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COVER CHANGE AND VEGETATION CARBON STOCKS OF MANGROVE FORESTS IN LAMU COUNTY, KENYA

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DECLARATION

This thesis is my original work and has not been presented for a degree in any other University.

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DEDICATION

This work is dedicated to my family and friends for their unwavering support, love, encouragement, and endurance during my study.

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LIST OF ABBREVIATIONS/ACRONYMS

AGB	Above-ground Biomass
ANOVA	Analysis of Variance
BA	Basal Area
BAU	Business - as - Usual
BC	Blue Carbon
BGB	Below-ground Biomass
CI	Complexity Index
CO_2	Carbon dioxide
DBH	Diameter at Breast Height
ESA	European Space Agency
FAO	Food and Agriculture Organization of the United Nations
GIS	Geographic Information System
GHG	Greenhouse Gas
GoK	Government of Kenya
GPS	Global Positioning System
IPBES	International Science Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel for Climate Change
IRD	Institut de Recherche pour le Développement
IV	Importance Value
KFS	Kenya Forest Service
KMFRI	Kenya Marine and Fisheries Research Institute
KWS	Kenya Wildlife Service
LRS	Linear Regeneration Sampling
$Mg\ C\ ha^{-1}$	Megagrams of Carbon per hectare
NbS	Nature-based Solution
NDCs	Nationally Determined Contributions
NMEMP	National Mangrove Ecosystem Management Plan
PA	Paris Agreement
PES	Payment for Ecosystem Services
r^2	Coefficient of determination
R^2	Pearson correlation coefficient
RC	Regeneration Class
SDGs	Sustainable Development Goals
UoEm	University of Embu
UNEP	United Nations Environment Program
UNFCCC	United Nations Framework Convention on Climate Change
WIO	Western Indian Ocean

CONVERSION TABLE

Value (grams)	Unit	Name
10^3	Kg	Kilogram
10^6	Mg	Megagram (tonne)
10^9	Gg	Gigagram
10^{12}	Tg	Teragram
10^{15}	Pg	Petagram

One meter (m) = 100 centimeter (cm)
One hectare (ha) = 10,000 meter squared (m^2)

DEFINITION OF TERMS

Above-ground biomass	All living parts of the plant above the soil including stem, stump, branches, bark, seeds, and foliage.
Allometric equations	Equation derived from the relationship between size and shape of an organism.
Anthropogenic	Human impact (influence) on the environment. It is the effect or the object on the environment resulting from human activity.
Basal area	The total area of the ground covered by trees measured at breast height and expressed in m^2.
Below-ground biomass	All living biomass of live roots. Fine roots of less than 2 mm diameter (suggested) are sometimes excluded because these often cannot be distinguished empirically from soil organic matter or litter.
Blue carbon	Carbon captured by vegetated coastal ecosystems particularly mangroves, salt marshes, and seagrasses.
Carbon dioxide-equivalent	An empirical value used to compare the emissions from various greenhouse gases based on their global warming potential.
Carbon pool	Carbon reservoirs such as sediment, vegetation, water, and the atmosphere that absorb and release carbon. Together, carbon pools make up a carbon stock (the total amount of organic carbon stored in an ecosystem of a known size).
Carbon sequestration	The removal of carbon from the atmosphere and the long-term storage in sinks both in aquatic and terrestrial ecosystems.
Carbon sink	A carbon pool in which more carbon flows in than out.
Cover change	The modification from initial physical land type to another different land type as a result of natural or human interruptions.
Double zonation	A situation in which a species may be abundant in two different disconnected zones of a forest.
Greenhouse gases	The atmospheric gases responsible for causing global warming and climate change.
Juvenile	A young organism: in a mangrove forest, refers to seedlings or saplings, that is, a young mangrove that has not reached exploitable stage.
Natural regeneration	Process where propagules of mangroves are naturally recruited. This may occur in either degraded or non-degraded forest.

ABSTRACT

Mangroves around the world are being threatened by a combination of natural and human factors. Losses of mangroves leads to reduced forest cover and enhanced carbon emission. This study assessed cover change, forest structure, natural regeneration, and carbon stocks of mangroves in Lamu County, Kenya. Landsat images were used to assess cover change from 1990 to 2019, and structural data were obtained in the field using the plot method. Using stratified random design, mangroves were sampled in 152 square plots of 400 m^2 along belt transects established perpendicular to the waterline. Within each plot, all trees with stem diameters $\geq$ 2.5 cm were identified, counted and position marked, while those $<$ 2.5 cm were counted and classified as juveniles. The following parameters were recorded: tree height (m), stem diameter (cm), and canopy cover (%); from which stem density (stems ha^{-1}), basal area (m^2 ha^{-1}), volume (m^3 ha^{-1}), and biomass (t ha^{-1}) were enumerated. Six mangrove species were encountered during this study. Based on importance value index, the dominant mangrove species in Lamu were *Rhizophora mucronata* (Lam.) and *Ceriops tagal* (perr.) C.B. Rob., that accounted for more than 70% of the mangrove formations. Mean standing density of the mangroves was estimated at 2,339±241 stems ha^{-1} (range:1,607-3,092 stems ha^{-1}), with a basal area of 24.26±3.18 m^2 ha^{-1}, and volume of 157.97±15.22 m^3 ha^{-1}. At least 42% of the forest was stocked with low-quality poles, indicating prolonged human pressure. However, natural regeneration rate of 7,342±450 juveniles ha^{-1} observed in the forest was considered adequate to support forest recovery following disturbance. The mean biomass was estimated at 354.98±49.81 Mg ha^{-1}. This translates to vegetation carbon storage of 166.56±23.41 Mg C ha^{-1}. Mangroves in Lamu were estimated at 35,678 ha, representing 62% of the country's total. Approximately, 1,739 ha of mangroves were lost between 1990 and 2019, mainly due to anthropogenic activities, representing a decline of 60 ha yr^{-1}. Total emission from loss and degradation of mangrove vegetation in Lamu was estimated at 41.64 Mg C ha^{-1}; which translates to 9,169.13 Mg CO_2e yr^{-1}. Assuming an offset price of US$10/Mg CO_2e, the estimated cost of avoided emissions in Lamu County is US$91,691.3 yr^{-1} plus other co-benefits such as fishery support and shoreline protection. Mainstreaming mangroves and associated blue carbon ecosystems into national development and climate change agendas could accelerate Kenya's achievements to the Paris Agreement and other processes.

CHAPTER ONE

INTRODUCTION

1.1 Background information

Mangroves occur along the shorelines of over 108 countries (Spalding & Leal, 2021). These forests and their associated coastal ecosystems are carbon-rich environments (Donato *et al.,* 2011; Zhu *et al.,* 2022). Despite occupying only 0.7% of the tropical forest area (Giri *et al.,* 2011; Alongi & Mukhopadhyay, 2015), mangroves account for approximately 3-4% of the global organic carbon sequestration by tropical forests (Alongi, 2020) and 10-15% of the total carbon sequestration in the coastal ocean (Alongi, 2014). This is in addition to the provision of other ecosystem functions and services, such as coastal fisheries (Whitfield, 2017; zu Ermgassen *et al.,* 2020), biodiversity conservation and ecotourism activities (Lee *et al.,* 2014; Runya *et al.,* 2022), reduction of shoreline erosion (Barbier, 2016), and provision of harvestable wood and non-wood resources to millions of people around the world (Lee *et al.,* 2014).

Despite the many benefits, mangroves have been lost and degraded at a rate of 1-2% per year globally, which is much higher than that of any other natural ecosystem (Hamilton & Casey, 2016). Higher rates of up to 8% per year have been reported for mangroves in Southeast Asia and Africa (Bosire *et al.,* 2014; Hamilton & Casey, 2016; Thomas *et al.,* 2017). A report by the International Science Policy Platform on Biodiversity and Ecosystem Services (IPBES) predicted catastrophic loss and degradation of ecosystems globally, including mangroves (IPBES, 2019). However, despite the continued global loss and degradation of mangroves, the rate of loss is decreasing with a recent estimate of approximately 0.16% per year (FAO, 2020; Goldberg *et al.,* 2020; Spalding & Leal, 2021). This is due to increased intervention measures for mangrove conservation and their recognition as effective natural carbon sinks (Flint *et al.,* 2018; Friess *et al.,* 2019).

When mangroves are lost or their areas are converted to other land uses, their services are greatly diminished, along with the capacity of the ecosystems to sequester carbon. This is in addition to the increased risk of carbon stored being released back into the atmosphere (Pendleton *et al.,* 2012; Adame *et al.,* 2021), leading to global warming. Estimates indicate that mangrove deforestation and degradation because of land-use changes can contribute up to 10% of annual Greenhouse gas (GHG) emissions (Alongi & Mukhopadhyay, 2015), with small mangrove harvesting, which is characteristic of the Kenyan utilization pattern, leading to emissions of up to 35.7±76.9 Mg C ha^{-1} yr^{-1} (Lang'at *et al.*, 2014).

Restoration and protection of mangroves is, therefore, recognized as a priority for both climate change mitigation and adaptation (Howard *et al.*, 2017; Taillardat *et al.,* 2018), and several countries have identified measures that harness these benefits in their Nationally Determined Contributions (NDCs) to the Paris Agreement (PA) (Herr & Landis, 2016, Gallo *et al.,* 2017; Lopez, 2021). The inclusion of concrete ocean-based mitigation and adaptation actions allows countries to increase their ambitions to realize their NDCs' commitments (Taraska, 2018). A total of 195 countries (including Kenya) have signed the PA with a commitment to reduce global temperature to below 2°C, preferably to 1.5°C, compared to pre-industrial levels by 2100 through carbon emission reductions (Paris Agreement, 2015). Although several countries have recognized the value of blue carbon (BC) in their NDCs, some are yet to integrate mangrove carbon into their NDCs.

The Kenyan coastline, which extends for approximately 600 km, has over 60,000 ha of mangrove forests (GoK, 2017). The mangroves occur in creeks, protected bays, estuaries, and lagoons (Kirui *et al.,* 2013; Bosire *et al.,* 2016). Sixty-two (62) percent of the mangroves in Kenya occur in Lamu and the surrounding islands. Kenya's mangroves contribute approximately USD 85 million yr^{-1} to the national economy and sustain the livelihoods of approximately 800, 000 artisanal coastal fishermen (Manzi & Kirui, 2021). In the context of climate change, Kenya's mangroves store up to 3% of the country's fossil fuel CO_2 emissions, which are in the order of 16 to 18 million megagram of CO_2e yr^{-1}, with a total of 77 Mg CO_2e currently stored in the country's mangrove areas (Erftemeijer *et al.,* 2022). Although Kenya has emission estimates in its NDCs and emission reduction strategies that include tree planting in the agriculture and forestry sectors, these reforestation and afforestation endeavors rarely include mangroves despite their high carbon sequestration rates and the multiple ecosystem services they provide. This gap provides an opportunity to influence the inclusion of BC ecosystems in the updated

Kenyan NDCs, as well as ensuring that coastal wetlands are accounted for in future global stock-take (GoK, 2020).

Mangroves in Kenya were included in the most recently submitted NDC targets of 2020. Concerning mitigation component goals, the NDC seeks to abate GHG emissions by 32% by 2030 in line with Kenya's sustainable development agenda through a low-carbon and climate resilient development pathway, covering the following sectors: energy, industrial processes and product use, agriculture, land use, forestry, waste, and dealing with the following gases: carbon dioxide (CO_2), methane (CH4), and nitrous oxide (N_2O). Among ocean climate actions in the updated NDCs, Kenya aims to harness the mitigation benefits of the sustainable blue economy, including coastal carbon Payment for Ecosystem Services (PES), implementing the National Mangrove Ecosystem Management Plan (NMEMP), incorporating Nature-based Solutions (NbS) in flood controls, and including coastal wetlands in GHG emissions and removals (GoK, 2020). The NDC tries to bear 21% (equivalent to USD 3,725 million) of the mitigation costs from domestic sources and the remaining 79% (equivalent to USD 14,000 million) from international support in the form of finance, technology, investment, development and transfer and capacity building. So far, despite Lamu County having the highest proportion of mangroves in Kenya (i.e., 62%), few of the past inventory studies on mangroves in this County (e.g., Roberts & Ruara, 1967; Ferguson, 1993; Kairo *et al.*, 2002a etc.,); report data on carbon stocks.

1.2 Problem statement

Mangrove ecosystems support the livelihoods of millions of people directly and indirectly in tropical coastal states, besides serving as very important carbon sinks storing about 3-4 times more carbon per unit area than all other forests. These forests thus have great potential to mitigate climate change and make important contributions to countries' nationally determined contributions (or NDCs) to the Paris Agreement. Although mangrove forests in Kenya have great potential for inclusion in national climate action plans because of their large carbon capture and storage rates, this opportunity has not been fully recognized for climate change intervention. This is despite the recent inclusion of mangroves in Kenya's updated NDCs in December 2020. There is a lack of robust and systematized information on this critical ecosystem, such as updated maps of the extent and distribution, their conditions and trends, inadequate information on carbon stocks as well as quantifiable carbon emission levels. There has also been a challenge with the tier of reporting where the default Intergovernmental Panel for Climate Change (IPCC) Tier 1 values are used instead of Tier 2 or 3 values. Moreover, mangroves in Kenya are at risk associated with over-harvesting, conversion pressure, pollution, and climate change, resulting in a 40% reduction in mangrove cover in the country. The selective removal of straight mangrove poles is prevalent in the country, resulting in the depletion of quality poles. In Lamu County, mangroves are declining mainly through over-harvesting of wood resources. The loss of mangroves leads to reduced forest cover and enhanced carbon emissions. Monitoring forest structure and regeneration are the backbone of successful forest management. Although common in terrestrial forestry, structural studies of mangroves are rare. Thus, this study was conducted in Kenya's largest mangrove cover (i.e., Lamu County) to provide updated data on the status and condition of the forest in terms of cover change, forest structure, natural regeneration, carbon stocks and emission levels.

1.3 Justification of the study

As countries explore the role of NbS in combating the challenges of global warming and climate change, there is a need to understand the status and contribution of ecosystems with limited information. The present study sought to contribute to sustainable mangrove forest management in Kenya through mapping, stock assessment, and planning for the benefits of community livelihoods, biodiversity conservation, and climate change mitigation. The findings of this study are expected to inform national and international processes for sustainable development, climate change, and biodiversity conservation. The findings of this study also have direct applications in the development of harvest plans for mangroves in Lamu County for sustained supply of desired goods and services. The novelty of this study is that the use of simple structural parameters such as stand density, biomass and natural regeneration can help us understand ecosystem health.

1.4 Objectives

1.4.1 General objective

To contribute towards sustainable mangrove forest management in Kenya through mapping, stock assessment, and planning.

1.4.2 Specific objectives

i. To assess the forest structure and carbon stocks in different mangrove management blocks in Lamu County.

ii. To determine the pattern of natural mangrove regeneration in different mangrove management blocks in Lamu County.

iii. To map changes in the areal extent of mangroves in Lamu County between 1990 and 2019.

1.5 Research questions

i. Is there a variation in forest structure and carbon stocks in different mangrove management blocks in Lamu County?

ii. What is the pattern of natural regeneration of mangroves in the different mangrove management blocks in Lamu County?

iii. How did the areal extent of mangroves in Lamu County change between 1990 and 2019?

1.6 Research hypotheses

H_o1. There is no significant difference in both forest structure and carbon stocks among the mangrove management blocks in Lamu County.

H_o2. There is no significant difference in the pattern of natural regeneration of mangroves among the mangrove management blocks of Lamu County.

H_o3. There is no significant variation in the areal extent of the mangroves in Lamu County between 1990 and 2019.

CHAPTER TWO

LITERATURE REVIEW

2.1 Mangrove environment

Mangroves are unique tropical and subtropical ecosystems that grow in intertidal areas between 32°N and 38°S of the equator (Spalding *et al.*, 2010; Bunting *et al.*, 2018; Friess *et al.*, 2019). They have formed a convergent type of evolution with unrelated species (Xu *et al.*, 2017). There are 73 true mangrove species in the world; 9 of these in Kenya and the Western Indian Ocean (WIO) region (Spalding *et al.*, 2010). These forests are established in diverse geomorphological settings, including tide, river, wave-dominated barrier lagoons, composite rivers (high wave-and river-dominated), and drowned bedrock valleys (Thom, 1984). Twilley (1995) added a reef environmental setting that includes carbonate processes. These geomorphological settings are based on the relative influence of tidal amplitude, rivers, rainfall, turbidity, and wave energies on coastal processes (Thom, 1982). Because of the allochthonous input of rich sediments from rivers, river-dominated mangroves are the most productive (Sawale & Thivakaran, 2013).

The ideal conditions for mangrove development include tropical temperatures (above 18 ^{o}C), coastlines free of strong tides and waves, brackish water, and fine-grained alluvial deposits (Tomlinson, 2016). Mangroves survive under variable flooding and salinity stress conditions imposed by the coastal environment through specialized adaptations, including possession of pneumatophores for gaseous exchange, salt exclusion, salt secretion mechanisms, viviparous seed development, and efficient nutrient retention mechanisms (Alongi, 2014). According to Tomlinson (2016), Rhizophoraceae is the most dominant family, because it is widely distributed globally. Moreover, the family also adapts to both extreme and non-extreme conditions. Therefore, in terms of floristic composition, species in this family are common and are easily found in mangrove habitats. The dominant mangrove species along the Kenyan coast are *Rhizophora mucronata*, *Ceriops tagal,* and *Avicennia marina* (GoK, 2017).

2.2 Mangrove forest structure and geomorphology

The structure of mangroves is determined by the magnitude and frequency of tides, seed predation, natural disturbance, human disturbance, nutrients, and stressors, such as siltation (Pool *et al.,* 1977; Smith, 1987; Githaiga *et al.,* 2020). The structural parameters determined by these factors include height, diameter, stem density, basal area (BA), species diversity, and classification of the forest stand into different mangrove community types. Based simply on their growth form, Lugo and Snedaker (1974) identified the following six mangrove community types:

(i) ***Fringing mangroves:*** occur mostly in the gentle coastline that is inundated by daily tides, transporting nutrients into and outside of the forest,

(ii) ***Riverine/estuarine mangroves:*** influenced by freshwater input and occur at the edge of major rivers draining into the oceans, they are highly productive due to the mixing of freshwater with seawater (brackish water),

(iii) ***Basin mangroves:*** have no direct link with the ocean and are mainly found at the back of both fringing and riverine mangroves,

(iv) ***Over-wash mangroves:*** occur on smaller low islands and projections in bays and estuaries, and are typically inundated during each tidal cycle. During every tidal cycle, the system is entirely inundated, unlike fringe forests,

(v) ***Dwarf or stunted mangroves:*** scrub forests common in abnormal or equinoctial tidal reaches, with tidal inundation of a few days per month. These mangroves are found in areas characterized by limited freshwater and nutrients as well as limited inundation by tides. Despite their small size and relatively low area to biomass ratios, dwarf mangroves typically have high leaf litter production rates,

(vi) ***Hammock mangroves:*** are similar to basin mangroves but are formed over accumulated mangrove-derived peat. In general, these six mangrove communities may coexist in the same area.

Earlier studies on mangrove forests in Kenya have observed that tree height, basal areas, and biomass values vary between areas south and north of the River Tana Delta, with the latter being superior. This difference has mainly been attributed to differences in climate, ocean currents, and human pressures (Njiru *et al.*, 2022). Despite the structural differences between mangroves in the north and south of the Tana Delta, the two regions have similar mangrove species (GoK, 2017; Njiru *et al.,* 2022). The coexistence of different community types within the same area is common in Kenya. Mangrove forests to the north of the Tana River are a mixture of basin, fringing, and riverine types that receive freshwater and nutrients from the Tana River, both directly and through seepage. Whereas mangroves within the Tana Delta are mostly riverine, those found south of the delta are a mixture of basin, fringing, riverine, and over-wash, a common occurrence in the mangroves of the Kilifi and Kwale Counties.

Mangroves are dynamic systems that exhibit major structural changes over time. Unsustainable exploitation of mangrove wood products negatively affects the structure and regeneration of the forest (Kairo *et al.,* 2002b; Gillis *et al.,* 2017; Scales & Freiss, 2019), threatening their sustainability, although this could be reversed by monitoring forest conditions and utilization over time. One way to characterize mangrove ecosystems and monitor changes is through the assessment of the forest structure (Cintron & Schaefer-Novelli, 1984). Easily measurable parameters for assessing forest structure include tree height, tree diameter, canopy cover, stem count and regeneration patterns. The datum is used to derive other vegetation attributes such as species importance value (IV), basal area, stand density, volume, biomass, and complexity index (CI) (Kairo *et al.*, 2002b). A similar approach has been used to characterize mangroves in Mexico (Velázquez-Pérez *et al.,* 2019; Snyder *et al.,* 2022), Indonesia (Cameron *et al.,* 2019; Yudha *et al.,* 2021), Bangladesh (Kamruzzaman *et al.*, 2018), Mauritius (Raghbor *et al.*, 2022), and Tanzania (Njana, 2020). Another important approach (De Liocourt's model) has been used to predict future size-class distribution because it applies particularly in mixed forests, where age classes and recruitment by natural regeneration are continuous, while diameter distributions have been used to assess the disturbance effect within forests (Clutter *et al.*, 1983; Sawale & Thivakaran, 2013; Okello *et al.*, 2022).

Previous studies on mangrove forest structure in Kenya have focused on ecosystem health (Mohamed *et al.*, 2009; Githaiga *et al.*, 2020; Okello *et al.*, 2013, 2022), dead wood stocks and productivity (Mugi *et al.*, 2022), value chains (Riungu *et al.*, 2022), and geomorphic and climatic drivers (Njiru *et al.*, 2022). Despite the existence of a structural inventory of mangroves in Lamu County (Roberts & Ruara, 1967; Ferguson, 1993; Kairo *et al.,* 2002a), updated data on forest conditions are lacking. This study aimed to characterize the present stand structure and its variability across different mangrove management blocks in Lamu County.

2.3 Mangrove natural regeneration

Mangrove restoration occurs via natural or artificial regeneration. Natural regeneration utilizes freely falling and dispersed propagules, which is recommended in areas where there is no limitation in the availability of propagules, and tidal water regimes have not been altered. This approach does not entail labor and costs; saplings establish more vigorously, there is less soil disturbance, and the forest is usually similar to the local mangrove species. However, with this approach, excessive wave action may cause poor establishment, the absence of parent trees may result in low propagule supply and there is less control over spacing as well as composition of seedlings.

Artificial regeneration, on the other hand, requires human intervention through direct planting in areas of limited propagule availability or altered tidal regimes. Using this approach, species composition, distribution, and pest infestation can be controlled, employment can be promoted during nursery establishment and out-planting, nursery establishments can be used for training, and community ownership can be enhanced. It also has its disadvantages; direct planting is expensive especially in areas where hydrological regimes have been altered, wrong species may be introduced, community conflicts may arise if they are not fully involved, and monoculture plantations may promote pest infestation (UNEP-Nairobi Convention/USAID/WIOMSA, 2020).

The critical site conditions that influence the natural regeneration of mangroves include inundation class, nature of the substrate, water salinity, erosive power as well as the accretive action of the sea. Threats to mangrove regeneration include high propagule predation rates, unfavorable soil conditions, pollution, lack of seeds and propagules, excessive tidal washing, and weed competition (FAO, 1994). According to FAO (1994), the regeneration potential of a site in terms of seedling size, abundance, and distribution can be determined using Linear Regeneration Sampling (LRS). Effective stocking is determined by the relative presence, abundance, and size of all regeneration classes. A regeneration potential of 2,500 juveniles ha^{-1} is considered adequate to support forest recovery after disturbance (FAO, 1994).

In Kenya, studies on the natural regeneration of mangroves have been conducted in Mwache and Manda Island (Alemayehu & Wekesa, 2017), Tudor Creek (Mohamed *et al.*, 2009), Mida Creek (Kairo *et al.*, 2002b) among others. Most of these studies have ascertained that natural regeneration of mangroves is possible without other forms of disturbance. Clear felling for fuel-wood has seriously impacted mangrove forests at Manda Island, while blank contiguous areas caused by the 1997-98 El Niño rains in Mongoni and Dodori Creek swamps have not recovered to date due to heavy siltation, which altered conditions for natural regeneration (GoK, 2017). Mangroves in Lamu County continue to be threatened, mainly by anthropogenic factors (Hamza *et al.,* 2020). As such, the present study assessed the natural regeneration potential of mangroves in Lamu County to restore forest functionality following disturbance.

2.4 Mangroves and cover change

The first attempt to map the global mangrove distribution, which was carried out as part of the FAO/UNEP Tropical Forest Resources Assessment in 1980, estimated the global mangrove area at 15.6 million ha. Deltares (2014) observed that species distribution across regions was uneven (Fig. 1). Globally, 75% of mangroves occur in 15 countries (Thomas *et al.,* 2017). Asia has the highest mangrove cover globally (38.7%), followed by Latin America and the Caribbean (20.3%), Africa (20.0%), Oceania (11.9%), North America (8.4%), and European overseas territories (0.7%) (Bunting *et al.*, 2018). Bunting *et al.* (2018) estimated the current global mangrove cover at 13, 760,000 ha.

In Africa, mangrove ecosystems occur in three major sub-regions: Western Atlantic region (49%), the Western Indian Ocean (WIO) region (37%), and the Central Atlantic region (14%). The WIO mangrove area is estimated to be 1 million hectares in Somalia, Tanzania, Mozambique, Seychelles, Mauritius, Madagascar, Comoros, South Africa, and Kenya (Bosire *et al.,* 2016). The largest continuous mangrove areas in the region are Lamu and Tana in northern Kenya, Rufiji Delta region in central Tanzania, Zambezi Delta in central Mozambique, and along the north-western coast of Madagascar (at Mahajanga, Nosy Be, and Hahavavy-Diana) (Erftemeijer *et al.,* 2022). In Kenya, there are over 60, 000 ha of mangroves with 62% occurring in Lamu County (GoK, 2017), which is the focus of the present study. Kenya hosts 7% of the total mangrove area in the WIO region. This is the fifth-largest mangrove area in the WIO region, representing approximately 2% of Africa’s mangroves (Erftemeijer *et al.*, 2022).

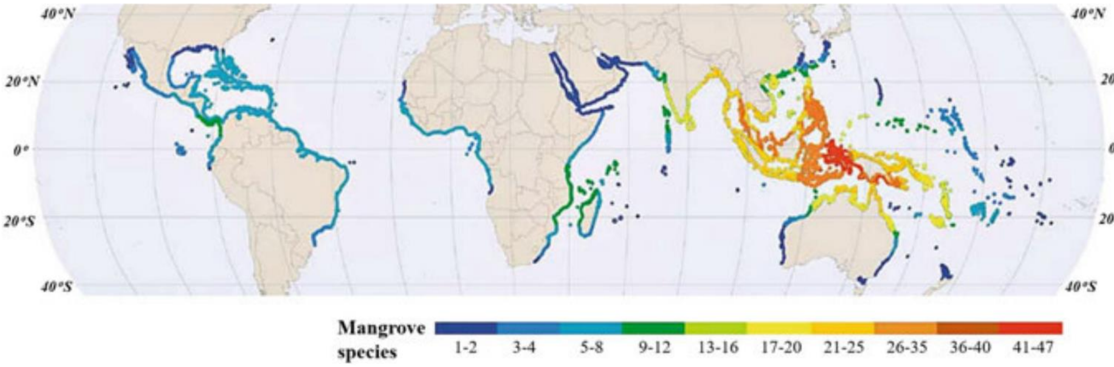


Fig. 1: Global distribution of mangroves and the number of species across regions. Source: Deltares (2014)

Previous studies on mangrove forest cover worldwide have indicated a decline in spatial coverage in recent years (Kirui *et al.*, 2013; Bosire *et al.*, 2014; Hamilton & Casey, 2016). Globally, mangrove deforestation has continued, but with a much-reduced rate of 0.16-0.39% annually (Hamilton & Casey, 2016). Mangrove cover changes in Africa have indicated a decline in cover, with Central and West Africa regions losing about 20-30% of mangrove cover in the last two and a half decades (Feka & Ajonina, 2011).

In Kenya, an assessment of mangrove cover change indicated a reduction in mangrove cover at a rate of 0.74% annually, mainly through anthropogenic activities (Kirui *et al.*, 2013). According to Erftemeijer *et al.* (2022), an overall net mangrove loss of 1,139 ha occurred in Kenya between 1996 and 2016. However, there was a significant increase by 578 ha between 2016 and 2020. In Tudor and Mwache Creeks (Mombasa), over 80% of mangrove loss was recorded and linked to land use changes experienced between 1992 and 2009 (Bosire *et al.*, 2014). There has been a significant increase in mangrove area between 2000 and 2019 in Vanga (235 ha), Ungwana Bay (424 ha), Kilifi (247 ha), and Ngomeni (665 ha), which has been attributed to natural regrowth following sedimentation, restoration efforts, and the implementation of conservation measures (Manzi & Kirui, 2021).

In Lamu County, there was a 12% reduction in mangrove cover between 1985 and 2010 (Kirui *et al*., 2013). Recent mapping of mangroves in Lamu County focused on spatial mapping using old aerial photographs and estimated a cover of 37,350 ha (GoK, 2017). There is no recent spatial-temporal mapping of mangroves in this County, hence, the focus of this study. Unlike GoK (2017), this study used Landsat images supplemented by SPOT and Sentinel images.

2.5 Values of mangroves

2.5.1 Socio-economic values

Mangroves constitute only 0.4% of the forested area worldwide (FAO, 2016). However, despite their small contribution to global forest cover, they make a significant contribution to the harvestable wood and non-wood resources to human society (Lee *et al.,* 2014). This is in addition to the ecosystem services provided by mangroves, including providing a habitat for fish and other animals (Lee *et al.*, 2014; Das *et al.*, 2017), protection against floods and hurricanes, reduction of shoreline erosion (Barbier, 2016), and support for biodiversity and ecotourism activities (Lee *et al.*, 2014).

In the WIO region, approximately 40 million people in coastal areas depend on mangroves for their livelihood (Samoilys & Kanyange, 2008; UNEP/WIOMSA, 2015). Mangroves in the WIO region provide timber, poles, fuel-wood, and protect coastal shorelines against strong storm surges, sea level rise and support artisanal and commercial fisheries (Lee *et al.,* 2014; Bosire *et al.*, 2016; Erftemeijer *et al.*, 2022). In Kenya, mangroves are harvested for construction poles and firewood, and serve as important breeding and feeding grounds for fish and other wildlife (Hamza *et al.*, 2020; Wanjiru *et al.*, 2021). They act as natural barriers by dissipating high-energy waves, and thus provide protection from cyclones, tsunamis, and storm surge impacts (Bosire *et al.*, 2016; Krauss & Osland, 2020). Mangroves also capture and store large amounts of carbon from the atmosphere in the form of biomass and organic matter (Donato *et al.*, 2011; Zhu *et al.*, 2022). The carbon storage potential of mangroves in Kenya, estimated at 600-1,500 Mg C ha^{-1}, is valued at USD 218.96 ha^{-1} yr^{-1} (Kairo *et al.*, 2009; Gress *et al.*, 2017). The total value of Kenya's South Coast mangroves is estimated at USD 6.5 million or USD 1166 ha^{-1} mainly from regulating services (Huxham *et al.*, 2015). Countywide, Kenya's mangroves contribute

Ksh 9.4 billion (equivalent to approximately USD 85 million) in annual economic net benefits to the national economy (Anonymous, 2021; Manzi & Kirui, 2021).

Mangrove ecosystems in Kenya also support research activities and education in addition to providing opportunities for tourism and recreation. The tourism industry around mangrove forests promotes employment, both directly and indirectly. Some local community groups in areas such as Wasini, Gazi, and Mida Creek have exploited the ecotourism potential of mangroves through the construction of boardwalks. At Wasini boardwalk alone, the total income generated through mangrove tourism has been estimated to be more than KSh 2.5 million per annum. Similar ventures are being promoted in other mangrove areas of Mombasa and Lamu Counties (GoK, 2017; Runya *et al.,* 2022). The use of mangrove wood products for construction and fuel-wood is ranked highest among the benefits accrued from Lamu mangrove forests (GoK, 2017).

2.5.2 Role of mangroves in climate change mitigation

Owing to their high carbon capture and storage capacity, mangroves have received increasing attention for their roles in climate change mitigation and adaptation (Lovelock & Duarte, 2019; Macreadie *et al.,* 2021). These forests exist in the United Nations Framework Convention on Climate Change (UNFCCC) mitigation context as important sinks and reservoirs for GHGs (Article 4.1d) (UNFCCC, 2012). Their global mean organic carbon stock is estimated at 738.9 Mg C ha^{-1}, translating to a total carbon stock of 6.17 Pg C (Along, 2020). Most of this carbon is stored in the sediment (Fig. 2). In total, these forests store about 24 Tg C yr^{-1} in the sediment, which represents about 10-15% of coastal sediment carbon storage. This occurs through fast accretion of sediment at 5 mm yr^{-1} and carbon burial at 174 g C m^2 yr^{-1} (Alongi, 2014). Mangroves have high sediment carbon due to their deep organic-rich soils, usually 0.1 to greater than 3 m (Kauffman & Donato, 2012), and anaerobic conditions (Macreadie *et al.,* 2019). Kauffman and Donato (2012) approximated the sediment organic carbon pool to be over 50% of the total mangrove ecosystem carbon stock, which is much higher than in other forest types (Fig. 2). This is due to the saturated nature of mangrove forest sediment, enabling it to remain in an anaerobic state in which carbon has low decomposition (Chmura *et al.,* 2003). Considering this high carbon storage capacity, mangroves are potential candidates for use by nations as a NbS to climate change mitigation and adaptation through their inclusion in the country's NDCs (Herr & Landis, 2016; Taillardat *et al.*, 2018). When degraded, the co-benefits provided by mangroves are greatly diminished, along with the ecosystem's capacity to sequester carbon (Pendleton *et al.,* 2012), and increased emissions further leading to adverse changes in climate (Lovelock *et al.*, 2017; Adame *et al.*, 2021).

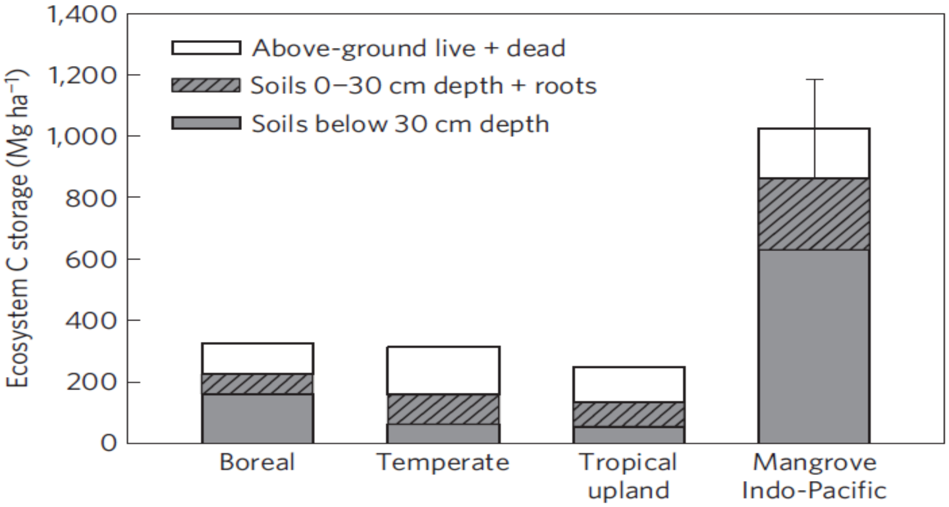


Fig. 2: Carbon storage per unit area in different forests.

Source: Donato *et al.* (2011)

Globally, carbon inventories in mangroves have been conducted in various mangrove ecosystems, such as Micronesia (Kauffman *et al.*, 2011), Sundarbans (Rahman *et al.*, 2015), the Indo-Pacific region (Alongi, 2014; Donato *et al.*, 2011), and the WIO region in the mangroves of Mozambique (Sitoe *et al.*, 2014; Stringer *et al.*, 2015), Madagascar (Jones *et al.*, 2015), and Tanzania (Alavaisha & Mangora, 2016). There is significant variation in mangrove carbon stock estimates in various mangrove ecosystems. This reflects differences in the age of the stands, geomorphology, and methods used (Alongi, 2020). Donato *et al.* (2011) estimated that total carbon stocks in mangroves generally range from 500-1,000 t C ha^{-1} globally, depending on forest type and conditions. In the WIO region, hotspots of blue carbon include Mahajamba (Madagascar), Rufiji Delta (Tanzania), Zambezi Delta (Mozambique), and Lamu (Kenya) (Erftemeijer *et al.*, 2022).

In Kenya, information on the carbon quantities in mangroves is still scarce. Assuming a global average carbon sequestration rate of 6 to 8 Mg CO_2e ha^{-1} yr^{-1} (Bouillon *et al.*, 2008;

Sanderman *et al.,* 2018), the total mangrove area of Kenya is potentially sequestering up to 3% of the country's total annual fossil fuel emissions, which are in the order of 16 to 18 million Mg CO_2 e yr^{-1} (Global Carbon Project, 2021). Earlier studies that focused on above-ground biomass (AGB) (Cohen *et al.,* 2013; Gress *et al.,* 2017), below-ground biomass (BGB), and sediment carbon (Gress *et al.*, 2017) mainly focused on mangroves on the southern coast of Kenya. As no studies have been conducted on the carbon stocks of mangroves in Lamu County, the present study sought to fill this gap.

2.6 Threats to mangroves

Mangrove forests are among the most threatened natural ecosystems on Earth (Giri *et al.*, 2011). The global loss of mangroves is largely attributed to over-exploitation of mangrove wood products, the conversion of mangrove areas to other land uses (Thomas *et al.,* 2017; Bryan-Brown *et al.,* 2020), pollution effects (Deng *et al.,* 2021), and climate change (Ellison, 2015; Ward *et al.,* 2016). Mangroves in the WIO face similar threats, which vary only in the extent of occurrence (Bosire *et al.*, 2016). Habitat degradation, coastal development, natural resource extraction, poor governance, economic drivers, population pressure, poverty, and climate change are the key root causes affecting coastal ecosystems and resources in the WIO region (Bosire *et al.*, 2016).

In Kenya, the major threats to mangrove ecosystems are over-exploitation of wood products, conversion of mangrove areas to other land uses, aquaculture, pollution & sedimentation, diversion & damming of rivers, infrastructure development, and climate change, particularly through sea level rise and aridity. There have also been reports of widespread dieback of the mangrove *Sonneratia alba* caused by wood-boring insect infestations in several areas along the Kenyan coast (Gordon & Maes, 2003; Jenoh *et al.*, 2016). The root causes of this loss and degradation are population growth, poverty, lack of awareness, economic pressure, and poor governance (Bosire *et al.,* 2016; GoK, 2017; Manzi & Kirui, 2021). The country lost 18% of its mangroves between 1985 and 2010, mostly due to human factors (Kirui *et al.,* 2013). Losses have been especially high in the peri-urban mangroves of Mombasa which have lost 70-80% in the past three decades (Bosire *et al.,* 2014). The area available for mangrove restoration in country is approximately 3,351 ha (Erftemeijer *et al.,* 2022).

Illegal harvesting is a major threat to mangroves in Lamu County (GoK, 2017; Hamza *et al.,* 2022; Okello *et al.,* 2022). Kenya's 2019 census indicated that the population of Lamu County has increased by 42,381 people since 2009, thereby increasing demand for mangrove poles. Moreover, the Lamu port in the Southern block is expected to cause human migration into the area, which is expected to increase the pressure on the forest. This will negatively impact mangroves (Bosire *et al.,* 2016). The enactment of the Forest Conservation and Management Act (2016) provided a framework in which sustainable forest management, including mangroves, could be achieved. However, governance and institutional problems persist, exacerbating mangrove degradation. These issues include (i) weak enforcement of existing legislation, (ii) lack of mangrove management policy, (iii) an uncoordinated sectoral approach to management due to overlapping or conflicting mandates, (iv) lack of effective coastal planning, (v) inadequate institutional capacities, and (vi) poor stakeholder or community participation (GoK, 2017).

CHAPTER THREE

MATERIALS AND METHODS

3.1 Description of the study area

This study was conducted on the northern coast of Kenya in Lamu County, at latitudes between 1^{o} 45’ and 2^{o} 20’ South and longitudes 40^{o} 44’ and 41^{o} 30’ East (Fig. 3).

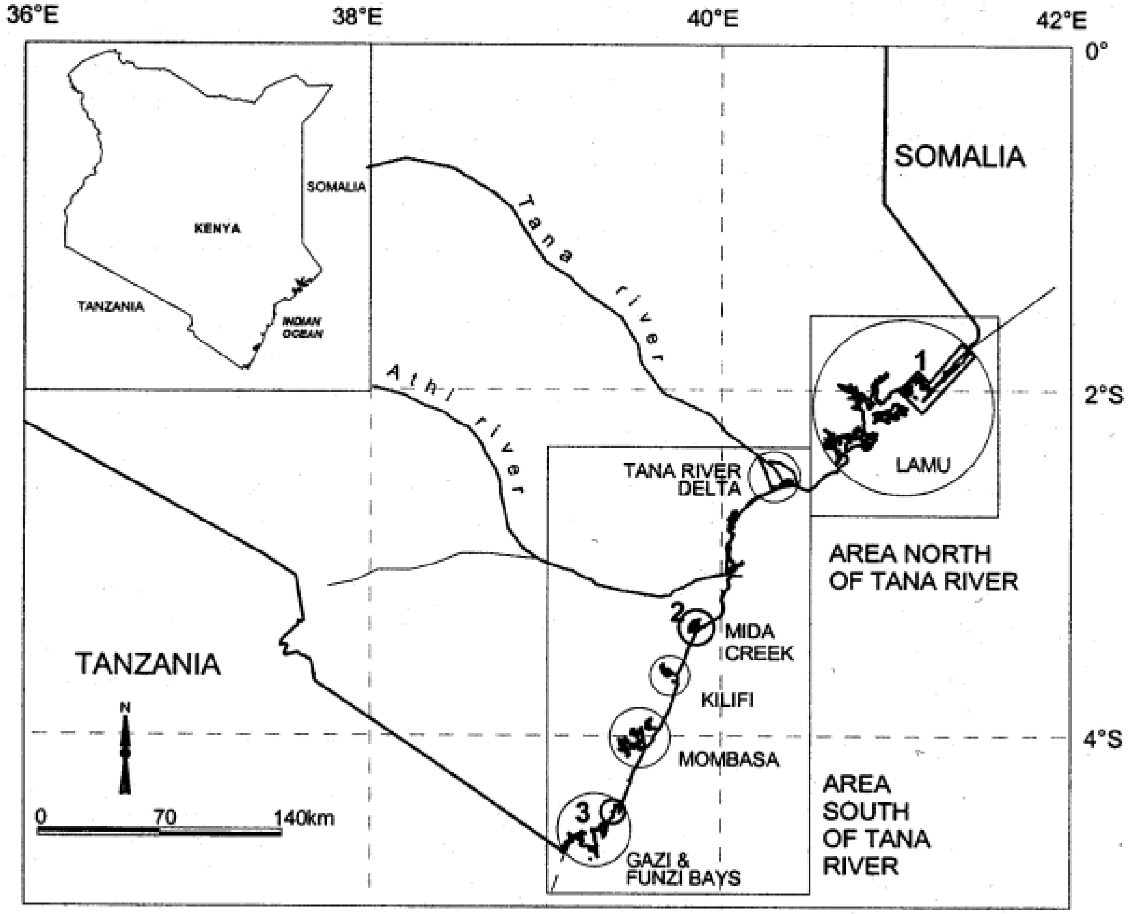


Fig. 3: Location of mangroves in Lamu County along the Kenyan coastline. The Kenyan mangroves are divided into two broad regions: north and south of the Tana River. Source: Modified from Kirui *et al.* (2013)

3.1.1 Geomorphology

Lamu County lies at an altitude of 0 and 50 m above sea level, with a generally flat topography and coastline extending to approximately 130 km. The major geomorphological features of the county are coastal lagoons, multiple small islands, large intertidal areas, sand dunes, and an expansive seascape that supports healthy mangrove stands and other coastal ecosystems (County Government of Lamu, 2018). The land comprises of geological structures with variations in sediment grain size, porosity, permeability, compaction, and texture. Lamu County is characterized by an ancient deltaic shoreline and significant upwelling of nutrient-rich bottom waters (Kamau *et al.,* 2020). Specific to mangrove areas, the sediments are predominantly unconsolidated colarines with poor water-holding capacity and extreme alkalinity (Boxem *et al.,* 1987). No permanent rivers drain into the mangroves of Lamu County. Freshwater is supplied by seasonal streams and groundwater aquifers (County Government of Lamu, 2018). Offshore, there is evidence of major underwater rivers draining into the Indian Ocean, which are now separated by fossilized underwater deltaic features (Caswell, 1953).

3.1.2 Climatic conditions

Lamu County is characterized by a hot and humid tropical climate, with annual rainfall ranging from 500 to 900 mm and a mean temperature of 27 °C (Camberlin, 2018). Relative humidity is high throughout the year, reaching 90% during the rainy season, which is an ideal environment for supporting mangrove growth and development (Spalding *et al.*, 2010). Monsoon winds strongly influence the rainfall seasons along the coast, resulting in two rainy seasons. Long rains occur between March and May during the South East Monsoon (Kuzi), while short rains occur during the North East Monsoon (Kaskazi) between October and December (Bosire *et al.,* 2016). The months from January to March and August to October are usually hot and dry. The temperature (^{o}C) and rainfall (mm) patterns in Lamu County are shown in Fig. 4.

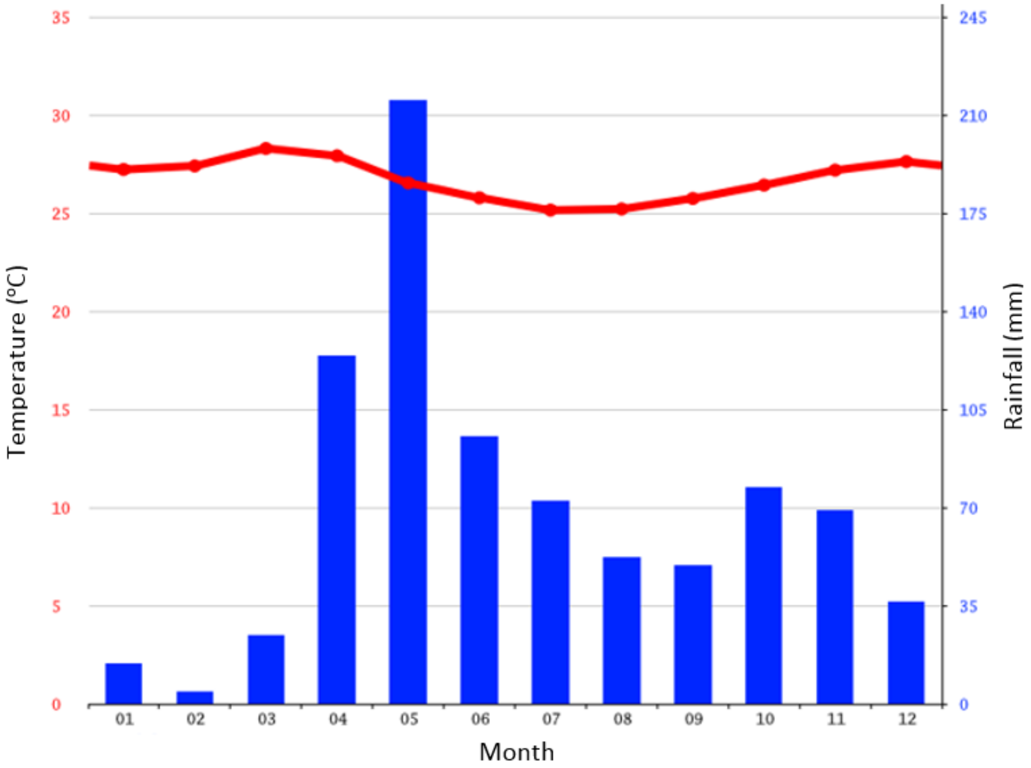


Fig. 4: Temperature (ºC) and rainfall (mm) patterns in Lamu County.
Source: Climate.data.org

3.1.3 Mangroves in Lamu County

Lamu County hosts the highest proportion of Kenya's mangroves, where the protective influence of barrier islands off the coast and groundwater seepage have resulted in an abundance of mangroves (Erftemeijer *et al.*, 2022). The total mangrove area in Lamu County is estimated at 37,350 ha, represented by nine species. The dominant mangrove forest types in the county are pure and mixed stands of *Rhizophora mucronata*, accounting for 38.1% of the total mangrove forest in the area. Other prominent mangrove formations in Lamu County are pure stands of *Avicennia marina* and *Ceriops tagal*, occurring in the landward and mid-zones, respectively (GoK, 2017). The mangroves of Lamu County, combined with the nutrient-rich Somali Current, create a conducive habitat for some of the greatest inshore densities of finfish and crustaceans in Kenya (Samoilys *et al.*, 2015). According to Kenya's NMEMP (2017 - 2027), mangroves in Lamu County are classified into five management blocks: Northern swamps, Northern central swamps, Mongoni and

Dodori creek swamps, Pate Island swamps, and Southern swamps. This study concentrated on the five management blocks (within the boxes in Fig. 5).

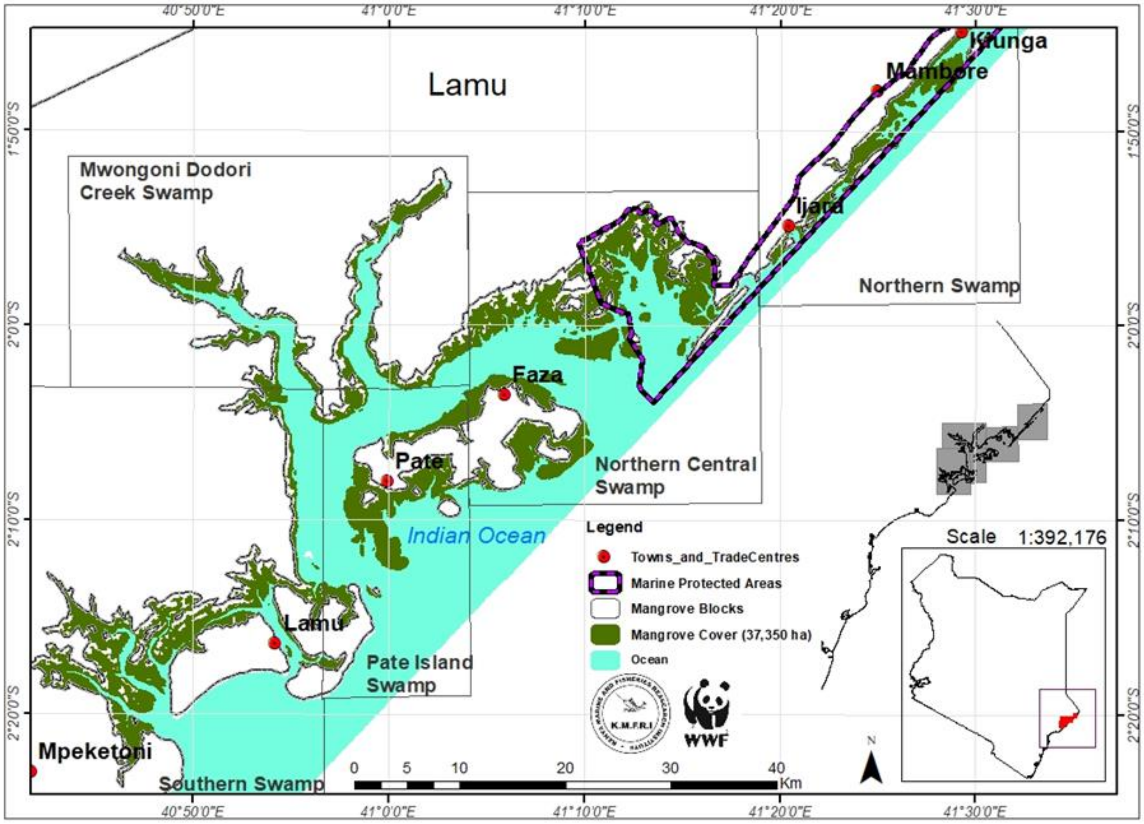


Fig. 5: Mangrove forests in Lamu County, Kenya; showing mangrove management blocks

Commercial harvesting of mangrove poles is a traditional activity of the Lamu people (Hamza *et al.,* 2020). For centuries, mangroves in Lamu County have served as important trade commodity to the Middle East and other treeless countries. Between 1910 and 1911, the export of mangrove poles was ranked the second most valuable item in Lamu County after wild rubber (Idha *et al.,* 1998). The export reached an annual peak of 75,000 scores (1.5 million poles) between 1947 and 1956 (Martin & Martin, 1978; Manzi & Kirui, 2021). Increased pressure on mangrove forests led to a Presidential ban on further export of mangrove wood from Kenya in 1982. Even with the export ban, domestic utilization of mangrove wood resources continued to grow with increasing demand (Bosire *et al.,* 2016; Hamza *et al.,* 2020).

Harvested poles are traded in urban centers along the coast for building and construction (Riungu *et al.,* 2022). Most of the commercial harvesting of mangroves is concentrated in the Northern central, Pate Island, and Southern swamps. Part of the Northern central and the entire Northern swamps constitute the Kiunga Marine Biosphere Reserve, where commercial harvesting of mangroves is not allowed. The reserve is jointly managed by the Kenya Forest Service (KFS) and Kenya Wildlife Service (KWS). Harvested mangrove products in Kenya are grouped into different utilization classes based on pole height and butt diameter. The *Mazio*-sized poles (butt diameter: 8.0-14.0 cm) are the most popular; followed by *Boriti* (11.5-13.5 cm), *Pau* (4.0-7.9 cm) and *Fito* (2.5-3.9). Others are *Nguzo* (13.6-20.4 cm), *Vigingi* (20.5-34.5) and *Mbao* (> 30.4 cm) (GoK, 2017).

3.2 Study design

A stratified random sampling design was used for the vegetation surveys. Stratification was performed at two levels: the land-use level that distinguished mangroves from non-mangrove forests, and the tree species level. Mangroves have a clear tonality and texture which makes it easy to discriminate them in satellite images (Howard *et al.*, 2014). The area boundary was defined using satellite imagery to cover the Lamu area and was restricted to mangrove ecosystems. Study sites were identified and corresponding zones that were representative of the area across the topographic gradient were classified based on vegetation type and stand conditions from satellite imagery.

A total of 152 square plots of 400 m^2 each, were established along belt transects perpendicular to the shoreline and sampled: 38 in Northern swamps, 39 in Northern Central swamps, 14 in Pate swamps, 45 in Southern swamps, and 16 in Mongoni and Dodori creeks. This depended on species formation and the spatial extent of the forest between seaward and landward edges. All sampling plots were georeferenced using a Garmin hand-held Global Positioning System (GPS) 76 receiver for future reference.

3.3 Field sampling

3.3.1 Assessment of forest structure

Within each plot, all trees with a stem diameter at breast height (DBH) ≥ 2.5 cm (treated as adult trees) were identified, counted, and their positions marked. The following vegetation parameters were measured: tree height (m), stem diameter (cm), and canopy cover (%). These data were used to derive the ecological importance of the species (%) (a measure that indicates the relative contribution of a plant species to the structure of a stand), stand density (stems ha^{-1}), basal area (m^{2} ha^{-1}), volume (m^{3} ha^{-1}) (Cintron & Schaffer-Novelli, 1984), standing biomass (Mg ha^{-1}) (Komiyama *et al.*, 2005, 2008), and complexity index (a measure of how complex or structurally developed a vegetation stand is) (Holdridge *et al.*, 1971).

Canopy cover (%) per plot was estimated from the area of bare ground that one would see if flying above the tree canopy and sky visibility (Zhou *et al.,* 1998; Okello *et al.,* 2022). Tree height and diameter were estimated using graduated poles and forest calipers, respectively. Stem diameter was measured 130 cm above ground following (Cintron & Schaeffer-Novelli, 1984). In the case of *R. mucronata*, a structurally complex species (Dahdouh-Guebas & Koedam, 2006), the stem diameter was measured 30 cm above the highest prop root (Komiyama *et al.*, 2005). For stems forked below 130 cm, individual branches in a clump were measured separately. Stems of mature trees were further grouped into utilization classes with different size categories for inventory purposes according to Kairo *et al.* (200b).

3.3.2 Assessment of forest quality

The quality of the *in-situ* mangrove trees was used to assess the harvesting pressure on the forest. All trees were categorized into three quality classes (or Forms) based on the suitability of the lead stem for construction. Form 1 trees have straight poles with a ready market for construction, Form 2 trees have crooked poles requiring minor repair before being used for construction, and Form 3 trees have crooked poles unsuitable for building (Kairo *et al.,* 2002b). A high proportion of Form 2 and 3 trees is a good indicator of degraded forest. Quality classes are usually complemented by stump count as an indicator of exploitation to identify anthropogenic disturbances in forests.

3.3.3 Assessment of natural regeneration

Mangrove trees with DBH < 2.5 were classified as juveniles. Linear regeneration sampling (LRS) was used to assess the status and pattern of natural regeneration, following the approaches of Sukardjo (1987) and FAO (1994). Juveniles of different species were identified and grouped according to their height classes and arbitrarily assigned Regeneration Classes (RC) I, II, or III. Seedlings less than 40 cm in height were classified as RCI. Saplings between 40 cm and 150 cm in height were classified as RCII, whereas those with heights greater than 1.5 m and less than 3 m but with a DBH < 2.5 cm were assigned as RCIII according to FAO (1994). The ratio of the three regeneration classes was used to determine the adequacy of natural regeneration. Saplings above 40 cm in height were further classified as "established regeneration", whereas those below were referred to as "potential regeneration" (FAO, 1994). A summary of the mapping and structural inventory methodology applied in this study is presented in Fig. 6.

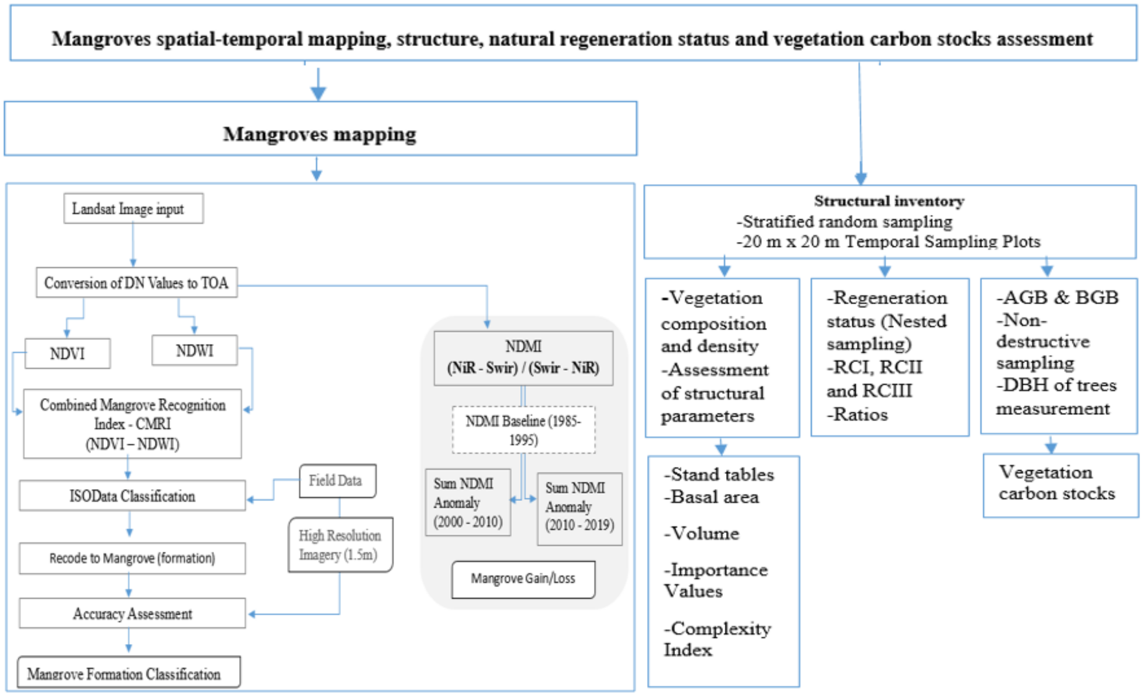


Fig. 6: A summary of mapping and structural inventory methodology applied in this study

3.3.4 Cover and cover change analysis

Landsat images dating from 1990 were accessed and reviewed for quality and cloud-free scenes. Data were freely acquired from the US Geological Survey (USGS) Landsat satellites 5, 7, and 8 (path 165, rows 61 and 62). This was complemented by SPOT and Sentinel images acquired from the European Space Agency (ESA; Table 1).

Table 1: Information on global survey data used in the analysis and mapping of mangroves in Lamu County

Sensor	Resolution (meters)	Raw and column	Epochs	Source
Thematic Mapper	30	165/061 & 165/062	1990-2000	Global land survey data
Enhanced Thematic Mapper plus	30	165/061 & 165/062	2000-2010	Global land survey data
Observation Land Imager	30	165/061 & 165/062	2018/2019	Global land survey data
Sentinel Imagery	10	073/159 & 074/233	2018/2019	Global land survey data
Spot Imagery	1.5	004/465	2018/2019	Global land survey data

3.3.4.1 Pre-processing of the acquired data

Geo-referencing of the acquired spatial data (Landsat and Sentinel imagery, Ground Control Points (GCP)) to a common global geo-referencing system that is World Geodetic System (WGS) 1984 was performed. The data were then registered to the local area coordinate system of Universal Transverse Mercator (UTM) Zone 37S with first-degree polynomial adjustment using ArcGIS geo-referencing tools. Normalization process was performed on all images to eliminate variations caused by the solar angle and the Sun-Earth distance. The normalization process entailed conversion of Digital Numbers (DN) to top of atmosphere reflectance in two steps. First, the DN was returned to values that could be compared between scenes. Second, the values obtained in step one were converted to account for differences in solar irradiance owing to the earth/sun geometry (orbital distance and tilt). The conversion was carried out in ArcGIS using a raster calculator tool, and the scene variables were sourced from metadata files acquired along with the imagery. Image enhancement was performed using the image analysis window of ArcGIS to improve the clarity and quality of the outputs.

Geometric correction was performed to improve the geo-location to a Root Mean Square (RMS) of 0.5 of pixels. The area of interest thus included mangrove cover and the adjacent land uses and cover along the area of interest area. The corrected images were then grouped into subsets and clipped to include only areas within and adjacent to areas where mangroves are likely to occur.

3.3.4.2 Data processing and image classification

Time series analysis was performed using Normalized Difference Moisture Index (NDMI) to generate long-term trends that informed the assessment of mangrove gain and loss. Unsupervised classification was performed prior to fieldwork to retrieve different spectral classes for comparison of the best result method. An NDMI layer was developed to act as a reference threshold (baseline) by averaging annual NDMI values from 1985 to 1995 and an annual NDMI layer was generated for successive years after 1995, which was subsequently subtracted from the threshold to create an annual NDMI anomaly layer. To compute the change layer within a given time span, the annual anomaly layers in the corresponding years within the predefined time lapse were summed and classified using the standard deviation. Using this approach, the study was able to determine clear-cut areas or naturally degraded areas (mangrove loss) because such areas record negative values, mostly occurring two standard deviations from the mean, whereas in contrast, areas that have experienced regeneration acquire positive values.

This study assessed mangrove gains and losses between 2000 and 2010 and 2010-2019. This was followed by intensive field validation campaigns. The combined mangrove recognition index (CMRI) was used to discriminate mangrove cover from other non-mangrove areas. NDMI was used to generate long-term trends in mangrove gain and loss within the study area. A supervised classification algorithm was then used to map the formation of the different mangrove species. Ground Control Points (GCPs) were collected using GPS 76 in the Universal Transverse Mercator (UTM) coordinate system. To minimize errors resulting from GPS accuracy, the collected GCPs were ensured to be within a 10 m radius of the same mangrove species formation. The acquired data were converted to top-of-atmosphere spectral reflectance using Joint Research Centre’s IMPACT Toolbox.

Accuracy assessment and validation of the results of the classification process are key to assessing the representativeness of classified phenomena in the real world in image analysis (Giri, 2016). This study used ArcGIS 10.6, Google Earth Pro, and part of the ground-truth data obtained during field campaigns to assess classification accuracy. A sample of the data generated from classified satellite imagery was confirmed using ground-truth data collected during field campaigns. A confusion matrix was then generated to calculate the producer's and user's accuracies and the kappa coefficient (K^) was assessed. An error matrix was generated using the equation proposed by Bishop *et al.* (1977) as in Kamal and Johansen (2017):

$$\mathrm{K}^{\wedge} = \frac{\sum_{\mathbf{i=1}}^{\mathbf{r}} \mathbf{xii} - \sum_{\mathbf{i=1}}^{\mathbf{r}} (\mathbf{Xi+} * \mathbf{X+i})}{\mathbf{N}^{2} - \sum_{\mathbf{i=1}}^{\mathbf{r}} (\mathbf{Xi+} * \mathbf{X+i})} \qquad \textbf{Eq. 1}$$

Where: K^ =Kappa coefficient, r = Number of rows in the matrix, xii = Number of observations in row i and column i (the major diagonal in the confusion matrix), xi+ and x+i = Marginal totals of row i and column i, N= Total number of observations.

3.4 Data analysis

3.4.1 Stand composition

All statistical analyses were performed using Minitab 17.0 program, and the *p-value* was set at a significance level of 0.05. Normality and homogeneity of variance tests were carried out using Kolmogrov-Smirnov and Levene's tests, respectively. All data that met the normality assumption were further analyzed for significant differences in structural attributes (i.e., height, diameter, biomass, and stocking rates of adult trees, juveniles, and stumps) and carbon stocks between the management blocks using One-way Analysis of Variance (ANOVA). Tukey's HSD test was performed to separate the means when a significant difference ($p < 0.05$) was observed. Pearson's correlation coefficient was used to determine the relationships between high-quality poles (Form 1) and stump density as well as canopy gaps and natural regeneration. Linear regression analysis was performed to test the relationship between above-and below-ground biomass.

Graphical presentation of the data was made using Microsoft Excel 2016 and STATISTICA 8.0 Software. Tree basal area, stand density, and importance value were derived according to Cintron and Schaeffer-Novelli (1984) and Kershaw *et al.* (2016) - Eq. 2, 3, and 7. The complexity index (Holdridge *et al.,* 1971) of each management block was assessed based on species composition, basal area, tree height and stand density using the relation in Eq. 8.

Basal area (m^2 ha^{-1}) = $\left(\frac{\text{sum of cross-sectional area}}{\text{plot area (m2)}} \text{ x } 10{,}000\right)$ **Eq. 2**

Stem density (stems ha^{-1}) = $\left(\frac{\text{number of stems in plot}}{\text{plot area (m2)}} \text{ x } 10{,}000\right)$ **Eq. 3**

Relative density = $\left(\frac{\text{number of individuals of a species}}{\text{total number of individuals}} \text{ x } 100\right)$ **Eq. 4**

Relative dominance = $\left(\frac{\text{total basal area of a species}}{\text{basal area of all species}} \text{ x } 100\right)$ **Eq. 5**

Relative frequency = $\left(\frac{\text{frequency of a species}}{\text{sum frequency of all species}} \text{ x } 100\right)$ **Eq. 6**

Importance Value (%) = relative density + relative frequency + relative dominance **Eq. 7**

Complexity Index = number of species × basal area (m^2 ha^{-1}) × mean height (m) × stem density (stems ha^{-1}) × 10^{-5} **Eq. 8**

Volume (m^3) = $(\pi D^2/4) \times h \times f$ **Eq. 9**

Where: $\pi = 3.141$, D=DBH (cm), h = tree height (m), and f = form factor.

In forestry, the form factor is determined by the manner in which the stem tapers, that is, by the decrease in diameter from the tip. A form factor of 0.7 for Lamu mangroves (Ferguson, 1993) was used to estimate the average volume in the present study. The average volume was further multiplied by stem density to obtain the total volume. With the rotation cycle of mangroves in Lamu estimated at 20 years (Roberts & Ruara, 1967; GoK, 2017), this was applied to the total volume obtained to estimate how much volume can be harvested annually during the first rotation if clear felling is employed.

3.4.2 Estimation of plant biomass and vegetation carbon stocks

This study used generalized biomass equations for mangroves, with stem diameter as the dependent variable (Komiyama *et al.*, 2005, 2008; Eq. 10 and 11) and localized species-specific wood densities (Gillerot *et al.*, 2018) due to the lack of robust allometric equations for Kenya mangroves. The total vegetation biomass was then computed as a sum of the living below-ground biomass (BGB) and above-ground biomass (ABG):

AGB = $0.251pD^{2.46}$ ($R^2 = 0.98$, Komiyama *et al.*, 2005) **Eq. 10**

BGB = $0.199p^{0.899}D^{2.22}$ ($R^2 = 0.95$, Komiyama *et al.*, 2008) **Eq. 11**

Where: AGB = above-ground biomass (kg), BGB = below-ground biomass (kg), ρ = wood density (g cm^{-3}), and D = DBH (cm).

Biomass values were then converted to carbon equivalents by multiplying with a conversion factor of 0.50 and 0.39 for AGB and BGB, respectively, following procedures in Kauffman and Donato (2012).

Total vegetation carbon stock (t ha^{-1}) **=** AGC + BGC **Eq. 12**

Where: AGC = above-ground carbon stock and BGC = below-ground carbon stock.

The vegetation carbon stock was then multiplied by 3.67 (the molecular weight ratio of CO_2 to C) to obtain the CO_2 equivalent (Kauffman & Donato, 2012; Howard *et al.*, 2014). To predict the nature of future mangrove forests in Lamu County, this study used De Liocourt's predictive model (Clutter *et al.*, 1983).

$y = aq^{n-1}$ **Eq. 13**

Where: y = predicted density for a given class, a = observed density in the highest class, q = a constant and is the ratio of number of stems between successive classes, and n = number of classes.

The densities in successive diameter classes were plotted against the diameter class, and the distribution was represented as an exponential curve of the form:

$z = ke^{-ax}$ **Eq. 14**

Where: *z* = number of trees in diameter class x, e = base of natural log (2.718), and k and a = constants.

CHAPTER FOUR

RESULTS

4.1 Floristic composition

A total of six (6) mangrove species, occurring in either mono or mixed stands, were encountered in this study in both the adult and juvenile stages. Based on the species importance value (IV) computation, mangroves in Lamu County are dominated by *R. mucronata* and *C. tagal*, with IV values ranging from 77.99% in Mongoni and Dodori creek swamps to 197.11% in Northern swamps (Table 2).

Table 2: Floristic composition and importance value of mangroves in different mangrove management blocks of Lamu County

Management block	Species	Relative Values (%)			IV* (%)
		Dominance	Density	Frequency	
Northern swamps	*Avicennia marina*	2.9	3.83	7.35	14.08
	Bruguiera gymnorhiza	0.83	1.43	7.35	9.61
	Ceriops tagal	4.41	15.25	19.12	38.78
	Rhizophora mucronata	75.4	68.76	52.94	**197.11**
	Sonneratia alba	16.46	10.72	13.24	**40.42**
Northern central swamps	*Avicennia marina*	7.62	3.08	6.49	17.19
	Bruguiera gymnorhiza	3.76	1.9	14.26	19.92
	Ceriops tagal	13.67	38.35	25.97	**77.99**
	Rhizophora mucronata	59.68	52.47	45.45	**157.63**
	Sonneratia alba	15.28	4.2	7.79	27.27
Mongoni and Dodori creek swamps	*Avicennia marina*	10.49	10.46	8.33	29.28
	Bruguiera gymnorhiza	2.4	2.96	16.67	22.03
	Ceriops tagal	19.3	31.25	25	**75.55**
	Rhizophora mucronata	34.39	40.4	30.56	**105.35**
	Sonneratia alba	25.06	10.81	11.11	46.98
	Xylocarpus granatum	8.36	4.12	8.33	20.81
Pate Island swamps	*Avicennia marina*	7.59	5.84	9.68	23.11
	Bruguiera gymnorhiza	3.49	18.09	12.9	34.48
	Ceriops tagal	0.63	3.99	16.13	20.75
	Rhizophora mucronata	48.22	54.87	41.93	**145.02**
	Sonneratia alba	29.37	12.19	16.13	**57.69**
	Xylocarpus granatum	10.7	5.02	3.23	18.95
Southern swamps	*Avicennia marina*	2.56	7.41	1.71	11.68
	Bruguiera gymnorhiza	4.29	20.37	6.96	31.62
	Ceriops tagal	26.85	26.85	10.64	**64.34**
	Rhizophora mucronata	64.99	39.82	77.61	**182.42**
	Sonneratia alba	1.32	5.56	3.08	9.96

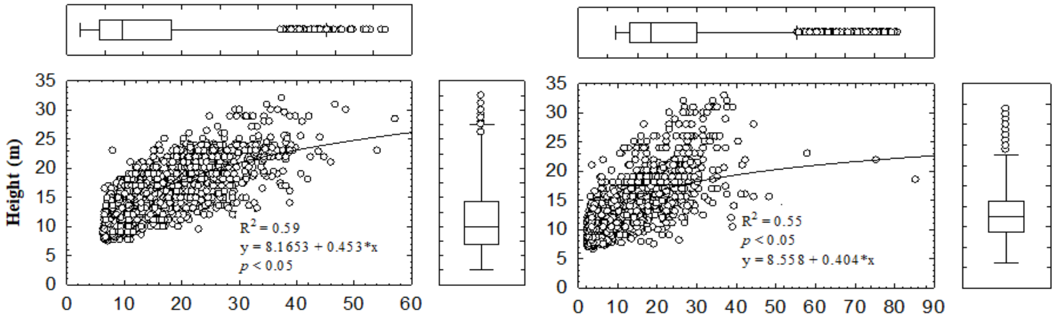
*Figures in bold represent the most important species per management block

The stocking rate ranged from 1,607±129 stems ha^{-1} in the Northern swamps to 3,092±213 stems ha^{-1} in the Southern swamps (mean:2,339±241 stems ha^{-1}). There was a significant difference in the stocking rates between the five management blocks ($F_{(4, 147)} = 6.65$, $p < 0.05$). *Rhizophora mucronata* was the highest contributor to the total stem density across the five management blocks, accounting for 55.8%, followed by *C. tagal* at 23.11%.

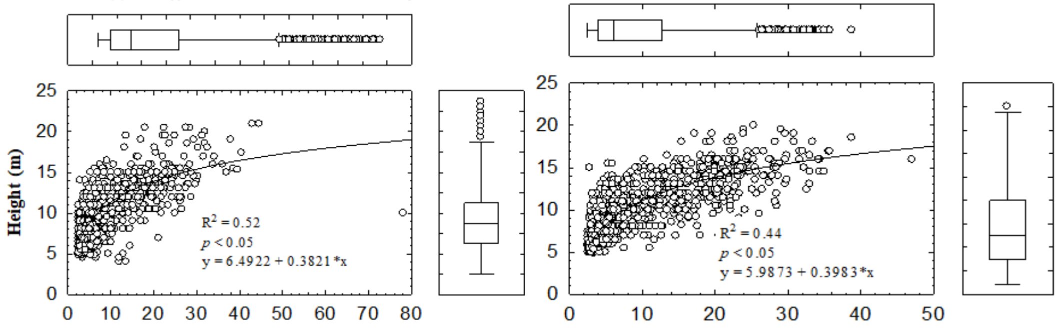
The lowest intertidal zone was occupied by *S. alba* in pure or mixed stands with *R. mucronata.* The mid-intertidal zone was occupied by pure or mixed stands of *C. tagal, A. marina,* and *X. granatum* while the highest intertidal zone was dominated by *A. marina. Bruguiera gymnorhiza* lacked a distinct zonation but occurred interspersed with *R. mucronata* and *C. tagal.* In some places, *A. marina* occurred on the seaward and landward edges, resulting in double zonation. Generally, it was the most widely distributed mangrove species in Lamu County. Despite *S. alba* dominating the lowest intertidal zone, *R. mucronata* and *A. marina* were the dominant species along small creeks.

4.2 Structural attributes

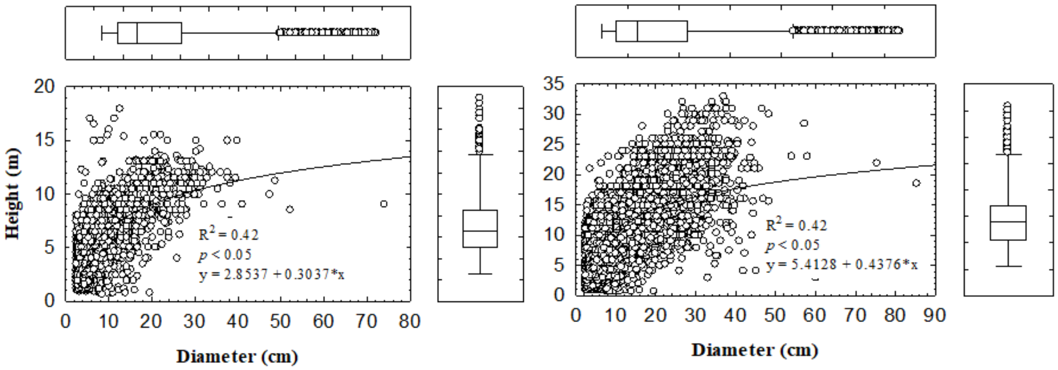
Across all the management blocks, 50% of the tree diameters and heights were between 3.9 - 10.8 cm and 5.5 - 12 m, respectively. Stem diameter ranged from 7.2 cm in the Southern swamps to 14.96 cm in the Northern swamps, with a mean value of 9.36 cm. The mean tree height ranged from 5.0 m in the Southern swamps to 15.0 m in the Northern swamps, with a mean height value of 14.9 m. There were significant differences in both diameter ($F_{(4, 147)} = 20.96$, $p < 0.05$) and height ($F_{(4, 147)} = 75.65$, $p < 0.05$) between the management blocks. A Tukey post hoc analysis showed that the mean diameter in Northern swamps was significantly greater than that of other blocks ($p < 0.05$). For height, Tukey's post-hoc analysis revealed that the mean height varied significantly across the blocks ($p < 0.05$), except in Pate Island and Mongoni & Dodori creek swamps ($p > 0.05$). Fig. 7 shows the scattergrams of height-diameter of mangroves in Lamu County.


(a) Northern swamps

(b) Northern Central swamps


(c) Mongoni and Dodori creek swamps

(d) Pate Island swamps


(e) Southern swamps

(f) Lamu combined

Fig. 7: Scattergrams on height-diameter of mangroves in Lamu County.
The box plots display the percentile distribution of DBH and height in the management blocks. The extremities of the plot correspond to the maximum and minimum observations in the dataset. The ends of the boxes are positioned at the 25% and 75% percentiles of the data

Stocking densities in all the management blocks decreased with increasing diameter classes, with Southern swamps recording the highest number of small-sized poles (Fig. 8). This pattern was, however, slightly distorted in mangrove harvesting areas of Northern central swamps and Southern swamp forests where *Boriti*-sized poles (DBH range: 11.5 to 13.9 cm) were significantly reduced (Fig. 8b, e, Table 3). *Fito* (DBH = 2.5 - 3.9 cm) and *Pau* (DBH = 4.0 - 7.9 cm) were the dominant utilization classes in the entire system while *Banaa/Mbao* (>30.5 cm) were the least abundant.

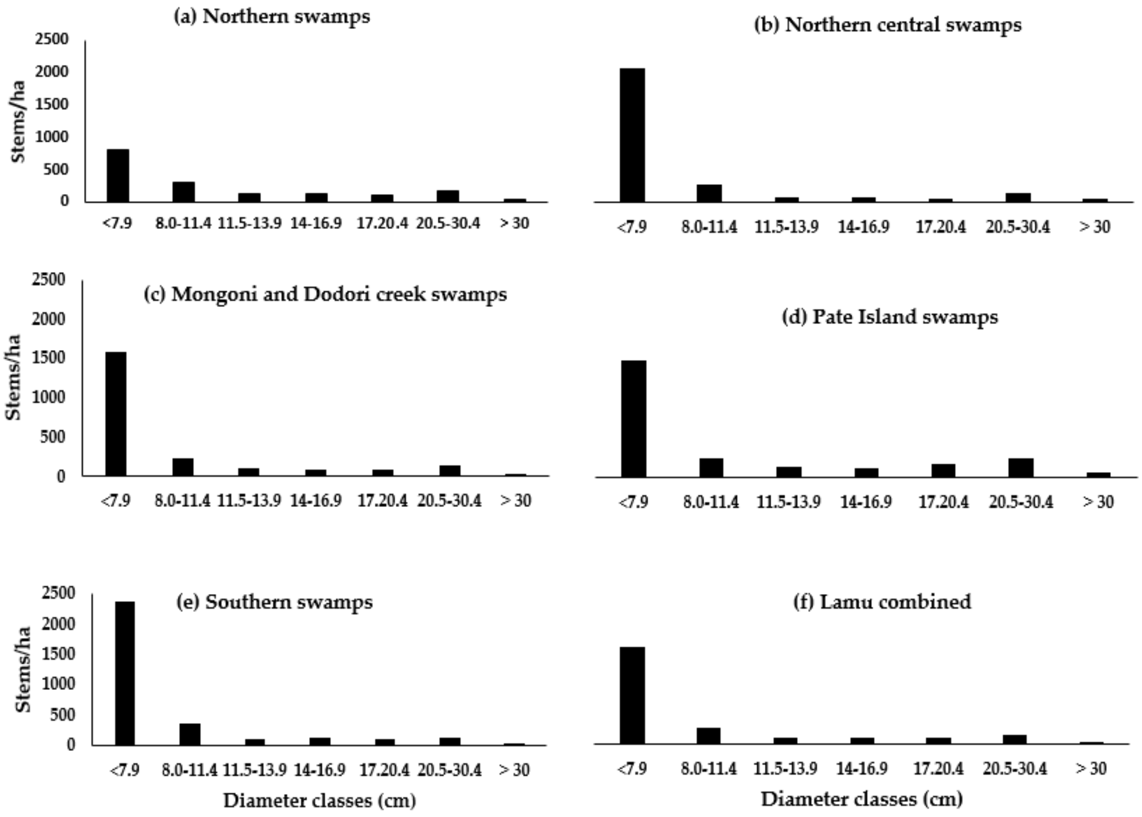


Fig. 8: Size-class distribution of mangroves in Lamu County.

A typical reversed-J curve was witnessed in all management blocks, which is an indicator of non-even-aged forests with natural regeneration

The standing biomass of mangroves in Lamu County ranged from 183.22 to 392.22 Mg ha^{-1} (mean±*s.e*:255.7±36.24 Mg ha^{-1}). Together with the root biomass, the mean vegetation biomass was 354.98±49.81 Mg ha^{-1} (range:254.63 to 541.98 Mg ha^{-1}) (Table 3). The highest mean biomass of 541.98±214.75 Mg ha^{-1} (range:154.54 - 1,010 Mg ha^{-1}) was recorded in the Northern swamps (Table 3). The mean biomass was significantly different across the five management blocks ($F_{(4, 147)} = 10.86$; $p < 0.05$). There was a strong positive correlation between above-and below-ground biomass ($r^2 = 0.99$, $p < 0.05$).

Table 3: Structural attributes of mangroves in Lamu County (x±*s.e*)

Management block	**Utilization class in Swahili names (DBH in cm)**							**Structural attributes**						
	Fito & Pau <8.0	**Mazio 8.0 - 11.4**	**Boriti 11.5 - 13.9**	**Nguzo 1 14.0 - 16.9**	**Nguzo 2 17.0 - 20.4**	**Nguzo 3 20.5 -30.4**	**Banaa >30.5**	**Density (stems ha^{-1})**	**Mean height (m)**	**Basal area (m^{2} ha^{-1})**	**Volume (m^{3} ha^{-1})**	**AGB (Mg ha^{-1})**	**BGB (Mg ha^{-1})**	**Total biomass (Mg ha^{-1})**
NS	197	521	222	190	169	219	89	1,607	15.0±7.7	35.64±1.86	193.34±20.53	246.44±25.88^{a}	149.76±9.03^{a}	541.98±34.84^{a}
NCS	1,891	254	75	74	65	134	40	2,523	11.7±3.5	19.44±1.42	155.45±25.78	214.89±25.42^{b}	82.87±8.26^{b}	297.76±33.63^{b}
MDS	1,536	213	108	97	103	147	27	2,169	10.8±6.9	18.44±1.76	130.08±17.86	183.22±21.38^{b}	71.41±6.97^{b}	254.63±28.25^{b}
PIS	1,420	234	120	102	149	231	46	2,302	9.7±3.3	26.69±1.48	191.4±3.38	258.87±36.57^{b}	102.39±13.52^{b}	361.26±50.07^{b}
SS	2,304	346	100	110	93	109	30	3,092	5.0±2.6	21.69±1.48	119.49±10.30	229.31±21.03^{b}	89.95±7.17^{b}	319.26±28.16^{b}
Entire	**1,467 ±357**	**315±57**	**125±25**	**111±17**	**112±21**	**163±33**	**46±11**	**2,339±241**	**10.4**±3.6	**24.26±3.18**	**157.9±15.22**	**255.7±36.24^{b}**	**99.28±13.58^{b}**	**354.98±49.81^{b}**
Proportion (%)	**62.72**	**13.47**	**5.34**	**4.3**	**4.4**	**6.9**	**1.97**							

NS = Northern swamps, NCS = Northern central swamps, MDS = Mongoni and Dodori creek swamps, PIS = Pate Island swamps, and SS = Southern swamps, AGB = Above-ground biomass, BGB = Below-ground biomass and DBH = Diameter at breast height.

4.3 Vegetation carbon stocks

Vegetation carbon stocks in mangroves of Lamu County followed the order of the size of the mean diameter across the management blocks. The total vegetation carbon was estimated at 6.3 million Mg C (mean: 166.56±52.35 Mg C ha^{-1}), with above-and below-ground biomass carbon contributing 77% and 23%, respectively. Northern swamps recorded the highest vegetation carbon (254.52±101.29 Mg C ha^{-1}), while Mongoni and Dodori creek swamps registered the lowest values (119.46±53.47 Mg C ha^{-1}) (Fig. 9). The mean vegetation carbon was significantly different across the five management blocks ($F_{(4, 147)} = 11.4$; $p < 0.05$). When expressed in terms of CO_2 equivalent, the vegetation carbon estimate of mangroves in Lamu County translated to 426.45 t CO_2e ha^{-1}.

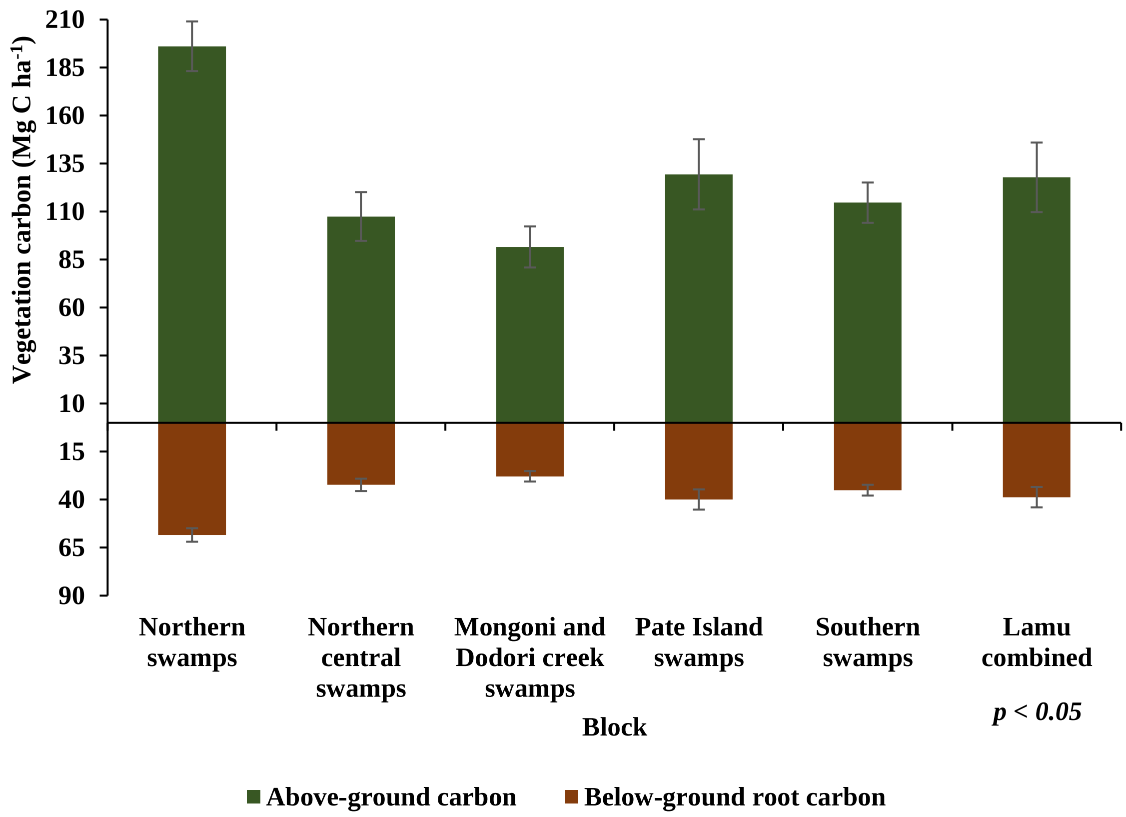


Fig. 9: Contribution of different carbon pools to the total vegetation carbon of mangroves in Lamu County (mean±*s.e*)

4.4 Forest quality

Lamu mangrove forest was dominated by low-quality (Form 3) poles which accounted for 42% of the forest, followed by Form 2 (36%), and Form 1 (22%) (Table 4). Southern, Pate Island and Northern central swamps recorded the highest proportion of Form 3 poles (Table 4). In addition, these blocks also had the highest mangrove degradation hotspots in areas close to human habitation or in sites targeted for coastal development. The pole quality was significantly different between the management blocks ($F_{(4, 147)} = 31$; $p < 0.05$). A Tukey post hoc test revealed quality classes 1 and 3 as the source of the difference. Stems of quality class 2 were not significantly different from either class 1 or 3 ($F_{(4, 147)} = 12$; $p > 0.05$). At the species level, *R. mucronata* (70.1%) and *C. tagal* (23%) recorded more straight poles, while *A. marina* (0.95%) and *X. granatum* had the most crooked poles (0.2%).

Table 4: Density of different quality classes (stems ha^{-1}) of mangroves in Lamu County (x±*s.e*)

Management block	**Quality class**		
	Form 1^{b}	**Form 2^{ab}**	**Form 3^{a}**
Northern swamps	264±40 (16.43)	690±37 (42.94)	653±57 (40.63)
Northern central swamps	685±51 (27.15)	840± 41(33.29)	998±85 (39.56)
Mongoni & Dodori creek swamps	595±33 (27.43)	719±35(33.15)	855±102 (39.42)
Pate Island swamps	413±27 (17.94)	821± 57 (35.66)	1,068±90 (46.39)
Southern swamps	603±48 (18.89)	1,117±77 (34.99)	1,372±94(42.98)
Lamu combined	**512±76 (22)^{b}**	**838±76 (36)^{ab}**	**989±119^{a} (42)**

*Column means (±*s.e.*) with same letter a or b attached are not significantly different (Tukeys HSD, $\alpha = 0.05$). Values in parenthesis are percentages

Northern central swamps recorded the highest density of stumps (492±101 stumps ha^{-1}), while Northern swamps recorded the lowest density (224±71 stumps ha^{-1}) (Fig. 10). The difference in stump density among the five management blocks was not significant ($F_{(4, 147)} = 1.4$, $p > 0.05$). Most of the stumps belonged to *R. mucronata* followed by *C. tagal*, where the majority were *Boriti*-sized poles. There was a strong positive correlation between high-quality poles (Form 1) and stump density ($r^2 = 0.76$, $p < 0.05$).

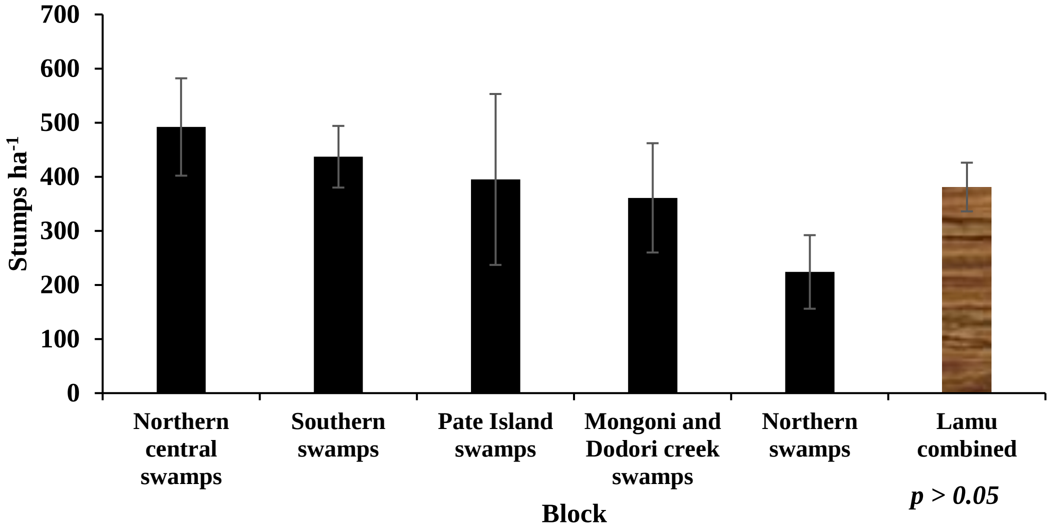


Fig. 10: Stump density (counts ha^{-1}) in different mangrove management blocks of Lamu County.

*The combined stump count for entire Lamu mangroves is included for ease of comparison

On an average, the number of poles harvested from the forest was 381±47 stems ha^{-1} (Fig. 10), giving a total of 13,593,313 poles harvested from the entire forest. According to the Kenya Forest Service, allowable cuts for *Mazio* (diameter:8.0 - 11.4 cm) and *Boriti*-sized mangrove poles (diameter:11.5 - 13.9 cm) in Lamu County are 90,000 and 150,000 scores per year, respectively. These are the main market-preferred mangrove poles in the country (GoK, 2017). Considering the 35,678 ha of mangroves (see Section 4.6) and stocking rates of 315 and 125 stems ha^{-1} for *Mazio* and *Boriti*, respectively, their total stock can be estimated at 561,929 and 222,988 scores, respectively. These estimates are far higher than the allowable cuts for *Mazio* and *Boriti*. With a rotation cycle of 20 years for Lamu County mangroves (Robert & Ruara, 1967), at least 28,096 and 11,149 Mazio and *Boriti* scores, respectively, can be removed annually.

4.5 Natural regeneration

Juveniles were frequently found growing close to the mother trees in clusters. The density of juveniles ranged from 5,636±404 juveniles ha^{-1} in Northern swamps to 8,096±383 juveniles ha^{-1} in Southern swamps (mean±*s.e*: 7,342±450 juveniles ha^{-1}). There was a significant difference in density of juveniles across the management blocks ($F_{(4, 147)} = 42$; $p < 0.05$). The forest was dominated by Regeneration Class I juveniles (42.66%) (Table 5). Majority of the juveniles were *R. mucronata* (52.93%) and *C. tagal* (25.57%), which was expected because of their parental dominance. This translated to regeneration ratios (for RCI: RCII: RCIII) of Lamu mangroves at 2:2:1. There was a positive correlation between the canopy gaps and regeneration ($r^2 = 0.61$, $p < 0.05$). When comparing management blocks, the potential regeneration category was highest in Pate Island and Mongoni & Dodori creek swamps, at 4,232 and 4,225 juveniles ha^{-1}, respectively. In contrast, the established regeneration category was highest in the Southern swamps (RCII = 3,404, RCIII = 2,089 juveniles ha^{-1}) and Northern swamps (RCII = 2,926, RCIII = 1,940 juveniles ha^{-1}).

Table 5: Juvenile densities (counts ha^{-1}) of mangroves in Lamu County

Management block	**Species**	**Regeneration class**			**Total (juveniles ha^{-1})**
		I (< 40 cm)	**II (40-150 cm)**	**III (150.1-300 cm)**	
Northern swamps	*Avicennia marina*	57	34	9	100
	Bruguiera gymnorhiza	24	16	1	41
	Ceriops tagal	508	232	171	911
	Rhizophora mucronata	1,510	2,134	913	4,557
	Sonneratia alba	7	20	0	27
	TOTAL	**2,106**	**2,436**	**1,094**	**5,636**
	Proportion (%)	**37.37**	**43.22**	**19.41**	**100**
Northern central swamps	*Avicennia marina*	23	7	1	31
	Bruguiera gymnorhiza	21	12	12	45
	Ceriops tagal	1,726	1,422	934	4,082
	Rhizophora mucronata	720	1,485	993	3,198
	Sonneratia alba	3	0	0	3
	TOTAL	**2,493**	**2,926**	**1,940**	**7,359**
	Proportion (%)	**33.88**	**39.76**	**26.36**	**100**
Southern swamps	*Avicennia marina*	2	2	4	8
	Bruguiera gymnorhiza	19	89	72	180
	Ceriops tagal	1473	1910	1174	4557
	Rhizophora mucronata	1109	1403	839	3351
	TOTAL	**2603**	**3404**	**2089**	**8096**
	Proportion (%)	**32.15**	**42.05**	**25.8**	100
Pate Island swamps	*Avicennia marina*	241	0	0	241
	Bruguiera gymnorhiza	393	286	61	740
	Ceriops tagal	789	627	443	1859
	Rhizophora mucronata	2796	1570	616	4982
	Sonneratia alba	13	0	21	34
	TOTAL	**4232**	**2483**	**1141**	7856
	Proportion (%)	**53.87**	**31.61**	**14.52**	100
Mongoni and Dodori creeks	*Avicennia marina*	2866	144	144	3154
	Bruguiera gymnorhiza	48	20	13	81
	Ceriops tagal	828	536	992	2356
	Rhizophora mucronata	464	1092	851	2137
	Sonneratia alba	19	6	0	25
	Xylocarpus granatum	0	2	6	8
	TOTAL	**4225**	**1800**	**1736**	**7761**
	Proportion (%)	**54.44**	**23.19**	**22.37**	100
All	**TOTAL**	**3,132**	**2,610**	**1,600**	7,342
	Proportion (%)	**42.66**	**35.55**	**21.79**	100

*Values in parentheses indicate percentages. Potential regeneration category = RCI = 3,132 (42.66%). Established regeneration category = RCII + RCIII = 3,210 (57.34%)

4.6 Mangrove cover and cover change between 1990 and 2019

The mangrove area in Lamu County was estimated at 35,678 ha, distributed in five management blocks (Table 6). The accuracy achieved in the overall mapping of mangrove coverage was 95%, while the accuracies achieved in discriminating mangroves by species formation stood at 71.3%, registering a Kappa coefficient of 0.6153 (61.5%). In most management blocks, the dominant mangrove formation was *R. mucronata*. About 35% of mangroves in Lamu County occurred in Northern central swamps, followed by 27% in Southern swamps (Table 6).

Table 6: Contribution of different species formations/types to the total areal coverage of mangroves in Lamu County in 2019

Forest type (species formation)	Northern swamps	Northern central swamps	Mongoni & Dodori creek swamps	Pate Island swamps	Southern swamps	Total area (ha)	Proportion (%)
Rhizophora dominant	1,542	5,104	1,104	2,815	3,419	13,983	39.19
Avicennia almost pure stand	-	1,038	2,733	446	9	4,225	11.84
Rhizophora almost pure stand	-	-	732	0	3,102	3,834	10.75
Avicennia dominant	252	2,455	1	864	1,222	4,794	13.44
Mixture of *Ceriops & Rhizophora*	-	3,884	1	1	-	3,886	10.89
Mixture of *Rhizophora* & *Sonneratia*	766	-	-	-	-	766	2.15
Ceriops dominant	419	1	1	1,910	1,860	4,191	11.75
Total	**2,979**	**12,481**	**4,571**	**6,036**	**9,611**	**35,678**	**100**
Proportion (%)	**8**	**35**	**13**	**17**	**27**	**100**	

Based on remotely sensed data, the mangroves in Lamu County are not static (Fig. 11). Of the total mangrove cover in Lamu County, 971 ha (2.7%) and 6,210 ha (17.4%) were highly and moderately degraded, respectively. Most of the degraded areas were close to human settlements or sites targeted for coastal development in concession sites in the Northern central, Pate Island, and Southern swamp forests. Mambore was the only area in the Northern swamps that was identified as highly degraded. Main areas for loss and degradation of mangroves in Lamu County were found to be areas in the Northern central swamps at Kiwayu, Ndau, Uvondo, Siyu, Rewa, Kizingitini, and Mbajumah sites, where commercial harvesting is allowed. Other sites that have suffered similar fates are Yowea and Manda Islands in both Pate Island and Southern swamps, where mangroves are clear-felled for fuel-wood (Kairo *et al.*, 2002a; GoK, 2017). Very few areas in the Mongoni & Dodori creek swamps were identified as degradation hotspots, with the majority being either moderately degraded or completely non-degraded (Fig. 11).

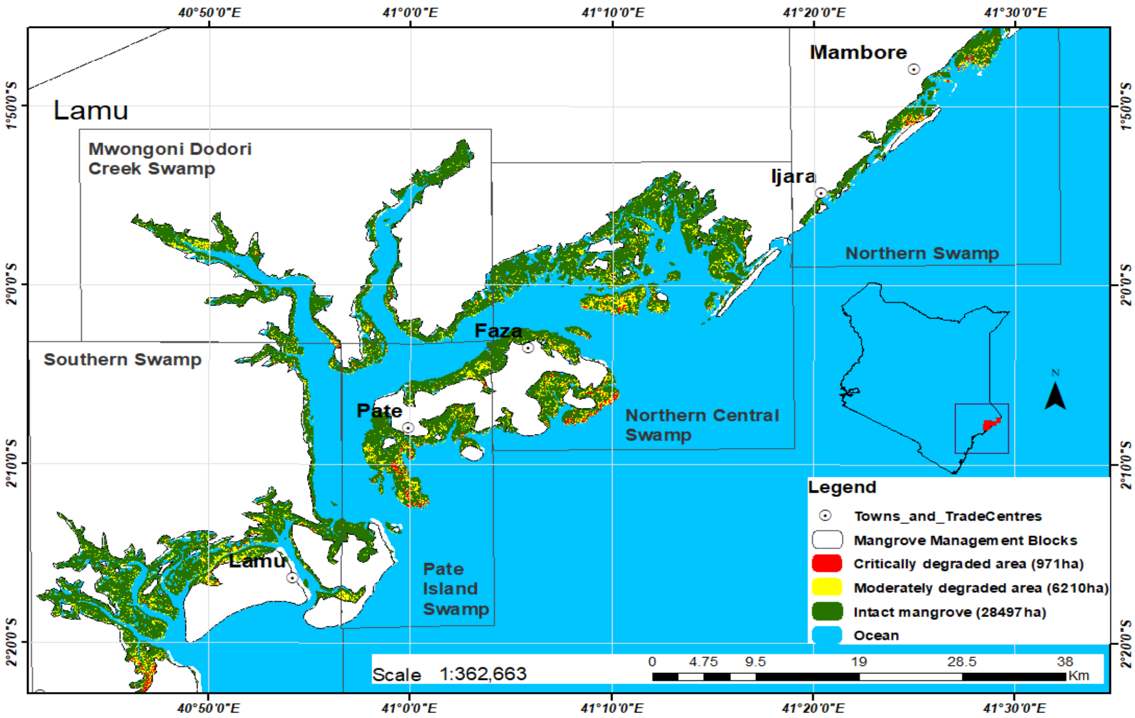


Fig. 11: Hotspots of mangrove loss and degradation in Lamu County.

Highted mangrove loss was witnessed close to human settlement in Pate Island, Northern central and Southern swamps

Overall, a cover loss of 1,739 ha was recorded from 1990 to 2019, translating to a loss of 60 ha yr^{-1}. However, the loss was not uniform between the different study periods, with 2010-2019 experiencing a higher loss (1,029 ha, at a rate of 114 ha yr^{-1}) than the 2000-2010 period (901 ha, at a rate of 90.1 ha yr^{-1}) (Table 7). Similar trends were observed in Northern central swamps, Northern swamps, and Pate Island swamps, whereby mangrove cover increased from the 2000 - 2010 period followed by a decline in the 2010 - 2019 period. However, mangroves in the Southern swamps recorded declining trends in the 2000 - 2010 period and continuously increased in the later period. Unlike other mangrove management blocks, Mongoni and Dodori creek swamps showed increasing trends throughout the study period.

Table 7: Cover and cover change of mangroves in Lamu County from 2000 to 2019. The 1990 data are included herein to serve as the baseline

Mangrove change	**1990 (Baseline)**	**2000-2010 (Gain/Loss)**		**2010-2019 (Gain/Loss)**	
		2000-2010 ha	**Total loss/gain 2000-2010**	**2010-2019 ha**	**Total loss/gain**
Very High Loss	-	6,317.10	8,453.20	6,585.50	8,592.60
High Loss	-	1,376.40		1,358.90	
Medium Loss	-	759.70		648.20	
No Significant Change	-	2,0934.60	2,0934.60	19,522.10	19,522.10
Moderate Gain	-	346.90	7,552.50	382.10	7,563.70
High Gain	-	831.70		971	
Very High Gain	-	6,373.90		6,210.6	
Total Area	**37,417**		**36,940.20**		**35,678.40**
Net change (Gain/Loss)		**-1739***	**-900.70**		**-1,028.90**
Net change (Gain/Loss) yr^{-1}		**-60***	**-90.10**		**-114.30**

* Net mangrove cover changes calculated over 29-year period (1990-2019). The gain or loss was determined block wise.

Based on cover change analysis, only two formations (forest types) showed a gain in areal coverage between 1990 and 2019, i.e., *Rhizophora* dominant (15.6% gain) and *Avicennia* dominant (8.2%). *Avicennia* stands, which were almost pure, recorded the highest loss (11.4%), followed by *Ceriops* stands (6.7%). *Rhizophora* dominant (39.2%) and a mixture of *Rhizophora* and *Sonneratia* formation (2.1%) are still the most and least dominant formations over the last 3 decades, respectively (Table 8).

Table 8: Areal coverage of mangroves in Lamu County by species formation and year

Forest type (species formation)	**Area (ha) 1990**	**%Area 1990**	**Area (ha) 2019**	**% Area 2019**	**Status**
Rhizophora dominant	8,839.0	23.6	13,983.5	39.2	**Gain**
Avicennia almost pure stand	8,693.4	23.2	4,224.8	11.8	Loss
Rhizophora almost pure stand	4,706.7	12.6	3,833.4	10.7	Loss
Avicennia dominant	1,937.7	5.2	4,793.4	13.4	**Gain**
Mixture of *Ceriops* & *Rhizophora*	4,629.1	12.4	3,886.0	10.9	Loss
Mixture of *Rhizophora* & *Sonneratia*	1,725.9	4.6	765.8	2.3	Loss
Ceriops dominant	6,885.5	18.4	4,191.5	11.7	Loss
Total	**37,417.4**	**100.0**	**35,678.4**	**100.0**	**Loss (0.2%)**

CHAPTER FIVE

DISCUSSION, CONCLUSION AND RECOMMENDATIONS

5.1 Discussion

5.1.1 Floristic composition and structural attributes

The present study only encountered six (6) mangrove species in Lamu County, which is lower than the eight (8) species reported by Kairo *et al.* (2002a) in the same area. Low species count witnessed in the study could be due to the sampling intensity and rarity of some mangrove species in Kenya, such as *Heritiera littoralis* (*dryand*.), *Lumnitzera racemosa* (*Willd*). and *Xylocarpus moluccensis* (*Lamk*.) *Roem.* (Bosire *et al.,* 2016; GoK, 2017). All species encountered in the present study are native to Kenya (Bosire *et al.,* 2016) and are part of the mangroves in the Indo-West Pacific bioregion (Spalding *et al.,* 2010, Saenger *et al.,* 2019).

Similar to previous studies on mangrove forests in Kenya (e.g., Mohamed *et al.,* 2009; Okello *et al.,* 2022), mangroves in Lamu County are dominated by single and mixed stands of *C. tagal* and *R. mucronata*. These forests grow luxuriantly in the Lamu area due to the influence of freshwater and nutrient input from Tana River during the northeast monsoon (NEM), when northern flowing East African Coastal Currents (EACC) meet the south-flowing Somali Currents causing upwelling and nutrient enrichment (Kamau *et al.,* 2020; Njiru *et al.,* 2022). This is in addition to the presence of groundwater seepage in the area, which supplies freshwater to the system (Kuria, 2013; Nijsten *et al.,* 2018). The structural variability of mangroves along the Kenya coast has been attributed to geomorphic and climatic drivers, which contribute to the high complexity indices for mangroves in Lamu County when compared to other areas along the coast (Njiru *et al.,* 2022).

The observed mangrove species zonation in Lamu County is typical of mangrove zonation in Kenya, which starts with *S. alba* on the seaward margin, followed by large *A. marina* and *R. mucronata*. Pure or mixed stands of *C. tagal, A. marina,* and *X. granatum* occupy the mid-intertidal zone, while the higher intertidal zone is dominated by *A. marina. Bruguiera gymnorhiza* lacks distinct zonation but is interspersed with *R. mucronata* and *C. tagal*. In the creeks, *Rhizophora-Avicennia* mix was the most dominant. In some places, *A. marina* occurred on the seaward and landward edges, resulting in double zonation. Generally, it was widely distributed across the five management blocks. Despite the dominance of *S. alba* in the lower intertidal zone, it was replaced by *R. mucronata* and *A. marina* as the first species along the small creeks. Similar to this study, double zonation of *A. marina* has also been observed in Gazi Bay (Dahdoub-Guebas *et al.,* 2004; Wang'ondu *et al.,* 2010). Dahdoub-Guebas *et al.* (2004) attributed the wide distribution of *A. marina* to its tolerance to different levels of salinity and tidal regimes, among others. Mangrove zonation is greatly influenced by the levels of inundation, geomorphology, and salinity (Tomlinson, 2016).

The standing density of mangroves in Lamu County was 2,339±540 stems ha^{-1} (range:1,607-3,092 stems ha^{-1}). This is significantly higher than the stocking rates recorded in other mangrove areas along the Kenyan coast, such as Mida (Kairo *et al.,* 2002b), Mombasa (Mohamed *et al.,* 2009), and Tana delta (Bundotich *et al.,* 2009). The stocking density of *Mazio* and *Boriti*-sized poles (DBH range:8.0 -11.4 and 11.5 -13.9 cm) was significantly lower in Northern Central swamps, Mongoni and Dodori Creek swamps, and Southern swamps possibly due to selective logging to satisfy market demand. These size classes are the most preferred for building and construction in the region (Bosire *et al.,* 2016). Selective logging of mangroves has been reported to contribute to low densities of merchantable wood products and a reduction in forest quality over time (Kairo *et al.,* 2002a; Okello *et al.,* 2022; Riungu *et al.,* 2022).

Maximum canopy height of half of the world's mangroves is less than 13.2 m (Simard *et al.,* 2019; Freiss, 2019). Hence, a mean of 10.2 m for Lamu mangroves indicates high canopy height. According to Tomlinson (2016), variations in canopy height can be attributed to climate, topography, and the extent of human disturbance. Simard *et al.* (2019) observed that tallest mangroves occur in less human populated areas, as well as areas with high precipitation and low cyclone frequencies. Basal areas of disturbed, secondary and pristine mangroves are > 10m^{2} ha^{-1}, about 15 m^{2} ha^{-1} and > 25m^{2} ha^{-1}, respectively (Komiyama *et al.,* 2008; Kauffman *et al.,* 2011). The basal area estimate of 24.26 m^{2} ha^{-1} recorded in this study indicates that the mangroves in Lamu County are semi-pristine. These mangroves had a complexity index of 34.9 which is higher than that of other mangrove forests along the Kenya coast (Table 9). These variations in forest structure along the Kenyan coast have been associated with geomorphic processes, climate drivers, and management regimes (Njiru *et al.*, 2022).

Table 9: Summary attributes of mangroves in Lamu County as compared to other mangrove sites in Kenya

Site	**Attribute**					
	Number of plots	**Number of species**	**Mean stem density (stems ha^{-1})**	**Mean basal area (m^2 ha^{-1})**	**Mean height (m)**	**Complexity Index (C.I)***
Lamu						
Lamu combined[1,*]	152[a]	6	2,339	24.38	10.40	34.9
Northern swamps[1,*]	38[a]	5	1,607	35.64	15.00	42.96
Northern central swamps[1,*]	39[a]	5	2,523	19.44	11.70	28.69
Mongoni and Dodori creek swamps[1,*]	16[a]	6	2,169	18.44	10.80	22.56
Pate Island swamps[1,*]	14[a]	6	2,302	26.69	9.70	35.76
Southern swamps[1,*]	45[a]	5	3,092	21.69	5.00	16.77
Other mangrove areas in Kenya						
Kirepwe, Mida[2*]	31[b]	6	1,197	23.62	8.88	6.97
Uyombo, Mida[2*]	60[b]	6	1,585	15.84	7.25	2.10
Ngomeni[3*]	44[b]	6	1,251	33.14	9.54	25.22
Gazi[4*]	9[b]	6	678	3.91	8.3	0.35
Tudor[5*]	230[b]	6	1,283	13.02	4.37	3.49

[1]This study, [2]Kairo *et al.*, 2002b, [3]Bundotich *et al.*, 2009, [4]Bosire *et al.*, 2003, and [5]Mohamed *et al.*, 2009. a = 400 m^2 plots, b = 100 m^2 plots.

Limit of inclusion: *individuals with diameter ≥ 2.5 cm and **individuals ≥ 5 cm

Vegetation biomass for mangroves in the county ranged between 254.63 and 541.98 Mg ha^{-1} (mean:354.98±137.82 Mg ha^{-1}). This is consistent with the reported biomass values of productive mangroves in Indonesia (Rozainah *et al.,* 2018), Malaysia (Rozainah *et al.,* 2018), Philippines (Thompson *et al.,* 2014), and Dominican Republic (Kauffman *et al.,* 2014), but significantly higher than values reported for degraded systems such as Sofala Bay in Mozambique (Sitoe *et al.,* 2014). Lamu mangroves are among the most productive systems in Kenya (Kairo *et al.,* 2002a; Njiru *et al.,* 2022) and indeed in the WIO region (Erftemeijer *et al.,* 2022). This attribute has been associated with both geomorphic and oceanographic drivers (GoK, 2017; Njiru *et al.,* 2022). Although there are no permanent rivers draining into the mangroves of Lamu, the forcing functions created by the East African coastal currents could be involved in pushing freshwater northward from the Tana River Delta to Lamu, thus contributing to enhanced marine productivity in the area

(Kamau *et al.,* 2020). High marine productivity in the area may also be associated with upwelling caused by the interaction between the southerly Somali Coastal Currents and the northward-flowing East African Coastal Currents (Kamau *et al.,* 2020). Northern swamps forests had significantly higher biomass values than other management blocks in the county. This is due to the management regime, as the area is classified as marine protected area where commercial harvesting is prohibited.

A previous study established the standing volume of mangroves in the Northern and Northern central swamps of Lamu at 145.88 m^3 ha^{-1} (Kairo *et al.*, 2002a). This is lower than the current estimates of 174.39 m^3 ha^{-1} for the same area; and 157.97 m^3 ha^{-1} for the entire mangrove area in Lamu County. The estimate is also higher than 100.44 m^3 ha^{-1} in a *Rhizophora mucronata* plantation in Gazi-Kenya (Lang'at, 2008), 93.04 m^3 ha^{-1} in Jor Bay-Indonesia (Zulhalifah *et al.*, 2021), 69.78 m^3 ha^{-1} in mangroves of mainland Tanzania (Njana, 2020), and 103.5 m^3 ha^{-1} in Mentawir-Indonesia (Kristiningrum *et al.*, 2019). This implies that mangroves in Lamu County have the potential for wood exploitation. Considering that the area of mangroves in Lamu County is 35,678 ha, the total standing volume of the mangroves in the entire county translates to 5,636,053.6 m^3. With the working plans for mangroves in Lamu County constituting a 20-year rotation cycle (GoK, 2017), approximately 281,803 m^3 can be harvested annually during the first rotation if a clear-felling system is employed. This information on the wood resources from the stand volume of mangroves in Lamu County is vital for guiding the design of harvest plans.

5.1.2 Vegetation carbon stocks

The mean vegetation carbon stocks of the Lamu mangroves (166.56 Mg C ha^{-1}); compares with values of 211.4, 103.3, 223, 147.5, and 190.2 Mg C ha^{-1} obtained in the Zambezi Delta-Mozambique (Stringer *et al.*, 2015), Geza-Tanzania (Alavaisha & Mangora, 2016), Mtimbwani-Tanzania (Alavaisha & Mangora, 2016), and global average (Siikamäki *et al.*, 2012), (Alongi, 2020), respectively. The global carbon density for forests is estimated to be 163 Mg C ha^{-1} (FAO, 2020), which is slightly lower than our estimate. However, much higher values of 340.6, 982, 339.7, and 334.8 Mg C ha^{-1} were obtained in undisturbed mangrove forests of Mozambique (Stringer *et al.,* 2015), West Papua, Indonesia (Murdiyarso *et al.*, 2015), Sumatra, Indonesia (Murdiyarso *et al.,* 2015)], and Yap, Micronesia (Kauffman *et al.,* 2011). On the other hand, lower values of 88.9, 53.2, and 64.7 Mg C ha^{-1} were obtained in Mahajamba Bay-Madagascar (Jones *et al.,* 2015), Sofala-Mozambique (Sitoe *et al.,* 2014), and mainland mangroves-Tanzania (Njana *et al.,* 2018). The observed differences are due to variations in sample size, species composition, geographical locations, wood density, and management regimes (Komiyama *et al.,* 2005).

Over 1990 - 2019, the mangroves in Lamu County decreased by a net loss of 1,739 ha. Such a change could have activated carbon dioxide (CO_2) emissions from lost vegetation cover. There are possible fates of "near-surface carbon" upon conversion, ranging from 25 to 100% emissions to the atmosphere depending on the land use type (Pendleton *et al.,* 2012; IPCC, 2014; Hamilton & Freiss, 2018). Using the lower end of the 25% emissions, the potential carbon loss from mangroves in Lamu County would amount to 41.64 Mg C ha^{-1}. To enable comparison with other assessments, values were expressed in terms of CO_2 equivalents by multiplying C stocks by 3.67, which is the molecular weight of C in CO_2. Therefore, the current carbon emission from mangroves of Lamu County using the lower end of 25% emissions is 9,169.13 Mg CO_2e yr^{-1}, mainly resulting from deforestation and forest degradation activities (GoK, 2017). These emissions from mangroves are large, especially compared to terrestrial ecosystems. Although mangroves occupy only 3% of Kenya's forests (GoK, 2017), the total volume of carbon they store is substantial. Globally, C emissions due to land-use change have been estimated to range between 90 and 450 million Mg CO_2e yr^{-1} over a global mangrove area of 13.8 - 15.2 million ha, which translates to 6.55 - 29.61 Mg CO_2 ha^{-1} yr^{-1} (Murray, 2012; Pendleton *et al.,* 2012).

Therefore, preserving mangroves may provide relatively low-cost opportunities to mitigate CO_2 emissions while simultaneously conserving biodiversity and supporting community livelihoods. Payment for Ecosystem Services (PES) schemes, such as Reducing Emissions from Avoided Deforestation and forest Degradation (REDD+), are potential revenue streams for compensating those involved in mangrove conservation activities. Assuming an offset price of US$10/Mg CO_2e, the estimated cost of avoided emissions from mangroves in Lamu County is USD 91,691.3 yr^{-1}, plus other co-benefits such as fishery functions and shoreline protection (GoK, 2017).

5.1.3 Forest quality

Lee *et al.* (2014) documented that the condition of mangroves locally depends on the pressure exerted by the communities living adjacent to the mangrove forests. At least 42% of the Lamu mangrove forest is stocked with Form 3 poles, indicating prolonged human pressure. This was followed by Form 2 poles (36%), further indicating the presence of selective removal of the desired Form 1 poles, which may subsequently affect forest structure and productivity. A high proportion of low-quality mangrove poles reported in Southern, Pate Island, and Northern swamps was expected. Most of the commercial harvesting of mangroves in the county is concentrated in the three management blocks. The removal of straight mangrove poles, leaving out crooked poles of low demand, may subsequently impact the future forest by minimizing its capability to provide quality poles as demanded by the market. The dominance of these crooked trees as mother trees may also result in gradual deterioration of the genetic quality of the forests, hence the need for exploitation of all pole quality classes (Kairo & Dahdouh-Guebas, 1995; Marten, 1996).

As noted by Kairo *et al.* (2002a) and Okello *et al.* (2022), stump prevalence is an indication of forest disturbance through human activities. In this study, stump counts were mainly attributed to harvesting by both illegal and licensed cutters. Most of the stumps were found in areas near human settlements because of their ease of accessibility. Similar observations linking stump density to forest degradation have been reported in Kenya in Mida Creek (Kairo *et al.*, 2002b), Mtwapa (Okello *et al.*, 2019), and Lamu mangroves (Okello *et al.*, 2022). These mangroves are exploited for construction purposes and as fuel-wood. Sawn timber is an emerging threat to mangroves in the county, involving the exploitation of aged trees, with possible consequences on forest undergrowth. In Kenya, mangroves are normally harvested through the selective removal of trees with diameters ranging from > 5 cm to < 20 cm to protect natural regeneration and ensure the sustainability of the forest (GoK, 2017). Unregulated removal of old-growth mangrove trees has negative effects on the structure and regeneration of the forest.

Among the mangrove species present, *R. mucronata* is the species most targeted by cutters and has the potential for over-exploitation. The preference for *R. mucronata* can be attributed to its capacity to produce quality poles that are tall, straight, and resistant to termite attack (Bosire *et al.*, 2014; GoK, 2017). These poles have a high calorific value when burned (Omodei-Zaroni & Cortini, 2000; Kathiresan, 2012; Bosire *et al.*, 2016) and are suitable for diverse uses. An earlier study in Mida Creek observed that in a mixed stand, selective removal of *R. mucronata* trees paved the way for the less desirable *C. tagal* (Kairo *et al.*, 2002b). The same has been observed in Mombasa, where *A. marina* replaced *R. mucronata* (Bosire *et al.*, 2014). Hence, ongoing selective harvesting in Lamu County may shift species composition, compromising the future availability of market-preferred poles.

5.1.4 Natural regeneration

The mangrove forests in Lamu County are dominated by parents and juveniles of *R. mucronata* and *C. tagal*. Natural regeneration of 7,342±450 juveniles ha^{-1} observed in the forest is considered adequate to support forest recovery following disturbance. This is significantly higher than the minimum of 2,500 seedlings ha^{-1} recommended by the FAO (1994) to restock a degraded mangrove stand without replanting. Fickert (2020) did not find a significant difference in the success of mangrove recovery between human-assisted and natural regeneration in the Gulf of Honduras. A healthy mangrove forest provides multiple services beyond wood. When lost or degraded, the provision of these services halts or declines. Restocking forests through natural regeneration restores their functionality and resiliency. The frequent occurrence of juveniles in clusters close to the mother tree observed in this study has been recorded in other studies of Kenyan mangroves, such as Gazi Bay (Bosire *et al.*, 2005) and Mtwapa (Okello *et al.*, 2013). Similar findings have also been reported in Mozambique (Nicolau *et al.*, 2017), and are attributable to the dispersal mechanisms of these species. Mangrove parents have nursing effects on their juveniles, as the clustering of the juveniles around the mother trees reduces the threat of washing by tides and predation by crabs (Olagoke *et al.*, 2013). At some sites, juveniles of different species from adult trees were present, indicating recolonization by a more competitive species (Bosire *et al.*, 2003). However, this may not be due to invasion, but rather a higher reproductive capacity of one species than the others. For instance, *C. tagal* has higher reproductive potential than *R. mucronata*. In a mixed stand of Ceriops and Rhizophora species, degradation favors the recolonization of Ceriops.

A regeneration ratio (RCI: RCII: RCIII) of 6:3:1 is recommended for effective mangrove regeneration (Chong, 1988). Hence, the adequacy of natural regeneration of Lamu mangroves (RCI: RCII: RCIII = 2:2:1) was significantly lower than the recommended value. However, based on juvenile densities, Lamu mangroves have the potential to have an excellent regeneration capacity.

Despite mangroves in the Northern swamps recording higher structural indices due to physical barriers, insecurity, low population, and the management regime whereby commercial harvesting is prohibited (Kairo *et al.*, 2002a), this block was characterized by low juvenile density. This is expected for old-growth forests because mangroves do not support understory (Janzen, 1985). According to Rasquinha and Mishra (2021), overgrown mangrove trees result in a closed canopy that hinders regeneration owing to insufficient light. Since harvesting of mangroves is prohibited in the Northern block, the trees formed a closed canopy, inhibiting light penetration. Viability of mangrove juveniles is generally lower in areas with full shade than in those with open canopy (Azad *et al.*, 2021).

The size class distribution of mangroves in Lamu County revealed that density decreased with increasing stem diameter (Fig. 12). There were lower stem densities of preferred poles (*Mazio* and *Boriti*) as well as higher densities of large poles (≥ 20.5 cm) - Fig. 12. This is an indication of selective logging of the preferred size classes. Theoretically, in an uneven-aged forest, there is a normal series of age gradations, depicted by the reversed J curve, whereby stand densities decrease with an increase in size classes (Clutter *et al.*, 1983; Cancino & Von Gadow, 2002; Ducey, 2006; Kerr, 2014). On the assumption that diameter classes express age of the trees, the density curves obtained in this study (Fig. 12) can be used to predict the nature of the future mangrove forest in the county. This can be done by harmonizing the irregularities in the size-class distribution by the deliberate removal of ‘excess’ trees in those size-classes (≥ 20.5 cm); where observed densities are higher than expected. This could be achieved through the diversification of the use of mangrove wood in the county. Old trees could be removed sustainably to serve market needs which is not the case at present. Removal should also be regulated through the use of a harvest plan to ensure the recovery of the forest after disturbance. The findings of this study is important in supporting sustainable mangrove management not only in Kenya, but also globally, where commercial wood harvesting is practiced.

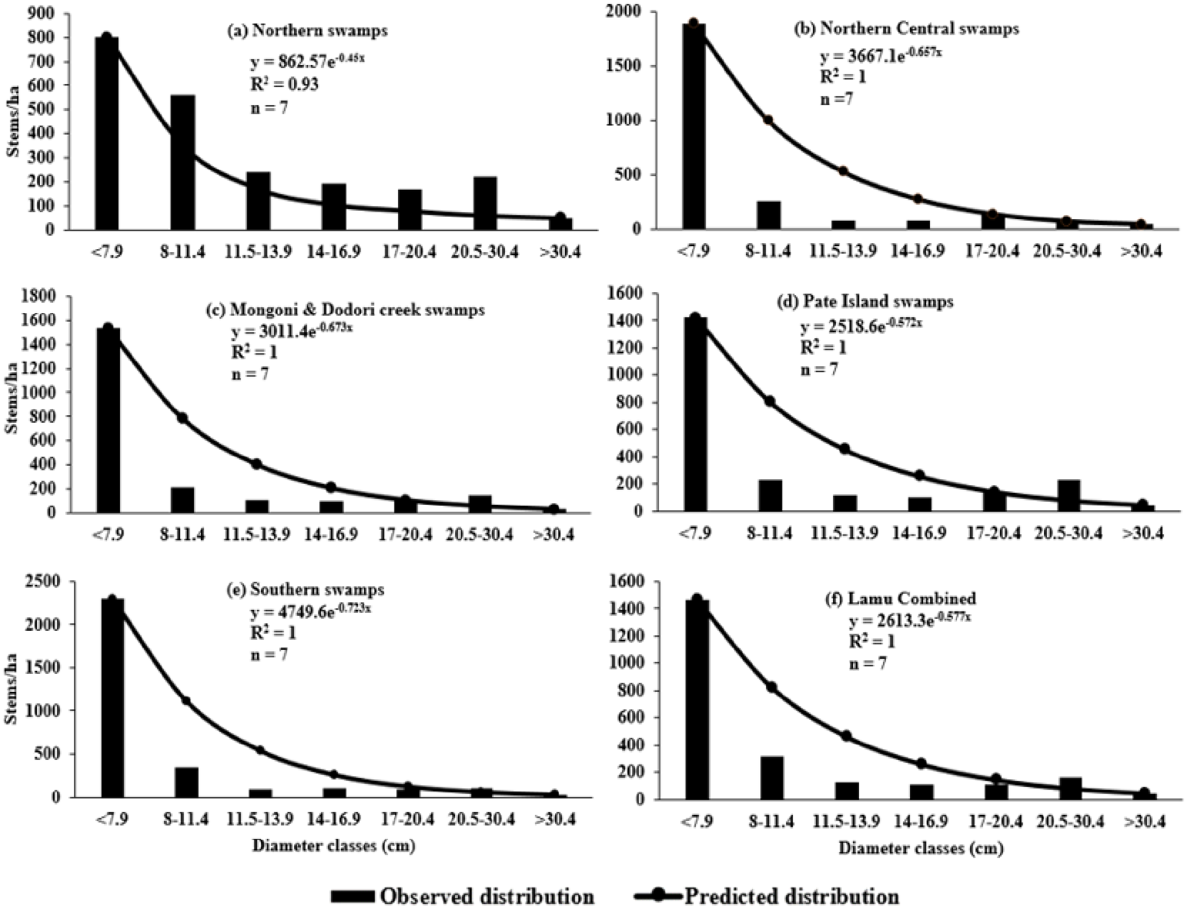


Fig. 12: Size-class distribution of mangrove forests in Lamu County. A high 'k' value in the stand curve $y = ke^{-ax}$ for southern, Northern central, and Mongoni and Dodori creek swamps reflects the occurrence of sporadic natural regeneration in the forest

5.1.5 Mangrove cover and cover change between 1990 and 2019

The mangrove area in Lamu County was estimated to be 35,678 ha. This value is significantly higher than the earlier estimate of 23,500 ha (Kirui *et al.* 2013), but lower than the estimate of 37,350 ha recorded in the National Mangrove Ecosystem Management Plan (GoK, 2017). Such differences are a result of the different methodologies, periods, and data sources (Xu *et al.*, 2022), considering that Kirui *et al.* (2013) and the present study utilized Landsat imagery, whereas the management plan used high-resolution aerial imagery (GoK, 2017). The 95% and 71.3% accuracy levels achieved in mapping the overall mangrove coverage and discriminating mangroves by species formation (Kappa coefficient of 0.6153), respectively, indicated strong agreement, proving satisfactory for use in this context.

The present study observed 7 out of the 9 mangrove forest types (species formations) found along the entire Kenyan coastline. However, despite these formations/types, only 6 of the 9 mangrove species found in Kenya (Bosire *et al.,* 2016; GoK, 2017) were recorded in this study. This can be attributed to the rarity of some species and sampling intensity, which influences species assemblages. Mangroves in Lamu County are not pristine. The underlying causes of loss and degradation of these mangroves have been identified as over-exploitation of wood products, conversion of mangrove areas for other land uses, and habitat encroachment (Hamza *et al.,* 2020). Climate change has also contributed to the loss of mangroves in the country, particularly through siltation (Bosire *et al.,* 2016).

This study observed that during the periods under investigation, there were gains and losses in mangrove coverage in Lamu County, with an overall loss of 60 ha yr^{-1} (0.16% yr^{-1}). This is similar to the current global rate of mangrove loss, estimated at 0.16% yr^{-1} (Hamilton & Casey, 2016; Hamilton & Freiss, 2018; Alongi, 2020). However, at the country level, the loss in this study is lower than the 0.7% reported by Kirui *et al.* (2013) for Kenyan mangroves. A comparison of the annual mangrove loss rate in Lamu County with those in other regions is provided in Table 10.

Table 10: Comparison of various studies on mangrove loss rates for different time periods using different methods

Location	Annual loss rate (%)	Period	Method	Reference
Lamu County, Kenya	0.16	1990-2019	Landsat images	This study (2019)
Whole of Kenya	0.74	1985-2010	Landsat images	Kirui *et al.,* 2013
Mwache, Kenya	5.1	1992-2009	SPOT images	Bosire *et al.,* 2014
Tudor, Kenya	2.7	1992-2009	SPOT images	Bosire *et al.,* 2014
Mtwapa Creek	0.04	1994-2000	SPOT images	Okello *et al.,* 2013
Kenya-Tanzania transboundary	0.69	1986-2016	Landsat images	Mungai *et al.,* 2019
Global estimate	0.16-0.39	2000-2012	Landsat images	Hamilton & Casey, 2016
Global estimate	0.13	2000-2016	Landsat images	Goldberg *et al*., 2020

5.2 Conclusion

This study assessed the status and condition of mangroves in Lamu County. There are 35,678 ha of mangroves, representing 62% of the country's total mangrove coverage. Forest structure differed significantly ($p < 0.05$) among the forest blocks, which was mainly attributed to anthropogenic and geomorphic factors. At least 42% of the forest is stocked by low-quality poles, which is an indicator of prolonged human pressure. Selective logging of mangroves slowly transforms the superior forest into inferior stands with low productivity (Kairo *et al.*, 2002a). However, natural regeneration rates of 7,342±450 juveniles ha^{-1} observed in the forest indicate a resilient healthy forest that can recover naturally if exploitation pressure is reduced.

At least 1,739 ha of mangroves were lost between 1990 and 2019, mainly due to anthropogenic activities, representing a decline of 60 ha yr^{-1}. Major areas of mangrove loss and degradation were associated with high human populations. The mean vegetation carbon was estimated at 166.56±23.41 Mg C ha^{-1} (or 611.28 Mg CO_2e ha^{-1}). The total emissions from the loss and degradation of mangrove vegetation in the county were estimated at 41.64 Mg C ha^{-1}, which translates to 9,169.13 Mg CO_2e yr^{-1}. Assuming an offset price of USD 10/Mg CO_2e, the estimated cost of avoided emissions is US$91,691.3 yr^{-1} plus other co-benefits such as fishery support and shoreline protection.

It would be unrealistic to consider stopping mangrove exploitation in Kenya and, by extension, in other developing countries where mangroves are exploited for wood and non-wood resources. This is because mangroves are lifelines for millions of coastal communities worldwide. For instance, in Kenya, mangrove forests meet at least 70% of the wood requirement of adjacent communities, which represent an economic value of approximately USD 24 million per year (Huxham *et al.*, 2015; Manzi & Kirui, 2021). This is in addition to people who frequently depend on mangrove areas for fishery products, traditional medicine, and leisure (Hamza *et al.*, 2020). Therefore, this study provides important outputs for the improved management of mangroves in Lamu County through mapping, resource assessment, and planning. These data may be used to revise harvest and restoration plans. These harvest plans will reduce the pressure on mangroves and ensure that mangrove harvesters continue to earn income while enhancing forest sustainability. In addition, this will bring other co-benefits such as biodiversity conservation, shoreline protection and support to fisheries. Furthermore, mainstreaming mangroves and associated blue carbon ecosystems into national development and climate change agendas could accelerate emissions reductions in Kenya, thereby contributing to the country's commitment to the Paris Agreement. This will help the country to conserve its mangroves as major carbon sinks, contributing directly to the Sustainable Development Goal (SDG) 13 (climate action). Improved mangrove management will, in turn, enhance marine productivity in the country, further contributing to SDG 14 (life underwater).

Kenya pioneered a carbon incentive scheme involving mangrove forests dubbed MIKOKO PAMOJA[1], which was the world's first community-led mangrove conservation and restoration project funded by carbon credits. Revenue generated from the sale of these carbon credits is used to support local development projects and mangrove reforestation activities. This carbon offset project is an excellent example of a "triple win" situation in Kenya, with benefits for climate, community, and biodiversity conservation (Wylie *et al.*, 2016; Flint *et al.*, 2018; Windham-Myers *et al.*, 2018; Kairo *et al.*, 2019; UNDP, 2020).

5.3 Recommendation

This study recommends the measurement of other carbon pools to determine total ecosystem carbon stocks. The present study focused on living vegetative carbon, excluding sediment organic carbon (the largest carbon pool in mangroves), litter, and deadwood. This is of significant interest as the country has recently incorporated blue carbon into its nationally determined contributions (or NDCs); hence, other carbon pools are important for national carbon accounting and reporting. There is also a need to develop a biomass growth model for Lamu mangroves by monitoring juvenile growth over a long period.

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[1] The project (MIKOKO PAMOJA) is accredited by Plan Vivo Foundation, an international non-governmental organization that supports smallholders and communities wishing to manage their land and natural resources more sustainably by selling Plan Vivo Certificates (PCVs), which are recorded and tracked through the independent Market Environmental Registry. This successful initiative is currently being replicated in a similar project at Vanga, Kwale County, Kenya.

REFERENCES

Adame, M. F., Connolly, R. M., Turschwell, M. P., Lovelock, C. E., Fatoyinbo, T., Lagomasino, D., ... & Brown, C. J. (2021). Future carbon emissions from global mangrove forest loss. *Global Change Biology*, *27*(12), 2856-2866.

Alavaisha, E., & Mangora, M. M. (2016). Carbon stocks in the small estuarine mangroves of Geza and Mtimbwani, Tanga, Tanzania. *International Journal of Forestry Research*, 1-11.

Alemayehu, F., & Wekesa, C. (2017). Structure and natural regeneration of mangroves in areas exposed to different agents of degradation in Kenya. *Journal of Global Biosciences*, *6*(1), 4695-4707.

Alongi, D. M. (2014). Carbon cycling and storage in mangrove forests. *Annual Reviews in Marine Science, 6*(1), 195-219.

Alongi, D. M., & Mukhopadhyay, S. K. (2015). Contribution of mangroves to coastal carbon cycling in low latitude seas. *Agricultural and forest meteorology*, *213*, 266-272.

Alongi, D. M. (2020). Global significance of mangrove blue carbon in climate change mitigation. *Science, 2*(3), 67.

Anonymous (2021). Roots of hope: the socio-economic value of mangroves in the Western Indian Ocean region. Global Mangrove Alliance, 1-13.

Azad, M. S., Kamruzzaman, M., & Kanzaki, M. (2021). Canopy gaps influence regeneration dynamics in cyclone affected mangrove stands in medium saline zone of the Sundarbans, Bangladesh. *Acta Ecologica Sinica*, *41*(4), 296-303.

Barbier, E. B. (2016). The protective service of mangrove ecosystems: A review of valuation methods. *Marine Pollution Bulletin*, *109*(2), 676-681.

Bishop, Y.M., Fienberg, S.E., Holland, P. W., Light, R. J., & Mosteller, F. (1977). Book Review: Discrete multivariate analysis: Theory and practice. *Applied Psychological Measurement*, *1*(2), 297-306.

Bouillon, S., Borges, A. V., Castañeda-Moya, E., Diele, K., Dittmar, T., Duke, N. C., ... & Twilley, R. R. (2008). Mangrove production and carbon sinks: a revision of global budget estimates. *Global biogeochemical cycles*, *22*(2), 1-12.

Bosire, J. O., Dahdouh-Guebas, F., Kairo, J. G., & Koedam, N. (2003). Colonization of non-planted mangrove species into restored mangrove stands in Gazi Bay, Kenya. *Aquatic Botany*, *76*(4), 267-279.

Bosire, J. O., Kairo, J. G., Kazungu, J., Koedam, N., & Dahdouh-Guebas, F. (2005). Predation on propagules regulates regeneration in a high-density reforested mangrove plantation. *Marine Ecology Progress Series*, *299*, 149-155.

Bosire, J. O., Kaino, J. J., Olagoke, A. O., Mwihaki, L. M., Ogendi, G. M., Kairo, J. G., ... & Macharia, D. (2014). Mangroves in peril: unprecedented degradation rates of peri-urban mangroves in Kenya. *Biogeosciences*, *11*(10), 2623-2634.

Bosire, J. O., Mangora, M. M., Bandeira, S. O., Rajkaran, A., Ratsimbazafy, R., Appadoo, C., & Kairo, J. G. (2016). *Mangroves of the Western Indian Ocean: Status and Management.* WIOMSA, Zanzibar Town, 1-161.

Boxem, H. W., De Meester, T., & Smalin, E. M. (1987). Soils of the Kilifi Area. *Reconnaissance Soil Survey Report No. R11*. Nairobi: Kenya Soil Survey, 1-136.

Bryan-Brown, D. N., Connolly, R. M., Richards, D. R., Adame, F., Friess, D. A., & Brown, C. J. (2020). Global trends in mangrove forest fragmentation. *Scientific Reports, 10*(1), 1-8.

Bundotich, G., Karachi, M., Fondo, E., and Kairo, J.G. (2009). Structural inventory of mangrove forests in Ngomeni. *Advance in Coastal Ecology, 20,*111-121.

Bunting, P., Rosenqvist, A., Lucas, R. M., Rebelo, L. M., Hilarides, L., Thomas, N., ... & Finlayson, C. M. (2018). The global mangrove watch—a new 2010 global baseline of mangrove extent. *Remote Sensing, 10*(10), 1669.

Cameron, C., Hutley, L. B., Friess, D. A., & Brown, B. (2019). Community structure dynamics and carbon stock change of rehabilitated mangrove forests in Sulawesi, Indonesia. *Ecological Applications, 29*(1), e01810.

Cancino, J., & Gadow, K. V. (2002). Stem number guide curves for uneven-aged forests development and limitations. In *Continuous Cover Forestry,* Springer, Dordrecht, 163-174.

Caswell, P.V. (1953). The Geology of the Mombasa Kwale Area. Coral Survey. Kenya Report No. 24. Nairobi: Ministry of Environment and Natural Resources, 1-65.

Chmura, G. L., Anisfeld, S. C., Cahoon, D. R., & Lynch, J. C. (2003). Global carbon sequestration in tidal, saline wetland soils. *Global Biogeochemical Cycles, 17*(4).

Chong, P. W. (1988). Proposed Integrated Forest Management Planning and Utilization of Mangrove Resources in the Terraba-Sierpe Reserve, Costa Rica. *FAO, Rome*.

Cintron, G., & Schaeffer Novelli, Y. (1984). Methods for studying mangrove structure. Monographs on *Oceanographic Methodology, 8,* 91-113.

Clutter, J. L., Fortson, J. C., Pienaar, L. V., Brister, G. H., & Bailey, R. L. (1983). *Timber management: A quantitative approach*. John Wiley & Sons, New York, 1-333.

Cohen, R., Kaino, J., Okello, J. A., Bosire, J. O., Kairo, J. G., Huxham, M., & Mencuccini, M. (2013). Propagating uncertainty to estimates of above-ground biomass for Kenyan mangroves: A scaling procedure from tree to landscape level. *Forest Ecology and Management, 310,* 968-982.

County Government of Lamu (2018). *Lamu County Integrated Development Plan 2018-2022 Lamu:* Department of Finance, Strategy & Economic Planning, 1-457.

Dahdouh-Guebas, F., Van Pottelbergh, I., Kairo, J. G., Cannicci, S., & Koedam, N. (2004). Human-impacted mangroves in Gazi (Kenya): predicting future vegetation based on retrospective remote sensing, social surveys, and tree distribution. *Marine Ecology Progress Series*, *272*, 77-92.

Dahdouh-Guebas, F., & Koedam, N. (2006). Coastal vegetation and the Asian tsunami. *Science, 311*(5757), 37-38.

Deltares (2014). Habitat requirements for mangroves. Available from: https://publicwiki.deltares.nl/display/BWN/Building+Block+Habitat+requirements+for+mangroves

Das, S. (2017). Ecological restoration and livelihood: contribution of planted mangroves as nursery and habitat for artisanal and commercial fishery. *World Development*, *94*, 492-502.

Deng, H., He, J., Feng, D., Zhao, Y., Sun, W., Yu, H., & Ge, C. (2021). Microplastics pollution in mangrove ecosystems: A critical review of current knowledge and future directions. *Science of the Total Environment, 753,* 142041.

Donato, D. C., Kauffman, J. B., Murdiyarso, D., Kurnianto, S., Stidham, M., & Kanninen, M. (2011). Mangroves among the most carbon-rich forests in the tropics. *Nature Geoscience*, *4*(5), 293-297.

Ducey, M. J. (2006). The reverse-J and beyond: developing practical, effective marking guides. In Workshop Proceedings: Implementing Uneven-aged Management in New England: Is It Practical? University of New Hampshire Cooperative Extension Natural Resource Network Report. Durham, NH, 1-45.

Ellison, J. C. (2015). Vulnerability assessment of mangroves to climate change and sea-level rise impacts. *Wetlands Ecology and Management, 23*(2), 115-137.

Erftermeijer, P., de Boer, M., Hilarides, L. (2022). Status of mangroves in the Western Indian Ocean Region. Wetlands International, 1-67.

FAO (1994). *Mangrove Forest Management Guidelines*. Rome: FAO Forestry Paper, *117*, 319

FAO (2016). *Global Forest Resources Assessment*. Rome: Food and Agriculture Organization of the United Nations, 1-54.

FAO (2020). *Global Forest Resources Assessment 2020 - Main Report*. Rome: Food and Agriculture Organization of the United Nations, 1-186.

Feka, N. Z., & Ajonina, G. N. (2011). Drivers causing decline of mangrove in West-Central Africa: a review. *International Journal of Biodiversity Science, Ecosystem Services & Management, 7*(3), 217-230.

Ferguson, W (1993). *A Landscape Ecological Survey of Mangrove Resource of Kenya.* Kenya Wildlife Service and Forest Department, Nairobi, Kenya, 1-165.

Fickert, T. (2020). To plant or not to plant, that is the question: reforestation vs. natural regeneration of hurricane-disturbed mangrove forests in Guanaja (Honduras). *Forests, 11*(10), 1068.

Friess, D. A., Rogers, K., Lovelock, C. E., Krauss, K. W., Hamilton, S. E., Lee, S. Y., ... & Shi, S. (2019). The state of the world's mangrove forests: past, present, and future. *Annual Review of Environment and Resources, 44,* 89-115.

Flint, R., Herr, D., Vorhies, F., & Smith, J. R. (2018). Increasing success and effectiveness of mangrove conservation investments: A guide for project developers, donors, and investors. IUCN, Geneva, Switzerland, and WWF Germany, Berlin, Germany. Supplementary documentation: Case studies of mangrove projects from Kenya, Madagascar and Viet Nam, 76-103

Gallo, N. D., Victor, D. G., & Levin, L. A. (2017). Ocean commitments under the Paris Agreement. *Nature Climate Change, 7*(11), 833-838.

Gamon, J.A., C.B. Gield, M.L. Goulden, & K.L. Griffin. (1995). Relationship between NDVI, canopy structure and photosynthesis in three Californian vegetation types. *Ecological Applications, 5,* 28-41.

Gao, B. C. (1996). NDWI—A normalized difference water index for remote sensing of vegetation liquid water from space. *Remote Sensing of environment, 58*(3), 257-266.

Gillerot, L., Vlaminck, E., De Ryck, D. J., Mwasaru, D. M., Beeckman, H., & Koedam, N. (2018). Inter-and intraspecific variation in mangrove carbon fraction and wood specific gravity in Gazi Bay, Kenya. *Ecosphere, 9*(6), e02306.

Gillis, L. G., Belshe, E. F., & Narayan, G. R. (2017). Deforested mangroves affect the potential for carbon linkages between connected ecosystems. *Estuaries and Coasts*, *40*(4), 1207-1213.

Giri, C., Ochieng, E., Tieszen, L. L., Zhu, Z., Singh, A., Loveland, T., Masek, J., & Duke, N. (2011). Status and distribution of mangrove forests of the world using earth observation satellite data: Status and distributions of global mangroves. *Global Ecology and Biogeography*, *20*(1), 154-159.

Giri, C. (2016). Observation and monitoring of mangrove forests using remote sensing: opportunities and challenges. *Remote Sensing*, *8*(9), 783.

Githaiga, M. N., Kotut, K., Kariuki, F., & Kairo, J. G. Structure and biomass accumulation of natural mangrove forest at Gazi Bay, Kenya. International Journal of Bonorowo Wetlands, *10*(1), 18-32.

Global Carbon Project (2021). Supplemental data of Global Carbon Project 2021 (1.0) [Data set]. Global Carbon Project. https://doi.org/10.18160/gcp-2021.

GoK (2017). *National Mangrove Ecosystem Management Plan.* Kenya Forest Service, Nairobi, Kenya. American Journal of Transplantation *18*(1), 1-115.

GoK (2020). *Kenya's Update Nationally Determined Contributions (NDCs).* Nairobi, Ministry of Environment and Forestry, 1-20.

Goldberg, L., Lagomasino, D., Thomas, N., & Fatoyinbo, T. (2020). Global declines in human-driven mangrove loss. *Global Change Biology*, *26*(10), 5844-5855.

Gordon, I., & Maes, K. (2003). Die-Back in Sonneratia Alba in Kenyan Mangroves is due to Attack by a Cerambycid Beetle and a Metabellid Moth. The Netherlands: Studies from Kenya Research African Studies Centre, Leiden, 281–290

Gress, S. K., Huxham, M., Kairo, J. G., Mugi, L. M., & Briers, R. A. (2017). Evaluating, predicting, and mapping belowground carbon stores in Kenyan mangroves. *Global Change Biology*, *23*(1), 224-234.

Gupta, K., Mukhopadhyay, A., Giri, S., Chanda, A., Majumdar, S. D., Samanta, S., ... & Hazra, S. (2018). An index for discrimination of mangroves from non-mangroves using LANDSAT 8 OLI imagery. *MethodsX*, *5*, 1129-1139.

Hamilton, S. E., & Casey, D. (2016). Creation of a high spatio-temporal resolution global database of continuous mangrove forest cover for the 21st century (CGMFC-21): CGMFC-21. *Global Ecology and Biogeography*, *25*(6), 729-738.

Hamilton, S. E., & Friess, D. A. (2018). Global carbon stocks and potential emissions due to mangrove deforestation from 2000 to 2012. *Nature Climate Change*, *8*(3), 240-244.

Hamza, A. J., Esteves, L. S., Cvitanovic, M., & Kairo, J. (2020). Past and present utilization of mangrove resources in Eastern Africa and drivers of change. *Journal of Coastal Research*, Special issue No. 95, 39-44.

Herr, D., & Landis, E. (2016). *Coastal blue carbon ecosystems. Opportunities for Nationally Determined Contributions.* International Union for Conservation of Nature, 1-27.

Holdridge, L.R., Grenke, W.C., Hatheway. W.H., Liang, T., & Tosie Jr, J.A., (1971). Forest environments in tropical life zones: a pilot study. *Pergamon Press, Oxford*, 747.

Howard, J., Hoyt, S., Isensee, K., Telszewski, M., and Pidgeon, E. (2014). Coastal blue carbon: Methods for assessing carbon stocks and emissions factors in mangroves, tidal salt marshes, and seagrass meadows. *Arlington, VA: Conservation International, Intergovernmental Oceanographic Commission of UNESCO, International Union for Conservation of Nature,* 99-107.

Howard, J., Sutton-Grier, A., Herr, D., Kleypas, J., Landis, E., Mcleod, E., ... & Simpson, S. (2017). Clarifying the role of coastal and marine systems in climate mitigation. *Frontiers in Ecology and the Environment*, *15*(1), 42-50.

Huxham, M., Emerton, L., Kairo, J., Munyi, F., Abdirizak, H., Muriuki, T., ... & Briers, R. A. (2015). Applying climate compatible development and economic valuation to coastal management: a case study of Kenya's mangrove forests. *Journal of Environmental Management*, *157*, 168-181.

Idha, M. (1998). The Mangroves of Lamu: History, Socioeconomic and Conservation Issues. *In Eastern Africa Regional Workshop on Mangroves and Shrimp Aquaculture*. East Africa Wildlife Society, 1-14.

IPBES (2019). Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. *Zenodo,* 1-1148.

IPCC (2014). 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. *Wetlands,* 2, 1-354.

Janzen, D. H. (1985). Mangroves: where's the understory? *Journal of Tropical Ecology*, *1*(1), 89-92.

Jenoh, E. M., Robert, E. M., Lehmann, I., Kioko, E., Bosire, J. O., Ngisiange, N., ... & Koedam, N. (2016). Wide ranging insect infestation of the pioneer mangrove Sonneratia alba by two insect species along the Kenyan coast. *PLoS One*, *11*(5), e0154849.

Jones, T., Ratsimba, H., Ravaoarinorotsihoarana, L., Glass, L., Benson, L., Teoh, M., Carro, A., Cripps, G., Giri, C., Gandhi, S., Andriamahenina, Z., Rakotomanana, R., & Roy, P.-F. (2015). The dynamics, ecological variability and estimated carbon stocks of mangroves in Mahajamba Bay, Madagascar. *Journal of Marine Science and Engineering, 3*(3), 793-820.

Kairo, G., & Dahdouh-Guebas, F. (1995). Conservation Status of Mangrove Resources in Kenya. *Aquaculture (marine and fresh water, tonnes)*, 97, 1-200.

Kairo, J. G., Dahdouh-Guebas, F., Bosire, J., & Koedam, N. (2001). Restoration and management of mangrove systems—A lesson for and from the East African region. *South African Journal of Botany, 67*(3), 383-389.

Kairo, J. G., Kivyatu, B., & Koedam, N. (2002a). Application of remote sensing and GIS in the management of mangrove forests within and adjacent to Kiunga Marine Protected Area, Lamu, Kenya. *Environment, Development and Sustainability, 4*(2), 153-166.

Kairo, J.G., Dahdouh-Guebas, F., Gwada, P., & Ochieng, C.A. (2002b) Regeneration Status of Mangrove Forests in Mida Creek, Kenya : A Compromised or Secured Future ? *Ambio 31*, 562-568.

Kairo, J. G., Wanjiru, C., & Ochiewo, J. (2009). Net pay: economic analysis of a replanted mangrove plantation in Kenya. *Journal of sustainable forestry*, *28*(3-5), 395-414.

Kairo, J. G., Hamza, A. J., & Wanjiru, C. (2019). Mikoko Pamoja: A demonstrably effective community-based blue carbon project in Kenya. In *A blue carbon primer: The state of coastal wetland carbon science, practice, and policy*; Windham-

Myers, L., Crooks, S., Troxler, T., Eds.; CRC Press Taylor & Francis: London, UK, 341-351.

Kamal, M., & Johansen, K. (2017). Explicit area-based accuracy assessment for mangrove tree crown delineation using geographic object-based image analysis (GEOBIA). In SPIE Earth Resources and Environmental Remote Sensing/GIS Applications VIII. Bellingham, WA: International Society for Optics and Photonics, 10428, 164-169.

Kamau, J., Ngisiange, N., Ochola, O., Kilionzi, J., Kimeli, A., Mahongo, S. B., ... & Osore, M. (2020). Factors influencing spatial patterns in primary productivity in Kenyan territorial waters. *Western Indian Ocean Journal of Marine Science*, 9-18.

Kamruzzaman, M., Ahmed, S., Paul, S., Rahman, M. M., & Osawa, A. (2018). Stand structure and carbon storage in the oligohaline zone of the Sundarbans mangrove forest, Bangladesh. *Forest Science and Technology, 14*(1), 23-28.

Kathiresan, K. (2012). Importance of mangrove ecosystem. *International Journal of Marine Science*, *2*(10), 70-89.

Kauffman, J. B., Heider, C., Cole, T. G., Dwire, K. A., & Donato, D. C. (2011). Ecosystem carbon stocks of Micronesian mangrove forests. *Wetlands*, *31*(2), 343-352.

Kauffman, J. B., & Donato, D. C. (2012). *Protocols for the measurement, monitoring and reporting of structure, biomass, and carbon stocks in mangrove forests*. Working Paper 86. CIFOR, Bogor, Indonesia, 1-50.

Kauffman, J. B., Heider, C., Norfolk, J., & Payton, F. (2014). Carbon stocks of intact mangroves and carbon emissions arising from their conversion in the Dominican Republic. *Ecological Applications*, *24*(3), 518-527.

Kerr, G. (2014). The management of silver fir forests: de Liocourt (1898) revisited. *Forestry*, *87*(1), 29-38.

Kershaw Jr, J. A., Ducey, M. J., Beers, T. W., & Husch, B. (2016). Forest mensuration. John Wiley & Sons, 10(5), 1-630.

Kirui, K. B., Kairo, J. G., Bosire, J., Viergever, K. M., Rudra, S., Huxham, M., & Briers, R. A. (2013). Mapping of mangrove forest land cover change along the Kenya coastline using Landsat imagery. *Ocean & Coastal Management*, *83*, 19-24.

Komiyama, A., Poungparn, S., Kato, S., (2005). Common allometric equations for estimating the tree weight of mangroves. *Journal of Tropical Ecology*, 21, 471-477.

Komiyama, A., Ong, J. E., & Poungparn, S. (2008). Allometry, biomass, and productivity of mangrove forests: A review. *Aquatic Botany*, *89*(2), 128-137.

Krauss, K.W. and Osland, M.J. (2020). Tropical cyclones and the organization of mangrove forests: a review. Annals of Botany, 125, 213-234

Kristiningrum, R., Lahjie, A. M., Yusuf, S., & Ruslim, Y. (2019). Species diversity, stand productivity, aboveground biomass, and economic value of mangrove ecosystem in Mentawir Village, East Kalimantan, Indonesia. *Biodiversitas Journal of Biological Diversity*, *20*(10), 2848-2857.

Kuria, Z. (2013). Groundwater distribution and aquifer characteristics in Kenya. In *Developments in earth surface processes.* Elsevier, 16, 83-107.

Lang'at, J.K.S. (2008). *Variability of Mangrove Forests along the Kenyan Coast*. Kenya Marine and Fisheries Research Institute: Mombasa, Kenya, 1-26.

Lang'at, J. K. S., Kairo, J. G., Mencuccini, M., Bouillon, S., Skov, M. W., Waldron, S., & Huxham, M. (2014). Rapid losses of surface elevation following tree girdling and cutting in tropical mangroves. *PLoS ONE*, *9*(9), e107868.

Lee, S. Y., Primavera, J. H., Dahdouh-Guebas, F., McKee, K., Bosire, J. O., Cannicci, S., ... & Record, S. (2014). Ecological role and services of tropical mangrove ecosystems: a reassessment. *Global Ecology and Biogeography*, 23, 726-743.

Lopez, O. (2021). Ocean-Based Climate Solutions in Nationally Determined Contributions. Washington, DC: Ocean Conservancy, 1-40.

Lovelock, C. E., Fourqurean, J. W., & Morris, J. T. (2017). Modeled CO_2 emissions from coastal wetland transitions to other land uses: tidal marshes, mangrove forests, and seagrass beds. *Frontiers in Marine Science*, 4, 143.

Lovelock, C. E., & Duarte, C. M. (2019). Dimensions of blue carbon and emerging perspectives. *Biology Letters*, *15*(3), 20180781.

Lugo, A. E., & Snedaker, S. C. (1974). The ecology of mangroves. *Annual Review of Ecology and Systematics*, 5, 39-64.

Macreadie, P. I., Atwood, T. B., Seymour, J. R., Fontes, M. S., Sanderman, J., Nielsen, D. A., & Connolly, R. M. (2019). Vulnerability of seagrass blue carbon to microbial attack following exposure to warming and oxygen. *Science of the total environment*, *686*, 264-275.

Macreadie, P. I., Costa, M. D., Atwood, T. B., Friess, D. A., Kelleway, J. J., Kennedy, H., ... & Duarte, C. M. (2021). Blue carbon as a natural climate solution. *Nature Reviews Earth & Environment*, *2*(12), 826-839.

Manzi, H., and Kirui, V.C. (2021). Assessment of the socio-economic role of mangroves and their conservation framework in Kenya. International Union for Conservation of Nature (IUCN) and Geo-Spatial Research International, Project Report, 1-229.

Martens, E. (1996). *The Mangroves of Kenya: general information.* Compiled for Netherlands Wetlands Conservation and Training Programme. Kenya Wildlife Service Coast Region Headquarters, Mombasa, Kenya, 1-54.

Martin, E.B., & Martin, C.P. (1978). *Cargoes of the East. The ports. Trade and culture of the Arabian Seas and Western Indian Ocean*. Elm Tree Books, London.

Mohamed, M. O. S., Neukermans, G., Kairo, J. G., Dahdouh-Guebas, F., & Koedam, N. (2009). Mangrove forests in a peri-urban setting: the case of Mombasa (Kenya). *Wetlands Ecology and Management*, *17*(3), 243-255.

Mugi, L. M., Kiss, D., Kairo, J. G., & Huxham, M. R. (2022). Stocks and productivity of dead wood in mangrove forests: A systematic literature review. *Frontiers in Forests and Global Change*, *5*, 767337

Mungai, F., Kairo, J., Mironga, J., Kirui, B., Mangora, M., & Koedam, N. (2019). Mangrove cover and cover change analysis in the transboundary area of Kenya and Tanzania during 1986-2016. *Journal of the Indian Ocean Region*, *15*(2), 157-176.

Murdiyarso, D., Purbopuspito, J., Kauffman, J. B., Warren, M. W., Sasmito, S. D., Donato, D. C., Manuri, S., Krisnawati, H., Taberima, S., & Kurnianto, S. (2015). The potential of Indonesian mangrove forests for global climate change mitigation. *Nature Climate Change, 5*(12), 1089-1092.

Murray, B. C. (2012). Economics: mangroves' hidden value. *Nature Climate Change*, *2*(11), 773-774.

Nicolau, D., Macamo, C., Mabilana, A., & Bandeira, S. O. (2017). Mangrove change detection, structure and condition in a protected area of Eastern Africa: The case of Quirimbas national Park, Mozambique. *Western Indian Ocean Journal of Marine Science*, *16*(1), 47-60.

Nijsten, G. J., Christelis, G., Villholth, K. G., Braune, E., & Gaye, C. B. (2018). Transboundary aquifers of Africa: Review of the current state of knowledge and progress towards sustainable development and management. *Journal of Hydrology: Regional Studies*, *20*, 21-34.

Njana, M. A., Zahabu, E., and Malimbwi, R. E. (2018). Carbon stocks and productivity of mangrove forests in Tanzania, Southern forests. Journal of Forest Science, *80*(3), 217-232.

Njana, M. A. (2020). Structure, growth, and sustainability of mangrove forests of mainland Tanzania. *Global Ecology and Conservation*, 24, e01394.

Njiru, D. M., Githaiga, M. N., Nyaga, J. M., Lang'at, K. S., & Kairo, J. G. (2022). Geomorphic and climatic drivers are key determinants of structural variability of mangrove forests along the Kenyan Coast. *Forests*, *13*(6), 870.

Okello, J., Dahdouh-Guebas, F., Koedam, N., Beeckman, H., Kairo, J., & Schmitz, N. (2013). Self-sustenance potential of peri-urban mangroves: a case of Mtwapa creek Kenya. *Journal of Environmental Science and Water Resource, 2*(8), 277-289.

Okello, J. A., Alati, V. M., Kodikara, S., Kairo, J., Dahdouh-Guebas, F., & Koedam, N. (2019). The status of Mtwapa Creek mangroves as perceived by the local communities. Western Indian Ocean *Journal of Marine Science, 18*(1), 67-81.

Okello, J. A., Osuka, K., Mbugua, J., Maina, G. W., & Samoilys, M. A. (2022). The structure of the mangrove forests of Kiunga-Pate Island conservancies in Kenya are shaped by selective harvesting and natural mortalities. *Estuarine, Coastal and Shelf Science*, 107885.

Olagoke, A. O., Bosire, J. O., & Berger, U. (2013). Regeneration of *Rhizophora mucronata* (Lamk.) in degraded mangrove forest: Lessons from point pattern analyses of local tree interactions. *Acta Oecologica*, *50*, 1-9.

Omodei-Zaroni L., & Cortini (2000). A multi-criterion analysis for the sustainable use of natural resources in mangrove ecosystems with the involvement of local community. Proceedings of the European Symbosium. On 'European farming and rural system and extension into the next Millenium', Volos, Greece, 3-7.

Paris Agreement (2015). Report of the Conference of the parties to the United Nations framework convention on climate change, HeinOnline, *4*, 2017.

Pendleton, L., Donato, D. C., Murray, B. C., Crooks, S., Jenkins, W. A., Sifleet, S., Craft, C., Fourqurean, J. W., Kauffman, J. B., Marbà, N., Megonigal, P., Pidgeon, E., Herr, D., Gordon, D., & Baldera, A. (2012). Estimating global "blue carbon" emissions from conversion and degradation of vegetated coastal ecosystems. *PLoS ONE, 7*(9), e43542.

Pool, D. J., Snedaker, S. C., & Lugo, A. E. (1977). Structure of mangrove forests in Florida, Puerto Rico, Mexico, and Costa Rica. *Biotropica*, 195-212.

Raghbor, P., Seeruttun, L. D., & Appadoo, C. (2022). First assessment of the blue carbon storage of Rhizophora and Bruguiera mangrove stands on the island of Mauritius (Western Indian Ocean). Southern Forests: *A Journal of Forest Science, 84*(1), 70-74.

Rahman, M., Nabiul Islam Khan, M., Fazlul Hoque, A. K., & Ahmed, I. (2015). Carbon stock in the Sundarbans mangrove forest: spatial variations in vegetation types and salinity zones. *Wetlands Ecology and Management*, *23*(2), 269-283.

Rasquinha, D. N., & Mishra, D. R. (2021). Impact of wood harvesting on mangrove forest structure, composition, and biomass dynamics in India. *Estuarine, Coastal and Shelf Science*, 248, 106974.

Riungu, P. M., Nyaga, J. M., Githaiga, M. N., & Kairo, J. G. (2022). Value chain and sustainability of mangrove wood harvesting in Lamu, Kenya. *Trees, Forests and People*, *9*, 100322.

Roberts, R.W., & Ruara, W.W. (1967). *Lamu mangroves inventory*. Part 1 and 2. Spartan Air Services (Canada) and Forest Department, Nairobi, Kenya. Ministry of Environment and Natural Resource.

Rozainah, M. Z., Nazri, M. N., Sofawi, A. B., Hemati, Z., & Juliana, W. A. (2018). Estimation of carbon pool in soil, above and below ground vegetation at different types of mangrove forests in Peninsular Malaysia. *Marine Pollution Bulletin*, *137*, 237-245.

Runya, R. M., Karani, N. J., Muriuki, A., Maringa, D. M., Kamau, A. W., Ndomasi, N., ... & Okello, J. A. (2022). Local perceptions, opportunities, and challenges of community-based ecotourism in Gazi Bay, Kenya. *Western Indian Ocean Journal of Marine Science, 21*(2), 95-108.

Saenger, P., Ragavan, P., Sheue, C. R., López-Portillo, J., Yong, J. W. H., & Mageswaran, T. (2019). Mangrove biogeography of the Indo-Pacific. In *Sabkha ecosystems*. Springer, Cham, 379-400.

Samoilys, M. A., & Kanyange, N. W. (2008). Assessing links between marine resources and coastal peoples' livelihoods: perceptions from Tanga, Tanzania. *IUCN Eastern and Southern Africa Regional Office, Nairobi*, 1-30.

Sanderman, J., Hengl, T., Fiske, G., Solvik, K., Adame, M. F., Benson, L., … & Landis, E. (2018). A global map of mangrove forest soil carbon at 30 m spatial resolution. *Environmental Research Letters*, *13*(5), 055002.

Sawale, A. K., & Thivakaran, G. A. (2013). Structural characteristics of mangrove forest of Kachchh, Gujarat. *Journal of the Marine Biological Association of India*, *55*(1), 6.

Scales, I. R., & Friess, D. A. (2019). Patterns of mangrove forest disturbance and biomass removal due to small-scale harvesting in southwestern Madagascar. *Wetlands Ecology and Management*, *27*(5), 609-625.

Siikamäki, J., Sanchirico, J. N., & Jardine, S. L. (2012). Global economic potential for reducing carbon dioxide emissions from mangrove loss. *Proceedings of the National Academy of Sciences*, *109*(36), 14369-14374.

Simard, M., Fatoyinbo, L., Smetanka, C., Rivera-Monroy, V. H., Castañeda-Moya, E., Thomas, N., & Van der Stocken, T. (2019). Mangrove canopy height globally related to precipitation, temperature, and cyclone frequency. *Nature Geoscience, 12*(1), 40-45.

Sitoe, A., Mandlate, L., & Guedes, B. (2014). Biomass and Carbon Stocks of Sofala Bay Mangrove Forests. *Forests*, *5*(8), 1967-1981.

Smith, T. J. (1987). Effects of light and intertidal position on seedling survival and growth in tropical tidal forests. *Journal of Experimental Marine Biology and Ecology*, *110*(2), 133-146.

Snyder, C. M., Feher, L. C., Osland, M. J., Miller, C. J., Hughes, A. R., & Cummins, K. L. (2022). The distribution and structure of mangroves (*Avicennia germinans* and *Rhizophora mangle*) near a rapidly changing range limit in the Northeastern Gulf of Mexico. *Estuaries and Coasts*, *45*(1), 181-195.

Spalding, M., Kainuma, M., & Collins, L. (2010). *World Atlas of Mangroves*. A collaborative project of ITTO, ISME, FAO, UNESCO-MAB, UNEP-WCMC and UNU-INWEH. Earth scan Publishers Ltd, London, UK and Washington, DC, USA, 319.

Spalding, M. D., & Leal, M. (2021). The state of the world's mangroves 2021. *Global Mangrove Alliance,* 1-49.

Stringer, C. E., Trettin, C. C., Zarnoch, S. J., & Tang, W. (2015). Carbon stocks of mangroves within the Zambezi River Delta, Mozambique. *Forest Ecology and Management*, *354,* 139-148.

Sukardjo, S. (1987). Natural regeneration status of commercial mangrove species (*Rhizophora apiculata* and *Bruguiera gymnorrhiza*) in the mangrove forest of Tanjung Bungin, Banyuasin District, South Sumatra. *Forest Ecology and Management*, *20*(3-4), 233-252.

Taillardat, P., Friess, D. A., & Lupascu, M. (2018). Mangrove blue carbon strategies for climate change mitigation are most effective at the national scale. *Biology Letters*, *14*(10), 20180251.

Taraska, G. (2018). Integrating Ocean and Climate Policy: A Next Step Forward in the Global Climate Effort. Washington, DC: Center for American Progress.

Thom, B. G. (1982). Mangrove ecology - A geomorphological perspective. Pages 3-17 *in* B. F. Clough, editor. *Mangrove ecosystems in Australia: structure, function and management*. Australian National University Press, Canberra, Australian Capital Territory, Australia.

Thom, B.G. (1984). "Coastal landforms and geomorphic processes". In *The Mangrove Ecosystem: Research Methods. Monographs on Oceanographic Methodology, 8*, 3-17.

Thomas, N., Lucas, R., Bunting, P., Hardy, A., Rosenqvist, A., & Simard, M. (2017). Distribution and drivers of global mangrove forest change, 1996-2010. *PLoS ONE, 12*(6), 1-14.

Thompson, B. S., Clubbe, C. P., Primavera, J. H., Curnick, D., & Koldewey, H. J. (2014). Locally assessing the economic viability of blue carbon: A case study from Panay Island, the Philippines. *Ecosystem Services*, *8*, 128-140.

Tomlinson, P. B. (2016). *The botany of mangroves.* Cambridge University Press, Cambridge, New York, U. S. A

Twilley, R. R. (1995). Properties of mangrove ecosystems related to the energy signature of coastal environments. Pages 43-62 *in* C. A. S. Hall, editor. *Maximum power: the ideas and applications of H. T. Odum*. University Press of Colorado, Denver, Colorado, USA.

UNDP (2020). Mikoko Pamoja, Kenya. United Nations Development Programme, Equator Initiative Case Study Series, New York, 1-16.

UNEP/WIOMSA (2015). The Regional State of the Coast Report: Western Indian Ocean. UNEP-Nairobi Convention and WIOMSA, Nairobi, Kenya, 1-546.

UNEP-Nairobi Convention/USAID/WIOMSA (2020). Guidelines on Mangrove Ecosystem Restoration for the Western Indian Ocean Region. UNEP, Nairobi, 77.

UNFCCC (2012). "Views on Specific Themes to Be Addressed at the Research Dialogue, Including Information on Technical and Scientific Aspects of Emissions and Removals of All Greenhouse Gases from Coastal and Marine Ecosystems, Submissions from Parties, Addendum 2." Subsidiary Body for Scientific and Technological Advice, Thirty-Sixth Session.

Velázquez-Pérez, C., Tovilla-Hernández, C., Romero-Berny, E. I., & Jesús-Navarrete, A. D. (2019). Mangrove structure and its influence on the carbon storage in La Encrucijada Reserve, Chiapas, Mexico. *Madera y Bosques, 25*(3), e2531885.

Wang'ondu, V. W., Kairo, J. G., Kinyamario, J. I., Mwaura, F. B., Bosire, J. O., Dahdouh-Guebas, F., & Koedam, N. (2010). Phenology of Avicennia marina (Forsk.) Vierh. in a disjunctly-zoned mangrove stand in Kenya. *Western Indian Ocean Journal of Marine Science*, *9*(2), 135-144.

Wanjiru, C., Rueckert, S., & Huxham, M. (2021). Composition and structure of the mangrove fish and crustacean communities of Vanga Bay, Kenya. *Western Indian Ocean Journal of Marine Science, 20*(2), 25-44.

Ward, R. D., Friess, D. A., Day, R. H., & Mackenzie, R. A. (2016). Impacts of climate change on mangrove ecosystems: A region by region overview. *Ecosystem Health and Sustainability, 2*(4), e01211.

Whitfield, A. K. (2017). The role of seagrass meadows, mangrove forests, salt marshes and reed beds as nursery areas and food sources for fishes in estuaries. *Reviews in Fish Biology and Fisheries*, *27*(1), 75-110.

Windham-Myers, L., Crooks, S., & Troxler, T. G. (Eds.). (2018). *A blue carbon primer: the state of coastal wetland carbon science, practice and policy*. CRC Press.

Wylie, L., Sutton-Grier, A. E., & Moore, A. (2016). Keys to successful blue carbon projects: lessons learned from global case studies. *Marine Policy*, *65*, 76-84.

Xu, L., Herold, M., Tsendbazar, N. E., Masiliūnas, D., Li, L., Lesiv, M., ... & Verbesselt, J. (2022). Time series analysis for global land cover change monitoring: A comparison across sensors. *Remote Sensing of Environment*, *271*, 112905.

Xu, S., He, Z., Guo, Z., Zhang, Z., Wyckoff, G. J., Greenberg, A., ... & Shi, S. (2017). Genome-wide convergence during evolution of mangroves from woody plants. *Molecular Biology and Evolution*, *34*(4), 1008-1015.

Yudha, R. P., Sugito, Y. S., Sillanpää, M., & Nurvianto, S. (2021). Impact of logging on the biodiversity and composition of flora and fauna in the mangrove forests of Bintuni Bay, West Papua, Indonesia. *Forest Ecology and Management*, 488, 119038.

Zhou, Q., Robson, M., & Pilesjo, P. (1998). On the ground estimation of vegetation cover in Australian rangelands. *International Journal of Remote Sensing*, *19*(9), 1815-1820.

Zhu, J. J., & Yan, B. (2022). Blue carbon sink function and carbon neutrality potential of mangroves. *Science of The Total Environment*, 822, 153438.

Zu Ermgassen, P. S., Mukherjee, N., Worthington, T. A., Acosta, A., da Rocha Araujo, A. R., Beitl, C. M., ... & Spalding, M. (2020). Fishers who rely on mangroves:

Modelling and mapping the global intensity of mangrove-associated fisheries. *Estuarine, Coastal and Shelf Science*, *247*, 106975.

Zulhalifah, Z., Syukur, A., Santoso, D., & Karnan, K. (2021). Species diversity and composition, and above-ground carbon of mangrove vegetation in Jor Bay, East Lombok, Indonesia. *Biodiversitas Journal of Biological Diversity*, *22*(4), 2066-2071

List of publications

I. **Mbatha Anthony**, Githaiga N. Michael, Kiplagat Kotut K, Kairo James, and Mungai Fredrick (2022). Structure and Sustainability of Mangrove Forests in Lamu, Kenya. Journal of Sustainable Forestry. *Taylor and Francis Publishers*. https://doi.org/10.1080/10549811.2022.2123357

II. Kairo James, **Mbatha Anthony**, Murithi Martin Mututa, and Mungai Fredrick (2021). Total Ecosystem Carbon Stocks of Mangroves in Lamu, Kenya; and Their Potential Contributions to the Climate Change Agenda in the Country. *Frontiers in Forests and Global Change.* https://doi.org/10.3389/ffgc.2021.709227

Conference presentations

I. **Mbatha Anthony,** Michael N. Githaiga; Kiplagat Kotut, and James Kairo. Vegetation Carbon Stocks of the Mangroves of Lamu County, Kenya. Presented at the 11th WIOMSA Scientific Symposium held at University of Mauritius.

II. **Mbatha Anthony**, Michael N. Githaiga, Kiplagat Kotut, and James Kairo. Carbon Stocks of the Mangroves of Lamu Archipelago and The Opportunities for Nationally Determined Contributions. Presented at the International Conference for YOUNG Marine Researchers at the University of Bremen, Germany.

III. **Mbatha Anthony**. Impacts of human activities onto the mangroves tree biomass. Presented at Aix-Marseille (CEREGE) from October 18 - 23, 2021. France.

IV. **Mbatha Anthony,** Michael N. Githaiga, Kiplagat Kotut, and James Kairo. Structure and Regeneration Status of Mangrove Forests in Lamu, Kenya. Presented during the 12th WIOMSA Scientific Symposium held on 10 -15 October 2022 at the Boardwalk Convention Centre, Nelson Mandela Bay, South Africa.

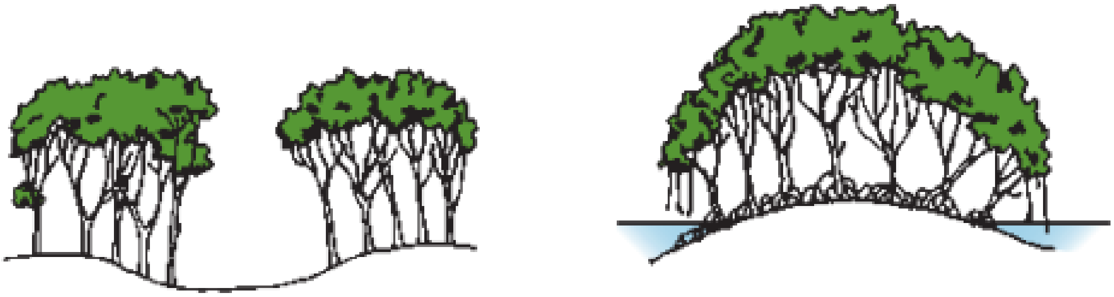
V. **Mbatha Anthony,** Michael Githaiga, Kiplagat Kotut, James Kairo. How Sustainable is Mangrove Harvesting in Lamu, Kenya? An Analysis of Forest Structure. Presented during Aquatic Resources and Blue Economy Conference (ARBEC) which was held on 27th November to 2nd December at Kisumu.

APPENDICES

Appendix 1: Mangrove community types (redrawn from Lugo & Snedaker, 1974 by UNEP-Nairobi Convention/USAID/WIOMSA, 2020)

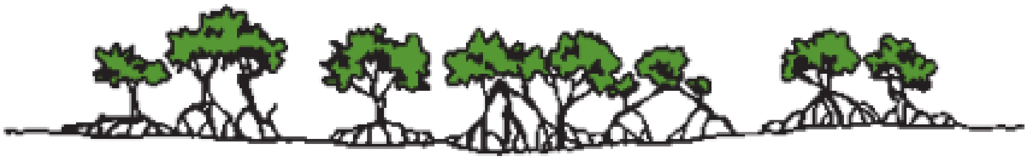
Fringe

Riverine



Basin

Over-wash



Dwarf

Appendix 2: Areal extent and proportions of degraded mangrove areas in Kenya

County	Mangrove area (ha)	Degraded mangroves (ha)	% Degraded area
Lamu	37,350	14,407	38.6
Kilifi	8,536	3,422	40.0
Kwale	8,354	3,725	44.6
Mombasa	3,771	1,850	49.1
Tana River	3,260	1,180	36.2
Total	**61,271**	**24,585**	**40.1**

Source: GoK (2017)

Appendix 3: Mangrove species found in Kenya and their main uses

Species	Local name	Main use
Rhizophora mucronata	Mkoko	Poles, dye, firewood, fencing, charcoal
Bruguiera gymnorhiza	Muia	Poles, firewood, charcoal
Ceriops tagal	Mkandaa	Poles, firewood, charcoal
Sonneratia alba	Mlilana	Boat ribs, poles, firewood
Avicennia marina	Mchu	Firewood, poles
Lumnitzera racemosa	Kikandaa	Fencing poles, firewood
Xylocarpus granatum	Mkomafi	Furniture, poles, firewood
Xylocarpus moluccensis	Mkomafi dume	Fencing poles, firewood
Heritiera littoralis	Msikundazi	Timber, poles, boat mast

Source: Kairo *et al.* (2009)

Appendix 4: Description of the mangrove management blocks in Lamu County

Management block	Location and characteristics
Northern swamps	Extends from Kiunga to Mlango wa Chano and is dominated by blocks of *Rhizophora* almost pure stands.
Northern central swamps	Extends from Mlango wa Chano to the mouth of Dodori Creek, including Uvondo and Ndau islands. The block is highly stocked with *Ceriops* and *Rhizophora.*
Mongoni and Dodori Creek swamps	Comprise the mangroves found on the banks of Mongoni & Dodori Creek and Manda Bay. It is stocked with *Ceriops* almost pure stands.
Southern swamps	Include mangroves of Mkunumbi and Kimbo Creeks. It is the largest of the five mangrove management blocks of Lamu County.
Pate Island swamps	Include mangroves surrounding Pate Island, Shindabwe, Kizingitini and Chongoni.

Source: Roberts and Ruara (1967) and GoK (2017)

Appendix 5: Indices applied in assessing mangrove forests in Lamu County

Index	Formulae	Reference
Normalized Difference Vegetation Index (NDVI)	$\frac{(NIR - RED)}{(NIR + RED)}$	(Gamon *et al.,* 1995)
Normalized Difference Water Index	$\frac{(Green - NIR)}{(Green + NIR)}$	(Gao, 1996)
Combined Mangrove Recognition Index (CMRI)	(NDVI - NDWI)	(Gupta *et al.,* 2018)

$NDMI = (NIR - SWIR)/(NIR + SWIR)$

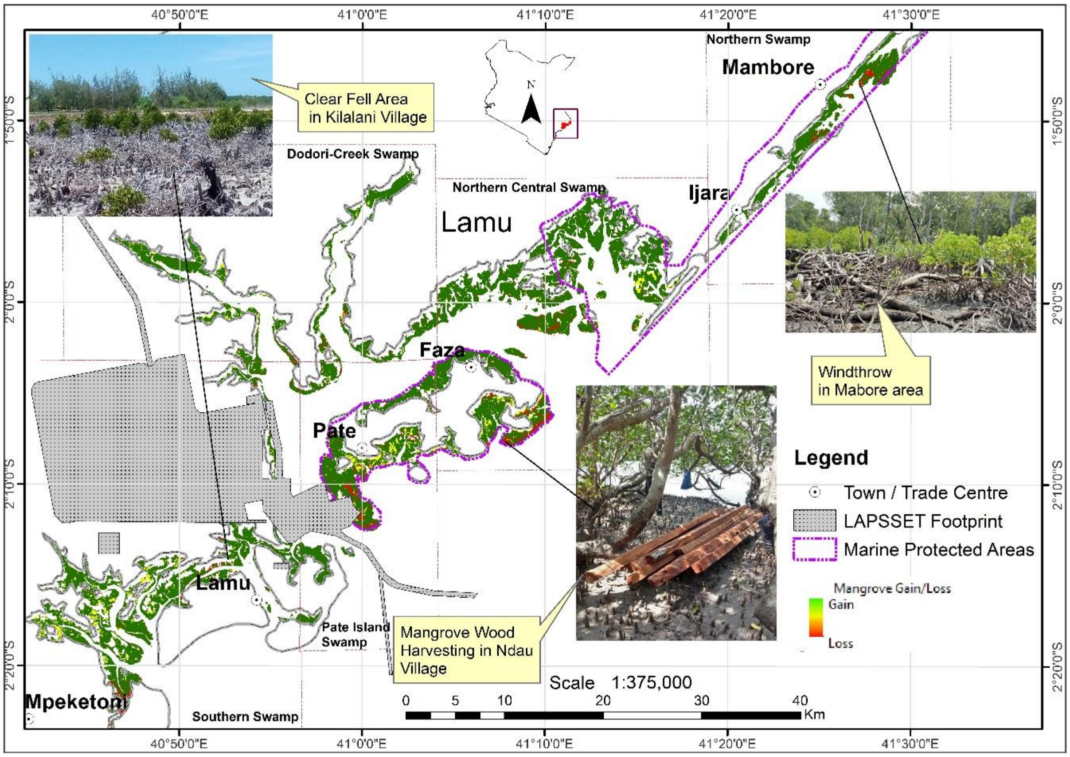
Appendix 6: A summary of the achieved classification accuracy in present study

Mangrove formation	AvD	AvP	CD	MixCR	MixRS	RhD	RhP	Total	PA
AvD	9	2	1	1	2	0	1	16	**56.3**
AvP	1	5	2	1	0	0	0	9	**55.6**
CD	0	0	21	1	1	0	0	23	**91.3**
MixCR	0	0	1	5	2	1	2	11	**45.5**
MixRS	2	0	0	1	6	2	3	14	**42.9**
RhD	0	0	4	0	4	66	5	79	**83.5**
RhP	0	0		3	2	4	10	19	**52.6**
Totals	12	7	29	12	17	73	21	171	
CA (%)	**75.0**	**71.4**	**72.4**	**41.7**	**35.3**	**90.4**	**47.6**	**71.3**	

where AvD (*Avicennia* dominant), AvP (*Avicennia* almost pure stand), CD (*Ceriops* dominant), MixCR (Mixture of *Ceriops* & *Rhizophora*), MixRS (Mixture of *Rhizophora* & *Sonneratia*), RhD (*Rhizophora* dominant), RhP (*Rhizophora* almost pure stand), PA (Producer Accuracy), and CA (Classification Accuracy)

Overall Accuracy (%) = 71.3% **Kappa Statistic =** 0.6153 (indicating good agreement)

Appendix 7: Anthropogenic and natural disturbance to mangroves in Lamu County



Appendix 8: Sampling point map of the study area showing the spatial location of the plots sampled in this study