# Carbon Stocks and Sequestration Potentials in Managed Mangrove Plantations of Gazi Bay, Kenya

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# DECLARATION

This research thesis is my original work and has not been presented for award of a degree in any university.

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# DEDICATION

This research thesis is dedicated to my dear parents; Mr and Mrs Justus Musyoka Maiyu, my motivation and Inspiration.

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# LIST OF ABBREVIATIONS AND ACRONYMS

| | |
|---|---|
| AGB | Aboveground biomass |
| BA | Basal Area |
| BGB | Belowground biomass |
| C | Carbon |
| $CO_2$ | Carbon (IV) oxide |
| $D_{130}$ | Stem diameter at breast height, measured at 130cm above ground level |
| DBH | Diameter at breast height |
| DOM | Dead Organic matter |
| FAO | Food and Agriculture Organization of the United Nations |
| g | Gram |
| GHGs | Greenhouse gases |
| Gt | Giga tones |
| IOI | International Ocean Institute |
| IPCC | Intergovernmental Panel for Climate Change |
| IV | Importance Value |
| $Km^2$ | Square kilometers |
| KMFRI | Kenya Marine and Fisheries Research Institute |
| LOI | Loss on Ignition |
| LULUCF | Land Use, land Use Change and Forestry |
| $Mg\ C\ ha^{-1}$ | Mega grams Carbon per Hectare |
| Pg | Pico gram |
| Ppm | Parts per million |
| $r^2$ | Coefficient of determination |
| REDD+ | Reducing Emissions from Deforestation and forest Degradation |
| SOC | Soil Organic Carbon |
| SOM | Soil Organic Matter |
| UNFCCC | United Nations Framework Convention on Climate Change |

# GLOSSARY

**Carbon sequestration:** The removal of $CO_2$ from atmosphere (source) into green plants (sink) where it can be stored indefinitely

**Allometric equation:** Refers to an equation derived from the relationship between size and shape of an organism.

**Stand:** A homogeneous plant community with distinctive plant association that may be recognized elsewhere.

**Tree biomass:** The total amount of above- and below-ground living organic matter in trees expressed as oven-dry weight per unit area.

**Necromass**: Dead fine roots that have not yet decomposed

**Carbon pool:** Components of the ecosystem that can either accumulate or release carbon and have classically been split into five main categories: living above-ground biomass (AGB), living below-ground biomass (BGB), dead organic matter (DOM) in wood, DOM in litter and soil organic matter (SOM)

**Carbon stock:** All carbon pools summed up together

**Juveniles**: Young organism; in mangrove forests it refers to seedlings and saplings i.e. a young mangrove that has not reached exploitable stage

**Allochotonous**: External sources of nutrients, especially from river run offs or anthropogenic discharges.

## TABLE OF CONVERSION FACTORS

Centimeter (cm) = 100 millimeters

1 Meter = 100 centimeters

1 Hectare (ha) = 10,000 squared meters

1 Kilometer (km) = 1000 meters

1 Mega tone (Mg) = 1,000 kg

1 Pico gram = $10^{-9}$ Kilograms

# ABSTRACT

Mangroves forests are important ecosystems found along the coastlines throughout the tropical coast. They provide goods and services that are of ecological, environmental and cultural importance to society. Mangroves store exceptionally large quantities of carbon stocks. This carbon is bound to be emitted when mangroves are degraded. The objective of this study was to quantify total ecosystem carbon stocks for mangroves forest at Gazi Bay, Kenya. Three categories were selected for this study; 13 year old El Niño plantation, 19 year old Kinondo plantation and a Natural stand of *R. mucronata*. Plots of 10×10 $m^2$ were selected in each study site. Four carbon pools were investigated; above and belowground vegetation carbon, dead wood and soil carbon. Additionally, Nitrates, Ammonium and Phosphates from the soil were also measured. Twenty five trees were harvested and used to develop biomass and volume equations of the form $y = ax^2 + bx + c$, and $Y = aX^b$ respectively for the 19 year old Kinondo plantation. Biomass for carbon determination in Natural and El Niño stands was estimated using a general equation. Mean total C stocks in El Niño and Kinondo plantations was 914.5±237.3 and 812.6±186.5 Mg C $ha^{-1}$ respectively, while the Natural stand contained significantly lower mean total C stocks of 692.9±142.9 Mg C/ha. (ANOVA, F; 2, 6 = 28.91, $P < 0.05$). Ammonium was the most abundant nutrient in the three sites. Belowground biomass was generally found to correlate negatively with the soil nutrients. This study show reforestation influences structural development of replanted mangroves and that replanted mangrove are significant carbon stores. Soil organic matter was signicantly high in plantations than in the Natural stand (F; 2, 92 = 8.89, $P < 0.05$). Proper monitoring is essential for continued verification of stored carbon stocks and improvement of data quality. Development of local allometric equations for all mangrove species can be useful estimating aboveground biomass carbon.

# CHAPTER ONE

# INTRODUCTION AND LITERATURE REVIEW

## 1.0 Background information

Atmospheric concentration of $CO_2$ on earth surface have increased from 280 parts per million (ppm) in pre-industrial times to 391.3 ppm in the year 2012 and are expected to increase by 2 ppm every year (WMO, 2015). Anthropogenic $CO_2$ emissions arise primarily from deforestation, burning of fossil fuels, agriculture and other land uses (IPCC, 2007). Forests store close to two-thirds of terrestrial carbon (FAO, 2007). Deforestation and forest degradation mostly in tropical forests emits 3.7 Gt of $CO_2$ or approximately 12% of the total anthropogenic $CO_2$ emissions (IPCC, 2007). Climate change is currently being regarded as one of the major threats to sustainable development (Mcleod & Salm, 2006). Enormous research efforts and studies are being carried out to find ways to slow down the climate change process and subsequently alleviate its adverse impacts (Kridiborworn *et al*., 2012; Pendleton *et al*., 2012). Technology and measures resulting to greenhouse gases (GHGs) emission reductions using LULUCF sector are amongst those that have received considerable attention (Siikamaki *et al*., 2012). Regarding this, forestry which forms significant part of the global carbon cycle, plays a significant role of balance of GHGs through carbon sequestration and emissions. Forestry can thus contribute significantly to low-cost global mitigation measures which are synergistic with adaptation and sustainable development (Barbier *et al*., 2008; Mitra *et al*., 2011). The need to reverse the ongoing climate change trends has interested many scientists and researchers. Many studies and researches have thus been carried to understand better the ecosystem sequestration functions and potentials of world’s forests (Clark, 2004). United Nations Program aimed at Reducing Emissions from Deforestation and Forest Degradation (REDD+) asserts that, stabilization global temperatures will be "practically impossible" to achieve without reducing emissions from the forestry sector in addition to other mitigation actions" (IPCC, 2007). Forests as carbon sinks have also been included in articles 3.3, 3.4, 6 and 12 of the Kyoto protocol as one of the mechanisms for mitigating climate change (Lovelock, 2008).

Mangrove forests are composed of salt tolerant plants that grow along the tropical and subtropical coastline regions in the area between land and sea. and have been recently reported to be one of the major stores of carbon (Bianchi *et al*., 2013; Sitoe *et al*., 2014). Clearance of these forests makes them a significant source of atmospheric $CO_2$ and also leads to permanent loss of sequestration ability or mitigation of $CO_2$ emissions thus making future mitigations targets difficult to achieve unless restoration is undertaken (Bouillon *et al*., 2008; Pendleton *et al*., 2012). There are few studies on C storage in mangrove ecosystems with extremely few studies in Africa. In Kenya Mangrove carbon studies are still limited and most of them are focused on natural mangroves with no studies on restored mangroves.

## 1.1 LITERATURE REVIEW

### 1.1.1 Mangrove species and diversity

Mangrove forests are a taxonomically diverse group of salt tolerant flowering plants growing along the coastline in the intertidal area between the land and the sea (Mcleod & Salm, 2006 Joshi & Ghose, 2014). They grow primarily in tropical and subtropical regions between latitudes $32^0$ North and $38^0$ South, and reach their maximum development between latitudes $25^0$ North and $25^0$ South with richest mangroves assemblages occurring in areas with water temperatures greater than $24^0$ centigrade in the warmest month (Mcleod & Salm, 2006; Kuenzer *et al*., 2011).

There are about 70 species of mangrove plants in 66 genera and 16 families (Kuenzer *et al*., 2011; Joshi & Ghose, 2014). They occur over a diverse geomorphological settings ranging from river dominated, wave dominated, river and wave dominated, tide dominated, drowned bedrock valleys to carbonate setting (Giri *et al*., 2011; Donato *et al*., 2012). The river dominated mangroves are the most productive system because of allochotonous input of rich sediments from rivers (Donato *et al*., 2012; Bianchi *et al*., 2013; Sawale & Thivakaran, 2013).

Mangrove ecosystems are estimated to have covered 75% of the tropical and subtropical coastlines (Mcleod & Salm, 2006). The current extent of mangroves coverage worldwide has been reported to vary between 14-24 million hectares (FAO, 2007; Spalding *et al.*, 2010). The most recent estimate suggest that true coverage value is near the low end of this range and is approximated to be 13 million hectares (Giri *et al*., 2011). This reflects to either improved mapping techniques (Kuenzer *et al*., 2011; Omo-irabor *et al*., 2011) or losses associated with deforestation or conversions (Kuenzer *et al*., 2011., Donato *et al*., 2012).

Asia alone supports the largest area of mangrove coverage, extending over 6.8 million hectares and representing 34-42% of the world’s total mangrove cover, followed by Africa (20%), North and Central America (15%), Oceania (12%) and South America (11%). Approximately, 75% of this world mangrove is concentrated in 115 countries and Indonesia alone contributes 22% of this (Giri *et al*., 2011; Murdiyarso & Kauffman, 2011).

### 1.1.2 Importance of mangroves

Mangrove ecosystems provide a wide array of environmental goods and services to the people at local, national and global levels (Barbier *et al.*, 2011a). They provide the local people with wood products including timber, poles, posts, fish traps, firewood and charcoal (FAO, 1993; Badola & Hussain, 2005). Much of the marine fisheries production largely depend the mangrove ecosystems. Thus mangroves form nursery and feeding grounds for commercial and artisanal fisheries, and are important habitats and feeding grounds for a range of benthic and pelagic marine animals and bird species (FAO, 2007; Barbier *et al*., 2002). In addition, they enhance environmental quality by reducing coastal erosion, trapping of sediments and other pollutants from activities upstream thereby maintaining water quality (Barbier, 2008). Mangrove forests also act as natural barriers against strong waves and other natural oceanic catastrophes and hence they protect local people from the damages arising from these events (Barbier 2008; Alongi, 2008).

Mangroves play significant role in global carbon sequestration by acting as sinks of carbon within the tropical coastal zones. Despite occupying less than 1% of the coastal area, mangroves are estimated to contribute approximately 20% of all carbon emissions arising from deforestation (Caldeira, 2012). They are able to sequester atmospheric carbon both in their above and belowground biomass and also in sediments (Adame *et al*., 2013; Sawale & Thivakaran, 2013). The global benefits associated with carbon sequestration in the mangrove is the removal of the harmful greenhouse gases (including carbon dioxide) from the atmosphere and therefore helping in mitigating effects of climate change (Pendleton *et al*., 2012; Siikamäki *et al*., 2012; Alongi, 2009).

### 1.1.3 Mangroves and climate change

Mangroves forests are among the most productive ecosystems (Bhomia *et al.*, 2013) with a standing plant biomass of 500 t/ha (Bianchi *et al*., 2013; Joshi & Ghose, 2014). Poungparn *et al*., (2012) estimated global C in mangroves as 4.03 Pg C while in their assessment of global primary production Bouillon *et al*., (2008) estimated the net primary production of mangroves to averagely be 218± 72 Tg C/year using a global area of 160, 000 km$^2$.

The amount of organic carbon stored in any mangrove ecosystem depends on several factors such as sources of carbon which include, tidally suspended organic matter, amount of rainfall in the hinterland or from local production by mangroves (Raza *et al*., 2011). Mangroves are able to sequester and store large quantities of carbon both in the plants biomass and also in the sediments below them (Murdiyarso, 2010; Pendleton *et al*., 2012; Adame *et al*., 2013). Their high carbon content in the soil is as a result of high sedimentation rates and sustained anoxic conditions in the belowground which results into low decomposition rates of soil organic matter hence building up of soil carbon (Ray *et al*., 2011). Carbon sequestration in mangrove ecosystem is a continuous process which often results into large deposits of carbon which together with carbon stored in salt marshes and seagrass is collectively referred to as “Blue carbon” (Matsui *et al*., 2012; Pendleton *et al*., 2012). In mangrove ecosystems, this carbon is often more than a thousand years old, making these habitats among the most carbon-rich ecosystems on earth (Donato *et al*., 2011; Webb *et al*., 2013).

Unlike the tangible benefits of provisioning goods and services that can be associated with a market value or price, benefits of regulating ecosystem services like carbon sequestration are less understood (Bouillon *et al*., 2008; Matsui *et al*., 2012). Their contribution as carbon sinks has been overlooked for several reasons; there was lack of carbon accounting methodologies capable of catering for the unique nature of mangrove different certification schemes. This is however being resolved following drafting of a supplement for IPCC guidelines in the year 2013 (Donato *et al.*, 2012). Secondly, while the carbon present in aboveground biomass is widely reported; reliable data still lacks for belowground biomass and soil C for most mangrove; (Donato *et al*., 2012; Adame *et al*., 2013).

Thirdly, mangrove soil despite being rich in organic matter and containing moderate to high carbon concentration, the amount of C as reported in most studies lacks information on the total

C stored within the mangrove ecosystem (Donato *et al*., 2011; Adame *et al*., 2013) and thus represents very conservative estimates (Donato *et al*., 2012; Ajonina *et al*., 2014).

This is because the methodologies used for quantification of C sequestration vary considerably (Brown, 2002a, 2002b; Donato *et al*., 2011). Despite the fact that mangroves fit within the general REDD+ framework, soil carbon which constitutes the majority of the mangrove carbon pool is not taken into consideration by carbon offset program (Siikamaki *et al*., 2012). Clean development mechanism which focuses mostly on terrestrial ecosystems has however accepted both small and large scale methodologies for mangrove restoration under its afforestation and restoration activity. This however presents limited opportunity for the mangroves which have a potential in mitigation of climate change (Herr *et al*., 2012).

Another reason is due to existence of complex institutional landscape in mangrove management. Mangrove management is rarely covered by one specific national policy despite existence of numerous policies covering various benefits provided. With no single environment protection ministry in different countries assuming responsibility to properly and responsibly manage mangrove ecosystem, they tend to be subjected to different threats and degradation pressures (Kridiborworn et al., 2012). The other factor is the lack of understanding of mangrove deforestation rates (Chen *et al.*, 2012). As part of national preparedness to REDD+, hefty investments are always made while assessing the deforestation rates with the aim of establishing baseline for future activities (Caldeira, 2012).

Few countries have included mangroves in national baselines inventories or their ongoing monitoring, reporting and verification (MRV) systems. This has resulted to REDD+ projects in mangrove forests being very costly compared to their terrestrial counterparts. This is because MRV for emissions in mangrove ecosystems needs to be integrated with the ongoing management cost (Matsui *et al*., 2012). Lastly, despite both gross and net primary production rates being used to assess the sequestration ability of mangroves, large uncertainties still exist. This is because more than 50% of the carbon fixed by mangroves is unaccounted for (Bouillon *et al*., 2008). Failure to link this essential ecosystem service to societal benefits leads to continued deforestation, forest degradation and inefficient decision making regarding mangrove ecosystems, (Bouillon *et al*., 2008; Barbier *et al*., 2011a).

### 1.1.4 Global threats to mangroves

Despite mangrove ecosystems providing tremendous value and benefits to coastal communities and other associated species, they are currently being destroyed at alarming rates (Giri *et al*., 2011). Over the last 50 years, about one-third of the world's mangrove forest cover has been lost (Giri *et al*., 2011; Caldeira, 2012) as a result of forest degradation and deforestation. Major threats facing mangroves include overexploitation of forest wood products, pollution and conversion of mangrove areas for other land uses (Murray, 2012; Farida-Hanum *et al*., 2012). Climate change now threatens the remaining mangrove forest cover through its associated impacts especially sea level rise (Krauss *et al*., 2011; Chen *et al*., 2012).

The greatest threat to mangroves is conversion to aquaculture ponds (Barbier *et al*., 2008; Pedgion, 2011). Globally, aquaculture has been reported to account for 20 to 50 percent loss of mangrove (FAO, 2007; Donato *et al*., 2011; Pedgion, 2011). Losses due to aquaculture development have greatly impacted on mangroves with Asia and Latin America having suffered the highest loss. Asia has lost over 30% of its mangrove cover due to aquaculture (Barbier, 2008). 56% loss has been reported in Thailand (Kathiresan & Bingham, 2001), 17% in Malaysia, 25% Indonesia, 50% in India and 30-70% in Latin America ( FAO, 2007; Hooijer *et al*., 2010).

In Africa aquaculture is not a well developed industry but could be a potential to similar degradation threat if commercial aquaculture is developed. In Kenya, it’s difficult to quantify and justify mangrove loss as a result of aquaculture and this is because the coastal communities living near mangroves forest have not fully embraced it (Kirui *et al*., 2012).

Given their large carbon stocks, the emissions arising from conversion and degradation of mangrove ecosystems are exceptionally high and hence a very significant source of greenhouse gases (Donato *et al*., 2011; Pendleton *et al*., 2012 Adame *et al*., 2013). Despite mangroves forests accounting for less than 1% of the total tropical forest area globally (Giri *et al*., 2011), clearing of mangroves forests generates approximately 10% of emissions from all global deforestation contributing to climate change (Caldeira, 2012; Pendleton *et al*., 2012; Siikamäki *et al*., 2012).

### 1.1.5 History of mangrove restoration, conservation and management

Mangrove restoration projects have been attempted with mixed results throughout the world (Ellison, 2000) with some being successful and others a total failure (Lewis, 2005; Bosire *et al.*, 2008). In his comprehensive review to examine existing goals for mangrove restoration projects, Ellison (2000) found out that most of the restoration efforts focused on silviculture alone.

Early reforestation and management of mangroves has been reported to be practiced mostly in Southeast Asia for decades mostly to produce forest products such as; wood, fuelwood, charcoal, thatching materials and legislative compliance (Gong & Ong, 1995; Kridiborworn *et al*., 2012) The longest recorded history of mangrove management involving 600,000 ha of mangrove forests occur in the Sundarbans region of India and Bangladesh (Gong & Ong, 1995; Kridiborworn *et al*., 2012). Sundarbans forest has been managed since 1769 mainly for charcoal and timber production. Detailed work-plans of the forest were reported to have been prepared between years 1893-1894 (Gong & Ong, 1995). Similarly, the mangrove forests of Matang in Malaysia, that cover 40,000 ha, have also been managed for fuelwood production since 1902 (Kridiborworn *et al*., 2012).

The rationale for mangrove restoration has now shifted from timber production to recognition of diverse ecological services provided by the mangroves in both natural and restored mangrove forest ecosystems (Ren *et al*., 2011). One of the ecological services of mangroves is their ability to store and sequester carbon and limit net emissions from forest degradation and deforestation (Bouillon *et al*., 2008). Mangroves are now being conserved to enhance forest carbon sinks and reservoirs so as to reduce atmospheric concentrations of $CO_2$ (Jin-ping *et al*., 2012) providing an important opportunity for climate mitigation and adaptation (Fontalvo-herazo *et al.*, 2011; Henry *et al*., 2011; IPCC, 2007).

Strategies as Clean Development Mechanism (CDM) and REDD+ of the UNFCCC which aim at mitigating climate promote afforestation, reforestation and enhancement of carbon stocks have been initiated in many countries. They promote innovative low carbon resilience projects in developing nations. Mangrove plantations have been established in different countries like China, India and Indonesia among others so as to promote carbon sequestration (Ren *et al*., 2010; Fontalvo-herazo *et al*., 2011; Chen *et al*., 2012). In Kenya, mangroves are also being replanted for carbon sequestration. More recently mangroves have also been restored for fish production (Primavera, 1998), eco-tourism (Mcnally *et al*., 2011), erosion control and experimental analysis of mangrove biology (Kairo *et al*., 2001) and restoration of forests damaged by oil spills (Di Nitto *et al*., 2008). With the recent realization of mangrove capability to sequester carbon, many small and large scale reforestation and afforestation projects are expected to be established (Caldeira, 2012; Siikamäki *et al*., 2012).

### 1.1.6 Mangroves in Kenya

#### 1.1.6.1 Mangrove vegetation along the Kenyan coast

Kenya has approximately 45,590 hectares of mangroves (Kirui *et al*., 2012). They are distributed along the coastline and particularly in creeks, bays, deltas and estuaries (Bosire *et al*., 2003; Kairo *et al*., 2008). Lamu and Tana River accounts for over 70% of the total mangrove forests cover at the Kenyan coast (Kairo *et al*., 2009; Kirui *et al*., 2006). Less extensive mangrove areas occur in Kilifi, Mida Creek, Mtwapa North of Mombasa, and Gazi and Vanga areas in the South (Kirui *et al*., 2012; Bosire *et al*., 2008). Nine mangrove species are found in Kenya (Appendix 1), with *Rhizophora mucronata* Lam and *Ceriops tagal (*Perr) C. B. Robinson being the most dominant species constituting 70 % of the total mangrove formation along the coast (Kairo *et al*., 2008). Other species include; *Sonneratia alba* Sm., *Bruguiera gymnorrhiza*, *Avicennia marina* Forsk. Vierh., *Lumnitzera racemosa* Willd., *Xylocarpus granatum* Koen., *Xylocarpus moluccensis* and *Heritiera littoralis* Dryand. In Ailon (Kairo, 2001).

Mangroves of Gazi Bay display the typical zonation pattern of the mangroves in Eastern Africa coastline (Kairo *et al*., 2008). The seaward side is dominated by *Sonneratia-Rhizophora*-giant *Avicennia* community. This is closely followed by *Rhizophora-Bruguiera-Ceriops* in the middle zone and dwarf *Avicennia-Lumnitzera-Xylocarpus* complex on the landward side (Matthijs *et al*., 1999).

#### 1.1.6.2 Value of mangroves in Kenya

Mangrove forests in Kenya provide many direct goods and services that are of economic, cultural and environmental importance to people (Kairo *et al*., 2001). Economically, mangroves provide wood products that are used as firewood, building poles, furniture, fencing posts, boat masts, tannins and dyes and charcoal. 70% of wood requirement by community adjacent the mangroves forest is met by the forest (Wass, 1995). Mangrove wood is also utilized by the local communities for furniture (Dahdouh-guebas *et al*., 2000). Non-timber products derived from mangrove forests include honey, food and medicinal products. Ecologically, mangroves of Gazi provide fishing grounds for the local fishermen (Daudouh-Guebas *et al*., 2000; Abuodha & Kairo, 2001). They also provide cultural services to the communities by providing religious and spiritual sites (Kairo, 2001).

### 1.1.6.3 Threats to Kenyan Mangroves

As in most parts of the world in Kenyan mangroves are also endangered. Overexploitation for wood products by the local community is the main agent of degradation (Bosire *et al*., 2008; Kairo *et al*., 2008). Salt extraction has also led to loss of mangroves whose environmental impacts include hyper salinity in areas close to mangroves leading to their deaths (Abuodha & Kairo, 2001; Kairo *et al*., 2001). Poor land use practices in the hinterland also increase sediment loads into mangrove leading to siltation and eventually death of breathing roots of the mangrove trees (Abuodha & Kairo, 2001). The situation has been reported to have worsened during the 1997/98 El Nino rains that affected the country causing massive death of mangroves in many areas along the coastline, most of which have experienced no recovery up to date (Bosire *et al.*, 2003; Kairo *et al*., 2008). Another threat facing Kenyan mangroves is oil pollution. For instance, between 1983 and 1993 Mombasa port and surrounding waters experienced 391,680 tonnes of oil spills that affected mangroves of Port Reitz and Makupa creeks (Abuodha & Kairo, 2001; Kairo *et al*., 2001). A new threat to mangroves in Kenya is the projected sea-level rise due to climate change (Olagoke *et al*., 2013). Overall degradation of mangroves in Kenya has been estimated to be 18 % for the last 25 years with an average rate of 0.7% per year and has been mostly attributed to overexploitation of mangroves by the local communities (Kirui *et al*., 2012).

#### 1.1.6.4 Restoration and conservation measures in Kenya

Information on earlier mangrove reforestation in East Africa is scanty (Kairo *et al*., 2001). Mangrove reforestation in Kenya dates back to 1918 when Smith and McKenzie Company undertook mangrove planting at Mobore in Lamu, after the forest was clear-felled during the First World War (Kairo *et al.*, 2001). More, recently mangrove restoration efforts has been carried out in Mida creek, Gazi, Mombasa and Ramisi estuary in activities coordinated mostly by KMFRI and other organizations (Bosire *et al*., 2008; Kairo *et al*., 2008). The present study is based on 7.0 ha and 3.0 ha of *R. mucronata* plantations in Gazi Bay, that were established in 1994 and 2001 respectively (Kairo *et al*., 2008). Although a lot of studies have been carried out on mangrove forest structure and productivity in this plantations (Kirui *et al*., 2006; Tamooh *et al*., 2008; Kairo *et al*., 2008) significantly few studies have been done on detailed quantification of total carbon stocks in mangroves. Quantification of the ecosystems total carbon stocks in this study will enable local communities to participate in carbon trade under CDM and REDD+ and be a source of income and reduce pressure on the mangroves.

## 1.2 Problem statement and Justification

Mangroves forests, in spite of their well known values and ecosystem services are least understood in their role in climate mitigations (Bianchi *et al*., 2013; Adame *et al*., 2013; Alemayehu *et al.*, 2014). They function as global carbon sinks (Adame *et al*., 2013; Alemayehu *et al.*, 2014). Few studies have quantified total C stocks in mangrove ecosystem (Adame *et al*., 2013; Sitoe *et al*., 2014; Ajonina *et al*., 2014). Mangrove store large quantities of carbon both in their above and below ground components (Donato *et al.*, 2012). For instance, studies in Thailand have established mangrove carbon stocks of 1400.9 Mg C ha$^{-1}$ (Kridiborworn *et al.*, 2012). Similar values have been reported in Micronesia (Kauffman *et al*., 2011), Central Africa

(Ajonina *et al*., 2014) and Kenya (Bosire, 2012). This carbon is likely to be emitted when mangroves are degraded and is equivalent to 3-19% of all emissions emanating from deforestation globally (Pendleton *et al*., 2012). Inorder to understand the role of mangroves in climate change mitigation , there is a need to understand their carbon sequestration in different mangrove systems (Kauffman *et al*., 2011; Kridiborworn *et al*., 2012). Most of carbon sequestration studies in mangroves have involved natural and degraded stands (Chen et *al*., 2012). There are few examples of similar studies in replanted forests. The situation is even worse in Africa (Sitoe *et al*., 2014; Ajonina *et al*., 2014) with extremely few studies in East Africa and more so in Kenya (Bosire *et al*., 2012). The present study examined carbon stocks and sequestration potentials in mangroves plantations of different age clusters in Gazi bay, Kenya.

## 1.3 Overall objective

To quantify carbon stocks and sequestration potentials of managed mangrove plantations at Gazi Bay, Kenya.

### 1.3.1 Specific objectives

The specific objectives were to

1. Assess forest structure and productivity of replanted mangroves in Gazi
2. Develop allometric equation for estimating biomass of a 19 year old *Rhizophora mucronata* plantation.
3. Determine total carbon stocks and sequestration potentials of replanted and natural *Rhizophora mucronata* stands
4. To determine how belowground biomass varies with nutrient status of the mangrove sediment.

## 1.4 Hypotheses

1. Reforestation may influence structural development of mangrove forests
2. Managed *Rhizophora Mucronata* plantations may contain significant carbon stocks compared to natural stand.
3. Belowground biomass distribution may be dependent on nutrient status of soil.

# CHAPTER TWO

# STUDY AREA, MATERIALS AND METHODS

## 2.0 Study Area

The study was conducted at Gazi bay, situated 50 km south of Mombasa in Kwale County (4˚25´S and 39˚30´E) (Figure 1). The total surface area of the bay is approximately 1800 hectares and it's sheltered from strong waves by the presence of the Chale peninsula to the east and a fringing coral reef to the south (Kirui *et al.*, 2006). Major ecosystems in the bay are mangroves, sea grasses and coral reefs. The area of mangroves in Gazi is about 615ha (Kairo *et al.*, 2001; Kirui *et al.*, 2006). The mangrove forests are not continuously under direct influence of salty water, two rivers that discharge into the bay, Kidogoweni in the North and Mkurumuji in the south.

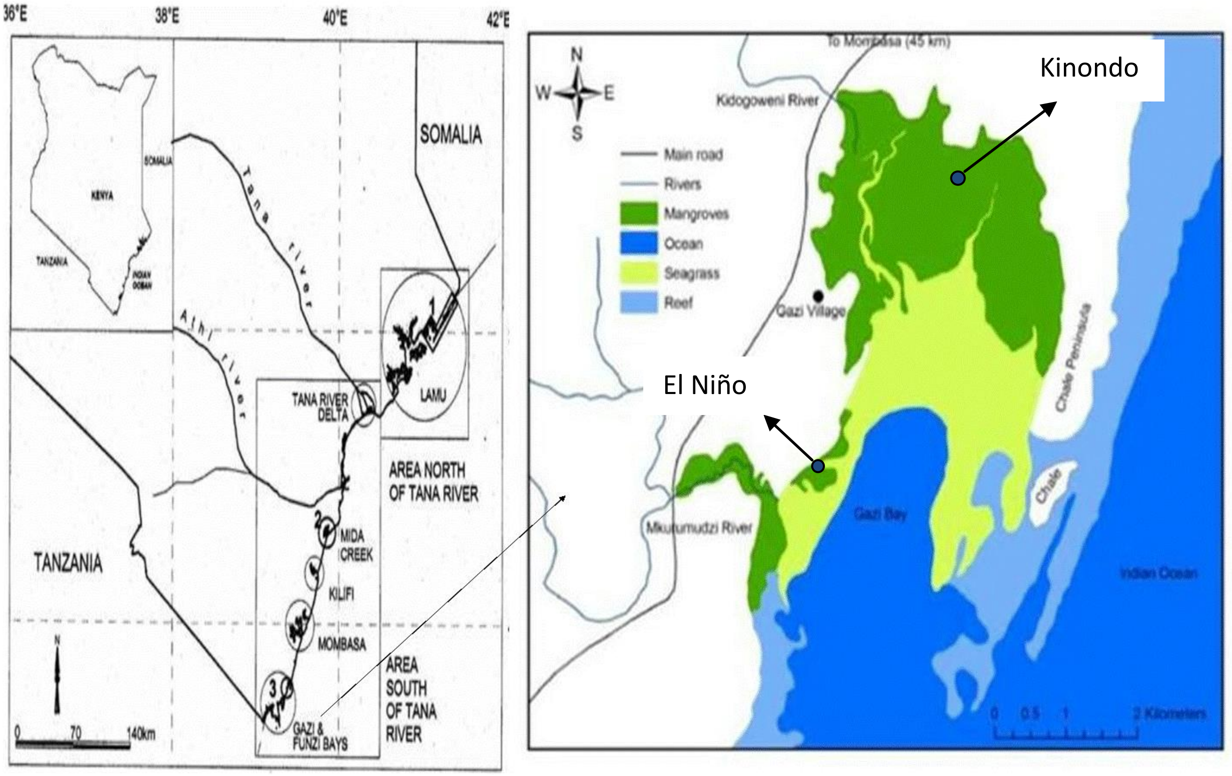

**Figure 1**: Map of Gazi Bay, Kenya, where the study was conducted. (Source; KMFRI).

## 2.1 Study Site

The study was carried out in two *Rhizophora mucronata* plantations aged 19 and 13 years old in Kinondo and Gazi respectively and a natural stand in Kinondo dominated by *R. mucronata*. The 19 year old stand was established in April 1994 as a 7.0 ha mono-culture of *Rhizophora mucronata*. The site in Kinondo had been clear felled in the 1970's to provide industrial fuelwood. The plantation was established with an initial spacing of 1x1 m, giving an initial stock of 70,000 *R. mucronata* saplings.

The 13 years plantation is also a monostand of *Rhizophora mucronata* having been established following dieback caused by 1997/98 rains El Niño (Kairo *et al*., 2001).The plantation is located close to the fish landing site of Gazi bay. About 20,000 propagules were established in a total area of approximately 3.0 ha.

For the two plantations, maintenance and monitoring has involved thinning and pruning aftere every 5 years (Bosire *et al*., 2003). Past studies, particularly, in the older plantation have been carried out on secondary successions (Bosire *et al*., 2003), biomass accumulation (Kairo *et al*., 2008), and mangrove phenology (Wang'ondu *et al*., 2010). The present study aimed at determining total carbon stock and sequestration levels of the plantations; compared with natural controls. The study will complement the activities of Mikoko Pamoja, a small scale mangrove carbon offset project in Gazi bay. The project aims to continue participating in carbon markets by promoting sustainable development by improving community livelihoods and ecosystem benefits through reforestation and afforestation activities.

## 2.2 Sampling design

Systematic sampling design was used for the plantations while in the natural stand characterised by heterogeneity, stratified random sampling was more applicable. Starting points in a transect line were established randomly by use of random numbers generated using a scientific calculator. The transect lines were established perpendicularly to the shoreline in each of the plantations. Quadrats of 10m × 10m were then systematically established along the transect line after every 20m and 12 quadrats selected for the study in the 19 year old Kinondo plantation. Similarly, equal numbers of quadrats in a natural stand were selected for comparison purposes. In the 13 years old El Niño plantation only 4 quadrats were selected.

### 2.2.1 Forest structure

Inside the 10 by 10 m plot, trees with a diameter at breast height (DBH) > 2.5cm were identified, counted and their height and stem diameter determined. Height was determined by use of graduated poles while diameter was measured by use of forest callipers. From the measurements, stand density (stems/ha), basal area (m$^2$/ha) and distribution patterns were calculated. Relative derivatives of density, dominance and frequency were also calculated from which the importance value (IV) was determined. Basal area (m$^2$) was calculated. The following formulas were used.

Basal area = л/4 dbh$^2$ = 0.00007854 $D_{130}^2$

Density per hectare (Stand density) = (No. of stems in plots x 10,000)/Area of the plot

Relative density = (Number of individuals of a species/total number of individuals) × 100

Relative dominance = (Total basal area of a species/Basal area of all species) × 100

Relative frequency = Frequency of a species/sum frequency of all species) × 100

### 2.2.2 Mangrove regeneration

Nested sampling was used to sample regeneration. 5 m by 5 m subplots within the 10 × 10 m were established and species and abundances of juveniles identified, recorded and grouped into three regeneration classes based on height, Regeneration Class I (< 40 cm), Regeneration Class II (40-150 cm) and Regeneration Class III (1.5-3.0 cm but with a diameter of less than 2.5 cm. The ratio of regeneration classes was then determined

### 2.2.3 Development of biomass allometric equation

Twenty five *Rhizophora mucronata* trees with a stem diameter ≥ 2.5 cm were harvested in the 19 year old Kinondo plantation. For all harvested trees, diameter at ground level ($D_o$) at 30 cm above ground level ($D_{30}$) cm, at 130 cm above the ground level ($D_{130}$) and height were measured. Height of each harvested tree was measured in the field by use of distance tape. The harvested trees were partitioned in to their component parts; the stem (trunk), branches, leaves and stilt roots, and the wet weight of each component weighed in the field.

A sub-sample of 500g for each tree component was weighed and oven-dried at $85^o$ C to constant weight in order to obtain wet to dry weight ratios (conversion factors). The wet weight of each tree component was converted to dry weight using the corresponding conversion factor and summed up to obtain the total dry weight of the tree. Regressions between the structural variables ($D_{130}$ and height) and the dry weight of each tree component and total dry weight of the tree were computed to derive allometric equations. Stem diameter above stilt root was used as the basis on which the data was arranged in ascending order. The Allometric relationship between $D_{130}$, $D_{130}{}^2$, $D_{130}H$ and $D_{130}H^2$ against biomass of different tree components were derived and their effectiveness the best predictor assessed by evaluation $R^2$ and SEE parameters.

### 2.2.4 Development of volume allometric equation

The harvested trees used for derivation of biomass equation were also used to develop allometric equation for estimation of stand volume. Both merchantable and non-merchantable volumes were estimated. The merchantable stem was divided into 1-m long billet till a top diameter of 2.5

cm is reached. The diameters of both ends of each billet were measured to the nearest 0.1 cm and the volume of each billet was calculated using the Smalian's formula (FAO, 1994);

$V= (D_1^2 + D_2^2/)/2 \text{ x } \pi/4 \text{ x } L$

Where;

V = volume

$D_1$ and $D_2$ are bottom and top diameters of the billet respectively

L is the billet length and $\pi = 3.14$.

The volume of the stem section above the top 2.5 cm diameter (stem tip) was calculated using the formula below,

$V = \frac{1}{3}AL$,

Where;

A and L are bottom end cross-sectional area and length of the stem tip, respectively.

A Sub-sample of 500g stilts roots, leaves and branches were submerged in water so as to determine their volumes. Volume regression equations were then developed so as to estimate the merchantable and non-merchantable volume in case of *R. mucronata*.

### 2.2.5 Carbon pools

#### 2.2.5.1 Aboveground carbon

The aboveground biomass (AGB) in the 19 year old plantation was estimated by applying derived equations to all the individuals in the plots. For the species encountered in the natural and 13 year old plantation, a general equation by Komiyama *et al.*(2005) was used.

$AGB = 0.251\rho D^{2.46}$

Where;

AGB= Tree Aboveground biomass (kg),

ρ= wood density (g/cm$^3$)

D = tree diameter at breast height (cm).

Carbon content was calculated by multiplying the biomass content of the mangrove tree by its specific carbon concentration using a default value of 0.5 (Kauffman & Donato, 2012; Mcleod & Salm, 2006). Wood density values by Bosire *et al* (2005) were used in calculating for tree biomass.

**Table 1**: Specific densities for different mangrove species used during calculation of biomass

| Mangrove species | Wood density (g/cm$^3$) |
|---|---|
| *R. mucronata* | 1.1 |
| *A. marina* | 0.9 |
| *B. gymnorrhiza* | 1.3 |
| *C. tagal* | 1.1 |
| *X. granatum* | 0.8 |

(Source; Bosire 2005).

### 2.2.5.2 Dead standing and dead downed wood

Trees that were dead and standing were recorded as such. Measurements similar to those of live trees were undertaken. Decay status for standing dead tree outline in Kauffman *et al.,* 2011 was used to determine the degree of decomposition of the tree. Status I included trees resembling live tree except for the lack of leaves. The allometric equation for a live tree was used to estimate the biomass of dead standing wood and then leaf or branch biomass which is 2.5% of the plant biomass subtracted to obtain the standing dead tree biomass (Pearson & Brown, 2005). Status II comprised of trees with branches and no twigs or leaves while status III comprised of a tree with standing stem only. Volume estimated using formulae for calculation of cylinder (Kauffman *et al.,* 2011). Biomass was the then determined by multiplying volume and density. Amount of carbon was determined by multiplying the amount of tree biomass by its specific carbon concentration of 0.5 or 50% of the total biomass (Kauffman & Donato, 2012).

Laying dead wood was sampled using the plot method which involves collection of all dead debris from selected plots from which density and mass are determined for the determination of biomass and consequently carbon (Kauffman & Donato, 2012). All dead wood in a plot was collected and weighed but only coarse dead wood with diameter of more than 2.5 cm was measured and that with diameter of less than 2.5 was considered as litter.

All laying wood debris in a plot was collected, assigned into three density classes namely: sound, intermediate and rotten. To determine what density class a certain piece of dead wood belonged to, each piece of wood was struck with a machete. If the blade sunk and bounced back, it was classified as sound. If it partly sunk and then there were some wood losses, it was classified as intermediate. If the blade sunk into the piece of wood with extensive wood loss it was classified as rotten (Kauffman *et al.*, 2011). To determine specific gravity of downed wood in each density class, a complete disc weighing 500g obtained, weighed and its volume determined. Volume was calculated using formulae of a cylinder assuming laying dead wood is a cylinder or a frustum (Teshome, 2005). The formula below was used to calculate volume and biomass of dead wood. Specific gravity was then determined as oven-dried weight divided by its volume.

Volume = $\pi$ x (average diameter/2)$^2$ x average length of the fresh sample.

Biomass was calculated using the formulae below;

Biomass (t/ha) = Volume x Density.

Downed wood was converted to C using a factor of 0.50 as recommended by (Kauffman & Donato, 2012).

### 2.2.5.3 Belowground biomass

Belowground biomass and necromass was estimated using Saintilan (1997) modified coring method. Each 10 m × 10 m plot was sub-divided into four equal quadrants from which four cores were taken. A total of 48 cores were collected for both the 19 year old Kinondo plantation and natural stand while 16 cores were collected for the 13 year old El Niño plantation. A core was made by vertically inserting a hollow metallic cylinder of 60 cm length and a diameter of 14 cm into the soil until the 1st mark (0-20 cm) at the lower end of the core was at level with the soil surface. This was repeated until all the depth profiles (0-20 cm, 20- 40 cm and 40-60 cm) were sampled. Each soil core was put in a stainless steel sieve with a mesh size of 1mm and washed to remove the mud. Roots that were brown in color were classified as live roots while those that were black were classified as dead and separated. Necromass comprised of fine roots (< 2 mm). Fresh roots and fine root necromass was weighed and then oven dried at $80^0$ C to a constant weight and then reweighed to obtain biomass. The amount of carbon then calculated in relation to the cross sectional area occupied by a single core and the results per plot obtained were pooled to obtain root carbon per hectare.

#### 2.2.5.4 Soil carbon

Soil carbon was sampled using protocols developed by Kauffman *et al.* (2012). After removing the top litter, the soil corer was inserted vertically into the soil until the top of the soil corer was level with the soil surface and then gently pulled out. The corer used was 1 m long and with a diameter of 5 cm. Once the soil corer was extracted, a ruler was used to determine the appropriate depth from which the soil sample was then obtained. Soil was sampled at depths of 0-15 cm, 15-30 cm, 30-50 cm and 50-100 cm and subsamples collected at the approximate mid-point of each sample depth. Upon collection, samples were placed in a labeled polythene bag with the site, date, plot number and soil depth. Soil samples collected were then transported to the laboratory where they were weighed and oven dried at $60^0$ C after which they were reweighed and two parameters namely; soil bulk density and soil organic carbon were determined. Soil bulk density was calculated as the ratio of soil dry weight to the soil corer volume and it's normally an indicator of soil compaction. Soil organic matter (SOM) was determined using loss-on-ignition (LOI), a quantitative method based upon the indiscriminant removal of all organic matter. For maximum efficiency, samples analyzed for bulk density were used. In the determination of organic carbon, the oven-dried samples were homogenized by grinding to a fine powder using a mortar and pestle, passed through a 2 mm sieve and placed in pre-weighed aluminum crucibles. 5g of the sample was obtained and set in a muffle furnace for combustion at 440˚C for 8 hours until only inorganic ash is left, after which it was cooled in dessicator and weighed. What was lost during oxidation represented the soil organic matter and SOM was multiplied by default value of 0.58 to obtain soil organic carbon (Kauffman & Donato, 2012).

Bulky density was calculated as;

$$\text{Soil Bulk Density} = \frac{\text{Oven} - \text{dry oven sample}}{\text{Sample Volume m}^3}$$

Soil organic matter was calculated as;

$$\text{Soil organic matter} = \frac{\text{Initial weight(g)} - \text{Final weight}}{\text{Initial weigh}}$$

To obtain soil organic carbon per hectare bulk density (g cm$^3$) was multiplied by soil depth interval (cm) and percent carbon expressed as a whole number.

### 2.2.6 Total carbon stock

The total carbon stock or pool was estimated by summing up all the component pools. First, each of the component pools was averaged across all plots and in each study site. The averaged values were then summed together to obtain total carbon pool.

Total Carbon stock per hectare = CtreeAG+ CtreeBg + Cwoodydebris+C soil

Where;

C = Carbon

AG = Aboveground carbon in live standing per hectare

Woody debris = dead standing and dead fallen trees per hectare

Soil = Belowground carbon in soil per hectare

BG = Root carbon per hectare.

## 2.3 Nutrients analysis

In this study, two nutrients were determined, which are, nitrogen in the form of $NO_3^-$ and $NH_4^+$ as well as phosphorus in the form of Phosphates ($PO_4^{3-}$). Soil samples were collected in the 0-20 cm, 20- 40 cm and 40-60 cm depth profiles using soil corer. After collection, they were put in polythene paper bags and stored in cooler boxes for transfer into the laboratory for analysis. For samples that could not be analyzed immediately, they were stored at $-4^0$ C which allows longer storage without any significant change in the content of the nutrient content. Procedure for soil nutrient analysis outlined below was used (Parsons *et al.*,1984)

### 2.3.1 Nitrates determination

10 grams of oven dried, ground and sieved soil was weighed and placed in a 150 ml conical flask. 0.1 g of Calcium sulphate dihydrate and 50 ml distilled water was then added. The mixture was shaken for ten minutes in a rotary shaker and then filtered using filter paper. The aqueous extract was serially diluted with distilled water to contain approximately 0 to 10 Ppm nitrate in the final volume based on colour change. Using scoop, 0.5 g of calcium sulphate dihydrate powder was added. The test tube was shaken immediately and the colored solution centrifuged and decanted after ten minutes. The colour intensity of the solution was read at 650 nm and compared to the standard curve of nitrate solution.

### 2.3.2 Ammonium determination

10 g of oven dried, ground and sieved soil sample was weighed and placed in the volumetric flask. Potassium chloride solution in the ratio of 2:1 was added and the mixture placed in a rotary shaker for I hour at room temperatures. The solution was filtered by use of filter paper and 5 ml of the filtrate pipetted in glass test tube. Ammonium ions in the filtrate were converted into ammonia by in-line neutralization with a concentrated buffer (6.5 g of sodium dissolved in potassium tartrate and 26.8 g of Sodium phosphate heptahydrate in deionized water). The concentrated ammonium filtrate was then heated with salicylate and hypochlorite to produce a blue colour which was read at 850nm and compared to standard to actual concentration.

### 2.3.3 Phosphate determination

10 g of the ground and sieved soil was weighed and placed in a 100 ml volumetric flask containing distilled water. The mixture was shaken for five minutes, centrifuged at 150 rpm and filtered into 25 ml by use of filter paper. Duplicates of 50 ml aliquots were used. For Ammonium-molybdate volumetric determination, 25 aliquots of the soil solution was transferred to 100ml volumetric flask and enough distilled water added to bring about 70 ml volume. 10 ml of 2.5% Ammonium-molybate and 4 ml of 0.25% Aminonapthosulfonic acid added to both soil extract and distilled water. If Phosphates are present, Phosphomolybidic acid is formed which is immediately reduced to form a blue color which stable for one hour. As little as 0.005 g of Phosphorous as phosphate has been known to give a distinct color in a volume of 100ml. The concentration was read using spectrophotometric machine at 850.

## 2.4 Data analysis

Data collected was analyzed using SPSS version 20 and Microsoft Excel spreadsheet statistical packages. Logarithmic transformation was carried out on soil and nutrients data. One way

ANOVA was used to determine if there was any significant difference s between carbon stocks, nutrient concentrations between sites and within blocks in Kinondo plantation. Whenever a significant difference ($P \leq 0.05$) was encountered, post hoc was carried out to determine the potential source of differences. Correlation analysis was carried out to determine if the BGB concentration was dependent on nutrient availability.

# CHAPTER THREE

# RESULTS

## 3.0 Structural characteristics

The structural characteristics for the plantation and natural stand are shown in the table (2) below. The mean height and stem diameter for 19 year old *R. mucronata* plantation in Kinondo was 11.6±1.7 m and 10.05±2.5 cm respectively. While mean height and stem diameter for El Niño plantation were 4.5±0.8 cm and 5.2±1.2 cm respectively. In the natural stand, mean height and stem diameter was 5.3±0.1 cm and 7.2±0.2 cm respectively. Based on importance values (IV), *R. mucronata* was the principal species both in the plantations and the natural stand.

**Table 2**: Structural characteristics of plantations and the Natural stand of *R mucronata* in Gazi Bay, Kenya (*N= Number of individual per species, IV = Importance value, x + s.d = Mean + standard error)

| | | | | | **Relative values in %** | | | |
|---|---|---|---|---|---|---|---|---|
| **Study Site** | **Species** | **N** | **Mean Height(m) (X+ s.d)** | **Basal Area (m ha$^{-1}$)** | **Density** | **Dominance** | **Frequency** | **IV** |
| El Niño | *R. mucronata* | 255 | 5.2±1.2 | 12.8 | 100 | 100 | 100 | 300 |
| Natural | *A. marina* | 66 | 6.2±1.5 | 27.8 | 8.4 | 13.2 | 5.9 | 27.9 |
| | *B. gymnorrhiza* | 93 | 5.4±2.3 | 12.5 | 11.8 | 13.2 | 17.7 | 42.7 |
| | *C. tagal* | 136 | 3.9±1.3 | 23.5 | 17.2 | 6.6 | 23.5 | 47.3 |
| | *R. mucronata* | 573 | 5.40±2.9 | 34.5 | 72.6 | 64.4 | 35.3 | 172.3 |
| | *X. granatum* | 11 | 6.2±1.2 | 5.9 | 1.4 | 3.5 | 5.9 | 10.7 |
| Kinondo | *B. gymnorrhiza* | 13 | 7.3±2.7 | 0.7 | 3.6 | 0.9 | 19.1 | 23.6 |
| | *C. tagal* | 3 | 5.3±1.9 | 0.2 | 0.8 | 0.2 | 14.3 | 15.3 |
| | *R. mucronata* | 346 | 11.6±1.7 | 28.4 | 94.5 | 97.6 | 57.1 | 249.2 |
| | *X. granatum* | 4 | 9.7±1.7 | 1.9 | 1.1 | 1.2 | 9.5 | 11.8 |

The overall basal areas for *R. mucronata* in Gazi and Kinondo plantations were 12.8 m$^2$ and 25.9 m$^2$ respectively while that of Natural stand was 18.1 m$^2$. Tree height, basal area and stem diameter showed significant difference between the plantations and the natural stand (H; 2,789, =

65.8, $P < 0.05$). Tree height, basal area and diameter were significantly larger in Kinondo plantation than in natural stand (Tukey test, $P < 0.05$).

The 19 year old Kinondo plantation was established as a monostand but there was recolonization by other mangrove species such as; *Bruguiera gymnorrhiza, Ceriops tagal and Xylocarpus granatum*. *B. gymnorrhiza* was the second most important or principal species after *R. mucronata*. Total stem density was 5650 stems ha$^{-1}$ and 3725 stems ha$^{-1}$ for Gazi and Kinondo plantations respectively. In the Natural stand, the total stem density was 2545 stems ha$^{-1}$ of which *A. marina, G. gymnorrhiza, C. tagal, R. mucronata* and *X. granatum* contributed 2.6%, 3.7%, 5.3%, 22.5% and 0.4% respectively.

Size class distribution *for R. mucronata* plantations followed a normal distribution curve while the Natural stand had a reversed J-shaped curve typical of natural stands (Figure 2–4). Size classes, 4.1-5, 11-13 and <5 cm in El Niño Gazi, Kinondo and Natural stands had the highest stem densities of ,450 stems ha$^{-1}$, 742 stems ha$^{-1}$ and 1374 stems ha$^{-1}$ respectively.

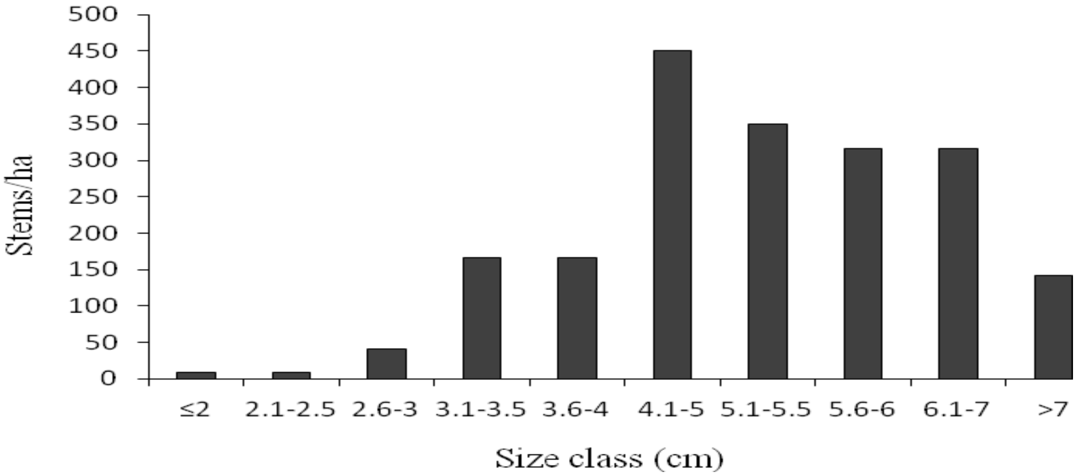

**Figure 2**: Size class distribution for 13 year old El Niño plantation

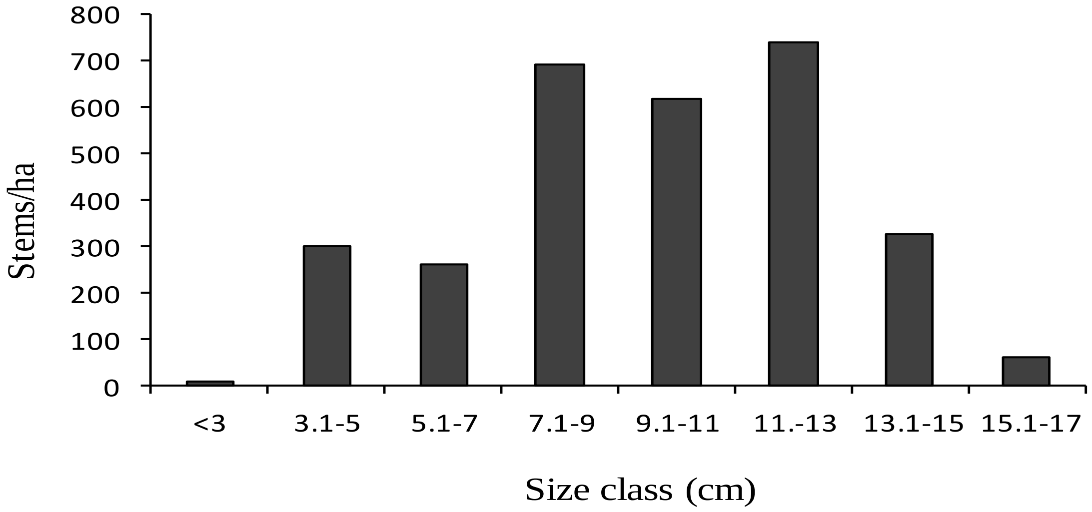

**Figure 3:** Size class distribution for 19 year old Kinondo plantation.

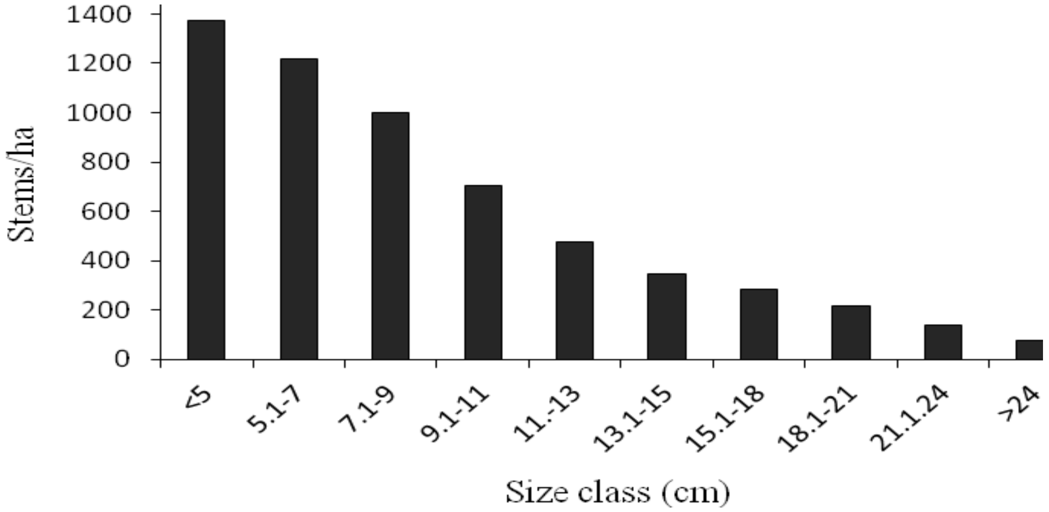

**Figure 4**: Size class distribution in Natural stand

Figures (5) below shows scatter grams for each of the study site. Fifty percent of the trees stem diameter in the 19 year old Kinondo plantation (Figure 5-A) show that majority of the trees in the stand were in the diameter class of between 11.5-13.5 cm and a height ranging between 7.1-15 m. In the El Niño (Figure 5-B) ranged between 4.1- 7 cm for diameter and height of between

4-7.5 m. In the Natural ranges were between 5- 13 cm for diameter and 3-9 m height.(Figure 5-C).

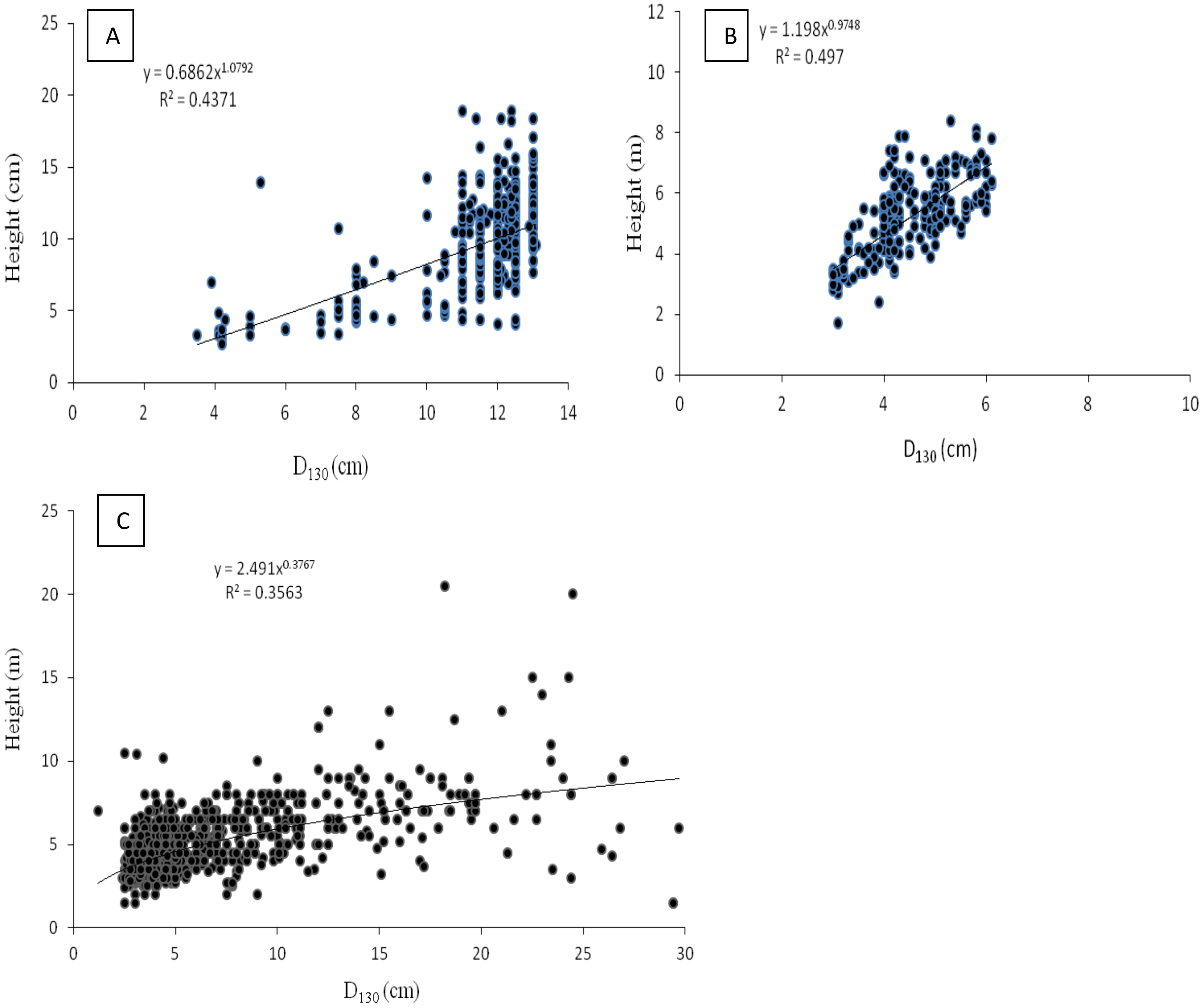

**Figure 5**: Height-diameter scatter-grams for 19 year old Kinondo plantation (A), 13 year old El Niño plantation (B) and Natural stand (C). of *R. mucronata* in Gazi Bay, kenya

## 3.1 Natural regeneration

Recolonization varied among sites and mangrove species (Table 3). Juvenile species comprising of *R. mucronata* and *C. tagal* were encountered in Eli Niño plantation whereas in the 19 year old Kinondo plantation juvenile species comprised of *R. mucronata, X. granatum* and *B. gymnorrhiza*. In the Natural stand, five juvenile species comprising of *A. marina, B. gymnorrhiza, C. tagal, R. mucronata* and *X. granatum* were encountered. El Niño plantation had an overall density of 950 juveniles/ha with *R. mucronata* contributing 95% of the total density while *C. tagal* contributed the remaining 5%.

**Table 3**: Regeneration potential in percentages for juveniles of *R. mucronata* in El Niño, Kinondo and Natural stands in Gazi Bay, Kenya (* = juveniles/ha).

| **Study site** | **Species** | **RCI*** | **RCI%** | **RCII*** | **RCII%** | **RCIII*** | **RCIII%** | **Total*** | **Total %** |
|---|---|---|---|---|---|---|---|---|---|
| El Niño | *C. tagal* | 0 | 0 | 25 | 20 | 20 | 25 | 50 | 5 |
| | *R. mucronata* | 725 | 100 | 100 | 80 | 75 | 75 | 900 | 95 |
| Total | | 725 | 76 | | 13 | | 11 | | 100 |
| Kinondo | *B. gymnorrhiza* | 175 | 5 | 75 | 39 | 0 | 0 | 250 | 7 |
| | *C. tagal* | 25 | 0. 95 | 25 | 13 | 25 | 11 | 75 | 2 |
| | *R. mucronata* | 3083 | 94 | 92 | 48 | 208 | 89 | 3406 | 91 |
| | Total | 3285 | 89 | 195 | 5 | 235 | 6 | 370 | 100 |
| Natural | *A. marina* | 4640 | 24.9 | 4880 | 27.9 | 5880 | 33.3 | 14410 | 28.4 |
| | *B. gymnorrhiza* | 1869 | 10.1 | 1508 | 8.7 | 1311 | 8.9 | 4688 | 9.3 |
| | *C. tagal* | 3345 | 18 | 2400 | 13.8 | 2436 | 16.6 | 8181 | 16.1 |
| | *R. mucronata* | 7244 | 39 | 7733 | 44.3 | 4622 | 31.5 | 19599 | 36.7 |
| | *X. granatum* | 1475 | 7.9 | 918 | 5.3 | 1410 | 9.6 | 3803 | 7.5 |
| | Total | 18739 | 36.7 | 17439 | 34.4 | 14659 | 28.9 | 50689 | 100 |

Kinondo had an overall juvenile density of 3708 per hectare, which comprised of 91.2% *R. mucronata*, 2.1% *C. tagal* and 4.9% *B. gymnorrhiza*. In the natural stand juvenile density was 28,109 saplings per hectare comprising of 11.6% *A. marina*, 9.3% *B. gymnorrhiza*, 14.6% *C. tagal*, 57.1% *R. mucronata* and 7.5% *X. granatum*. In the three study sites, most of the juveniles were of regeneration class I (Table 3). In El Niño plantation regeneration Class III had the least number of juveniles while in Kinondo and Natural stand least number of saplings were of Regeneration Class II. The regeneration ratio; RCI: RCII: RCIII was 29: 5: 4, 73: 4: 5 and 11: 10: 8 for Gazi, Kinondo and Natural stands respectively.

There was no significant difference in regeneration class I abundance between the three study sites ($F_{2, 51} = 1.527$, $P > 0.05$). There was also no difference in abundance in RCII between the sites ($F_{2, 50} = 3.055$, $P > 0.05$). There was however, a significant difference in abundance of RCIII between the sites ($F_{2, 50} = 19.2$, $P < 0.05$). There was a significant difference in regeneration density of RCIII between the 13 year old El Niño plantation and Natural stand (Tukey test, $P < 0.05$) and between the 19 year old Kinondo and Natural stand (Tukey test, $P < 0.05$). There was no significant difference between the two plantations in the abundance of RCIII (Tukey test, $P > 0.05$).

## 3.2 Biomass allometric equation

Power function and polynomial functions of were tested during development of the equations. A polynomial equation (2nd order) in of the form; $y = ax^2 + bx + c$ (*where*; y = biomass, x = DBH alone and a, b and c are constants) was used. Model A used diameter alone, model B used quadratic diameter and height, model C used quadratic diameter while model D used diameter and quadratic height. Coefficient of determination using the tree diameter ($D_{130}$) were found to be 0.91, 0.86, 0.85, 0.85, 0.78 and 0.68 with total AGB, total volume, stem biomass, stem biomass, branch biomass, leaf biomass and root biomass with standard errors of 6.4, 9.3, 11.1, 11.8, 13.5 and 14.8 respectively (Table 4 and figure 6). Regression coefficients using a combination of diameter squared and height with total AGB, total volume, stem biomass, stem biomass, branch biomass, leaf biomass and root biomass were 0.89, 0.89, 0.86, 0.77 and 0.74 with standard errors of 7.6, 11.3, 12.7, 13.8, 14.3 and 15.3 respectively (Figure 6).

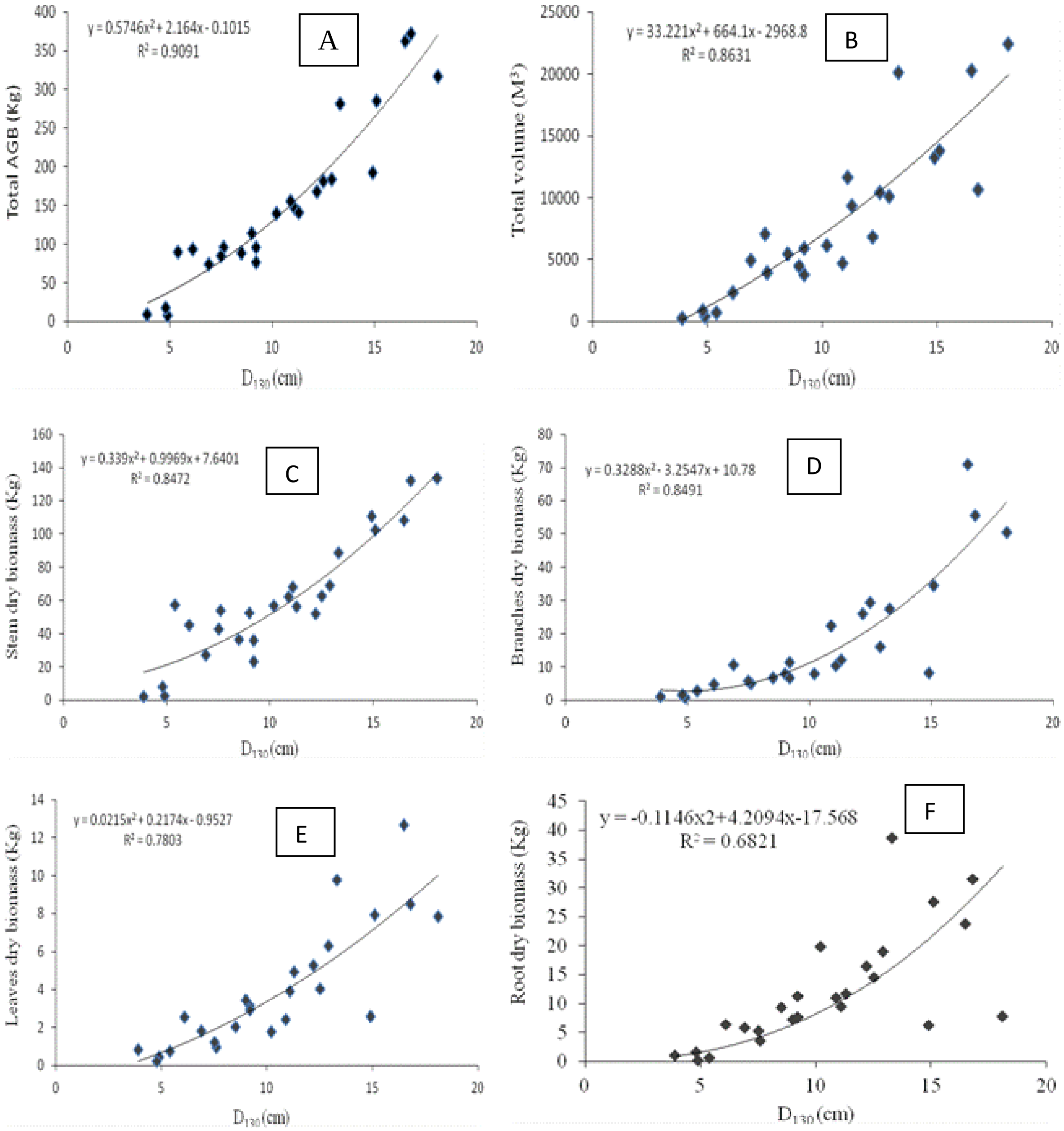

**Figure 6:** Allometric relationship between stem diameter ($D_{130}$) and (A) total AGB, (B) total volume, (C) stem, (D) branches, (E) leaves and (F) root biomass respectively of *R. mucronata* in Gazi Bay.

**Table 4:** Allometric equations for estimating biomass in the 19 year old *R. mucronata* plantation in Kinondo, Gazi Bay, Kenya.

| Model | X | a | b | c | $R^2$ | SEE | N | P value |
|---|---|---|---|---|---|---|---|---|
| A | $D_{130}$ | 0.5746 | 2.164 | -0.1015 | 0.9091 | 6.4 | 25 | $<0.05$ |
| B | $D_{130}^2$ | 0.0602 | 0.7192 | 7.2847 | 0.9083 | 7.6 | 25 | $<0.05$ |
| C | $D_{130}^2H$ | 3E-06 | 0.0502 | 17.542 | 0.8862 | 11.8 | 25 | $<0.05$ |
| D | $D_{130}^2H^2$ | -3E-08 | 0.0046 | 13.099 | 0.8697 | 12.5 | 25 | $<0.05$ |

Comparatively the $r^2$ and standard error of stem diameter against other tree variables were higher than those found for $D^2H$ against tree variables. The dry weight of the trees was estimated using polynomial function of second order using the stem diameter as the independent variable. Table (4) summarises the standard error, significance values, regression coefficients for all models developed. When the stem diameter was substituted with quadratic diameter, (Square diameter multiplied by height) the $r^2$ values of the subsequent equation declined (Table 4). Model A (Table 4) was found to be superior to other models bases on evaluation parameters and therefore used to estimate ABG in the 19 year old Kinondo plantation.

## 3.3 Volume allometric equation

Allometric relationship between $D_{130}$, $D_{130}^2$, $D_{130}H$ and $D_{130}H^2$ against volume of different tree components were derived (Figure 7). Coefficient of determination ($r^2$) when using the tree diameter ($D_{130}^2H$) were found to be 0.97, 0.97, 0.83, 0.80 and 0.72 with total volume, stem volume, branch volume, leaf volume and root volume with stand errors of 2.1, 2.1, 0.008, 0.006 and 0.01 respectively (Table 5). Coefficient of determination when using the tree diameter alone ($D_{130}$) were found to be 0.95, 0.95, 0.84, 0.81 and 0.62 with total volume, stem volume, branch volume, leaf volume and root volume with stand errors of 1.001, 1.001, 0.0035, 0.0027 and 0.001(Table 5) respectively.There was no significant difference between merchantable (stem volume) and total volume, ($F_{,\ 1,\ 24}$= 85.9, p= 0.057) and hence similar $r^2$ values when the two were regressed against DBH. Volume was estimated Model A which had the highest $r^2$ value and smaller SEE while taking into consideration that height is a difficult parameter to measure in the mangroves and is mostly estimated. The stand volume was 182.5 ±13.00 $m^3$/ha for the 19 year old *R. mucronata* plantation.

**Table 5:** Equations derived for volume determination in the 19 year old *R. mucronata* plantation in Kinondo

| **Component** | **Model** | **N** | **X** | **$r^2$** | **SEE** | **P value** |
|---|---|---|---|---|---|---|
| Total volume | A | 25 | $D_{130}$ | 0.95 | 0.026 | <0.05 |
| Total volume | B | 25 | $D_{130}^2$ | 0.95 | 0.028 | <0.05 |
| Total volume | C | 25 | $D_{130}^2H$ | 0.97 | 0.024 | <0.05 |
| Stem Volume | D | 25 | $D_{130}^2H$ | 0.97 | 0.025 | <0.05 |
| Branch volume | D | 25 | $D_{130}^2H$ | 0.83 | 0.008 | <0.05 |
| Leaf volume | D | 25 | $D_{130}^2H$ | 0.80 | 0.007 | <0.05 |
| Stilt root volume | D | 25 | $D_{130}^2H$ | 0.72 | 0.010 | <0.05 |

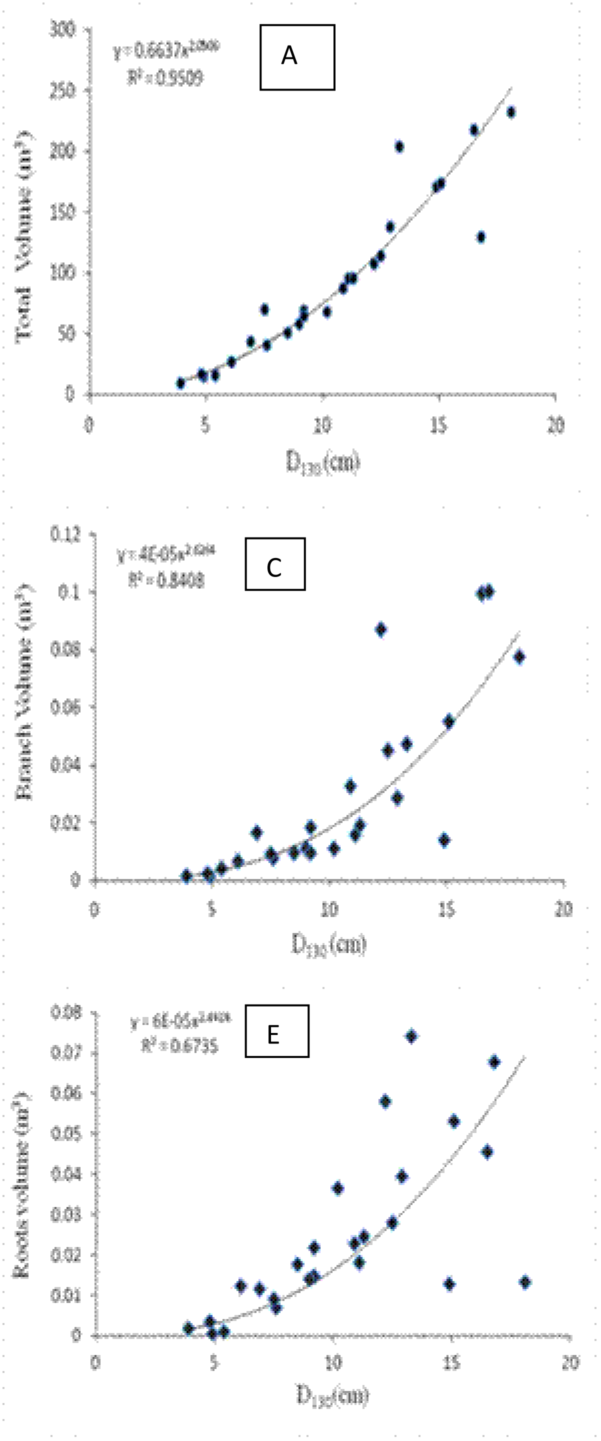

**Figure 7:** Allometric relationship between stem diameter (cm) and (A) total volume, (B) stem volume, (C) branch Volume, (D) leaves volume and (E) root volume respectively. of *R. mucronata* in Gazi Bay, Kenya.

## 3.4 Carbon pools

### 3.4.1 Aboveground

Aboveground biomass in El Niño, Kinondo and natural stands was estimated at 74.2 ± 11.6 t/ha, 196.5 ± 2 t/ha and 335.5± 71 t/ha respectively. In Kinondo and Natural stand where other species were encountered, biomass contribution varied as shown in Table 4 below. *A. marina* contributed the second highest biomass after *R. mucronata* in the Natural stand contributed while in the 19 year old Kinondo plantation, *B. gymnorrhiza* was second highest biomass contributor after *R. mucronata*.

**Table 6**: ABG biomass contributions by different species encountered in 19 year old Kinondo plantation and Natural stand in Gazi Bay, Kenya.

| **Site** | **Species encountered** | **ABG t/ha** | **% contribution** |
|---|---|---|---|
| Natural stand | *A. marina* | 85.96 | 26% |
| | *B. gymnorrhiza* | 26.38 | 9.3% |
| | *C. tagal* | 10.2 | 3% |
| | *X. granatum* | 57.78 | 17.1% |
| | *R. mucronata* | 155.24 | 44.4% |
| **Total** | | **335.5** | **100%** |
| 19 year old Kinondo | *B. gymnorrhiza* | 77.06 | 39.2 |
| | *C. tagal* | 0.14 | 0.1 |
| | *X. granatum* | 1.44 | 1.2 |
| | *R. mucronata* | 116.82 | 59.5 |
| **Total** | | **196.5** | **100%** |

Assuming 50% of biomass is carbon; the average above ground carbon for Kinondo plantation was estimated at 98.2 ± 7.1 Mg C ha$^{-1}$ and ranged between 47.5-138.8 Mg C ha$^{-1}$. El Niño plantation had a mean aboveground carbon of 37.2 ± 5.8 Mg C ha$^{-1}$ ranging between 27.3-53.2 Mg C ha$^{-1}$ while the Natural stand above ground biomass carbon averaged at 167.9± 35.5 Mg C ha$^{-1}$ and ranged between 37.5- 349.65 Mg C ha$^{-1}$ with three plots (1, 3 and 11) which had the highest stem density contributing over 50% of the total AGB carbon. There was a significant difference in aboveground carbon among different sites (F; $_{2,33}$= 5.09, $P < 0.05$).

There was a significant difference between the Natural stand and El Niño plantation ($P < 0.05$, Tukey test). Aboveground carbon was found to be more in the Natural stand than in the 13 year old El Niño plantation. In the Natural stand, five mangrove species were encountered and included *A. marina, B. gymnorrhiza, C. tagal, R. mucronata* and *X. granatum* and contributed 26.3%, 9.3%, 3%, 44.4% and 17.1% of the total biomass carbon respectively (Table 6). There was a significant difference on aboveground biomass carbon among different magrove species encountered in Natural and Kinondo plantation (Natural, F, 4, 333,= 48.92 , $P < 0.05$: Kinondo plantation, F; 3, 149, = 34.8, $P < 0.05$). There was a significant difference between *R. mucronata* and other species ($P < 0.05$, Tukey test).

### 3.4.2 Dead organic matter

Dead wood was sampled for the 19 and 13 year old plantations but only the older plantation had standing and fallen dead wood. Dead wood carbon in the 19 year old Kinondo plantation was 7.8± 0.67 Mg C/ha.

### 3.4.3 Belowground biomass

Gazi El Niño recorded the lowest root carbon concentration of 56.2± 2.5 Mg C $ha^{-1}$ in 0-60 cm depth profile compared to 83.8±4.5 Mg C $ha^{-1}$ and 66.3 ±2.5 of the same depth in natural stand and 19 years old Kinondo plantation respectively, (fig 8). There was a significant difference in total root carbon concentration among the three study sites ($F;_{2, 69} = 89.35$, $P < 0.05$). There was significant difference between the plantations and the natural stand ($P < 0.05$, Tukey test). There was a significant difference in root carbon distribution within the different depth profiles among sites ($F;_{2, 69} = 2.69$, $P < 0.05$). Using Tukey test, it was found out that there was no significant difference between the 19 year old plantation and Natural stand ($P > 0.05$) but there were significant differences between 0-20 cm and 20-40 cm depth profiles of 13 year old El Nino plantation and those of other two study sites ($P < 0.05$). Among the three study sites, the Natural stand recorded the highest amount of root carbon in all three depth profiles sampled i.e. 21.8±4.3 Mg C $ha^{-1}$,36.7±3.2 Mg C $ha^{-1}$ and 25.78±3.7 Mg C $ha^{-1}$ in 0-20 cm, 20-40 cm and 40-60 cm respectively. In all study sites, most roots were concentrated in 20-40 cm depth profile, with Gazi El Niño, Kinondo and Natural stand recording 23.4±1.8 Mg C $ha^{-1}$, and 26.7±2.1 Mg C $ha^{-1}$ 36.7±3.2 Mg C $ha^{-1}$ respectively in that particular sampling depth profile.

The high standard errors in different depth profiles of the Natural stand were as a result of reduced number of replicates in some of the plots sampled. This was because of the presence of coral reef beyond 0-20 cm depth profiles. In 0-20 cm few roots were encountered here.

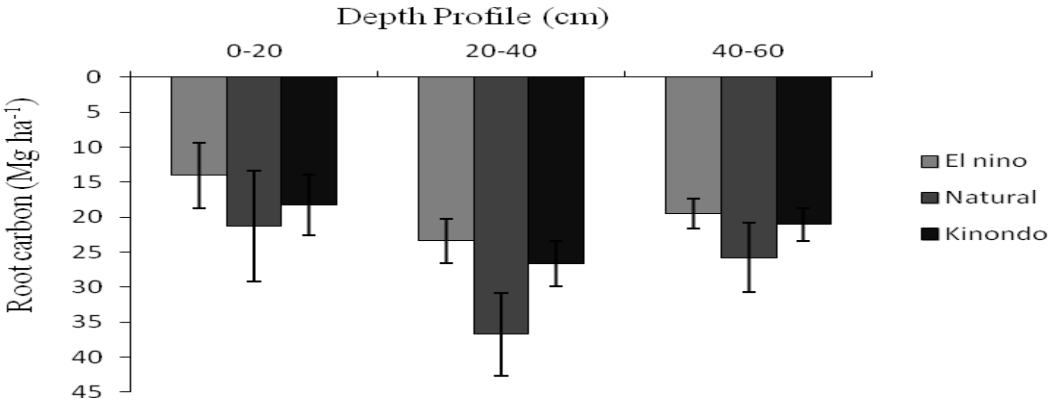

**Figure 8**: Total root carbon distribution among different depth profiles across different study sites in Gazi, Kenya.

### 3.4.3.1 Necromass

Natural stand recorded the highest concentration of necromass carbon in all sampled depth profiles 496.5±16.8 Mg C $ha^{-1}$ while El Niño and Kinondo plantations had 470.5 ± 28.9 Mg C $ha^{-1}$ and 148.4± 5.1 Mg C $ha^{-1}$ respectively (Figure 9). Most of the necromass was concentrated in the middle depth profile (20-40) in all the study sites. There was a significant difference in necromass carbon concentration within and between sites ($F;_{2, 69} = 89.35$, $P < 0.05$). There was a significant difference in necromass carbon concentration between Kinondo and the other two study sites was noticed ($P < 0.05$, Tukey test).

Of the three study sites, natural stand and Kinondo plantation did not show a trend in necromass carbon concentration unlike in El Niño plantation where the necromass carbon concentration within different depth profiles increased downwards.

There was a significant difference in necromass carbon concentration within different depth profiles in El Niño and Natural sites (El Niño F; $_{2, 9}$ = 11.80, $P < 0.05$, Natural F; $_{2, 27}$ = 11.49, $p < 0.05$) but there was no difference in necromass carbon distribution in Kinondo (F; $_{2, 33}$ = 2.628, $p > 0.05$). A significant difference was also noted in necromass carbon in the three depth profiles between the different study sites (F; 2, 69 = 0.89, $P < 0.05$) .There was significance difference in necromass concentrations between different depth profiles in El Niño and Natural stand ($P < 0.05$, Tukey test).

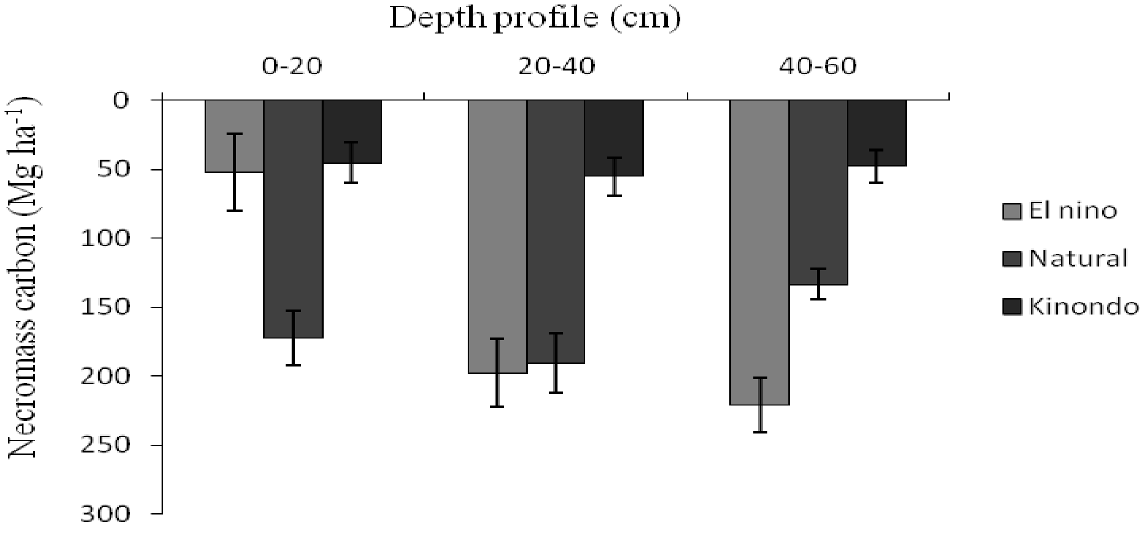

**Figure 9**: Total necromass distribution within different depth profiles in different study sites at Gazi Bay, Kenya.

### 3.4.4 Soil

#### 3.4.4.1 Bulk density

Bulk density was highest in Gazi and ranged from 0.61 to 0.73 g/ $cm^3$ with a mean of 0.67±0.03 $g/cm^3$. Kinondo and natural stand had mean bulky densities of 0.43±0.01 $g/cm^3$ and 0.3±0.02 $g/cm^3$ respectively. There was a significant difference in bulk densities among different study sites ($F;_{2,92} = 48.349$, $P < 0.05$). There was a significant difference between the plantations and the natural stand ($P < 0.05$, Tukey test). In the plantations, bulk density increased with increase in depth (Figure 10). Natural stand revealed a fluctuation trend in bulk density. In the stand, the bulk density increased with depth in the first two sampling depth profiles then decreased in the last two sampling depth profiles (Figure 10). There was no significant difference in bulk density among the various depth intervals ($F;_{2,92} = 0.235$, $P > 0.05$).

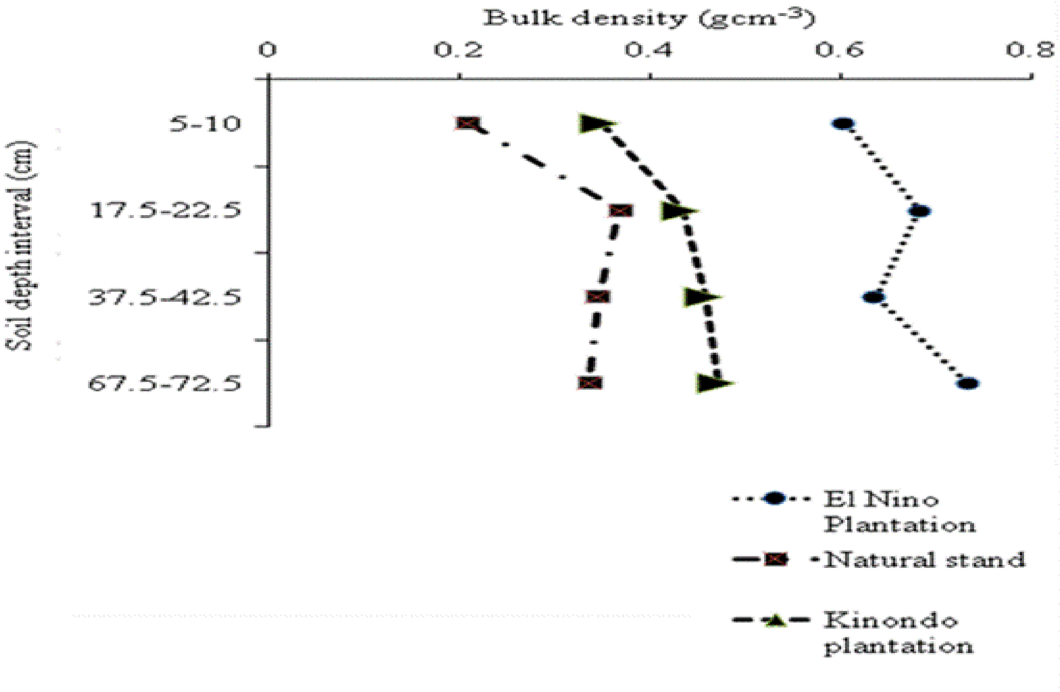

**Figure 10:** Bulk density across different depth intervals in the plantations and natural stand of *R mucronata* in Gazi bay, Kenya.

### 3.4.4.2 Soil organic carbon

Soil organic carbon across the plots ranged from 488.9- 1040.3 Mg C $ha^{-1}$ with a mean of 848.2±176.2 C $ha^{-1}$ in the Gazi El Niño plantation followed by the 19 year old Kinondo plantation whose soil organic carbon ranged from 483.9-780.1 C $ha^{-1}$ with a mean of 640.5 ± 27.6 Mg C $ha^{-1}$. Natural stand had the least amount of soil carbon, ranging from low as 122.2 C $ha^{-1}$ to 629.2 with a mean of 442.1 ± 46.5 Mg C $ha^{-1}$ (Figure 11). Soil carbon densities differed significantly between sites ($F;_{2, 92} = 8.89$, $P < 0.05$). There was no significant difference between the two plantations ($P > 0.05$) while there was a very significant difference in soil organic carbon concentration between the two plantation and the natural stand ($P < 0.05$, Tukey test).

Soil carbon densities also differed greatly among different depth profiles sampled (H; 3, 92= 90.21, $P < 0.05$). The vertical patterns of soil carbon densities were similar, increasing with increase in depth in both natural and replanted forests. 50-100 cm depth interval had the highest concentration while 0-15 cm having the lowest concentration. There was high concentration in SOC in the 50–100-cm depth as compared to the rest of the depth profiles. SOC at 50-100 cm accounted for 54.2 % of the total carbon in the upper 100 cm sampled in the 19 year old Kinondo plantation, 85.7% in the 13 year old El Niño plantation and for 43.4% in the natural stand. For all three study sites, SOC was highest in 50-100 cm and lowest in 0-15 cm. There was a more significant difference in soil organic carbon concentration between 50-100 depth profile and other depth profiles ($P < 0.05$, Tukey test).

SOC densities changed with depth in the three sites sampled. To understand the distribution patterns at different sediment layers in different sites, SOC densities were calculated for the study sites as shown in figure 10 above. It was found that SOC densities increased consistently with depth but the incremental varied among the three study sites.

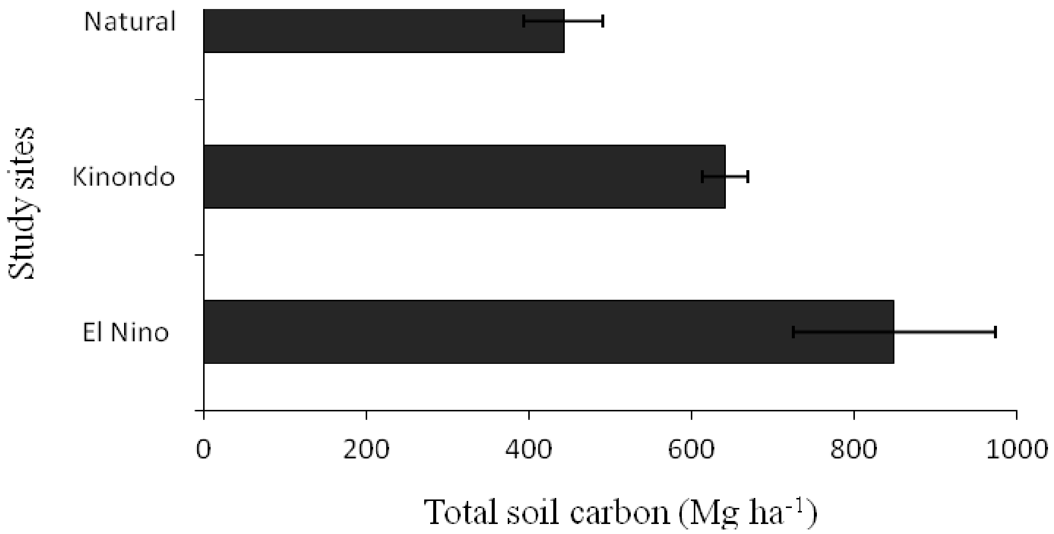

**Figure 11**: Mean soil carbon stocks for the *R. mucronata* plantations and natural stands in Gazi Bay, Kenya

### 3.4.5 Total carbon stocks

Pooling all the carbon compartments accounted for during the study, mean total carbon stocks were estimated at 914.5±237.3 Mg C/ha, 812.6±186.5 Mg C/ha and at 692.9±142.9 Mg C/ha for the El Niño plantation, Kinondo plantation and Natural stand respectively. The total carbon stocks differed significantly between the sites (F; $_{2,\,6}$ = 280.91, $P < 0.05$). There was a significant difference between the two plantations and natural stand ($P < 0.05$, Tukey test). Plantations had a significant amount of soil organic carbon compared to the natural stand (Table 7).

**Table 7**: Carbon stocks in Mg C $ha^{-1}$ for various carbon pools of Natural stand and plantations of *R. mucronata* in Gazi Bay, Kenya.

| Carbon stock | El nino Plantation | Kinondo Plantation | Natural stand |
|---|---|---|---|
| **Aboveground carbon** | 37.2 ±3.8 | 98.2 ±7.1 | 167.9 ±35.5 |
| **Belowground carbon** | 56.2 ±2.5 | 66.3 ±2.5 | 83.8 ±186.5 |
| **Soil organic carbon** | 848.2 ±176.2 | 640.5 ± 27.6 | 442.1 ± 46.5 |
| **Total Carbon Stock** | 914.5 ±237.3 | 812.6±186.5 | 692.9±142.9 |

The deeper soils layers; 37.5-42.5 and 67.2-72.5 contributed the highest carbon stocks, which far exceeded all other pools combined and contributed 90%, 78.8% and 63.8 % in Gazi El Niño plantation, Kinondo plantation and natural stand respectively.

## 3.5 Nutrients

### 3.5.1 Ammonium

On average the 19 year old Kinondo plantation had the highest concentration of ammonium 5.5±0.22 g/L, followed by Natural stand with a concentration of 4.3±0.0.04) g/L. El Niño had the least ammonium concentration of 3.2±0.16 g/L (Figure 12). There was a significant difference in ammonia concentration between sites (F; $_{2,\ 66}$ = 0.521, $P < 0.05$). There was no significant difference in ammonium concentration between Kinondo plantation and Natural stand ($P > 0.05$, Tukey test) but there was significant difference in ammonium concentration between El Niño and other two study sites ($P < 0.05$, Tukey test). Along the four depth profiles ammonium increased with increase in depth and was highest in 50-100 depth profile. There was however, no significant difference in ammonium concentration among the different depth profiles in a site (F;

$_{3,\ 66}$ = 6.98, P > 0.05) but there was a significant difference in concentration among different depth profiles in the study sites (P < 0.05, Tukey test).

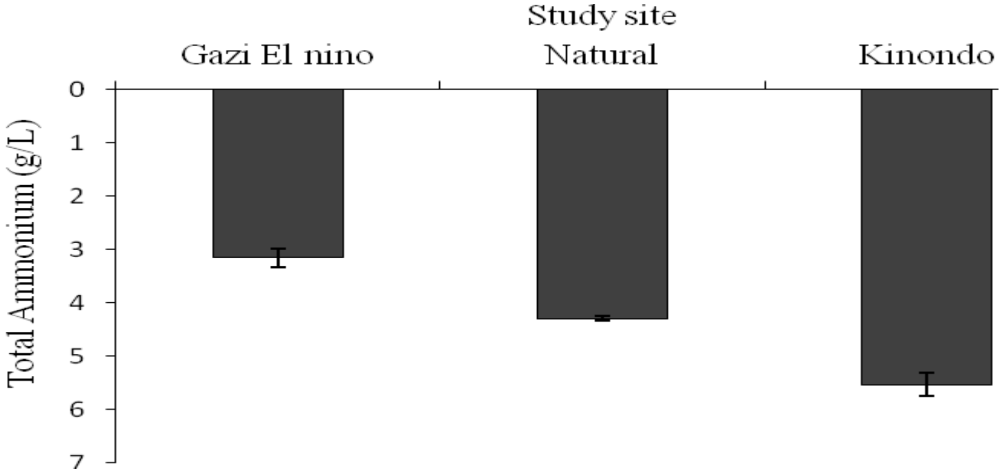

**Figure 12**:Ammonium in the sediment of plantations and natural stands of *R. mucronata* in Gazi Bay, Kenya.

### 3.5.2 Nitrates

Natural stand recorded a nitrate concentration of 1.5± 0.3 g/L while Gazi El nino and Kinondo plantation recorded a nitrate concentration of 1.3± 0.3 and 1.5± 0.6 g/L respectively. There was no significant difference in nitrate concentration between the three study sites (F; $_{2,\ 66}$ = 0.528, P > 0.05). Concentration decreased with increase in depth. There was no significant difference in nitrates concentration between different depth profiles within (F; $_{3,\ 366}$ = 0.168, P > 0.05).

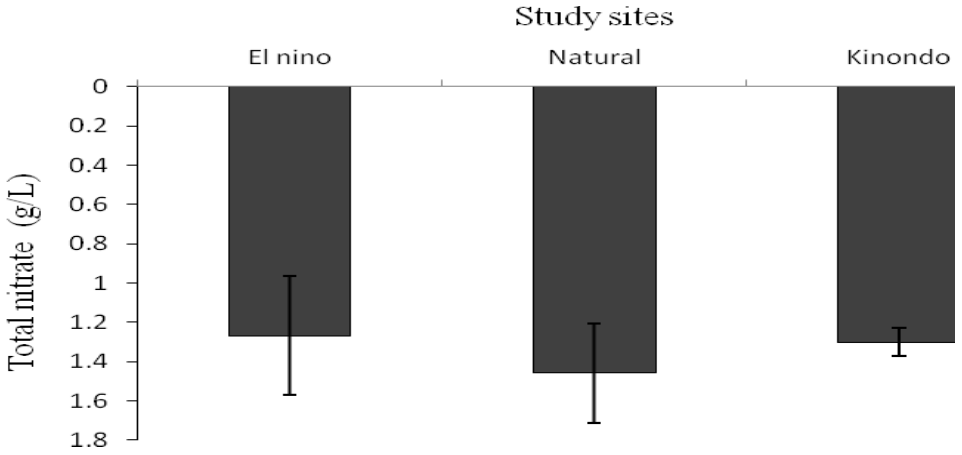

**Figure 13**: Nitrate concentration in the sediment of plantations and natural stands of *R. mucronata* in Gazi Bay, Kenya.

### 3.5.3 Phosphates

The Natural stand had a mean phosphate concentration of 0.4± 0.2 g/L while El Nino and Kinondo plantations recorded a phosphate concentration of 0.34± 0.3 g/L and 0.34± 0.4 g/L respectively. There was no significant difference in phosphate concentration among the three study sites ($F;_{2,\ 66} = 0.521$, $P > 0.05$). In El Nino plantation, the phosphate concentration increased with increasing depth profile while Kinondo plantation and the Natural stand displayed a flactuating trend with 0-15 cm depth profile having the highest concentrations. The concentration were 0.80 g/L 0.45 g/L and 0.39 g/L for the 13 year old El Nino plantation, Natural stand and Kinondo plantation respectively. There was a significant difference in P concentration between different depth profiles within study sites ($F;_{3,\ 66}$, $= 6.633$ ,$P < 0.05$).There was a signficant difference in phosphorous concentration especially in 50-100 cm depth profile and other depth profiles ($P < 0.05$, Tukey test).

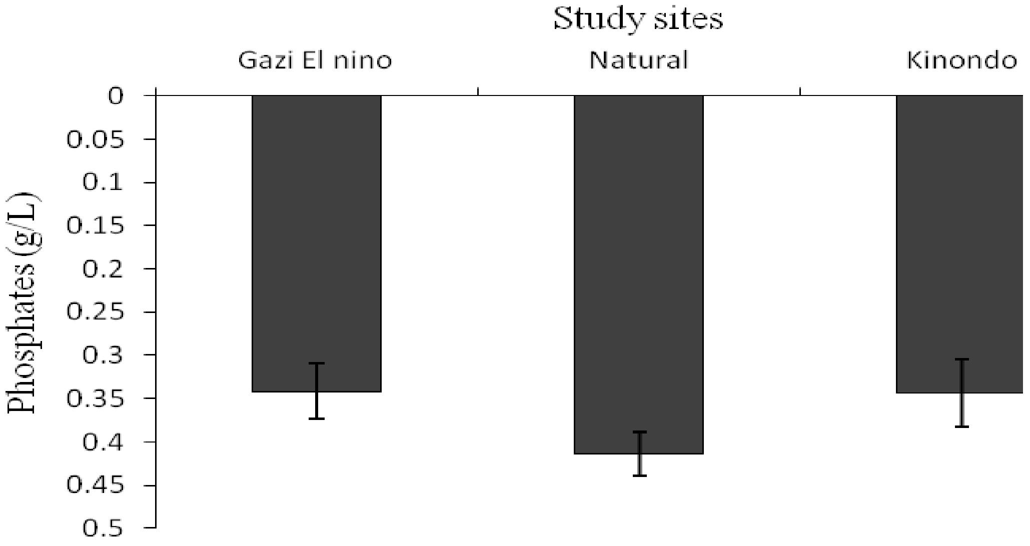

**Figure 14**: Phosphate concentration in the sediment of plantations and natural stands of *R. mucronata* in Gazi Bay, Kenya.

## 3.6 Belowground biomass and Nutrients

Belowground biomass across the vertical dimensions was also correlated with nutrients. In the Natural, there was no correlation between P and BGB in all depth profiles sampled (n = 48 r = 0.42, P > 0.05). Same was noted in the plantations. Nitrogen was however negatively correlated with BGB in all depth profiles and sites sampled (n = 48, r = -0.081, P < 0.05). Ammonium on the other hand was positively but weakly correlated with BGB (n= 48, r = 0.63, P > 0.05). Generally, correlation between BGB and soil nutrient availability appeared to be negatively correlated across the total vertical dimension of 0-60 cm (r = -0.095, P > 0.05).

# CHAPTER FOUR

# DISCUSSION, CONCLUSIONS AND RECOMMENDATION

## 4.0 Forest structure

In all stands, Importance values (IV) were high for *Rhizophora mucronata* making it the most dominant species. The high IV value of *R. mucronata* in the plantations is because the plantations were established as monostands of *R. mucronata* with high density. In the natural stand *C. tagal* was the second dominant species after *R. mucronata* (Table 2).

The overall basal area for *R. mucronata* in El Niño plantation, Kinondo plantations and the Natural stands was 12.8, 28.4 and 20.2 m$^2$ respectively. Bigger stem diameters, height and basal area were noted in Kinondo plantation than in the natural stand. This can be attributed to rapid growth as a result of silvicultural treatments such as pruning and reduced interspecies competition in the 19 year old Kinondo plantation while the small diameters and basal areas in the natural stand can be attributed to the interspecies competition by other species which may have inhibited the growth of *R. mucronata* ).

Stand table data (Table 2) shows occurrence of high stem density of 5650 stems/hectare in the 13 year old El Niño plantation followed by the 19 year old Kinondo plantation which recorded a density of 3725 stems/hectare. The mangrove trees in Kinondo plantation had been thinned, thus reducing the stand density but allowing increase in biomass. El Niño plantation had the highest stem density; this is because unlike in the plantations, no thinning had been carried out unlike in the Kinondo plantation. The Natural stand had the lowest stem density and this is because the trees here were more spaced with comparatively bigger diameters (Table 2). Disturbance such as logging which was much more pronounced in the Natural stand also caused immediate decline in tree stem density (Table 2). The high density of Juveniles in 13 year old El Niño plantation than in Kinondo plantation (Table 3) gives a clear indication that this is a relatively younger forest compared to the other two study sites. A study by Lugo, (2002) found a high abundance of regeneration mangroves in a 11 year old *R. mangle* plantation in Florida. Difference in stand density between replanted and natural stand are expected because of the planting density, survival of the saplings and recruitment of the seedlings at early stages of forest development (Sawale & Thivakaran, 2013). Other reasons for different stem density and regeneration potentials could also be as a result of space exploitation, resource exploitation and increased light competition between adult trees (Sawale & Thivakaran, 2013). Despite Gazi El Niño plantation having the highest stem density compared to Kinondo plantation; its basal area is relatively smaller. A decline in stand density and an increase in basal area is typical of a developing forest (Laclau, 2003). Stem diameters in Kinondo plantation are slightly larger than those reported by Kairo *et al.* (2008) when the plantation was 12 years old.

In the natural stand, the pattern of distribution of trees by diameter class follows the reverse-J curve in which the number of large trees decreases with increasing diameter class (figure 4). This happens because the growth rate of seedlings and saplings trees is higher than matured trees and theoretically because of age-gradations in an uneven-aged forest (Kairo *et al*., 2002). The curve can however be modified by environmental factors such as selective tree harvesting, interspecies resources competition or competition between mother trees and seedlings resource competition, regeneration patterns, differences sediment characteristics, irregular or seasonal climatic events. Diameter distributions have been used to assess the disturbance effect within forests (Sawale & Thivakaran, 2013) and to detect trends in regeneration patterns (Laclau, 2003). Tree density distribution across different diameter classes also shows how well the growing forest is utilizing the site resources (Farida-Hanum *et al.*, 2012; Sawale & Thivakaran, 2013).

Similarities in some of the structural attributes and even the nutrient status between the 19 year old *R. mucronata* and Natural stand indicate that reforestation can be used to reverse the trend of human induced deforestations restoring both the associated ecosystem functions and also the lost biological communities.

## 4.1 Natural regeneration

Differences in forest conditions affect growth and survival of juveniles and saplings in mangroves ecosystem. Lack of significant difference between Regeneration Class I and Regeneration Class II in the three sites indicate that factors that determine emergence and growth are similar among the sites. Differences in abundance of Regeneration Class III between the Natural stand and plantations could be due to effect of shading. At the plantations the canopy was more closed and might have limited light penetration which is required for growth and survival unlike in the Natural where canopy was more open. There was no shading effect and this increased photosynthesis hence survival of Regeneration Class III. This could also be due to the fact that Natural stand produces more seeds than the plantations because of presence or abundance of mature trees which act as seed sources. It could be also due to the species richness as many species were encountered and some have different regenerating potentials. Generally, the Natural stand had a higher number of recruits compared to the plantations suggesting a higher regeneration potential. However, Natural stand which had the highest regeneration density had lowest mature tree density. This shows that regeneration potential is high in areas characterized by low overstorey and low stand density. Mangrove ecosystems have also been reported to support little or no understorey vegetation (Field, 1998, Krauss *et al*., 2003). Light has been reported to positively influence mangrove reestablishment. Increased growth rate has been reported in un-shaded environments in mangroves. This is attributed to increased assimilation rate in mangrove trees despite them being adapted to low light regime (Krauss *et al*., 2003). Higher densities of saplings recorded in natural stand than in the plantations could be due to prevailing conducive factors in the stand. Factors like elevation, tidal inundation, soil texture, seed dispersal or trapping and water current pattern have been reported to favor natural regeneration and so could be the reason for this high regeneration potential in the natural stand (Sawale & Thivakaran, 2013). This indicates that there is high regeneration potential in the natural than in plantations. This shows that proper management and conservation of natural system could provide a better alternative compared to replanting.

*R. mucronata* seedling and sapling survival was high in the three sites compared to other species as witnessed by the high number of seedlings and saplings in different regeneration classes. This could be due to its good seed propagule and good rooting habit. This enables it to grow and survive better, outcompeting other species which are less suited to survive under such conditions. This could also be due differences in the physicochemical conditions, where by conditions for regeneration in particular specie could be sub-optimal for others. Conditions under the selected study sites could have favored regeneration of *R. mucronata*. It could also be due to the fact that majority of mangroves in the three sites were mainly *R. mucronata*. While the low abundance of *X. granatum* juveniles compared to other juvenile species encountered. It could also be due to its heavy fruit which limited dispersal and limited distances as it seedlings and saplings were encountered in very few plots in the natural stand.

Chong (1988) formulated ‘equivalent regeneration values’ for different regeneration classes (RCI: RCII: RCIII). A regeneration ratio of 6:3:1 was found to be an effective stocking rate for saplings. In the present study, the regeneration ratio; RCI: RCII: RCIII was 29:5:4, 73:4:5 and 11:10:8 was recorded for Gazi ,Kinondo and natural stands respectively which indicates that there is no need for replanting degraded patches in natural stand as the above juvenile ratios show that the forest is able to regenerate on its own when under no disturbance.

## 4.2 Biomass equation

In this study, various independent variables such as stem diameter, quadratic diameter, height or combination of both were tested based on the $r^2$ and SE values obtained. Total aboveground biomass and biomass of different tree components was best estimated by model A with $D_{130}$ as the independent variable. The correlation coefficient for model A when total AGB used was highest compared to other models derived using different parameters or their combinations. Even though height data was collected during the study and even used to derive some models, it was not used as predictive variable because of reduced practicability in mangrove ecosystem. This is because other than in harvesting, it’s not always easy to measure the tree height due to their interlocking canopies (Mitra *et al*., 2011). Most sophisticated equipments for determining height measurements in other forest ecosystems are not able to accurately or efficiently work in mangrove ecosystems. Stem diameter alone was used as the predictive variable which indicates in this study, biomass could be confidently estimated using DBH as the only predictor. Biomass accumulation rate in the plantation was 10.32 t/ha/yr. This was slightly higher than 8.89 t/ha/yr reported for the same plantation when it was 13 years old (Kairo *et al.,* 2008). The biomass accumulation is similar to those observed for managed plantations in Southeast Asia, where annual biomass increment values ranging from 14 to 34 t/ha/yr for plantations of *Rhizophora* species (Ong *et al.*, 1984; Ong *et al.*, 1995). Biomass accumulation ranging from 6.3 to 45.4 t/ha/yr has been observed for Australian mangroves (Clough, 1992).

## 4.3 Volume equation

The mangrove tree volume in the 19 year old Kinondo plantation increased proportionately with increase in tree diameter ($D_{130}$). The tree with the smallest diameter of had the smallest volume of 9.87 $m^3$ while tree with the biggest diameter had the highest volume of 231.93 $m^3$. There was no significant difference between total volume (both merchantable and unmerchantable) and merchantable volume. This could be due to pruning which has been administered as one of the silvicultural treatments during management of the plantation. Few branches and leaves left after the pruning exercise contributed to significantly low amount of unmerchantable volume. Stand volume in *R. mucronata* plantations has been reported to range between 3-280 $m^3$/ha (Chandra *et al*., 2011) while (Kairo *et al*., 2002) reported stand volume ranging between 28-700$m^3$/ha in natural stand of *R. mucronata.*

A natural mangrove forest in Sarawak, Malaysia was reported to have a stand volume of 65 $m^3$/ha (Chandra *et al*., 2011) which is lower than that reported for this study. In his study, Langat (2006) reported a stand volume of 100.44 ± 22.53 $m^3$/ha for the 19 year old Kinondo plantation when it was 12 years old. This shows stand volume increases as the tree matures. Stand volume for the 19 year old *R. mucronata* is within the range of the values reported for other *Rhizophora* species in other studies (Kairo *et al*., 2002). The difference in stand volume at different ages can be attributed to changes in tree diameter and height which are closely related to changes in tree volume.

## 4.4 Carbon pools

### 4.4.1 Aboveground

In this study, AGB carbon differed significantly between sites. High aboveground carbon in the natural stand than in the plantations could reflect to the age difference. During this study the plantations sampled were comparatively younger compared to the natural forest which is a secondary stand more than 50 years old. Age difference meant higher foliage in both natural and 19 year old plantation which means higher primary production compared to 13 year old plantation. The high aboveground carbon in the Natural stand than in the plantations can also be explained by high species richness encountered in the Natural system compared to the plantations. Chen *et al.*, (2012) reported high above and belowground for a mixed plantation than for a mono stand. Low amount of aboveground biomass in Kinondo plantation could also be attributed to loss of biomass as a result of pruning which was part of silvicultural treatment for the plantation.

In terms of size class distribution, despite Kinondo plantation mean size being slightly larger than the Natural stand. Few large *R. mucronata* with extreme diameters in the natural stand contributed significantly to the aboveground biomass. The variation in biomass has also been reported to vary based on ecology, species, plant density, growing season, plant age and global positioning of mangrove forests (Abib & Appadoo, 2012; Matsui *et al*., 2012). These factors are never constant within an ecosystem and may differ across different ecosystems.

Aboveground biomass for plantations in this study compares well to those reported for other mangrove plantations around the world. In Mauritius Abib & Appadoo, (2012) reported an aboveground biomass of 16.63 t/ha which is equivalent to 8.31 Mg C $ha^{-1}$ in 15 year old *R. mucronata* mangroves. This is lower compared to that reported for 19 and 13 year *R. mucronata* plantations in this study. In Central Thailand, AGB was estimated at 140.49 Mg C $ha^{-1}$ in a 12 year old *R. apiculata* plantation (Kridiborworn *et al*., 2012) which is higher than that reported for the 13 year old *R. mucronata* in this study. In Japan, ABG was reported as 108 Mg/ha in a mangrove stand dominated by *R. mucronata* which translates to 54 Mg C $ha^{-1}$ of carbon

(Komiyama *et al*., 2008; Abib & Appadoo, 2012) while in Matang forest in Malaysia, aboveground biomass carbon for 18 and 23 year *old* *R. apiculata* plantation was reported to be 60 and 77.5 Mg C ha$^{-1}$ (Gong & Ong, 1995), figures which are lower than that reported for 19 year old *R. mucronata* plantation in this study but higher than those reported for 13 year old Gazi plantation.

ABG carbon values in natural stand for this study are similar to those reported for mangroves in West Africa mangroves reported as 225.5 Mg C ha$^{-1}$ in Cameroon, 170.5 Mg C ha$^{-1}$ in Gabon, 125.5 Mg C/ha in republic of Congo and 204.5 in Democratic republic of Congo (Ajonina *et al*., 2014) and also similar to 176 Mg C ha$^{-1}$ for Mexican mangroves (Adame *et al*., 2013) and similar to 249 Mg C/ha for mangroves found Yap, Micronesia (Donato *et al*., 2012) . Based on these studies, there is a clear indication that a great variability in AGB and subsequent aboveground carbon stocks, exists for *R. mucronata* mangroves across the world.

### 4.4.2 Dead organic matter

In this study, dead wood carbon pool was 7.8± 0.67 Mg C ha$^{-1}$ for the 19 year old Kinondo Plantation. The 12 year Gazi El Niño plantation and natural stand had no wood debris and this can be attributed to the fact that the sites are accessible and close to the village where there is much high fuel wood demand and also because of its young age. Few studies have quantified dead organic mass in mangrove forests (Adame *et al*., 2013; Ajonina *et al*., 2014) despite it representing a significant carbon and nutrient pool in mangrove ecosystems (Krauss *et al*., 2005). In this study, dead wood estimated is similar to 7. 3 Mg C ha$^{-1}$ in Cameroon and 10.5 Mg C ha$^{-1}$ for Gabon (Ajonina *et al*., 2014). Its however lower compared to 34 Mg C ha$^{-1}$ reported for DRC (Ajonina *et al*., 2014), 16.7 4.2 Mg C ha$^{-1}$ for Mexican Caribbean (Adame *et al*., 2013) and 28.5 Mg C ha$^{-1}$ for federated states of Micronesia (Kauffman *et al*., 2011). Wood debris can persist for many years in tropical mangrove ecosystems (Krauss *et al.*, 2005) and thus form a significant source of organic carbon.

### 4.4.3 Belowground biomass

Below ground root carbon was high in the natural stand than in the plantations with the 13 year old plantation having recording the least concentration of root biomass carbon. In the three study sites, most of the roots were found to be abundant in (20-40 cm) depth profile. High concentration of roots biomass carbon in the middle depth profile could be due to the fact that, most of the nutrients seemed to be concentrated in this layer. Also this could be due to the fact that as mangroves grow in soft wet and muddy substrate, their roots need to be deep enough for proper anchorage to withstand tide inundation and strong wind but also in a depth where there is some free air circulation. In his study, (Castaneda-moya *et al*., 2011) also found out that the highest root concentration was in 0-40 cm zone compared to the deeper root zone (Beyond 40) where soil condition was more anoxic. High root biomass carbon concentration in the natural stand than in the plantations could be as a result contribution by other mangrove species present and probably because the trees in this stand are mature trees with well developed roots unlike in a monostand. Low root biomass in Gazi could be as a result of its age, a young forest which has not reached maturity and probably with incomplete developed root system. It could also be because the plantation is a monostand with no other species to contribution hence low root biomass and subsequently carbon.

The root carbon concentration in this study is much higher compared to 18.1 Mg C ha$^{-1}$ and 17.9 Mg C ha$^{-1}$ respectively reported for natural stand dominated by *R. mucronata* and *R. mucronata* 12 year old plantation Gazi Bay, Kenya (Tamooh *et al*., 2008). In this study carbon content for the natural stand was almost 9 fold greater than that reported by Tamooh *et al.* (2008) for the same site and this can be attributed to different estimations methods used and also because of different plots used during the study. They are however similar to 75.5 Mg C $ha^{-1}$ in Gabon and 61 Mg C/ha in Democratic Republic of Congo (Ajonina *et al*., 2014) but lower than 153 Mg C $ha^{-1}$ reported in Cameroon and 92.5 Mg C $ha^{-1}$ for *R. racemosa* in Democratic republic of Congo (Ajonina *et al*., 2014) and 98.05 Mg C $ha^{-1}$ reported by Komiyama *et al*. (2008) for *R. apiculata* mangrove stand in Halmahera Island, eastern Indonesia. Other studies have reported total belowground root carbon ranging between 70.3 -176.3 Mg C $ha^{-1}$ for a natural *R. mucronata* forest and 70.3 - 134.5 Mg C $ha^{-1}$ for younger stands in Sawi Bay mangroves in Thailand (Matsui *et al*., 2012). In Cuban mangroves total belowground root carbon was reported to average at 16.3 Mg C $ha^{-1}$ for *R. mucronata* forest within a sampling depth of up to 40 cm (Lovelock, 2008). Belowground a root carbon to storage has been reported to range between 19.5 - 142 Mg C $ha^{-1}$ for most *Rhizophora* species (Fujimoto *et al*., 1999; Kridiborworn *et al*., 2012). The high amount of biomass in mangrove ecosystem could be as an adaptation of for living in soft sediment and tidal inundation as they could be unable to mechanically support their aboveground weight without a heavy root system.

The carbon values reported in this study seem to be higher than most of those reported in the literature. These differences in belowground root carbon estimates in different studies can be attributed to different methodological methods used to estimate belowground biomass, sampling depth, age of mangrove forest and probably ecological conditions of study sites.

The coring method used during this study may have resulted to overestimation of root biomass due to unoccupied substrate. However underestimation may have also occurred due to uneven distribution of roots and also sampling to a depth of up to 60 cm only.

#### 4.4.3.1 Necromass

Abundance of necromass in the middle depth profile might be a physiological adaptation by the mangrove to facilitate efficient uptake of water and nutrients which have been shown in this study to be abundant in this layer. A significantly higher proportion fine root necromass carbon in Gazi El Niño plantation and natural stand could be as a result of high density of roots and short life span of fine roots in natural stand while in Gazi El Niño could be as a result of massive sedimentation due to erosion following extremely heavy rainfall in the year 1997 which caused mangrove dieback in Gazi Bay. Similar findings for the same site have been reported by Tamooh *et al*., (2008).

### 4.4.4 Soil

#### 4.4.4.1 Bulk density

The 19 year old Kinondo had higher bulk density compared to the Natural stand. Bulky density has been reported to increase with increasing depth and natural stand which had coral reefs in most plots sampled could have resulted to the low bulky density. Donato *et al*., (2011) reported a bulky density to range from 0.35 to 0.55 g/cm$^3$ for Indo pacific mangroves while Ajonina *et al*., (2014) reported a bulky density to range from 1.12-1.05 with a mean 1.09 ± 0.03 g/cm$^3$ for mangroves in West and Central Africa and in these studies bulky density seemed to increase with depth unlike in this study where bulky density showed no trend but fluctuated with depth.

The bulky density in the study sites can thus be said to be within the range of that reported in the literature for mangroves in indo pacific region (Donato *et al*., 2011) but lower compared to that reported for mangroves in Central and West Africa (Ajonina *et al*., 2014).

#### 4.4.4.2 Soil organic carbon

Soil organic comprised the largest carbon pool in mangrove ecosystems sampled during this study. A higher SOC content was found in the plantations compared to the natural stand. Previous deposition by pre-existing stands and probably sediment deposition from hinterland could explain the high SOC in the plantations and more so In El Niño plantation. The relatively high carbon stocks in Gazi El Niño plantation which is the youngest also show that not all carbon was oxidized and emitted to the atmosphere after destruction of pre-existing stand indicating substantial input and accumulation of carbon in the plantation compared to natural stand. Results of this study are therefore proves the fact that mangrove forest restoration increases soil organic carbon accumulation in the sediments.

The results of this study compare well with the high other carbon stocks associated with mangrove sediment elsewhere in the world. A study by Chen *et al*., (2012) found out that mangrove plantations accumulated more sediment carbon compared to natural stand. Donato *et al*., (2011) reported an average of 864 Mg C ha$^{-1}$ for mangroves in the Indo-pacific region but lower than SOC of 1166 Mg C ha$^{-1}$ reported by Adame *et al*. (2013) in sediment of mangroves in the Mexican Caribbean. The results are also in conformity with those reported by Ajonina *et al*., 2014 for West and Central Africa where SOC averaged at 827± 170 Mg C ha$^{-1}$ with undisturbed mangrove stands recording the highest amount of mean SOC 967 ± 58 Mg C ha$^{-1}$ with heavily and moderately sites recording an average of 774 ± 163 and 741 ± 190 Mg C ha$^{-1}$ respectively. 50-100 depth profile recorded. Different values for the soil organic carbon have been reported for sediments of various mangrove types, from different sites and depth profiles. Pandey *et al* (2013) while working in the mangroves of Gujarat, India found out that there was more carbon sequestration in lower levels (16-30 cm) as compared to the upper layer (0-15).

Efficiency of carbon sequestration in sediments improves with age of mangrove forests from 16% for a 5 year old forest to 27% for an 85 year old stand, (Alongi, 2002). El Niño plantation which was the youngest of the three study sites recorded the highest concentration of soil organic carbon which suggests a probable contribution from previous mangroves which were destroyed in the same stand as a result of increased sedimentation during the El Niño rains of 1997. High root turnover in young plantations could also have contributed to the high soil organic carbon.

High soil organic carbon in the plantations could also be as result of more sustained anoxic conditions in the plantations than in the natural which has been degraded and exposed to the sun unlike in the plantations. During the study, it was observed that during some neap tides, the plantations were inundated while some parts of the Natural system were not. Anoxic conditions in the plantations slowed down the decomposition process of organic carbon in the sediments hence high soil organic carbon in the plantations. The Natural stand was quite degraded, exposed and inadequately inundated. These conditions could have accelerated decomposition rates in the stand resulting to low organic carbon in the sediment of Natural stand. Pandey & Pandey (2013) reported that mangroves ecosystem receiving proper and adequate inundation, its soil organic carbon was protected from high rates of decomposition unlike those receiving inadequate inundation.

High soil organic carbon in El Niño plantation mangrove sediment could also be as a result of importation through soil erosion from hinterland. Accumulation of soil organic carbon in the mangrove sediment has also been attributed to balancing between inputs of organic C (Litter, tide and root turnover) and on the other hand, output through dilution by inorganic material, mineralization and export by tide (Bouillon *et al*., 2008; Jin-ping *et al*., 2012).

*R. mucronata* mangrove species has a unique and complex root system including stilt roots and some detritus materials such as litter and fallen dead wood which provides excellent environment for organic carbon to accumulate, sink and become sequestered in the sediment. The plantations which were dominated by *R. mucronata* have high abundance of stilt roots per unit area probably facilitated high sedimentation rates and as a result a reduced water flow, facilitate organic matter deposition in the sediment hence more sediment carbon in mangroves plantations than in natural mangrove forest. In her study (Lovelock, 2008) found out that over 80% of the suspended soil particles brought into the mangroves from coastal waters was trapped in mangroves whereby the particles were trapped into stagnation zones around the mangrove root areas. In their study, (Fujimoto *et al*., 1999; Jin-ping et al., 2012) also reported that substantial production of leaf litter combined with low rates of organic matter decomposition in the mangrove sediment all contributed to high organic carbon accumulation

Minimal SOC accumulation was found to occur in natural mangrove stand where the SOC density in the 0-50 cm and 50-100 were (56%) while maximal accumulation occurred in the plantation with (10%) between 0-50 and (34%) between 0-100 cm, however half of the total stock was stored at 50-100 cm.

From the study and published literature its evident that, not only does mangrove restoration result in direct carbon sequestration in the sediment but also helps lock the previous carbon left in the soil after destruction of a pre existing mangrove stand. Furthermore, forest plantation with different sequestration rates results to difference in carbon accumulation rates in the sediment (Ren *et al*., 2010).

The reason for the high organic carbon content in the mangrove sediment is due to the accretion rates of these ecosystems as they try to keep up with sea-level rise, sometimes over thousands of years, and trap detritus and sediments from tidal movement and alluvial deposits (Krauss *et al*., 2003; Mckee *et al*., 2007). Unlike mangrove ecosystems which can keep on accreting sediment over millennia, most terrestrial ecosystems reach maximum carbon content in their soils over decades or even centuries thus making mangrove ecosystem critical carbon sinks (Jin-eong, 1995; Sakho *et al*., 2011).

### 4.4.5 Total carbon stocks

Gazi mangroves; both natural and replanted comprise significant carbon stocks which differed significantly among sites. The variability of total carbon stocks between the study sites was evident from the aboveground vegetation structure and species composition encountered in the sites studied. The replanted stand had a homogenous appearance with uniform diameters while those in the natural system showed structural heterogeneity in diameter distribution. The number of larger trees decreased with increasing diameter class. Low carbon stocks in the natural system than in the plantations system could also be as a result of illegal logging and degradation which was more pronounced in the natural stand. Restoration coupled with good management plans in the 19 year old Kinondo plantation could have contributed to high carbon stocks. It’s therefore important for mangroves to stay completely undisturbed to maintain maximum carbon stocks and large quantities carbon in the sediments. Differences in C stocks in stocks between the plantations and natural stand could also be attributed to; species composition, forest structure, elemental C concentration in trees, tree density, age, management regime and soil depth sampled while analyzing soil carbon.

The total carbon stock values for the three study sites are comparable to other studies carried elsewhere around the world. Donato *et al* (2012) reported total carbon stocks to be 1023± 88 Mg C $ha^{-1}$ in the Indo pacific region, 987± 338 Mg C $ha^{-1}$ for mangroves in Mexican Carribean

(Adame *et al*., 2013) while Mudiyarso *et al* (2009) reported a total C stock of 986 Mg C ha$^{-1}$ for mangroves in Indonesia. The figures are however lower than 1520 ± 164 Mg C ha$^{-1}$ reported by Ajonina *et al* (2014) for mangroves in west and central Africa. In the present study the replanted mangroves had the highest total carbon stock per unit area compared to the natural stand.

Total carbon stock for the Kinondo plantation was higher than 65.8 Mg C ha$^{-1}$ reported by (Kairo *et al*., 2008) for the same plantation when it was 12 year old. This is probably due to the fact that, soil carbon which forms the largest carbon pool in any mangrove ecosystem was not accounted for as in the present study and it could also be due to age difference. Biomass production has been reported to increase as tree matures (Joshi & Ghose, 2014). The biomass production for the 19 year old Kinondo plantation was similar to those observed for managed plantations in Southeast Asia, (Ong *et al.*, 1984; Ong *et al.*, 1995) and too increased with increase in age. Other factors such climatic conditions, nutrient limitation and soil factors have also been thought to influence biomass production however the complexity of interactions between these factors and forest structure and growth makes it difficult to correctly point out the main factors affecting biomass production in any given site (Clough, 1992).

Total carbon storage reported in mangroves is exceptionally high relative to other major forest domains (Donato *et al.*, 2011; Kauffman *et al.*, 2011). In Kenya, carbon stocks have been reported to ranged between 53-80 Mg C ha$^{-1}$ in Arabuko Sokoke, an indigenous coastal forest (Glenday, 2006). Average AG carbon pool for riverine forests in Tana river county were 257±43 Mg C ha$^{-1}$ in levee forests, 170 ±13 Mg C ha$^{-1}$ in evergreen forests and 163±15 Mg C/ha in transitional/woodland areas (Glenday, 2006). In Kakamega forest, which is a rain forest, total carbon stock was estimated at 218 ± 17.7 Mg C ha$^{-1}$ Lung (2008), which is also lower than those reported for mangroves, underscoring the value of mangroves as significant carbon sinks and a potential mitigation to climate change. In this study, it was found that soil carbon stocks in replanted mangroves were higher compared to the natural mangrove stand.

Even though the aboveground carbon of natural mangrove system was low, the high concentration of soil carbon resulted in a relatively large ecosystem carbon stock though lower than for the plantations. In all study sites, belowground, both soil and roots, made up approximately over 60% of the total carbon stock while AG carbon pool made up 30%. Deadwood contributed less than 2% of the total in Kinondo plantation.

Mean total carbon stocks for the three study sites were statistically different and same was the case for different carbon pools sampled. Differences in C stocks between the plantations and natural stand could also be attributed to; species composition, forest structure, elemental C concentration in trees, tree density, age, management regime and soil depth sampled while analyzing soil carbon.

## 4.5 Nutrients

Nutrient availability is a major factor influencing mangrove productivity (Lovelock *et al*., 2005) Nitrogen and phosphorous are the key elements limiting production in mangrove ecosystem (Feller *et al.*, 2003; Lovelock & Feller, 2003). These nutrients occur in mangrove soils in extremely low amounts (Lovelock *et al*., 2005). Given the large disturbance that lead to sedimentation and degradation before establishing the replantations, it was hypothesized that there was a significant difference in nutrient status between the plantations and natural stand. This was however not the case because results of this study suggests that even after disturbance, nutrient distribution pattern in the plantations was able to re-establish and be similar to that of natural stand.

Ammonium is the most abundant and primary form of nitrogen in mangrove sediments due to the anoxic conditions of the sediment in combination with high organic matter and high abundance of denitrifying bacteria such as *Pseudomonas denitrificans* which deplete the nitrates and nitrite pools as a result of accelerated dinitrification rates (Krauss *et al*., 2008; Kristensen, 2008). This explains the high ammonium and low nitrate concentrations concentration in the three study sites. A study on mangrove sediments in Dominican Republic, reported an insignificant nitrate concentrations with vast majority of inorganic N being in the form of ammonium (Sherman *et al*., 1998).

Phosphorous in mangrove soils is immobile and unavailable for plant use making organisms that solubilize Phosphorous important for plant growth especiallly in such a nutrient limited environment like a mangrove ecosystem (Lovelock *et al*., 2005; Lovelock, 2008). Bacteria solubilizing P occur in areas where the sediment is oxygenated especially near the the mangrove roots (Krauss *et al*., 2003). This explains why phosphate concentration in the upper most depth profile in the three study sites was higher compared to deeper layers. The presence of crab holes in the top layers also increases aeration creating a conducive environment for the solubilzing bacteria (Lovelock, 2008; Mckee & Faulkner, 2000)

Studies on mangroves show that when nutrient availability is high, mangroves invest more in ABG that maximise carbon acquisition and when low redirect resources to enhance root biomass (Hwang & Chen, 2001). This was also evident in this study, where by all the nutrients tested in this study showed a negative correlation with belowground biomas and especially in the middle depth profile where BGB was high compared to upper layers with abundant nutrients. Similarly in his study Bouillon *et al*. (2008) reported a negative correlation between soil carbon and C:N nutrients. In his study Castaneda-moya *et al*., (2011) found out that root dynamism was depended on P availability. He also found out that there was a significant increase in in fine root biomass allocation with increasing P limitation which indicates a strong coupling between P availability and carbon allocation to fine root production which in the end translates to increased belowground biomass and hence carbon. This also facilitates nutrient acquisition (Middleton & Mckee, 2001)

Lack of correlation between BGB and P may reflect a weak relationship between total phopshate and the seasonal dynamics of available P. The negative correlation between BGB and nitrates could indicate that N is the most limiting element in mangrove in mangroves of Gazi bay. Total nitrogen has been reported to be one of the elements that limit both below and aboveground production in mangrove ecosystem (Feller *et al.*, 2003). Without detailed information about the above nutrients availability. It was difficult to confidently conclude on the role of each element in driving root distribution and accumulation. Further detailed studies are needed to investigate the relationships between BGB accumulation, various nutrients and their dynamics.

## 4.6 CONCLUSIONS

Restoration combined with management influenced structural development of the replanted mangroves. This is witnessed by the homogeneous distribution of stem diameters in the plantations unlike in the natural stand where there was high heterogeneity in stem distribution.

Mangrove plantations of Gazi Bay are a significant carbon stocks despite occupying a relatively small cover area. They are extremely valuable for their long-term carbon sequestration capacity and underscore the potential of using reforestation as a tool in mitigation of climate change.

The study also demonstrates that biomass and carbon storage capacity varies with mangrove type and age of forest as witnessed by significant difference observed in carbon stocks between the plantations and naturally growing mangroves.

Species-specific allometric equations are the best for estimating biomass. Species-specific allometric should be developed to significantly improve the capacity to accurately estimate biomass and carbon stocks in natural ecosystem.

Root biomass allocation is negatively correlated to nutrient availability. Mangroves invests more in root biomass when in  nutrient limiting environment.

## 4.7 RECOMMENDATIONS

Based on the study, the following recommendations could be made out about the study;

- There should be long term monitoring and management of restored mangroves so as verify the continued storage of carbon stocks observed in this study.
- Proper and continuous monitoring and management of both restored mangroves and natural mangroves to improve the data quality. Monitoring would enable gauging of not only dynamics of carbon but also general mangrove ecosystem dynamics like growth, mortality, recruitment among other factors.

- Development of more species specific allometric equations for different mangrove species can enable accurate estimation of aboveground biomass. This would therefore improve the quality of the data on total carbon stocks on these ecosystems.

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APPENDIX 1: **Mangrove species in Kenya in their local names and uses.**

| Family | Species name | Local name | Uses |
|---|---|---|---|
| *Rhizophoraceae* | *Rhizophora mucronata* | Mkoko | Timber, firewood and charcoal |
| | *Bruguiera gymnorrhiza* | Muia | Timber and firewood |
| | *Ceriops tagal* | Mkandaa | Timber and firewood |
| *Sonneratiaceae* | *Sonneratia* alba | Mlilana | Timber and firewood |
| *Avicenniaceae* | *Avicennia marina* | Mchu | Firewood and fencing. |
| *Combretaceae* | *Lumnitzera racemosa* | Kikandaa | Firewood and boat ribs |
| *Meliaceae* | *Xylocarpus granatum* | Mkomafi | Timber, firewood and curving |
| | *Xylocarpus moluccensis* | Mkomafi dume | Firewood and fencing |
| *Sterculiaceae* | *Heritiera littoralis* | Msikundazi | Poles, timber and boat mast. |

(Source; Kairo *et al*., 2001)