Inter-university Master of Science in Ecological Marine Management (ECOMAMA)

THE IMPACT OF INDIRECT EFFECTS OF CLIMATE CHANGE ON MANGROVE ASSOCIATED BIODIVERSITY

Thesis submitted in partial Fulfilment of the Requirements for the Degree of Master of Science in Ecological Marine Management (ECOMAMA)

By

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Dedication

For your prayers and love, Mzee Jenoh Mrabu, Mama Kadzo Mwabonje and Dama Jenoh, my dear wife Umazi, Children; Rolleen and Tune. Siblings Ndaa, Mwaka, Malombos, Neema, Kiringis and M'rabu. Lastly Sara and Majoy- this work is wholly dedicated to you.

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Abstract

Periodic episodes arising out of global climate changes seem to pose a reasonable threat to the integrity of mangrove ecosystem. Mangrove macrofauna, which are residents of mangrove areas throughout their adult life, stand to be highly affected by the Periodic episodes arising out of global climate changes. During the 1997/8 8 El-Niño event, massive sedimentation due to erosion of terrigenous sediments caused mangrove dieback in many areas along the Kenyan coast. Mwache Creek a peri-urban mangrove forest in Mombasa was the most affected resulting in mangrove death covering about 200ha. Biodiversity in El-Niño impacted sites was compared to reference (natural forests) sites in order to assess the impact of climate change to mangrove associated biodiversity. Transects (sea-landward transect) were laid in both impacted and natural sites where relevant physico-chemical variables were measured and mangrove biodiversity determined as an indicator of ecosystem change. Molluscs densities and diversity were found not to be significantly different between treatments (impacted and reference sites) while crabs diversities was significantly higher in reference sites than impacted sites. Faunal diversity of Molluscs in impacted sites was found to be sustained by invasive shrubs while crab densities and diversity was highly reduced by the mangrove dieback.

Keywords

Macrofauna, impacted site, ecosystem, associated biodiversity.

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Chapter 1

1. INTRODUCTION

1.1 Definition of Mangroves

The term mangroves refer to an assemblage of trees and shrubs that dominate the intertidal zone along tropical and subtropical coastlines (Kathiresan and Bingham, 2001). Mangroves could also mean the tidal forest, oceanic woodlands or mangrove swamps-which are typical wetland ecosystems found in coastal deposits of mud silt (Gang and Agatsiva, 1992; Mohammed et al., 2008). These swamps are predominantly intertidal habitats that occur worldwide in the tropics and subtropics along sheltered and shallow water coastlines (Krauss et al., 2008; Nagelkerken et al., 1992). Mangroves can grow on sand, peat, rocks and corals even though they are mostly associated with muddy soils that are usually found along deltaic coast, lagoons and along estuarine environment (Saenger, 2002). The mangrove ecosystem is composed of flora and fauna which have a mutual interdependent relationship that enhances their survival in an environment that is constantly under flux (Krauss et al., 2008; Robert et al., 2009).

Mangroves form one of the most productive ecosystems in the world (Badola and Hussein, 2008; Kristensen et al., 2008) having been valued at approximately US $ 181 billions (Constanza et al., 1997; Gilman et al., 2008). Thus, they offer diverse forest goods and services to indigenous people and the ecosystem (Dahdouh-Guebas and Koedam, 2008; Walters et al., 2008). Mangrove trees have developed different morphological and physiological adaptations to make them cope with extreme intertidal conditions like salinity, wave action, anoxic soils and rhythmic inundation (Robert et al., 2009; Saenger, 2002). Due to this, they have been associated with aerial roots, vivipary, salt exclusion and salt secretion even though these are not exclusive characteristics for mangrove trees (Popp et al., 1993; Tomlinson, 1986). These adaptations partially explain the ability of mangroves to thrive in this peculiar harsh environment.

1.2 Distribution

Mangroves are distributed between latitudes 30° north and 30° south. The northern upper limit occurs in Japan (31°22'N) and Bermuda (32°20'N) whereas Australia (38° 03'S) and South Africa (32°59'S) form the southern limit of the mangrove distribution. Globally, mangroves cover an area of about 181 077 to 198 818 km² (Spalding et al., 1997), they are composed of about seventy taxonomically diverse trees, shrubs and ferns (in twenty seven genera, twenty families, and nine orders) majority of them being found in the new world. Species richness of this ecosystem is higher in the Indo West Pacific than in the Atlantic Caribbean and Eastern Pacific (Ellison et al., 1999).

The latitudinal distribution of mangroves is highly influenced by climatic factors such as sea surface and air temperature (Dahdouh-Guebas and Koedam, 2001). Fresh water input (from rainfall or water runoff) reduces soil salinity, thereby influencing the forest structure and distribution. On the other hand, temperature which has a direct influence in the upper limit of mangroves is thought to be influenced by frost stress (Kangas and Lugo, 1990) whereas the landward movement of mangroves is restricted by their inability to compete with terrestrial vegetation.

1.3 Importance of mangroves

Mangrove forests are among the most productive ecosystems in the world (Badola and Hussein, 2008). Where they occur, mangrove ecosystems have provided a plethora of goods and services (Dahdouh-Guebas and Koedam, 2008; Krauss et al., 2008). Goods provided by mangroves include all the products which can be extracted from the ecosystem for the direct or indirect usage by humans such as providence of sea food, medicines, timber, fire wood, honey and fodder for domesticated animals. Services offered by this ecosystem refer to the conditions and processes through which ecosystem and the species that make them up sustain and fulfil human life (Daily 1977 *in* Bosire, 2006). These services such as nutrients recycling, sediments accretion, and moderation of hydrological processes are the life support functions at the foundation of the mangrove ecosystem (Badola and Hussein, 2008). The goods and services derived from mangrove ecosystem can be split to four major categories namely: Provisioning, regulation, supporting and cultural (Bosire, 2006).

1.3.1 Provisioning

Coastal communities living by mangrove forests have for a long time drawn their livelihood from mangrove ecosystem. These ecosystems provide the local people with wood in order to use it for timber, fuel, poles as well as for boat construction (Ewel et al., 1998; Dahdouh-Guebas and Koedam, 2008). Mangrove wood is popular to the local people due to its anti-rot and anti-insect boring properties and its very high calorific energy value useful for fuel wood Dahdouh-Guebas et al., 2000; Field 1999). In Kenya, the use of mangrove for the purpose of house construction is the largest and most significant (Dahdouh-Guebas et al, 2000). Apart from local use of mangrove poles as building materials, the coastal communities harvest the poles for sale to licensed dealers in exchange for money. In Asia and Africa, charcoals from mangroves have been a booming business, which unfortunately due to the unsustainable harvesting practises have led to mangrove degradation (Mohammed et al., 2008).

Mangroves have been documented to support both commercial and recreational fisheries to a tune of about 70% to 90% in the Gulf of Mexico and Australia (Caddy and Sharp., 1986). Artisanal fishery, which is one of the major activity for the coastal communities, provides the locals with a source of income and protein (Dahdouh-Guebas and Koedam, 2001; Mwaluma, 2002). Fishing-normally for subsistence, is done by use of dhows and traps (uzio) in many parts of Africa (Richmond, 2002). Apart from fish, crustaceans, molluscs and prawns are also harvested mainly for sale in touristic hotels and for domestic consumption (Fratini and Vannini, 2002; Mwaluma, 2002). Crustacean species of value are *Scylla serratta* Forskål, *Thalamita crenata* Latreille and *Portunus pellagicus* L. where as mollusc of economic value are mainly oysters (Mwaluma, 2002).

Other goods harvested from mangroves include tannin which is used as dye for coating and preserving wood, nets, fishing gear, and dyeing cloths; honey harvested mainly from *Avicennia* species (Field, 1999). In Sri Lanka, mangroves have been exploited to make juice, ice cream and jam from *Sonneratia* species. Traditionally, mangroves provide also medicine to the communities. In Indonesia the net potential value of traditional medicine has been valued at € 1066 / km^2 (Ruitenbeek , 1992). Mangrove leaves have been used as fodder for livestock in some countries. Lactating cows have been found to produce more milk when foraging mangrove leaves especially *Rhizophora*. (Bosire, 2006).

1.3.2 Regulation services

Since mangroves occur at the border between land and the sea, they provide a number of important regulation services to the terrestrial world and the sea. Mangroves play an important role in shore line stabilization. This is possible due to the intricate root system which helps in controlling soil erosion (Badola and Hussein, 2008; Field, 1999). Mangrove vegetation and their root system form a barrier against storm surges (Furakawa et al., 1997) thus protecting the terrestrial world. This protection property was very evident during the 2004 Indian Ocean tsunami which destroyed human life and property. During this event shorelines with healthy mangroves were less affected in comparison to those with degraded mangroves or no mangroves at all (Dahdouh-Guebas, 2005; Kathiresan and Rejendran, 2005). Dahdouh-Guebas et al., (2005) found out that the degree of protection depended on the mangrove health. The presence of real mangrove species provided a good protection whereas areas where the mangrove had undergone some degradation and thus the mangrove was colonised by inferior species, little protection was obtained.

Furthermore, mangroves play a vital role in protecting the adjacent critical habitats namely coral reefs and sea grass beds. This is possible since mangroves act as a trap of sediments from land, heavy metals, nitrogen from domestic waste and other pollutants (Wong et al., 1995; Cannicci et al., 2008) Sedimentation alone is harmful for both sea grass and coral reef having the ability to destroy the whole system encase of severe sedimentation. Similarly, nutrient influx from the terrestrial world can lead to algae blooms which are destructive to corals reefs (Ewel et al., 1998).

Recently mangroves have been suggested to be capable of absorbing organic waste from sewerage. Hence they are being suggested as a novel method for domestic sewage disposal. (Mohammed et al., 2008b; Cannicci et al., 2008; Wong et al., 1995) Through the process of photosynthesis Mangroves fix and store significant amounts of carbon and thus play an important role in carbon sequestration by absorbing an estimated 25.5 x 10 ton C a year (Alongi, 2002; Ong, 1993).

1.3.3 Supporting services

Mangroves have been suggested to support near shore fisheries production due to its high organic productivity (Lee, 1998). The organic materials produced by the system are thought to be a basis for complex food-webs which in turn support a wide variety of marine life in the adjacent ecosystems. In addition, they offer rich nursery and breeding ground for many fish species (Richmond, 2002). The link between mangroves and the adjacent habitats is considered to be tides, currents and the nursery role of mangroves to many fish species. Tides are able to transport organic matter to the nearby ecosystems during out going tides, whereas currents lead to trapping of larvae in mangrove fringed channels (Chong et al., 1990). Otherwise the link is by the nursery role of the mangroves (Crona and R□nnb□ck, 2005) in the case of some fish species feeding directly on mangrove detritus. The mangrove system harbour wide range of resident and visiting fauna, they include mammals, insects, reptiles and even migratory birds. (Kairo et al., 2008).

1.3.4 Cultural services

Many coastal communities living by the mangroves have set aside parts of mangrove forest for religious or cultural practises. In these areas, which are mainly used as shrines, harvesting of trees is highly forbidden therefore maintaining the pristine condition of the forest. Recently tourism has targeted this cultural forest that has played a major role in cultural preservation (Bosire, 2006).

1.4 Mangroves of East Africa

Mangroves of east Africa occur along many coastal stretches of the Western Indian Ocean (WIO) region. This region is composed of Southern Somalia in the north to the Kwa Zulu-Natal coast of South Africa and along the coast of Madagascar. Other countries where mangroves exist in the WIO region include Comoros, Kenya, Mauritius, Mayotte, Mozambique, Reunion, Seychelles and Tanzania (Richmond, 2002).

The east Africa region is home to ten species of mangrove trees and several shrubs. Mangrove tree species found in East Africa are: *Rhizophora mucronata* Lamk, *Sonneratia alba* J.E. Smith, *Avicennia marina* (Forsk.) Vierh, *Ceriops tagal* (Perr.) C.B. Robinson, *Bruguiera gymnorrhiza* (L.) Lam., *Heritiera littoralis* Dryand, *Xylocarpus granatum* König, *Lumnitzera racemosa* Willd, and *Pemphis acidula* Forst. *Xylocarpus moluccensis* (Lamk.) Roem (Dahdouh-Guebas et al., 2000; Kairo, 2001). How ever the true identity of *Xylocarpus moluccensis* is still debatable with scientists agreeing on its genus but doubting the species identity (Dahdouh-Guebas and koedam, pers. Comm., 2009). In the landward side of east Africa mangrove swamps exists vegetation assemblages of shrubs grasses and other lower plants. Dominant among these are *Sesuvium portulacastrum* L, *Arthrocnemon indicum (*Wild.) Mog. *Salicornia* species and several species of sedges (Richmond, 2002). Table 1 shows mangrove cover in the Western Indian Ocean (WIO) region.

Table 1: Mangrove species of the WIO countries and the respective mangrove forest area coverage.

Species	Somalia	Kenya	Tanzania	Madagascar	Seychelles	Mayotte	Reunion	South Africa
R. mucronata	√	√	√	√	√	√		√
C. tagal		√	√	√	√			√
B. gymnorrhiza		√	√	√	√			
A. marina		√	√	√	√			
S. alba		√	√	√	√			
H. littoralis		√	√	√	√			
X.granatum	x	√	√	√	√			
L. racemosa		√	√	√	√			
A. offinalis		x	x	√	x			x
P. acidula		√						
X. moluccensis		√						
Total species		9	8	9	9	1		2
Total cover Km²	910	530	1,155	3,403	29	10	NA	11

Compared to Southeast Asia, African mangroves have a relatively low diversity of mangrove trees. These mangroves provide critical services for maintaining nearby coral reefs and populations of fish and birds as well. In East Africa, Tanzania has the largest mangrove forests which are found along Rufiji River that runs through Tanzania creating an enormous delta (Richmond, 2002).

1.4.1 Mangroves of Kenya

Kenya has ten species of mangroves listed in east Africa. These *are A. marina, R. mucronata, S. alba, B. gymnorrhiza, C. tagal, L. racemosa, X. moluccensis, H. littoralis, X. granatum* and *P. acidula* (Dahdouh-Guebas et al., 2000; Kairo and Dahdouh-Guebas, 2008). Among these mangroves, *H. littoralis* and *X. moluccensis* are rare species whereas *R. mucronata, A. marina, C. tagal* and *B. gymnorrhiza* are the most encountered (Kairo and Dahdouh-Guebas, 2008; Richmond 2002).

In Kenya, mangrove forests are found along the coastal strip extending from Lamu (which harbours the largest mangroves in Kenya), Malindi, Kilifi, Tana River and Mombasa districts (figure 1) (Duet et al., 1981; Kairo and Dahdouh-Guebas, 2008). Different government departments and individual researchers have reported different areal coverage for the mangrove forests (Table 2). According to a Kenyan survey of land resources 1982, mangrove area is reported to cover 52,980 ha whereas the Kenya forest department reports 64, 426ha. The world mangrove atlas gives a figure of

96,100 ha and 61,000 ha as reported by Taylor et al., (2003). However, the figure 53,000ha is thought to be more reliable (Spalding et al., 1997). These inconsistencies are thought to be due to differences in estimation techniques, time when the survey was done, classification and delineation of areas considered to be mangrove ecosystem (Kairo and Dahdouh-Guebas in press). Figure 1 shows the major mangroves forests of Kenya divided into two broad regions north and south of Tana River.

Table 2: Distribution and area (ha) of mangrove forests coverage in Kenya

District	Doute et al., 1981	Forest Department
Lamu	33,500	46,229
Tana River	2,665	3690
Kilifi / Malindi	6,606	6,378
Mombasa	1,960	3,059
Kwale	8,795	6,345.5
Total	52,980	64,426.9

*Formerly, Kilifi District also included Malindi. There are no separate data on the mangroves for the present-day Kilifi District and Malindi District (Source: Kairo & Dahdouh-Guebas, 2008.)

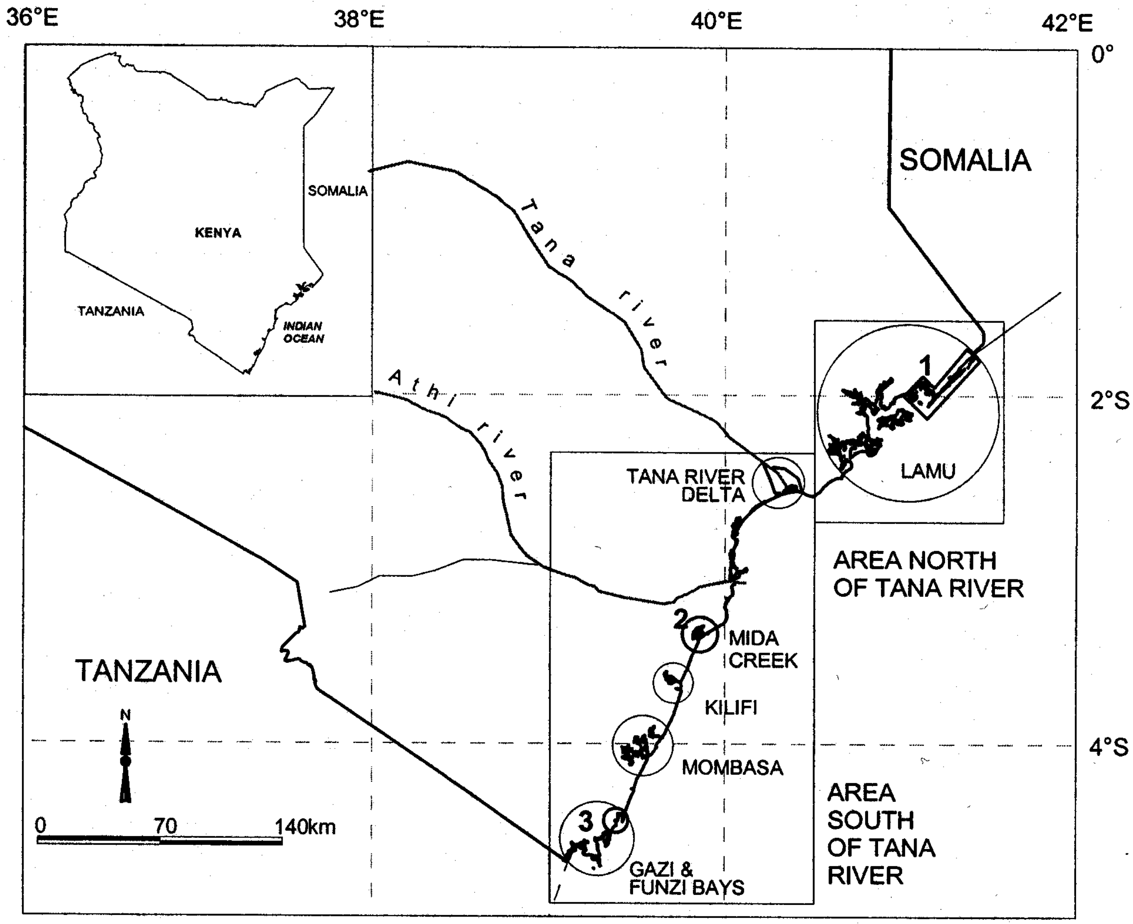

Figure 1: The coastline of Kenya showing major mangrove areas, (adapted from Kairo, 2001).

Mangroves zonation in Kenya displays a distinct pattern which is highly influenced by soil water, salt content and level of inundation (Kairo and Dahdouh-Guebas in press). In most cases, the zonation follows a sea-landward pattern in the following order: *S. alba, R. mucronata, B. gymnorrhiza, C. tagal, A. marina, X. granatum, L. racemosa.* However, there exists a high variability, thus this systematic pattern is not fully evident in all mangrove forest in Kenya (Dahdouh-Guebas et al., 2002). *A. marina* displays two distinct zones; one on the landward side and another on the seaward side (Dahdouh-Guebas, 2002b). On the other hand, *S. alba* forms a narrow often interrupted strip at the seaward forest margin. Following this zone there is a mixed stand of *R. mucronata* and *A. marina* which may be followed by either a pure or mixed stand of *C. tagal* and *A. marina*. *B. gymnorrhiza* does not form an independent zone but occurs interspersed within *A. marina, R. mucronata* and *C. tagal* stands. *L. racemosa* usually occurs as a small intermittent fringe beyond the higher *A. marina* zone, but it is also encountered within this zone. *Xylocarpus* and *H. littoralis* occur in a more localised distribution thus does not contribute to the zonation pattern.

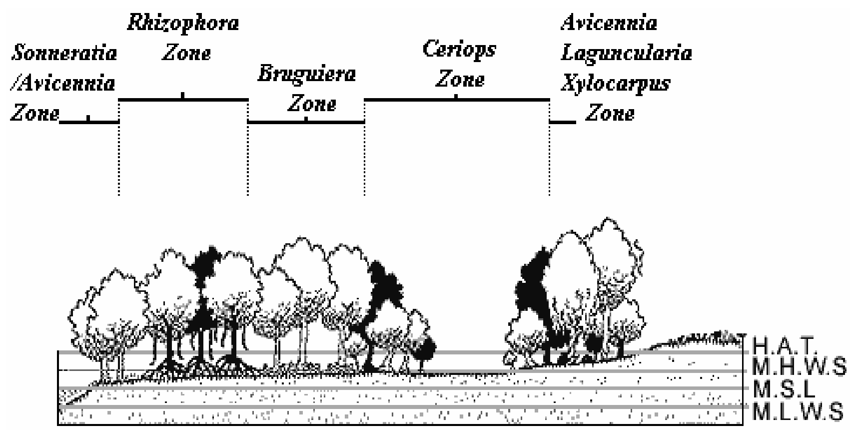

Figure 2: A diagram depicting the zonation of mangroves in Kenya.

In Kenya all forests, mangrove forest included, are managed by the Kenya Forest Department. This department is responsible for issuing licences to individuals involved in harvesting. However they have a limited capacity to supervise cutting operations or any illegal harvesting due to lack of resources. Lack of capacity to effectively supervise forest harvesting has greatly compromised the sustainable forest management in Kenya (Bosire, 2006).

1.4.2 Use of mangroves by Kenyan coastal communities

Kenyan coastal communities living nearby mangrove forests have for a long time drawn their livelihood from mangrove forests (Kairo, 2001; Mwaluma, 2002). These communities exploit this resource to gain mangrove poles which are useful for building. Mangrove poles from different species are used in different parts of the house during construction. *Rhizophora* is preferred for the walls especially the thicker supportive poles and corner pillars of a house. *Ceriops* which is locally known as “Fito” is used on the walls interweaving the *Rhizophora* poles. Whereas *Bruguiera* poles are used on the roof tops where palm thatching materials commonly referred as “Makuti” are attached to them (Dahdouh-Guebas and Koedam, 2001). Apart from local use of these poles as building materials the local communities harvest the poles for sale to licensed dealers. (Mohamed, 2008; Dahdouh-Guebas et al., 2000).

Artisanal fishery is one of the major activities for the coastal communities providing the locals with a source of income and protein (Dahdouh-Guebas and Koedam, 2001; Mwaluma, 2002). Fishing is normally subsistence done by use of dhows and traps (uzio) (Richmond, 2002). Apart from fish, crustaceans, molluscs and prawns are also harvested mainly for sell in the tourist hotels and for domestic consumption (Fratini and Vannini, 2002; Mwaluma, 2002; Nagelkerken et al., 1992). Crustacean species of value are *Scylla serrata* (Forskål) *Thalamita crenata* (Latreille) and *Portunus pelagicus* (L) where as molluscs of economic value are mainly oysters (Mwaluma, 2002).

Mangrove forest have for long time provided cultural benefits to the local people. Deep in the forest the local people have sacred areas ("Kaya") which they use for their religious and cultural activities.

Currently, the government is encouraging eco-friendly ways of sustainable utilisation of these forests. This includes ecotourism, where the local communities benefit directly from the local mangrove forests by the construction of walk boards and training of tour guides. Local communities also participate in other activities like beekeeping, reforestation and controlled wood harvesting (Mohammed et al., 2008). Sustainable aquaculture is now being introduced to the local communities with the aim of finding an alternative source of income and food (Mwaluma, 2002) although a lot must be done to reach a reasonable level.

Beside the importance of mangrove forest to local communities, these forests support a vast community of fauna which play an important role in the overall functioning and health of the mangrove ecosystem (Bosire et al., 2003). Mangroves of Kenya serve as breeding grounds for many species of fish, molluscs, crustaceans and migratory birds (Kimani et al. 1996; Taylor et al., 2003). They also support mammals, insects, and a lot other lower taxa like bacteria and arthropods. Reptiles like snakes, lizards and even crocodiles have been reported in the mangroves of Kenya (Kairo and Dahdouh-Guebas, 2008).

Mangrove fauna are broadly classified into two groups namely; infauna which are the animals that burrow in the sediment, and epifauna which refers to the animals that live on the sediments and the trees. The focus of this study was on epifauna specifically the study focused on crabs and molluscs.

1.5 Challenges and threats facing mangroves management

Despite the ecological, social and economical importance of mangroves, they have one of the highest rates of degradation of any global habitat - exceeding 1% of mangrove area per year (FAO, 2005; Valiela et al., 2001). The global loss of mangrove forests, estimated at a reduction of 35% of historical area, is recognized as an ecologically important phenomenon (Valiela et al., 2001). Over exploitation, clear cutting of forests and pollution are amongst the major courses for decline of mangroves (Alongi, 2002; Farnsworth and Ellison, 1997). Even though accurate data on mangrove degradation in different world regions is currently not availlable, it is known from existing information that Asia and Latin America - where the world's most expansive and developed mangroves exist-has suffered great mangrove loses (Bosire, 2006; Dahdouh-Guebas and Koedman, 2006; Duke et al., 2007). Figure 3 summaises the courses of mangrove degradation.

In many parts of Asia and Latin America, shrimp farming has been the main course for the loss of mangrove forests. Countries like Vietnam, Indonesia, Thailand Ecuador and the Phillipines lost huge areas of mangrove forest during the shrimp farming boom (Field, 1998; Spalding et al., 1997). Even though this venture provides an important source of income to the affected countries, it is not sustainable since it is a resource intensive venture (Spalding et al., 1997) and the fact that it turns the multi-use mangrove systems to an unsustainable single-use enterprise (Bosire, 2006) . In many cases, some biodiversirty components in such disturbed mangrove forest may persist, even though the conversion of this ecosystem to other unsustainable uses leads to a net loss of goods and sevices and a "cryptic" degradation (Bosire, 2006).

Apart from shrimp farming, mangrove systems have been degraded by reclaimation of forest land to pave way for residential houses, tourist installations, agricultural ventures, diversion of fresh water and unsustainable mangrove harvesting for local or commercial use (Dahdouh-Guebas et al., 2004; Dahdouh-Guebas et al 2005; Kairo et al., 2002). Other economic activities like salt extraction and attempted but neglected shrimp farms have coursed mangroves degradation. Worthy of remark is the example of Ngomeni area in Kenya. In addition, both Asia and Africa mangrove forests face the problem of increased deforestation due to cutting down of mangroves for charcoal and fire wood. This is a major problem especial in peri urban mangroves forest (Mohammed et al., 2008b).

Accidental oil pollution in mangrove areas is also a major problem especially for mangroves that are near harbours. In developing countries, mangroves, which have for long been viewed as waste lands have been used as damping sites, sewage disposal areas in peri urban setting and have also suffered from oil spill from tanker accidents as was the case of Tsunza bay in Kenya (Mohammed et al., 2008). Whereas the effects of the use of mangroves as sewerage disposal sites have yet to be fully understood, (Cannicci et al., 2009). A recent research done in East Africa shows that some species of crabs i.e. *U. vocans* totally disappear on exposure to sewerage whereas some mollusc species increase in size compared to the molluscs in pristine sites. In this study, Cannicci et al., (2009) could not fully ascertained the real effects of sewage disposal on flora and fauna since the sampling sites were also experiencing other problems like sedimentation and severe human logging which greatly compromised the forest integrity. However, other than the obvious human stresses on mangrove ecosystems, periodic episodes arising out of global climate changes seem to pose a reasonable threat to the integrity of mangrove. Thus climate change related events, have been classified as the most destructive courses of mangrove ecosystem degradation. (Mohammed et al., 2008).

1.5.1 Climate change as a threat

In addition to anthropogenic induced mangrove degradation, global climate change is further threatening the resilience of mangroves and other ecosystems like the coral reefs (Kitheka et al., 2002; Mclanahan et al., 1988;2008 Mcleod, 2006). Climate change in the globe is predicted to have +60.0cm sea level rise, 840 ppmv (parts per million volumes) for atmospheric CO_2 increase, and +3.0 °C increase in global mean temperature changing hydrologic regimes, sedimentation and increasing tropical storms and intensity (Field, 1995; WWF, 2003). These rates of change if it doesn't get worse, worlds mangrove ecosystems will be affected by temperature increase, rising sea levels, increase of CO_2 and increase in tropical storms and their magnitude. Even though global climate change is a reality, (IPCC, 2007) the effects of this phenomena on mangrove ecosystems need to be further investigated in order to fully understand them since up to now they remain unclear. (Alusa, and Ogallo, 1992)

Since mangroves forests are among the most prominent ecosystems in the low lying coastal areas of the tropics, they are likely to be the first ecosystems to be affected by global climate change especially in respect to sea-level rise. A rise in sea level, for instance, is predicted to increase flooding of the low-lying coastal areas and drown mangroves (Field, 1995) thus affecting not only the floral component of mangroves, but much more the fauna which reside in the mangrove in an intricate association with the floral component (Hekstra). The flooding condition closely simulates the effect of flooding on mangroves experienced during episodes of abnormally high rainfall during the 1997/8 El-Niño flooding in Kenya.

Climate is an integral part of ecosystems, organisms have thus adapted to their regional climate over time. A change in the frequency and intensity of climate related events have the potential to alter the mutual interdependent, relationship between the mangrove plants and the macrofauna existing in the mangal ecosystem. This scenario may lead to the stoppage of the many resources and services provided to each other between the mangrove trees and the macrofauna (IPCC, 2007; Nagelkerken et al., 2008).

The destruction of mangrove ecosystem due to climatic events will have a negative effect on mangrove biodiversity which is widely regarded to be important in maintaining genetic richness, ecological functioning and the resilience of the whole ecosystem (Nagelkerken et al., 2008; Bosire et al., 2008). Improved understanding of biodiversity, the foundation of the broad range of forest goods and services is critical for conservation and mitigating impacts of expected climate change on mangrove ecosystems.

1.5.2 The El-Niño/ IOD phenomenon

El Niño-Southern Oscillation is a periodic change in the atmosphere and ocean of the tropical Pacific region. It is manifested in the atmosphere by changes in the pressure difference between Tahiti and Darwin, Australia, and in the ocean by warming or cooling of surface waters of the tropical Eastern Pacific Ocean. El Niño occurs when water in that region is warmer than normal while La Niña refferes to the period when the water there is colder than normal (Eisenman et al., 2005).The periodicity of the oscillation has no well-defined period, but instead occurs every three to eight years. Mechanisms that sustain the El Niño - La Niña cycle remain a matter of research.

The Indian Ocean Dipole (IOD) is an oceanographic phenomenon affecting climate in the Indian Ocean region. The IOD involves an aperiodic oscillation of sea-surface temperatures, between "positive" and "negative" phases. A positive phase sees greater-than-average sea-surface temperatures and greater precipitation in the western Indian Ocean region, with a corresponding cooling of waters in the eastern Indian Ocean—which tends to cause droughts in adjacent land areas of Indonesia and Australia. The negative phase of the IOD brings about the opposite conditions, with warmer water and greater precipitation in the eastern Indian Ocean, and cooler and drier conditions in the west.

The IOD also affects the strength of monsoons over the Indian subcontinent. A significant positive IOD occurred in 1997-8, with another in 2006. The IOD is one aspect of the general cycle of global climate, interacting with similar phenomena like the El Niño-Southern Oscillation (ENSO) in the Pacific Ocean. The positive IOD in 2007 evolved together with La Niña which is a very rare phenomenon that happened only once in the available historical records (in 1967). Its current occurrence is thought to be due to the on going global climate change. Also the occurrences of consecutive positive IOD events are extremely rare with only one such precedence within the records (during 1913–14). A study by Ummenhofer et al., (2009) has demonstrated a significant correlation between the IOD and drought in the southern half of Australia, in particular the south-east.

The 1997/8 and 2006 abnormal high precipitation in the Indian Ocean region are a good example of extreme events caused by climate change. The abnormal high precipitation led to erosion, high temperatures, massive sedimentation flooding in the coastal lowlands and extensive mangrove dieback (Kitheka et al., 2002; Mclanahan et al., 2008). Documented information indicates that El-Niño coupled the more pronounced phenomenon referred to as the Indian Ocean Dipole (IOD), which is climate change related caused this abnormally high precipitation in the Western Indian Ocean region, with the Eastern Indian Ocean experiencing severe droughts (Saji et al., 1999; Overpeck and Cole., 2007, Bosire et al., 2009).

1.5.3 Threats of climate change to associated biodiversity

Worldwide, ecosystem biodiversity under threat from a number of natural as well as human induced pressures, climate change will be an additional stressor (Desanker, 2002; Alongi and Carvalho, 2008). The potential threats of climate change include raising temperatures which will trigger melting of the continental glaciers. Which intern will leads to the sea level rise. Leavens et al., (2006) predicts that Climate change will reduce biodiversity in coastal wetlands as a consequence of compression of wetlands gradients, resulting in inefficient trapping of sediments and nutrients. Great percentage of plant species's suitable habitats will decrease in size or shift due to climate change consequently triggering species migration. McClean, (2005) predicted by 2085 between 25 percent and 42 percent of the African species' habitats are expected to be lost due to climate change alone.

In the face of impending global climate change, the danger of biodiversity reduction is today a reality. Prolonged Flooding for instance, may lead to habitat loss for some land dwelling and burrowing faunal species. It may also lead to the death of some floral species and the fauna that depend on them as their habitat and food. The resultant death of trees for instance can initiate a cascade of ecological harmful events such as canopy gap creation, sedimentation due to soil erosion, change in canopy micro climate, loss of associated flora and fauna and their associated biodiversity, and change in hydrological and biogeochemical cycles (Alongi and Carvalho, 2008). Temperature increase could impact the montane biodiversity especially those with limited ability to move up in elevations, (UNFCCC, 2006). Wetlands ecosystem could be impacted by a decrease in runoff since climate change may lead to change in hydrological patterns whereas some wildlife species could be affected by the expansion of the range of some vectors and infectious diseases (UNFCCC, 2006).

Habitat loss and migration triggered by climate change, may lead to changes in faunal assemblage as migrating species move to suitable areas (Parmesan and Yohe, 2003). Changes in biota of ecosystems reduces genetic and species diversity. These biotic changes will likely influence ecosystem Processes to sufficiently alter the future state of the world's ecosystems and the services that they provide to humanity. (Chapin et al., 1997; UNFCCC, 2006).

It is evident that biodiversity affect both the day-to-day functioning of ecosystems and the resilience with which ecosystems respond to environmental stress. Biodiversity also provide a framework for predicting how future changes in biodiversity might influence ecosystem processes that are relevant to society. Reduction of biodiversity leads to reduced efficiency in ecosystem resilience and functioning and a loss of a valuable indicator of ecosystem health (Levin et al., 2001; UNFCCC, 2006).

In the ecosystem organisms possess different traits and they occur in different abundances and diversity hence they play an important role in determination of the ecosystem and the landscape traits it possesses. Individual organisms, for instance, gain carbon and nutrients from the environment, transfer plant tissues to higher trophic levels thus nutrients cycling, and decompose plant litter. (Levin et al., 2001; UNFCCC, 2006). Species also have substantial indirect effects on ecosystem processes through shading, thermal insulation, tissue-quality effects on decomposition, and the like (Levin et al., 2001). Global Climate change could therefore have substantial indirect effects on ecosystem processes through its effects on the species composition and diversity and assemblage of communities (UNFCCC, 2006).

1.6. Mangrove crabs

Mangrove crab species are adapted to occupy all sorts of microhabitats, from permanent mangrove channels to the mangrove canopy (Vanini et al., 1995; Kairo and Dahdouh-Guebas, 2008) and are known to play an important ecological role within the ecosystem (figure 4) (Jones, 1984; Lee 1988; Dahdouh-Guebas et al.,2002). Within the mangroves of east Africa, brachyuran crabs are the dominant taxa, both in terms of biomass and species richness (Jones, 1984; Ruwa, 1997; Kathiresan and Bingham). Some of the commonly encountered crabs families are; Calappidae, Eriphiidae, Gecarcinidae, Grapsidae, Macrophthalmidae, Ocypodidae, Oziidae, Pilumnidae, Portunidae, Sesarmidae, Scopimeridae and hermit crabs. (IPCC, 2007; Nagelkerken et al., 2008; Cannicci et al., 2009).

Mangrove crabs play a significant role in the functioning of the mangrove ecosystem. These crabs are known to aerate the anoxic mangrove soil through the burrows they make on the sediments. These burrows allow air to reach depth that would otherwise be very anoxic for organisms to survive hence creating a microhabitat for fauna and affecting the sediment chemistry and forest productivity (Smith et al., 1991; Gillikin and Kamanu, 2005; Cannicci et al., 2008). The crab burrows also provide an efficient mechanism for exchanging water between the anoxic substrate and the overlying tidal water in the process they helps in salinity regulation in the sediments (Gillikin et al., 2004). Studies done on a sersarmid crab burrow which was also inhabited by a piston prawn indicated that the burrow was completely flashed within 1h by the activity of the two within one tidal event (Nagelkerken et al., 2008). Thus Crab burrow not only help in water exchange in the depth but also provide habitat to other fauna like the piston prawn and some mangrove mosquitoes (personal observation).

Crabs have been noted to be key players in determining the mangrove community structure by actively determining the growth of mangrove seedlings through predation of propagules (Cannicci et al 2008). In the *A. marina, R. mucronata* and *Bruguiera* forest zones an inverse relationship exists between the dominance of a given tree species in the canopy and the amount of seed predation by crabs (Nagelkerken et al., 2008). Mangrove trees and different crab species have evolved a mutual relationship with crabs. In this relation, the crabs benefit from getting suitable habitat provided by the trees whereas the mangrove trees benefit from reduced competition between mangrove plants species through selective predation on seedlings (Bosire et al., 2005) Apart from the sersarmid crabs, land crabs and hermit crabs also have been found to play the role of selective predation. Selective predation of propagules is not all positive since in some instances it can course negative effects on regeneration of mangrove stands (Dahdouh-Guebas et al., 1997; 1998).

The mangrove under story is often free from any fallen leaves. This is due to their removal by graspid crabs that quickly remove the fallen leaves and take them to their burrows (Fratini and Vannini, 2002; Olafsson et al., 2002). In this way the graspid crabs play a major roll in mangrove litter turnover and nutrients cycling in the forest (Fratini and Vannini, 2002; Olafsson et al., 2002). Although sesarmids and ocypodids can consume up to 100% of the mangrove leaf litter, crabs' assimilation rate of the leaf litter is generally low (<50%), and about 60% of the dry mass of the material consumed is egested as faecal matter (Lee, 1993), resulting in high faecal rate production by crabs. The physical and chemical conditions of mangrove leaf litter changes noticeably during the digestion process of crabs, enhancing the nutritional qualities of crab's faecal material, which is thus exploited by both small autochthonous and alloctonous benthic invertebrate consumers (Olafsson et al., 2002; Cannicci et al., 2008) hence enhancing the biodiversity and functioning of the mangroves.

Through their burrowing activity, litter removal and faecal matter; Crabs tend to support many other benthic organisms which in turn become food to inshore fishes that visit the mangrove during high tide and other vertebrate predators like birds and reptiles. (Cannicci et al., 2008; Olafsson et al., 2002). Thus mangrove ecosystem have been thought to support the adjacent ecosystem by offering breeding grounds, feeding grounds and by sediment traps thus protecting the adjacent coral reefs and sea grass beds.

1.6.2. Mangrove molluscs

Mangrove molluscs and crustaceans (decapods) are the most well represented taxon of marine origin in mangrove forests. Molluscs high diversity is thought to be due to the availability of a varied range of microhabitats in the mangrove (Cannicci et al., 2008). Mangroves, molluscs are represented in all the levels of food web, as predators, herbivores, detritivores and filter feeders. They are zoned both horizontally (i.e. along the sea-land axis) and vertically and include both mobile and sessile species (Cannicci et al., 2008; Vannini et al., 2006). However, the overall ecological role of molluscs' and the effects they exert within the mangrove ecosystem is not fully understood (Cannicci et al., 2008).

Mangrove molluscs have an important role in the determination of mangrove community structure through predation of propagules. Gastropods have been known to actively compete with crabs for food resources (propagules and leaves) (Fratini et al., 2004) hence affecting propagule establishment and eventually influencing the forest structure (Dahdouh-Guebas et al., 1997; Dahdouh-Guebas et al., 1998; Vannini *et al.,* 2008). Gastropods perform diurnal and nocturnal movements either vertical or horizontal in search for food and while avoiding the high tide (Vannini et al., 2007; 2008). *Terebralia palustris* has been recorded to reach a distance of 0.80 m in 24hrs whereas *Cerithidea decollata* has been known to migrate vertically during high tide and horizontally during low tide (Vanini et al., 2006; 2007; 2008). As *C. deccollata* L. and young *T. palustris* Linnaeus move on the surface they ingesting sediments. As they forage on sediments, bacteria and leaves, they release some slime which binds the soil surface together hence talking part in soil stabilization in the mangrove (figure 4).

Mangrove gastropods play a vital role in trapping additional primary production before it is removed by ebbing currents. Leaves that fall from mangrove trees form their diet. Once a leaf has fallen gastropods like the mud whelk actively pursues the leaves or propagule by using chemical cues to locate the leaves during both low and high tide (Fratini et al., 2004). *C. decollata* and young *T. palustris* feed on mud surface (during low tide for *C. decollata*) whereas *Littorinids*, the only molluscs in East Africa that occur on foliage, feeding on green leave thus contributing to nutrient recycling (Vannini et al., 2006; 2007;2008).

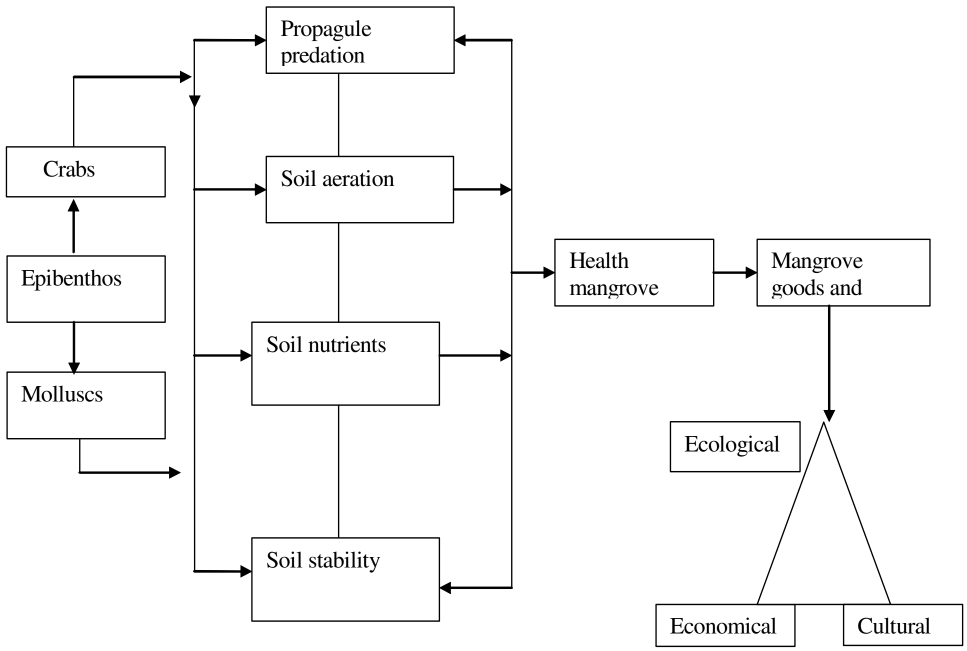

Figure 3: A flow chart showing the function of mangrove crabs and molluscs

Crabs are important in propagule predation hence influencing forest tree recruitment they also play a major role in soil aeration through burrow formation and they play a role in nutrient recycling by feeding on the mangrove vegetation that has fallen on the mangrove floor. Molluscs contribute to propagule predation and thus also influence the forest natural regeneration. The slime they leave behind as they move help in soil stabilization. By feeding on leaves and propagules, Molluscs contribute to the soil nutrients recycling within the forest. In overall this roles significantly contribute to making a healthy mangrove ecosystem which in turn lead to better mangrove goods and services.

1.7 Justification

The El-Niño rain of 1997/8 is a good example of the devastating effects of events triggered by the global climate change on the whole mangrove ecosystem. During the El-Niño event, massive sedimentation due to erosion of terrigenous sediments caused mangrove dieback in many areas along the Kenyan coast. Mwache Creek a peri-urban mangrove forest in Mombasa was the most affected resulting in mangrove death covering about 200ha. Mangrove macro fauna, which are residents of mangrove areas throughout their adult life (Fratini et al., 2004; Skov and Hartnoll, 2002) stands to be highly affected by like events of climate change.

This study therefore aimed to assess the mangroves impacted by 1997/98 El-Niño phenomenon in Kenya. In particular, the study assessed the impact of El-Niño phenomenon on mangrove biodiversity particularly crabs and molluscs.

1.7.2 Scientific hypotheses

Biodiversity in El-Niño impacted sites is low compared to reference (natural forests) sites.

Biodiversity is the variability of all living organisms - including animal and plant species - of the genes of all these organisms, and of the terrestrial, aquatic and marine ecosystems of which they are part. In this paper biodiversity is used to refer two aspects

1. The species diversity –which refers to the both flora and fauna species richness, and the interaction between them.

2. The ecosystem biodiversity referring to both biotic and abiotic part of the community and their interaction

1.7.3 Main Objectives

The main objective of this study was to assess the indirect impact of climate change to mangrove associated biodiversity.

1. Measure relevant physico-chemical variables e.g. sediment nutrients, temperature, salinity, organic matter, moisture and grain size.
2. Determine mangrove biodiversity (crab, mollusc and species abundance and diversity in the impacted sites as an indicator of ecosystem change).

Chapter 2

2 Materials and methods

2.1 Study area

This study was conducted at Mwache creek (fig 5). This creek is located in the upper Port Reitz ria (Kitheka et al., 2003), Mwache creek (403.01’ S & 39.06038.06’E) is found 20 km Northwest of Mombasa city in Coast Province of Kenya. The total area of the wetland is approximately 17 km² with about 70% of the surface area being covered with mangroves. The creek has both basin and riverine mangroves and a distinct mangrove-fringed channel in the lower sections. The mangrove species found in Mwache Creek are*: A. marina, R. mucronata, C. tagal and S. alba* (Kitheka et al., 2002).

The creek receives freshwater from Mwache River, which is seasonal and thus there is usually no flow during the dry season especially between December and March, and July and September. The rate of sedimentation within Mwache River basin reaches a high of 3,000 tons yr due to poor land-use activities e.g. overgrazing, shifting cultivation, cultivation on steep slopes without the application of soil conservation measures, high rainfall intensity during the rain season and steep land gradient among others (Kitheka et al., 2002).

The high erosion rate and sedimentation led to severe mangrove dieback due to smothering of mangrove roots as a result of excessive input of terrigenous sediments especially at the landward zone (Kitheka et al., 2002) during the October – December 1997 El-Niño Southern Oscillation (ENSO) related flooding of Mwache River. The most extensively affected species was *R. mucronata*, whereas *A.marina* was relatively less affected. The area affected is about 17% of the total mangrove forest acreage of the creek.

Reversing monsoon winds are the determining factors of Climatic condition in the area. Between October and March there is the North East Monsoon wind (NEM) whereas between April and October there is the South East Monsoon wind (SEM).

Hot and dry periods of the year fall during NEM whereas the cool wet season occurs during the SEM (Mclanahan, 1988). Due to the sea the area experiences the convectional kind of rainfall.

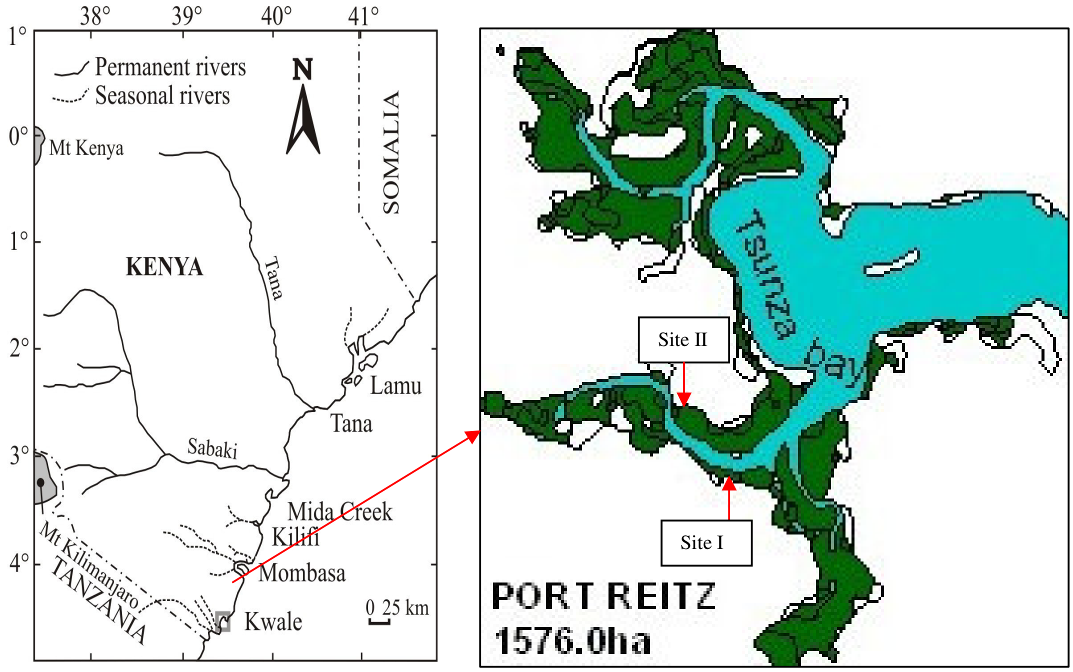

Figure 4: A map of the study area showing location of the two study sites.

2.2 Sampling design

The study was conducted during the spring tides of July and August 2008. To assess the impact of the mangrove die-back on associated biodiversity, two degraded sites (S1D and S2D) within Mwache Creek were used as the experimental units for this study (figure 4). Each degraded area was sandwiched between natural forests (S1F, S2F and S1F2) which were either not impacted or relatively less impacted during the El-Niño event and were thus used as reference sites. The first degraded site had a parallel reference site and an additional one at the landward side of the impacted site. Two transects perpendicular to the shoreline were made at the first site. In the second degraded site, the reference plot was wider than long hence only four quadrats could fit, thus three parallel transects were made. Transects were placed 10m apart to ensure independence of sampling units. Along each transect, 5m x 5m quadrats were made after every 100m. Within the 5m x 5m quadrats, one 2m x 2m sub-quadrant was randomly placed in the 5m x 5m quadrant for actual sampling. A minimum distance of 100m between the two treatments (impacted and natural forest) was maintained in placement of transects in order to avoid 'noise' during sampling.

2.2.1 Physico-chemical variables

In each sub-quadrat above, a total of five sediment cores (replicates) were randomly taken using a hand corer of diameter 6.4 cm to a depth of 15 cm during low spring tide. The samples were put in a plastic bag and stored in cooler boxes with ice. Samples were then transported to the lab for analysis. Three replicates were used for nutrients (NO_2^-, NO_3^-, NH_4^+, and PO_4^{3-}) and organic matter analysis while the remaining two replicates were used for grain size analysis. From one of the holes made by the removal of the sediment cores, interstitial salinity, temperature, dissolved oxygen and conductivity were determined by direct reading using a universal multimeter (Hanna instruments).

2.2.2 Faunal colonisation

Determination of crabs and molluscs was done a day after placement of the quadrats in order to give time for the fauna to be acclimatised to the rope thus avoiding sampling error. For every 2m x 2m sub-quadrat above, all crabs species were identified and counted using a direct binocular count to assess the ratios among species and the sex ratios within the species (Skov and Hartnoll, 2002). In order to avoid underestimating those species not active during the direct binocular counts, three sub-quadrants (0.5m x 0.5m) were randomly placed in the 2m x 2m quadrant after the binocular counts in order to count the crab burrow (Skov and Hartnoll, 2002). The burrows were counted in three categories <1cm, 1cm and >1cm. Dichotomy identification keys by Cannicci et al., (1997) were used for species identification. Molluscan species were identified and counted within the 2m X 2m quadrant using keys by Richmond, (1997). Molluscs species on trees falling within the sub-quadrant up to 1m height from the ground were also identified and counted**.**

2.3 Statistical analysis.

Crabs and molluscs species data were log transformed (log x+1) and subjected to non-metric multidimensional dimensional Scaling (MDS) ordination using Bray Curtis similarity coefficient. To test for crabs variability between sites and among sites, the analysis of similarity randomisation test (ANOSIM), (Clarke and Corley 2006) was used. Crab and mollusc species diversity in the different sites was calculated using Shannon diversity index and was then subjected to one way ANOVA while Post hoc was done with Tukey's HSD test for data that passed the normality and homogeneity test otherwise a Mann-Whitney U test was performed. Environmental parameters were analysed using 2-way ANOVA and their correlation with fauna species determined using BIOENV application in primer v6 software (Clarke and Gorley, 2006). Multivariate analysis of environmental parameters was done using Principle Component Analysis (PCA) for spatial display.

2. 3.1 Analysis of Similarity (ANOSIM)

One way layout technique ANOSIM was applied to investigate significant differences between groups. The null hypothesis (H_0) in this test is by default that there are no differences in community composition at the study sites, in this case the two study sites. In this test we reject the H_0 when the value of R tends to go towards negative value or towards zero. This means there are more similar groups between the groups than within thus we accept the null hypothesis. The reverse is true for positive R value or value leaning towards 1. On the other hand the value of range between 0 and 1 indicating some discrimination between the sites.

2.3.2 Non-metric Multidimensional Scaling (nMDS)

This technique was applied using bray-curtis similarity resemblance for biological data. The purpose of this technique is to represent the sample as points in a low-dimensional space (usually 2-D) so that the relative distance apart of all point are the same rank order as the relative dissimilarities of the species as measured by the similarity matrix. The final nMDS graphic representation allows easy visualisation and interpretation of the data. Points that are closer together represent samples that are closer together (Clarke and Warwick, 2001). The stress value needs to be as low as possible to be sure that enough iteration have been performed thus showing the degree of faithfulness of the high dimensionality represent the relationship among the samples represented in the ordination plot.)

2.3.3 BIOENV

This test is used to search the high rank correlations between a secondary fixed sample similarity matrix- mostly from biological data and resemblance matrices- (mainly environmental matrix) generated from different variables subsets of a primary matrix. In this study, the data was log transformed and then normalised since it had different units. This test works by finding the match between the multivariate among the environmental and the biological matrices. The extent to which these two pattern match reflects the degree to which the chosen abiotic factor explains the biotic variable. Bioenv does this work by carrying a full search of all possible combination of variables from the primary data.

2.3.4 Principal Component Analysis (PCA)

This is also a multivariate technique that was used for environmental data. It results to a map that places samples in two or three dimensions in which they reflect the similarity of the environmental data. Samples are point's referred to environmental axes. The environmental data used was first transformed by Euclidian distance then it was normalised due to the different units involved in an environmental data.

2.3.5 Diversity Indices.

These indices are important in every community investigation since they reflect the ecosystem health. Pillans et al., 2007 recommend the use of multiple measures of diversity in order to gain an understanding on how communities are impacted by change. In this study Hill's diversity indices were used in order to reduce the multivariate (multi-taxa) complexity of assemblage data into a single number (or small number of indices) that were evaluated for each sample. Ultimately all the diversity, density were statistically tested using ANOVA or Mann-Whitney U test in Statistica (version 8) Homogeneity of variance was tested using Levene's test and post hoc pair wise comparison was done using Tukey HSD test.

2.3.6 Taxa richness (S)

This was simply given as a total number of taxa for each of the taxonomic groups (Molluscs data and crab data). Mean taxa richness was calculated according to treatment (impacted and natural forest) and sites (site 1 and site 2).

2.3.7 Shannon- wiener diversity index (H)

Shannon diversity measures diversity of taxas in categorical data by treating taxa as symbols and their relative population sizes as the probability. This index takes into account the taxas number and their evenness. The index increases by either having more unique species or having greater taxa evenness.

Formular.

$$H' = - \sum pi \log (pi)$$

Where pi is the proportion of the total count (or biomass) arising from the *i*th species.

2.3.8 Dominance index (N_{inf})

Dominance reflects the distribution of traits in a community, which in turn affects the strength and sign of both intraspecific and interspecific interaction (Hillebrand et al., 2008). This is a quantitative estimate of biological variability that was used to compare taxa in communities by expressing how individuals were distributed among the different taxa, taking larger values when no taxa dominated the total abundance. Higher values implies no dominance whereas lower values implies high dominance

$N_\infty = 1/\max\{P_i\}$

2.4 Laboratory procedure

2.4.1 Nutrients

Three sediment samples per quadrat were randomly taken for nutrient analysis using a hand corer diameter 6.4 cm to a depth of 15 cm. Soil nutrients were analysed spectrophotometrically for NO_2^-, NO_3^-, NH_4^+, and PO_4^{3-}. Using the same sample, sediments from all samples were placed in pre weighed 10 ml beaker, and the second reading was done to get the weight of the sediments. The sediments in the beaker were dried. After drying the weight of dry sample and the beaker was recorded.

2.4.2 Sediment nutrients

Pore water in the sediment was first extracted by taking approximately 10g of sediment then adding 40mls 1M KCl flash with nitrogen gas (2 minutes) and shaking for 2 hours, to ensure maximum extraction. The sample was then centrifuged at a speed of 2000x g.r.m. for 10 minutes. The extract was then decanted and diluted with distilled water and used for the determination of nutrients.

NH_4^+ was determined according to the procedure of (Parsons et al., 1984). By this method ammonium ion in the sample is buffered in alkaline citrate medium and then treated with sodium nitroprusside (which acts as a catalyst). The reaction in this mixture gives a complex, indophenols whose blue colour intensity is measured calorimetrically at 630nm. Calibration standards were prepared using analytical-grade ammonium sulphate. The limit of detection is 0.005μMN and precision is at 0.25 μMN.

NO_3^- was determined according to the procedure of ALPHA (1992) in which NO_3^- is reduced to nitrite NO_2 by running the sample through a reduction column containing copper-coated cadmium filings. The nitrite is reacted with sulfanilamide in acidic solution where the resulting diazo compound was complexed with N-(-1-naphthyl) – ethylene diamine to form a highly coloured azo dye whose intensity is measured at

543nm. Calibration standards were prepared using analytical-grade potassium nitrate. The limit of detection is 0.05μMN and precision is at 5 μMN.

To analyse for phosphates (PO_4^{3-}), the sample was reacted with a phosphate reagent containing molybdic acid, ascorbic acid and potassium antimony III tart rate. The resulting complex formed was reduced by ascorbic acid with trivalent antimony ion as catalyst to give a blue colour solution whose intensity is measured at 885nm (Parsons et al 1984). Calibration standards were prepared using analytical grade potassium dihydrogen phosphate. The limit of detection is 0.03 μM P and precision is at 3μM P.

2.4.3 Grain size

Sediments grain size was determined using the dry sieving method. This is because the sediment samples had low or no percentage of fine silts and clays. 100g of the wet sediment sample was spread over an enamel pan and dried in an oven at 105°C until the weight was constant. The sediment was then passed through a series of sieves (2.00mm, 1.60mm, 1.00mm, 500µm, 250µm, 125µm, 63µm, 38µm). The remainder, if any was to be collected in a pan below. The sieves were then removed and their content weighed and recorded.

2.4.4 Organic matter

About 10g sediments of each sample was weighed, dried and homogenised, and placed on a pre weighed labelled aluminium foil. The samples were combusted at 450 °C for 4 hours in an oven. Samples were then kept in a desiccator to cool while dry. The samples were then weighed to at least one decimal point (Erftemeijer and Koch, 2001). The percentage of organic matter was calculated using the following formula

% organic matter = (initial weight (g) – final weight (g))/ 100

2.4.5 Porosity

Sediments from all samples were placed in pre weighed 10ml beaker, and the second reading was done to get the weight of the sediments. The sediments in the beaker were dried. After drying the weight of dry sample and the beaker was recorded. Porosity was calculated by:

Pore volume in cubic centimetres = weight of saturated sample in grams - weight of dried sample in grams (Erftemeijer and Koch, 2001).

Chapter 3

3.0 Results

3.1 Environmental parameters

All the environmental parameters were not significantly different between the study sites (Table 3). A BIOENV analysis isolated Salinity, Total dissolved oxygen, temperature and porosity as the abiotic factors that played a great role in the faunal assemblage although this analysis showed a poor correlation between both crabs and molluscs and environmental parameters (0.542). With the exception of NH_4^+-NH_3, phosphates, porosity, temperature, nitrates, sand (%), and organic matter, all the other measured parameters had a significant difference between treatments (natural forest vs. impacted forest) (table 3) with significance of P=0.0003 for interstitial salinity and P=0.0007 for clay (%),P=0.002 for total dissolved Oxygen whereas P=0.00003 for conductivity. Salinity was generally higher in the impacted sites reaching a maximum average high of 66.03±17.63 (PSU) in the impacted forests and a maximum average high of 43.00±4.73 (PSU) on the natural forests. Organic matter values were low in the disturbed sites compared to the reference forest reaching an average value of 4.12±1.73 ml/g for the impacted forest whereas the natural forest had an average high of 4.44±4.58. A principal component analysis (PCA) of environmental factor salinity, total dissolved Oxygen, temperature and porosity with sampling site superimposed as a bubble plot depicts this distribution in the sampling sites (Figure 5). A summary of the measured environmental parameters is given in table 4.

Table 3: A summary of 2- way ANOVA analysis table of physico-chemical parameters the two sampling sites (Site 1 and Site 2) and the treatment (Reference forest and the impacted forest)

Variable	Source	DF	MS	F	P
Salinity	Site	1	7.81	0.0417	0.839080
	Treatment	1	7248.72	38.6582	**0.000000**
	Site*Treatment	1	1820.24	9.7075	**0.003038**
Conductivity	Site	1	218.0	0.8241	0.368339
	Treatment	1	5518.2	20.8580	**0.000033**
	Site*Treatment	1	2421.2	9.1516	**0.003918**
TDSO	Site	1	15.10	0.1464	0.703611
	Treatment	1	1084.45	10.5177	0.002109
	Site*Treatment	1	481.54	4.6702	0.035505
Temperature	Site	1	7.38	1.776	0.188704
	Treatment	1	3.22	0.775	0.382743
	Site*Treatment	1	5.31	1.277	0.263875
SAND	Site	1	2296.1	3.9752	0.051643
	Treatment	1	1504.9	2.6054	0.112793
	Site*Treatment	1	23.1	0.0400	0.842275
CLAY	Site	1	0.159	0.00203	0.964258
	Treatment	1	1015.897	12.98226	**0.000723**
	Site*Treatment	1	439.354	5.61455	**0.021713**
Porosity	Site	1	0.01	0.001	0.973392
	Treatment	1	19.11	2.293	0.136238
	Site*Treatment	1	39.90	4.788	0.033365
Organic	Site	1	37.7775	1.51592	0.223998
	Treatment	1	8.3996	0.33706	0.564141
	Site*Treatment	1	0.0838	0.00336	0.953980
Phosphorus	Site	1	0.01	0.001	0.973392
	Treatment	1	19.11	2.293	0.136238
	Site*Treatment	1	39.90	4.788	0.033365
Nitrates	Site	1	931.734	1.658502	0.203734
	Treatment	1	816.319	1.453063	0.233711
	Site*Treatment	1	669.494	1.191712	0.280217
NH3/NH4+	Site	1	19204.88	1.394333	0.243261
	Treatment	1	19728.65	1.432361	0.237024
	Site*Treatment	1	21161.74	1.536407	0.220940

***Numbers in bold are indicate significant difference.**

Table 4: Site averages (Mean ± SD) Sediments physico-chemical characteristics

Site	Salinity (psu)	Temp (°C)	% Sand	% Clay	O/matter	PO4^{3-}	NO3-/NO2-	NH3/NH4+	porosity
S1D	55.91±10.78	29.20±1.34	82.27±32.67	2.74±4.89	6.19±11.15	3.23±4.90	0.28±0.32	3.26±1.12	11.99±2.23
S1F1	32.05±6.32	25.56±1.69	83.08±7.49	16.81±7.58	3.64±0.94	1.73±1.67	0.44±0.21	3.07±1.37	15.18±1.40
S1F2	26.33±10.91	26.78±1.36	72.91±24.72	18.54±10.98	3.49±1.30	17.56±50.37	0.80±0.36	78.14±25.11	16.27±1.94
	F1 D F2*	F1 D F2*	F1DF2*	F1D F2*	F1DF2*	F1F2*D	F1DF2*	F1F2*D	F D F*
S2D	66.03±17.63	27.80±1.95	69.70±33.10	4.09±3.29	2.38±1.43	2.66±1.06	0.51±0.27	2.05±0.93	13.51±1.06
S2F	43.00±4.73	24.97±0.45	78.82±6.03	17.71±3.08	5.87±2.02	1.41±0.30	0.91±0.40	5.44±0.97	15.91±1.11
	FD	FD	DF	DF	FD	FD	FD	FD	FD

Underlined sites (D=degraded, F = forested F* = forested 2) represent significantly different sites. These not underlined were statistically similar sites.

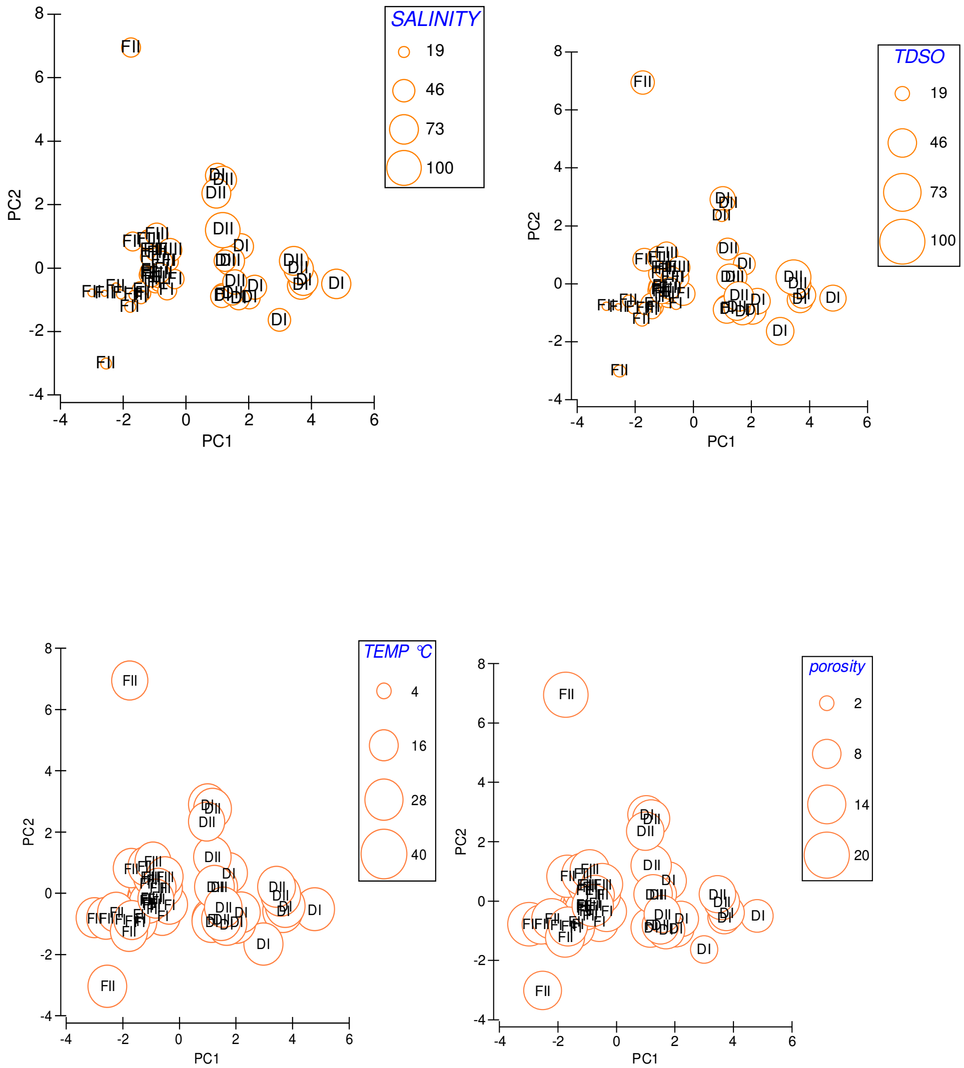

Figure 5: Principal Component Analysis (PCA) of the environmental parameters.

3.1.1 Molluscs density

Molluscs were represented by three species namely *C. decollata, C. cucullata* and *L. scabra.* However, *C. cucullata* was mostly present in the natural forests. The densities of the molluscs were low in the impacted forest compared to the natural forest.

Table 5: Average density (no/m²) of Mollusc species at the different sampling sites (mean±sd)

Species	S1D	S1F1	S1F2	S2F	S2D
Cerithidea decollata	13.5±21.5	11.4±19.7	32.6±16.1	30.5±23.7	30±19.5
Littoraria scabra	0.2±0.6	0	3.8±6.5	2.3±3.2	1.0±1.2
Crassostrea cucullata	0	0	0	0.4±1.4	1.2±2.9
Totals	13.7±22.2	11.4±19.7	36.4±22.7	32.8±28.4	33.7±47.2

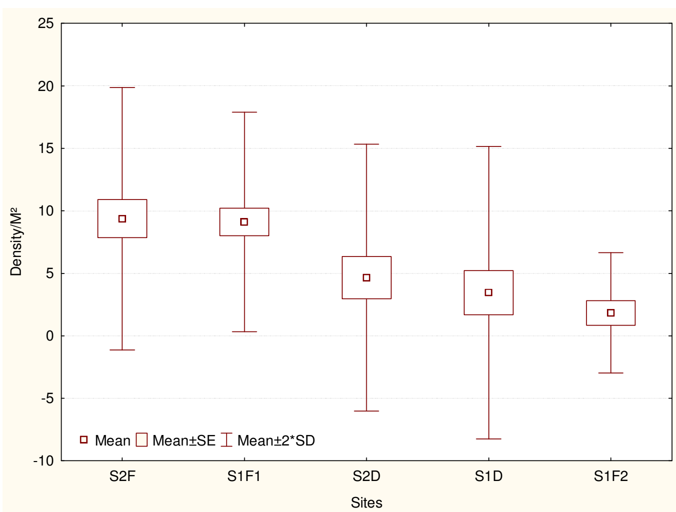

Figure 6: Box plots showing the mean densities of molluscs between the sampling sites

3.2.1 Mollusc diversity

Table 6: A summary of mean taxas richness by treatment and site) and mean dominance index of the same categories.

Indices		S1			S2	
	Natural	Impacted	Site	Natural	Impacted	Site
S	0.96±0.72	1.09±0.30	1±0.61	1.67±0.49	0.78±0.83	1.23±0.78
N_∞	0.81±0.54	0.05±0.18	0.89±0.46	1.15±0.19	0.6±0.58	0.92±0.49

Table 6 shows that site 1 had less taxas richness compared to site 2. Species dominance was higher in site 1 (lower value) compared to site 2. In site 1, impacted site had a higher species richness of molluscs and higher dominance of the same. Whereas in site 2, the natural forest recorded a higher mean species richness and a lower (high value) species dominance.

Table 7: A summary of Mann-Whitney U test results of indices of Hill for molluscs.

Variable	Source	Z	P
S	Site	0.2376	0.8122
	Treatment	-1.3677	0.1714
N1	Site	-0.8466	0.3972
	Treatment	-1.6423	0.1005
N_∞	Site	0.4960	0.6199
	Treatment	-1.2597	0.2078

Table 7 shows no significant difference in taxas richness (S) between site (site 1 and site 2) and between treatments (impacted and natural). This was the same with the Shannon-Weiner diversity index (N1 = expo (H')) and the dominance index (N_∞).

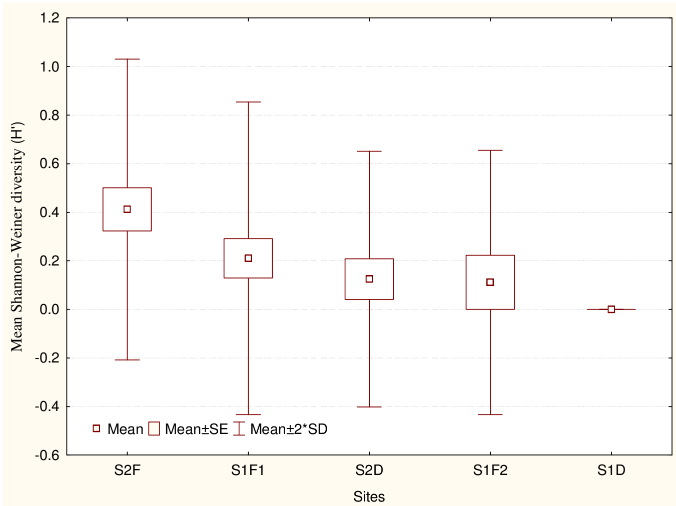

Figure 7: Box plots showing the mean Shannon-Weiner diversity of molluscs on the sampling sites

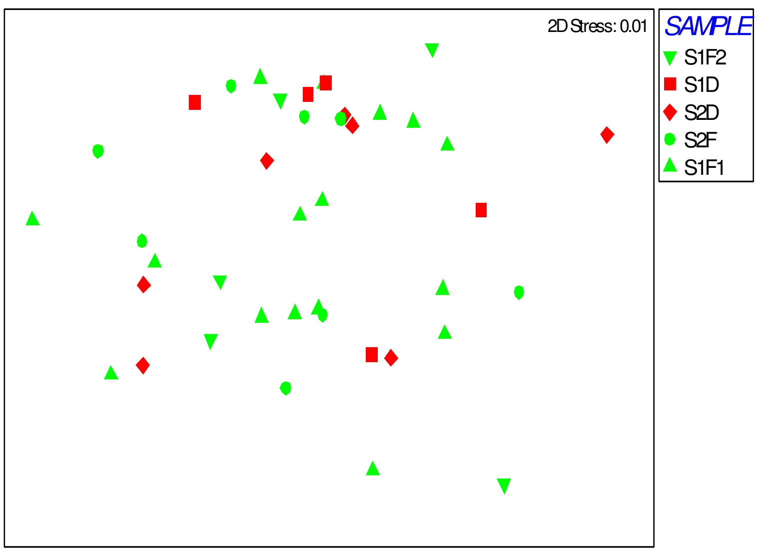

Figure 8: An nMDS plot showing molluscs species assemblages among the sampling

One way analysis of similarity (ANOSIM) shows no significant difference in molluscs species composition among the 2 sites (R=0.357, P=0.1). This was also depicted from the nMDS (Figure 9). Species diversity was not significantly different between sites (Mann-Whitney U test, Z=1.734701, P=0.8685) and between treatments (Mann-whitney U test Z=2.559041, P=0.01497) in mollusc species. However species densities significantly differed between treatments (natural forest and impacted forest) (Mann-Whitney U test, Z=4.223, P=0.0008), with the natural forest having an overall higher densities than the disturbed forest (Table 5). Indices of hill did not differ between the two sites and between treatments (table 6). In total there were only three species of molluscs found namely *Cerithidea decollata* (Linn), *Crassostrea cucullata* (Born) and *Littoraria scabra* (Linn). Apart from *Crassostrea cucullata* which was encountered mostly in site 2 the other two species were evenly found in all impacted and natural sites alike (Table 6) though in low densities.

3.3.1 Crabs density

Table 8: Average crab species density (no/m²) in the two study sites.

Species	S1D	S1F1	S1F2	S2F	S2D
U. annulipes	1.7±5.4	6.9±9.6	2.3±2.9	13.3±11.2	1.7±4.0
U. chloropthalmus	0.0±0.0	7.3±9.5	4.0±4.3	4.4±7.1	0.0±0.0
U. inversa	13.8±12.4	2.8±7.7	0.0±0.0	2.2±7.5	16.2±14.7
U. urvillei	0.0±0.0	0.4±0.9	0.0±0.0	1.3±2.2	0.1±0.3
U. vocans	0.0±0.0	0.3±0.6	0.0±0.0	0.0±0.0	0.0±0.0
P. guttatum	0.4±1.0	5.6±3.4	5.2±1.9	5.8±4.4	0.2±0.4
P. leptosoma	0.0±0.0	0.6±1.1	0.0±0.0	0.4±1.0	0.0±0.0
N. meinerti	0.0±0.0	3.6±3.8	0.5±1.2	0.5±1.2	0.2±0.4
C. ortmanni	0.0±0.0	0.1±0.3	0.0±0.0	0.0±0.0	0.2±0.7
N. smithii	0.0±0.0	3.2±4.0	4.2±2.7	2.3±3.1	0.0±0.0
M. thukuhar	0.0±0.0	0.2±0.5	0.3±0.8	0.3±0.7	0.0±0.0
Total	15.9±17.8	31±41.4	16.5±13.8	30.5±38.4	18.6±20.5

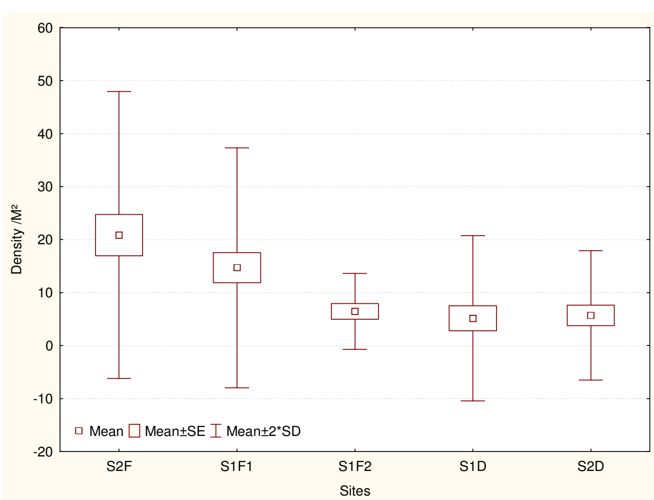

Figure 9: Box plots showing the mean densities /m² of crabs between the sampling sites.

Table 9: A summary of sex ratio differences between the burrow and binocular densities.

Crab	Burrow counts	Binocular counts	Z	P
mean densityM²	6.6914	3.5741	-5.2259	**0.0000**
Ann female	1.0037	0.6019	-4.4161	**0.0001**
Ann male	1.0037	0.8843	-3.8520	**0.0001**
Inv female	1.6728	1.0185	-4.8313	**0.0000**
Inv male	3.0111	1.0694	-5.8607	**0.0000**

***Numbers in bold are indicate significant difference.**

Table 9 shows the summary of Mann-Whitney U test for the comparison of sex ratio as calculated using the burrow counts and the binocular counts. Burrow counts and binocular counts showed significant difference with all the sexes of both *U. annulipes* and *U. inversa* with burrow count having higher readings than the Binocular counts.

3.3.2 Crab diversity

Table 10 shows that the site 1 has more taxas richness compared to site 2. Species dominance was higher in site 2 (lower value) compared to site 1. In both sites the natural sites registered higher species richness compared to the impacted sites. Site 1impacted site registered higher dominance compared to impacted site 2 whereas the natural forest of site 1 had less dominance than Site 2.

Table 10: A summary of mean taxas richness by treatment and site and mean dominance index

Indices	S1			S2		
	Reference	Impacted	Site	Reference	Impacted	Site
S	4.09±1.27	1.18±075	3.12±1.78	3.75±0.75	1.56±1.33	2.81±1.5
N∞	2.29±1.27	0.92±0.51	1.83±0.96	1.93±0.7	0.96±0.62	1.52±0.82

There was no significant difference between the sampling sites (site 1 and site 2) and the same is for the effect of both treatment and site of study in all the indices of Hill (Table 11). However there is significant difference between the treatments for all the indices.

Table 11: A summary of 2- way ANOVA analysis table of indices of Hill

Variable	Source	Df	MS	F	P
S	Site	1	0.2313	0.2019	0.655170
	Treatment	1	85.8434	74.9121	**0.000000**
	Site*Treatment	1	3.2477	2.8342	0.098512
N1	Site	1	0.0528	0.0750	0.785382
	Treatment	1	41.1432	58.3837	**0.000000**
	Site*Treatment	1	0.7968	1.1307	0.292733
N_∞	Site	1	0.0051	0.0101	0.920528
	Treatment	1	18.6111	36.9312	**0.000000**
	Site*Treatment	1	0.0516	0.1025	0.750195

***Numbers in bold are indicate significant difference.**

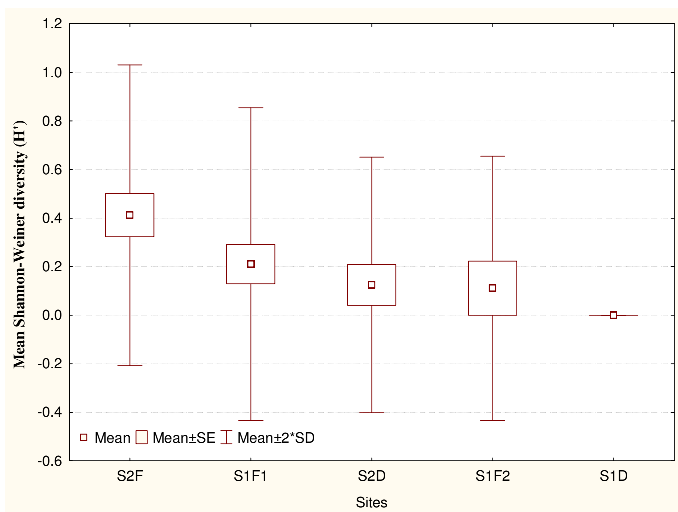

Figure 10: Box plots showing the mean Shannon-Weiner diversity (H) of crabs

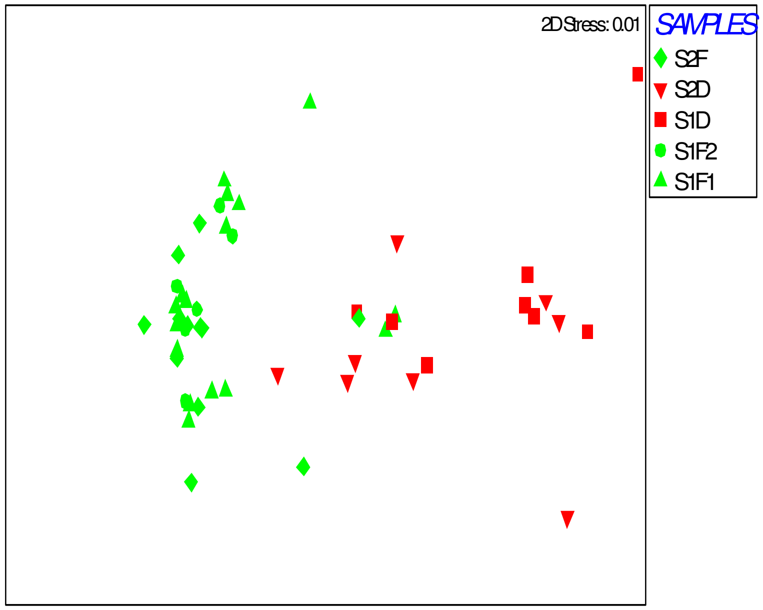

Figure 11: An nMDS plot for crabs showing crab species assemblages

Crab species composition showed a significant difference between the natural forests (SIF, S1F2 and S2F) and the impacted forest (S1D, S2D). The above nMDS (figure 11) depicts the difference in crab assemblage between these two treatments.

Overall species composition was not significantly different ($R_{site1-\ Site\ 2}$ = 0.009, P = 0.53) between the two study sites (site 1 and site 2). Whereas treatments (natural sites and impacted sites) were significantly different ($R_{natural\text{-}impacted}$ =0.754, P= 0.001). Species diversity between the two study sites was not significantly different (p=0.865) while among treatments (impacted and forested of both sites) there was a significant difference (p=0.001) with forested sites registering higher diversity compared to the impacted site.

In site1, there was significant difference in crab assemblage between the reference forests (S_IF and S_IF_2) and the disturbed site (S_ID_I) ($_{S_IF\text{-}S_ID}$ R =0.658, p=0.001; $_{S_IF_2\text{-}S_ID}$ R =0.66, p=0.001). An nMDS ordination (Figure 12) showed that crab species composition from the impacted areas (site 1 and site 2) were completely different from the reference forest (S_IF and S_IF). Species diversity in the natural forest 1 was higher than at the impacted forest I (p= 0.0016), with a similar pattern being observed among natural site 2 and impacted site 1 (p=0.0017). A BIOENV analysis isolated salinity and temperature TDSO and porosity as the main variables influencing crab species distribution.

In the second study site, crab species assemblage showed a significant difference between natural forest (S2F) and the impacted site (S2D) ($R_{S2F2\text{-}S2D}$= 0.754 P=0.001) Species diversity between the two treatments was also significantly different (P= 0.0017) with natural forest having higher diversity compared to the impacted site.

Chapter 4

4.0 Discussion

4.1 Environmental parameters

The general high interstitial salinity and temperature readings in the degraded sites could be explained by the fact that the degraded sites were without canopy resulting to the high evaporation which led to high salinity. In the mangrove, High temperatures and salinity inhibit the growth and establishment of some fauna and mangrove species therefore indirectly affecting forest structures and the associated biodiversity. Low organic matter and nitrates in the degraded sites may be due to reduced litter fall owing to the mangrove die back. Crabs and molluscs play a major roll in recycling nutrients thereby maintaining the nutrients level in the mangrove. The absence of the mangroves trees coupled with low abundance of sesarmid crabs reduces nutrient recycling thus leading to the low reading of nitrates and organic matter. Sand had no significant difference between treatments, implying the effect of massive erosion that happened during the El-Niño event. Massive sedimentation in turn resulted to great habitat modification in the forested area and thereby affecting faunal densities and composition. Other studies on mangroves have found similar results and reported that sedimentation within mangrove habitats had resulted in negative functional and structural effects on benthic communities and was responsible for lower densities and biodiversity of macro-fauna (Ellis et al., 2004; Alfaro, 2006).

Due to the habitat modification in the reference forests, some species have been able to move in the forested area resulting in competition of resources for the strict forest species. The alteration in soil conditions in the study area are quite similar to those found in other disturbed tropical forests (Alongi and Carvalho, 2008; Kaly et al., 1997). However the extend of change in soil physiochemical characteristic highly depend on the severity of the disturbance (Alongi, 2008). Change in environmental parameters like temperature, salinity, moisture, and their interactions with additional abiotic and biotic factors, has direct effect on seedling performance. Therefore effects of extreme climatic events on seedlings may have pervasive consequences at the community level, and ultimately drive vegetation shifts which eventually can lead to a reduction in biodiversity.

4.2 Crabs

A total of 11 crab species were found in this study (table 8), most of them being found in the natural sites (table 8) whereas the impacted sites had 5 species. Burrow counts and binocular counts showed significant difference with burrow counts registering a higher density than binocular counts (table 9). This difference between burrow count and binocular counts is in harmony with other studies done in this region (Skov and Hartnoll, 2001 and Skov et al., 2002). In their study, where they compared binocular counts and burrow counts to excavated counts, they found out that burrow counts over estimated the densities whereas binocular census underestimated the crab densities. They attributed the differences to discrepancies in juvenile counts (Skov and Hartnoll, 2001 and Skov et al., 2002). Total densities recorded in this study were lower (table 6) than those recorded by in other studies (Skov and Hartnoll, 2001; Skov et al., 2002) though these studies are not very comparable to this study since this study was done in a generally degraded mangrove whereas the studies by Skov were conducted in pristine mangroves.

The sex ratio recorded in this study is in harmony with other findings done in the region. Studies done by Skov and Hartnoll, (2001) during one spring tide showed that sex ratio for *U. annulipes* was almost 1:1 male to female. This ratio also corresponded to the ratio of excavated crab densities. They thus recommended binocular census as a reliable method of crab quantification during spring tides since both male and female crabs are active during this period.

Faunal composition, diversity and abundance had no differences among the two study sites (S1 and S2); however the treatments indicated differences in faunal diversity (all the indices of Hill) (table 11) and abundance with the reference sites having both higher abundance and diversity of crabs (table 11, fig 10). This finding suggests that the mangrove die-back has had a negative cascading impact on associated biodiversity, agreeing with observations from other studies on degraded mangroves (Fondo and Martens, 1998; Bosire et al., 2004; Worm et al., 2006; Alongi and Carvalho, 2008). Of most significance is the fact that *Sesarma* spp were under represented at the impacted site, despite their critical role in ecosystem functioning

(Olafsson et al., 2002; Bosire et al., 2004; Worm et al., 2006). Other studies have also found the same results and concluded that forests offer a wide variety of ecological niches for crabs that often segregate in space and time to reduce inter specific competition for food and other resources (Lee, 1998; Fondo and Martens, 1998; Kathiresan and Bingham, 2001; Fratini et al., 2005).

Different crab species feed on different food types. This leads to colonisation of different microhabitats within the same mangrove forest locality (Vannini et al., 1997; Bosire et al., 2004; Nagelkerken et al., 2008). Apart from food, mangrove vegetation provide different niche thus supporting more groups of epifauna (Fondo and Martens, 1998). For instance, mangroves provide hiding places from predators and offer protection from desiccation. The presence of high crab density in the natural site attracts predator crabs and also supports bacteria and micro algae which feed on crab faeces or use the nitrogen rich faeces for growth. Micro algae on the other hand are important food source for Ocypodid crabs (Micheli et al., 1991; Dahdouh-Guebas et al., 1999, 2000; Bouillon et al*.,* 2002; Cannicci et al*.,* 2008). This provision was unlikely in the impacted site owing to the death of mangrove trees. Therefore it seems the natural sites were more preferred by crabs probably due to being functionally better than the impacted sites. This finding suggests the importance of a healthy mangrove forest in supporting the associated biodiversity. Studies done by Fondo and Martens ,1998 also found that the presence of mangrove epi-fauna in mangrove areas to some extend depended on plant cover.

In the degraded *A. marina* zone, *U. annulipes, U. inversa, C. ortmanni* and *N. meinerti*, which are typical inhabitants of *Avicennia* forest, were still present in this zone regardless of the fact that it was only the dry *A. marina* stumps present. However *C. ortmanni* and *N. meinerti* occurred in very low densities compared to the *Uca* species which are also known dwellers in *Avicennia* zone in a mangrove forest (Olafsson et al., 2002). The presence of these *Uca* spp in high densities could be attributed to the fact that this species inhabits the desert area of mangroves even when the mangrove is healthy (Olafsson et al., 2002). Therefore they are not much affected by degradation since their food supply (detritus and micro algae) can still be available even after the mangrove die-back through tidal input and they are known to be tolerant to harsh conditions of high temperature and salinity (Fondo and Martens, 1998).

Uca spp have been thought to graze sparingly on micro algae around their burrows in a way not to deplete their food resource i.e. macro algae. This is due to the fact that as they move and feed they alter and mix the sediment there by playing a bioturbation role hence they are regarded as system engineers.It has been found that the bioturbation activity encourages the growth of the micro algae in a way that there is a fast growth of this resource every time the *Ucas* feed on them. This “farming” behaviour makes macro algae to always be sufficient within the *Uca* grazing environment. This explains their occurrence in high density on the degraded area (Fabrizio *et al.,* in prep; Cannicci et al*.,* 2008). However, this theory needs to be further investigated for certainty. In addition, *U. annulipes* and *U. Inversa* have been recorded to be able to tolerate very harsh condition of high salinity and temperature (Fondo and Martens, 1998). The tolerance ability and the bioturbating trait make the Ucas to be the dominant taxa in the impacted sites.

The presence of *C. ortmanni* and *N. meinerti* in the degraded sites despite the high temperatures, salinity and general absence of the *A. marina* species is an illustration of home fidelity and resilience by some mangrove crabs. This kind of home fidelity was also observed by Crona and R□nnb□ck (2005) on mangrove shrimps. Even though these species had the ability to move to the natural sites, they remained in the impacted sites. Their fidelity to this zone despite the degradation could be explained by the fact that this crabs preferred *A. marina* leaves compared to *R. mucronata* leaves as food (Ólafsson et al., 2002). Even though they are herbivores, these crabs have been observed to consume almost 70% of sediments in their diet and that sediments comprised higher percentage of their diet than leaves (Micheli, 1993; Ólafsson et al., 2002; Skov and Hartnoll, 2002). It could therefore be thought to be energy wise none profitable for this species to move to the natural sites where the leaves have high tannin levels and the competition for the mangrove leaves is higher. Basically Food availability, sediment kind and competition for leaves from other crabs could be major factor influencing the fidelity of these crabs to their ‘home’.

Both *C. ortmanni* and *N. meinerti* species were also found in the reference forests but strictly in regions that *A. marina* occurred even if it was a single tree. However it is worthy noting that apart from *U. inversa*, the densities of these crabs in the degraded sites formally occupied by *A. marina* was very low compared to a normal healthy *Avicennia* forest. This kind of 'home' fidelity highlights the importance of different mangrove tree species to support a unique faunal species. Since particular mangrove species provide critical unique services to ecosystem and support specific associated biodiversity (Worm et al., 2006), it is clear that efforts of reforestation that mainly target single stands (Walton et al., 2006), could highly compromise biodiversity due to low functionality in single stand mangrove species. This effect is also experienced in degraded areas where different species are affected differently hence leaving behind particular species that are more tolerant. In this scenario the ecosystems undergoes cryptic degradation as is the case in selectively harvested mangroves. However mangrove replantation has been proven to slowly restore biodiversity (Bosire et al., 2004).

In the degraded site, there was high growth of grasses and an opportunistic shrub (S*ueda maritima*) with S2 having higher cover of these opportunistic species. Most of the time *C. ortmanni* and *N. meinerti* were encountered around the *A. marina* stumps or under the opportunistic shrub. From the field observations, it is worthy noting that opportunistic shrubs can maintain some mangrove associated biodiversity but in much lower densities (Stevens et al., 2006), and worse still such shrubs do not provide the goods and services derived from mangroves to the ecosystem and the community.

Therefore mangrove death seems to have significantly reduced crab densities and led to loss of some critical species. Even though this invasive species supports some mangrove fauna, the effect of cryptic degradation is a factor not to be ignored. Studies have indicated that an increased number of species inversions overtime also coincided with loss of biodiversity and that invasion does not compensate for loss of native biodiversity and services since they compromise other species groups mostly microbial, and small invertebrates which have an equal important role in the ecosystem functioning and stability (Worm et al*.,* 2006).

In less disturbed conditions, the sediment composition in the Rhizophora and Sonneratia zone is normally clay and silt. Due to the massive sedimentation, the soil characteristic in the Rhizophora zone was modified to resemble that of upper shoe sandy desert zone. The presence of sandy substrate for easy burrowing and the numerous open canopies in the reference forest encouraged colonisation by *U. inversa and U. annulipes*. Sedimentation therefore led to modifying the natural forest, which in turn led to colonisation by other species which otherwise would not be in the reference forest. Eventual colonisation can lead to reduced biodiversity due to intraspecies competition and functional shifts of the system (Worm et al., 2006; Bosire et al., 2008). Sedimentation within mangrove habitats has been reported to result in negative functional and structural effects on benthic communities (Andrea Alfaro, 2005).

Majority of the crab species found in this study were sersamids which are known to inhabit the whole mangrove forest. In this study they were distinctively absent from the impacted forest except in patchy occurrences. Sesarmids inhabit forest where the moisture content is high and temperature is low, a situation that is different in the impacted forest which had high temperatures and low moisture content leading to a possibility of sesarmid crabs desiccation. In the forest sesarmids play a major role in nutrient recycling by feeding on the fallen leaves and aeration of the forest through burrow formation. Therefore food unavailability, high temperatures and low moisture content could be a major factor which made sesarmids conspicuously absent in the impacted site. These results suggest that after the mangrove die back, these crabs migrated to the natural forest. The absence of sesarmids which impairs nutrient recycling required for plant growth is thus likely to slow down mangrove natural regeneration and consequently impair recovery at the impacted sites.

4.3 Molluscs

Molluscs were represented by three species namely *C. decollata, C. cucullata* and *L. scabra.* However, *C. cucullata* was present mainly in the reference forests. This is because this species is mainly found attached to mangrove roots mainly *Rhizophora* and some times on *Bruguiera* roots besides being submerged round the year (Pinto & Wignarajah, 1998). Therefore the occurrence of this species is majorly dependent on tidal regime and availability of attaching substrate mainly *Rhizophora* or *Bruguiera* roots. Those that appeared on the impacted site were strictly found on the opportunistic shrub.

C. decollata and *L. scabra* were present in both treatments. *C. decollata* is adopted to the dry condition and high salinities associated with the outer *Avicennia* zone where the tide reaches only during high spring tide for a short time, hence its occurrence relatively high densities at the impacted sites. They can also occur in regions where the tide reaches twice a day during spring tides (Vannini et al., 2006; Vannini et al., 2007). However in their natural occurrence they are always under the *Avicennia* trees where the moist content is higher than in the degraded site. In this study, *C. decollata* were found under the invasive shrub and in places where there was a dead mangrove stump together with a more or less permanent pool of water, suggesting that this species may not depend on mangroves directly but tidal input for microalgae and moist conditions (Vannini et al., 2006).

In the degraded sites *L. scabra* was only found on the leaves of the invasive shrubs although their densities were low compared to the reference sites. The low density of *L. scabra* in the degraded area suggests the lack of optimal conditions for full colonization. It is therefore likely that this species may disappear altogether from the impacted site if the forest does not re-establish. Studies have found that extinction of species (at least on a local scale) is a real possibility due to either degradation and mainly due to habitat loss (Thomas et al., 2004). It is worth to not that the results indicate these mollusc species can also be supported by the invasive species (*Sueda maritima*) even though not at the population supported by healthy mangroves.

The results strongly confirm the hypothesis that biodiversity in impacted sites is impoverished following the massive sedimentation and mangrove die-back. The alteration of the forest soil characteristics has had the effect of reducing and changing the faunal composition and assemblage of the mangroves. Consequently this has led to the overall reduced functionality of the ecosystem. In this study there was an evident reduced diversity and density of species in the degraded site. This could be explained by the loss of functional attributes of the degraded area e.g. food, shelter and appropriate environmental conditions. The results clearly show that mangrove fauna are inextricably linked to integrity of the forest component of the ecosystem.

The mangrove die-back has thus reduced habitat/structural complexity thus leading to biodiversity reduction (Worm et al., 2006; Stevens et al., 2006). The extent to which mangrove degradation will affect faunal species will depend on the level of dependence of each species on the mangroves.

The succession patterns observed here seem to strongly negate successful recovery of the impacted sites. Although the invasive flora may support different fauna and bring some degree of ecosystem functioning, they do not provide the goods and services provided by the mangroves like wood products, coastal protection and organic matter production among others. These results confirm that climate change will have far reaching effects by compromising mangrove ecosystem integrity and the associated biodiversity and eventually threaten livelihoods of dependant communities. A landscape approach is therefore recommended linking mangrove conservation downstream to land-use practices upstream. Human intervention to restore the impacted sites using smart species adaptable to the changed sediment characteristics is therefore urgently required to halt the retrogressive succession patterns observed and thus support recovery of this critical and strategic (peri-urban) mangrove. Comprehensive community engagement as has already been tried in this project will be key in mitigation and adaptation measures as climate change accelerates.

Chapter 5

5.0 Conclusion and recommendation

5.1. Conclusion

Extreme climate triggered events like the El-Niño rains may lead not only to the massive death of mangrove but also in the modification of the remaining less disturbed forest through extensive deposition of sediments. This alteration of the forest soil characteristics has the effect of changing the faunal assemblage of the mangroves which may affect the functionality of the forest by bringing more resource competition and functional shift of the forest.

Further more, the biodiversity of disturbed forest is dramatically reduced in density and diversity, this has an effect on the ecosystems ability to provide goods and services to the ecosystem itself and the society that depend on the resource in many ways. The dependence range not only in food providence but also in protection of the society from the increasing frequencies and intensity of harsh climatically events and the ability of the ecosystem recovery from perturbation.

In this study there was an evident reduction in diversity and density of species in the degraded site. This could be explained by the loss of functional attributes of the degraded area e.g. food, shelter and appropriate environmental condition. The result clearly shows that mangrove fauna assemblages, diversity and density change with change of mangrove forest health. However though the resulting vegetation after degradation may provide food and refuge for molluscs and crabs, the greater volume and structural complexity of mature pristine mangrove forest supports more fauna biodiversity (Stevens et al., 2006; Worm et al., 2006).

In the event of mangrove degradation, opportunistic invasive flora like *Sueda maritima* may support different fauna and bring some degree of ecosystem functioning. How ever the other goods and services provided by the mangroves like coastal protection may never be effectively attained. To mitigate impacts of expected climate change, efforts should be put in reforestation. This reforestation should target multi species since replantation of single species may not encourage the restoration of all faunal groups because different mangrove species seems to have offer specific function for specific groups.

Increased biodiversity in healthy mangroves offer not complementary services but distinct specific role played by each fauna and flora species thus enhancing functionality of the mangal ecosystem. By early mitigation of fore seen effect of the impending climate change we could evade reduction of biodiversity and thus indirectly investing in productivity and reliability of goods and services offered by these ecosystems to current and future generation (Worm et al., 2006).

According to this work it is clear that the increasing intensity and frequency of extreme climatic events affect the ecosystem functioning consequently reducing biodiversity. This have a direct implication on the goods and services offered to the community, the ecosystem, the resistance of mangrove to stress and even food security.

5.2 Recommendation

Having been severely affected the replantation efforts in Mwache Creek need to be fully accelerated. Local villages need to be sensitized on the benefits of restoring this degraded system so that they can fully participate with complete understanding of the returns of their efforts. However further work need to be done to investigate the right mangrove species to plant in this highly modified environment so as to get a full system recovery.

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