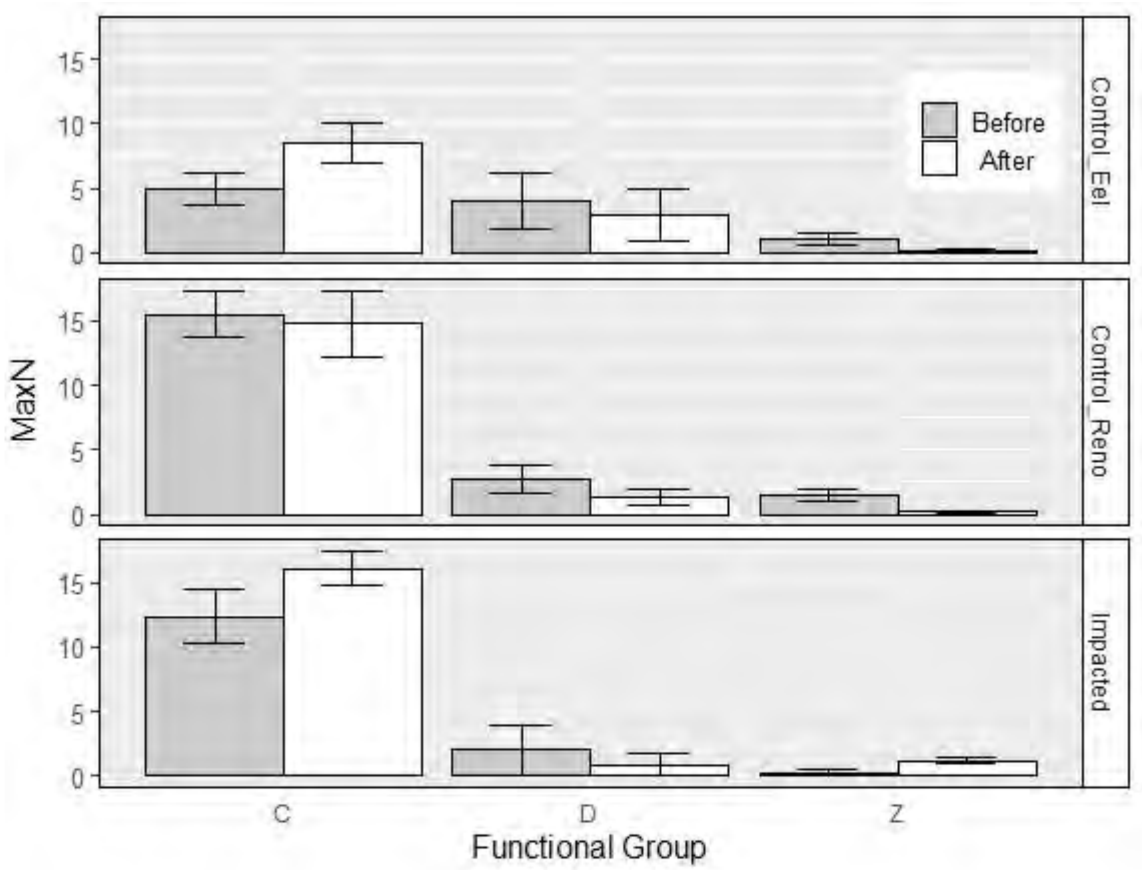

Figure 3.15. Mean (±se) MaxN of three major fish functional feeding groups (C= carnivorous, D= detritivorous, Z= zoobenthic predator) during different installation phases (before and after) of Reno mattress at the Impacted Habitat within three habitats (Control_Eel, Control_Reno, and Impacted).

Table 3.4. Results of the post hoc Tukey tests for differences in MaxN of each functional feeding group (Carnivorous, detritivorous, and zoobenthic predator) of fish between before and after installation phases within each habitat (Impacted, Control_Eel, and Control_Reno).

Functional group	**df**	**t value**	**p value**
	Habitat III (Impacted)		
Carnivorous	3595	0.57	$p > 0.05$
Detritivorous	189	-0.41	$p > 0.05$
Zoobenthic predator	1326	1.02	$p > 0.05$
	Habitat I (Control_Eel)		
Carnivorous	3800	0.56	$p > 0.05$
Detritivorous	193	-0.42	$p > 0.05$
Zoobenthic predator	1392	-0.91	$p > 0.05$
	Habitat II (Control_Reno)		
Carnivorous	3800	-0.105	$p > 0.05$
Detritivorous	193	-0.51	$p > 0.05$
Zoobenthic predator	1392	-1.47	$p > 0.05$

3.3.3 Colonization of Reno-mattress at Impacted Habitat

Following the installation of Reno mattress, the eastern bank of the Keurbooms Estuary showed marked changes in bank stabilisation at the section where Reno mattresses were placed (Appendix A). Prior to the installation, the bank showed severe erosion, the installation of Reno mattresses preventing further erosion of this section of bank.

Within a month of installation (May 2019), only two taxa (*Amphibalanus amphitrite* and *Littoraria* sp.) were found colonising the newly installed Reno mattress. Percentage cover of *A. amphitrite* within the Impacted Habitat, which was not normally distributed ($p < 0.05$, Shapiro-Wilk), differed significantly across months ($H_9 = 38.543$, $p < 0.05$, Kruskal-Wallis). Post hoc tests revealed that percentage cover in January 2020 was significantly greater than in April, May and July 2019 ($p < 0.05$, Tukey, Fig. 3.16). An increasing trend in the percentage cover of *A. amphitrite* was observed over time.

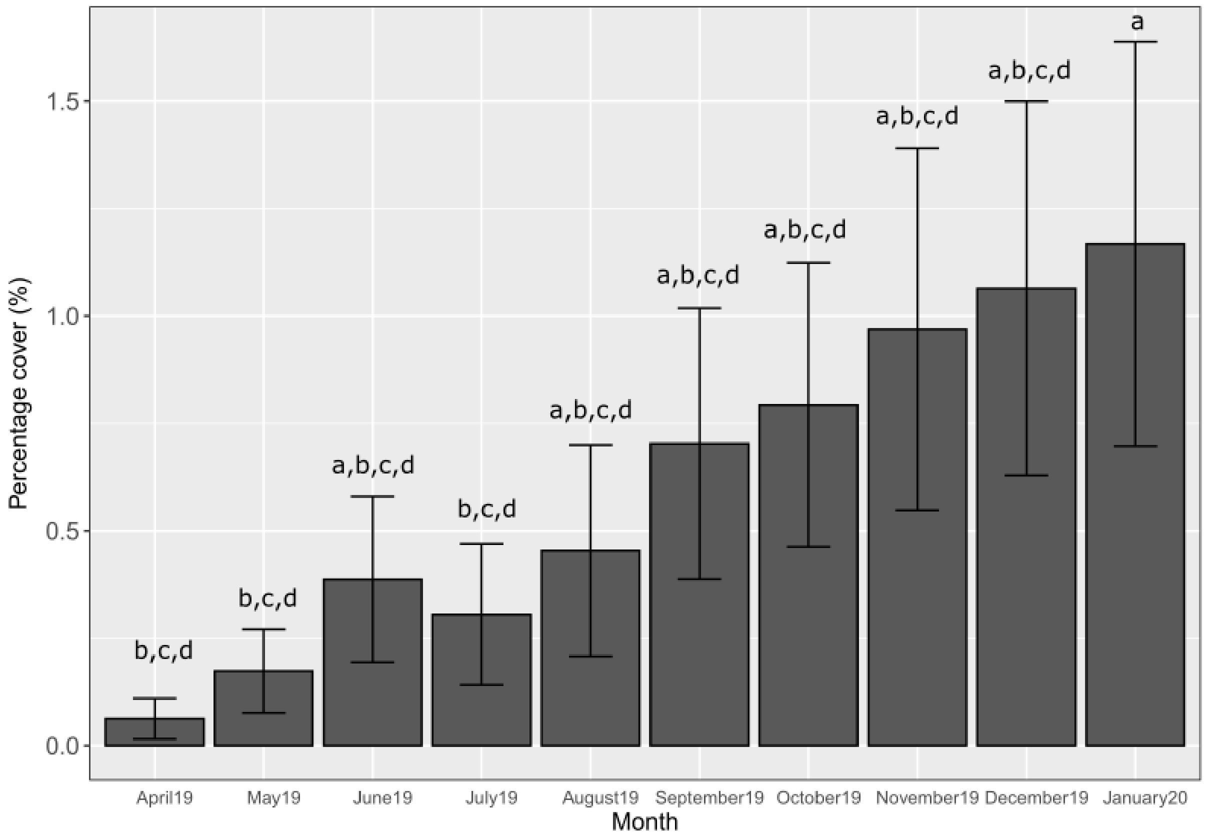

Figure 3.16. Monthly mean (±se) percentage cover (%) of *Amphibalanus amphitrite* on the newly constructed Reno mattress within the Impacted Habitat in 2019. Different letters above error bars indicate significant difference between means.

Average number of individuals of *Littoraria* sp. per 0.25 m^2 was found to be non-normally distributed ($p < 0.05$, Shapiro-Wilk) and differed significantly across months ($H_9 = 18.53$, $p < 0.05$, Kruskal-Wallis). Post hoc tests revealed that the average number of individuals per 0.25 m^2 in January 2020 was significantly greater than in April, August, July, October, and December 2019 ($p < 0.05$, Tukey, Fig. 3.17).

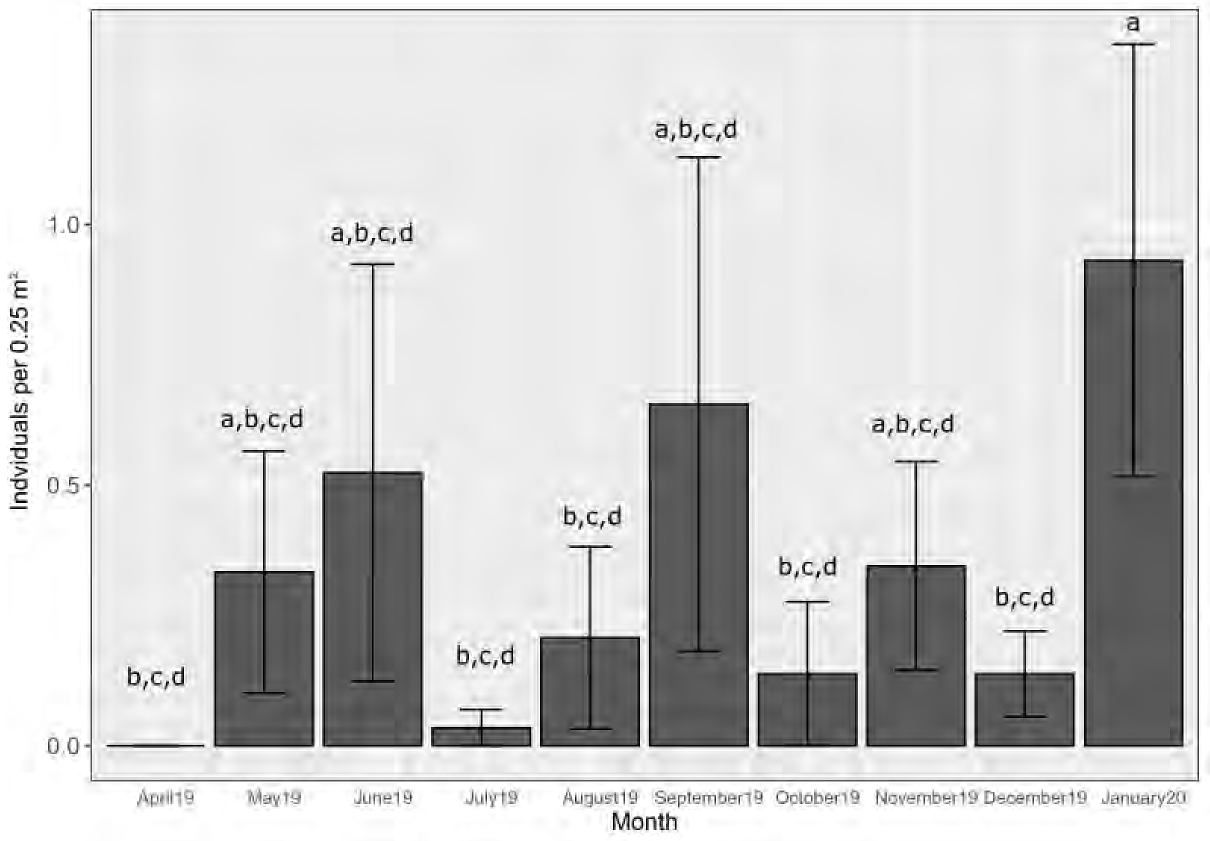

Figure 3.17. Monthly mean (±se) density of *Littoraria* sp. on the newly constructed Reno mattress within the Impacted Habitat in 2019. Different letters above error bars indicate significant difference between means.

3.4 Discussion

Coastal ocean sprawl has been thought to drive the loss of habitats, especially seagrasses (Airoldi & Beck, 2007; Bulleri & Chapman, 2010; Morris *et al*., 2018). Estuarine macrophytes act as nursery areas for fish and invertebrates (Polte & Adams, 2006), which is especially evident in areas that are monitored before and after the loss of macrophytes (Whitfield, 2016). For example, Whitfield (1984) reported the decline in abundance of two juvenile fish species after a reduction in macrophyte beds (*Potamogeton pectinatus*) in the Swartvlei estuarine lake in South Africa, and Pihl *et al*. (2006) recorded the reduction of juvenile cod from sites where the seagrass (*Zostera marina*) beds had disappeared in Sweden. The loss of seagrass beds due to anthropogenic pressures is of great concern (Orth *et al*., 2006; Short *et al*., 2006; Adams, 2016). Measurable differences in both the assemblages and abundances of fish were found between artificial Reno mattress structures and natural eelgrass beds in the Keurbooms Estuary (see Chapter 2). The present study investigated whether these differences could be attributed to the installation of a Reno mattress structure and the transformation to a hybrid habitat of eelgrass and Reno mattress.

Most seagrasses exhibit cyclical growth rates, with maximum plant density and biomass usually recorded in summer (Hanekom & Baird, 1984; Whitfield, 2016; 2019) which, coincides with the arrival of juvenile marine fishes. In the present study this general seasonal trend was not observed at the Control_Eel Habitat, percentage cover of eelgrass remained constantly at 95%, and at the Control-Reno Habitat there was an increase in percentage cover throughout the study continuing into autumn and winter. Variability in eelgrass extent can be caused by various factors, therefore, this result may be due to various other environmental changes such as changes in temperature, salinity, or turbidity.

Within the Impacted Habitat, eelgrass percentage cover decreased during the installation phase (summer) which indicates the decrease was not due to seasonal changes but rather the installation of Reno mattress. After the installation eelgrass recovered quickly and no differences were found between installation phases. The length of eelgrass blades were found to be shorter after the installation of Reno mattress at the Impacted Habitat which may be due to the disturbance during installation. Adams (2016) noted that *Z. capensis* biomass displays aseasonal fluctuations and responds greatly to physio-chemical conditions and sedimentary disturbances (Duarte, 2001). This decline of *Z. capensis* at the Impacted Habitat may be due to increased turbidity during the installation period, as this has been seen to lead to loss of eelgrass within KwaZulu Natal estuaries (Adams, 2016). Despite this decline, *Z. capensis* recovered after the completion of the installation. The use of shade cloth at the edge of the installation, to control any sediment and prevent increases in turbidity, and the higher positioning of new Reno mattress on the bank landward of eelgrass beds may be the reason for this. Therefore, this installation resulted in the creation of a hybrid habitat consisting of intertidal Reno mattress habitat and adjacent eelgrass. The habitat created is similar to that of Habitat II discussed in Chapter 2. The merging of both habitats further increases the complexity thus providing new refuge for animals.

Hybrid habitats or “living shores” refers to areas where both soft and hard approaches are implemented (Currin *et al*., 2010; Dugan *et al*., 2011; Gittman *et al*., 2016). As the Impacted Habitat is a hybrid habitat it was necessary to have two control habitats for comparison. Comparisons needed to be made between the natural eelgrass beds as well as the Reno mattress present to account for natural system wide variability, with the hypothesis that the

Impacted Habitat would change from being more similar to the control eelgrass habitat, prior to the installation, to the control Reno mattress habitat after completion. The performance of artificial habitats can be strongly dependant on where structures are implemented (Franzitta & Airoldi, 2019). The fact that the Keurbooms Estuary has an estimated 64 ha of *Z. capensis* (Adams, 2016) may explain why the Reno mattress installed had no net negative impact on assemblages of invertebrates and fish and rather provided additional habitat. Many species require multiple habitat types in their life histories (Whitfield, 1990; 1999; Barnes & Hughes 1999) and therefore the hybrid habitat may allow for this (Gittman *et al*., 2016a). Benthic invertebrates are often associated with eelgrass beds (Barnes, 2010; 2017), as there is a large amount of sediment and mud for burrowing. It is interesting to note that although the installation of Reno mattress altered the type of substrata and decreased the area of soft sediment for burrowing (Appendix A), *Upogebia africana* (mudprawn), which is usually associated with soft sediment, was still present after the Reno mattress was installed (Table 3.1). The placement of Reno mattress has not resulted in the local displacement of *U. africana* in this area.

The taxa of invertebrates found in the present study were similar to those previously recorded within the Keurbooms Estuary (Duvenage & Morant, 1984; Bornman & Adams, 2005) and are also typical of those found in the lower reaches of South African warm temperate estuaries (Day, 1981; de Villiers *et al*., 1999). Fish taxa recorded during this study are typically found in warm temperate estuaries in South Africa (Whitfield, 1999) and similar to those found during the surveys within the Keurbooms Estuary carried out by Duvenage & Morant (1984) and Whitfield (1994a). One species (*Caffrogobius caffer*) in particular was only noted at the older Reno mattress control habitat and only at the Impacted Habitat after the installation of the new Reno mattress. This indicates that the installation of this erosion control structure has allowed this species, which is typically found in high shore rock pools (Butler, 1982), to utilise this habitat as noted in Chapter 2. There was no change in the abundances of dominant invertebrate or fish functional feeding groups at the Impacted Habitat after the installation of Reno mattress. This indicates that this installation did not lead to immediate changes in the composition of functional feeding groups, and therefore ecosystem functioning. Invertebrate deposit feeders were amongst the most dominant functional feeding group within the Impacted and the Control_Reno habitat, which suggests that there may be sufficient detritus and nutrients for animals to feed off within Reno mattress habitat. Artificial structures have been noted to enhance production of detritus within sedimentary habitats, however, very few studies have investigated the effects this can have on a habitat (Heery & Sebens, 2018). Carnivorous fish were found to be the most dominant group of fish across all habitats indicating that within the Keurbooms estuarine habitats these fish are abundant in this system. However, it is also a result of bias through the use of the mini BRUVs as they are known to attract carnivorous species (Harvey *et al*., 2007; Logan *et al*., 2012).

At first glance, artificial structures are found to improve the abundance of invertebrates (Connolly & Glasby, 1999) and promote the aggregation of fish (Clynick *et al*., 2008; Lowry *et al*., 2013; Franzitta & Airoldi, 2019). This may give the superficial appearance that the structure has provided a suitable habitat for species (Chapman & Underwood, 2011). In this study, only the richness of benthic invertebrates increased after the installation of the Reno mattresses, while diversity was unaffected, and the abundances of one phylum in particular Annelida increased after the installation of Reno mattress. This finding may be the reason for the increase in abundance of the functional feeding groups omnivorous filter feeder (i.e. an increase of *F. enigmaticus*) and omnivorous deposit feeder (i.e. an increase in the taxa

Capitella). Variation of richness, diversity, and abundances of invertebrates from 2018 to 2019 were also recorded at the control habitats, which suggests that changes at these sites were owing to other environmental factors and were regarded as natural variability.

Much of the research done on areas where ecoengineering has taken place is at small spatial and temporal scales (Morris *et al*., 2019). This study only looked at the immediate impact of the installation of Reno mattress, and therefore only provided a snapshot of this area. Comparisons of artificial structures of different ages revealed that differences can be persistent over time (Perkol-Finkel *et al*., 2006; Ponti *et al*., 2015) and therefore differences between habitats may only become evident over a longer time scale. The long term effects of Reno mattress on the communities of both invertebrates and fish present within the Keurbooms Estuary will require a longer term study as results in Chapter 2 show that there are clear differences between an established artificial structure (>3 years old) and eelgrass habitat. The colonisation of sessile invertebrates to new areas can take time, and during this study only two species (*Amphibalanus amphitrite* and *Littoraria* sp.) were recorded to colonise the newly installed Reno mattress. Artificial structure colonisation studies are rare within South African estuaries, however, recruitment of invertebrate species on areas experimentally cleared of mussels in the embayment of the Knysna Estuary took about five months (Radloff, 2018). It is therefore likely that the newer Reno mattress will have a greater number of species living on it in the future.

Connectivity within an aquatic environment can be important in community structure and ecosystem functioning (Bishop *et al*., 2017). It is generally accepted that artificial structures become fouled with species over time (Glasby *et al*., 2007). The newly installed Reno mattress was colonised within the first month of installation with two species of sessile invertebrates and, therefore, served as a corridor for typical rocky shore species to colonise further up into the estuary. Structures can act as stepping stones for species, especially for those marine species that have life stages dependant on hard substrata (Bishop *et al*., 2017). It has been noted that artificial structures provide a place for a number of plant and animal species that normally attach to rocky shore habitats, however, it is important to note that these epibiotic assemblages may differ from those at a natural rocky shore (Connell & Glasby, 1999; Connell, 2001). Increased connectivity may not only provide new dispersal routes for species that are of conservational importance (Perkol-Finkel *et al*., 2012) but also facilitate in the spread of invasive species (Bulleri & Airoldi, 2005; Glasby *et al*., 2007), which can be seen in the spread of the Mediterranean mussel (*M. galloprovincialis*) in the Keurbooms Estuary found within the Control_Reno Habitat. The spread of this invasive mussel has been noted in the Langebaan Estuary (Hanekom & Nel, 2002), however, these beds subsequently died off by 2001 (Robinson *et al*., 2007). Recently the spread of *M. galloprovincialis* has been recorded in the Knysna Estuary (Allanson *et al*., 2014; Pollard & Hodgson, 2016).

Although this study found no obvious immediate negative impacts from the installation of Reno mattress, site specific assessments of such structures need to be taken into consideration when implementing them. Assessing the community changes (native or non-native) on artificial structures are vital to understanding how such structures impact environments and how to manage such changes (Firth *et al*., 2013a). These assessments need to take place not only at the time of impact but also following the impact. The efficacy of Reno mattress needs to be compared to other artificial structures used in erosion control. Chapter 4 aims to discuss the overall insights gained from this study as well as the way forward within a South African context.

Chapter 4

Artificial structures- the good, the bad, and the management

4.1 The good and the bad

Globally, there is a loss of natural coastal habitats because of shoreline hardening using a variety of engineering structures (Gittman *et al*., 2016b). Because artificial structures are becoming more prevalent along coastlines (Bulleri, 2005; Airoldi & Beck, 2007; Firth *et al*., 2016a; Bishop *et al*., 2017), the ecological effects of different forms of shoreline armouring need to be assessed. There has been much debate whether coastal artificial structures are good or bad. Findings from multiple studies about the ecological efficacy of artificial habitats compared to natural habitats are contradictory (Sanabria-Fernandez *et al*., 2018), highlighting the difficulties in determining the overarching ecological impacts from coastal development. The addition of artificial structures have both positive and negative effects on the abundances and diversity of species of fish (Rilov & Benayahu, 2000; Clynick, 2007; Clynick *et al*., 2008) and invertebrates (Chapman, 2003; Firth *et al*., 2013b; Loke *et al*., 2016). For example Gittman *et al*. (2016b) found that seawalls support 23% lower species diversity and 45% fewer organisms than natural habitat. By contrast, Rilov & Benayahu (2000) found greater abundances and richness of fish at an artificial reef compared to a natural one. There have also been conflicting reports of the ability of artificial structures to provide more habitat to species (Rilov & Benayahu, 2000; Chapman & Blockley, 2009; Firth *et al.*, 2016b). Whilst studies suggest that some artificial structures can create new habitats and increase species diversity (Rebele, 1994; Connell & Glasby, 1990), others argue these structures are colonized by common and high occupancy species creating a false sense of similarity to natural habitats and giving only a superficial appearance of providing suitable habitat (Chapman & Underwood, 2011; Sanabria-Fernandez *et al*., 2018). Artificial structures have been thought to only serve as aggregation devices and do not lead to any real increase of biomass (Bohnsack, 1989; Pickering & Whitmarsh, 1997). Nevertheless, some studies point out the potential for artificial structures to be used in conservation of threatened species (Perkol-Finkel *et al*., 2012; Garcia-Gomez *et al*., 2014; Gittman *et al*., 2016a).

The shorelines of estuaries are being converted at an accelerated rate (Able *et al*., 1999; Chapman *et al*., 2018) shifting from unconsolidated soft bottoms dominated by seagrasses or saltmarshes to hard artificial structures (Airoldi & Beck, 2007; Bulleri & Chapman, 2010; Lai *et al*., 2015). Alterations of these habitats have numerous impacts, such as differing assemblages compared to natural habitats (Bulleri & Chapman, 2004; Moreira, 2006; Chapman *et al*., 2018), and shifts in species populations as certain species of functional feeding groups do not occupy artificial structures (Chapman, 2003). The present study investigated the faunal differences between an artificial erosion control structure (Reno mattress) and a natural habitat (*Zostera capensis* i.e. eelgrass) and the impact the new installation of this structure had on faunal communities in the Keurbooms Estuary in South Africa. Not all artificial structures are created equal (Heery *et al*., 2018), their designs differ resulting in different complexity of structures and therefore differ in their ability to provide habitat and sustain species (Dugan *et al*., 2011). This once again highlights the importance of studying the ecological impacts of specific artificial structures.

An important finding of the present study was the differences in species assemblages, diversity, and richness between eelgrass (*Z. capensis*) and Reno mattress (Chapter 2). For example the presence of reef associated fish species recorded at the Reno mattress habitat. Furthermore, the invertebrate community found within the Reno mattress resembled that of a rocky shore environment and did not seem to change the functional feeding group structure of the fish. However, this does not mean Reno mattresses act as suitable surrogates for rocky shores rather that they introduce hard substratum not usually associated in soft bottom habitats. Unlike other types of shore hardening structures, where decreases in species richness and diversity and changes in community structure are common, Reno mattresses did not have these impacts. Complexity of artificial structures is key in proving animals with refuge (Loke *et al*., 2014; Evan *et al*., 2016; Gittman *et al*., 2016b) and is often regarded as the best way to improve already existing artificial structures (Hellyer *et al*., 2011; Firth *et al*., 2014a). Reno mattresses are already complex structures, and while simultaneously providing protection to an eroding bank they are clearly effective in providing habitat for both fish and invertebrate species. Reno mattresses in Thesen Islands Marina in the Knysna Estuary of South Africa were also found to benefit the endangered Knysna seahorse providing additional habitat to this species (Claassens & Hodgson, 2018). Coastal artificial defence structures (breakwaters, rip-rap, concrete blocks and tetrapods) have been found to be useful in conservation of species of algae (e.g. *Cystoseira barbata* (Perkol-Finkel *et al*., 2012)), sponges (e.g. *Spongia agaricina* (García-Gómez *et al.*, 2014)), molluscs (e.g. *Patella ferruginea*, *Cymbula nigra*, and *Charonia lampas* (García-Gómez *et al.*, 2014)), echinoderms (e.g. *Centrostephanus longispinus* (García-Gómez *et al.*, 2014)), and fish (e.g. *Hippocampus brevirostris* (García-Gómez *et al.*, 2014), *Oncorhynchus tshawytscha* (Toft *et al*., 2013)).

One of the more well-known negative effects of shore hardening is that this can allow the introduction and spread of invasive species (Bulleri & Airoldi, 2005; Glasby *et al*., 2007; Airoldi & Bulleri, 2011; Dugan *et al*., 2011; Duarte *et al*., 2013; Dafforn, 2017). Invasive species can outcompete native species thus leading to community structure and trophic level changes (Grosholz & Ruiz, 2009; McQuaid & Arenas, 2009). Estuaries, in particular, are vulnerable to marine invasions most likely because of their close association with anthropogenic activities (Ruiz *et al*., 1999; Miranda *et al*., 2011). Two invasive species, the Mediterranean mussel *Mytilus galloprovincialis* and estuarine tube worm *Ficopomatus enigmaticus*, were recorded attached to older Reno mattresses (see Chapter 2) during this study. Both of these species are regarded as ecosystem engineers as they aggregate into beds and provide a substratum for other species to settle on or find refuge in (Borthagaray & Carranza, 2007; Heiman & Michel, 2020). Their presence can modify the physical environment therefore having direct and indirect effects on assemblages (Heiman & Michel, 2020). Marine mussels and oysters are not usually found in estuaries, and *M. galloprovincialis* has been found to displace native species by taking up all the space (Griffiths *et al*., 1992; Robinson *et al*., 2005). However, there is a limited amount of hard-substratum in the Keurbooms Estuary (Duvenage & Morant, 1984), and therefore few hard-substratum species to displace. The mussels present on the Reno mattress have, in fact, the potential to further increase the complexity of this structure as *M. galloprovincialis* forms complex multi-layered mats (Griffiths *et al*., 1992) possibly providing additional habitat and food sources. This raises the question: should these ecological engineering species be considered as having a large impact and therefore the use of Reno mattress harmful to an estuary, or could we consider this increase of complexity a positive impact? If more Reno mattresses are used for erosion control in south coast South African estuaries it is likely that these invasive organisms will benefit from them.

The installation in 2018 of new Reno mattress for erosion control at one site in the Keurbooms Estuary had no apparent net negative impacts on the fauna or adjacent eelgrass (Chapter 3). However, instead of replacing *Z. capensis* the newly installed Reno mattresses were placed on bare sediment thus creating a hybrid habitat. Combining hard structures and natural vegetation/habitat is referred to as a hybrid design, and may reduce negative impacts. This is because these hybrid habitats have the potential to provide more habitat for species and enhance ecosystem functioning thus merging the positives of both natural and artificial habitats (Dugan *et al*., 2011; Gittman *et al*., 2016a; b; Chapman *et al*., 2018; Morris *et al*., 2018). For example, using a combination of granite sills and saltmarsh lead to an enhancement of nursery function within an estuary in Hatteras Island, USA (Gittman *et al*., 2016a). Therefore the effectiveness of Reno mattress within the Keurbooms Estuary may be due to the presence of eelgrass within the Impacted Habitat. Claassens (2016) demonstrated, within the Knysna Estuary, that even in instances where Reno mattresses were placed in a new aquatic environment in Thesen Islands Marina they were able to provide habitat to species.

Ecological engineering studies have their own limitations including lack of replication at multiple locations (Morris *et al*., 2017), and limited time at relatively small scales. Therefore there is inadequate information to make large scale recommendations (Chapman *et al*., 2018). In the present study investigating the impact of newly installed Reno mattress was limited to the small area covered by this erosion control structure. In addition, the time scale for this study was too short to determine the long-term effects the newly installed Reno mattress had on the macrofauna in the Keurbooms Estuary. Furthermore, implementing ecological engineering practises within ecosystems can be difficult as there are different legislative aspects to consider in different countries. Currently, there is no standardised protocol for assessing the impact of installing an artificial structure on a habitat within South Africa, making comparisons between studies difficult. The use of the Before-After-Control-Impact (BACI) approach, while helpful in determining if an immediate impact has taken place, only provides a snap shot of the area in question. Long term monitoring would provide a holistic view of how Reno mattress can alter an area and eventually result in the significant differences reported in Chapter 2. Experimental studies, involving the placement of structures into areas, may be useful in determining how these structures could lead to changes in faunistic assemblages and ecosystem functioning. Extensive surveys of fish, and invertebrates in all South Africa estuaries have not been carried out since the 1980s (van Niekerk *et al*., 2011; Adams *et al*., 2020) and observing changes to an environment is difficult owing to these gaps in historical data.

4.2 Paving the way forward: management of shore hardening

When introducing coastal artificial structures, especially within estuaries, ecological based planning and management need to be taken into consideration. Over the past few decades South Africa has developed various environmental legislation, specifically focused on the coastal environment. This includes the National Environmental Management (Act No. 107 of 1998): Integrated Coastal Management Act, 2008 (ICM Act No. 24), which ensures the socially, economically justifiable and ecological sustainable development of the coast, as well as aiming to stop inappropriate development of the coastal environment. Under section 35 of the ICM Act, an Estuarine Management Plan must be developed for all estuaries of South Africa. Within the Western Cape, within which the Keurbooms Estuary is located, the Estuary Management Programme forms a priority area at both the provincial and municipal level (Western Cape Department Environmental Affairs and Development Planning, 2017). Despite having various legislation in place enforcement and compliance are often lacking (Adams *et al*., 2020), and at the national level implementation of these functions is severely under capacitated (Western Cape Department Environmental Affairs and Development Planning, 2018). The results of the present study highlight the increasing need for further study of shoreline development and armouring within South Africa, especially studies focused on artificial structures used in the building of marinas and for erosion control. Currently there are no other studies investigating the impacts artificial structures have on faunal communities within South African coastlines and estuaries. Identification and quantification of artificial structures along the coastline of South Africa are required as there is no estimate of the extent of armouring of the entire coastline, and thus no baseline to monitor and compare future developments against. While there are sections of the South African coastline that have minimal development, South Africa has a long history of coastal armouring (Swart, 1996). There are armoured hotspots (Table 1.1), particularly around coastal cities and ports for example the Eastern Cape coast at Port Elizabeth and East London (Fig. 1.1) have sections of dolosse (Tulsi & Phelp, 2009), and numerous structures have been implemented along the coastline surrounding Durban Harbour (Corbella & Stretch, 2012). Despite this there is very limited research on the ecological impacts of armouring.

4.3 Future Research

Based on research outcomes of the present study an erosion management guideline is recommended for the South African coastline, especially where there is increased urbanization and an increased use of erosion control structures as climate change continues to alter coastal environments. Marine spatial planning and standardised protocols for the implementation of erosion control structures, which take ecological sustainability into consideration, are required. Although coastal development mainly focuses on the benefits to humans, for example to support the growing population living close to the coast (Dugan *et al*., 2011, Firth *et al*., 2016b), incorporating ecological benefits as a goal would allow for enhanced, ecologically sensitive designs to the substratum. Management can aim to reverse negative impacts by implementing restoration ecology (Chapman *et al*., 2018), or by improving habitat for some species in irreversibly human altered areas- reconciliation ecology (Rosenzweig, 2003). Management of artificial structures is typically aimed at already existing structures and a number of recommendations involve altering structures through ecological engineering practises (e.g. drilling rock pools, specially designed tiles, manipulating concrete (Perkol-Finkel & Stella, 2014), and creating novel habitats) (Firth *et al*., 2016b). Ideally, management options should be considered during the design and construction phase of a project, using interdisciplinary research to achieve a synergy of benefits through the design of multifunctional structures (Firth *et al*., 2016b). It is suggested that the use of structures that already have complexity should be used over others that do not mimic a natural habitat. As not all artificial structures are the same it is recommended that comparisons of faunal assemblages between Reno mattress and other erosion control structures should be carried out. A standardised scheme outlining and quantifying the impacts of structures should also be developed to facilitate the comparison between difference types of artificial structures. It would also be beneficial to determine the ecological impact of Reno mattresses in other estuaries within South Africa to determine if this structure could be used to replace existing erosion control structures. Preliminary work carried out by Firth *et al*. (2014b) on rock-filled gabion baskets and mattresses in Wales and the Netherlands found that selection of rock sizes can enhance diversity, however, further research is needed to test whether these could be used in future management of coastlines.

The present study is the first to investigate the ecological efficacy of Reno mattress within a South African estuary, and therefore there is still inadequate data for conclusions to be drawn as to the ecological impacts on a wider scale. Before Reno mattresses can be suggested as the most suitable erosion control structure, further investigations needs to be made. For example, conducting field experiments in different biogeographic localities and types of estuaries, comparing different types of structures. Estuaries are dynamic systems (Chapman *et al*., 2018) therefore when investigating the ecological impacts of artificial structures it is important not to make sweeping statements as not one structure fits all.

Chapter 5

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Photographs of the installation of the new erosion control structure Reno-mattress along the eastern bank of Keurbooms Estuary (before installation on the left, after installation on the right).