CARBON STOCKS AND COVER CHANGE IN THE MANGROVE FOREST OF MTWAPA CREEK, KENYA

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A thesis submitted to the Graduate School in partial fulfillment of the requirement for the Award of the Degree of Master of Science in Geography of Egerton University

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DECLARATION AND RECOMMENDATION

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

I dedicate this Thesis to my Mother Lilian, Sisters; Irene, Everline, Caroline, Janet and Winnie
and the entire Kormom's family for their faith in me
and the endless prayers and encouragement
throughout my studies.
May God bless you abundantly.

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ABSTRACT

Most ecological assessments often focus on one ecosystem without investigating the integrity of contiguous ecosystems and implications thereof on the habitat of interest. From a conservation perspective, ignored ex situ processes may counteract management interventions in an ecosystem of interest if the former aren't taken into account. This study sought to assess land-use cover change and its potential relationship or impact on the integrity of contiguous mangroves downstream. Carbon stocks data was collected from 36 systematically established 10 x 10 plots set along stratified transects. Biomass carbon was estimated based on general allometric equations applicable globally and wood densities derived within WIO region. Soil carbon estimation was done using soil samples extracted from the 36 plots. Loss on ignition procedure was used to get %SOM then calorimetric equation from adjacent creek was applied to get soil organic carbon. Mangrove cover and land use changes within Mtwapa system in Kenya was assessed using multi-temporal medium resolution Landsat TM (1992) and SPOT images taken in 2000 and 2009. Maximum likelihood classification method was used in ERDAS 9.1 and ArcGIS 10.0 softwares. Land-cover changes around the creek from 1990 to 2009 revealed a high rate of upland deforestation (3.85% yr^{-1}) and an increase in agricultural land (13.9% yr^{-1}). Between 1992 and 2009 the mangrove forest lost 21% of the cover, translating to 1.2% cover loss per annum which fell within the global mean of 1 – 2%. The stocking rates of mangroves in Mtwapa were estimated at 2870±295 stems/ha. The mean biomass carbon for the study area was 49.46±8.49 Mg C ha^{-1}, with no significant variation between sites ($p>0.5$). Mean SOC of the study area was 196.09±19.31Mg C ha^{-1} giving a total ecosystem carbon of 245.54±20.95Mg C ha^{-1}. This was quite low compared to in-country and off-shore carbon stocks and is likely due to poor forest structure in Mtwapa creek associated with anthropogenic disturbance as noted by high stump count 2,425±423 stumps/ha. There was a highly positive correlation between land use cover change (agricultural expansion – $R^2 = 0.70$) and mangrove cover Although these mangroves recorded high stocking densities, high degradation rates and observed sedimentation due to poor land-use practices upstream have led to poor stand structure hence low carbon stocks. A landscape approach which combines sound land husbandry upstream and mangrove conservation is recommended.

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LIST OF ACRONYMS AND ABBREVIATIONS

FAO	Food and Agriculture Organization
GCP	Ground Control Points
GIS	Geographic Information System
GHG	Green House Gas
IPCC	Intergovernmental Panel for Climate Change
IUCN	International Union for Conservation of Nature
KMFRI	Kenya Marine and Fisheries Research Institute
LOI	Loss on Ignition
LULC	Land use Land cover
LULUCF	Land Use, Land Use Change and forestry
MASMA	Marine Science for Management
REDD+	Reduced Emissions from Deforestation and forest Degradation
RS	Remote sensing
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
UNEP	United Nations Environmental Programme
UNFCCC	United Nations Framework Convention on Climate Change

Definition of Terms

Biomass:	Is used to refer to the total dry weight or organic content of standing living forest vegetation and is expressed in Mg C ha^{-1}.
Built-Up:	An area densely covered by houses or other buildings and infrastructure.
Bulk Density:	Refer to the measure of compactness of soil, and calculated by dividing the dry weight of soil by its volume.
Carbon stocks:	This refers to the amount of carbon contained in the living and dead biomass together with carbon fixed and trapped in the column of organic soil lying beneath coastal habitats as a result of prior sequestration (Sifleet *et al*., 2011).
Cover change:	Refers to the modification from initial physical land type to another different land type as a result of natural or human interruptions.
Cropland:	These are lands used for the cultivation of crops, including tree crops and shrub crops.
Forest:	These are natural or planted stand of trees of all ages covering an area.
Land-use:	Is the arrangement of activities and inputs people undertake in a certain land cover type for production or for maintenance (FAO, 1997 and UNEP, 1999)
Land cover:	Refers to physical land type whether occurring naturally or as a creation of man.
Mangrove forest:	These are woody inter-tidal trees and shrubs growing in saline sediment in the coastal zone with low oxygen soils and hyper salinity.
Sand flat:	This is an area of land covered with sand or mud at the edge of the creek waters and edges of the mangrove forest
Water:	The main water body of the study area extending landwards from the ocean; The Mtwapa creek.

CHAPTER ONE

1.0 Introduction

This chapter provides the general introduction to the mangrove ecosystem, emphasizing on the motivation for the thesis. It highlights both global and local environmental and socio-economic importance of this habitat and threats compromising the integrity of the system that necessitated need for current research with scientific justification based on available information. Statement of problem, aims and objectives of the research are well spelt out, with clear statement of research hypotheses. At the close of this chapter, scope of the study is stated.

1.1 Background of the Study

Mangroves are woody inter-tidal trees up to medium height and shrubs growing in saline coastal sediment habitats occurring along the sheltered coastlines, mudflats and riverbanks between land and sea in tropical and subtropical regions (Neukerman *et al*., 2008). They sustain a wide bio-diversity of flora and fauna and provide many ecosystem services such as coastal protection from storms, reduction of shoreline and riverbank erosion, stabilizing sediments and absorption of pollutants (Duke *et al*., 2007; IUCN, 2006).

Globally, mangroves are disappearing at alarming rates throughout the world, mainly owing to anthropogenic activities (Ong and Khoon, 2003). With recent changes in global climate and related effects, over the last 50 years, about one-third of the world's mangrove forests have been lost (Alongi, 2002). Projections suggest that mangroves in developing countries are likely to decline by more than 25% in the next three decades (Ong and Khoon, 2003).

Traditionally mangroves in Kenya have served the local communities with a number of goods and services such as resources for building huts, for boat construction, fencing, making fish traps, source of fuel as firewood, medicine, insecticides and even fodder for livestock (Kairo, 1995). Up to date mangrove forest ecosystems have considerable economic potential in terms of timber, wood products, fisheries and tourism (UNEP, 2003a). Although economic valuation of mangroves in Kenya is still scanty (Kaino, 2012), available information indicates estimates of US$3000/ha/yr in Gazi Bay for a 12-year plantation (Kairo et al., 2009).

Recently, mangroves have been found to play an important role of sequestering and storing carbon more than any other forest ecosystem on Earth (Donato *et al*., 2011). However, anthropogenic activities and climate change related impacts continue to accelerate degradation of these ecosystems.

Research studies on mangrove biodiversity and conservation have increased the understanding of values, functions and attributes of mangrove ecosystems (IUCN, 2006). Mangrove in Mtwapa creek is capable of providing sustainable economic and ecological benefits to its residents (Kairo, 2002). However it is located in a peri-urban setting with close proximity to settlements and hotels (Okello *et al*., 2013). Studies on peri-urban mangrove forests have reported human activities as the main cause of mangrove forest degradation, (Omar *et al*., 2009 and Kaino, 2012).

Mtwapa creek is subjected to problem related to land use effects like urbanization, Aquaculture and agricultural activities from surrounding areas, besides unsustainable indiscriminate deforestation and pollution from raw sewage from nearby hotels, residential quarters and a government prison, (Okello *et al*., 2012).

1.2 Statement of the Problem

Mangrove forests sequester and store large amounts of carbon especially due to the fact that mangroves can store large amounts of carbon in the sediments besides its biomass making it very important within the frameworks of mitigating climate change. However, these ecosystems are threatened by overexploitation, conversion, land use changes and sea-level rise. About 1-2% of global mangrove cover is lost annually (Giri *et al*., 2011). In Kenya peri-urban mangroves have recorded higher loss rates of about 2.7% annually (Olagoke, 2012). Loss of mangroves means loss of ecosystem services.

Despite that, a lot of gaps still exist as most of the mangrove carbon stocks are unaccounted for. Spatio-temporal cover change dynamics are limited owing to complexity associated with availability and analysis of past cover change data which can only be derived remotely using satellite images and past aerial photographs. Furthermore these are complex wetland ecosystems that have been for long neglected and degradation having immensely taken place over the years.

Mtwapa mangroves are peri-urban systems subjected to a lot of anthropogenic pressure from deforestation and degradation due to upland activities. This study was carried out to estimate carbon stocks and determine the temporal cover change dynamics in order to add information on status of mangrove degradation in Mtwapa creek besides the general effects related to 1997 ENSO events.

1.3 Broad Objective

To assess carbon stocks and status of temporal cover change dynamics of peri urban mangroves of Mtwapa creek, for enhanced conservation and better management of mangrove forest resource

1.3.1 Specific Objectives

1. To estimate below ground and above ground carbon stocks in Mtwapa creek

2. Determine spatio-temporal changes in mangrove cover and Land cover changes between 1990 and 2009 in Mtwapa Creek

3. To establish spatio-temporal relationship between mangrove cover change and land use changes in Mtwapa creek

1.3.2 Research hypothesis

Ho_1 There is no statistically significant variation in carbon stocks between the study sites in Mtwapa Creek.

Ho_2 There is no statistically significant changes in mangrove cover and land use changes between 1990 and 2009 in Mtwapa Creek.

Ho_3 There is no relationship between spatio-temporal mangrove cover change and land use changes in Mtwapa Creek.

1.4 Justification/ Significance of the Study

There is a recent information need on area, biomass and carbon entities of the coastal and marine ecosystems (Rabiatul *et al*., 2012). The carbon stored in a hectare of mangroves contribute to emissions as three to five hectares of other tropical forest (Murray *et al*. 2011) and halting current rates of mangrove deforestation would lead to mitigation of approximately 170 – 490 million tCO_2e per year (Murray *et al*. 2011). Therefore, quantification of carbon and cover trends will highlight the importance of these ecosystems in carbon capture and storage, hence providing the basis for its conservation.

Mangroves also play a crucial role to the coastal ecosystem among them: breeding ground for fisheries, habitats for wildlife, sources of livelihood to people in the tropics and sub-tropics and provides coastal protection from storm surges (Kairo, 2001). However, degradation from unsustainable harvesting, conversion to other uses, coastal urban development, household and industrial pollution and sedimentation continue to be a threat (Giri, 2010, Bosire, 2010, Rabiatul *et al*., 2012 and Duke *et al*., 2007).

Studies on Mangrove forest cover change in Mombasa Kenya, (with satisfactory levels of accuracy at (87.5%)) estimated forest cover loss rate at 0.7% per annum (Kirui *et al*., 2012). Site specific forest cover assessment, in Mwache and Tudor Creeks indicates 2.7% annual loss of mangrove cover. However, spatial and temporal variations exist possibly because of legislative inadequacies and difference in habitat alteration pattern, calling for a site specific study.

Mtwapa creek mangrove forest is a peri-urban system that prompted for assessment of cover changes to ascertain threats facing this ecosystem. Because of the complexity associated with coastal fringe ecosystems, the management of these regions requires an integral view. Hence this study employed both Field survey and satellite Imagery in analyzing the Spatio-temporal dynamics of land cover and land use changes in order to provide a synoptic view which incorporates the time scale and allows a comprehensive spatial analysis of change.

1.5 Scope of the study

This study was conducted in Mtwapa creek located in the Northern Coast of Mombasa, Kenya. It focused on estimating the carbon stocks within the mangrove ecosystem and assessing surrounding spatio-temporal cover change between 1990 and 2009.

This study applied generic allometric equations and species specific wood densities for biomass estimation and analysis of carbon for above and below ground biomass. Generic allometric equations were sourced from general equations derived in Palau Micronesia (Komiyama *et al*., 2008). Wood densities derived from a study in Mozambique were used. This is because it falls within Western India Ocean (WIO) region, and lack of local mangrove species specific wood densities. Soil carbon was calculated based on lab analysis results and calorimetric equations derived from the analysis of soils from adjacent Tudor creek.

Image analysis was done using Landsat of April 1990, SPOT of May 2000 and SPOT of Jan 2009 which fairly falls within the dry season of between January and April. Low rainfall season in Mombasa falls between January and April, although in the year 2000 SPOT image of May had least cloud obscurity despite being captured in May and hence its inclusion in the study.

CHAPTER TWO

LITERATURE REVIEW

2.0 Introduction

This chapter reviews the global and local literature on mangrove species and ecology, with focus on factors that determine their distribution, total area of coverage, type of species and their distribution. This chapter also reviews literature on mangrove biomass and its sequestration potentials, this ecosystem being one of the most carbon-rich tropical forests with extremely high carbon storage rates. Lastly and important this section looks at literature on mangrove and land use cover change and other threats to this fragile ecosystem that formed the spine and basis of this study.

2.1 Mangrove species and ecology

Mangroves are salt tolerant trees or shrubs located within high and low tidal zone along the shoreline and restricted to tropical and sub-tropical regions of approximately between 25^0N and 25^0S (Giri *et al*., 2011 and Tomlinson, 1986). The factors that determine their distribution are; major ocean currents, 20^0C isotherm of sea water and are distributed from mean sea level to highest spring tide, colonizing sheltered shores and estuarine areas of tropical ocean waters (Alongi, 2009). The total area of mangroves in the world has been approximated at 181,000 square kilometers (sq. km) and accounting for about 2.4 % of tropical forests in the world, comprising of 9 orders, 20 families, 27 genera and about 70 species according to World Mangrove Atlas (Spalding *et al*.,1997).

There are around 80 species of mangroves found throughout the world (Saenger *et al*., 1983). Most commonly they occur within tropical and subtropical sheltered coastal areas subjected to tidal influences (Giri *et. al*., 2011). Mangroves can be divided into two distinct groups: exclusive and non-exclusive. Exclusive mangroves are the largest group, comprising around 60 species (Saenger *et al*., 1983). These mangroves are confined to intertidal areas and have not been found to exist within any other type of vegetation community (Alongi, 2009).

The remaining 20 plant species considered to be mangroves are referred to as non-exclusive, and are not restricted to the typical mangrove environment and are often found within drier, more terrestrial areas (Tomlinson, 1986). Areas, which have a great variety of mangrove species, are found along coasts that receive high rainfall, heavy run-off and seepage into the intertidal zone from the hinterland. Such areas are commonly subject to extensive sedimentation, which provides a diverse range of substrate types and nutrient levels, which in turn are favorable for mangrove growth (Gilbert and Janssen 1998).

Mangrove communities most commonly occur in areas where the average temperature of the coldest month is higher than 20°C and where the seasonal range does not exceed 10°C. Temperatures of around 5°C and frosts also limit mangrove distributions (Tomlinson, 1986). Overall, there are 15.2 million ha of mangroves around the world; down from 18.8 million ha in 1980 (FAO, 2006; Spalding *et al*., 2010). Mangroves represent less than 0.7 % of tropical forests (Giri *et al*., 2011) and despite their limited area they are of global economic, environmental and social importance to humans (FAO, 1994; Costanza *et al*., 1997; Kathiresan and Bingham, 2001). The word is used in at least three senses: most broadly to refer to the habitat and entire plant assemblage or mangal for which the terms mangrove forest biome, mangrove swamp and mangrove forest are also used; to refer to all trees and large shrubs in the mangrove swamp, and narrowly to refer to the mangrove family of plants.

Mangrove plants require a number of physiological adaptations to overcome the problems of anoxia, high salinity and frequent tidal inundation (Duke *et al*., 1998). Hence, the mix of species is partly determined by the tolerances of individual species to physical conditions, such as tidal inundation and salinity, but may also be influenced by other factors, such as predation of plant seedlings by crabs (Abuodha and Kairo, 2001).

Mangrove forests play a central role in transferring organic matter and energy from the land to marine ecosystems from fallen leaves and branches, and forms the base of important marine food chains (Hogarth, 1999). Bacteria break down the detritus, releasing useful nutrients into the water that can then be used by marine animals (Kristensen *et al*. 2008). The dense root systems form a home for fish, crabs, shrimps, and molluscs and also serve as nurseries for juvenile fish like coral reef fish, for example, spawn in mangrove forests where young fish stay until they are old enough to move to the reef (Hemminga *et al*. 1994). In addition, mangrove forests are nesting and migratory sites for hundreds of bird species, as well as home to a wide variety of reptile, amphibian, and mammal species (Dahdouh-Guebas, 1998).

The intertidal existence to which these trees are adapted represents the major limitation to the number of species able to thrive in their habitat (Hogarth, 1999). High tide brings in salt water, and when the tide recedes, solar evaporation of the seawater in the soil leads to further increases in salinity (Kathiresan and Bingham, 2001). The return of tide can flush out these soils, bringing them back to salinity levels comparable to that of seawater. At low tide, organisms are also exposed to increases in temperature and desiccation, and are then cooled and flooded by the tide. Thus, for a plant to survive in this environment, it must tolerate broad ranges of salinity, temperature, and moisture, as well as a number of other key environmental factors-thus only a select few species make up the mangrove tree community. Though the trees themselves are few in species, the ecosystem that these trees create provides a home for a great variety of other organisms (Saenger and Snedaker, 1993).

In Kenya, Mangroves are the major ecosystem along the sheltered shoreline, estimated to cover approximately 54,000 ha along the coastline with dominance in Lamu and Tana River counties (Kairo, 2002).

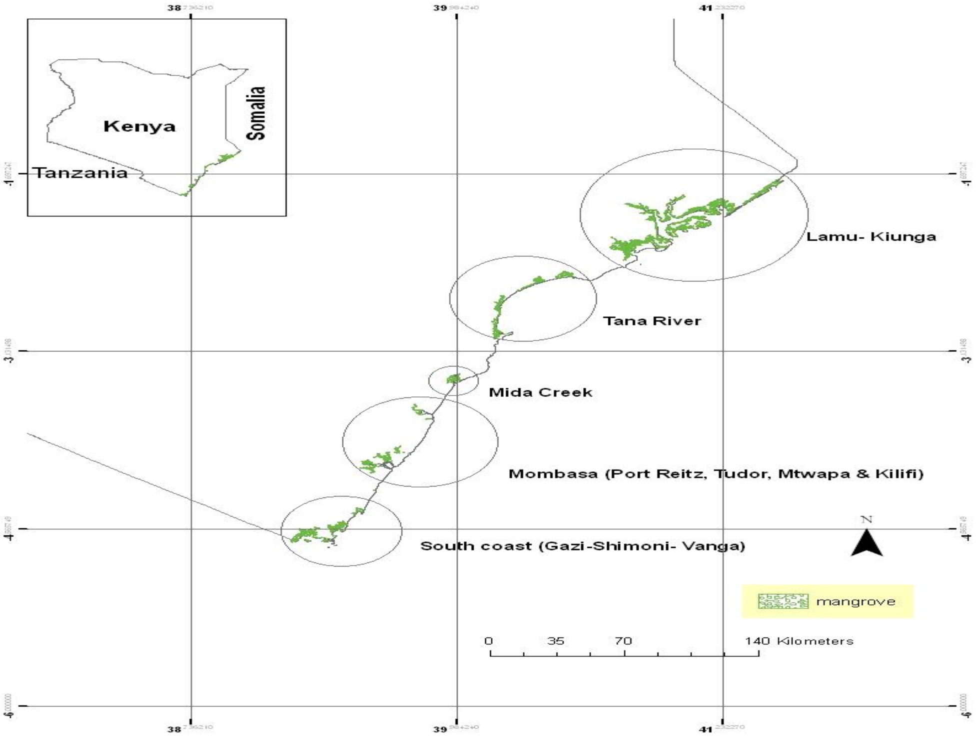

Figure 2.1: Map of the Kenyan Coast showing mangrove areas. Source: (Kirui *et al*., 2012)

They have been estimated recently to cover 45,590 ha (Kirui *et al.*, 2012); representing 3% of natural forests and 1% of the state land (Wass, 1995). These forests occur in creeks, protected bays and estuaries spread along the 600km coastline from Kiunga at the Kenya Somali border to the north, to Vanga at the Kenya-Tanzania border to the south (Figure 2.1). Altogether there are 73 true mangrove species in the world (Spalding *et al*., 2010); only 9 of these have been described in Kenya and the Western Indian Ocean (WIO) region (Spalding *et al*., 1977; Spalding *et al*., 2010). The dominant mangrove species in Kenya are *Rhizophora mucronata and Ceriops tagal* that represent more than 70% of mangrove formation (Kokwaro, 1985; Ferguson, 1993). Other common species are *Sonneratia alba, Avicennia marina* and *Bruguiera gymnorrhiza.* Less frequent species, include; *Xylocarpus granatum, Xylocarpus mollucensis, Lumnitzera racemosa* and *Heritiera litoralis* (Kokwaro, 1985; Dahdouh-Guebas *et al*., 2000b).

Mangroves ecosystem along Kenyan coast harbor a broad range of animal life including both terrestrial and aquatic. The terrestrial life includes mammals, birds, reptiles and insects. A study in Tana Delta showed presence of numerous animal species including Crocodiles, hippopotamuses, rodents, and fruit bats while Mida Creek has numbers of families of birds that nest and feed in the mangrove forests (UNEP, 2003a). Aquatic life supported by the mangrove forests include prawn, shrimp, crab, mollusk and oysters in addition to common fish species like stripped catfish, *Plotosus lineatus*, gobies common silver biddy, *Gerres oyena*, herring and Spratellomorphia (UNEP, 2003b).

2.2 Mangrove biomass and carbon stocks

The ability of forests to absorb and sequester carbon is an essential component of climate change mitigation (Chhatre and Agrawal 2009). Carbon sequestration by vegetation refers to the process whereby carbon is removed from the atmosphere through photosynthesis, and then stored in the biomass and soil of the capturing ecosystem (Nelleman *et al*., 2009). Carbon sequestration has always been considered in terrestrial forest systems (Nelleman *et al*., 2009). However, mangroves are one of the most carbon-rich tropical forests (figure 2.2) and have extremely high carbon storage rates (Donato *et al*., 2011).

Despite their relatively small global extent of <1% of all tropical forests worldwide FAO (2006), vegetated coastal ecosystems are disproportionately important in sequestering carbon dioxide as compared with terrestrial ecosystems (McLeod *et al*., 2011). Current studies indicate that they sequester up to four times better than tropical forests (Rabiatul *et al*., 2012) hence making this ecosystem attract attention in recent research on their capacity to contain and remedy climate change (Mwihaki, 2013). Carbon stocks inventory and emissions from mangrove forest ecosystem have gained great attention due to rising awareness on climate change resulting from forest deforestation and degradation (Angelsen, 2007).

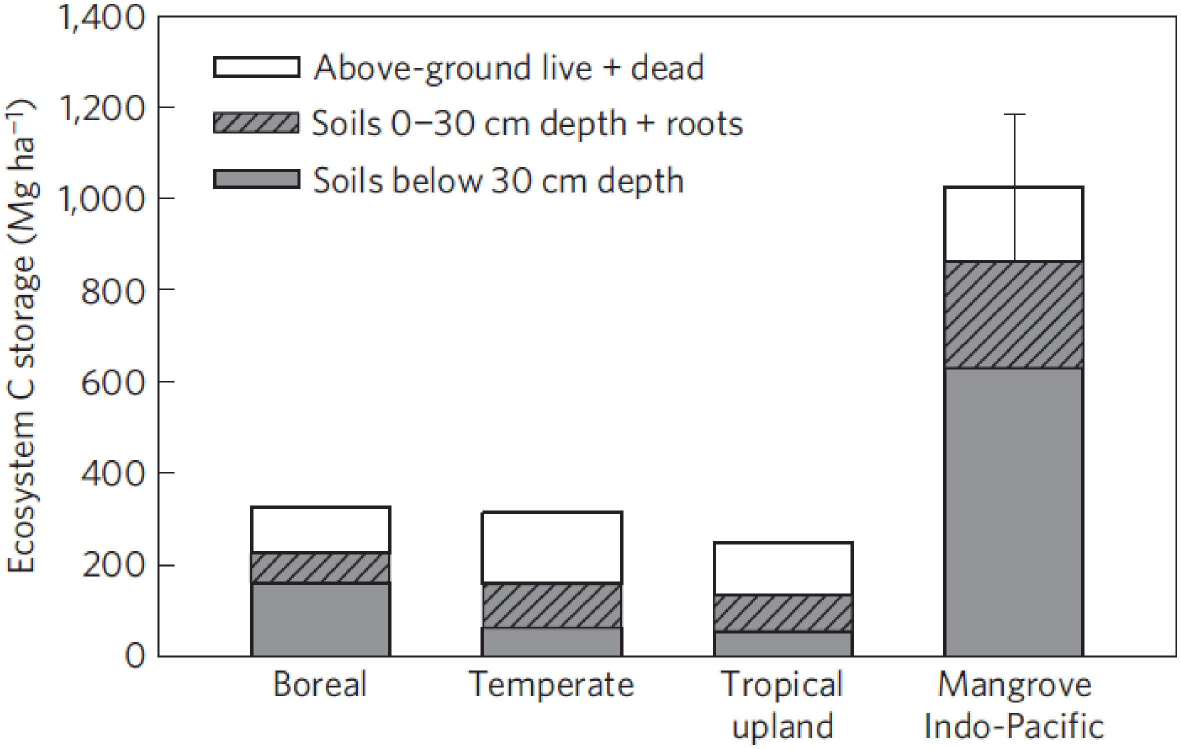

Figure 2.2: Graph showing ecosystem carbon storage per unit area in different forests. Source: (Donato *et al*., 2011)

The carbon sequestration process occurs as a result of the autotrophic nature of coastal ecosystems (Murray *et al*., 2011). Coastal ecosystems, including mangroves, remove CO_2 from the atmosphere through photosynthesis, return some to the atmosphere through respiration and oxidation, and store the remaining carbon in two pools: living biomass (both aboveground and belowground vegetation) and soil organic carbon (Murray *et al*., 2011).

There is increasing attention from the climate and conservation communities with focus on climate change mitigation and adaptation and a boost of ecosystem services through the endorsement of the concept of carbon financing schemes under the United Nations Framework

Convention on climate Change (UNFCCC) e.g. REDD+ (reduced emissions from deforestation and degradation through enhanced forest conservation) (Alongi, 2011). Carbon trading is becoming a key in motivating local communities to conserve and protect mangrove forests. This follows the agreement of 11th session of UNFCCC stipulating on the compensation of countries that shall reduce emissions from deforestation and degradation (Angelsen, 2008).

The importance of immediate protection measures and conservation activities to prevent loss of vegetation including mangroves is essential worldwide (Angelsen, 2007). Biomass loss and carbon emissions resulting from deforestation and degradation from anthropogenic and natural factors like climate, natural fires and desertification are uncertain (Murray *et al*., 2011). The information on amount of biomass of forests for estimation of carbon stocks is needed because when the changes occur much of carbon stocks in the ecosystem is released to the atmosphere causing greenhouse effect and subsequently climate change.

However, losses of biomass associated with selective wood harvest may lead to change in carbon stocks in forests but without a change in forest area (Laurence *et al*., 1998). Likewise, accumulations of biomass in growing and recovering forests may lead to increase in biomass without change in forest area (Barlow, 2003). These changes in biomass are generally more difficult to detect with satellite data than changes in forest area and more difficult to document from census data; yet, the changes in carbon may be significant (Laurence *et al*., 1998).

2.3 Land use and mangrove cover change

Mangrove forests have been lost to urban expansion, tourism development, and other infrastructure needs (Spalding *et al*., 2010). Mangroves have been cleared for urban expansion in a number of major cities in the world including Singapore, Jakarta, Bangkok, Mumbai, Lagos, Free town, and Douala (Spalding *et al*., 2010). In these areas large tracts of mangroves have been converted to waterfront property, marinas, tourist resorts and golf courses. Moreover, in cases where mangroves are not entirely cleared, development activities can still negatively affect forest health.

Mangroves thrive in areas that receive freshwater run-off and tidal water flushing. The building of infrastructure such as roads, sea defences and drainage canals can create barriers to natural water flow (Spalding *et al*., 2010). This can have a devastating effect on mangroves because regular flushing with saltwater or freshwater prevents the hyper-salinization of the mangrove environment and protects the supply of nutrients and sediments. Together, the obstruction of both tidal and freshwater flow results in increased salinity of the mangrove environment and leads to reduced forest growth.

Human precautions to cope with climate change may increase the amount of development in the coastal zone. In order to cope with rising sea-levels, heavy engineering is often used to increase the elevation of land through infilling (often with materials dredged from offshore), or to build sea defences to protect against coastal erosion. Both of these methods incurs considerable financial costs, and often provides an only temporary solution (Spalding *et al*., 2010). The rate of sea-level rise associated with climate change is expected to increase in the coming decades, which will further exacerbate these challenges.

Aquaculture is another land-use activity, which often involves the creation of extensive pond systems in intertidal areas and largely associated with the worldwide losses of mangroves (Wolanski *et al*., 2000). Aquaculture is responsible for more than half (52%) of global mangrove losses (Valiela *et al*., 2001), with shrimp farming rising in the last decade to more than 50% of global shrimp production (Asche *et al*. 2011). According to FAO (2007), Indonesia, Malaysia and South America have recorded highest conversion since 1980's partly due to conversion for shrimp ponds (FAO, 2007).

Shrimp aquaculture has other serious environmental costs; High yields in intensive shrimp farming can only be maintained through the heavy application of antibiotics, pesticides, and fungicides (Wolanski *et al*., 2000). This results in contamination of surface and ground waters in the form of excess lime, organic wastes, pesticides, chemicals, and disease microorganisms which flush into neighbouring mangroves and environments (Primavera, 1991); Shrimp ponds are not sustainable over long time and often abandoned whenever yields or profits drop (Flaherty and Karnjanakesorn 1995). Very few trees are able to re-colonize the area even 10 years after shrimp ponds are abandoned (Wolanski *et al*., 2000). Moreover, abandoned ponds cannot be restored unless extensive efforts are made to rehabilitate soils which lead to continued clearance of mangroves for new ponds (Spalding *et al*., 2010).

Agriculture is another land-use activity that is associated with mangrove loss. This has been attributed to the flat and rich organic soils of mangrove forests which make them prime locations for conversion into cash crops farms, especially rice paddies and palm oil plantations. When mangrove areas are converted for agricultural purposes they are first deforested. Then rain water is used to remove salt from the soil and together with costly embankments constructions to protect the area from seawater intrusion. When the soil salt levels are sufficiently low, the area is then ready for cultivation. However, this conversion is generally not profitable due to the high cost and low return income (Spalding *et al.*, 2010). Deforestation and alteration of natural hydrology can cause mangrove soils to dry out and become irreversibly acidic. Such soils are no longer useful for growing crops. Additionally, clearing of mangroves for agriculture can lead to a loss in soil elevation. This requires engineering interventions to prevent flooding (Spalding *et al*. 2010).

In Kenya, Satellite imagery indicated more than 20% decrease in coverage of *R. mucronata* between 1965 and 1992, but an increase of almost 35% in sand cover over the same period (Obade *et al.*, 2004). These changes were attributed to human influence as the most probable trigger of the observed changes (Obade *et al.*, 2004). In another study, remote sensing and GIS techniques were successfully employed in a carbon distribution and mangrove vegetation change detection studies in Tudor and Mwache creeks, Mombasa, Kenya revealing loss in mangrove forest coverage by over 80% between 1992 and 2009 with loses closely being linked to land use changes within the study area (Olagoke, 2012).

2.4 Mangrove threats

Mangrove forests are among the most threatened habitats on earth (Valiela *et al.*, 2001), with human activities being primary driver of mangrove degradation, destruction and loss (Spalding *et al*. 2010). Compounded with effect from natural disturbances, global forest destruction and degradation could likely worsen, unless drastic measures are taken to protect these fragile ecosystems (Kathiresan and Bingham 2001).

Most of mangrove ecosystem areas in the world have received very little attention with respect to management resulting in large areas of mangroves being depleted and degraded at an alarming rate (Kairo, 2001 and Hamilton, 1984). Total mangroves cover by area in the year 2000, was estimated at 137,760 km2 in 118 countries and tropical and subtropical regions of the world (Giri, 2011). Despite 75% of world's mangroves being found in just 15 countries, only 6.9% are protected under the existing protected areas network (IUCN, 2010).

With approximately 90% of all mangroves growing in developing countries (Duke *et al*., 2007), they are bound to be depleted at higher rates. In 2007, the Food and Agriculture Organization of the United Nations (FAO) did a comprehensive assessment on the changes in global mangrove area over the last 25 years. Its report showed extensive losses in all regions since 1980. About a fifth of all mangroves have been lost, while many that remain are in a degraded condition. The global rate of decline over this time frame is reported to be 1.04 % during the 1980s, 0.72% during the 1990s, and 0.66 % from 2000 to 2005 FAO (2007).

Anthropogenic disturbance of any forest ecosystem majorly occurs either through deforestation or conversion to other land uses leading to a large net carbon release into the atmosphere (Hamilton, 1984). This process happens in a matter of hours, in case of fire; over a number of years, due to decomposition; or over decades, where wood products enter domestic or urban systems FAO (2007). Studies on the estimation of carbon change in terrestrial pools enable determination of net carbon emission.

Tropical forests naturally contain more aboveground carbon per unit area than any other land cover type (Gibbs, *et al*., 2007), this makes them important to consider within effort to mitigate climate change. However, Continuous and consistent accounting of all carbon inflows and outflows is complex. The current estimates shows that 'land use, land use change and forestry' (LULUCF) accounts for 15-20% of total greenhouse gas emissions based on this type of stock accounting, with large net emissions occurring at the tropics (IPCC, 2006). Studies have shown that carbon released in 5 years is balanced by the uptake of carbon for up to 30 years (Houghton, 2005). For about two decades ago Pond aquaculture is recognized worldwide as the major source of large scale clearing of mangrove forests and other coastal wetlands (Valiela *et al*., 2001).

In Kenya, salt farming has been mainly responsible for the destruction of large tracts of mangroves with 50% of the total salt farm located within former mangrove areas. Aquaculture projects executed under the FAO/ UNDP funding has also been responsible for the destruction of up to 60 ha of mangrove forests in the Ngomeni creek (Rasowe, 1992). Recently, extraction of wood for use as industrial wood-fuel (Kairo 1995) was the major cause of mangrove degradation in Gazi Bay. Unfortunately, the extent of degradation by these various causes in the Kenyan situation has not been quantified but is only evident from expansive bare areas (Kairo, 1992 and Dahdouh-Guebas *et al*., 2000a). Recent estimates indicate that mangrove loss in Kenya since 1992 is about 20% (Kirui *et al*. 2011) and the highest degradation rates have been recorded in Tudor (86%) and Mwache Creek mangroves (46%) over the same period (Olagoke O, 2012).

Mombasa peri-urban Creeks (Mwache, Tudor and Mtwapa) tend to suffer the proximity effects associated to urban settlements and faced with both direct and indirect human disturbances (Olagoke, 2012). Domestic wood-fuel and industrial energy, clearance for house building, sewage dumping and other forms of pollution, are the highlighted significant threats linked to these ecosystems (Duke *et al*., 2007). Proximity to settlement of mangrove areas and its accessibility are factors that put this resource at higher risk of degradation; this is well documented by decreased in Rhizophora mucronata by 20% between 1962 and 1992, with sand cover increasing by 35% in Gazi (Obade *et al*., 2004).

2.5 Conceptual Framework

The interplay of climate change impacts, forest and habitat disturbances within the mangrove forest ecosystem results to a reaction and ecological feedbacks. Altered precipitation, ENSO events, flooding, sedimentation and sea level rise are among the climate induced variables, while exploitation and conversion of mangrove forest through deforestation and conversion to other land uses constitute the human aspect that is directly affecting the mangrove forest ecosystem. With regard to land use, crop farming, aquaculture and urban infrastructure development are the most noted forest cover replacements. This results in a range of ecological feedback from the ecosystem including, reduction in aerial extent of the forest, mangrove dieback, health decline, interruption of species zonation and general disruption of the ecosystem, hence inability to perform its functions of providing ecosystem services and importantly sequestration of carbon leading to increase in GHG concentration.

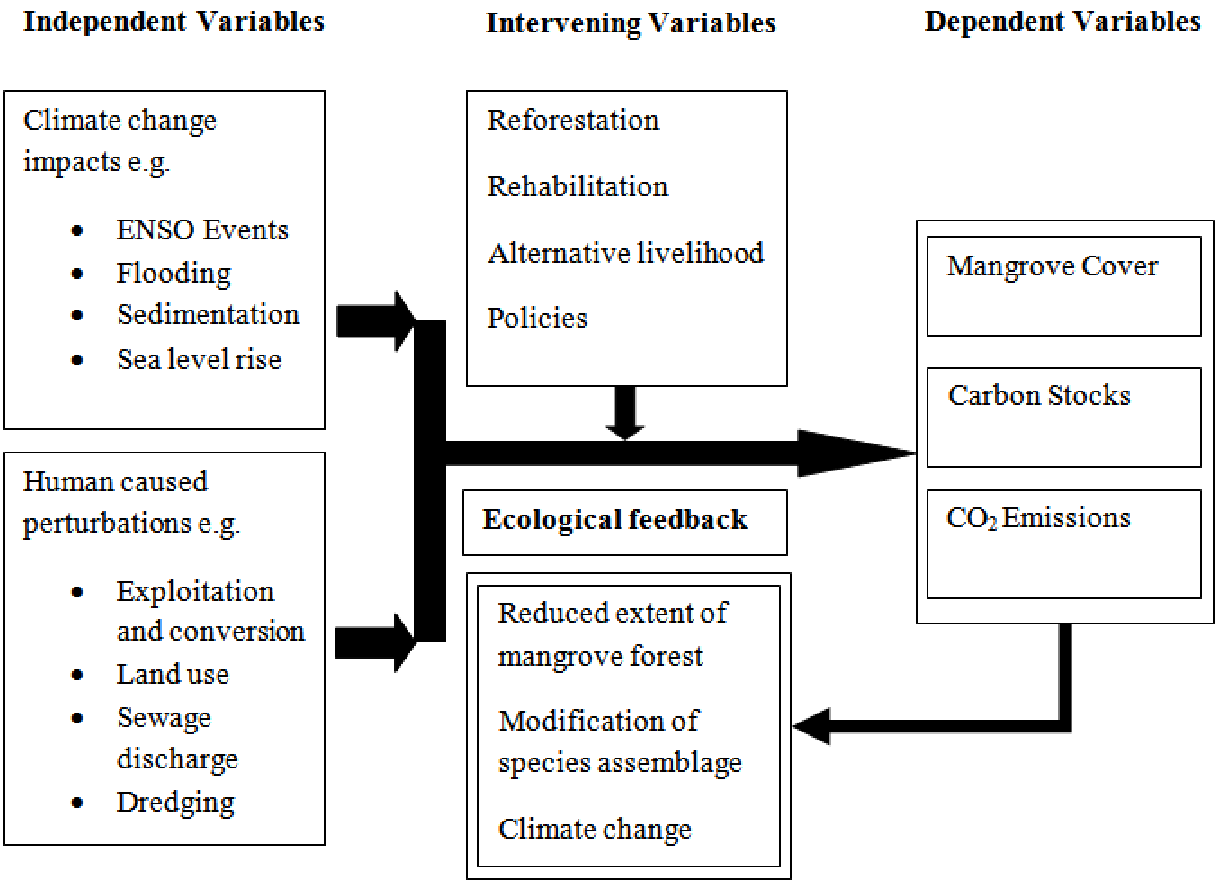

Figure 2.3: Conceptual Framework of the study

CHAPTER THREE

RESEARCH METHODOLOGY

3.0 Introduction

Deals with the research methodology: It gives the geo-description of the study are and climate of Mtwapa area; Research design and sampling methods of biomass and soil sediments by Kauffman and Donato are described in this chapter. Ground Truthing survey methods is also stated in this section. Important laboratory procedures (Loss on ignition and Bulk density) for soil carbon determination are also described here. Finally data analysis and presentation of Biomass and carbon estimation, cover change analysis and emission estimates are described.

3.1 Study area

Mtwapa Creek is located 15km from Mombasa City and lies between the Northern Coast of Mombasa county and southern coast of Kilifi county in Kenya about 4° 00'S and 39°45’E (Figure 3.1).

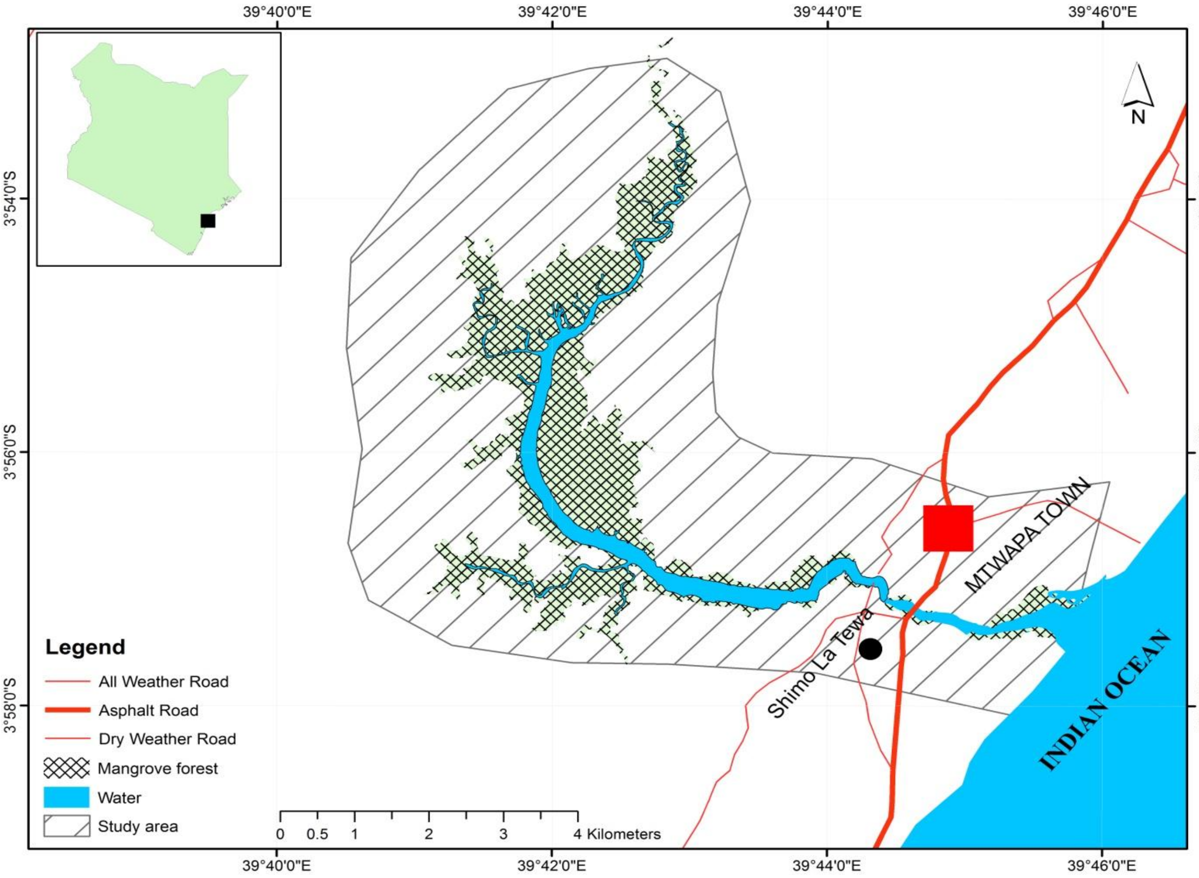

Figure 3.1: Map of the study area showing the creek and mangrove distribution

The creek is adjacent to Mtwapa Town and is approximately 13.5km in length and opens to the Indian Ocean through a long narrow channel (Okello *et al*., 2013). It is a tropical estuarine surrounded by vast mangrove swamps, while its offshore area is shielded by extensive share parallel coral reefs. It receives runoff from three seasonal rivers, (KwaNdovu, Kashani and Kidutani).

Mtwapa creek is covered by a multi-species mangrove stand including the species *Rhizophora mucronata, Ceriops tagal Avicennia marina, Sonneratia alba and Xylocarpus granatum.* It consists of three forest patches (Gung'ombe, Kitumbo and Kidongo: named after adjacent large villages) which are situated further landward from the mouth (Okello *et al*., 2013).

3.1.1 Climate

The creek lies in the coastal zone and its climate is determined by factors among them, the inter-tropical convergence zone (ITCZ) and the monsoon winds coming in two seasons. Between December and March the area experiences The Northern Easterly Monsoon (NEM), while from May to October, Southern Easterly Monsoon (SEM) are evident. Mean annual Rainfall within the study area is about 1038mm, with peaks in June and July; and the mean annual temperatures range between 23.9^0C and 28.5^0C, for the two seasons respectively (Mohamed *et al.*, 2008). The highest temperatures of 28-29˚C occur following the Northeast Monsoon in the months of December to April.

3.2 Research Design

This study adopted a cross-sectional survey to determine the organic carbon stocks and in establishing training sites for analysis of spatio-temporal cover change using Landsat image of 1990 and SPOT images of 2000 and 2009 of Mtwapa creek.

3.3 Sampling Design

3.3.1 Biomass and soil sampling

For carbon estimation, this study sampled above ground, below ground and sediments pools. Dead wood and forest litter are minor components of the ecosystem and were not included in the sample. They make up a minor part of the organic carbon pools and can be excluded from the measurements without compromising the accuracy of the sample (Kauffman and Donato, 2012).

Stratified systematic sampling technique was used to establish representative transects at 90^0 with respect to the navigational direction along the shoreline of the Creek with the aid of a 2009 unsupervised classified map of mangrove species done prior to the field survey. The plots measuring 10 x 10m were set up systematically along each transect with 100m interval between successive plots. The total minimum numbers of plots for the study area were 28, calculated from Pearson's model. Plots per transect varied according to the length of cross-section of the creek. Soil and biomass sampling were done concurrently in the same plots during a low tide. Biomass data was collected by recording species and measuring two variables; DBH or D_{130} (cm) and tree height (m) of all trees within the sampling plot with $d_{130} \geq 2.5$ cm. DBH was measured using the Forest Calipers, while tree height was measured by Suunto Hypsometer. Geographic positions of the sampling plots were taken using a Garmin 26cs GPS.

Sampling was done to a depth of 100cm as soil organic carbon has been established to be found within this depth (Pearson *et al*., 2007). Coring was done at the center of every plot using a 120cm corer. After undisturbed soil core had been extracted, four samples were collected at depths of between 10–15cm, 15-30cm, 30-50cm and 50-100cm. Measuring was done by a ruler and 5 cm sub samples of mid-points were collected using a spatula in the four depths as follows respectively, 5-10, 20-25, 37.5-42.5 and 72.5-77.5. After collection every sample was placed in a labeled container, stored at 4^0c and transported for analysis in the laboratory (Kauffman and Donato, 2012).

3.3.2 LULC Ground Truthing Survey

As field survey is essential in identifying features of interest prior to image processing in Remote Sensing (Green *et al*., 2000), ISO-CLUSTER unsupervised classification was done on the 2009 image prior to field work to retrieve different spectral classes for creating specific classification training sites. Field survey was carried out across mangrove species aggregation area and across the land cover types within the study area up to a distance of 2km from the mangrove zone. Particular land-uses and land-cover were located and marked using Garmin GPS in UTM coordinates system with the help of the unsupervised map. A Google earth image was used to ground truth areas that could not be accessed during the field survey by comparing it with unsupervised classification image of 2009 by selecting specific regions observed to have similar specific land cover or land use for verification.

3.4 Lab Analysis

3.4.1 Soil Bulk Density

Bulk density was calculated from mass of oven-dried soil per unit volume. The soil samples from the field were placed in the pre-weighed aluminum foils and placed in the oven at a constant temperature of 60˚C for about 48 hours after which they were weighed. The bulk density was calculated using equation i below:-

$$\boldsymbol{Bulk\ density} = \frac{\boldsymbol{oven\ dried\ sample\ mass\ (g)}}{\boldsymbol{sample\ volume(cm3)}} \text{.......................................Equation } \mathbf{i}$$

3.4.2 Loss on Ignition

Loss on ignition (LOI) was used to calculate Soil organic matter (SOM). This is a semi-quantitative technique where all organic matter is removed via ignition in the furnace. After drying the samples for 48 hours at 60^0c, they were grinded using pestle and mortar into fine soil powder and passed through a 2mm sieve. After sieving, the samples were placed in duplicates into a pre-weighed aluminum foils (5g). This was then combusted at 450˚C for 8 hours then cooled in a desiccator and weighed. Soil organic matter (SOM) content was determined by equation (ii) below:-

$$\boldsymbol{SOM} = \frac{\boldsymbol{initial\ mass(g) - final\ mass(g)}}{\boldsymbol{initial\ mass(g)}}\ x\ 100 \text{......................................Equation } \mathbf{ii}$$

3.5 Data Analysis and presentation

Analysis of field data was done using Microsoft EXCEL and STATISTICA Program version 8.0. Biomass and carbon stocks were analyzed followed by analysis of remotely sensed Landsat TM image of January 1990 and SPOT images of May 2000 and January 2009 to determine mangrove cover and LULC changes over time in the study area. Descriptive statistics were done on species composition and structure (Height and DBH). Biomass, Bulk density and Organic carbon stocks data were subjected to Shapiro-Wilk's test to check for Normality. All variables were found to be non-parametric, and Kruskal-Wallis test was employed to test for their variation between sites. Mangrove forest cover was regressed against land use changes to determine the existing temporal relationship.

3.5.1 Biomass and carbon Estimation

General allometric equations derived from similar species and within similar geographic regions were used to estimate above and below ground biomass of Mtwapa creek. These were sourced from allometric equations for mangrove biomass in Komiyama *et al.,* (2005 and 2008) and Kauffman and Donato, (2012). Wood density obtained from the empirical data collected from field survey in Mozambique were applied, as it falls under the same East African WIO region (Bosire *et al*., 2012). Table 3.1 shows the allometric equations and wood densities used for biomass estimation.

Table 3.1: Allometric equations and wood densities used in estimating Biomass of mangrove forest in Mtwapa Creek.

Species	**Species specific wood density(gcm^{-3})**	**Allometric equation (AGB)**	**Allometric equation (BGB)**
Avicennia marina	$\rho = 0.94$	$0.199\rho 0.899DBH^{2.22}$	$0.251\rho DBH^{2.46}$
Ceriops tagal	$\rho = 1.13$	$0.199\rho 0.899DBH^{2.22}$	$0.251\rho DBH^{2.46}$
Rhizophora mucronata	$\rho = 1.10$	$0.199\rho 0.899DBH^{2.22}$	$0.251\rho DBH^{2.46}$
Sonneratia alba	$\rho = 0.80$	$0.199\rho 0.899DBH^{2.22}$	$0.251\rho DBH^{2.46}$
Xylocarpus granatum	$\rho = 0.70$	$0.199\rho 0.899DBH^{2.22}$	$0.251\rho DBH^{2.46}$

Carbon content of every tree was calculated by summing the carbon content of AGB and BGB of the same tree. Carbon content of AGB was calculated by multiplying biomass by its carbon concentration of 47.1% for *Sonneratia alba*, and an average of 46.4% for all other species according to Kauffman *et al*., (2010). Carbon mass of roots was calculated by multiplying BGB by root carbon concentration of 39% following estimation by Jaramillo *et al*., (2003).

Soil organic carbon was calculated based on the laboratory bulk density and LOI results. The following equation was used to get soil organic carbon (SOC):-

SOC=Bulk Density x Sample Depth x %C……………………..……………….………Equation **iii**

3.5.2 Image Analysis

Image processing and classification was done using ERDAS IMAGINE 9.1 image processing software, while final maps were prepared using Arc MAP in ArcGIS 10.1. Landsat TM image of January, 1990, SPOT images of May, 2000 and January, 2009 and a 1992 mangrove vector map were acquired from a mapping project in (KMFRI) for this study. Figure 3.2 shows the raw composite bands 432 images used for cover change analysis.

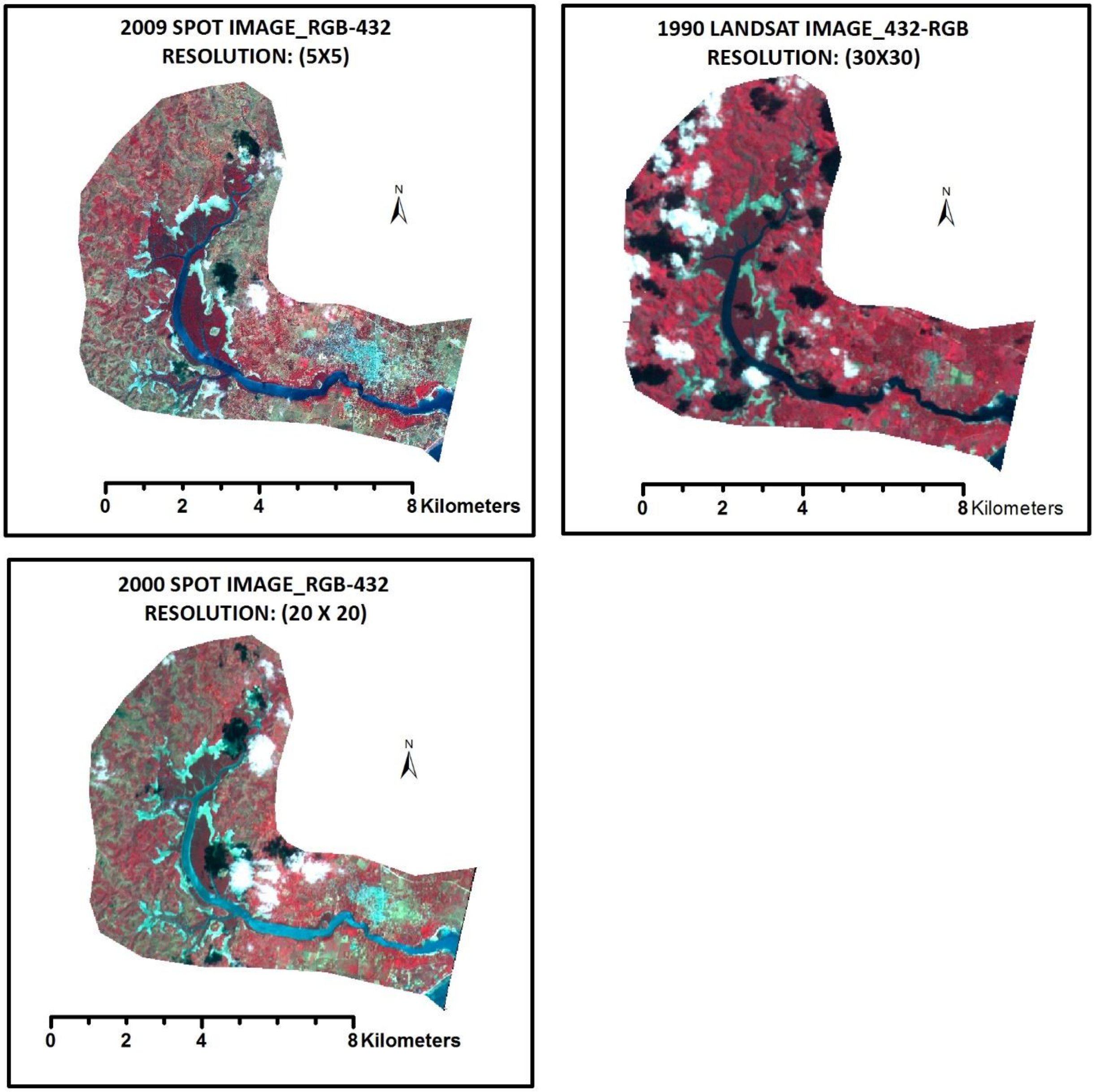

Figure 3.2: Composite RGB of Landsat TM and SPOT images used for cover change detection

All the images were registered to WGS 84 UTM Zone 37S projection. Nearest Neighbor re-sampling method was applied for geometric correction for all the images as per (Reddy and Roy, 2008; Ardli and Wolff, 2009). Image registration was done to 2009 image as the base image, followed by the three other images of 1990 and 2000. Obscurity was removed using atmospheric correction on images to remove effects of the different atmospheric conditions on the reflectance for the three images taken at different temporal periods (Song *et al*., 2001).

Training sites were digitized to create polygons representing the identified classes in the three images and saved as the training file. Maximum Likelihood classification method was performed on the three images using the training file leading to generation of 6 Land use Land cover classes. Classification accuracy was performed using 180 randomly generated points across the study area as shown in figure 3.3. Points from the output classified map were arranged as row and reference data (Google earth points) as columns (Table 4.3). Overall Accuracy (OA), User's Accuracy (UA), Producer's Accuracy (PA) and Kappa coefficient were calculated to measure accuracy of classification prior to post classification analysis.

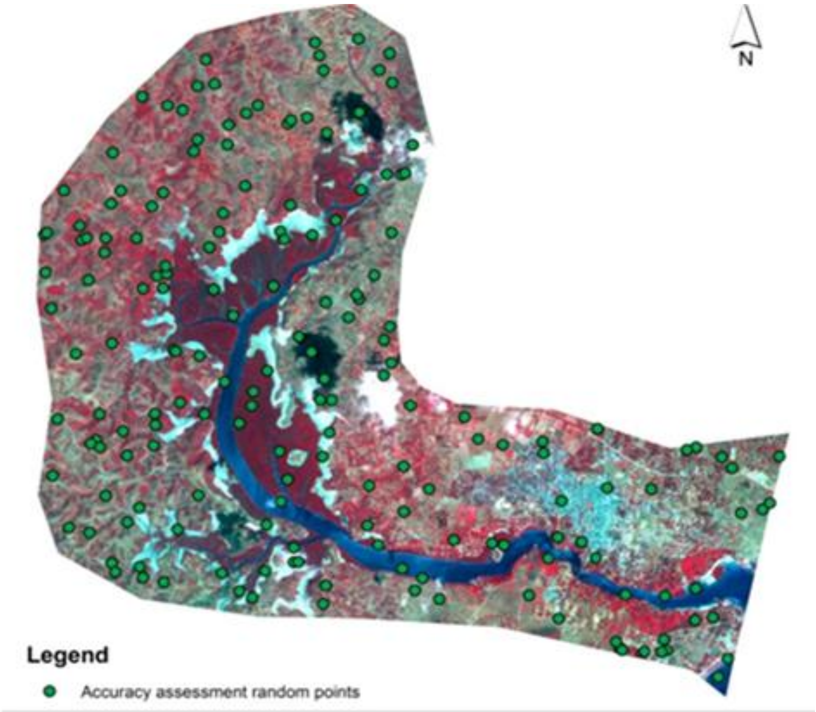

Figure 3.3: 180 randomly generated points used to test the accuracy of the classification

3.5.3 Emission estimates due to Mangrove Cover Change

Carbon estimates were done based on 2006 IPCC method involving estimation of changes in carbon stock from biomass carbon pool (Above-ground biomass and Belowground biomass for this study). *Stock-Difference Method* which estimates the difference in total biomass carbon stock at time t2 and time t1 (Equation 2.8 of IPCC, 2006) was used to get the carbon stock loss annually.

Stock-difference method (Equation 2.8 of IPCC, 2006**)** in the equation below.

$$\Delta C_B = \frac{C_{t_2} - C_{t_1}}{t_2 - t_1}$$ ……………………………………………………………………Equation **iv**

Where:

ΔC_B = Annual change in carbon stocks in biomass
C_{t2} = Total carbon in biomass at t_2, tonnes (Mg) C
C_{t1} = Total carbon in biomass at t_1, tonnes (Mg) C

Then a conversion factor of 3.67 was used to get the CO_2 emissions estimates annually based on live biomass carbon loss as estimated from cover change loss.

CHAPTER 4

RESULTS AND DISCUSSION

4.0 Introduction

Presents results of analysis and discussion. It is divided into two sections but based on the specific objectives: Results on carbon stocks from biomass and soil sediments and results from cover change analysis of remotely sensed images of 1990, 2000 and 2009 on mangrove changes and land cover changes. It also presents the regression of identified and land use patterns and mangrove cover change over the study time. The last part of the chapter looks at the carbon stocks in relation to mangrove cover changes and uses this to estimate CO_2 emissions based on 2006 revised IPCC Guidelines for National Greenhouse Gas Inventories' Stock-difference method.

4.1 RESULTS

4.1.1 Vegetation Structure and Composition

Species occurrence in Mtwapa creek was found to be dominated by *Rhizophora mucronata* with stand density of 3305±315 stems/Ha representing 86% of the species occurring in all the study sites. It was followed by *Ceriops tagal* with 350±40 stems/Ha representing 8% of the species, then *Avicennia marina* had 164±25 stems/Ha representing 4% and *Xylocarpus granatum* had a minimum but not least stand density of 130±15 stems/Ha equivalent to 2% of the species in the study area (Figure 4.1). *Sonneratia alba* was noted to occur very minimally at the edge of the forest. The mean stem density for the entire mangrove forest was estimated to be 3776±275 stems/Ha (Table 4.1).

Avicenia marina and *Xylocarpus granatum* were evidently noted to occur at the transect start, along the creek shores, with *Avicenia marina* also occurring at the edges of the forest near mudflats. *Rhizophora mucronata* dominated most parts of the mid-forest closely followed by *Ceriops tagal*, which occurred just after *Rhizophora mucronata* near the edge of the forest before the mudflats together with *Avicenia marina*. There was however no variation in the occurrence of all the four species between the sites Kruskal Wallis ($p>0.05$).

The mean DBH of the mangroves in the study area was 5.09 ±0.43 cm. The largest DBH was recorded in Mabirikani 6.08±0.59cm, while the lowest was recorded in Kidongo 4.45±0.13cm. There was however significant statistical variation in DBH between study sites within the study area, ($p<0.005$).

Table 4.1: Stand table of mangrove forest of Mtwapa Creek.

Site	Species	Tree density (stems/ha) ± S.e	Mean tree Height(m) ±S.e	Basal area (m^2/ha) ±S.e	Relative values (%)			I.V (%)	C.I
					Dom	Den	Freq		
Majaoni	*R. mucronata*	4375±785^a	3.2±0.06^a	33.3(±6.2)^b	92.8	88.8	89.2	270.8	56
	C. tagal	350±164^b	2.6±0.11^a	1.8(±0.7)^a	5.0	7.1	6.9	19.0	
	A. marina	200±87^b	6.6±0.34^b	0.8(±0.5)^a	2.2	4.1	3.9	10.2	
	Mean		**3.2±1.7^1**	**11.9(±5.1)^1**					
	Total	**4925±22^1**			**100**	**100**	**100**	**300**	
Mtepeni	*R. mucronata*	3789±413^a	3.7±0.06^a	9.8(±3.2)^b	93	96.5	96.7	60.2	15.9
	C. tagal	72±83^b	3.4±0.07^a	0.1(±0.05)^a	1	1.8	1.7	5.3	
	A. marina	66±55^b	7.4±0.61^b	0.6(±0.06)^b	6	1.7	1.6	102.1	
	Mean		**3.87±0.12^1**	**3.5(±2.1)^2**					
	Total	**3927±18^1**			**100**	**100**	**100**	**300**	
Mabirikani	*R. mucronata*	2917±782^a	3.9±0.07^a	8.7(±3.0)^b	82.3	82.2	86.1	250.6	14.9
	C. tagal	367±166^b	3.3±0.17^a	1.0(±0.5)^a	9.5	10.3	6.2	26.0	
	A. marina	166±120^b	8.0±0.44^b	0.57(±0.3)^a	5.4	4.7	4.8	14.9	
	X. granatum	100±100^b	4.3±0.42^a	0.3(±0.03)^a	2.8	2.8	2.9	8.5	
	Mean		**4.06±0.29^1**	**2.6(±1.2)^2**					
	Total	**3550±45^1**			**100**	**100**	**100**	**300**	
Kidutani	*R. mucronata*	2480±768^a	4.3±0.08^a	6.2(±2.6)^a	63.3	72.5	72.5	208.3	13.4
	C. tagal	780±682^b	2.6 ±0.11^a	3.2(±1.2)^a	32.6	22.8	22.8	78.2	
	X. granatum	160±160^b	5.4±0.31^b	0.4(±0.03)^a	4.1	4.7	4.7	13.5	
	Mean		**3.97 ±0.42^1**	**3.3(±2.4)^2**					
	Total	**3420±44^1**			**100**	**100**	**100**	**300**	
Mibuyuni	*R. mucronata*	2383±618^b	3.4±0.09^a	5.9(±2.1)^a	81.9	91.7	90.4	264	5.6
	A. marina	217±151	4.5±0.67^a	1.3(0.07)^b	18.1	8.3	9.6	36	
	Mean		**3.51 ±0.41^1**	**3.1(±2.1)^2**					
	Total	**2600±44^1**			**100**	**100**	**100**	**300**	
Kidongo	*R. mucronata*	3883±406^a	3.3±0.15^a	6.6(±2.7)^a	77.6	91.7	91.5	260.8	11.8
	C. tagal	183±183^b	2.1 ± 0.04^a	0.4(±0.4)^a	4.7	4.3	4.3	13.3	
	A. marina	170±162^b	4.4± 0.04^a	1.5(±0.8)^a	17.7	4.0	4.2	25.9	
	Mean		**3.32 ±0.21^1**	**2.8(±1.6)^2**					
	Total	**4236±44^1**			**100**	**100**	**100**	**300**	

Dom = dominance, Den = density and Freq = frequency

Same alphabet superscript means no significant difference ($p>0.05$) in a column within a site; whereas same numeric in superscript means no significant difference ($p>0.05$) in totals or means in columns for all the sites.

The mean height of mangrove trees for the study area was found to be 3.67±0.27m, with maximum height of 4.06±0.29m recorded in Mabirikani and minimum of 3.32±0.17m in Majaoni and 3.32±0.21m in Kidongo as shown in Table 4.1. There was no significant statistical variation in the mean heights between study sites, ($p>0.05$).

4.1.2 Biomass

The mean biomass for the study area was 112.48±19.14Mg ha^{-1}.This was achieved by summing Above Ground Biomass (AGB) and Below Ground Biomass (BGB) means. AGB contributed a mean of 75.56±13.30Mg ha^{-1} while BGB had 36.92±5.90Mg ha^{-1}. The highest biomass was recorded at Mabirikani 140.26±13.94Mg ha^{-1}, while the lowest biomass was recorded in Kidongo 86.09±5.42Mg ha^{-1}. The other sites registered biomass as given in figure 4.1 below.

Biomass of sites did not show statistical significant variation between the study sites Kruskal Wallis ($H=5.94$; $p>0.05$).However, biomass values in Kidongo and Mibuyuni were statistically significantly lower than values for Mabirikani, Mtepeni and Kidutani ($p<0.05$).

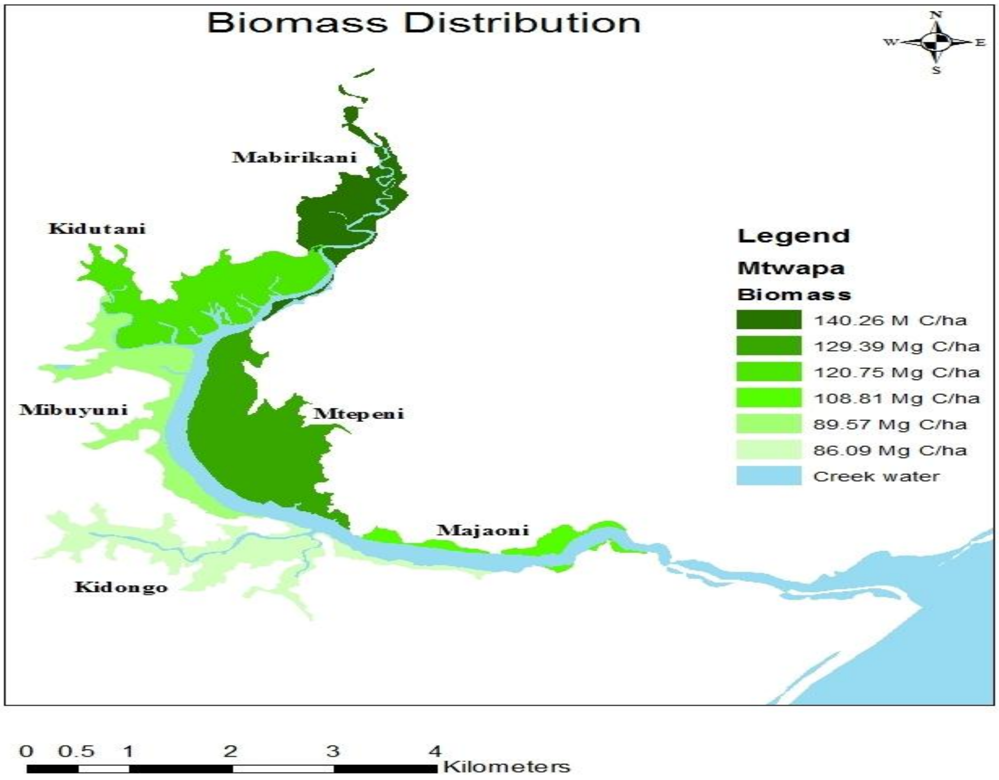

Figure 4.1: Map showing Biomass distribution of mangrove forest in Mtwapa Creek.

4.1.3 Bulk Density

The mean soil bulk density of the study area was 0.71±0.04g/cm^3. Lowest bulk density was recorded in Mibuyuni (0-15) cm interval (0.59±0.03 g/cm^3), while the highest value (0.88±0.06 g/cm^3) was recorded at Kidutani (50-100) cm Interval. The site means indicated significantly high bulk density in Kidutani of 0.85±0.07gcm$^-$3 ($p<0.05$) and significantly Low in Mtepeni with 0.62gcm^{-3} ($p<0.05$).

There was a significant statistical difference in mean bulk density between sites, (H=12.88; $p<0.05$). However the bulk density at same depth between sites showed no significant statistical variation ($p>0.05$) except in the (50-100) cm depth (H=12.68; $p<0.05$).

Bulk density increased down the depth interval in all the sites except for Kidongo which declined in the second depth interval (15-30). Only Majaoni bulk density declined in the third depth interval (30-50) and in the final depth decline was observed in Mibuyuni and Mabirikani. This is illustrated in the multiple line graph of depth interval (cm) against bulk density (g cm^{-3}) shown below (Figure 4.2).

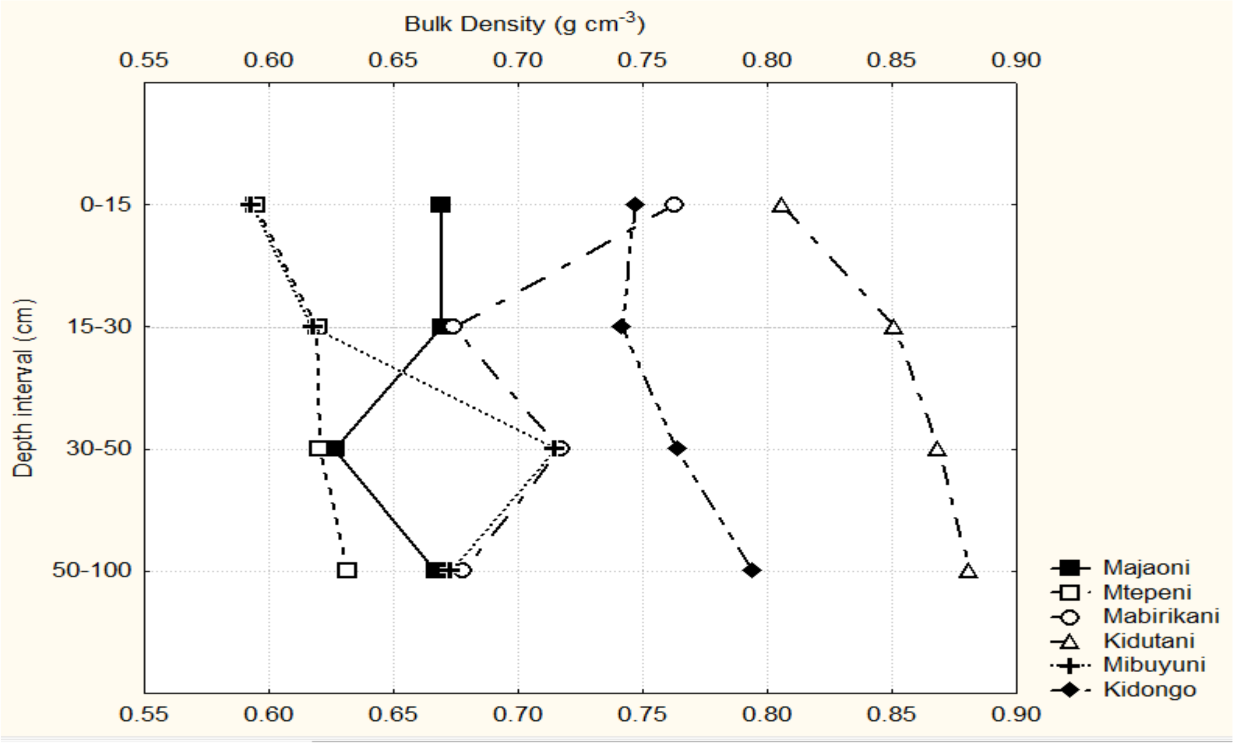

Figure 4.2: Mean soil bulk density (g cm^{-3}) of Mtwapa Creek forest per depth.

4.1.4 Biomass Carbon

Biomass conversion factors of 47.1 for *Sonneratia alba*, and 46.4 for the other species in AG biomass and 39 for BG biomass were used to convert biomass to carbon equivalents (Kauffman *et al.,* 2010 and Jaramillo *et al* 2003). The average biomass carbon of the study area was 49.46±8.45Mg C ha^{-1}. The highest biomass carbon was at Mabirikani 61.79±6.17Mg C ha^{-1} while the lowest figure was recorded in Kidongo 37.75±2.38Mg C ha^{-1}. The above ground carbon was higher than below ground carbon, with means of 35.06±6.17Mg C ha^{-1} and 14.40±2.30Mg C ha^{-1} respectively. Figure 4.3 shows the distribution of biomass carbon within the study area.

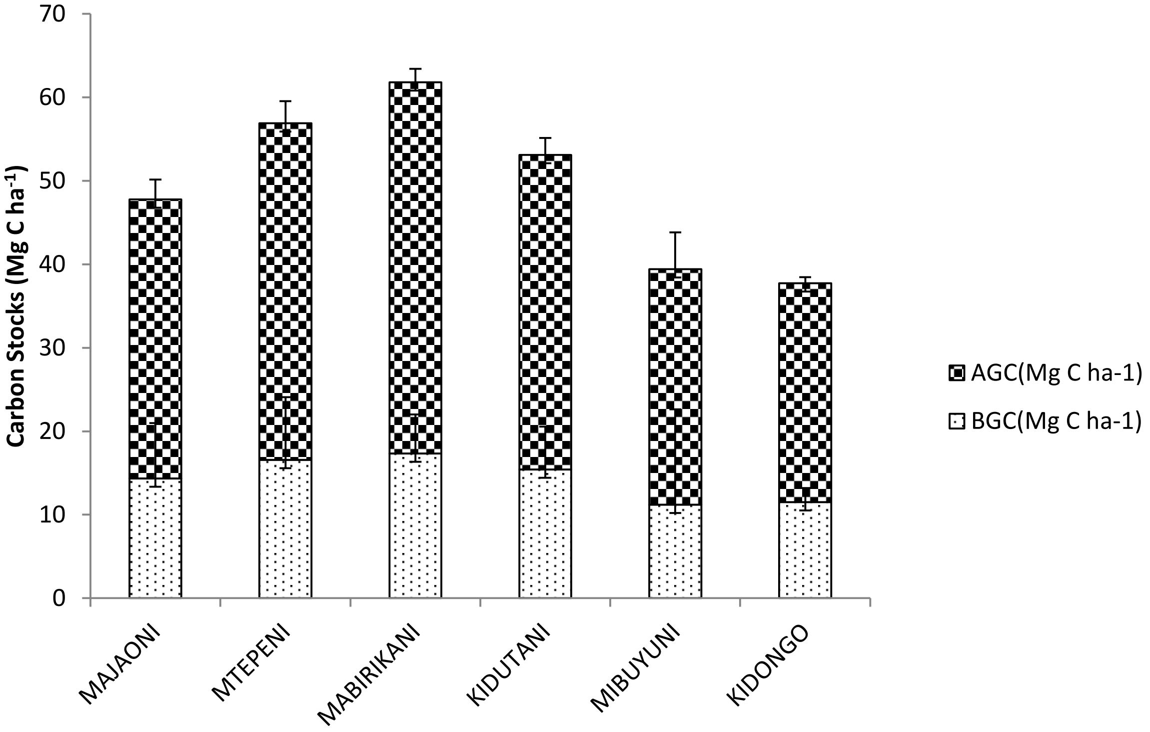

Figure 4.3: Graph of Biomass carbon obtained from mangrove forest of Mtwapa Creek.

4.1.5 Sedimentary Carbon

There were two calorimetric equations available from studies from adjacent creeks for the conversion of SOM to SOC; Mugi's equation (***Y=0.4328X - 0.3769***; R^2=0.7991) and Olagoke's equation (***Y=0.4563X - 1.1863***; R^2=0.9551). Both equations were used and t-test performed to check on their variability. There was a significant variation on the results from the two equations (t-test, p=0.00887), Olagoke's equation was used due to high co-efficient of determination (R^2= 0.95) and because bulk density in Mtwapa creek lies within the figures of 0.65g/cm^3 to 1.10g/cm^3 obtained in Tudor creek, besides topographical similarities.

The mean SOC of the study area was found to be 196.09±19.31Mg C ha^{-1}, with highest value in Mibuyuni of 214.89±20.95Mg C ha^{-1} and lowest figure in Mabirikani 178.33±16.05Mg C ha^{-1} as illustrated in Table 4.2. SOC increased along the depth gradient. The (0-15) cm interval recorded the lowest mean of 29.79±1.77 Mg C ha^{-1}, (15-30) cm interval had 31.33±1.50 Mg C ha^{-1}, (30-50) cm interval recorded 44.30±2.20 Mg C ha^{-1} and the last depth (50-100) cm interval registered a highest SOC of 90.67±4.80 Mg C ha^{-1} (Figure 4.4). No statistical significant difference existed between the study sites (p>0.05). However, Mibuyuni had a significantly higher SOC as compared to Mabirikani and Kidongo (P<0.05).

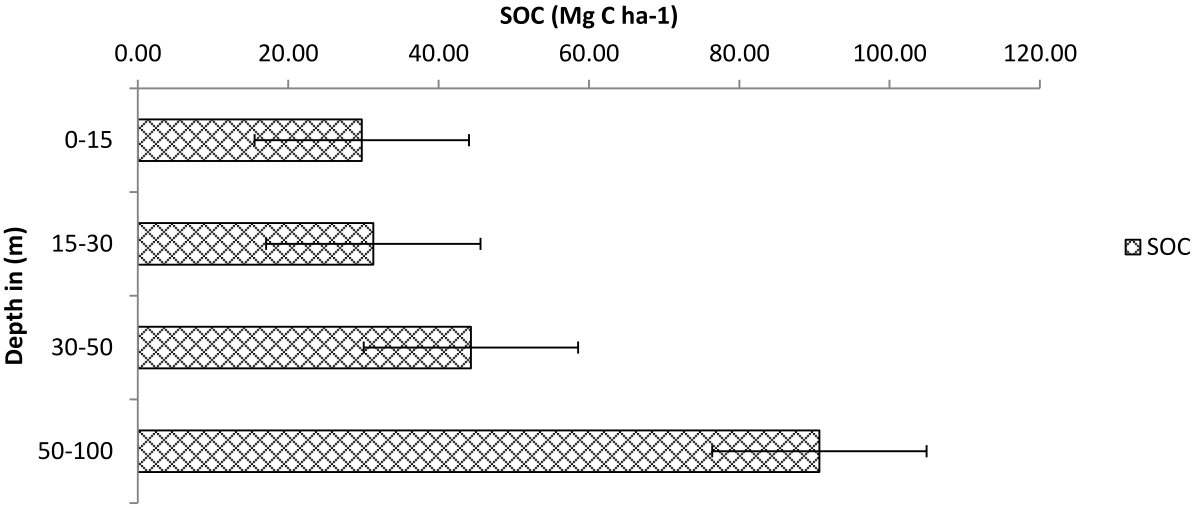

Figure 4.4: Soil Organic Carbon along the sampled soil profile in mangrove forest of Mtwapa creek.

4.1.6 Total Ecosystem Carbon

The total carbon stock for the entire ecosystem was obtained by adding biomass carbon and organic sedimentary carbon. The carbon stock for the entire Mtwapa ecosystem was estimated to be 245.54±20.95Mg C ha^{-1}. The highest carbon stock was recorded in Mtepeni with a mean of 260.04±27.07Mg C ha^{-1} while Kidongo recorded the least carbon stock of 218.07±13.75Mg C ha^{-1} as shown in Table 4.2.

Table 4.2: Biomass carbon of mangrove forest of Mtwapa Creek in (Mg C ha^{-1})

LOCATION	**AGC**	**BGC**	**SOC**	**Total**
Majaoni	33.42±6.63	14.35±2.38	204.82±17.44	252.58±21.37
Mtepeni	40.33±7.53	16.56±2.66	203.14±19.41	260.04±27.07
Mabirikani	44.46±4.67	17.33±1.61	178.33±16.05	240.12±20.37
Kidutani	37.66±5.12	15.44±2.05	195.03±30.39	248.13±32.64
Mibuyuni	28.21±11.40	11.22±4.40	214.89±20.95	254.32.±10.52
Kidongo	26.25±1.68	11.50±0.72	180.3±11.66	218.07±13.75
Mean	35.06±6.17	14.40±2.30	196.09±19.31	254.543±21.00

Sedimentary pool contributed 196.09±0.9 Mg C ha^{-1} representing 79.86% of the total carbon stock, this was followed by AG pool with 35.06±6.17 Mg C ha^{-1} representing 14.28% of the total carbon stock and BG pool was the least with, 14.40±2.30 Mg C ha^{-1} representing 5.86% of the total carbon stock (Table 4.2). There was no statistically significant variation in total ecosystem carbon within the study area, Kruskal Wallis test ($p>0.05$) however, Kidongo and Mabirikani had a significantly lower carbon content than the rest of the sites ($p<0.05$). Figure 4.5 illustrates the contribution of the three pools on the carbon stocks of Mtwapa creek.

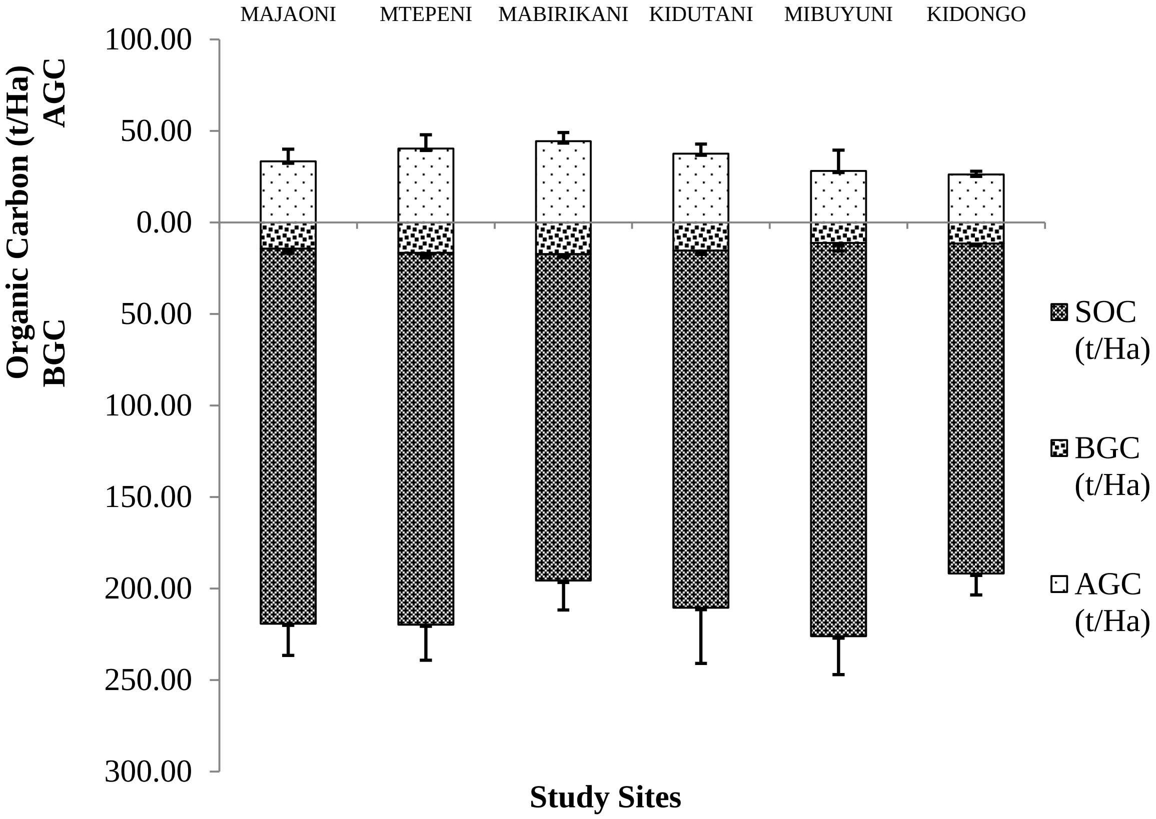

Figure 4.5: Carbon stocks from the three sampled pools in mangrove forest of Mtwapa Creek.

4.1.7 Accuracy assessment

Classification error matrix was generated based on 2009 SPOT image classes. Both the correctly and incorrectly classified pixels, based on 180 randomly generated points were obtained (Table 4.3). Most points (140) were correctly classified, obtaining an overall accuracy of 77.78%. User's accuracy and Producer's accuracy for each of the classes showed that satisfactory levels of accuracy were obtained (PA and UA > 50%). Mangrove cover had the lowest UA (33.33%), although its PA was as high as 75%.

Table 4.3: Accuracy assessment of supervised classification of 2009 SPOT image for Mtwapa Mangrove forest and surrounding land cover.

	Reference Data						**Row totals**	**Producer's accuracy**	**User's accuracy**
	Wa	**Bu**	**Sa**	**Agr**	**UF**	**MF**			
Class Data									
Wa	**8**	0	2	0	0	0	**10**	**66.67%**	**80.00%**
Bu	0	**3**	1	2	0	0	**6**	**100.00%**	**50.00%**
Sa	4	0	**23**	5	3	1	**36**	**74.19%**	**63.89%**
Agr	0	0	0	**74**	9	0	**83**	**83.15%**	**89.16%**
UF	0	0	0	7	**29**	0	**36**	**70.73%**	**80.56%**
MF	0	0	5	1	0	**3**	**9**	**75.00%**	**33.33%**
COLUMN TOTAL	**12**	**3**	**31**	**89**	**41**	**4**	**180**		

Diagonal sum =140; Overall accuracy = 77.78%. Kappa = 0.68. Wa = Water, Bu = Built up, Sa = Sand flat, Agr = Agriculture, UF = Upland forest and MF = Mangroves forest.

4.1.8 Mangrove Cover Changes

Training polygons on species cover and extent were digitized on 2009 SPOT image and a supervised classification done to determine the distribution and cover of mangrove species. This was compared with a 1992 mangrove species vector map to determine changes in species cover over time. Figure 4.6 shows the thematic map on the species classes in 1992 and classification results for 2009 in Mtwapa creek.

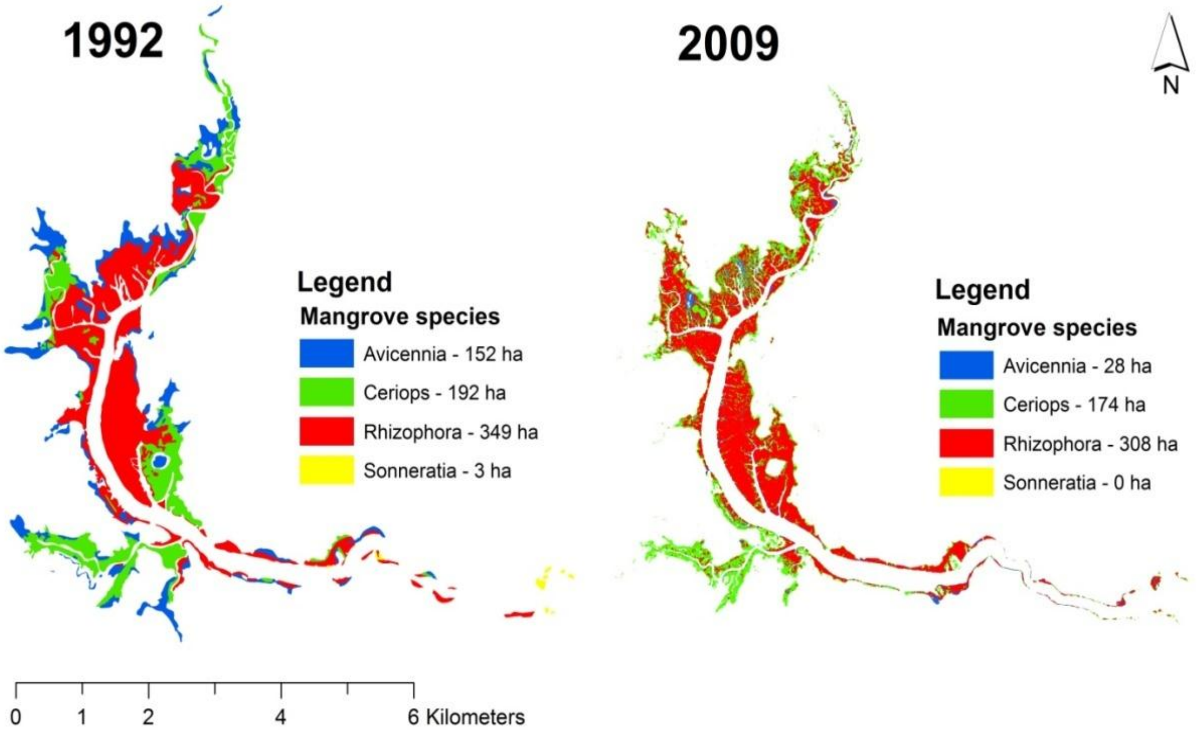

Figure 4.6: Map showing mangrove cover change between 1992 and 2009 in Mtwapa mangrove forest.

Between 1992 and 2009, mangrove species have undergone noticeable cover changes (Figure 4.6). This was accompanied by shifts of species and an example is *Ceriops tagal* zone which in some areas been replaced by *Rhizophora mucronata* in 2009 especially in Majaoni and Mabirikani (Figure 4.6). All the species recorded a reduction in extent of coverage. *Sonneratia alba* was the most affected and was not detected in 2009 imagery while previously having occupied 3ha representing 0.43% of the mangrove area. *Avicennia marina* reduced by 81.58%, this is after previously occurring abundantly at the edges of mangrove forest almost in all sites with a total area of 152 ha, which had reduced to 28ha in 2009. *Ceriops tagal* reduced in cover by 9.38%, from192ha in 1992 to 174ha in 2009. Despite dominating most parts of the forest, *Rhizophora mucronata* declined by 11.74%, from 349ha in 1992 to 308ha in 2009. For the 17 year period from 1992 to 2009 mangrove cover reduced by 26.72%. This translates to 1.5% of mangrove forest lost annually in Mtwapa creek. Table 4.4 shows the summary in (ha) and (%) of the species cover dynamics for the 17 years.

Table 4.4: Mangrove cover changes between 1992 and 2009 in mangrove forest of Mtwapa Creek.

Species	**1992 (ha)**	**2009 (ha)**	**Change**
Rhizophora mucronata	349 (50.14)	308 (60.39)	-11.74%
Ceriops tagal	192 (27.59)	174 (34.12)	-9.38%
Avicennia marina	152 (21.84)	28 (5.49)	-81.58%
Sonneratia alba	3 (0.43)	0 (0)	-100.00%
Total	696 (100)	510 (100)	-26.72%

4.1.9 Land cover changes

Training polygons for Upland forest, Agriculture, Built up areas, Mangrove forest, Sand flat and Water were digitized on the 1990 Landsat TM images, 2000 SPOT image and 2009 SPOT image before the classification procedure that output thematic maps with the land cover categories for the study area. The Classification yielded three Land cover maps of the study area which were classified into 6 broad classes. The six classes are Built-up, Agriculture, Upland forest, Mangrove forest, Sand flats and Water as shown in the classified maps (Figure 4.7) and summary of area in ha and percentage covers in Table 4.5. These maps are the basis of presented information on the primary Land cover changes that have occurred from 1990 to 2009 within the study area.

The spatial extent of the 1990 Land cover Classes indicates that Upland forest occupied the highest percentage cover of 3301. 10ha (60.29%) and was distributed across the map with the highest concentration observed towards the sea. The second highest was Agriculture (838.37ha, 15.31%) which occurred in patches and scattered around North, South, and the Western parts of the area with very small patches within the mangrove forest reserves. Mangrove forest was third highest and covered 700.00ha (12.78%) on the either sides of the creek. Sand flat occurred next to Mangroves with a cover of (324.64ha, 5.93%) mostly at the edge of the mangrove forest. This was followed by the water 279.36ha (5.10%) which covered the creek channels. Built up area covered 32.03ha (0.58%). It had the least area coverage and appeared as a cluster in the south eastern part of the map.

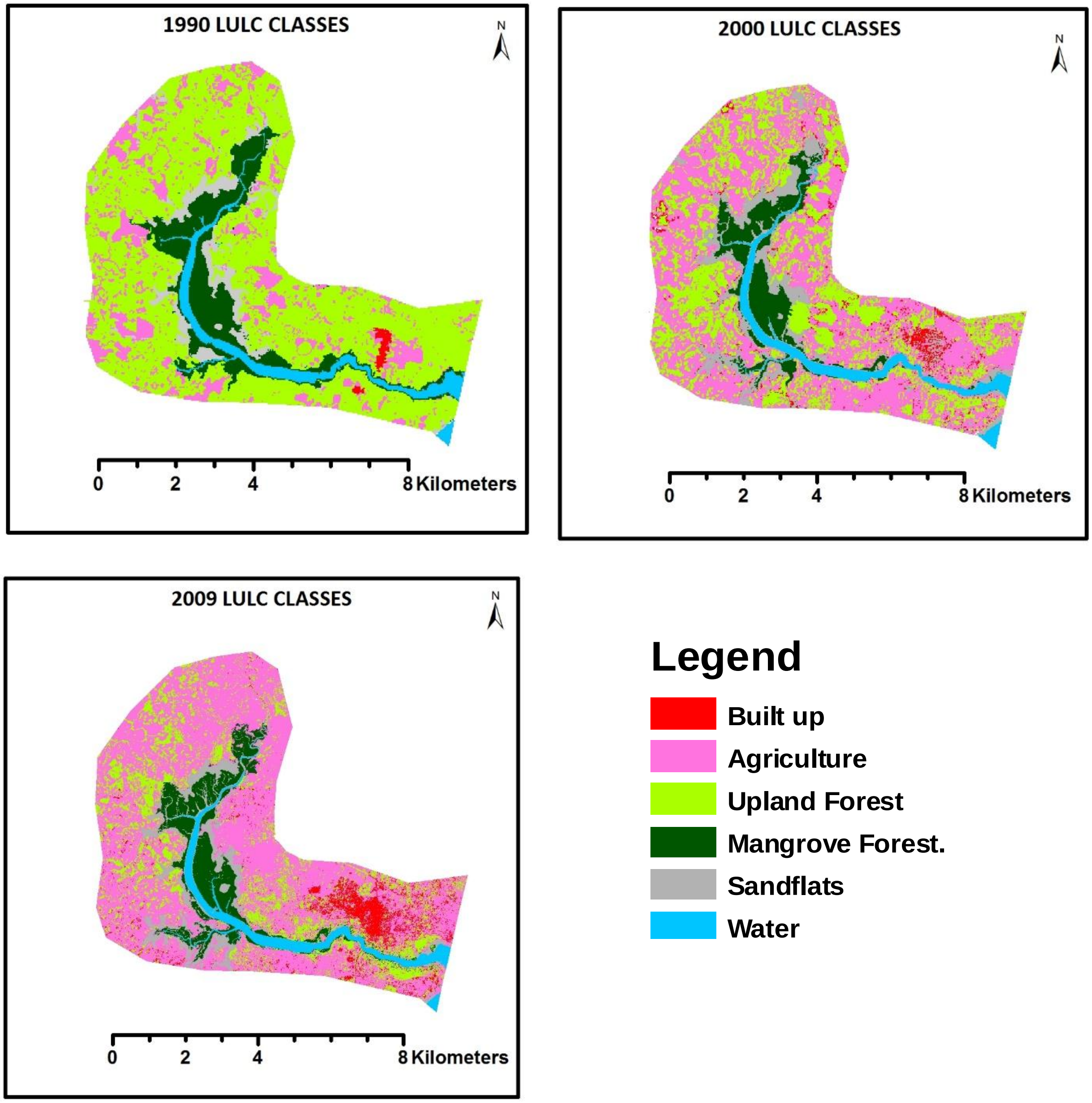

Figure 4.7: Land cover supervised classification Maps of the study area

The SPOT image of 2000 yielded Land-cover map (Figure 4.7) with Agriculture occupying the largest area of 2465.86ha (45.03%) as compared to other land-cover classes and distributed almost equally across the study area. This was a 194.13% Increase after previously having a cover of 838.37ha (Table 4.5). Upland forest was second with an area of 1575.66 ha (28.78%), a decline of 53.94% from a previous cover extent of 3301.10ha. Sand flat had an area of 575.30ha (10.51%) along the Mangrove forest edges and small patches in the northern and southern parts of the map. This was an increase by 77.21% from 324.64ha in 1990. Mangrove forest occupied an area of 438.85ha (8.01%) in the inter-tidal zone along the edges of the channel**.** This was a decrease by 29.44% for the 10 years from 1990. Built up area had an area of 143.78ha (2.63%) and occurred in clusters in the south eastern part of the map which was an increase by 348.89% as compared to its cover in 1990.

After classification of SPOT image of 2009 Agriculture was found to occupy the largest area of 3052.14ha (56.52%) and occurred in all parts of the map but with more concentration on the northern part. This was a 264.06% increase from 1990. Upland forest was second and covered an area of 898.62ha, (16.64%) with concentration near the Mangrove forest edges in the Northwestern part of the map. This was a very high decrease of 73.22% from 1990 coverage (Table 4.5). Mangrove forest had an area of 288.63ha (4.97%), representing 21.30% decrease for the 19 years from 1990. Sand flat had an area of 401.45ha (7.43%) along the mangrove forest edges representing increase from 1990 by 23.66%. Built up had an area of 347.09ha (5.04%) with concentration on the southeastern part of the map and very small patches within the entire scene except the creek Channel.

Table 4.5: Land cover changes between 1990 and 2009 within the study area in Mtwapa mangrove forest.

Land-cover	**Change (ha)**			**% Change**		
	1990-2000	**2000-2009**	**1990-2009**	**1990-2000**	**2000-2009**	**1990-2009**
Built-Up	111.75	203.31	315.06	348.89%	141.40%	983.64%
Agriculture	1627.49	586.28	2213.77	194.13%	23.78%	264.06%
Upland Forest	-1780.53	-677.04	-2457.57	-53.94%	-42.97%	-73.22%
Mangroves	-206.06	68.72	-137.34	-29.44%	15.66%	-21.30%
Sandflat	250.66	-173.85	76.81	77.21%	-30.22%	23.66%
Water	-3.32	-7.41	-10.73	-1.19%	-2.67%	-3.84%

From the analysis above major changes occurred within the19 years between 1990 and 2009. Figure 4.8 shows the comparative bar graph on changes within a cover type from 1990 to 2009. The great and significant increase and decrease happened within Agricultural area and upland forest area respectively. Agricultural area increased over the 19 years with 194.13% between 1990 and 2000 and 23.78% between 2000 and 2009. Upland forest cover declined over the same period by 53.94% between 1990 and 2000 and by 42.97% between 2000 and 2009. Mangrove cover decreased by -29.44% between 1990 and 2000 and increased by 15.66% between 2000 and 2009.

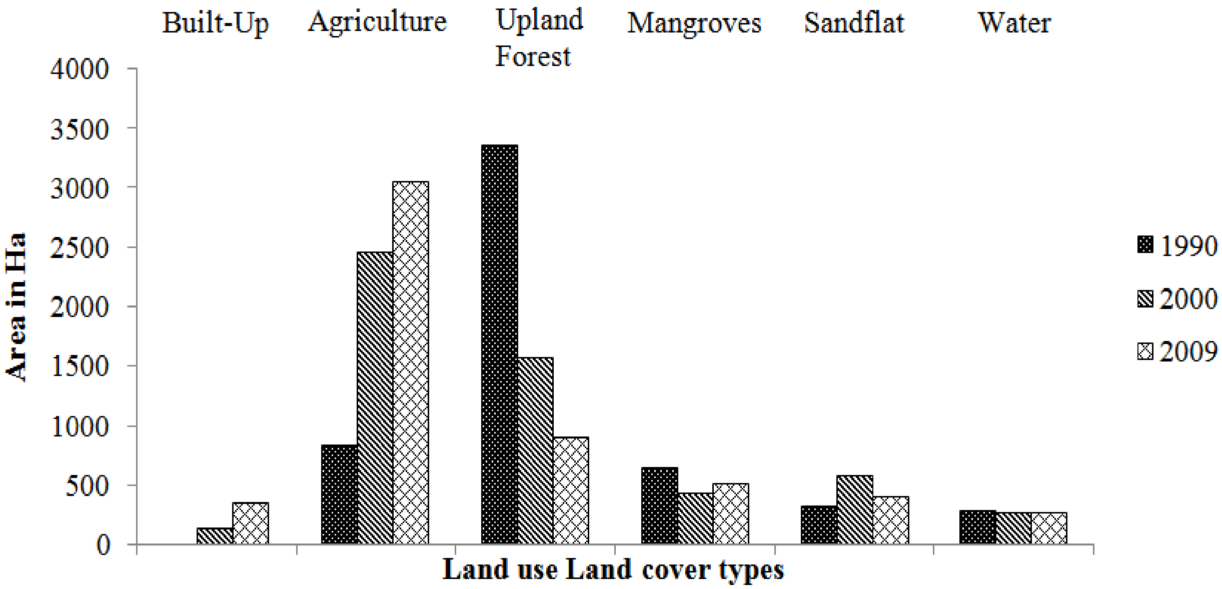

Figure 4.8: Comparative bar graph for Land cover changes from 1990 to 2009 within the study area in Mtwapa mangrove forest.

4.1.10 Mangroves cover change regression against Land use changes

Simple linear regression analysis of Mangrove cover against other Land-cover types indicated existence of a statistically significant relationship between Land-cover dynamics and changes in mangrove cover ($p<0.05$), except for built up area where there was no sufficient evidence to show existence of a linear relationship to Mangrove cover changes ($p=0.06516$) (Table 4.6).

Mangrove cover and Upland forest cover depicted a statistically significant positive correlation coefficient of $R=0.823$ ($p<0.05$). Agriculture had a strong but inverse significant correlation of

R= -0.83 ($p<0.05$), while Sandflat had a negative correlation of R=-0.917 with a coefficient of determination of R^2=0.841 ($p<0.05$). Built up had a negative linear relationship of R=-0.52 and a very low coefficient of determination of R^2= 0.27, with no statistically significant relationship to Mangrove cover change ($p>0.05$).

Table 4.6: Regression results of mangrove cover change against other Land cover changes within the study area in Mtwapa mangrove forest.

Mangrove VS LULC Type	Linear Equations	Correlation coefficient	P-Value	Coefficient of determination
MF:UF	y = -3340.3675 + 9.961*X	r = 0.8233	p = 0.03843	r^2 = 0.0677
MF:BU	y = 594.55650 - 0.7923*X	r = -0.5204	p = 0.06516	r^2 = 0.2708
MF:Agr	y = 6930.5413 - 9.0712*X	r = -0.8298	p = 0.03769	r^2 = 0.6885
MF:Sa	y = 1029.3081 - 1.1227*X	r = -0.9172	p = 0.02609	r^2 = 0.8412

Wa =Water, Bu = Built up, Sa = Sand flat, Agr = Agriculture, UF = Upland forest and MF = Mangroves forest.

4.1.11 Estimated CO_2 Emissions from mangrove Biomass loss between 1992 and 2009

CO_2 emissions per annum were derived from carbon stock loss per ha based on temporal mangrove cover change analysis (Table 4.2). The mangrove forest of Mtwapa was estimated to have lost about 26.72% of its cover between 1992 and 2009.

Based on the mean biomass carbon per hectare from the field assessment and the cover loss for the 17 years, this study estimated the carbon stock loss at 541.1506 Mg C Yr^{-1}, based on 2006 revised IPCC's **Stock-difference method.** This was a low end estimates by the virtue that this study assumed the biomass carbon per hectare to have been constant for the two periods. Hence, the total carbon emissions estimates due to mangrove cover loss in Mtwapa mangrove forest was found to be 1986.02 Mg CO_2 yr^{-1} (Table 4.7).

Table 4.7: Estimated carbon Emissions from biomass due to mangrove cover change between 1992 and 2009 in Mtwapa creek

Species	**1992 C stock (Mg) Estimates**	**2009 C stock (Mg) Estimates**	**Estimated C stock loss from live biomass (Mg C Yr^{-1})**	**Estimated CO_2 emissions from live biomass (Mg CO_2 Yr^{-1})**
Rhizophora mucronata	17261.54	15233.68	119.2859	437.7792
Ceriops tagal	9496.32	8606.04	52.36941	192.1957
Avicennia marina	7517.92	1384.88	360.7671	1324.015
Sonneratia alba	148.38	0	8.728235	32.03262
Total	34424.16	25224.6	541.1506	1986.023

*****molecular weight ratio of CO_2 to C = 3.67.

4.2 DISCUSSIONS

4.2.1 Vegetation structure and composition

Mangroves zonation, patterns and distribution is a unique aspect to mangrove forest ecosystem, with its possible causes having been debated extensively (Tomlinson, 1986). Different causes have been hypothesized including flooded conditions that may contribute to spatial patterns (McKee, 1993), water logging which results in virtual exclusion of free Oxygen (Beeckman *et al.,* 1990) and importantly the chemical composition of the mangal soil sediments (Matthijs S. *et al.,* 1999). Mtwapa creek has four main dominant mangrove species with Rhizophora mucronata overwhelmingly dominating with about 86% , Ceriops tagal is second with 8%, Avicennia marina 4% and Xylocarpus granatum 2% with traces of Sonneratia alba at the forest edges after the mudflats. These findings are consistent with what was reported by Okello *et al*., (2012), in assessment of self-sustenance of mangroves in Mtwapa creek where the study recorded presence of five species, with more than 70% dominance of Rhizophora mucronata. Occurrence of all identified species in the study sites portrayed the zonation pattern same as identified along the Kenyan coast (Ruwa, 1997).

Mtwapa creek has a mean tree density of about 2600-4425stems/Ha. This was comparable but slightly higher than results found in Mwache creek of 2050-2633stems/Ha (Kaino, 2012). However, this was quite higher compared to the one recorded in Tudor of 1264-1301stems/ Ha (Mohamed *et al.,* 2008). Despite the high tree density, the mean DBH and Height of trees in the study area were very low with a mean of 5.09cm and 3.67m respectively. These figures were lower compared to the values in Tudor and Mwache creeks where mean DBH and Height were 6.4cm and 4.5m respectively (Olagoke, 2012).

The number of cut stumps in the study area was very high standing at mean of 2425 stumps Ha^{-1} and ranging from 1580 to 4417 stumps/Ha, suggesting possibilities of high rates of logging and could explain the low figures of DBH and tree Heights in the study area. There was significant variation in cut stumps between sites suggesting difference in the rate of deforestation, which could be influenced by a number of factors including forest accessibility and proximity to human settlements. Mibuyuni, Kidongo and Majaoni, which are close to the Mtwapa trading center and settlements, recorded the highest stump count and lowest canopy, whereas Kidutani and Mtepeni recorded lowest likely due to remoteness and inaccessibility due to dense growth respectively.

Correlation analysis showed that Canopy cover correlated highly with the cut stumps, with a significant correlation coefficient of R= -0.8024 and (p=0.015) at 95% confidence level. This is an indication that cut stumps resulting from deforestation could be directly leading to low canopy cover and importantly reduced biomass.

4.2.2 Biomass distribution

The total biomass of the study area was contributed by AGB and BGB with an overall mean biomass of 112.48Mg C ha^{-1}. This figure was closely similar to estimate in Gazi Bay of 106.7Mg C ha^{-1} reported for a 12- year old restored mangrove plantation, (Kairo *et al.*, 2008) and recent estimates of biomass in Africa where Kenya was estimated to have a mean biomass of 119 Mg C ha^{-1} as shown in Table 4.8 (Fatoyinbo, 2013) . The highest biomass was recorded at Mabirikani 140.26Mg C ha^{-1}, while the lowest biomass was recorded in Kidongo with a value of 86.09Mg C ha^{-1} with the difference likely being as a result of difference in vegetation structure determined by influence of human disturbances through direct deforestation, sedimentation and pollution (Okello *et al*., 2012).

Biomass figures in this study were slightly higher than those found in similar study in highly degraded Tudor creek, which recorded a total biomass mean of 70.5Mg C ha^{-1} (Olagoke, 2012). However, it was lower than what was found in Mwache creek which recorded a mean total biomass of 313Mg C ha^{-1}, with mean AGB of 230.60Mg C ha^{-1} and mean BGB of 82.7Mg C ha^{-1} (Kaino, 2012).

This study's biomass was very low though compared to Global values from similar studies in Palau, Micronesia where total live biomass ranged between 225Mg C ha^{-1} to 363Mg C ha^{-1} (Kauffman, 2011), but fell within the Biomass range of between (41- 460)Mg C ha^{-1} as reported earlier in the same region (Komiyama et al., 2008) and also within range reported in Bocas Del Toro of (8-194)Mg C ha^{-1} (Lovelock *et al*., 2005).

Table 4.8: Area of mangrove cover and mean biomass per hectare in selected African countries

Country	Area (Km^2)	Total biomass (Mg)	Mean biomass ($Mg\ ha^{-1}$)
Angola	1541	1,441, 200	93
Equatorial Guinea	181	2, 922, 420	161
Ghana	76	742, 925	97
Guinea	1, 889	18, 153, 800	108
Kenya	**192**	**2, 294, 820**	**119**
Madagascar	2, 059	24, 856, 900	121
Mozambique	3, 054	30, 974, 100	101
Nigeria	8, 573	94, 788, 000	111
Somalia	30	436, 907	143
South Africa	12	40, 018	100
Tanzania	809	11, 037, 800	136
Africa	25, 960	301, 665, 553	116

(Source: Fatoyinbo, 2013)

There was no significant difference in Biomass between study sites; this could be due to homogeneity in species occurrence throughout the study area and the fact that most of the sites were dominated by two species (*Rhizophora mucronata* and *Ceriops tagal*). AGB contributed 67.11% while BGB contributed 32.89% of the total biomass, conforming to the ones reported in similar studies (Komiyama *et al*., 2008; Olagoke, 2012; and Mwihaki, 2013).

High biomass in Mabirikani compared to other study sites, is likely to be as a result of luxuriant forest growth driven by fresh water runoff from upcoming river, which increases sediment input and adequate nutrient supply which is known to promote mangrove growth and productivity (Roy, 1997; Bosire *et al*., 2012). It was observed that this site had minimal human disturbances equally evident by low value of stump count with highest DBH and Height except at the very end where conversion to cropland was noted to have occurred. Remoteness of this site could also be another reason for the observed reduced human disturbances. Comparatively, the low biomass recording sites of Kidongo, Mibuyuni and Majaoni which are close to human settlements and located close to the sea, recorded the lowest biomass with causes closely linked to increased human related disturbances including indiscriminate deforestation for poles and charcoal burning.

4.2.3 Bulk Density

Bulk density in mangrove ecosystem has been found to be affected by a number of factors including soil texture, soil minerals, and forest disturbances by human activities like deforestation and influence of siltation effects from land uses (Arshad *et al*., 1996). Mwihaki (2013) explained fluctuation of bulk density in Mwache creek as being a result of varying vegetation density, the morphology and heterogeneity in the rooting systems of mangroves. Typically, bulk density increases with increase in soil depth (Ceron-Breton *et al*., 2010), this results from soil compaction due to the weight of the upper sediment layers (Calderon *et al*., 2011). Soil organic carbon is in turn a function of bulk density, soil depth interval and soil organic carbon concentration (Kauffman and Donato, 2012).

In this study a mean soil bulk density of 0.71g/cm^3 was obtained with depth trend showing steady increase in bulk density down the soil profile though with a small range of about 0.005g/cm^3. This was consistent with what was obtained in Tudor creek (Olagoke, 2012) and similar study in Campeche Mexico (Ceron-Breton *et al*., 2010). There was no variation in bulk density between study sites at same depth, probably because of similarity in vegetation pattern, distribution and root system in most sites. The lowest bulk density ranged from 0.59g/cm^3 recorded at Mibuyuni within (0-15)cm depth interval, to the highest bulk density of 0.88g/cm^3 recorded at Kidutani within (50-100)cm interval. However in (50-100) cm depth interval, there was a significant variation between sites recorded which could be as a result of different sedimentation rates between sites and influence from vegetation stand densities.

The variation in mean bulk density between study sites was also attributable to different sedimentation rates resulting from differences in rates of sediment input by tidal fluxes. Moreover, soil properties of sites like Mabirikani are influenced by sediments from upland erosion that is brought in by the river joining the creek and could also explain the variability in bulk density between sites. Compaction rates is a possible reason for the variation seen in the lowest depth interval of (50–100) cm as described above.

4.2.4 Ecosystem Carbon

Biomass carbon of 49.46Mg C ha^{-1} found in this study, is lower than what was obtained in Mwache of 142.8Mg C ha^{-1} (Mwihaki, 2013) but higher than that of Tudor of 32.90Mg C ha^{-1} (Olagoke, 2012). The highest biomass carbon was at Mabirikani 61.79Mg C ha^{-1} which is linked to high biomass recorded in that site, while the lowest of 37.75Mg C ha^{-1} in Kidongo is linked to low biomass resulting from low figures of DBH and Height with high cut stump count. The above ground carbon was higher than below ground carbon, with means of 35.06Mg C ha^{-1} and 14.40Mg C ha^{-1} respectively. This follows the 67.11% to 32.89% ratio of AGB and BGB above.

Overall biomass carbon did not show any statistical difference between sites and this is explained by the similar species distribution and pattern throughout the study area. However, it is important to note however that biomass carbon was low in Kidongo, Mibuyuni and Majaoni compared to other sites with likely reason being disturbance due to proximity to settlements.

Mangrove sediments' ability to store large amounts of carbon is a unique factor that distinguishes it from other forests in terms of their ability to store carbon (Donato *et al.*, 2011). Results on SOC indicated high amounts of carbon concentration throughout the soil profile with a mean of 3.02% which is consistent with the findings by Kauffman *et al.* (2011) in similar mangrove ecosystem. Furthermore, lower profile layers had highest overall SOC along the depth interval. This is explained by the fact that the buried organic matter and dead trees accumulate in layers with the sub layers having decomposed to form peats for carbon compared to the upper layers.

Sediment carbon storage value in Mtwapa forest is comparably higher than the global average for upland forest in the tropics stated by Kauffman *et al., (*2011) despite the degradation rate, indicating its importance in carbon storage. The overall soil carbon of 196 Mg C ha^{-1} in Mtwapa is relatively lower than the estimates of (820 – 920) Mg C ha^{-1} reported by Donato *et al.* (2011). Biomass loss has been explained to be the cause of decreased rate of soil carbon sequestration in mangrove forests (Ceron-Breton *et al.,* 2011; Lovelock *et al.*, 2011). It is likely that recurrent anthropogenic and climate related disturbances in this forest could have resulted to a significant reduction in the carbon sequestration potential, hence carbon estimate which is way lower than estimates from other mangrove ecosystems in the world.

The mean carbon stock for the entire Mtwapa creek ecosystem of 245.54Mg C ha^{-1}, is close to the results from earlier and similar studies in Tudor and Mwache creeks as illustrated in Table 4.9. This is a confirmation that they indeed store a lot of carbon more than the terrestrial systems in Kenya which store about 137.8Mg C ha^{-1} (FAO, 2009). However this values are low compared to other similar ecosystems in the world which have reported carbon stocks of between 820Mg C ha^{-1} to 920Mg C ha^{-1} in Indo-Pacific region (Donato et al., 2011) and 479-1068Mg C ha^{-1} in Palau, Micronesia (Kauffman *et al*., 2011). This low values are attributed to the fact that this forest ecosystem was dominated by Rhizophora mucronata and Ceriops tagal with low decomposition rates than species like Avicennia marina, (Huxham *et al*., 2010). Furthermore, suspected high rates of logging as shown by high stump count and low percentage canopy cover is suggestive that the recurrent anthropogenic and Natural-related disturbances could be the result of significant reduction in the carbon sequestration potential of this forest.

Greatest contribution to the carbon stocks of the ecosystem was from sedimentary soil pool with 79.86% of the total carbon stocks, followed by AG biomass pool with 14.28% and BG biomass pool was the least contributing 5.86% of the total carbon stocks. This was consistent with local results from Mwache and Tudor creeks indicating higher carbon stocks in the soil sediment pool than live biomass, 63% and 88% respectively (Mwihaki, 2013; Olagoke, 2012) and indeed similar studies globally with reported figures of about 66% - 77% soil carbon (Ceron-Breton *et al*., 2010; Howe *et al*., 2009 and Kauffman *et al*., 2011).

Table 4.9: Comparison of carbon stocks of the study area with other similar studies in the world

Study site	Total carbon (Mg C ha^{-1})	Author(s)
China	185.5	Zhong and Qiguo, 2001
Campeche, Mexico	12.0 -222.0	Ceron-Breton *et al*,.2010
Indo-Pacific region	1023.0	Donato *et al*., 2011
Mtwapa, Kenya	**218.07 – 260.04**	**This study**
Mwache, Kenya	189.5-676.3	Mwihaki, 2013
Palau, Micronesia	479.0 -1068.0	Kauffman *et al*., 2011
Tudor, Kenya	242.0 – 334.0	Olagoke, 2012
Yap, Micronesia	853.0 -1385.0	Kauffman *et al*., 2011

4.2.5 Mangrove Cover change

Mangrove forest cover in Mtwapa was estimated to have reduced by 27% in the 19 years between 1990 and 2009, translating to 1.5% loss per annum. This falls within cover loss rates obtained in mangrove forests from coastal lagoon systems in Mexico, where annual deforestation rates were about 0.6–2.4% (Ruiz-Luna *et al*., 1999). Similarly this rate fall within the global mangrove cover loss range of 1-2% annually (FAO, 2007). Locally this study's cover loss fell between higher, mangrove cover loss in adjacent Tudor and Mwache creeks of 2.7% annually (Olagoke, 2012 and Kaino, 2012), and lower degradation rate of Kirui, (2012) who estimated mangrove cover loss in the Kenyan coastline at a rate of 18% for 25 year period between 1985 and 2010 which is equal to 0.7% loss annually.

This study found this cover change to be significant and accompanied with species loss attributed to mangroves diversity decline due to extreme harvesting of most valuable trees (Okello, 2012). Plate 4.1 shows some of the activities in Mtwapa creek forest leading to degradation and cover loss.

Plate 4.1: Illegally cut trees for firewood and charcoal burning in Mtwapa mangrove forest reserve. (Source: Author)

Cover loss was accompanied by species shift, which is explained by change in forest structure and species from secondary growth (Kairo, 2002). It is proven that mangrove species distribution, which is quite a striking aspect depends on the precise interplay of factors among them water level, salinity, pH, sediment fluxes and oxygen potential (Thom, 1984). Among these factors, salinity is a key factor which links the physical environment through the physiology of mangroves to patterns of spatial organization (Snedaker, 1982). Upland deforestation is another likely factor for cover loss as it leads to increase in flooding and increase in fresh water flushing especially during rainy season causing change in salinity levels. This then affects species zonation and is exhibited by species shift with *Rhizophora mucronata* dominating in the entire forest and loss of the salt tolerant species *Avicennia marina* that previously dominated the mangrove forest edges. It is certain that shifts in species due to climate change, forest degradation and loss of habitat connectivity may reduce the protective capacity of mangroves (Lee *et al*., 2014).

According to Kirui et al., 2012, the rate of mangrove cover loss in Kenya reduced immensely between 2000 and 2010. The finding from this study is a further proof, with recorded decrease in cover loss rate between 2000 and 2009 compared to the first period of 1990 and 2000. Presidential ban on harvesting of mangroves for domestic market in 1982 which took effect in 2000 (Abuodha and Kairo, 2001), could have lessened deforestation activities in this period. However, several other factors could have likely been responsible for reduced rates in Mtwapa.

These include reforestation and restoration campaigns by Kwetu Training centre under UNDP project whose primary objectives is conservation and sustainable utilization of mangrove resources *(UNDP, 2012). Over 190,000 mangrove seedlings were planted between 2007 and 2010 to repopulate areas of coastline forests that had been over-exploited by local communities (UNDP, 2012).*

Most of the coastal ecosystems are currently under environmental stress, mainly caused by the growth of the human population and increasing demand for food and services (Alonso-Perez *et al*., 2003). Apart from pressure due to local impacts, they also receive cumulative effects as a result of increased anthropogenic activities uplands (Victor *et al*., 2004). Agriculture, forestry, and urbanization have been the main transformers of the natural land cover (Bedford, 1999). This has impacted the coastal systems particularly mangrove swamps, where degradation occurs through indiscriminate tree cutting, sedimentation, addition of toxins and species dieback which leads to decreased area, and subsequent loss of connectivity between coastal wetlands and upland ecosystems.

4.2.6 Land cover changes

It has been established that the effectiveness of mangroves for coastal protection depends on a range of factor scales related to landscape, community and species (Lee *et al*., 2014). Deforestation is considered to be one of the most significant environmental problems globally (Zaitunah, 2004). Based on this study, the landscape surrounding Mtwapa creek was predominantly terrestrial forest land in 1990 with 61.29% cover. This has reduced at a very high rate to 16.64% in 2009 with conversions to agriculture and built up areas significantly taking its places. Agriculture and urbanization have been found to be the major accelerators of upland deforestation resulting to loss of biodiversity, change in soil profile and initiation of downstream erosion (Harris and Miller, 1984). A similar situation was observed in Mexico where Maass, (1995), reported conversion of terrestrial forest to agriculture and pasture at high rate of more than 60% entailing nearly to destruction of the entire forest structure and composition within a lagoon system. Plate 4.2 shows the agricultural farms that have extended up the shores of the creek.

Plate 4.2: Agricultural activities at the banks of Mtwapa Creek. (Source: Author)

When soils are exposed due to destruction of vegetation for agricultural purposes and wood harvesting, surface layers dry and impervious surfaces are produced causing acceleration of surface runoff during rainy season (Ellison, 1998). Flood peaks go up quickly and lead to erosion from the bare areas. The high rate of terrestrial deforestation and increase in cropland in Mtwapa is likely to have resulted in flooding and sedimentation within mangrove zone. This is known to affect mangroves (Gilman *et al*., 2008 and Ellison, 1998) and could be the reason for enlarged Sandflats taking the spaces of choked mangroves in the study area.

Growth of agriculture in Mtwapa is alarming with current surface area equivalent to 56.52% of the study area from 15.31% in 1990. Together with urban expansion, whose current cover is 5.04% of the study area, they are exerting a lot of pressure on terrestrial forest and the ecosystem downstream. Urbanization can lead to degradation through siltation and changes in water temperature, water flow and salinity (Macintosh *et al*, 2002). Effects trickle down to loss of biodiversity and pollution that becomes a threat to the mangrove system’s health. Similar outcomes were observed in a related study which assessed use of Remote Sensing Data to evaluate the extent of anthropogenic activities and their impact on Lake Naivasha, Kenya between 1986 and 2007. There was 37.2% decrease in forest cover, 103.3% increase in horticultural and irrigated farms and 90% increase in urban settlement placing great pressure on both the quality and quantity of the lake's water resources (Onywere *et al*., 2012).

Anthropogenic degradation of coastal systems is widely known to result to both climate and environmental changes which reduce the resilience of mangroves making them vulnerable ecosystems (Ellison and Farnsworth, 1996). Population growth has been identified to be a key force behind environmental change, especially in developing countries (Cheng, 1999). In this study Increase in population is highly linked to increase in cropland and built up areas and decrease in upland forest in Mtwapa. This affects the hydrological processes especially land surface flow leading to sedimentation downstream. The trends in population of major urban centers in Kenya show high increase in urban population within the last three decades. In the coastal region of Kenya, the statistics have been similar with a rise in population from 1.3 million people in 1979 to 3.3 Million people in 2009 (GOK, 2010). This represents 60.6% population growth between 1969 to 2009 equivalent to 2.02% population growth rate annually.

Table 4.10: Population of urban centres along the Kenya coast between 1969 and 2009

Name	**1969**	**1979**	**1989**	**1999**	**2009**
Kilifi	2,662	_____	14,145	30,394	44,257
Lamu	7,403	_____	_____	_____	13,243
Malindi	10,757	23,275	34,047	53,805	84,150
Mombasa	247,073	341,148	461,753	665,018	915,101
Msambweni	_____	_____	_____	_____	11,985
Mtwapa	_____	_____	**< 10,000**	**18,397**	**48,625**
Ukunda	_____	_____	_____	43,946	62,529

Source: Kenya National Bureau of Statistics (web). http://www.citypopulation.de/Kenya-Cities.html.

Mtwapa urban centre has seen increase in population from below 10,000 inhabitants in 1989, to about 18,397 inhabitants in 1999 and exponential growth to 48,625 inhabitants in 2009 (Table 4.10) (GOK, 2001; GOK, 2009). This is equal to annual population growth rate of 3.97%. The increase in population is mainly attributed to immigration from other parts of the country (GOK, 1996). New settlers obtained pieces of land to settle and do farming at small scale level, while others have engaged in business activities promoting growth of Mtwapa urban centre. This has led to clearance of upland forests for farming land and putting up of structures that have resulted to immense land cover changes.

4.2.7 Emissions from Mangrove Cover Change

Emissions estimates of 1986.03 Mg CO_2 yr^{-1} was obtained in Mtwapa creek and is equivalent to 48.50 Mg CO_2 ha^{-1} for the 17 years in which this study covered. In comparison with the Mwache creek study whose emissions were 328 Mg CO_2 ha^{-1} (Mwihaki, 2013), this study's emissions were way lower though cumulatively adds to the CO_2 GHG accumulation. Though the two creeks are peri urban systems, Mwache creek had higher degradation of 2.7% (Kaino, 2012) as compared to Mtwapa with 1.5% and explains the lower emission rates in Mtwapa. Furthermore, Mwache creek had better tree structural characteristics than Mtwapa leading to a big difference in the CO_2 emissions resulting from cover loss.

CHAPTER 5

CONCLUSION AND RECCOMMENDATIONS

5.0 Introduction

Provides summarized synthesis of all other chapters, based on the main findings and their implications. Concluding remarks and recommendations are based on the results of this study as provided on the previous chapter.

5.1 CONCLUSION

Mtwapa creek is a peri-urban mangrove ecosystem with a capacity to provide ecological, social and economic benefits to the local communities. However, findings from this study show that human driven activities have altered the mangrove forest structure as characterized by poor tree form, high cut stumps and low biomass carbon estimates of 49.46±8.45Mg C ha^{-1}. Although degradation was not significantly found to vary by location, it is paramount to note that areas close to villages recorded high rates of degradation, as seen by high cut stumps of 2425±423 stumps/Ha, and these areas subsequently recorded low biomass values of 112.48±19.14Mg ha^{-1}. Wood fuel is heavily relied upon by local communities besides being a major source of building materials for huts and provision of other important services. Total carbon stocks in Mtwapa mangrove forest of 245.54±20.95Mg C ha^{-1} was very low compared to global as well as locally available estimates of (350 – 450) MG C ha^{-1}. This is an indication that the ecosystem is under stress.

Land use changes upstream significantly changed during the study period within Mtwapa creek, with very high percentage of terrestrial forest loss to agricultural activities (>50%). High upland deforestation of 264.06% has likely led to absence of alternative source of wood resources in the area. This indicates that more pressure will be exerted on mangrove forest resources (current loss of 1.5% annually) as the only source to serve the rapidly increasing population. Soil carbon highly depends on the mangrove cover and its state. It is obvious that soil sediments cannot capture and store carbon independently, hence placing mangroves as a very important integral part of the process. Soil carbon of 196 Mg C ha^{-1} was very low due to the poor form of mangrove forest resulting from degradation over time. This therefore means any activity leading to deforestation or degradation of this forest will jeopardize the natural process of carbon fixing and further handicap the functionality of the ecosystem from its crucial role of carbon storage.

Highly significant relationship exist between land use changes and integrity of mangrove ecosystem downstream in Mtwapa creek. Mangrove cover change is subject to a number of human caused activities within and without the forest. Among the major activities are direct indiscriminate and selective cutting of mangrove trees and land use change related activities upland. There was high link between mangrove degradation and terrestrial forest loss to cropland and built up areas. The rapidly changing land uses to Agriculture and settlements in Mtwapa poses a lot of threat to mangroves sustainability and the entire coastal environment. High rates of terrestrial forest loss leads to disturbance of upland biodiversity, modification of topography, alteration of soil properties and destruction of habitat conditions. These activities continue to slowly but detrimentally change abundance and spatial pattern of Mangrove forest environment in Mtwapa.

5.2 RECOMMENDATION

To improve the carbon sequestration and storage potential of Mtwapa creek ecosystem and increase its integrity and service provision potential, a landscape approach which combines sound land husbandry upstream and mangrove conservation is recommended. Specifically this study recommends the following:

(i) Mangroves are very important ecosystems in sequestration of carbon, calling for keen monitoring and protection of these habitats. Protective zone of mangroves should be established especially to act as a buffer against coastal erosion from up-stream activities which can have major impacts on down-stream ecosystems and consequently negative socio-economic impacts.

(ii) The Protective measures in place should be revised by Kenya Forest Service (KFS), Kenya Wildlife Service (KWS) and Kenya Marine and Fisheries Research Institute (KMFRI) who jointly are mandated to protect these unique forest reserves. This is especially to stop indiscriminate and selective tree cutting by the local communities and seek for alternative sources of fuel and resources. Hot spot areas for mangrove logging which are close to the settlements should be identified and more attention given to secure and rehabilitate the areas.

(iii) Restoration efforts by KFS, KWS and KMFRI should be initiated on degraded areas of the mangrove forest to regain the initial state, however, many of the functions and attributes of mangroves, including their productivity and biodiversity support, can be regained through artificial restoration.

(iv) Forest managers should adopt GIS and RS technique as a valuable tool in locating and predicting forest cover changes. Thematic maps of forest cover types and various LULC classes can be distinguished by the satellite image interpretations and used to evaluate their conversions as well as analyzing their trends. These will aid in forest cover change detection and identification of areas under risk of conversion and deforestation.

(v) This study recommends further research on assessment of site specific effects of upland activities on mangroves forest integrity along coastal region of Kenya using Remote sensing techniques to aid in up scaling the database and provide the basis for the national conservation.

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