

**THE ROLE OF SEA BLIGHT (*Suaeda monoica* FORSSK. EX GMEL) IN RECOVERY OF  
DEGRADED MANGROVES AT MWACHE CREEK, KENYA**

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Science Degree in Environmental and Occupational Health of Egerton University**

**EGERTON UNIVERSITY**

**OCTOBER, 2015**

**DECLARATION AND RECOMMENDATION**

**DECLARATION**

I hereby declare that this thesis is my original work and has never been presented for the award of a degree in any other university and that all the sources I have used have been acknowledged.

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## **DEDICATION**

To Pa, Ma, Linda, Ruby, baby Janelle, Pastor Mordecai's family, Walter, Harriet and Harry, for the smiles and encouragement.

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## ABSTRACT

Given the subsequent degradation and potential encroachment of mangrove ecosystems, the slow recovery nature of these systems and huge restoration failure efforts (100% mortality after planting in some cases) of mangrove ecosystems, quantification of the role of early colonizing vegetation to mangrove systems recovery is necessary. While sea blight (*Suaeda monoica*) is common in the degraded sites of Mwache Creek mangrove forest in Kenya and within its growing patches a marked regeneration of mangrove seedlings, its role in mangrove forest recovery is not known. This study assessed the effects *S. monoica* initiates on the biotic and abiotic factors of the degraded mangrove system to enable subsequent re-entry and functional development of the system as a contribution to mangrove intertidal restoration. Using stratified systematic sampling, in sites of naturally growing *Avicennia marina* Forssk. (Vierh), bushes of *S. monoica*, adjacent open canopy and adjacent bare sites as controls respectively, measurements of sediment conditions, vegetation structure, species composition, regeneration, faunal densities and diversities, soil; structure, bulk density and carbon stocks were determinants of recovery. Except for nutrients, significant differences in all sediment conditions ( $p < 0.05$ ) were observed amongst the four sites. Fauna densities and diversity were higher in the vegetated sites than their respective adjacent controls and their numbers significantly different in all the sites. *A. marina* site had the highest juvenile mangrove vegetation density and higher live biomass proportion, but there was no significant difference ( $p > 0.05$ ) in these two parameters between the vegetated sites. The bare areas had the highest bulk densities and low soil organic carbon, while the two vegetated sites had higher ecosystem carbon stocks ( $t\ ha^{-1}$ ) than their respective controls. The results suggest that *S. monoica* sites are functionally developing towards and becoming more akin to the natural mangrove sites. The primary mechanism proposed for improving recovery success is sediment stabilization and hydrology moderation provided by *S. monoica* bushes. These findings support the use of pioneer species where natural regeneration has been impeded as a tool for management in conservation and restoration of the functional integrity of degraded mangrove habitats.

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## **ABBREVIATIONS and ACRONYMS**

|                        |   |
|------------------------|---|
| DBH(D <sub>130</sub> ) | Stem diameter at breast height, measured at 130cm high from ground level. |
| FAO                    | Food and Agriculture Organization of the United Nations                   |
| GHG                    | Greenhouse Gases  |
| IOD                    | Indian Ocean Dipole   |
| IPCC                   | Intergovernmental Panel for Climate Change                                |
| IUCN                   | International Union for Conservation of Nature                            |
| REDD+                  | Reducing Emissions from Deforestation and Degradation                     |
| LOI                    | Loss on ignition  |
| UNEP                   | United Nations Environmental Programme                                    |
| UNFCCC                 | United Nations Framework convention on Climate change                     |
| WHO                    | World Health Organization of the United Nations                           |
| WIOMSA                 | Western Indian Ocean Mangrove Scientists Association                      |

## CHAPTER ONE

### INTRODUCTION

#### 1.1 Background to the Problem

Ecosystem services are indispensable to the wellbeing of all people. They include provisioning, regulating and cultural functions that directly affect people and support needed to maintain other services. Human health ultimately depends upon ecosystems (WHO, 2013). Unfortunately, human actions are depleting earth's natural capital, putting such strain on the environment that the ability of the planet's ecosystem to sustain future generations can no longer be taken for granted (MEA, 2005). Mangrove forests are one such ecosystem.

The term 'mangrove' refers to an assemblage of tropical trees and shrubs that grows in the intertidal zones. It is a diverse group of plants that are well adapted to wet and saline habitat and are circumglobally distributed, with majority of the populations occurring between the latitudes of 32<sup>0</sup>N and 38<sup>0</sup>S (Tomlinson, 1986). Spalding *et al.*, (1997) and Mitch & Gosselink, (2007); define mangroves as halophytic trees and shrubs (including ferns and palms) that are a characteristic of mudflats and banks, that grow in brackish to saline tidal waters of tropical and sub-tropical rivers and coastlines in many parts of the world. One of the most diverse forests, mangrove wetlands is known as the "rain forests by the sea" and is an important part of the marine habitat.

Mangroves are a backbone of the tropical ocean coastlines and are far more important to global ocean's biosphere than previously thought (Dittmar *et al.*, 2006). They play an immense role in the global carbon cycle. Further they store more carbon than most ecosystems including tropical rainforests (Donato *et al.*, 2011). This high carbon storage suggests that mangroves may play an important role in climate change management. Mangrove ecosystems are prime nesting and migratory sites for hundreds of bird species, support extensive coastal food webs, provide shoreline stability, prevent erosion, storm protection, catch sediment and alluvial materials and protect corals and sea grass beds from siltation damages due to their filtering effect (Dittmar *et al.*, 2006). They are also key pollutants sink at the coast. Wells *et al.*, (2006); estimated the value of both direct (fisheries, timber, fuel-wood, fodder and tourism) and indirect benefits of mangrove ecosystems at USD 2,000 - 9,000 per hectare of mangroves per year in areas of extensive mangrove forests and

areas close to and directly utilized by human populations. About 70% of wood requirements at the Kenyan coast are met by mangroves.

Despite their immense benefits to humans, mangrove forests continue to disappear all over the world. About 90% of global mangroves are growing in developing countries and they are under the condition of critically endangered and near extinction in 26 countries (Kathiresan & Qasim, 2005). Long term survival of mangroves is at a great risk and the services offered by the mangroves may likely be lost within 100 years (Duke *et al.*, 2007). This massive loss can be attributed to, anthropogenic activities such as; urbanization and settlement, agriculture, aquaculture practices, salt manufacture (Aboudha & Kairo, 2001), cutting for timber, fuel and charcoal, oil pollution and other pollution issues as waste dumping and mining operations as well as natural phenomena. Most recently, climate change has also been proven to affect mangroves (Bosire, 2006).

The total area of mangrove has continually reduced in Kenya since 1985. According to Kirui *et al.*, (2012); in 1985, mangrove covered an estimated 55,280ha, by 1992, this had been reduced to 51,880ha (approximately 6.2% or 0.89% yr<sup>-1</sup> on average loss over the period). The cover stood at 46,930ha in 2000 (a further loss of 9.5% or 1.19% yr<sup>-1</sup>) and 45,590ha by 2010 (2.8% or 0.28% yr<sup>-1</sup>), with Tudor Creek recording the highest loss of 82% followed by Mwache at 46% (Olagoke, 2012; Kaino, 2013). The 1997/1998 and 2006 heavy rainfall caused massive sedimentation due to erosion of terrigenous sediments that led to massive mangrove die backs. The upper region of Mwache Creek was the most affected, losing close to 500ha of forest (Bosire, 2006). Mangroves forests situated in peri-urban settings/areas are exposed to more threats than those further away (Kaino, 2013). This makes Mwache mangroves more vulnerable as they are also impacted upon by anthropogenic activities.

Just like all ecosystems of the earth, mangroves play a critical and important role in the coastlines of the world and thus their restoration is of great importance. Restoration of marine ecosystems has many examples, but possibly a shorter history than on land (Fairweather, 2004). Management of mangroves as renewable resource poses severe problems in that natural regeneration seems to be insufficient where large scale die backs have taken place. To sustain yield of these forests, therefore, there is need to address both artificial and natural regeneration methods (Kairo, 2001). Over the years, restoration efforts through reforestation have been and are still being implemented in Kenya (Bosire, 2006; Kairo *et al.*, 2008; Kirui *et al.*, 2008), as well as studies on natural

regeneration, but little has been done on native vegetation that establish particularly after a large scale disturbance (Kaino, 2013). Although re-colonization may be slow and unpredictable especially if there are no remote seed sources, recovery is still shaped positively or negatively by interacting component species with facilitation being largely supported as an influencing factor for plant distribution (Callaway, 1995; Bruno *et al.*, 2003). Ecological restoration may therefore involve nurse species that improve seed trappings and establishment, attract seed carriers, enhance soil condition through organic matter or nutrient accumulation or provide protection of sensitive seedlings apart from artificial reintroduction of the original community dominant. According to McKee *et al.*, (2007b); this approach must be based on thorough understanding of natural successional dynamics of the system as well as the growth requirements of the dominant plant species. On this account therefore, this study tried to find out the role of one such pioneer species- *Suaeda monoica* in supporting the recovery of degraded mangroves in Mwache Creek – Kenya that had a large die back during the 1997/98 and 2006 El Niño apart from human induced disturbances.

## **1.2 Statement of the Problem**

International initiatives over the years have led to the appreciation of the value of mangroves and upsurge of restoration efforts but unfortunately, restoration has emphasized on planting of mangroves as the primary approach, rather than assessing wholistic recovery opportunities and how to facilitate these efforts. Consequently, the scientific resource community has been reluctant to apply facilitation to restoration practices. Naturally through succession, opportunistic species colonize areas previously occupied by mangroves. One such species is the *Suaeda monoica* at the Mwache Creek. A large extent of it has grown back in the area that was affected by sedimentation after the Indian Ocean Dipole effect and with its presence, there has been slow but marked regeneration of mangroves especially *Avicennia marina* species in such areas. *Suaeda monoica* is known to recolonize degraded mangrove sites, but little is known on its role, ability and extent to facilitate regeneration and recovery of mangroves in degraded areas. Moreover, few studies have experimentally examined facilitation in the context of restoration (most noted being the facilitative role of herbaceous plants in recovery of mangroves in the Caribbean and saltwort in Southwest Florida). This study therefore, sought to find out the role of *S. monoica* using natural regeneration, faunal colonization, carbon stocks and sediment conditions as indicators of recovery at Mwache Creek along the Kenyan coastline.



### **1.3 Study Objectives**

#### **1.3.1 Broad Objective**

To understand the role of *S. monoica* in the recovery of degraded mangroves and mangrove ecosystems

#### **1.3.2 Specific Objectives**

- i. To measure sediment conditions in patches with and without *S. monoica* in the study area.
- ii. To assess faunal diversity and abundance in patches with and without *S. monoica* in the study area.
- iii. To characterize vegetation survival and growth in patches with and without *S. monoica* in the study area.
- iv. To measure carbon stocks in patches with and without *S. monoica* in the study area.

### **1.4 Hypotheses**

- i. *S. monoica* has an effect on sediment conditions in the study area.
- ii. *S. monoica* has an effect on faunal diversity and abundance in the study area.
- iii. *S. monoica* has an effect on vegetation survival and growth in the study area.
- iv. *S. monoica* has an effect on carbon stocks in the study area.

### **1.5 Justification of the Study**

Mangroves are considered to be one of the most endangered ecosystems in the world with approximately 35% of the original area degraded or destroyed since 1980 (Valiela *et al.*, 2001). Kenya's mangroves have reduced in cover over the years and Mwache Creek has lost approximately 45.4% of its cover of mangroves since 1992 (Kaino, 2013). Such losses may be reversed through application of the principle of ecological restoration. For developing countries with few resources and incentives to restore degraded forests, complimentary approaches rather than ecological engineering techniques such as facilitation can be a great opportunity for restoration at minimum cost and less effort (McKee *et al.*, 2007b).

Mangrove ecosystems constitute not only a critical habitat with important ecological and societal benefits, but are a system in which facilitative interactions may be applied to improve restoration

techniques (McKee *et al.*, 2007b). There had been various restoration attempts in this degraded site using various trial species; examples being: *Avicennia marina*, *Rhizophora mucronata* Lamk. and *Ceriops tagal* (Perr.) C. B. Robinson 1980, but this was met with great failures. The site was originally dominated by *R. mucronata* but site conditions such as elevation, hydrology and salinity regimes significantly changed after the sedimentation and ensuing die-back, thus making *R. mucronata* less suitable as a candidate species. Through various experimental trials, *A. marina* was found to perform better in terms of survival (Pers. Comms, Bosire). With regeneration of *A. marina* in the growing patches of *S. monoica*, therefore, there is need to document the potential for use of nurse plants to promote regeneration of mangrove in large disturbed areas where harsh conditions limit recovery (Kaino, 2013). A further understanding of how benefactors may facilitate survival and growth of mangroves will lead to identification of vegetative characteristics to screen potential candidates for restoration projects as well as provide knowledge on sustainability and management of mangrove forests which is a key component of the coastal environmental health. Restoration will help improve the lives of coastal communities who depend on the mangrove forest services and whose livelihoods have been jeopardized over time due to massive degradation of this ecosystem.

### **1.6 Scope of the Study**

Located 20km Northwest of Mombasa city in the Coast of Kenya, Mwache Creek covers approximately 17Km<sup>2</sup>. Roughly 70% of its total 1,500ha (Bosire, 2010) area is covered with mangroves of the *Avicennia marina*, *Rhizophora mucronata*, *Ceriops tagal* and *Sonneratia alba* Sm. (Kitheka *et al.*, 2002). This study focused on measurements of; sediment conditions, diversity of epifauna and infauna, above and below ground as well as the soil carbon stocks and growth and survivability of mangroves growing in four different treatments i.e.: plots with *S. monoica*, adjacent bare plots, plots of natural *A. marina* forest and their adjacent bare plots. The experimental plots were confined to Bonje area of the whole Creek's mangroves where the El Niño of 1997/98 and 2006 hit (Bosire, 2010). The study gave insight on the supporting role of mangrove associate *S. monoica* on the recovery of degraded mangroves in a span of 11 months.

## 1.7 Definition of Terms

***Avicennia marina*:** *Avicenniaceae* genus comprises eight species occupying diverse mangrove habitats (Tomlinson, 1986). *A. marina* is the widely distributed species (Duke, 1990) and it's the only representative of the genus in Kenya (Kokwaro, 1985; Tomlinson, 1986). It is often considered to play an important pioneering role in plant succession (Osborne & Berjak, 1997). Kaino, (2013); noted that *A. marina* has increased in the creek, owing to its tolerance to wide range of environmental conditions (Dahdouh-Guebas *et al.*, 2004b). It can tolerate a wide range of salinity and flooding (Tomlinson, 1986), but does poorly under shade in stands dominated by other species (Kirui, 2006).

**Epifauna:** Benthic animals that live in the surface of a substrate such as rocks, pilings, marine vegetation or the sea or lake floor itself. They may attach themselves to such surfaces or range freely over them as by crawling or swimming. Mussels, crabs, starfish, flounder are some Epifauna animals.

**Facilitation:** A succession mechanism where an early colonizing species changes the abiotic conditions thereby allowing subsequent species entries into previously intolerable habitats (Milbrandt and Tinsley, 2006).

**Infauna:** Benthic animals that live in the substrate of a body of water especially in a soft sea bottom. Infauna usually constructs tubes or burrows and are commonly found in deeper and subtidal waters. Clams, tube worms and burrowing crabs are some examples.

**Peri-urban:** The transition zone or interaction zone, where urban and rural activities are juxtaposed and landscape features are subject to rapid modification induced by anthropogenic activities.

***Suaeda monoica*:** A shrub to 6 meters high with succulent leaves of xerophytic and brackish sites that occur in Mali, East Africa and Asia. It is halophytic, saline soil indicator and able to tolerate frequent sea-water flooding. It is used in agri-horticulture, as an indicator of soil type and water availability and for land conservation. A root decoction of this plant is drunk for sore throat in Kenya, (naso-pharyngeal medicine) (Burkill, 1985).

## CHAPTER TWO

### LITERATURE REVIEW

#### Introduction

This chapter gives insight on the research subject, describes the biology of mangroves, their distribution, goods and services mangroves provide to man, degradation and destruction drivers of mangroves, mangrove associate species, diversity and distribution, regeneration, succession, restoration as well as facilitation in a global, regional, country and study site levels.

#### 2.1 Biology of Mangroves and Diversity

Mangroves are non woody plants that grow at the interface between land and sea in the tropical and sub-tropical latitudes where they exist in conditions of high salinity, extreme tides, strong winds, high temperatures and muddy anaerobic soils. There may be no other group of plants with such highly developed morphological, biological, physiological and ecological adaptations to extreme conditions (Kathiresan & Bingham, 2001). These plants and the associated microbes, fungi, plants and animals constitute the mangrove forest community or mangal. In order to survive in their environments, mangroves have evolved structural and physiological adaptations notably; aerial breathing roots, support roots and buttress, high salt tolerance, salt secreting leaves and viviparous water dispersed propagules (Walsh, 1974; Tomlinson, 1986; Saenger & Snedaker, 1993; Kathiresan & Bingham, 2001).

Mangroves often display horizontal distribution of species or zonation with certain species at the seaward fringes of swamps and others commonly in the upland reaches with considerable overlaps (Dahdouh- Guebas *et al.*, 2004a). This zonation is largely attributed to micro topography or tidal elevation, particle size characteristics and chemistry of underlying sediments, response to geomorphological factors, salinity and differential dispersal of propagules. Distribution patterns of mangrove species depend on rainfall and frequency of runoff from riverine catchments, salinity inputs and gradient, climate, sediments input and tides (Duke *et al.*, 1998). These factors and their interactions have led to classification of mangroves into Overwash; formed through tidal overwash of small, low islands and fingerlike projections of larger land masses in shallow bay and estuaries, Fringe; along fringes of islands and protected shorelines and influenced by tidal ranges, Riverine; along rivers and creeks and flooded daily by the tide, Basin; are found in island areas along

drainage depressions, channeling terrestrial runoff towards the coast, Hammock; are similar to basin except that they occur on ground which is elevated relative to surroundings areas and Scrub or Dwarf; found along flat coastal fringes. They are typically < 1.5m tall because the environment tends to be poor on nutrients (Lugo & Snedaker, 1974). Based on this classification, Mwache Creek has fringing, overwash and dwarf mangroves.

About 73 true mangrove species occur in the world (Spalding *et al.*, 2010). Depending on classification, there are about 34 species belonging to 9 genera and 4 families that are major components of mangroves and about 20 species from 11 genera and 10 families that are minor components. The 3 most well-known mangrove species are Red mangrove (*Rhizophora mangle*), the Black mangrove (*Avicennia germinans*) and the White mangrove (*Laguncularia racemosa*) (Ellison, 2000). In Kenya, mangroves are found in creeks, bays and estuaries (Kirui, 2006) and estimates of total area under mangrove vary according to different sources

## **2.2 Mangroves Distribution**

Globally, mangroves are distributed circumtropically, occurring in 112 countries and territories (Kathiresan & Bingham, 2001). Over the years, mangrove coverage has been estimated at 10 million hectares, 14-15 million hectares, 24 million hectares with a recent Spalding, (1997); estimate of over 18 million hectares. They have broader ranges along the warmer eastern coastlines of the Americas and Africa than along the cooler western coastlines. Mostly abundant in broad, sheltered, low lying coastal plains where topographic gradients are small and tidal amplitudes are large (Kathiresan & Bingham, 2001). The tropical shorelines of the world currently contain 146,530 Km<sup>2</sup> of mangrove cover (FAO, 2003).

Kenya's mangroves are well developed in many areas of the coastline (Mohamed *et al.*, 2008). Recent sources estimate the cover to be 54,000ha, 70% of which occurs in Lamu District, others are in Tana District, with less extensive mangroves in Mida, Kilifi, Mombasa and Gazi-Funzi near the Kenya-Tanzania border (The bulk of these forests occur in Lamu (33,500ha), with smaller forests in Kwale (8,800ha), Kilifi (6,600ha) and Mombasa (2,000ha)). These mangroves may be divided into two blocks, either areas north or south of Tana River, with those on the north being river and tidal dominated systems and having low human pressure than those south of Tana River (Kairo, 2001). Of the 73 species recorded worldwide, only nine mangrove species belonging to six families have

been described in Kenya. The principal being *Rhizophora mucronata* and *Ceriops tagal* that forms over 70% of the forests (Kairo, 2001). They exhibit typical zonation pattern. According to Kairo, (1995a); *Sonneratia-Rhizophora-giant Avicennia* community occupy the seaward side, followed by *Rhizophora- Bruguiera-Ceriops* in the mid and dwarf *Avicennia-Lumnitzera-Xylocarpus* complex occupying the landward side.

### **2.3 Ecosystem Goods and Services of Mangroves**

According to Brown & Lugo, (1982); mangroves are the third most productive ecosystems after Amazon and coral ecosystems. It is one of the renewable marine resources with immense economic, ecological and environmental values. Mangroves supply a variety of ecosystem goods (Kirui, 2008) and services. According to MEA, 2005; these can be classified into four:

Provisioning; the forest products include wood for timber, fuelwood poles and boat construction among other uses (Dahdouh-Guebas *et al.*, 2000). Mangrove timber is valued for construction by the locals given its anti-rot, anti-insect boring properties and valued as fuelwood because of its high calorific energy (Bosire, 2006). Nipa shingles and tannins (for coating and preserving wood, nets and other fishing gears as well as clothe dyes), leaves as livestock fodder in many countries, honey from *Avicennia* spp (Field, 2000) and a source of medicine are some of the valued traditional products that can be extracted from mangroves. Offshore fishfin and shellfish which use mangrove ecosystems as a nursery are the indirect goods produced (Mumby *et al.*, 2004). 70% of the local people along the coast of Kenya depend on mangroves for fisheries, wood for construction, energy needs among other services (Kitheka, 2000).

Cultural; indirect interest in the species found in ecosystem stems from the appreciation of the diversity of life and/or individual species (eco-tourism) (Kirui, 2008). Many local communities have also designated locations in mangrove forest as sacred shrines, where tree extraction is forbidden (UNEP-WCMC, 2006, pers obs).

Supporting; they are known to support fisheries production. They are primary nursery areas for commercially important species of fish. Due to this productivity via detritus, the organic material produced by this system serves as the base for a complex chain of food web that supports a variety of marine life. However, the link between mangroves and fisheries is still subtle and more empirical

evidence is needed for this important interaction (Bosire, 2006). Mangroves also provide a habitat for diverse fauna communities (Ashton *et al.*, 2003).

Regulating; Apart from this supporting role, mangroves root system retard water flow and create a quiet environment that encourages sedimentation and inhibits resuspension (Field, 1999). This encourages sediment stabilization that leads to shoreline protection by controlling soil erosion and acting as a barrier against storm surges, thereby protecting farmlands and human habitations in the backyard of mangroves (Bosire, 2006). During the December 2004 tsunami, shorelines with healthy mangrove forests experienced less damage than degraded ones (Dahdouh-Guebas *et al.*, 2005; Kathiresan & Rajendran, 2005). Mangrove belts also reduce impacts of cyclones (Badola & Hussain, 2005) and mitigate losses that could be experienced. They are known to perform waste disposal services (Kirui, 2008). They trap land based sediments, heavy metals, nitrogen from domestic wastes, industrial wastes and removal of fertilizer and pesticide surpluses (Levin *et al.*, 2001), thus moderating water quality and protecting the integrity of adjacent seagrass beds and coral reef ecosystems. In the context of climate change (major environmental and human health threat worldwide), mangroves are known to capture and store large quantities of CO<sub>2</sub> from the atmosphere within their biomass and sediments. According to Ong, (1993); carbon sequestration via absorption by mangroves was estimated at 25.5 x 10<sup>6</sup> ton C a year. Recent studies estimate about 388Mg C ha<sup>-1</sup> retention of carbon in mangrove sediments at a rate of 3.0-3.5 Mg C ha<sup>-1</sup> yr<sup>-1</sup> (Chmura *et al.*, 2003; Lovelock & Ellison, 2007). Research done recently at the Mwache Creek showed a total of 189.5-676.3 Mg C ha<sup>-1</sup> of carbon stocks in the mangroves.

### **2.3.1 Mangrove Forests and Climate Change**

Over the next decades, it is predicted that billions of people, particularly those in developing countries will face shortages of water and food and greater risks to health and life as a result of climate change (UNFCCC, 2007). Worldwide local climate variability can influence people's decisions with consequences for their social, economic, political and personal conditions and effects on their lives and livelihoods. The effects of climate change imply that the local variability that people have previously experienced and have adapted to is changing and changing relatively fast (UNFCCC, 2007). This therefore calls for an urgent need of reducing vulnerability of developing countries to climate change and their capacity to adapt increased. One of the strategies recommended by the UNFCCC secretariat is biodiversity conservation. Recent international climate

agreements highlight Reduced Emissions from Deforestation and Degradation (REDD+) as key relative cost effective option for mitigating climate change; the strategy aims to maintain carbon store through financial incentive for forest conservation (for example, carbon credits). REDD+ and similar programmes requires rigorous monitoring of C pools and emissions (IPCC, 2007; IPCC, 2003). Tropical wetland forest, e.g peatlands contain organic soils upto several meters deep and are among the largest C reserves in the biosphere (Murdiyarso *et al.*, 2010). Known as the “Blue Carbon” ecosystems; tidal marshes, mangroves and sea grass sequester and store large quantities of carbon in both the plants and sediments below them (Herr *et al.*, 2011) and have a potential to mitigate climate change. Perhaps the least investigated, yet critically important ecosystem service of mangroves is that of carbon storage (Kauffman & Donato, 2012). Mangrove carbon pools are among the highest of any forest type (Kauffman & Donato, 2012). They sequester 3-4% times more carbon than any other productive ecosystem. Mangroves are responsible for 14% of ocean carbon sequestration yet they occupy less than 0.5% of the coastal ocean (Alongi, 2012). This they store both in the below and above ground components, where mangrove sediments capture the largest amount of 50-90% (Donato *et al.*, 2011; Kauffman *et al.*, 2011). Development and implementation of Blue Carbon-based activities now requires strategic policy and incentive mechanisms for coastal conservation, restoration and sustainable use as well as disincentives to drain or damage this important ecosystem (Herr *et al.*, 2011).

#### **2.4 Degradation of Mangroves**

Historical records indicate that the original extent of mangrove forests have declined considerably. International proportions of original mangrove cover lost vary from 4 to 84% with the most rapid losses occurring in the recent decades (FAO 2003; 2007). These losses have largely been attributed to anthropogenic pressures such as, clearance for human inhabitation, reclamation for agriculture, aquaculture and salt ponds construction (Primavera, 1995), over-harvesting for timber and fuel wood production (Dahdouh-Guebas *et al.*, 2004a), oil pollution or gas exploration, petroleum production and accidents by large tankers also cause significant damage (Farnsworth & Ellison, 1997). Mining, pollution and damming of rivers greatly affect water salinity. Conversion of marine wetlands to aquaculture farms can cause massive aquatic organisms death and is a threat to human health because of potential acidification of the pond waters. Degradation can also be caused by nature induced changes. This include; tropical storms and tsunamis, diseases as the top dying



disease (Hussain & Acharya, 1994), biological pests and parasites. Recently, the greatest threat to loss of mangroves is attributed to global climate change related effects. It threatens the survival of a diversity of species, humans and the integrity of ecosystems worldwide (King, 2004). The major concern when it comes to mangrove forests and climate change is the possibility of increase in sea level rise. The continued existence of mangrove localities depends on their ability to keep up with the sea level rise in the future (Slob, 2012).

History has it that; Kenya experienced a flourishing mangrove trade in the 20<sup>th</sup> century from Lamu until 1982 presidential ban that stopped further exports due to over exploitation (Bosire, 2006). About 20% of mangrove forests in the country have been lost due to conversion into ponds for salt extraction (Abuodha & Kairo, 2001). The rest of the forest is degraded by pollution, unsustainable agriculture, extraction of trees for fuelwood and timber and largely threatened by climate change. Predation on natural propagules has also been known to limit their availability for natural regeneration (Bosire *et al.*, 2005). A comprehensive survey is however still needed at the Kenyan coast to get data on current standing stocks of various locations on which adaptive management guidelines can be based depending on mangrove status and needs of the locals (Bosire, 2006).

## **2.5 Mangrove Associate Species**

Mangroves provide a unique ecological environment for diverse bacterial communities which are fundamental to the functioning of these habitats (Kathiresan & Bingham, 2001). They are also a home to a group of fungi called ‘mangilicolous fungi’ which are important to nutrient cycling in these habitats (Hyde & Lee, 1995). On the Indian Ocean coast of Africa, Steinke & Jones, (1993); identified 93 species of marine fungi including 55 from mangrove wood, particularly *A. marina*. Phytoplankton and benthic micro algal communities make part of mangal productivity. One of the rich flora in mangrove environments is macroalgal flora that contributes to production while also providing habitat and food for a number of invertebrate and fish species (Kathiresan & Bingham 2001). Seagrasses of *Thalassia hemprichii*, *Halophila ovalis* and *Halodule wrightii* have been identified in the intertidal mangrove areas of Gazi Bay, Kenya and many more in mangrove areas of the world. Salt marsh plants are also present in these environments, though not common. A large number of non-mangrove plant species may be found existing with mangroves or in areas of mangrove die backs. A floristic survey of the tidal mangrove flora in the Sunderbans, India documented 1175 angiosperm species in 680 genera and 154 families (Nasakar & Bakshi, 1993).

Olmsted & Gommez, (1996); found approximately 100 families of non-mangrove species in the tropical mangrove forests of the Yucatan Peninsula. Of these, they mentioned *Chenopodiaceae* family where *Suaeda spp* belong. In Kenya, preliminary observations indicated that mangrove associate species of the grass family such as *Sporobolus spicatus* and the shrub, *S. monoica* and *Suaeda maritima* display early colonization of open mangrove areas after die backs of mangroves following the 1997/98 El Niño, but little is known on their role in mangals (Kaino, 2013).

Diverse communities of zooplanktons exist in mangrove habitats and abundances can be extremely high. Because they are often surrounded by muddy or sandy sediments, submerged mangrove roots, trunks and branches are islands of habitat that attract rich epifauna communities. These epifauna can show distinct distributional patterns correlated with desiccation, wave action, temperature and salinity. This may include; sponges, hydroids, anemones, polychaetes, bivalves, barnacles (cause damage to mangroves) bryozoans and ascidans (Kathiresan & Bingham, 2001). The muddy or sand sediments of the mangal may be home to a variety of epibenthic, infaunal and meiofaunal invertebrates. The composition and importance of these communities vary greatly from habitat to habitat depending on the sediment characteristics of the individual mangal. Mangrove sediments support higher densities of benthic organisms than adjacent non- vegetated sediments (Edgar, 1990; Sasekumar & Chong, 1998). Mangroves and prawns/shrimps population are highly linked in many regions. Analyses of commercial prawn catches have repeatedly shown strong correlations between abundance and biomass of prawns and extent of surrounding mangrove areas (Vance *et al.*, 1996). Apart from these, crabs are characteristic members of the mangrove invertebrates' fauna that have received much attention. Insects on the other hand are known to be either permanent resident of the mangal or transient visitors. Either way, they play important roles in the ecology of the system and contribute to the unique character of the habitat (Kathiresan & Bingham, 2001). Mollusks, reptiles, amphibians and other crustaceans are also part of mangrove habitats. Besides these, mangroves have a rich and diverse assemblage of fish, some of commercial value. Other fish species are important links in the mangrove food web. Human activities that impact mangroves have cascading effects on the reptile and amphibian fauna. Landbirds, shorebirds, water fowl also find habitat in mangroves. This include threatened species such as; spoonbills, large snowy egrets, scarlet ibis, fish hawks, royal terns, West Indian- whistling ducks and storm's stocks (Danielsen *et al.*, 1997; Panitz 1997). A variety of mammals (Bengal Tiger, chital deer, flying fox, and buffaloes among others) make their homes in the mangal. However, their ecology within the mangal and their associations

with the mangroves themselves have been little studied and poorly known (Kathiresan & Bingham, 2001). Mangrove associate species play an important role in the functioning of the ecosystem but silviculture management more often than not ignores assessing this component (Ellison, 2007) which can be a useful indicator of the state of the forest.

## **2.6 Mangrove Plant Community Interactions**

### **2.6.1 Natural Regeneration in mangrove ecosystems**

Natural regeneration is the renewal of forest plants through natural means; naturally occurring mangrove propagules act as the source of regeneration (Kairo *et al.*, 2001). The regeneration of mangrove seedlings takes place successfully under favorable conditions such as frequent tidal inundation, abundant space and light (Lugo, 1989). Their recruitment and survivorship are also influenced by pre-dispersal insect colonization of propagules and post-dispersal propagules predation by crabs (Allen *et al.*, 2003; Bosire *et al.*, 2006). Shoreline disturbance is also an important factor in structuring ecological communities in mangrove ecosystems (Mckee *et al.*, 2007a). The composition of regenerated species depends on the species mix of the neighboring population. Forest conditions (cut or not cut), tides and soil stability influence mangroves dispersal through self-planting or stranding strategies (Kairo *et al.*, 2001). Information on natural regeneration is important because it will assist in assessing needs for restoration in the presence of slow or no full natural forest recovery (Kaino, 2013). Artificial regeneration on the other hand involves hand planting of desired mangrove propagules and saplings at selected intertidal areas (Kairo *et al.*, 2001) with *Rhizophoraceae*, *Avicenniaceae* and *Soneratiaceae* families being the widely used. The most common technique is artificial regeneration is the use of propagules. Sometimes, saplings of less than 1.2m high are used but rarely the case in small trees of upto 6m high (Kairo *et al.*, 2001).

### **2.6.2 Facilitation in mangrove ecosystems**

Previous studies have demonstrated that plant survival and ultimate succession is driven by tolerance to physiological stress and plant-plant interactions (Milbrandt & Tinsley, 2006; Gedan & Siliman, 2009). Facilitation has been noted as the mechanism of succession where an early colonizing species changes the abiotic conditions thereby allowing subsequent species entries into previously intolerable habitats (Milbrandt & Tinsely, 2006).

The amelioration of abiotic stressors such as high temperatures, hyper-salinity and drought conditions by pioneer/superior species has been inferred in salt marsh and grassland communities (Bertness & Ewanchuk, 2002) where plants have to cope with stresses such as salinity, flooding and variable sediments and nutrient supplies. In mangroves of Florida, *Batis maritima* L. was found to improve mangroves seedlings success by slightly increasing elevation due to its dense root networks (Milbrandt & Tinsley, 2006). In the Caribbean where mangrove ecosystems is characterized by low sediment supply and low nutrients, low elevations, high temperatures, hypersalinity and strongly reducing soils with accumulation of plant phytotoxins such as sulfides; investigations revealed that *Distichlis spicata* and *Sesuvium portulacastrum* (herbaceous plants) facilitated recolonisation of mangroves in a disturbed forest. Trapping of dispersing propagules, promotin establishment and rooting and/or enhancing survival and growth of seedlings through amelioration of physicho-chemical conditions were the factors stated for facilitation (McKee *et al.*, 2007b). Whereas the findings have supported the idea of positive nursing effect of the pioneer species in degraded areas, it has not been put to test in areas where mangroves grow to tall heights and where sedimentation is a recurrent disturbance.

Rapid establishment of native vegetation, particularly after a large scale disturbance can be critical in; preventing soil erosion, invasion by exotic species and other unwanted outcomes. The current challenge in ecological restoration though is to manipulate development so that recovery of the entire suite of structural and functional features is achieved as quickly as possible (Dobson *et al.*, 1997). This is however quite difficult given the information necessary to make critical decisions about species introductions is least available (McKee *et al.*, 2007b). Often, mangrove plant communities contain herbaceous species which are a common component of the tropical beach habitats, salt marshes or other wet coastal communities (Tomlinson, 1994). Such mangrove associates may occur naturally as understory, inhabit a back-mangal ecotone or invade only upon disturbance of the dominant mangrove vegetation (McKee *et al.*, 2007b) and their presence could have potential influence in mangrove re-establishment. Although factors such as seed and seedling predators, flooding and salinity and sedimentation influencing mangrove recruitment have been studied in tropical forests, effects of herbaceous associates are relatively under studied (Milbrandt & Tinsley, 2006).

## **2.7 Restoration of Mangrove Ecosystems**

In this context, restoration is ‘the act of bringing an ecosystem back into, as nearly as possible, its original condition, renewing it or bringing it back into use’ (Field, 1996); ‘any process that aims to return a system to a pre-existing condition that includes natural restoration or recovery following basic principles of secondary succession’ (Lewis, 2005) while rehabilitation denotes any activity , including restoration and habitat creation, that converts a degraded system to a stable alternative. Increasing awareness of the true value of mangrove ecosystems has led to renewed efforts to protect and restore them (Macintosh *et al.*, 2002). Mangrove restoration and rehabilitation has been initiated successfully in various parts of the world including Thailand (Field, 1996), Bangladesh (Lewis, 2005), Malaysia, Florida, Philippines and Vietnam. In spite of the success stories, most attempts to restore mangroves often fail completely, or fail to achieve the stated goals (Lewis, 2000), because; there is a misunderstanding of mangrove forest hydrology or acceptance of the false assumption that simply planting mangroves is all that is required to establish a fully-functional mangrove ecosystem (Lewis & Gilmore, 2007). Many mangrove restoration projects unfortunately, move immediately into planting mangroves without determining why natural recovery has not occurred (Lewis, 2005). There may even be a very large capital investment in growing mangrove seedlings in a nursery before stress factors are assessed. This often results in major failures of planting efforts (Lewis, 2005). For an ecosystem to call for attention and need restoration, it implies that such a system has been altered or degraded in a way that conflicts with the defined management or conservation objectives (Kairo *et al.*, 2001).

In the East African region, sectorial approach of mangrove resource management, lack of community inputs into management efforts, poverty status of many local coastal communities and lack of awareness among decision makers about the true values of mangroves are the major obstacles in management of mangrove forests (Semesi, 1998; Kairo, 2001). These management problems are compounded by inadequate knowledge of silviculture, multiple uses potential of mangrove resources as well as inadequate techniques of natural regeneration and reforestation (Kairo *et al.*, 2001). Apart from plantation experiments for the rehabilitation of deforested mangrove areas (Kairo 1995b, 2001), little effort has been made to restore degraded mangrove systems in Kenya (Kairo *et al.*, 2001).

## 2.8 Theoretical Framework

Plant communities along tropical coastlines are often affected by natural and human disturbances, but little is known about factors influencing recovery. Based on Mckee *et al.*, (2007b) and Milbrandt & Tinsley, (2006); studies on how herbaceous plants facilitate mangrove recruitments after forest disturbance in the Caribbean and the role of Salwort in regeneration of degraded mangrove forests in Florida respectively, development of a broader picture of how these associate plants interact with mangroves to influence forest regeneration rates and patterns in a variety of environmental conditions is needed apart from just the experimental planting of seedlings in areas with the herbaceous plants and showing the possibility that herbaceous plants might promote mangrove regeneration. Application requires site specific information and mechanisms involved; this is what influenced this study.

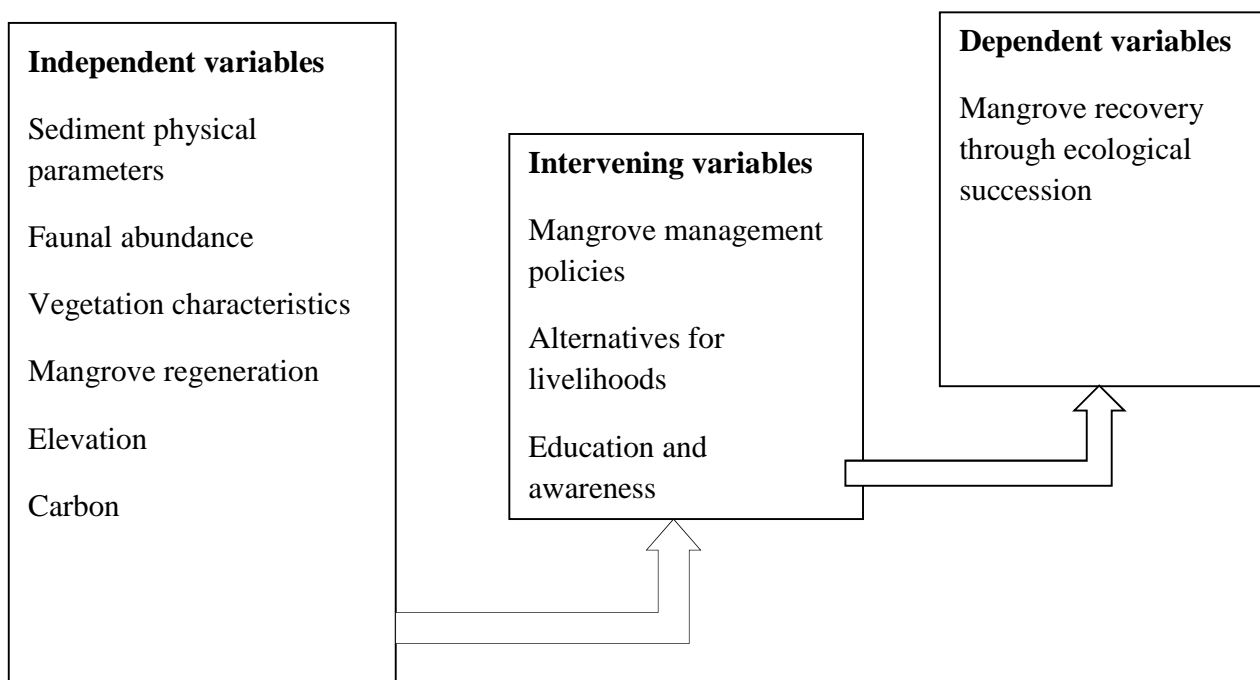


Figure 2.1: Framework showing how the study variables relate.

## CHAPTER THREE

### MATERIALS AND METHODS

#### Introduction

This chapter describes the details of the study area, the methods for field sampling, laboratory procedures and data analysis.

#### 3.1 Description of study site

Mwache Creek is one of the two tidal creeks found in Mombasa. It is located ( $04^{\circ}3.01'S$  and  $39.06^{\circ}38.06'E$ ) in the upper zones of Port-Reitz, 20Km Northwest of Mombasa city (Kitheka, 2000), (Figure 3.1). Roughly 70% of its total 1,500ha (Bosire, 2010) area is covered with mangroves of the *Avicenia marina*, *Rhizophora mucronata*, *Ceriops tagal* and *Sonneratia alba* (Kitheka *et al.*, 2002). Mangrove associates such as *Suaeda* spp and some grass are also present in the landward degraded sites. The creek receives freshwater from Mwache River which is seasonal and thus, no flow during the dry seasons between December and March and between July and September (Kitheka *et al.*, 2002). It experiences semidiurnal tides, with tidal ranges of 3.2m and 1.4m during spring and neap tides respectively. Other than this, it is ebb dominant in the front water zone main channel and flood dominant in the back water zone.

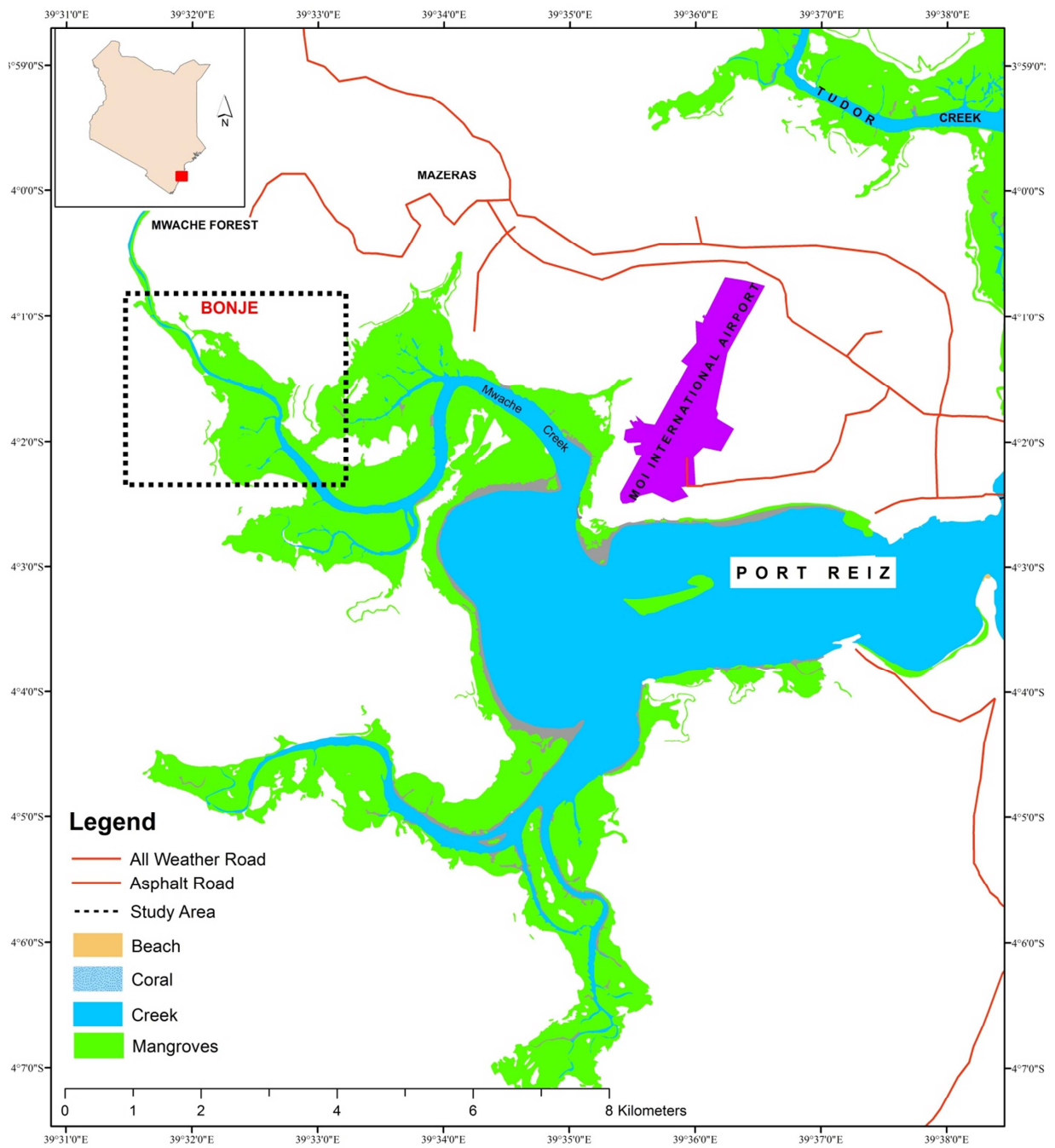


Figure 3.1: Map of Mwache Creek showing the confine location of the study (Bonje), where the study plots were based; (Modified from Kaino, 2013).



### **3.1.1 Socio economic characteristics of communities adjacent to Mwache Creek**

The 2009 census results, recorded a total of 939,370 persons in Mombasa County (KNBS, 2010). Of this about 6,226 households live in the sub-locations adjacent to the Mwache creek mangrove forest, with a population of 200 people per square kilometer. Most of the inhabitants of this communities are illiterate (90% illiteracy levels) and live below the poverty line. The socio-economic activities undertaken by these communities are mainly farming; which has been affected by the erratic rains and fishing for both subsistence and income generation. According to Kaino, (2013); fishing is directly connected to mangrove forests with the forest providing other services as wood for fuel and timber for building/construction. Basic amenities within the area are insufficient with very poor access roads and lack of electricity in majority of the houses. Most houses are made of simple mangrove frames and clay.

### **3.1.2 Status of Mwache Creek mangroves**

During rainy seasons, Mwache creek receives freshwater and terrigenous sediments from the seasonal Mwache River. Heavy supply of sediments during the Indian Ocean Dipole effect, led to huge deposition of sediments ( $10^6$  tonnes) (Kitheka *et al.*, 2002) in the wetland leading to massive destruction (smothering of mangrove roots as a result of excessive inputs) of mangrove forest in the upper region (Bonje) (Kitheka *et al.*, 2002). Sedimentation is also compounded by poor land use practices upstream; overgrazing, shifting cultivation, cultivation on steep slopes without application of soil conservation measures among others (Kitheka *et al.*, 2002; Bosire *et al.*, 2006). In addition to sedimentation, the forest is also facing degradation through high extraction pressure. Further degradation of the forest was as a result of oil spill in 2005, which affected mangroves in Port Reitz, where Mwache creek lies (Kairo *et al.*, 2005). Assessment of this impacted sites within the creek reveals that the post impact recovery of mangroves is still limited (Kaino, 2013; Bosire *et al.*, 2006).

### **3.1.3 Climate of Mwache and its environs**

Mwache climate is typical of the Kenyan coast. The coastal weather is dominated by the dry North East Monsoon from November to early March. Towards April; the monsoon wind starts to blow in an east to south-easterly direction bringing in streams of maritime air from the Indian Ocean bringing heavy rains. The South East Monsoon influence that is stable and cooler then gradually sets in from around May to August and then the North East Monsoon re-establishes. According to

Kitheka, (2000); temperature varies in the region from a minimum of 24<sup>0</sup>C in July to a maximum of 32.5<sup>0</sup>C in February. Total annual rainfall mean values are in the order of 900mm. Annual evaporation is in the order of 1800mm (much higher than the normal annual total rainfall with the result of freshwater deficit in the dry seasons). The waters are characterized by semi-diurnal tides with the spring and neap tidal variations of 3.2 and 1.4m respectively (Kitheka *et al.*, 2002) and a comparatively high relative humidity all year round, reaching its peak in the months of April to July (Aura *et al.*, 2010).

### **3.2 Study Design**

Stratified systematic sampling was adopted based on the sites history and their proximity. An area of *A. marina* natural stand and another colonized by *S. monoica* formed the two strata. To minimize environmental variation and maximize on paired matching; the *A. marina* and the *S. monoica* sites were about 250m away and had adjacent open canopies and adjacent open bare sites (2m to 5m away) as controls respectively. The environmental variables sampled included pre-determined sediment characteristics; while biotic variables included vegetation structure and composition, diversity of epifauna and infauna, above and below ground biomass estimates as well as soil carbon of sites. This was to determine any effect or change *S. monoica* initiates on soil sediments, mangrove regeneration and associated biodiversity.

### **3.3 Sampling Methods**

#### **3.3.1 Sediment Conditions**

**3.3.1.1 Salinity and temperature:** Sediment interstitial water samples were randomly collected by digging holes in the soil of 10-15cm (depending on inundation class, 10cm for class 1 and 15cm for class 2), using a machete. Salinity and temperature were measured using an optical refractometer (Atago brand) and a pH meter respectively.

**3.3.1.2 Height above datum:** For height above datum, in each of the four sites, 40 poles wrapped with dyed plotting papers were erected randomly in the whole area and left to the next day. After 24 hours, the height of the water (height to where the dye had been soaked off the plotting papers) was measured. The difference between the day's maximum tidal height (from the tide table) and the recorded height gave the height above datum for each point.

### **3.3.2 Vegetation Characteristics**

Transects were made perpendicular to the shoreline depending on the areas colonized by *S. monoica* as well as on areas of natural *A. marina* stands. Vegetation sampling was carried out using standard 10m<sup>2</sup> quadrants laid 5 meters away from each other along the transects in all the sites. Bare/open canopy sites adjacent to each of the vegetated plots above were chosen at least 2m away as controls. Within each quadrant, tree height (m) and diameter of stems (D130) were measured for all trees with a diameter greater than 2.5cm. Trees with diameter of less than 2.5cm were classified as juveniles in three regeneration classes (RCs) i.e. RCI (<40cm height), RCII (>40cm height but less than 1m) and RCIII (>1m height). In all plots, all juveniles were identified, counted and classified into the above respective RCs. The percent cover of *S. monoica*, mangrove and any other flora species was also determined in all the vegetated plots.

### **3.3.3 Faunal Diversity and Abundance**

**3.3.3.1 Epifauna:** In 10m<sup>2</sup> plots, subplots of 5m<sup>2</sup> were randomly made. Gastropods within each respective subplot were identified and counted. Additionally, for every tree that fell within each plot, gastropods were counted upto a height of 2m from the ground. For crabs, there were actual burrow counts as a measure of density.

**3.3.3.2 Sediment infauna:** A subplot of 5m<sup>2</sup> was randomly measured out in each of the 10m<sup>2</sup> plots for sediment macrobenthic sampling. Three replicate sediment cores (per subplot) were taken using a hand corer of diameter 6.4 cm to a depth of 10 cm into the soil at low tide. The samples were fixed with formalin in the laboratory before being washed with a gentle jet of tap water over a set of 0.5 mm mesh sieves to separate fauna from sediments and detritus. All that remained on the sieves was picked by forceps and put into respective sieved samples. The benthos were then stained using Rose Bengal for ease of identification (to taxonomic class level) and counted under a dissecting microscope (Days, 1974).

### **3.3.4 Soil Carbon of the sediments**

In each of the plots, a soil core was extracted to a depth of 100cm from the center of each of the plots using an open face soil corer (points GPS marked). To prevent contamination, the corer and sampling tools were washed and wiped with each use. Consistent with Indo pacific published sources on mangrove forests (Kauffman *et al.*, 2011 and Donato *et al.*, 2011), the soil profile was divided into four depth intervals: 0-15, 15-30, 30-50 and 50-100 cm. Subsamples of 5 cm in length

were then collected at the approximate midpoint of each depth interval for bulk density, grain size and carbon analysis, another 20grams of each of the remaining subsample was collected for nutrient analysis from all the depth intervals, sealed, labeled, stored at 4<sup>0</sup>C and transported for laboratory analysis.

### **3.4 Laboratory Analysis of the Samples**

**3.4.1.Nutrients analysis:** In this procedure, nitrogen in the form of NO<sub>3</sub><sup>-</sup> and NH<sub>4</sub><sup>+</sup> as well as phosphorus in the form of PO<sub>4</sub><sup>3-</sup> were extracted from soil samples using potassium chloride and analyzed using standard-sea water analysis methods described by Parsons *et al.*, (1984).

**3.4.2 Sediment particle sizes analysis:** To determine fractions of silt and clay, fine, medium and course sand, granulometric analysis was done. Samples were oven dried at 60<sup>0</sup>C for 48hours. Twenty five grams was weighed, placed in a labeled beaker with 250ml water and 10ml aqueous sodium hexametaphosphate (6.2g/l dilution) was added to separate the soil particles. This was stirred for 10minutes and left to sit for atleast 4hours and then stirred again for 10minutes. The beakers contents were poured into a 63µm sieve and flushed with water while brushing until no more silt was lost. Residue was brushed into a marked and pre weighed petri-dish and left to dry in the oven for 8hours. It was then passed through a 500µm sieve, stacked on a pan; they were then separated, weighed and recorded appropriately.

**3.4.3 Bulk density:** This is the dry weight of soil per unit volume and an indicator of soil compaction. The 5cm sub samples were placed on pre-weighed crucibles and oven dried at 60<sup>0</sup>C to constant dry weight and then weighed (Kauffman & Donato, 2012). Bulk density was calculated using:

$$soilbulkdensity(gm^{-3}) = \frac{oven\ dry\ sample\ mass(g)}{sample\ volume(m^3)}$$

Where volume = cross-sectional area of the corer X the height of the sample sub-section.

**3.4.4 Soil organic carbon analysis:** soil organic matter (SOM) was determined using loss-on-ignition (LOI), a semi quantitative method based on the indiscriminant removal of all organic matter. Samples analyzed for bulk density were used. The oven dried samples were homogenized by grinding to fine powder using a mortar and pestle, passed through a 2mm sieve and placed in a

pre-weighed aluminum crucible. This was set in a muffle furnace for combustion at 450<sup>0</sup>C for 8hours, cooled in a dessicator and weighed. Organic matter content was determined by:

$$\frac{\text{initialweight}(g) - \text{finalweight}(g)}{\text{initialweight}(g)} \times 100$$

The total soil carbon pools were determined by summing up the mass of each sampled soil depth. The soil carbon mass per sampled depth interval was calculated by:

$$SOC(Mgha^{-1}) = \text{bulkdensity}(gcm^{-3}) \times \text{soildepthinterval}(cm) \times \%OC$$

Where %C is expressed as a whole number.

**3.4.5 Vegetation (mangrove) carbon:** Data on the study on vegetation structure was used to estimate the Above and Below Ground biomass (AGB and BGB). Specific wood densities on studies carried out in mangroves of Zambezi and Mozambique in the E. African coast (Bosire *et al.*, 2012) were used. Using the general equation from Komiyama *et al.*, (2005); AGB was calculated as;

$$\text{Above ground biomass: } AGB = 0.251 * \rho * D^{2.46}$$

Where; AGB= Tree above ground biomass (Kg),  $\rho$  = wood density (g/cm<sup>3</sup>) and D = Tree diameter at breast height.

The AG carbon pool was derived by multiplying the biomass of individual component tree species by their specific wood density carbon concentrations, 47.1% for *S. alba*, 46.3% for *B. gymmnorrhizia* and 46.4% for all other species encountered (Kaufman *et al.*, 2011).

Because of the complexity in the collection and measurement of the BG biomass in mangroves, most scientific works have not been able to harvest, measure or come up with allometric equations, thus few of these equations exist. BG biomass was thus derived following the general equation for mangroves as reviewed in Komiyama *et al.*, (2008) and vegetation structure data collected.

$$\text{Below ground biomass: } BGB = 0.199 * \rho^{0.899} * D^{2.22}$$

Where; BGB= Tree below ground biomass (Kg),  $\rho$ = wood density (g/cm<sup>3</sup>) and D = Tree diameter at breast height

The carbon stock in the below ground biomass was calculated as the product of BG biomass and C concentration with a default value of 39% being used as the BG biomass C concentration as illustrated:

$$BGC = BGB \times 0.39$$

The result was then scaled to a per hectare basis to report C pool estimates.

Total Carbon stock in each of the four sites was calculated as:

$$Total\ C\ stock\ (MgCha^{-1}) = CA_{TB} + CB_{TB} + SOC$$

### **3.5 Statistical Analysis**

Statistical analyses of sediment conditions, vegetation structural indices and juvenile densities, faunal abundance and diversity, as well as carbon stocks data were done using two way ANOVA. Multiple comparisons were done using Tukey's Honest Significant Difference (HSD) test. In all cases, the quadrants/plots mentioned were treated as replicates.

## CHAPTER FOUR

### RESULTS

#### Introduction

This chapter presented the findings of the study. Based on the four sites of the study, significant differences and similarities were compared for environmental variables of sediments, biotic variables, biomass and carbon.

#### 4.1 Sediment Conditions

##### 4.1.1 Salinity and temperature of sediment interstitial water

Salinity and temperature were highest in the *S. monoica* bare areas. The vegetated areas both recorded low salinity and temperature as compared to their adjacent controls (Table 4.1). There was a significant difference ( $F_{(3,36)} = 11.99$ ,  $p < 0.05$ ) in salinity and temperature ( $F_{(3,36)} = 7.98$ ,  $p < 0.05$ ) amongst the four sites.

Table 4.1: Mean  $\pm$  SE salinity (ppt) and temperature ( $^{\circ}\text{C}$ ) in the four study sites at Mwache Creek. Sampling sites bearing same letters are not significantly different ( $n = 4$ )

| Sites                        | Parameter                   |                                    |
|------------------------------|-----------------------------|------------------------------------|
|                              | Salinity (ppt)              | Temperature ( $^{\circ}\text{C}$ ) |
| <i>S. monoica</i>            | 50.4 $\pm$ 5.4 <sup>a</sup> | 30.6 $\pm$ 0.2 <sup>a</sup>        |
| <i>S. monoica</i> bare       | 84.8 $\pm$ 8.1 <sup>b</sup> | 32.0 $\pm$ 0.3 <sup>b</sup>        |
| <i>A. marina</i> open canopy | 49.0 $\pm$ 4.5 <sup>a</sup> | 31.7 $\pm$ 0.2 <sup>b</sup>        |
| <i>A. marina</i>             | 39.3 $\pm$ 3.5 <sup>a</sup> | 30.2 $\pm$ 0.4 <sup>a</sup>        |

##### 4.1.2 Height above datum of the study sites

For elevation, height above datum was recorded. The highest average height was recorded in the *S. monoica* patches (including bare controls) at  $3.57 \pm 0.008\text{m}$  (3.57m – 3.58m) while the lowest average height was recorded in *A. marina* stand (including open canopy controls) at  $3.49 \pm$

0.01m (3.50m – 3.57m). There was a significant difference;  $F_{(3,27)} = 7.19$ ,  $p < 0.05$  in the height above datum amongst the four different sites. Areas of *S. monoica* bushes tended to be more raised than all other areas in the four sites.

#### **4.1.3 Sediment particle sizes distribution of the study sites at Mwache Creek**

Grain size distribution was expressed as percent silt and clay ( $<63\mu\text{m}$  particle size), fine and medium sand size ( $<500\mu\text{m}$  particle size) and coarse sand ( $>500\mu\text{m}$  particle size). The percent coarse sand proportion was negligible in all the four sites with the highest being only about  $1.62\pm 0.3\%$ . There was a significant difference ( $F_{(3,9)} = 17.85$ ,  $p < 0.05$ ) and  $F_{(3,9)} = 27.39$ ,  $p < 0.05$ ) in percent fine and medium sand amongst the four different sites and by depth respectively. Percent fine and medium sand exhibited a decrease in amount with increasing depth in all the four sites. Higher proportions of fine and medium sand were recorded in the *S. monoica* bare adjacent plots  $75.68\pm 2.0\%$  and the least in the *A. marina* open canopy plots  $36.78\pm 3.5\%$  (Figure 4.1). There was a significant difference ( $F_{(3,9)} = 15.49$ ,  $p < 0.05$ ) and ( $F_{(3,9)} = 18.38$ ,  $p < 0.05$ ) in the percent silt and clay amongst the four sites and by depth respectively (Figure 4.2). Proportions of fine and medium sand did not differ statistically between *S. monoica* and *A. marina* stand sites ( $p = 0.09$ ). However, Silt and clay proportions differed significantly  $F_{(3,7)} = 18.42$ ,  $p < 0.05$  between this two sites. Sediment grain sizes in *A. marina* stand areas and their open canopy controls exhibited no significant differences which was also the case between *S. monoica* and their bare adjacent controls.



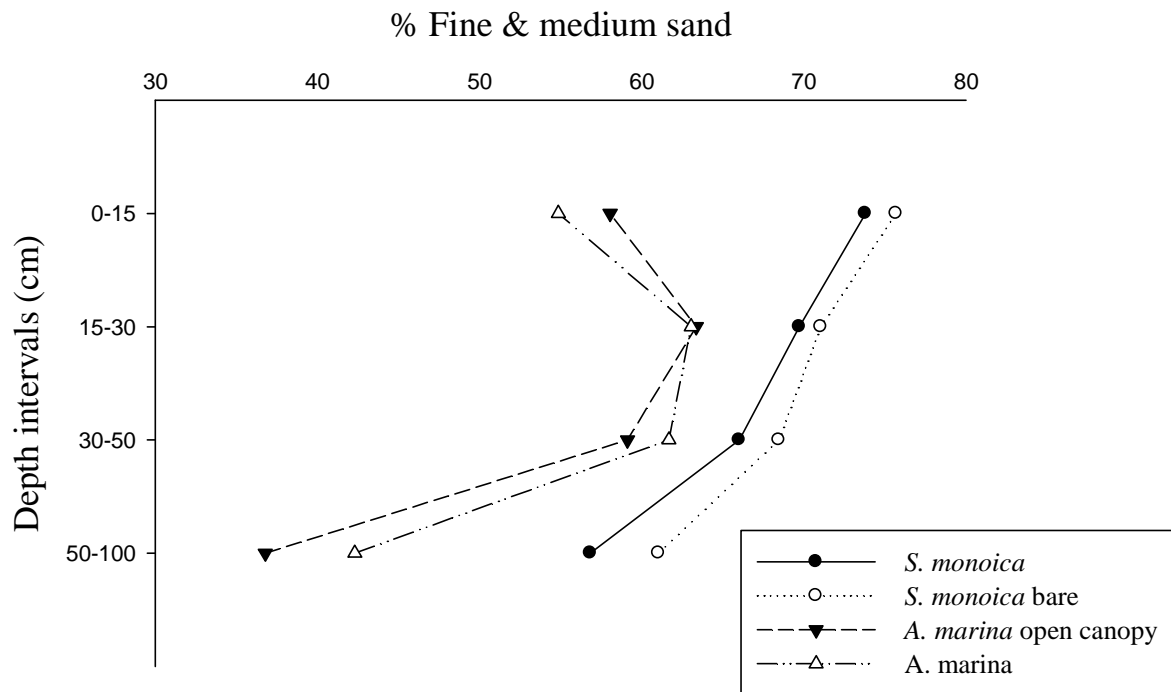


Figure 4.1: Mean  $\pm$  SE fine & medium sand percentage in the different depth intervals of the four sites at Mwache Creek.

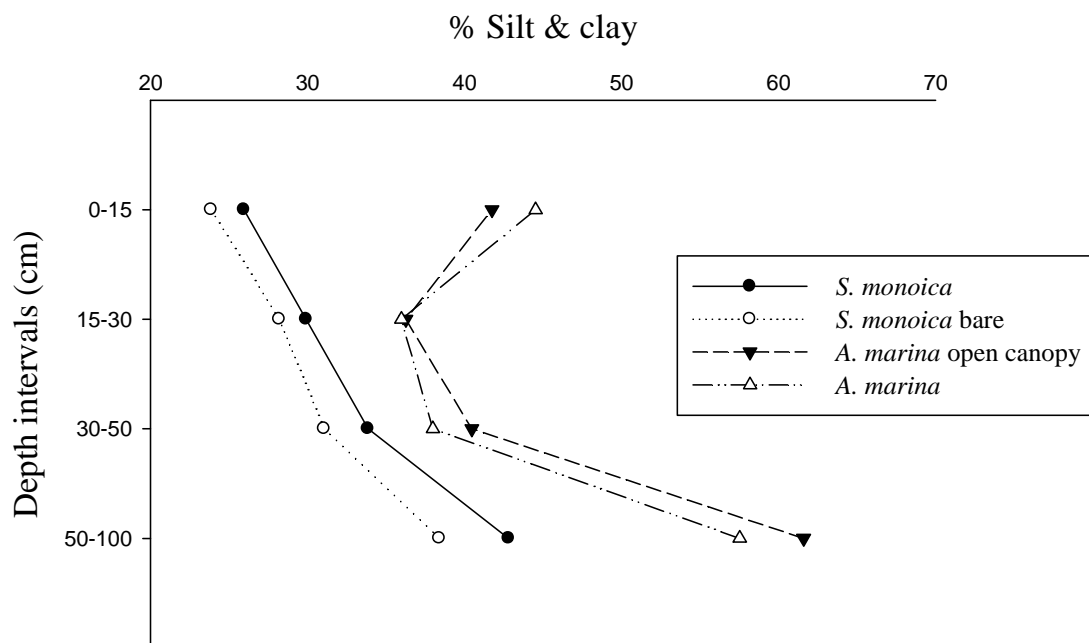


Figure 4.2: Mean  $\pm$  SE silt & clay percentage in the different depth intervals of the four sites

### 4.1.3 Soil organic matter of the sediments

The mean soil organic matter differed significantly both: amongst the sites ( $F_{(3,9)} = 4.46$ ,  $p < 0.05$ ) and by soil profiles (depth) ( $F_{(3,9)} = 8.87$ ,  $p < 0.05$ ) in all the sites of the study area (Table 4.2).

Table 4.2: Mean  $\pm$  SE percentage soil organic matter at different depth intervals in the four different sets of sites at Mwache Creek.

| Depth Interval | Sites            |                       |                      |                  |
|----------------|------------------|-----------------------|----------------------|------------------|
|                | <i>S.monoica</i> | <i>S.monoica</i> bare | <i>A.marina</i> open | <i>A. marina</i> |
| 0-15cm         | 1.71 $\pm$ 0.4   | 1.32 $\pm$ 0.2        | 1.34 $\pm$ 0.3       | 1.64 $\pm$ 0.3   |
| 15-30cm        | 1.49 $\pm$ 0.2   | 0.89 $\pm$ 0.2        | 1.23 $\pm$ 0.2       | 1.54 $\pm$ 0.3   |
| 30-50cm        | 1.24 $\pm$ 0.2   | 1.25 $\pm$ 0.4        | 2.01 $\pm$ 0.7       | 2.35 $\pm$ 0.54  |
| 50-100cm       | 2.87 $\pm$ 0.7   | 1.76 $\pm$ 0.3        | 2.18 $\pm$ 0.6       | 3.31 $\pm$ 0.3   |

The natural *A. marina* sites had significantly high soil organic matter content that all the three sites and the bare area had significantly ( $p < 0.05$ ) low soil organic matter content ( $n = 4$ ). Although combined comparison showed a significant difference in the soil organic matter amongst the four sites, paired comparisons between; *A. marina* and *S. monoica* sites, *A. marina* and their adjacent open canopy controls as well as *S. monoica* and their bare adjacent control sites exhibited no significant differences in the percent soil organic matter.

### 4.1.4 Sediment nutrients

The concentration of the various nutrients varied in all the four different sites, with ammonia recording the highest concentration in soil sediments of *A. marina*, their open canopies and *S. monoica* sites compared to all the other nutrients measured (Table 4.3). There was no significant difference ( $p > 0.05$ ) in the mean (mg/l) of phosphorous, nitrogen and ammonia by depth in all the sites. However, there was a significant difference: in phosphorous ( $F_{(3,12)} = 10.023$ ,  $p < 0.05$ ), in nitrogen ( $F_{(3,12)} = 11.2792$ ,  $p < 0.05$ ) and in ammonia ( $F_{(3,12)} = 10.7404$ ,  $p < 0.05$ ) amongst all the

four different sites. There was a statistical difference in the phosphate ( $F_{(3,7)} = 16.34$ ,  $p < 0.05$ ), but no statistical difference in ammonia and nitrate concentrations between *S. monoica* and *A. marina* sites. Although different in values, the nutrients in *S. monoica* plots recorded the same type of trend as in the *A. marina* natural stand areas (higher amounts of ammonia than nitrates and phosphates).

Table 4.3: Mean  $\pm$  SE nutrients concentrations (phosphates, nitrates and ammonia) in mg/l of soil sediments in the four different sites at Mwache Creek per depth.

| Treatment               | Nutrients  | Depth intervals |                |                |                |
|-------------------------|--|-----------------|----------------|----------------|----------------|
|                         |  | 0-15 cm         | 15-30 cm       | 30-50 cm       | 50-100 cm      |
| <i>S.monoica</i>        | PO <sub>4</sub> <sup>3-</sup>                              | 0.19 $\pm$ 0.1  | 0.18 $\pm$ 0.1 | 0.17 $\pm$ 0.1 | 0.18 $\pm$ 0.1 |
|                         | NO <sub>3</sub> <sup>-</sup> /NO <sub>2</sub> <sup>-</sup> | 0.54 $\pm$ 0.3  | 0.16 $\pm$ 0.1 | 0.35 $\pm$ 0.2 | 0.73 $\pm$ 0.1 |
|                         | NH <sub>4</sub>  | 1.25 $\pm$ 0.1  | 1.42 $\pm$ 0.2 | 1.19 $\pm$ 0.1 | 1.55 $\pm$ 0.2 |
| <b><i>S.monoica</i></b> |  |                 |                |                |                |
| <b>bare</b>             | PO <sub>4</sub> <sup>3-</sup>                              | 0.57 $\pm$ 0.1  | 0.77 $\pm$ 0.2 | 0.83 $\pm$ 0.1 | 0.92 $\pm$ 0.2 |
|                         | NO <sub>3</sub> <sup>-</sup> /NO <sub>2</sub> <sup>-</sup> | 0.06 $\pm$ 0.6  | 0.13 $\pm$ 0.8 | 0.05 $\pm$ 0.9 | 0.13 $\pm$ 0.3 |
|                         | NH <sub>4</sub>  | 0.73 $\pm$ 0.1  | 0.80 $\pm$ 0.1 | 0.66 $\pm$ 0.1 | 0.85 $\pm$ 0.1 |
| <b><i>A.marina</i></b>  |  |                 |                |                |                |
| <i>A.marina</i>         | PO <sub>4</sub> <sup>3-</sup>                              | 0.16 $\pm$ 0.1  | 0.12 $\pm$ 0.1 | 0.15 $\pm$ 0.1 | 0.13 $\pm$ 0.1 |
|                         | NO <sub>3</sub> <sup>-</sup> /NO <sub>2</sub> <sup>-</sup> | 0.33 $\pm$ 0.1  | 0.55 $\pm$ 0.2 | 0.38 $\pm$ 0.1 | 0.20 $\pm$ 0.1 |
|                         | NH <sub>4</sub>  | 1.09 $\pm$ 0.1  | 0.84 $\pm$ 0.1 | 1.06 $\pm$ 0.1 | 0.97 $\pm$ 0.2 |
| <b><i>A.marina</i></b>  |  |                 |                |                |                |
| <b>open canopy</b>      | PO <sub>4</sub> <sup>3-</sup>                              | 0.12 $\pm$ 0.1  | 0.11 $\pm$ 0.1 | 0.09 $\pm$ 0.1 | 0.11 $\pm$ 0.1 |
|                         | NO <sub>3</sub> <sup>-</sup> /NO <sub>2</sub> <sup>-</sup> | 0.33 $\pm$ 0.1  | 0.44 $\pm$ 0.1 | 0.70 $\pm$ 0.2 | 0.61 $\pm$ 0.1 |
|                         | NH <sub>4</sub>  | 1.31 $\pm$ 0.1  | 0.95 $\pm$ 0.1 | 1.29 $\pm$ 0.2 | 0.93 $\pm$ 0.1 |

## 4.2 Fauna Diversity and Abundance

Fauna investigated were epifauna those that live above the sediments and infauna, those within the sediments.

#### 4.2.1 Crabs in the sampled sites of Mwache Creek

The largest population of crabs identified in the study site was *Uca*. Although not restricted to the two species mentioned in Table 4.4, these two were the most recorded in the four different sites. There was a significant difference in the density of *Uca* (spp);  $F_{(3,27)} = 13.24$ ,  $p < 0.05$  and in the density of *Perisesarma guttatum*;  $F_{(3,27)} = 6.88$ ,  $p < 0.05$  amongst the four sites but no significant difference in the density of *Cardisoma carnitex*  $F_{(3,27)} = 0.98$ ,  $p > 0.05$  amongst the four different sites (it was only sighted in areas of *A. marina* natural stand and their adjacent controls). No statistical difference was recorded in the average sum of crab densities (individuals/m<sup>2</sup>) between the two vegetated sites and between each vegetated site and their respective adjacent controls. The natural *A. marina* sites had however, a significantly ( $p < 0.05$ ) higher density of crabs than the other three sites.

Table 4.4: Mean  $\pm$  SE crabs density (individuals/m<sup>2</sup>) in the four study sites at Mwache Creek

| Species  | Sites             |                        |                       |                         |
|--|-------------------|------------------------|-----------------------|-------------------------|
|  | <i>S. monoica</i> | <i>S. monoica</i> bare | <i>A. marina</i> open | <i>A. marina</i> canopy |
| <i>Uca</i> (spp)<br>especially;( <i>Uca</i><br><i>annulies</i> and<br><i>Uca chlorophthalmus</i> ) | 7.2 $\pm$ 0.7     | 3.1 $\pm$ 0.9          | 34.5 $\pm$ 8.8        | 55.1 $\pm$ 10           |
| <i>P. guttatum</i>   | 1.2 $\pm$ 0.7     | 0                      | 1.8 $\pm$ 0.8         | 3.9 $\pm$ 0.8           |
| <i>Cardisoma carnitex</i>  | 0                 | 0                      | 0.1 $\pm$ 0.1         | 4.6 $\pm$ 4.6           |

#### 4.2.2 Mollusks in the sampled sites of Mwache Creek

There was a significant difference in the density; of *Cerithidae*  $F_{(3,27)} = 10.88$ ,  $p < 0.05$  and *Terebralaia*  $F_{(3,27)} = 9.1$ ,  $p < 0.05$ , but no significant difference in the density of *Littoraria* species amongst the four sites. The two vegetated sites recorded higher numbers of gastropods than

respective controls. There was no statistical difference in the density of *Cerithidae* and *Littoraria*, but a significant difference  $F_{(9,19)} = 7.01$ ,  $p < 0.05$  was recorded in the density of *Terebralaia* species between the two vegetated sites. Apart from *Littoraria*, gastropods species were abundant in numbers in the *S. monoica* site than all the other three sites. *Terebralaia* was more in the *S. monoica* and their bare control sites than in the *A. marina* stand and their open canopy control sites. Generally, the vegetated sites recorded a higher number of gastropods than their respective adjacent controls.

Table 4.5: Mean  $\pm$  SE gastropods density (individuals/ m<sup>2</sup>) in the four study sites at Mwache Creek

| Sites                        | Gastropods        |                    |                   |
|------------------------------|-------------------|--------------------|-------------------|
|                              | <i>Cerithidae</i> | <i>Terebralaia</i> | <i>Littoraria</i> |
| <i>S. monoica</i>            | 66.7 $\pm$ 11.3   | 2.3 $\pm$ 1.08     | 2 $\pm$ 0.5       |
| <i>S. monoica</i> bare       | 1.4 $\pm$ 1.7     | 0.4 $\pm$ 0.2      | 0.06 $\pm$ 0.06   |
| <i>A. marina</i> open canopy | 4.2 $\pm$ 0.4     | 0.08 $\pm$ 0.04    | 2 $\pm$ 1.0       |
| <i>A. marina</i>             | 64.9 $\pm$ 18.6   | 0.48 $\pm$ 0.3     | 4.4 $\pm$ 1.7     |

#### 4.2.3 Infauna in the sampled sites of Mwache Creek

The *A. marina* sites and their adjacent open canopy controls recorded higher densities of infauna than the *S. monoica* sites and their adjacent controls. The former sites also had more infauna taxa than the latter sites (Table 4.6). There was a significant difference ( $F_{(3,36)} = 4.31$ ,  $p < 0.05$ ) in infauna densities among the four different sites. The number of taxa also differed significantly ( $p < 0.05$ ) among the sites. The vegetated sites had higher density of sediment infauna than their respective controls. *Eleptera* taxon was exclusive to the *S. monoica* sites while *Tubelleria* and *Polychaeta* taxa were found in all the four sites. The bare areas recorded the least number of sediment infauna.

Table 4.6: Mean  $\pm$  SE sediment infauna density (nr. m<sup>2</sup>) recorded in the four different sites at Mwache Creek.

| <b>Sediment infauna</b> |                          |                               |                         |                                     |
|-------------------------|--------------------------|-------------------------------|-------------------------|-------------------------------------|
| <b>Infauna</b>          | <b><i>S. monoica</i></b> | <b><i>S. monoica</i> bare</b> | <b><i>A. marina</i></b> | <b><i>A. marina</i> open canopy</b> |
| <i>Brachyura</i>        | 0                        | 0                             | 73 $\pm$ 30             | 62 $\pm$ 21                         |
| <i>Caridea</i>          | 0                        | 0                             | 22 $\pm$ 18             | 0                                   |
| <i>Copepoda</i>         | 0                        | 0                             | 257 $\pm$ 121           | 121 $\pm$ 67                        |
| <i>Cumacea</i>          | 0                        | 0                             | 13 $\pm$ 14             | 0                                   |
| <i>Tubellaria</i>       | 527 $\pm$ 142            | 19 $\pm$ 12                   | 10261 $\pm$ 921         | 947 $\pm$ 225                       |
| <i>Insecta</i>          | 17 $\pm$ 18              | 0                             | 93 $\pm$ 47             | 33 $\pm$ 30                         |
| <i>Isopoda</i>          | 17 $\pm$ 18              | 0                             | 458 $\pm$ 42            | 73 $\pm$ 31                         |
| <i>Nematoda</i>         | 0                        | 0                             | 15693 $\pm$ 1284        | 911 $\pm$ 89                        |
| <i>Oligochaeta</i>      | 0                        | 0                             | 8296 $\pm$ 4332         | 1235 $\pm$ 329                      |
| <i>Ostracoda</i>        | 674 $\pm$ 249            | 0                             | 583 $\pm$ 177           | 425 $\pm$ 66                        |
| <i>Polychaeta</i>       | 54 $\pm$ 29              | 17 $\pm$ 18                   | 22740 $\pm$ 3982        | 11251 $\pm$ 521                     |
| <i>polycladida</i>      | 0                        | 0                             | 0                       | 15 $\pm$ 7                          |
| <i>Eleptera</i>         | 234 $\pm$ 91             | 0                             | 0                       | 0                                   |

### 4.3 Vegetation Characteristics

The average canopy height of mangrove (mean of tree heights) was 3.34 $\pm$ 0.41m and 4.30 $\pm$ 0.21m in the *S. monoica* and *A. marina* sites respectively with *A. marina* as the dominant species in the two sites. There were 3 mangrove species (*A. marina*, *C. tagal* and *R. mucronata*) encountered within the *S. monoica* patches and 4 mangrove species (*A. marina*, *C. tagal*, *R. mucronata* and *Bruguiera gymnorrhiza* (L) Lamk.) encountered in the *A. marina* natural forested areas. On average, plots with *S. monoica* had an extensive growth of *S. monoica* bushes that covered an approximate 53.8% of the 10m<sup>2</sup> plots. *S. monoica* was also encountered in the natural forest area

but only covered about 2.2% of the 10m<sup>2</sup> plots. Unlike the natural stand *A. marina* areas, the *S. monoica* patches also had salt tolerant grass that occupied on average 7.1% of the sampled area.

All regeneration classes of *Avicennia marina* species exhibited no significant differences in plots of natural *A. marina* forests and plots of *S. monoica* ( $p > 0.05$ ). Plots in both controls recorded no regeneration at all. Other mangrove species noted were *R. mucronata* and *C. tagal* (Table 4.7). There was a significant difference ( $F_{(2,5)} = 17.77$ ,  $p < 0.05$ ) in juvenile densities of all mangrove species encountered in the two sites. *A. marina* had the largest juvenile density.

Table 4.7: Mean  $\pm$  SE regeneration density (nr. ha<sup>1</sup>) of mangrove species encountered in the sampled sites at Mwache Creek.

| Sites             | <i>A.marina</i> | <i>R. mucronata</i> | <i>C. tagal</i> | Sum              |
|-------------------|-----------------|---------------------|-----------------|------------------|
| <i>S. monoica</i> | 7600            | 100                 | 200             | 7900 $\pm$ 1050  |
| Bare Control      | -               | -                   | -               | -                |
| <i>A.marina</i>   | 12800           | 1200                | 400             | 14400 $\pm$ 2800 |
| Open Control      | -               | -                   | -               | -                |

#### 4.4 Carbon Stocks of the Study Sites at Mwache Creek

##### 4.4.1 Bulk density of the sediment

The average bulk densities for the four sites were; 1.3055 $\pm$ 0.07gcm<sup>-3</sup>; 1.3076 $\pm$ 0.03gcm<sup>-3</sup>; 1.1984 $\pm$ 0.03gcm<sup>-3</sup> and 1.2254 $\pm$ 0.03gcm<sup>-3</sup> for *S. monoica*, their adjacent bare plots and *A. marina* and their adjacent open canopy plots respectively ranging from 0.76gcm<sup>-3</sup> to 1.768gcm<sup>-3</sup>. A marked significant difference was recorded in the mean bulk densities ( $F_{(3,9)} = 6.08$ ,  $p < 0.05$ )

amongst the four sites. In contrast, there was no significant difference ( $p>0.05$ ) in the mean bulk densities by depth (Figure 4.3).

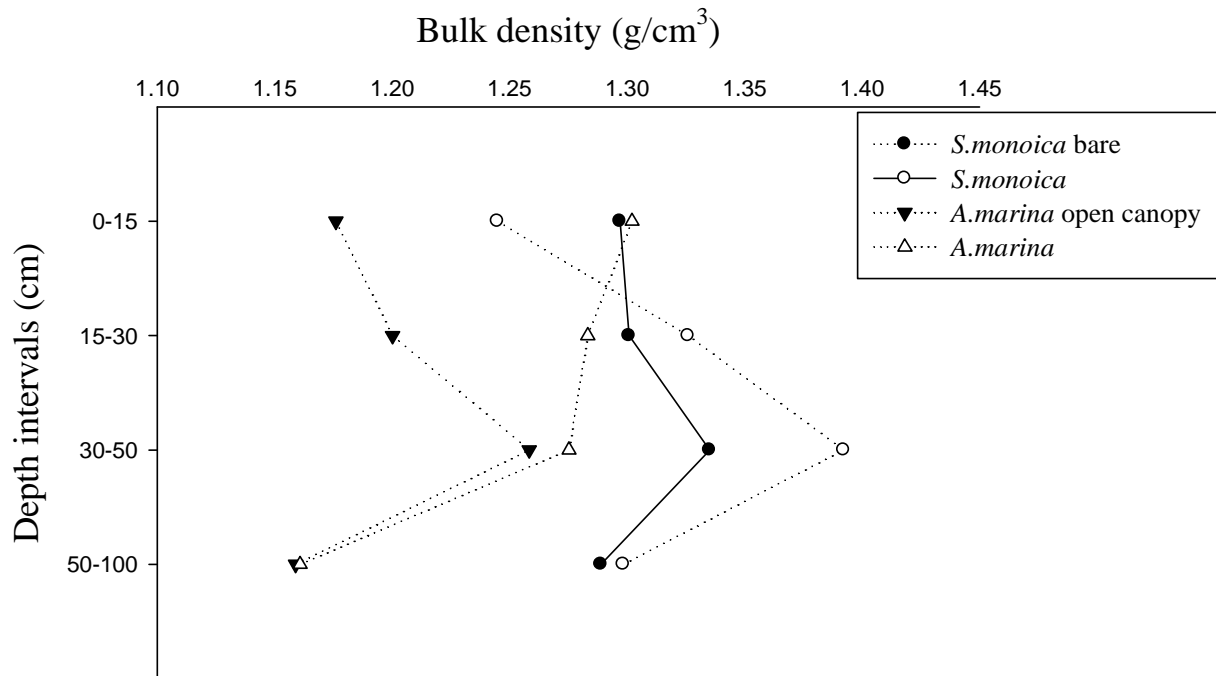


Figure 4.3: Mean bulk density trends through the different soil depth intervals in the four study sites.

#### 4.4.2 Soil organic carbon concentration

The mean percent soil organic carbon was highest at the 50-100cm soil depth in all the four study sites (an observed increase with depth) (Figure 4.4). There was a significant difference in the mean % soil organic carbon; amongst the study sites ( $F_{(3,9)} = 7.25$ ,  $p<0.05$ ) and with soil depth ( $F_{(3,9)} = 13.94$ ,  $p<0.05$ ). Soil organic carbon concentration exhibited the same patterns as soil organic matter.



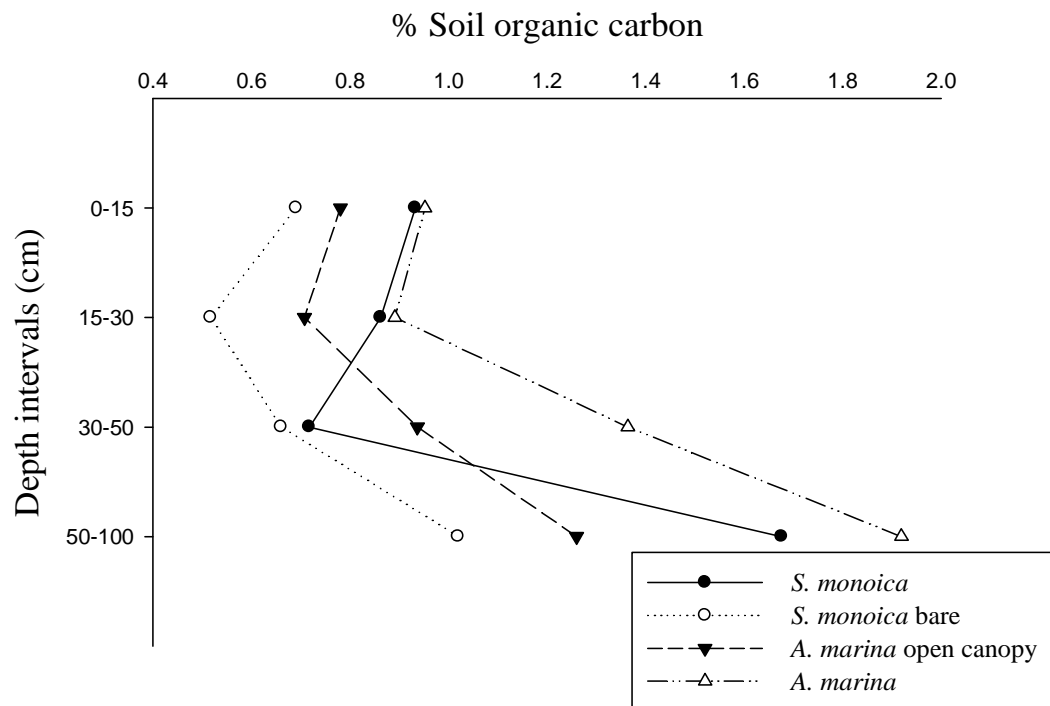


Figure 4.4: Percent soil organic carbon concentration trends through the different soil depth intervals in the four study sites.

#### 4.4.3 Soil organic carbon pool

The mean SOC was highest in the *A. marina* site at  $179.983 \pm 6.8 \text{ Mg C ha}^{-1}$  and lowest in the bare *S. monoica* sites at  $114.301 \pm 8 \text{ Mg C ha}^{-1}$ . A significant difference of  $F_{(3,9)} = 20.73$ ,  $p < 0.05$  was observed in the mean SOC amongst the study sites. Further a significant difference ( $p < 0.05$ ) in the mean SOC with respect to depth at which the soil was collected (Figure 4.5). The vegetated sites had higher mean SOC than their adjacent controls.

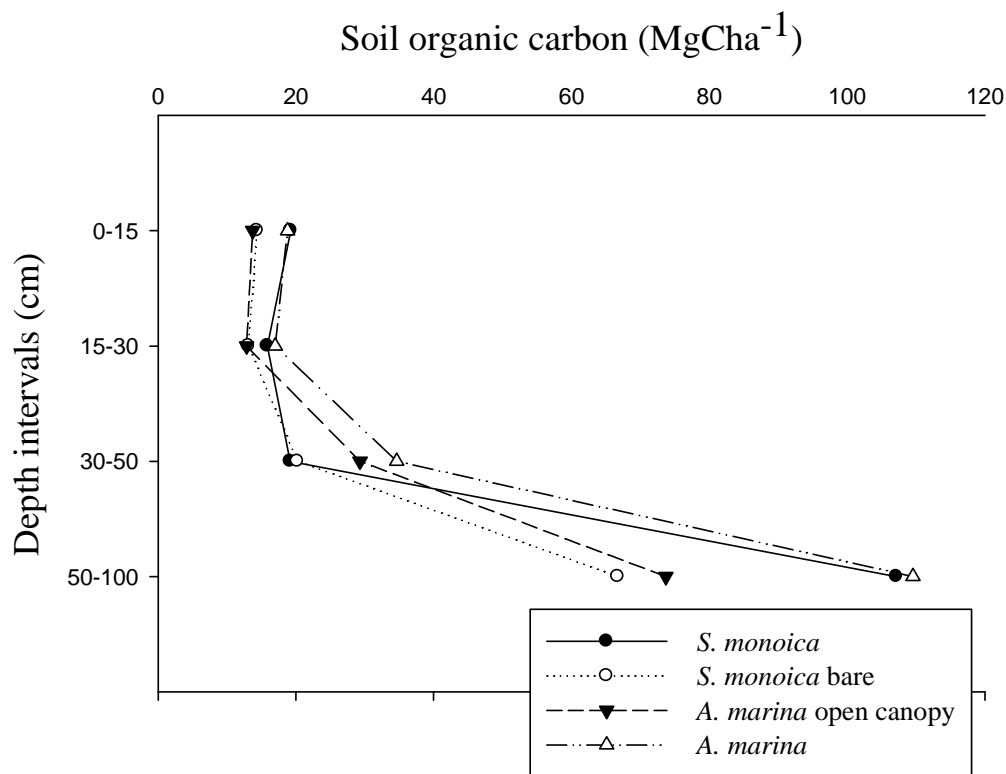


Figure 4.5: Mean  $\pm$  SE SOC mass through the soil depth profiles in the different sites at the study area.

#### 4.4.4 Mangrove vegetation organic carbon pool

Above and below ground biomass: in the *S. monoica* site was  $17.29 \pm 8.93 \text{ t ha}^{-1}$  and  $7.3 \pm 3.54 \text{ t ha}^{-1}$  respectively while in the *A. marina* site it was  $49.22 \pm 13.6 \text{ t ha}^{-1}$  and  $21.4 \pm 5.7 \text{ t ha}^{-1}$  respectively. Vegetation (mangrove) carbon pool was also estimated in the vegetated sites i.e: *S. monoica* and *A. marina* sites because the controls had no vegetation. The above ground carbon represented 4.64% and 10.8% of the organic carbon stock in the *S. monoica* and *A. marina* sites. Further the below ground root biomass also represented 1.75% and 3.95% of the organic carbon stock in the *S. monoica* and *A. marina* sites.

#### 4.5 Total organic carbon

There was a significant difference ( $p < 0.05$ ) in the total organic carbon amongst the four sites. The average total organic carbon in the study *S. monoica*, *A. marina*, *A. marina* open canopy and *S. monoica* bare sites were;  $172.978 \pm 15.51 \text{ Mg C ha}^{-1}$ ,  $211.14 \pm 15.35 \text{ Mg C ha}^{-1}$ ,  $129.62 \pm 5.26$

Mg C ha<sup>-1</sup> and 114.301±8.0 Mg C ha<sup>-1</sup> respectively (Figure 4.6). The total organic carbon was significantly (p<0.05) high in the natural *A. marina* and the *S. monoica* sites than their adjacent controls with the bare areas recording significantly (p<0.05) low total organic carbon. The *A. marina* area recorded the highest amount of total organic carbon than all the other three sites (a difference of 31.162±0.16 Mg C ha<sup>-1</sup> with the areas of *S. monoica*).

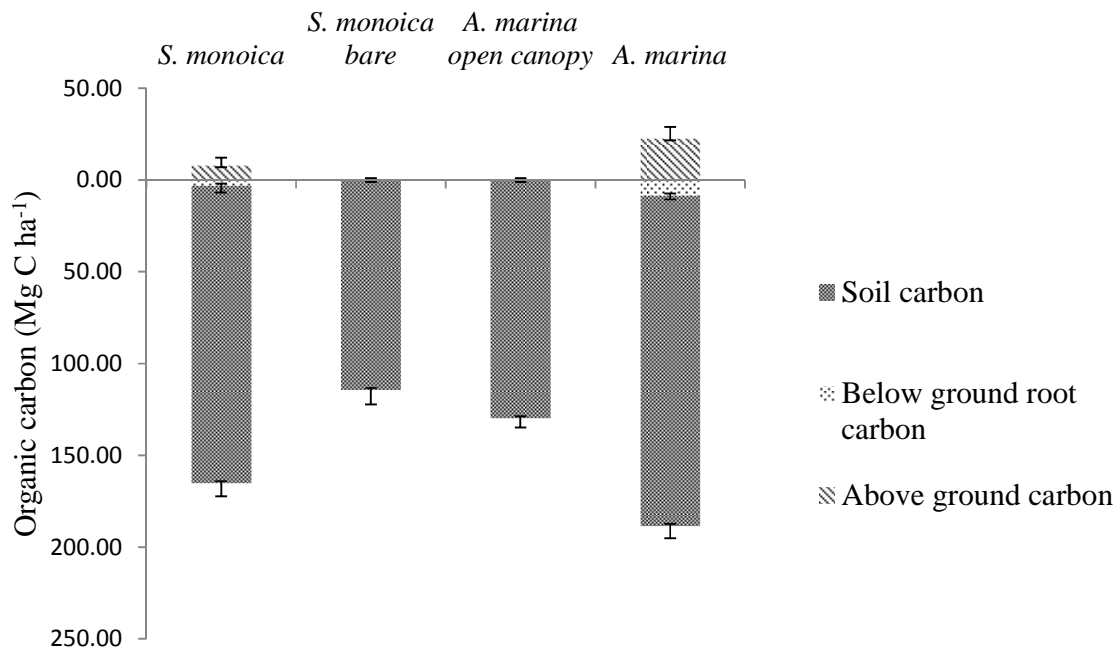


Figure 4.6: Comparison of the distribution of mean ± SE organic carbon stocks in the sampled pools of the four study sites at Mwache Creek.

## CHAPTER FIVE

### DISCUSSION

#### Introduction

This pioneering (in Kenya) study sought to determine the role (if any) of *Suaeda monoica* in facilitating recovery of a severely degraded mangrove site, a rarely investigated phenomenon in mangrove ecosystems. In this chapter, the findings and interpretations from the results section were done based on the study objectives and hypotheses.

#### 5.1 Sediment Conditions

Salinity and interstitial water temperature were lower in the vegetated sites as compared to their respective bare and open canopy control areas due to the shading by the canopy above (Frith & Brunnenmeister, 1980; Bosire *et al.*, 2003, 2005) which reduces water loss. In all cases, salinity directly corresponded with interstitial water temperature suggesting the strong role of irradiance due to exposure in driving salinity regimes. Past studies as Bosire *et al.*, (2003) and (2005); have recorded similar patterns. Temperature and salinity are controlling factors for seedling growth (Hastuti *et al.*, 2012). The temperatures recorded in all the sites were suitable for *A. marina* growth (its growth is inhibited at 37 °C (Hastuti *et al.*, 2012)). According to Ball (2002); salinity and growth of *Avicennia* have a positive correlation until optimum (growth of mangroves, like other halophytic species is typically enhanced under low to moderate salinities). Wells (1982); reported that *A. marina*, was among species that could grow in soils of over 65% salinity. As a major factor limiting mangrove regeneration, the *S. monoica* bare sites recorded salinity beyond the upper threshold.

Height above datum was higher in the *S. monoica* plots. Generally, *S. monoica* prefers to colonize elevated sites, which were created due to the past El Nino related flooding and sedimentation episodes (Kitheka, 2002). Sediment dynamics is an important driver controlling pioneer seedling establishment. The *S. monoica* patches (including their bare controls) had a higher proportion of sand and higher height above datum as compared to the *A. marina* areas (including their open canopy controls). Vegetation enhances sediment trapping thus allowing finer particles to settle within the two sites as compared to bare or open areas. Tidal currents is one mechanism responsible for dispersal and accumulation of sediments in mangrove forests

(Kitheka, 2000) with the trapped materials originating from within or transported from other areas into the mangrove area. The *S. monoica* roots possibly stabilize sediments and facilitate seedling establishment and growth.

Mangroves are among the most productive systems due to nutrients inputs from terrestrial sources, decomposition within the forest and inputs from the sea i.e. autochthonous litter and allochthonous inputs of natural and anthropogenic origins (Lee, 1990; Bouillon *et al.*, 2002. Nutrients concentration was low in the four sites as was the study in Bonje area by Bosire, (2010) save for low phosphorus observed in this study. Amounts of nutrients varied among the four study sites with the bare areas recording the highest concentration of phosphorous than the vegetated sites. Nitrates and ammonia were on the other hand higher in the two vegetated sites than the bare areas. The bare areas have had their substrates exposed to intense irradiation, suggesting that desiccation coupled with low faunal abundance could be playing a role in retarding nutrients availability (Bosire, 2006). This case proves that *S. monoica* modifies physico – chemical conditions and faunal colonization to support nutrients cycling. Nutrients between *A. marina* stand and their open canopy controls did not vary significantly probably due to the latter being a natural mangrove ecosystem occurrence.

## **5.2 Faunal Diversity and Abundance**

Mangroves associated fauna plays a significant role in the functionality of this ecosystem (Lee, 2008). Comparing the numbers of the epifauna assessed in the *S. monoica* patches and the forested *A. marina*, their open canopies and bare areas; epifauna numbers was highest in the natural forested *A. marina* areas. Similar observations have been made by Fondo and Martens; (1998) while investigating the effects of mangrove deforestation on macrofauna densities that reported higher numbers of epifauna in the natural mangrove areas than deforested areas. According to Ashton *et al.*, (2003) and Bosire *et al.*, (2004); species diversity, density and distribution of crabs and other gastropods respond predictably to disturbance, exploration and management of mangroves. This was exhibited by distribution of crabs and mollusks in the area.

The *A. marina* sites and their open adjacent controls recorded the highest crab species richness and high density than the *S. monoica* areas and their bare controls. Apart from *Uca* species that was recorded for all the four sites, most crab species prefer shaded areas for both shelter and

food. *Uca* spp is known to occupy open areas of sand and mud (Gillikin, 2000). The more numbers and high species richness of crabs in the *S. monoica* area than the bare areas suggests that *S. monoica* offers shaded habitats favorable for a variety of crab species. High number of mollusks in the *A. marina* and *S. monoica* sites is likely to be attributed to the shading effect that these two sites provide. Mollusks prefer moist conditions that are assured under closed canopy (Bosire, 2006). *Cerithidae* spp seemed relatively well adapted to the vegetated sites thus their abundance. The less density of these mollusks in both controls could most likely be attributed to lack of structural complexity and exposure to heat. Epifaunal distribution in an area is always influenced by organic matter and moisture content, an observation that was also made in this study. The similar species and high numbers of epifauna registered in the *S. monoica* areas as in the natural forested *A. marina* areas compared to the low numbers registered in the bare areas suggest that *Suaeda monoica* has potential to support faunal colonization and thus enhance ecosystem recovery.

Thirteen different types of sediment infauna taxa were identified from the four study sites. Of this, the natural *A. marina* forest recorded the highest numbers of infauna, atleast twelve of the thirteen sighted. According to Edgar, (1990) and Sasekumar & Chong, (1998); mangrove sediments generally support higher densities of benthic organisms than do adjacent non vegetated sediments which was the case in the bare sites. The presence of significant density of sediment-infauna in the *S. monoica* sites than in the bare areas suggests a recovering ecosystem. According to Schrijvers *et al.*, (1995); open areas have less fine grained size sediments, do not slow down incoming tides hence less organic deposition. Substrate type, salinity, oxygen, water table level, and organic material are among factors responsible for infauna distribution even in the four sites studied here. These, especially soil organic matter could be most responsible as infauna feed on particulate organic matter (POM).

### **5.3 Vegetation Characteristics**

#### **5.3.1 Vegetation structure and composition**

Description of vegetation characteristics often always includes measures of species composition, diversity, stem height, diameter, basal area, tree density as well as age-class distributions and spatial distribution patterns of these components in the forest (Robertson & Alongi, 1992). In coastal wetlands, vegetation composition and structure is determined by a suite of environmental

parameters, including salinity, nutrient concentration, frequency of inundation, topography, sediment type and inter species competition (Chen & Twilley, 1999; Robertson & Alongi, 1992). In the sites with vegetation, *A. marina* was the dominant mangrove species encountered. This pattern of growth can be closely linked to soil properties and disturbance history of the study site (Feller *et al.*, 2002). The relatively high density of smaller *A. marina* trees in the area recorded in this study suggest that *A. marina* is a pioneer mangrove species that can adapt well to changes in intertidal habitats and develop tolerance making it a high possible candidate for recruitment into a disturbed area as long as a micro-site is available and a window of opportunity appears.

### **5.3.2 Mangrove seedlings regeneration**

Population growth is principally driven by seedling recruitment and survivorship (Burns & Ogden, 1985; Kraus *et al.*, 2008) thus determining crop quality and productivity of forest stands. Natural regeneration has improved in the study site by about 53.6% (all regeneration classes) since 2010 (Bosire, 2010) supporting Kaino, (2013); who noted that the juvenile density per ha within the creek was sufficient for natural regeneration to occur. On average, the two vegetated sites had 10,233 juveniles/ha which is sufficient to support self-natural regeneration because it falls within the 5,000-10,000 juveniles/ha range given for the managed forests of Matang (UNDP/UNESCO, 1991) and the 7,000-11,000 juveniles/ha range in the Northern coast (Kairo *et al.*, 2002; Bosire *et al.*, 2003). Variation in juvenile densities of mangrove species in the two vegetated sites could be because; the *A. marina* forest stand site had enough number of mixed species of parent trees unlike the *S. monoica* site with few (*A. marina*) parent trees. On average, twelve standard parent trees are required per hectare to serve as seed sources for regeneration (FAO, 1994).

According to Di nitto *et al.*, (2008); strong waves prevent propagules establishment, expose the shallow root systems and prevent accumulation of fine sediment. The *S. monoica* thus plays an important role in stabilizing sediments, making the site suitable for seedling establishment and growth unlike the exposed bare sites subjected to relatively higher hydrodynamic activity and loose sediments. Mangrove regeneration has generally remained low in the larger bare area at the study site. Possible causes of this failure in seedling regeneration may include: a limited influx of propagules, propagules predation, high wave energies, hydrodynamic trapping or damage of propagules by floating debris or tidal position of the bare sites, with associated strong tidal

currents. However, with reproductive sea blight sites adjacent, propagules supply can be ruled out as a cause. The most probable cause could be because of inundation and high currents (Bosire *et al.*, 2003). The low silt and clay content (26.4%) in this bare site may be indicative of the impact of these hydrodynamic processes.

## **5.4 Carbon Stocks**

### **5.4.1 Soil organic matter and carbon concentration**

Mangrove ecosystems have the capacity to efficiently trap suspended materials from water columns (Kristensen *et al.*, 2008), litter from trees and subsurface root growth to provide a significant input to soil organic matter (Alongi, 1998) due to long flooding periods and low decomposition rates sustaining anoxic conditions (Ceron-Breton *et al.*, 2011). The two vegetated sites in the study area had higher percentage of soil organic matter and higher soil carbon concentrations than their adjacent controls, due to lower bulk densities and higher percentage of silt and clay than in their respective adjacent controls which enhances burial of carbon. Other than this, because tides are responsible for importation and exportation of organic materials into and from mangrove systems (Kitheka *et al.*, 2005; Kristensen *et al.*, 2008) the vegetation of the *A. marina* and *S. monoica* sites increase roughness by their root networks and hold water, leading to cohesion enhancing conditions favoring carbon accumulation in these soils. Variations with increasing depth in the organic matter and organic carbon concentrations of the four study sites were similar to trends of Tudor (Olagoke, 2012) and the larger Mwache Creek mangroves, while the higher amounts of these two parameters recorded in the subsurface layers of the soils than in the mid layers could be due to inputs from external land use practices surrounding the study area.

### **5.4.2 Bulk density of the sediments**

Well-structured soils are often expected to have a low bulk density that increases with depth. Low bulk density means proper aeration and inherent conditions for edaphic life and nutrient turnover (Hakanson & Lipiec, 2000) in contrast with those of high bulk density. The study site bulk density figures were however very high compared to figures from studies in Tudor (Olagoke, 2012), Micronesia (Kauffman *et al.*, 2011) and Mexico. The latter two are known for extensive mangrove forests. The relatively higher bulk density may be due to poor vegetation structure and low productivity in the study site compared to Tudor, Micronesia and Mexico. The



bare sites had the highest bulk density because they are exposed thus suffer more water loss, hence more compaction.

Previous studies have different reports on bulk densities by depth. In Tudor Creek, bulk densities by depth did not have clear trends, some decreased by depth others increased by depth (Olagoke, 2012), the later phenomena was also observed in a study in Campeche, Mexico (Ceron-Breton *et al.*, 2011). In contrast, Donato *et al.*, (2011); reported fluctuations of the same to a meter deep and an increase thereafter. The unusual trends of bulk density in all the four sites could represent heavy and intermittent deposition in the area. This Creek, according to Kitheka, (2000); experience heavy sedimentation. Though not significant, the difference in bulk density between the vegetated sites and their adjacent controls might have been influenced by vegetation densities, morphology and heterogeneity in the vegetation rooting system and presence of fauna. Other than this, the bare soils might be exposed to direct solar radiation that might have led to a substantial loss of organic carbon.

#### **5.4.3 Carbon Stocks**

The above - ground biomass for the two vegetated sites was very low and does not represent the AGB of Mwache Creek that stands at  $229.38 \pm 53.28 \text{ t ha}^{-1}$  (Kaino, 2013). This low above - ground biomass may have resulted because of the nature of the site. Being landward, the site is poor in nutrients and has drier ground. Standing biomass in a mangrove forest is a function of the system's age, productivity and organic matter allocation and exportation strategies (Kasawani *et al.*, 2007). In this study the total above - ground biomass of  $49.216 \text{ t ha}^{-1}$  and  $17.28 \text{ t ha}^{-1}$  for *A. marina* and *S. monoica* sites respectively is close to values by Shafiq & Chandani, (2012); in the Island of Mauritius. As expected, the *A. marina* site recorded high above ground biomass than the *S. monoica* site; this was because of the number of standing grown trees in the *A. marina* sites compared to the few old and very young trees in the *S. monoica* site. Usually, as stand develops so does its accumulation of biomass in woody tissues increase. The size of trees and density principally determine stand biomass. The wood density of the tree further affects carbon content of the plants and hence that of the stand. The ratio of above ground biomass to below ground biomass in the two vegetated sites was 2.4 for the *S. monoica* bush areas and 2.3 in the *A. marina* stand area. These ratios lie in the 0.8 to 4.0 ratio ranges for Tudor creek (Olagoke, 2012).

Estimates of ecosystem carbon stocks in mangroves indicate a large proportion of carbon in the soils (Bouillon *et al.*, 2008; Alongi, 2009; Kauffman *et al.*, 2011; Olagoke, 2012). This was also the case of soil carbon stocks of the four sites; a phenomenon attributed to accumulation of peat from roots and anoxic conditions that slow decomposition. The vegetated sites had higher soil organic carbon than their adjacent controls. This is because bare areas are exposed and lead to emission of carbon IV oxide due to loss of aboveground carbon stocks and increased rates of soil decomposition (Donato *et al.*, 2011). Although the *A. marina* stand site had the highest soil organic carbon, this was low compared to the average of  $246.1 \pm 71.5 \text{ Mg C ha}^{-1}$  (Bosire *et al.*, 2013) recorded for the Creek.

Variability of carbon stocks within the four sites was evident from the above-ground vegetation structure and composition. Mangroves have a wide range of ecosystem carbon stocks and a great variability in structure. Higher carbon stocks of the sampled sites in the study were measured in the vegetated sites than their bare adjacent controls, thus emphasizing the impact of mangrove degradation in the exacerbating carbon emissions. Once again, the role of *S. monoica* in enhancing carbon storage as another ecosystem service (beyond encouraging regeneration and faunal colonization) was apparent.

## CHAPTER SIX

### CONCLUSIONS and RECOMMENDATIONS

#### Introduction

This chapter presents a summary of study findings conclusions based on study objectives and way forward in terms of recommendations and further research.

#### 6.1 Conclusions

Natural and human induced disturbances modify mangrove habitats resulting in the loss of mangrove ecosystems functional attributes, in this study's case; support of rich fauna taxa, regeneration of mangroves, carbon sink and controlling of natural sediment conditions. These then may alter the functioning and structure of the ecosystem which was apparent from the impoverished conditions of the bare sites, whereas the *S. monoica* sites had significantly rich and diverse fauna recruited, high mangrove juvenile densities, high ecosystem carbon stocks, low temperatures and salinity as well as high soil organic matter content.

Strong waves prevent propagules establishment, expose the shallow root systems and prevent accumulation of fine sediment. The *S. monoica* thus plays an important role in stabilizing sediments, making the site suitable for seedling establishment and growth unlike the exposed bare sites subjected to relatively higher hydrodynamic activity and loose sediments. Additionally, the near similarity in species and high numbers of fauna registered in the sea blight sampling sites as in the natural *A. marina* stand sites compared to the low numbers registered in the bare areas suggest that sea blight has potential to support faunal recolonisation and thus enhance ecosystem recovery. Further, the high carbon stocks in the *S. monoica* sites as the natural mangrove areas emphasize on the role of this herbaceous plant species in enhancing carbon storage of the recovering ecosystem.

If the co-occurrence of faunal species/taxa, sediment conditions, carbon storage and vegetation growth in different sites in the study is a reflection of ecological function equivalency, then the *Suaeda monoica* sites were more akin to natural *A. marina* stands and their adjacent open canopy sites and departed generally from their bare adjacent control sites (these sites seemed to be functionally developing towards the original natural forest), emphasizing on the nursing role of *S. monoica* in recovery of degraded mangrove systems. The findings of this study indicate that

*S. monoica* enhance recovery of the entire suite of structural and functional features of the forest. And that natural regeneration can occur much faster if beneficial species such as *S. monoica* are present.

## **6.2 Recommendations**

Restoration of Mwache Creek mangroves have been met with a lot of failures in the past, since mangroves seedlings are usually planted at a young age (< 1 year) and as such may be highly susceptible to a range of environmental stresses. To improve this efforts therefore, the study recommends that;

- Since the sites undergo hydrological and other physical changes and that restoration success rate of the degraded sites largely hinges on planting species with tolerant traits, *A. marina* seedlings can be planted in this site because of higher survivability and growth rate than all other species and because *S. monoica* sites have shown changes from harsh to tolerable conditions, planting can therefore be done in areas of extensive *S. monoica* growth to help nurse the plants and facilitate these efforts.
- Although this work directly compared the effects of only one mangrove ecosystem associate species, there is an extensive list of mangrove associate species worldwide of more than 25 plant families of grasses, rushes, sedges, succulents, forbs and ferns. Future works therefore need to screen these plants for potential beneficial species in other geographical areas.
- Additionally, there is need for research on site-specific information and mechanisms involved if these plant associate species are to be applied to mangrove intertidal restoration projects.

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