ASSESSMENT OF THE DISTRIBUTION, ABUNDANCE AND CARBON STOCKS IN SEAGRASS MEADOWS WITHIN EASTERN AND WESTERN CREEKS OF GAZI BAY, KENYA

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A Thesis Submitted to the Graduate School in Partial Fulfillment of the Requirements for the Award of the Degree of Master of Science in Environmental Science of Chuka University

CHUKA UNIVERSITY

SEPTEMBER 2019

DECLARATION AND RECOMMENDATIONS

Declaration

This thesis is my original work and has not been presented for an award of a Diploma or conferment of Degree in this or any other University.

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Recommendations

This thesis has been examined, passed and submitted with our approval as the University supervisors.

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DEDICATION

I dedicate this work to my parents, Mr. Agustinus Juma and Mrs. Rosemary Atieno who were always very proud of me.

ACKNOWLEDGEMENT

The journey to the development of this thesis has been momentous and challenging but its success is one of the greatest milestones in my academic life. I accord my utmost appreciation to the people who nurtured the dream and passion in me, supported and guided me to see the fulfillment of this academic endeavor. To a greater extent, I am grateful to Prof. Adiel Magana of Chuka University for his mentorship and supervision. The inspiration, in-depth scrutiny of my work and guidance he gave were of great significance. Special appreciation is also accorded to Dr. James Kairo of Kenya Marine and Fisheries Research Institute (KMFRI), Mombasa for his supervision and support during my field work and thesis development.

I sincerely thank the panel members of Chuka University at the various developmental stages of this thesis, particularly, the Director of Chuka University Board of Post Graduate Studies, Dr. Moses Muraya and Chair of Chuka University Department of Environmental Studies and Resource Development, Dr. Lemmy Muriuki for the necessary support and insights. Their suggestions and professional comments helped me to maintain focus leading to successful finishing of this work.

In a special way, I wish to acknowledge the support of the Field Assistants during data collection in Gazi. Special thanks is to KMFRI team including, Tom Kisiengo, Laitani Suleiman, Peter Musembi, Derrick Omollo, Charles Muthama, George Onduso and Julia Oluoch for assisting in the field sampling and laboratory sample processing. I am also grateful to the community members Hassan Omar and Hassan Amiri for offering boat transportation during the field work.

Finally, I am grateful to the National Commission for Science Technology and Innovation (NACOSTI) for the issuance of the permit (NACOSTI/P/18/52699/24288), to undertake this study. This M.Sc. thesis was funded by Punguza Hewa Kaa: Partners in Carbon Project, awarded to KMFRI by Newton Fund for Institutional Links Grant, to whom am very grateful.

ABSTRACT

Seagrass meadows are one of the most important blue carbon ecosystems within the seascape environment providing both ecological and economic benefits. They act as breeding and feeding grounds for fish and other organisms; perform carbon sequestration, nutrient cycling and other ecosystem services. Through their carbon capture and storage ability, seagrass can be incorporated in carbon offset schemes. However, they are highly degraded from both anthropogenic and natural factors. Carbon stocks assessment is required in understanding dynamics of seagrass meadows. In Gazi Bay, Kenya, carbon storage in the seagrass meadows has been determined in the open waters of the Bay. The present study aimed at contributing to carbon dynamics of Gazi bay by assessing the distribution, abundance and carbon storage in seagrass within the mangrove fringed creeks. The objectives included assessing distribution and abundance of seagrass in the creeks, determining above and below ground seagrass biomass and comparing sediment carbon stocks between vegetated and un-vegetated sites. Stratified random sampling strategy was used in collecting data within 80 square plots of 0.25m by 0.25m. Five species formation viz; *Thalassia hemprichii, Cymodocea rotundata, Cymodocea serrulata, Enhalus acoroides,* and *Thalassidendron ciliatum* were encountered as either single or mixed stand. A total of 480 samples were collected for sediment and biomass determination in the laboratory. The results showed a higher seagrass diversity in the Eastern creek, ($H = 1.71$), than Western creek, ($H = 1.67$). There was also a significant difference in the total biomass between the creeks ($t= -8.44$, df. = 53, $p < 0.0001$) and among species ($F = 14.6$, df = 79, $p < 0.0001$) with a mean of 7.25 ± 4.2 Mg C ha^{-1}, (range: 4.1 - 12.9 Mg C ha^{-1}). Sediment carbon varied between species within the 1.2 km^2 creeks area; with a range from 97.6 to 302.4 Mg C ha^{-1}, (mean: 183.4 ± 100.5 Mg C ha^{-1}). This is lower than 236 ± 24 Mg C ha^{-1}, reported in the open bay but within the global range. In all the species, vegetated areas showed significantly higher carbon values than the un-vegetated sites ($t = 12.02$ $p < 0.0001$). Based on this study, the total seagrass carbon stocks can be estimated at 21,118.8 Mg C. Using the IPCCC emission value of 7.9 tonnes of C ha^{-1}, values for organic soils for wetlands, conservation of seagrass in these two mangrove fringed creeks will prevent emission of 2,682.13 Mg of CO_2 equivalent yr^{-1} to the atmosphere. The avoided emission could be bundled with the existing offset scheme in the bay involving mangroves. Inclusion of seagrass in carbon offset scheme has a long term benefits of climate, community livelihood and biodiversity conservation.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF ABBREVIATIONS AND ACRONYMS

AGB:	Above Ground Biomass
ANOVA:	Analysis of Variance
BGB:	Below Ground Biomass
C:	Carbon
C.I:	Confidence Interval
C_{org}:	Organic Carbon
DOC:	Dissolved Organic Carbon
DW:	Dry Weight
GHG:	Green House Gas
IPCC:	Intergovernmental Panel for Climate Change
KMFRI:	Kenya Marine and Fisheries Research Institute
LOI:	Loss of Ignition
OC:	Organic Carbon
OM:	Organic Matter
PES:	Payment for Ecosystem Services
WIO:	Western Indian Ocean

CHAPTER ONE

INTRODUCTION

1.1 Background of the Study

Seagrass meadows are one of the most productive blue carbon ecosystems on Earth and in the coastal seascape environment providing a wide range of benefits to humans and the ecosystem (Duarte, 2002). These include regulating, supporting, provisioning and cultural services (Hejnowicz *et al.,* 2015). They have the ability to perform nutrient cycling (Constanza *et al.,* 1997 & Githaiga *et al.,* 2017), sediment stabilization, shoreline protection and commercial fisheries support. As habitats for fish, they provide breeding ground and feeding ground for fish hence increasing fish stocks. In addition, they perform exceptional carbon sequestration, and offer protection against climate change impacts (Beck *et al.,* 2001; Duarte & Prairie, 2005; Gedan *et al.,* 2009).

Seagrasses store organic carbon either in the sediments or in the adjacent ecosystems accounting to approximately 48 – 112 tC/ha/yr and an estimated total ocean carbon burial of 10 – 18% (Kennedy *et al.,* 2010). As ecosystem engineers, they stabilize sediments and alter the hydrodynamic environment by trapping the suspended sediments. This enhances carbon stocks and sequestration rates which are influenced by sediment composition and processes outside the system (or allochthonous processes) (Agawin & Duarte, 2002). Seagrasses similarly regulate the nutrient cycle and water quality thereby enhancing trophic transfers and production of organic carbon production to the adjacent blue carbon habitats (Duarte *et al.,* 2008).

There has been a rapid decline in seagrass beds due a combination of natural and anthropogenic activities. Some of the natural factors include diseases, storm surges and strong waves, sedimentation and herbivory (Cabaco *et al.,* 2008 & Githaiga *et al.,* 2017). Increased anthropogenic factors have globally threatened seagrass meadows, with a loss estimated at 340,000 to 980,000 ha/year (Pendleton *et al*., 2012). Activities such as boat mooring and anchoring, introduction of invasive species and unsustainable fishing methods such as trawling greatly affect seagrass meadows, leading to their degradation (Orth *et al.,* 2006 & Waycott *et al.,* 2009). When degraded, they can become potential sources of carbon emission thereby leading to further global warming.

There is, therefore, a concern that the roles seagrasses have played in the marine ecosystems may diminish if proper conservation measures are not implemented. Finding new and more effective approaches to seagrass restoration is a conservation and climate mitigation priority. This is why it is important to restore and manage these blue carbon ecosystems. With proper conservation, there will be increased storage of C_{org} in the below- ground biomass as a result of the high root production and the rapid turnover (Duarte *et al.,* 1998).

Globally, among the blue carbon pools including seagrass, mangroves and intertidal salt marshes, seagrass is the most extensive one, yet it has received less scientific attention worldwide (Nelleman *et al.*, 2009; Githaiga *et al.,* 2016). Many of the studies have been done in the Caribbean, Western Europe, Mediterranean and American coasts indicating that seagrass meadows store about 19.9 Pg of organic carbon (Fourqurean *et al.,* 2012). They are net sinks of CO_2 within the biosphere storing about twice more carbon per unit area than the tropical and temperate forests (Duarte *et al.,* 2005).

In Africa, recent studies have shown that the mean above-ground and below-ground C_{org} is 174.4 and 474.6 g DW m^{-2} respectively while the total biomass is 514 g DW m^{-2} with the world biomass values ranging between 461 and 738 g DW m^{-2} (Githaiga *et al.,* 2016). However, there is still a great paucity regarding seagrass studies in Africa (Fourqurean *et al.,* 2012; Harcourt *et al.,* 2018). The scarcity exists in mapping of the seagrass beds, management approaches and assessments involving sediment carbon despite this pool contributing to over 97% of total C_{org} (Mazarrasa *et al.,* 2018). In particular, large gaps exists in West Africa where very little has been done (Githaiga *et al.,* 2016). This gap means that the potential of seagrass as a carbon sink in Africa could be underestimated. Limited knowledge on these critical ecosystems may also deter conservation initiatives which could ensure their resilience. Bridging the knowledge gap in seagrass alongside other critical ecosystems through research and adopting sound management approaches should therefore be prioritized alongside other management initiatives in Africa. This will help in sustainable management of resilient and critical ecosystems hence contributing to the achievement of the sustainable blue economy agenda.

Studies in East Coast of Africa have indicated that *Thalassia hemprichii* and *Enhalus acoroides* are dominant in these areas and have a high growth pattern (Lymo *et al*., 2016). The biodiversity of seagrass, their biomass and abundance increase towards the tropics and harbor a variety of species including 18 species of algal epiphytes and over 50 species of algae. Other organisms harbored include gastropods, lobster, sea cucumbers bivalves and several species of crabs (Ochieng & Erftemeijer, 2003; Alcoverro & Mariani, 2004). In terms of seagrass biomass, the mean values for East African Coast are highest at 587.1 and 256.8 g DW m^{-2} for below ground and above ground biomass respectively when compared to other African regions including South Africa and the South Mediterranean regions (Lymo *et al.*, 2006; Githaiga *et al.*, 2016).

In Kenya, seagrass research has focused mainly on species composition, distribution and community ecology (Bouillon *et al.*, 2007), and productivity (Githaiga *et al*., 2016). Studies along the Kenyan coastline have indicated the occurrence of seagrass meadows along the back reef lagoons which exist between the cliffs or beaches and adjacent fringing reefs. They also inhabit channels of creeks that run through the mangroves. These places include *Fuzi*, *Gazi*, *Tudor Mtwapa*, *Kilifi* and *Mida* (Ochieng & Erftemeijer, 2003). Recent seagrass studies have estimated the extent of seagrass along the Kenyan coastline to cover 317.1 km^2 with a decline estimated at 0.85% yr^{-1}. Within the past thirty years, about 12 ha of seagrass cover have been lost per year (Harcout *et al.*, 2018). This translates to emission of about 35 tonnes $ha^{-1}yr^{-1}$ of CO_2 based on tier 1 IPCC emission factor of 7.9 tonnes of C ha^{-1}, values for organic soils for wetlands (IPCC, 2014), hence calling for the need to conserve these critical ecosystems.

At Gazi bay, where most of the blue carbon research has been carried out in Kenya, seagrass carbon stocks have been assessed within the bay. The C_{org} stocks have been estimated at 168,642 Mg C in the open waters with high biomass recorded in the vegetated areas (Githaiga *et al*., 2017). This carbon in seagrass meadows comprise of allocthonous materials from the surrounding mangroves and a mixture of re-suspended organic matter that are derived within the meadow (Hemming *et al.*, 1994). The seagrass meadows in this bay also significantly support the coastal communities through small scale fisheries in the nearshore shallow waters (Musembi *et al.*, 2019). There are four dominant species of seagrasses in Gazi Bay including *Thalassia hemprichii*, *Thalassodendron ciliatum*, *Enhalus acoroides* and *Syringodium isoetifolium* (Githaiga *et al*., 2016). Seagrass beds continue to degrade due to poor fishing activities, sedimentation and herbivory by sea urchins. However, no study had been done to determine carbon stock and sequestration rates within the mangrove fringed creeks. This could lead to the underestimation of the total carbon stored and sequestered in the entire bay. A deeper understanding of the eastern and western creeks in terms of the carbon stock would add onto the known figures of the stock in the seagrass meadows of the open bay waters that ranges between 160.7 and 233.8 Mg C ha^{-1}.

1.2 Statement of the Problem

Seagrass meadows are important blue carbon ecosystems due to their economic and ecological significance. They help in nutrient cycling, carbon storage, shore line protection and serve as breeding and feeding ground for fish. As such, they are significant in climate regulation, community livelihood improvement and biodiversity conservation. However, they are under severe threats from various anthropogenic activities such as use of seine and dag nets by the artisanal fishermen and reclamation of coastlines. Natural factors including storm surges, herbivory, sedimentation and strong waves also threaten seagrass growth. To enhance conservation and restoration of seagrass meadows, proper assessment is required to determine the amount of carbon stored in the entire system. Past studies in Gazi have determined carbon stocks of seagrass meadows in the open waters of the bay. However, there is paucity of information on the carbon storage and the sequestration rate within the western and eastern creeks of the bay. This information is important in determining the carbon budget of the entire blue carbon ecosystems in Gazi Bay. The study therefore focused on abundance, distribution and carbon stocks within the mangrove fringed creeks of the bay. This would provide baseline information for bundling seagrass ecosystems into Mikoko Pamoja Carbon Mangrove Offsetting scheme, thereby enhancing increased conservation and management.

1.3 Objectives

1.3.1 Broad Objective

The broad objective was to assess the distribution, abundance and amount of carbon stored in seagrass meadows of Eastern and Western creeks of Gazi Bay, Kenya for sustainable conservation and management of seagrass ecosystems.

1.3.2 Specific Objectives

The specific objectives were to:

i. Determine the distribution and abundance of seagrass species within the meadows of Eastern and Western creeks of Gazi Bay, Kenya.
ii. Determine the vegetation carbon stocks of seagrass meadows in the above ground and below ground components of Eastern and Western creeks of Gazi Bay, Kenya.
iii. Compare the sediment organic carbon stocks between seagrass vegetated areas and the un-vegetated areas in Eastern and Western creeks of Gazi Bay, Kenya.

1.4 Research Hypotheses

H_{01}. There is no significant variation in the distribution and abundance of seagrass species within Eastern and Western creeks of Gazi Bay, Kenya.

H_{02}. There is no significant difference in above-ground and below-ground vegetation carbon stocks in seagrass meadows of Eastern and Western creeks of Gazi Bay, Kenya.

H_{03}. There is no significant difference in the sediment organic carbon stocks between seagrass and un-vegetated areas of Eastern and Western creeks of Gazi Bay.

1.5 Significance of the Study

The study is important since it forms an ecological and socio-economic research with the aim of understanding the carbon dynamics within the creeks. Underpinned in the KMFRI/Punguza Project it will facilitate the bundling of the seagrass ecosystem services with those of mangroves. The current carbon offset project, MIKOKO

PAMOJA, concerns restoration and projection of mangroves through sale carbon credits. Income generated amounting to US$ 13,000 per year is used to finance community development projects in water and sanitation, education, health and environmental conservation. The information from the current study will help in expanding MIKOKO PAMOJA to include mangrove ecosystem. Prior to the inclusion of seagrass in carbon offset program, there is need to carry carbon baseline of the ecosystem, conditions of the seagrass beds and trends; as well as determining hotpot areas requiring interventions. Past studies on seagrass carbon in Gazi has focused on open day (Githaiga *et al.*, 2017). The current study builds on previous studies by assessing carbon stocks in the mangrove fringed creeks of the bay. Information on distribution, abundance and carbon stocks of seagrass beds will facilitate expansion of carbon offset to include seagrass beds. The information from the study will also add to the literature on seagrass thereby benefiting in research and academia.

1.6 Scope of the Study

The study focused on seagrass distribution, abundance and carbon stocks within Eastern and Western creeks of Gazi Bay, Kenya. A total of 80 quadrats were established and 480 samples collected for above ground biomass, below ground biomass and sediment carbon determination. The study was done between the months of May and September 2018.

1.7 Limitation of the Study

The seagrass meadows are easily accessible during the low spring tides when the seagrass plants are exposed. The field work activity was therefore limited to the low spring tides, thereby decreasing sampling time. Natural processes such as strong waves in the nearshore also reduced water clarity. This limited the data collection process to times of the day when the water is disturbed. .

1.8 Operational Definition of Terms

Allochthonous Carbon: Carbon produced in one location or ecosystem and deposited in another location. The associated carbon and sediments are transported from the neighboring terrestrial and offshore ecosystems such as the mangroves into the seagrass beds of Eastern and Western creeks.

Autochthonous Carbon: Carbon produced and deposited in the same ecosystem or place. It results from the uptake of carbon dioxide by the seagrass plants from the atmosphere or ocean, which gets converted for use by the plant tissues, and then gets decomposed into the surrounding soils. This is the carbon produced and deposited in the seagrass beds of the study area.

Bay: A recessed water body that is about three quarters surrounded by land and connects to another bay, a lake or a sea. In our study, Gazi Bay connects to the Indian Ocean and from which the two creeks extend.

Blue Carbon: a term used to denote carbon stored by vegetated coastal ecosystems especially tidal salt marshes, mangroves and seagrass meadows within the living biomass below ground (roots and rhizomes and necromus), living biomass above ground (shoot, branches and leaves) and non-living biomass.

Blue Carbon Ecosystem: Are coastal and marine ecosystems including mangroves, seagrasses and tidal marshes that store blue carbon

Carbon Inventory: Is the accounting of carbon losses and gains from the ocean or atmosphere by sea grass, mangroves and other vegetation over a given time frame.

Carbon Stocks: Total amount of organic carbon stored in mangroves, seagrass or tidal salt marsh ecosystems of a known size. In the current study, it is the amount of organic carbon in sea grass of the eastern and western creeks.

Carbon Sequestration: Is the long-term storage of carbon in soils, plants, ocean and geologic formations. It involves the long term storage of carbon in the different carbon pools within the seagrass meadows of the creeks.

Carbon Pools: Includes the above-ground biomass (the living vegetation both herbaceous and woody plants, stumps, barks, seeds, branches, and stems), below-ground biomass, dead wood, litter and the soil organic matter.

Creek: A small bay or inlet, narrower and extending further into the land than a recess in the shore of a river or the sea. In the present study, eastern and western creeks extend further in to the land from the open waters of Gazi bay.

Inorganic Soil Carbon: Refers to the carbon component of carbonates and occurs in coastal soils in the form of pieces of corals or shells.

Payment for Ecosystem Services: This is a concept whereby the user or beneficiary of an ecosystem service contributes directly or indirectly to individuals, communities or organizations that conserve or manage a resource to keep providing the ecological services. In this regard, they are the benefits that will accrue to the community for conserving seagrass in the bay.

Seagrass Abundance: This refers to the quantity of the various seagrass plant species in a given area.

Seagrass Distribution: Is the extent or spatial coverage of seagrass species in an area.

Seagrass Meadows: Seagrass are flowering underwater plants in the order *Alismatales* and occur in meadows in saline environments providing food, shelter and nurseries for various ecologically, recreationally and commercially important species including seahorse, dugongs, crustaceans, manatees, sea turtles and fish.

Soil Organic Carbon: Is the carbon component of the organic matter. The quantity of soil organic carbon depends on climate, soil texture, vegetation and the current and historical land use or management of the creeks.

Tidal Range: Is the variation in height between the lowest and the highest tidal water marks.

CHAPTER TWO

LITERATURE REVIEW

2.1 Overview of Seagrass Ecology, Species Composition and Adaptations

Seagrasses comprise of a taxonomic group of polyphyletic marine angiosperms adapted to a life in fully submerged seawaters by the development of a specific morphology (Green & Short, 2003). They have an anchorage systems made up of roots and rhizomes, air lacunae for oxygen supply in the roots and flowers with hydrophilous pollination (Bandeira & Bjork, 2001). The strong structures of the roots also enable these plants to withstand strong waves and ocean current in cases of storm surges. They also have the ability to take up nutrients by both the leaves and the roots (Heminga, 1998). Seagrasses inhabit shallow zones of photic environments forming the seagrass meadows and beds that cover about 0.2% of the ocean cover (Green & Short, 2003; Duarte *et al.*, 2005).

The colonization of seagrass involves a slow process in which the species require years to centuries to form the meadows (Duarte *et al.*, 2013). The colonies are formed through initiation of patches, formed by seeds or rooted fragments and subsequent growth of patches through clonal rhizome growth (a process which is exponential and self-accelerating) (Duarte & Sand-Jansen, 1990). They produce filamentous pollen grains which get transported through water current. Through planting, patch formation and survival can be catalyzed leading to seagrass colonization and restoration of seagrass meadows (McGlathery *et al.,* 2012; Mazarrasa *et al.,* 2018).

In many cases, most populations of seagrasses rely largely on asexual production and are clonal to maintain population (Short *et al.,* 2007; Waycott *et al.,* 2006). In other cases, some seagrass plants vary their strategies of reproduction depending on the surrounding conditions or produce many propagules thereby enhancing sexual reproduction. The phylogenetic diversity is therefore relatively limited leading to a small range in the strategies of life history (Inglis & Waycott, 2001). Asexual reproduction is exhibited in all the seagrass species as they produce the ramets (modular units) through the growth of rhizomes horizontally. These units may be identical to the genet (parent plant) but are usually physiologically independent (Kuo & Kirkman, 1987).

Sexual reproduction is also encountered in some species of seagrass plants. They produce viviparous seedlings or fruits in which some seeds may be long lived thereby forming seedbanks (Inglis & Waycott, 2001; Short *et al.,* 2007). Seed dispersal in seagrass is limited since they get released at the plant stem below the surface of sediment and are poorly adapted (Orth *et al.,* 1994). Through the combined strategies of reproduction, including seed production and clonal growth enhance evolutionary adaptive advantage to these plants that exhibit unpredictable, highly disturbed and changing environment (Rasheed, 2004).

Seagrass have high tolerance ability and are therefore able to thrive in different climate ranges. They can withstand high salinity range from a limit of 42 parts per thousand (ppt) to fresh water. In many cases, dense and healthy seagrass inhabit saline waters between 10 and 30 ppt. However, seagrass have been observed to tolerate up to 50 ppt in Florida bay - Laguna Madre. They can also thrive in temperatures as low as - 6˚C to high readings of 40.5 ˚C (Phillips & Menez, 1988; United States Geological Survey, 2011).

2.1.1 Ecosystem Services of Seagrass

Seagrass occupy only about 0.2% of the coastal oceans yet they play a great role in providing different ecosystem services with a total estimated value of $ 19,004 ha/yr (Duarte, 2002; Duffy, 2006; Fourqurean *et al.,* 2012). These services include cultural, provisioning, regulating and supporting. More established seagrass meadows may have higher abundance of faunal assemblages than a less established meadows (Duarte *et al.,* 2008; Cullen – Unsworth and Unsworth, 2013; Hejnowicz *et al.,* 2015). Similarly, carbon storage may differ among species due to the morphological difference, for example, broadleaved seagrass species such as *Enhalus spp* may have more biomass than other stocks such as *Halodule spp* that are narrow leaved (Mazarrasa *et al.,* 2018).

In provisioning services, seagrass meadows provide habitat for feeding, hiding and breeding grounds for fish and other marine fauna. Seagrass meadows act as sources of food supply for the mega herbivores including sea urchins, manatees, herbivorous fish, water birds, sea turtles and the dugongs (Barbier *et al*., 2011). In fisheries, seagrass support the productivity of prawns valued at US$1,150/ha/yr (Kirsch *et al.,* 2002). They also support commercial fisheries valued at U.S$ 47.8 million/yr in Florida (Green & Short, 2003) and fish productivity valued at $ 103.74 million/yr (McArthur *et al.,* 2006). Additionally, they store organic materials within their systems which when transported to the adjacent ecosystems, support different terrestrial and marine consumers (Unsworth & Cullen-Unsworth, 2014).

Seagrasses also play regulatory roles. They mitigate climate change by capturing and storing huge stocks of carbon in both above and below ground components (Oreska *et al.,* 2017). Standing biomass in seagrass ecosystems has been estimated at 76 - 151 Tg C (Fourqurean *et al.,* 2012). More than 50% of this carbon is derived from the terrestrial and the adjacent ecosystems. Additionally, more than 97% of the carbon is stored in the sediment (Bouillon *et al.,* 2007; Githaiga *et al.,* 2017). They therefore constitute important carbon sinks globally and their degradation will exacerbate global warming. Seagrass also filter particles and other wastes from the water column thereby enhancing water quality. Below ground components including rhizomes, roots and necromas can also store C_{org} making it stable for millennia. However, carbon storage below ground depends on the interplay of various biotic and abiotic factors (Duarte *et al.,* 2013). The economic valuation of seagrass in capture and storage of carbon is estimated at U.S. $ 394/ha/yr (Pendleton *et al.,* 2012; Dewsbury *et al.,* 2016).

Seagrass as critical habitats also offer supporting services by facilitating nutrient cycling to the global economy valued at $ 3.8 trillion/year (Dewsbury *et al*., 2016). They perform nutrient cycling via the water columns, and store nutrients in seagrass detritus, biomass and sediment. Seagrass also modify abiotic environment through their leaves hence act as ecosystem engineers. The seagrass leaves increase the pH through absorption of CO_2 creating a conducive habitat to carbon associated organisms, improve light conditions by trapping suspended nutrients and sediments and reduce hydrodynamic stress through attenuation of waves and currents (Terrados & Duarte, 2000; van der Heide *et al*., 2007; Hendricks *et al*., 2014; Lymo, 2016). As such, they contribute to marine resilience by reducing the microbial contamination of sea water hence contributing to human health (Lamb *et al.,* 2017).

In addition to the other ecosystem services, seagrass meadows provide a variety of social, cultural and economic benefits to the coastal society. Seagrass are breeding and feeding grounds for fish, hence promoting artisanal and commercial fishing thereby promoting food security. The role played by seagrass in food security has been estimated at US$ 3500 ha/yr (Waycott *et al.,* 2009). In Africa, gleaning of shellfish among the artisanal fishers and invertebrate harvesting have improved the rural livelihoods through income generation ranging between US $ 8.51 and US $ 17.01 per catch (Nordlund *et al.,* 2011; Hejnowicz *et al.,* 2015). Other benefits include use of seagrasses as bio-fertilizers, medicine, baits and substrate for seaweed farming (Githaiga *et al.,* 2017). Since 16th century, litter from *Posidonia* has been used in the filling of beddings. Cottars also used seagrass in Scotland, Orkney in the 18th century in thatching the flagstone roofs as a substitute for the straws in Orcadian houses (Willis, 1983; Terrados & Bodrum, 2004). In coastal Kenya, seagrass has been used for aesthetic and religious significance to the communities (Unsworth & Cullen, 2010), hence improving livelihoods.

2.1.2 Threats to Seagrass

Seagrass meadows continue to decline due to natural and anthropogenic factors, at the rate of 1.5% per year threatening some of the species to extinction (Duarte 2002; Pendleton *et al*., 2012; Roca *et al.,* 2016). Increasing anthropogenic activities as a result of increased human settlement along the coastline subject the seagrass vegetation to disturbance. These activities include: boat anchoring and mooring, fishing and aquaculture, unsustainable development along the coastline, introduction of invasive species, reclamation of coastline and dredging. Additionally, poor fishing methods such as use of seine nets and trawling may disturb the seagrass ecosystems through removal (Waycott *et al.,* 2009; Githaiga *et al.,* 2016). Agricultural activities and effluent discharge upstream cause sedimentation and nutrient over-enrichment in marine waters increasing turbidity and lowering water clarity. This reduces the productivity of seagrass leading to degradation of the ecosystem (Larkum, *et al*., 2006).

Natural factors that threaten seagrass ecosystems include storm surges, herbivory, strong waves, diseases and sedimentation (Cabaco *et al.,* 2008; Githaiga *et al.,* 2017).

Surface runoff and nutrient fluxing may also attenuate light leading to invasion by epiphytes and macro-algae which may out-compete seagrass for nutrients and light. When light is unavailable beyond the minimum limits, the meadows and the associated biological community may be affected (Cardoso *et al*., 2004).

The anthropogenic and natural disturbances threatening seagrass growth can affect their ability to capture and sequester carbon in a number of ways. One is that severely disturbed seagrass can die off leading to loss of carbon which is stored in the material if the C is not buried in the sediments. Two, the suspended allochthonous carbon which adds to the below ground biomass is reduced when the filtering capacity of seagrass is reduced (Burden *et al.,* 2013). Three, sedimentary carbon buried in seagrass meadows can be released through erosion, microbial mineralization and leaching (Macreadie *et al*., 2014; Dahl *et al.,* 2016). As a result, other greenhouse gases such as nitrous oxide and methane can be released. Finally, the total amount of inorganic carbon which is fixed by seagrass may be reduced due to activities such as shading or grazing which reduces the photosynthetic capacity (Dahl *et al.,* 2016).

2.1.3 Management Practices

The occurrence of seagrass meadows at the interface of land and sea, pose a great challenge to their management (Rudd & Lawton, 2013). Their pressures and threats sometimes differ with those in terrestrial environments due to their "no-man's land," nature hence require sustainable and multi-sectoral responses that are conceived and implemented consistently (Herr *et al.,* 2017). Drivers of degradation may originate from either landside or seaside environment thereby heavily impacting on the seagrass meadows and the socio-economic vitality. Yet, the management regimes and the policy mandates may be isolated so that the economic interests such as fishing and aquaculture production are prioritized leading to high degradation rates in these *terra – nullius* environment (Herr *et al.,* 2017). This is still the case for Africa where large knowledge gap still exists on seagrass ecosystems (Githaiga *et al.,* 2016). Initiating new management mechanisms for global and the African seagrass extent will therefore enhance the resilience for improved functionality. Some of the mechanisms may include restoration, monitoring, educational efforts, conservation and management practices (Cullen-Unsworth & Unsworth, 2013).

Monitoring is key in any seagrass conservation program. Baseline geographic parameters can be established followed by consistent data set that aid in decision making from time to time. Other monitoring techniques include identification of the seagrass species, measurements of sediment and water quality, taking aerial photos and ground verifications. These monitoring activities are best undertaken by the local fisher community who best understand the dynamism of the ecosystems and the history of the programs. However, the programs can be coordinated by larger organizations, such as research institutions (USGS, 2011).

Management practices implemented in the seagrass ecosystems can similarly help in their protection and restoration. Development of comprehensive management policies and plans facilitate framework which enable the decision-making process of various stakeholders to be easy (Waycott *et al.,* 2009). The management plans are implemented at the ground level despite being directed at the national or states level thereby enabling managers and scientists to use conservation mechanism that are specific to a given region. Certain policies and regulations including restrictions on boat-dredging reduce propeller scarring thereby helping in improving quality of seagrass habitats (USGS, 2011). These regulations also control coastal development practices, and non-point and point pollution hence reducing the anthropogenic threats on seagrass. Enforcement of policies and implementation of the set guidelines reduce both human and natural pressure on seagrass and other critical ecosystems hence preventing their degradation.

2.2 Seagrass Species Distribution and Abundance

2.2.1 Seagrass Abundance

Seagrass species occur either in mixed stands of multi species or in monospecific stands with their coverage extending from the intertidal to the subtidal areas in the sandy and rocky substrates, along estuaries or on mud flats (Githaiga *et al.*, 2017). Their distribution and abundance are influenced by biophysical factors such as water currents, day length, epiphytes, wave action, temperature, light availability, salinity and substrate type and depth, which regulate their morphology and physiological activity (Hemminga & Duarte, 2000; Green & Short, 2003).

Colonizers and other small species of seagrass also facilitate large carbon deposition mainly formed by allochthonous C_{org}. These meadows are usually found in depositional and sheltered bays where the mud content of the soil provides a good proxy for C_{org} deposition (Serrano *et al.,* 2016). Due to excess bioturbation or overgrazing, meadows that have suffered disturbance in their trophic webs usually have lower carbon stocks and sequestration capacity due to lack of top-down control (Macreadie *et al.,* 2017).

Additionally, human factors upstream or along the coastline such as effluent discharge and agricultural activities facilitate sediment and nutrient loading thereby affecting seagrass health status and abundance. Through the processes including eutrophication, habitat fragmentation and alteration of the sediment C_{org} deposits human threats indirectly cause detrimental conditions to the survival of seagrass or directly affect the seagrass abundance through mechanical means (Lymo, 2016; Ricart *et al.*, 2017; Macreadie *et al.,* 2017).

2.2.2. Seagrass Distribution

Distribution of seagrass varies within short periods of time and within a small scale due to their continuous and timely response to the changes in their immediate environment. These changes in the environment majorly occur in areas of high biodiversity, bioregional boarders and places with more human influence (Short *et al.,* 2007). In all the seagrass species, distribution is, therefore, a combination of clonal growth and sexual reproduction in plants due to environmental limitations and dispersal (Spalding *et al.,* 2003). In a broader perspective, the geographic range and extent of seagrass are limited to either tropical or temperate regions. Globally, seagrass meadows are found in six bioregions based on their species distribution, assemblages and the temperate and tropical influences. The tropical bioregion contains the Tropical Indo-Pacific and the Tropical Atlantic regions.

The regions covered under Tropical Indo-pacific include South Asia, East Africa, and tropical Australia extending to the eastern Pacific. This bioregion is known to have the highest and the largest diversity. It contains 24 tropical seagrass species that are found predominantly in intertidal, subtidal deep waters, and on reef flats where they are majorly grazed by the mega herbivores such as dugongs, manatees and sea turtles (Short *et al.,* 2001; Short *et al.,* 2007). Some of the species found in this region include *Halodule uninervis, Halodule pinifolia, Halophila beccarii, Halophila decipiens, Halodule wrightii, Halophila ovalis, Halophila minor, Halophila carpricorni, Halophila ovata, Enhalus acoroides, Cymodocea rotundata, Cymodocea angustata, Syringodium isoetifolium, Zostera japonica, Zostera muelleri* and *Zostera capensis* among others.

On the other hand, the Tropical Atlantic region covers areas such as the Bahamas, Bermuda, the Gulf of Mexico, Caribbean Sea and both of the Atlantic's tropical coasts. In this bioregion, higher abundance of tropical seagrasses comprising 10 species defines its distribution. These seagrass plants grow on shallow banks and back reefs in clean water. Some of the species found in this region include; *Halophila engelmanni*, *Halophila decipiens*, *Halophila stipulacea Halophila baillonni, Halophila johnsonii*, *Halodule beaudettei*, *Thalassia testudinum, Syringodium filiforme,* and *Ruppia maritima* (Short *et al.*, 2001). The dominant species in this region include *Halodule. wrightii, Syringodium filiforme* and *Thalassia testudinum.* In many cases they occur as mono-species but are also found in mixed stands or sequentially occur in ecological successions in the Caribbean Sea and the Gulf of Mexico (Wabnitz *et al.,* 2008). Mega herbivores including sea turtles and manatees feed on seagrasses of this region. In areas such as Laguna Madre, environmental conditions (higher temperatures) in the lagoons and bays restricts seagrass diversity and abundance hence is only dominated by *Ruppia maritima* and *Halodule wrightii* that can withstand hypersaline conditions and higher temperatures (Onuf *et al.,* 2003).

The temperate bioregion has Temperate Southern Ocean, Temperate North Atlantic, Mediterranean region and the Temperate North Pacific region (Green & Short, 2003; Short *et al.,* 2007). In particular, the Temperate Southern Oceans region includes South America, New Zealand, South Africa and the temperate Australia. About 18 seagrass species have been identified in this bioregion having low to high diversity and known to grow under extreme conditions. Some of the common species found in this bioregion include; *Amphibolis griffithii, Amphibolis antarctica, Posidonia angustifolia, Posidonia sinuosa, Posidonia ostenfeldii, Posidonia australis, Ruppia* *tuberosa, Zostera tasmanica, Zostera. capensis, Thalassodendron pachyrhizum* and *Thalassodendron ciliatum* among others (Short *et al.*, 2007).

In the Temperate North Pacific bioregion, 15 species of temperate seagrasses are encountered occurring in high diversity along the coastal surf zones, estuaries and lagoons. Some of the species that are specific to this region include *Phyllospadix torreyi, Phyllospadix serrulatus, Phyllospadix iwatensis, Phyllospadix scouleri, Phyllospadix japonicus, Zostera japonica, Zostera asiatica, Ruppia maritima, Halophila euphlebia* and *Halophila decipiens* among others. This bio region covers Mexico and Korea to Baja. *Zostera marina* is dominant in the northern parts occurring near the Pacific Rim in areas such as China, Korea and Japan and proceeds to the Gulf of California and Bering Sea. The Zostera species found in this region have deeper to limited depths as they are found between 10 m to 20 m. Studies have revealed the existence of tropical species (*Halophila euphlebia* and *Halophila ovalis*) in Japan, a place where they are least expected as the conditions majorly support temperate species (Nakaoka *et al.*, 2003; Uchimura *et al.*, 2008). In Baja, *H. decipiens* overlaps with *Zostera marina* with perches of *Halodule wrightii* also found.

The Temperate North Atlantic however has low diversity of seagrass plants that are majorly found in lagoons and estuaries. Five species of temperate seagrasses have been found here, covering USA, Portugal and North Carolina. The species found include *Halodule wrightii, Cymodocea nodosa, Zostera marina, Ruppia maritima,* and *Zostera noltii* (Short *et al.*, 2001; Short *et al.*, 2007). In this bioregion, most of the intertidal areas are covered by *Z. marina* occurring in the bottom soft sediments of the clear waters up to depth of 12 m. The size of the seagrass plants increase with the latitude range with the vegetative seagrasses in northern U.S.A growing to 3 m while those in North Carolina record 0.2 m in height (Moore & Short, 2006). Similarly, some of the species form mixed stands with other tropical species, for example, *Z. marina* in this region intermixes with *Halodule wrightii* while *Cymodocea nodosa* is found in mixed stands with other temperate species in Portugal.

The Mediterranean as a bioregion covers northwest of Africa, the Black sea, Aral Sea, Caspian Sea, and the Mediterranean Sea. The seagrasses in this region have moderate diversity growing in deep and wide meadows of clear waters. Nine species comprising of both tropical and temperate species have been found in this area. These include *Halophila stipulacea, Halophila decipiens, Halodule wrightii, Ruppia maritima Ruppia cirrhosa, Cymodocea nodosa, Posidonia oceanica, Zostera noltii* and *Zostera marina*. This region is distinctly characterized by *Cymodocea nodosa* that grow to a lesser extent in clear waters to depth between 30 – 40 m and *Posidonia oceanica* that grow in deep waters up to 45 m forming deep rhizomes and roots and is long lived, growing for thousands of years (Procaccini *et al.,* 2003; Short *et al.,* 2007). The deepest occurring seagrass species (*Halophila stipulacea*), recorded globally has also been identified in this region below 50 m although its spread is scarce (Lipkin *et al.*, 2003). *Zostera marina* occupies the Adriatic Sea and the open coastal areas of Spain, Italy and France, and occurs in close association with *Zostera noltii* in the coastal lagoons in the western Mediterranean. In the Northern parts of Africa, *Zostera noltii* and *Cymodocea nodosa* occur from outside and inside of the sea while *Halophila decipiens* and *Halodule wrightii* occur from outside of Mediterranean Sea only.

In the African continent, seagrass beds are distributed in five regions with different species dominance and biomass productivity rates. The East African Coast have species including *Thalassodendron ciliatum, Thalassia hemprichii* and *Enhalus acoroides* dominating the region and has the highest mean below ground, above ground and total biomass at 587.1, 256.8 and 778.1 g DW m^{-2}, respectively (Githaiga, 2016). *Posidonia oceanica, Cymodocea nodosa* and *Thalassodendron ciliatum* dominate the South Mediterranean region with AGB & BGB of 155.6 and 299.3 g DW m^{-2} while the Western Indian Ocean Islands and South African region is dominated by *Zostera capensis* and *Cymodocea serrulata* and have mean biomass of 95.7 and 413.3 g DW m^{-2} respectively. Data from North West Africa show that the region is dominated by *Cymodocea nodosa* and *Syringodium isoetifolium* and have the least amount of biomass for AGB (61.06 g DW m^{-2}), BGB (145.2 g DW m^{-2}) and total biomass (159.4 g DW m^{-2}) (Fourqurean *et al.,* 2012; Githaiga *et al.,* 2016).

In the East African region, the coastline consists of back-reef lagoons with narrow channels connecting the sea with the lagoons during ebb tides. Areas such as Inhaca, Kilwa, Muhoro, Tanga, Gazi, Funzi and Mida among other places have the seagrasses, corals and mangroves occurring adjacent to one another and forms interrelated ecosystems (Ochieng & Erftemeijer, 2003). Some of the species that have been found for this region include *H. stipulacea, Z. capensis, Cymodocea. rotundata* and *Cymodocea. serrulata* among others. Despite *Halodule pinifolia, Halophila beccarii* and *Halophila ovata* being reported in East Africa, their observation may need further confirmation since it could be a misidentification. The seagrasses in this region are of significance since they support a variety of species including the endangered dugong, *Dugong dugong* and *Chelonia mydas* (green turtle) that graze on the grass.

Natural and human influence has led to recent changes in the distribution of species globally. These activities include movement of species from their endemic environment or alteration of such environment causing shifts in seagrass diversity and distribution. For example, the opening of Suez Canal has led to invasion of *Halophila stipulacea* into the Mediterranean from the Red Sea consequently leading to its distribution in the eastern parts of Mediterranean and further into the western regions of Sicily. This shift has extended to the Caribbean due to boat and ship movement leading to development of new perches (Ruiz & Ballantine, 2004). Similarly, the introduction of *Zostera japonica* into the western parts of North American coastline during the World War II has led to its spread throughout major coastal areas of USA and Canada overlapping with *Zostera marina* and *Ruppia maritima* in the intertidal regions.

Other factors such as competition among the species and from alien species also contribute to shift in diversity and distribution. For example, the extent of *Cymodocea nodosa* has expanded in the western parts of Mediterranean due to the competition it gets from *Posidonia oceanica. Cymodocea nodosa* usually extends further to harsh hydrodynamic conditions such as the estuaries and lagoons when *Posidonia. oceanica* is absent and competition is reduced. Existence of invasive species within the meadows, may lead to disappearance or regression of seagrasses within their habitat. The introduction of algal *Caulerpa taxifolia* in 1980s and its spread in the Mediterranean has resulted into localized regression of some of the dominant species' meadows including *Posidonia oceanica* and *Cymodocea nodosa* (Ceccherelli and Cinelli, 1997; Short *et al.*, 2007).

2.2.3 Seagrass Composition and Distribution in Kenya

Seagrass is found along the 600 km Kenyan coastline making the marine ecosystem rich and productive. The coastline is characterized by limestone cliffs, sand dunes and beaches, fringing coral reefs, mangroves forests and numerous sheltered creeks and bays (UNEP, 2001; Ochieng & Erftemeijer, 2003). With the large tidal amplitude, the intertidal zones between the coast and the fringing reefs are fairly extensive. Macro-algae and seagrasses predominantly determine the productivity of these ecosystems as they grow in the shallow depressions which retain water during the low tides. In many cases, extensive seagrass meadows are found in the back-reef lagoons which occur between cliffs or beaches and the adjacent fringing reefs.

There are about 12 species of seagrass in Kenya including *Halodule wrightii, Halodule uninervis, Syringodium isoetifolium, Enhalus acoroides, Halophila stipulacea, Halophila ovalis, Halophila minor, Thalassia hemprichii, Thalassodendron ciliatum, Cymodocea serrulata Cymodocea rotundata and Zostera capensis.* These species occur in both single and mixed stands. *Thalassodendron ciliatum* is usually a dominant species occurring mostly as mono-stand with higher biomass. The seagrass species also exhibit the common channels and creeks that run through the mangroves, thereby functioning as traps and also aid in reduction of flux rate of nutrient and particulate matter between the ocean and mangroves (Coppenjans *et al.,* 1992; Githaiga *et al.,* 2017).

Major seagrass areas along the Kenya coast can be found at *Funzi, Gazi, Kilifi, Chuda, Mida* and *Mtwapa*. In these areas, the coral reefs, mangroves and sea grass meadows occur adjacent to one another in the interrelated ecosystems. Other areas in Kenya where seagrass meadows are found include Kiunga Marine Reserve, Lamu, Diani – Chale lagoon and Shimoni (Gullstrom *et al.,* 2002; Ochieng & Erftemeijer, 2003). In 2016, seagrass coverage in Kenya was estimated at 308.4 ± 40.8 km^2 (Harcourt *et al.,* 2018).

Different organisms' life processes such as breeding and feeding have been supported within the seagrass meadows hence these habitats play critical biodiversity conservation and ecosystems functionality. The Lamu Archipelago in Kenya and Pemba – Zanzibar channels in Tanzania are home to the endangered dugongs while as they graze in these places (Muir *et al.,* 2003; van der Haide *et al.*, 2012). Fish species including the surgeonfish *(Acanthridae)* and rabbitfish (*Siganidae)* also directly feed on seagrasses while *Leptoscarus spp*. (parrotfishes) feed on the epiphytes that attach on seagrass plants. Other species such as the barracuda, groupers, snappers and grunts also feed on the have their juveniles feed on detritus derived from seagrasses while the adults consume infauna of the seagrass beds (Ochieng & Erftemeijer, 2003).

Table 1: Seagrass species distribution in Kenya (Coppejans *et al.*, 1990; Gullstrom *et al.,* 2002; Ochieng & Erftemeijer, 2003; McMahon & Waycott, 2006; Uku, 2006)

Seagrass species	Family	Where found in abundance
Thalassodendron ciliatum	*Cymodoceaceae*	Gazi, Chale lagoon, Kiunga Marine Reserve, Lamu, Mida
Thalassia hemprichii	*Hydrocharitaceae*	Gazi, Nyali, Kiunga,
Cymodocea rotundata	*Cymodoceaceae*	Nyali, Kiunga Marine Reserve, Gazi, Kilifi, Lamu
Cymodocea serrulata	*Cymodoceaceae*	Gazi, Kiunga, Lamu, Shimoni, Funzi
Halodule uninervis	*Cymodoceaceae*	Vanga, Gazi, Kiunga Marine Reserve, Lamu, Chuda
Halodule wrightii	*Cymodoceaceae*	Nyali, Gazi, Lamu,
Halophila ovalis	*Hydrocharitaceae*	Mida creek, Kiunga Marine Reserve, Gazi
Halophila minor	*Hydrocharitaceae*	Gazi, Kiunga Marine Reserve
Halophila stipulacea	*Hydrocharitaceae*	Kiunga Marine Reserve, Mtwapa, Gazi
Syringodium isoetifolium	*Cymodoceaceae*	Lamu, Kiunga Marine Reserva, Mida creek, Gazi bay
Enhalus acoroides	*Hydrocharitaceae*	Gazi bay, Kiunga Marine Reserve, Mida Creek,
Zostera capensis	*Zosteraceae*	Gazi bay, Kiunga Marine Park

The rate of loss of seagrass in Kenya has been increasing from 0.29% per year in 1986 to 0.85% yr^{-1} in 2000 and to 1.59% yr^{-1} in 2016. Although these rates are below the global average seagrass decline rate of 7% per year, necessary measures need to be put in place (Harcourt *et al.,* 2018). In the Kenyan coastline, the loss has been aggravated by anthropogenic activities such as boat mooring and natural factors such as herbivory by the sea urchins (Musembi *et al.*, 2019). Seagrass areas such as the Diani-Chale lagoon have experienced a net loss of more than 50% of *T. ciliatum* within the last two decades (Harcout *et al.,* 2018). Nutrient loading and destruction through boat dredging also contribute to the degradation along the coastline (Orth *et al.,* 2006). Upon reduction of the human activities on the seagrass growing areas, the plants have been observed to naturally recover due to stress reduction.

2.3 Above and Below Ground Carbon Stocks in Seagrass Meadows

Seagrass ecosystems are morphologically structured with their anatomy consisting of elaborate above ground and below ground systems. The blue carbon refers to the carbon which is stored in the salt marshes, seagrass and the mangroves. The carbon is stored in the living above ground biomass (stems, branches and leaves), the living below ground biomass (necromass, roots and rhizomes), and the dead non-living matter and in the sediments. The sequestration of carbon in the soil can occur over a long period of time making these ecosystems to become carbon sinks (Duarte *et al.,* 2005).

Due to the high accretion rates and the anaerobic condition which slows the rate of decomposition, the remineralization process is similarly lowered thereby facilitating the long term storage in these ecosystems. In the seagrass ecosystems, seagrass meadows have continually suffered habitat loss thereby decreasing their ability to capture and sequester carbon. Once there is loss of the ecosystem functions, the stored sedimentary carbon is released in the form of CO_2 as a result of remineralization or the process of carbon sequestration ceases (Marba *et al.,* 2015). Seagrasses that are subsequently transplanted after degradation of the initial meadows do not usually attain the sequestration capacity of carbon which is similar to the initial un-degraded meadows.

Carbon capture and storage in seagrass is cumulative and complex process involving preservation, delivery production and storage of carbon (Nellemann *et al*., 2009). Most studies on seagrass carbon for example Di Carlo and Kenworthy (2008), and Kaldy and Dunton, (2000) have focused on seagrass above ground vegetation carbon with little focus on sediments carbon. Understanding of the biogeochemical processes involved is therefore vital in determination of seagrass sequestration rates. Recent studies have indicated that organic carbon below the sediment is in anoxic condition, thereby having a high likelihood of preservation (Wakeham & Canuel, 2006; Burdige, 2007). Decrease in the availability of oxygen may lead to diagenetic conditions including inhibition or promotion of the remineralization process.

As flowering plants, the seagrass manufacture their own fo od through the utilization of carbon dioxide and the light energy in the process of photosynthesis. The fixation of carbon in seagrass exceeds the immediate metabolic needs hence the long term storage of the excess carbohydrates in roots and rhizomes. This carbon is then exuded to the sediments where it forms the rich anaerobic organic autochthonous carbon. The seagrass leaves and stems can trap the suspended allochthonous organic carbon from the water and settle it down. This makes the seagrass ecosystems highly efficient carbon storage pools (Howard *et al.*, 2018).

The high belowground biomass usually enhances sedimentary C_{org} storage due to the low turnover and high root production. In addition to the metabolic sources of the sedimentary C_{org}, from the tissues of the seagrass, the suspended organic matter is similarly trapped by the canopies and retained in the sediments as accumulated organic matter (Kennedy *et al.*, 2010; Gullstrom *et al.*, 2017) (Figure 1).

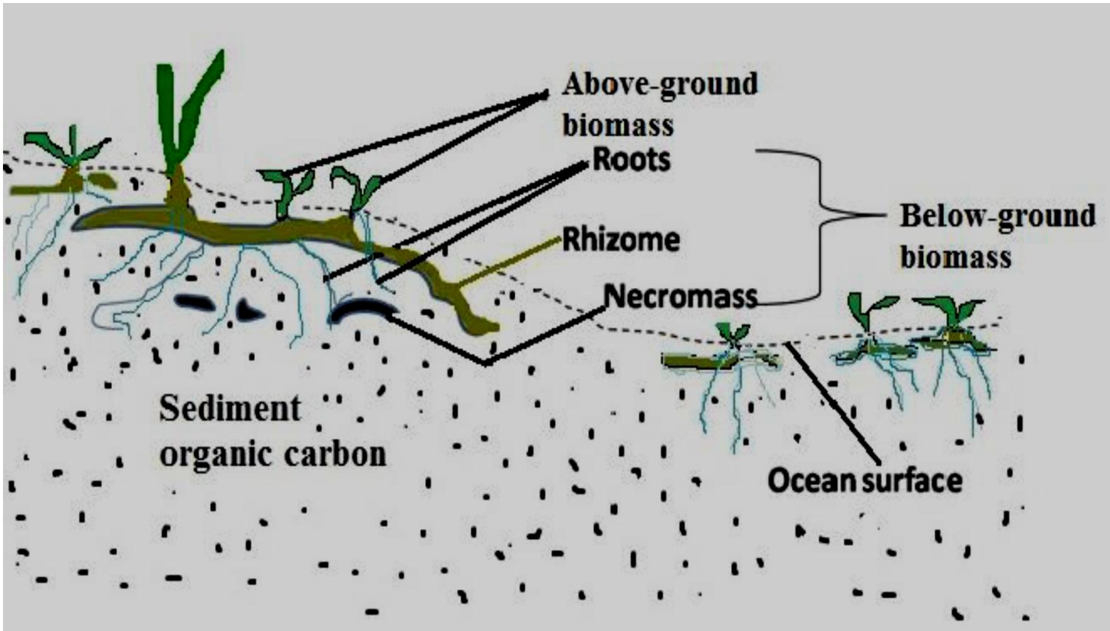

Figure 1: Diagrammatic Illustration of Various Carbon Pools in Seagrass Beds (Githaiga *et al.*, 2017).

Globally, the mean below ground and above ground biomass has been estimated at 474.6 and 174.4 g DW m^{-2} respectively while the annual production rates ranges between 816 and 1012 g DW m^{-2} $year^{-1}$ (Githaiga *et al.,* 2016). The mean production rate for the standing stock is estimated at 5.0 g DW m^{-2} day^{-1} (Duarte & Chiscano 1999). A higher diversity of the above ground vegetation stock comprising of up to 19 species has been identified in Southeast Asia, extending to the Great Barrier Reef to the North Tropical Australia (Green & Short, 2003; Short *et al.,* 2007).

In Africa, the highest values for total biomass have been recorded for at Jambiani in Zanzibar for meadows with mixed species at 3063.3 g DW m^{-2} (Lymo *et al.,* 2008), while the lowest values have been recorded at 0.6 g DW m^{-2} at Inhaca Island in Mozambique among meadows dominated by *Halophila ovalis* (Martins & Bandiera, 2001). Studies in Africa also reveal that *Thalassia hemprichii* has got the highest total biomass at 928.0 g DW m^{-2} while the lowest total biomass has been recorded in *Halodule wrightii* at 19.2 g DW m^{-2} (Githaiga *et al.,* 2016). Larger seagrass species tend to record higher biomass values in the above ground and below ground pool due to reduced turnover rates of the below ground materials (Duarte & Chiscano, 1999). With regards to seagrass productivity, studies in Africa reveal that average rate of rhizomes production is 2.5 g DW $shoot^{-1}$ day^{-1} while that for a leaf is 0.07 g DW $shoot^{-1}$ day^{-1} (Lymo *et al.,* 2006). In terms of regions, East African coast is more studied when compared to the regions. Despite the extensive coastline and seagrass cover in West Africa, there is still limited spatial extent of study hence paucity of information on seagrass vegetation carbon (Githaiga *et al.,* 2016).

In Kenya, studies on above ground and below ground biomass have been done in Gazi bay in which *Syringodium isoetifolium* recorded the highest total biomass at 1984.7 g DW m^{-2} (Githaiga *et al.,* 2017). Other studies have shown that explosion of sea urchins have led to excess herbivory thereby degrading seagrass above ground component. In Watamu and Chale - Diani Marine Reserve, herbivory on seagrass by sea urchin has led to 50% degradation of *T. ciliatum* (Uku *et al.,* 2007).

2.4 Seagrass Sediment Organic Carbon

Seagrass sediments form one of the most important blue carbon pools. It is composed of the allochthonous and autochthonous organic materials which have been buried for a longer timespan forming the sediment $C_{org.}$ (Gullstrom *et al.,* 2017). The efficiency of carbon storage in seagrass sediment is primarily influenced by its properties such as grain size, porosity and density (Dahl *et al.,* 2016). Storage and sequestration of carbon is therefore partially driven by sediment composition and processes that influence the amount of derived allocthonous carbon.

Organic Carbon that is primarily deposited in the marine environment is usually classified as detrital biopolimers which undergoes various types of diagenetic processes and has inherent refractory and liability potential (Jamaludin, 2015). The primary deposition leads to depolymerization of compounds into high and low molecular weight intermediates. These intermediates can be re-mineralized in the presence of micro-fauna, especially the sediment microbes. Some categories of organic matter in detritus however inherently resist degradation and have low remineralization rates. They tend to undergo selective preservation when buried in the sediments resulting to increased concentration with increased age and depth (de Leeuw *et al*., 2006; Zonneveld *et al.,* 2010). Mineral shielding may prevent OM from enzymatic attack while other processes in the geo-chemical environment may prevent OM from further degradation (Rothman & Forney, 2007). These processes can combine or solely preserve OM in the seagrass meadow over different time scales, ranging from months to millennia (Howard *et al.,* 2018).

The below ground carbon storage in the seagrass sediments is twice as much the amount stored per hectare in the terrestrial forests. Additionally, carbon in the seagrass sediments can be stored for thousands of years (Fourqurean *et al.,* 2012). Despite the high plant biomass in the terrestrial ecosystems when compared to the ocean environment, the rate of carbon cycling in the marine environment is equally high (Nellemann *et al.,* 2009).

2.4.1 Effects of Habitat Features on Long-term C_{org} Storage in Soils within the Seagrass Meadows

The biotic interactions that exist between seagrass plant species and other organisms and the role played by the trophic webs can determine the ability of the meadows to perform C_{org} sequestration. Burrowers and grazers help in maintenance of the meadows health as they also benefit from seagrass protection and productivity (Siebert & Branch, 2007; Mazarrasa *et al.,* 2018). The organisms reduce sulfide levels in seagrass by preventing excessive detritus from accumulating in the soil and improving soil oxygenation. This prevents seagrass mortality (Bertics & Zierbis, 2010). However, overpopulation of burrowers and grazers in seagrass meadows may threaten accumulation, carbon stocking and sequestration of autochthonous and allocthonous C_{org} due to significant removal. Overgrazing may also shorten the leaves and reduce canopy cover or change the composition of meadow species thereby allowing for colonizer species dominance hence reducing the sequestration capacity (Atwood *et al.,* 2015; Dahl *et al.,* 2016)

Seagrass species composition within the seagrass meadows affects the sequestration capacity since the species usually vary in dynamic characteristics such as turnover rates, primary productivity and lifespan. They also differ in biomass distribution and size leading to different ecological zones (Duarte & Chiscano, 1999). Small species such as *Halodule spp* and *Halophila spp* are usually colonizers having lower biomass accumulation due to high turnover and growth rates. However, these meadows dominated by small and narrow leaved species may sometimes record higher Corg stocks if they are found in areas where deposition occurs despite the labile nature of the carbon due to low below ground biomass (Lavery *et al.,* 2013). On the other hand, species such as *Thalassia spp* and *Posidonia spp* are long-lived, large and persistent hence tend to have high biomass accumulation (Serrano *et al.,* 2016). The difference in biomass within the system facilitates higher autochthonous carbon accumulation since there is high proximity of the plant material to the refractory conditions of below ground tissues and anoxic nature of seagrass soils (Klap *et al.,* 2000) (Figure 2 (a)).

The complexity of the meadow canopy and landscape patchiness may also influence the amount of C_{org} sequestered in the soil compartment. Seagrass meadows vary in their canopy complexity which is dependent on factors such as leaf area, shoot density and species specific features such as biomass (Samper-Villarreal *et al.,* 2016). Higher canopy complexity increases the ability of seagrass to reduce sediment resuspension and hydrodynamic energy. This explains why seagrass such as *Thalassia spp* and *Zostera* that have blade like leaves trap more particles from the water column and form muddy soils (Peterson *et al.,* 2004; Koch *et al.,* 2006; Peraltra *et al.,* 2008). The epiphytic algal assemblages may also enhance canopy complexity accounting for up to 50% of the above-ground biomass.

The landscape distance to the patch edge, patchiness and patch size may also determine seagrass ecological features such as grain size of sediment, the distribution, sedimentation patterns, biodiversity and detritus transfer (Jackson *et al.,* 2006; Ricart *et al.,* 2015). This affects C_{org} sequestration. Continuous meadows have a greater capacity for sediment C_{org} capture and storage due to their higher ability to accumulate autochthonous carbon (Oreska *et al.,* 2017; Ricart *et al.,* 2017). Recent studies have also revealed that deposition of organic soil carbon increases in magnitude with distance from the edge of the patch into the meadow. This shows that well established and large meadows are more effective in carbon stocking and sequestration (Oreska *et al.,* 2017) (Figure 2 (c)).

The seagrass plant leaves also play significant roles in reducing water flow and accumulating allochthonous C_{org}. When overgrazing occurs, the canopy is reduced thereby hindering the ability of seagrass leaves to trap sediment. The leaves also get highly exposed to erosional forces hence leading to their degradation (Dahl *et al.,* 2016) (Figure 2 (d)).

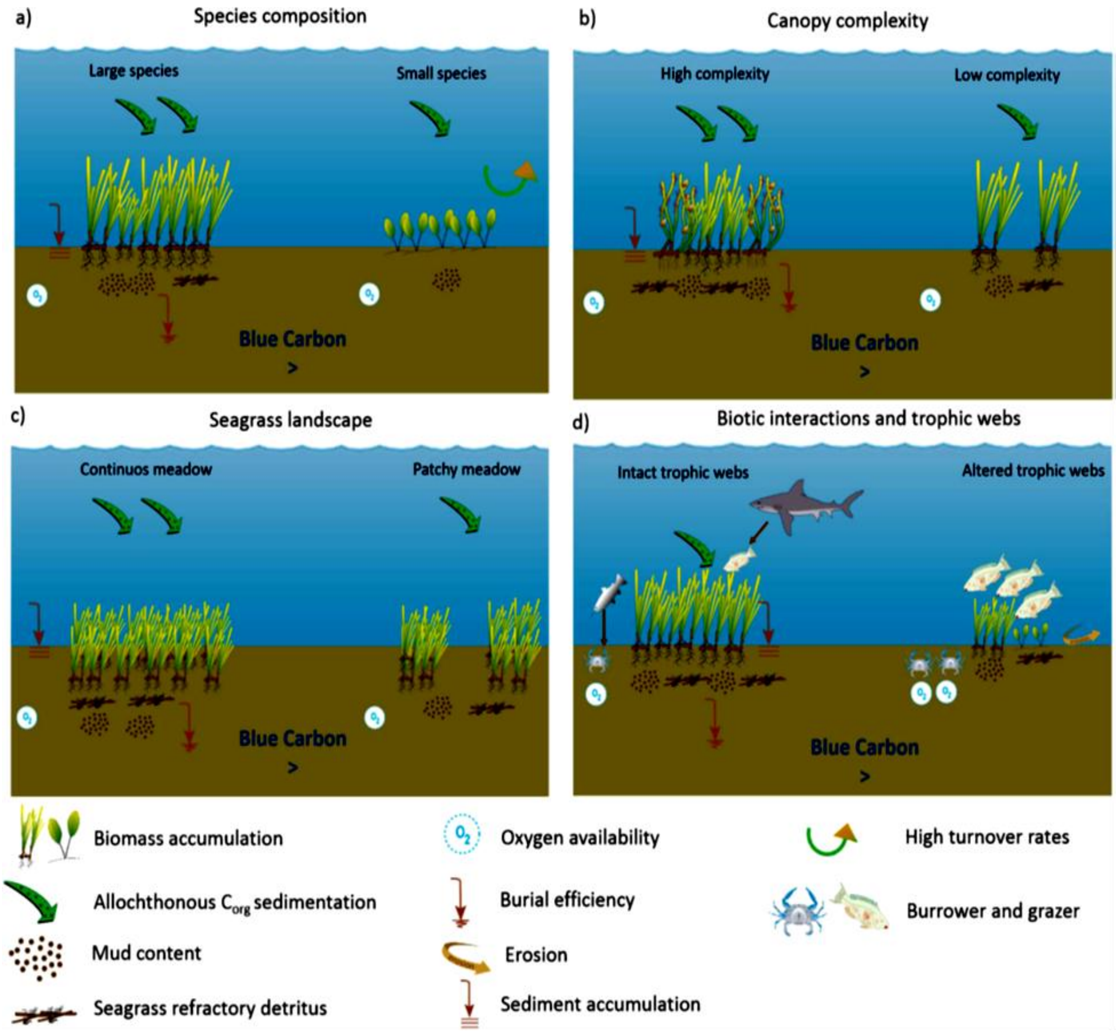

Figure 2: Diagrammatic Illustration of the Effect of Habitat Biotic Features on Blue Carbon Formation, Accumulation and Storage in Seagrass Meadows (Mazarrasa *et al.*, 2018)

2.4.2 Effects of Abiotic Factors on Long-term C_{org} Storage in Soils within seagrass meadows

Water depth affects light attenuation and irradiance thereby affecting seagrass growth, shoot density and productivity. The distribution of seagrass meadows occurs widely from the intertidal zones to the sub tidal zones extending to 40 m of depth. In the deep sub tidal areas, irradiance constrains light penetration thereby inhibiting seagrass growth and productivity. Seagrass meadows occurring in shallow photic zones are therefore, more productive than meadows found in the deeper areas (Collier *et al.*, 2007; Serrano *et al.*, 2016; Dahl *et al.*, 2016). However, the changes in the hydrodynamic conditions may make the shallow meadows more susceptible to wave action thereby favoring soil aeration and seagrass detritus export with reduction in sediment. In this case, deposition of C_{org} in deep seagrass areas may be enhanced than in shallow meadows as the case of *Amphibolis antarctica* and *Posidonia sinuosa* (Koch *et al.,* 2006; Lavery *et al.,* 2013).

Turbidity affects the seagrass growth and carbon storage ability by affecting irradiance which is a prime factor in photosynthetic process and also affects the seagrass morphology, growth and distribution. Earlier studies and experiments have revealed that reduction in the availability of light for seagrass leads to a reduction in growth rate and shoot density (Lee *at al*., 2007; Lavery *et al.,* 2009; Collier *et al.,* 2012). The reduction in shoot density and biomass leads to a decrease in autochthonous carbon sequestration ability and simultaneously, the canopy's capacity to capture and store allocthonous carbon from other ecosystems (Peralta *et al.,* 2008). It can therefore be concluded that higher sediment trapping and sequestration occurs in environments in which turbidity is low. However, high turbidity can also lead to deposition and storage of fine sediment and allocthonous C_{org} after burial due to the low oxygen levels in the fine sediments (Serrano *et al.,* 2016; Samper – Villarreal *et al.,* 2016; Mazarrasa *et al*., 2018).

The degree of seagrass exposure to different hydrodynamic energy forces such as currents, tides and waves determines erosion and sedimentation patterns in the marine environment thereby affecting organic carbon storage. Seagrass meadows usually have more fine sediments in the high canopied and sheltered areas when compared to un-vegetated bare areas (Mazarrasa *et al.,* 2018). The deposition and storage of C_{org} in seagrass meadows is therefore affected by the hydrodynamic energy in various ways including; potential exposure of the buried organic carbon to aerobic conditions, muddy soil formation and the grain size of sediments and the balance between export and accumulation for both the allochthonous and autochthonous C_{org} (Serrano *et al.,* 2016).

The morphology, growth and abundance of seagrass are equally strongly affected by nutrient availability thereby affecting seagrass carbon storage capacity. More autochthonous carbon is likely to be accumulated in seagrass meadows with high nutrient availability where shoot density, shoot length, biomass and productivity are also enhanced (Lee & Dunton, 2000). There is also high efficiency in these meadows to water flow reduction thereby enhancing allocthonous C_{org} reduction when compared to the seagrass meadows found in oligotrophic waters. The ability of nutrient availability to influence seagrass morphology and both autochthonous and allochthonous carbon makes it a key factor in sediment C_{org} storage in seagrass (Armitage & Fourqurean, 2016). However, availability of nutrients in excess may lead changes in the composition seagrass species within the meadow. These changes may enhance colonization by the fast-growing species *Halodule spp* and *Halophila spp* which are persistent and usually sequester less autochthonous carbon (Howard *et al.*, 2016).

The occurrence of the seagrass meadows either in subtidal or intertidal areas is another factor affecting carbon sequestration capacity. In many cases, seagrass meadows found in intertidal areas are subjected to various unfavorable and extreme environmental conditions than the meadows found in subtidal areas (Silva *et al.*, 2009). These conditions include higher hydrodynamic forces such as tide currents and wave actions, higher variations in water levels, excessive irradiance and exposure to air. During the ebb tide, there is complexity in the response of seagrass productivity with regards to their exposure to air. Despite some of the earlier studies finding that seagrass in the intertidal areas assimilate CO_2 in the atmosphere rapidly during low tides leading to higher photosynthetic rates than in sub-tidal areas (Silva *et al.*, 2009), other studies have shown lower rates of photosynthesis and productivity in intertidal areas than submerged areas (Clavier *et al.*, 2011).

The amount of water in seagrass plant tissue constraints accumulation of CO_2 during the periods when the plants are exposed to air hence the desiccation process limiting carbon gains (Leuschner *et al.*, 1998). Due to the complex interactions taking place between intertidal and subtidal meadows of seagrass, more carbon storage is expected in submerged regions. However, the only study that assessed this variation in C_{org} stocks in these meadows under the same species did not find a significant difference (Lavery *et al.*, 2013).

The climatic region in which the seagrass plant is found may also influence the ability of seagrass plants to capture and store carbon. Globally, seagrasses vary in their distribution across different latitudes that experience different day length, irradiance and temperature (Short *et al.,* 2007; Olesen *et al.,* 2015). The variation in latitude brings about differences in ecosystem dynamics and seagrass rates of metabolism which consequently affect C_{org} capture and sequestration ability in the plants' sediment. From the recent studies, the annual rates of seagrass productivity and biomass in the above-ground component have been increasing from lower to higher latitudes (Duarte & Chisano, 1999). On the other hand, the turnover rates for biomass have had the tendency of decreasing from regions of lower latitudes to higher latitudes (Olesen *et al.,* 2015).

Seagrass meadows in the tropical region have lower seasonal biomass variability when compared to the meadows in temperate regions. Their sequestration capacity is higher during the summer or spring season when the productivity rates are high. This also brings about seasonal fluctuation C_{org} capture and storage (Bos *et al.,* 2007). However, the trend may be different and the variation may come about due to difference in species. For example, *Z. marina* may have higher biomass and sequestration capacity in the tropical region when compared to the temperate regions (Clausen *et al.,* 2014). In tropical regions, the temperatures are high when compared to temperate regions. This may increase the remineralization rate of organic matter thereby lowering burial efficiency of carbon in seagrass soils. In other studies, the biomass and composition of microbial community across the latitudinal range in terrestrial soils has been considered an important factor accelerating the sedimentary organic matter remineralization than temperature variation (Bradford *et al.,* 2017).

In comparing seagrass soils carbon between temperate and tropical regions, some studies have revealed no significant variations between the regions, (Lavery *et al.,* 2013), while others have found significant differences. Miyijama *et al.,* (2015) found a significant variation between the soil C_{org} stocks of subtropical and tropical meadows of Southeast and Eastern Asia, and those of the temperate regions. The variation can be brought about as a result of temperature difference and other habitat conditions affecting seagrass morphology and productivity at a smaller scale.

In the future, seagrass meadows may vary as a result of the increasing anthropogenic pressures and climatic changes. This may similarly cause changes in sequestration potential of the seagrass meadows. With regards to the changing climatic conditions, factors such as rise in temperature, ocean acidification and sea level rise will impact on seagrass growth thereby affecting carbon storage capacity (Duarte *et al.,* 2002; Mazarrasa *et al.,* 2018). Sea level rise is likely to impact on reduced ocean irradiance and increased depth due to the suspended sediments resulting from soil erosion along the coastline. Primary productivity and seagrass carbon sequestration potential in subtidal and deeper regions will therefore be reduced resulting in meadow expansion on the intertidal and landward regions (Saunders *et al.,* 2013).

Increase in temperature globally is likely to impact on various features including seagrass turnover rates, annual biomass, species distribution, productivity and phenology. Increased temperatures in the temperate regions will particularly reduce seagrass productivity and biomass and consequently increase the plants' turnover rates while decreasing sequestration of autochthonous carbon (Clausen *et al.,* 2014; Kendrick *et al.,* 2017). Seagrass meadows may similarly expand to the northern latitudes especially the eelgrass species. However, this expansion may be limited by the constrained seed dispersal due to lower temperatures hence low propagule supply. Sequestration of autochthonous carbon including the above and below ground biomass could similarly increase with increased acidification (Russell *et al.,* 2013).

The increasing human activities and threats such as reclamation of coastlines and agricultural practices upstream has led to accumulation of terrestrial sediment and nutrient into the seagrass region, thereby increasing turbidity and hindering the plants' survival (Waycott *et al.,* 2009). However, this process may favor allochthonous C_{org} accumulation and deposition including the macro algae (Samper – Villarreal *et al.,* 2018). However, the carbon deposited from terrestrial sources may be highly vulnerable to remineralization and labile when compared to autochthonous one. Additionally, excess supply of nutrients from river in-flow may trigger higher microbial rates further increasing the rate of remineralization. It is therefore apparent that human pressure and changes in climatic conditions may threaten the long-term functionality of seagrass meadows as carbon sinks (Ricart *et al.,* 2017; Oreska *et al*., 2017).

Empirically, studies on global seagrass carbon stocks in the sediment have been estimated at 165.6 Mg C ha^{-1}. The stocks from different regions have also been established to range between 115.3 to 829.2 Mg C ha^{-1} (Fourqurean *et al.,* 2012). Another study by Mateo *et al.,* (2006) has revealed that carbon is stored inform of seagrass biomass over a short period while it can stay up to 4 years inform of detritus material and up to millennia in the sediment. In general, there is still paucity of information globally regarding seagrass sediment carbon. This is due to few studies in carbon dynamics and uncertainties in seagrass spatial extent (Fourqueran *et al.,* 2012; Githaiga *et al.,* 2017).

In Africa, a number of studies on seagrass sediment carbon stocks have been done in the Western Indian Ocean region, along the East African coastline with very few studies in West Africa. In a study by Lymo *et al* (2016) in Zanzibar, the findings revealed that disturbances on seagrass meadows including shading and grazing reduce biomass in seagrass thereby lowering the ability of seagrasses to store in the sediment. In another study by Gullstrom *et al* (2017) in Mozambique and Tanzania, the findings indicated that seagrass vegetated areas have higher C_{org} stocks than the un-vegetated ones (adjacent bare patches).

In Kenya, sediment carbon stocks have been determined in the seagrass meadows of Gazi bay and estimated at 236 ± 24 Mg C ha^{-1} (Githaiga *et al.,* 2017). These studies also revealed that sediment is the carbon pool that stores the highest percentage of carbon. Other studies in Gazi bay have indicated that seagrasses and mangroves comprise the major organic carbon sources for storage within these ecosystems or are out-welled in the surrounding ecosystems. Similarly, about 50% of the carbon stored in seagrass meadows originates from the adjacent ecosystems including the mangroves and the terrestrial ecosystems (Bouillon *et al.,* (2007).

2.5 Policy gaps and Implications

Based on existing literature and policies regarding seagrass conservation and management, the following are gaps that will be filled by the current study:

i. The information received from the current study will be used to strengthen the policy and legal framework regarding the management of seagrasses in Kenya.
ii. Understanding seagrass composition, distribution and stocks will help in the implementation of Seagrass Ecosystems and Coral Reefs Conservation Strategy.
iii. It will also help increase awareness on the importance of these critical ecosystems and the impacts of their degradation.

2.6 Theoretical Framework of the Study

The current study based on the Driving force, Pressure, State, Impacts and Responses (DPSIR) framework. This model describes the interactions between the society and environment (Cutter, 1996), and is used in assessment and management of various environmental associated problems. The driving forces are the various socio-cultural and socio-economic forces driving human influence which mitigate or increase pressure on the environment. These include poverty, population growth, lower education levels and economic development. Pressures refer to the stresses placed on the environment as a result of human activities for example, pollution and emissions, and shore line change, over-exploitation of resources, sea level rise, ocean acidification and eutrophication (Kristensen, 2004). State refers to the environmental condition and could include biological or physical-chemical state while impacts are the effects resulting from environmental degradation for example, decline in fisheries catches, modification and loss of critical habitat and loss of biodiversity. Finally, the responses are the different manner in which the society responds to the situation in the environment for example response in terms of health, ecosystems or political response, changes, in policies, laws and management options (Cutter, 1996).

2.7 Conceptual Framework of the Study

The study has the independent variables including the seagrass distribution, seagrass abundance and seagrass species which show the current state of the ecosystem. These factors can be affected in one way or another by human and natural factors. The dependent variables on the other hand include above ground biomass, below ground biomass and sediment carbon that is influenced by the different species, their richness and abundance. The intervening variables include carbon out-welling, the natural factors such as sea urchin herbivory, sedimentation, wave action and human factors such as reclamation of the coastline and unsustainable fishing methods, that acts as the drivers and pressures. There are also the moderating variables including the physical-chemical parameters such as turbidity, current, depth and salinity which moderate the interactions between independent and the dependent variables (Figure 3).

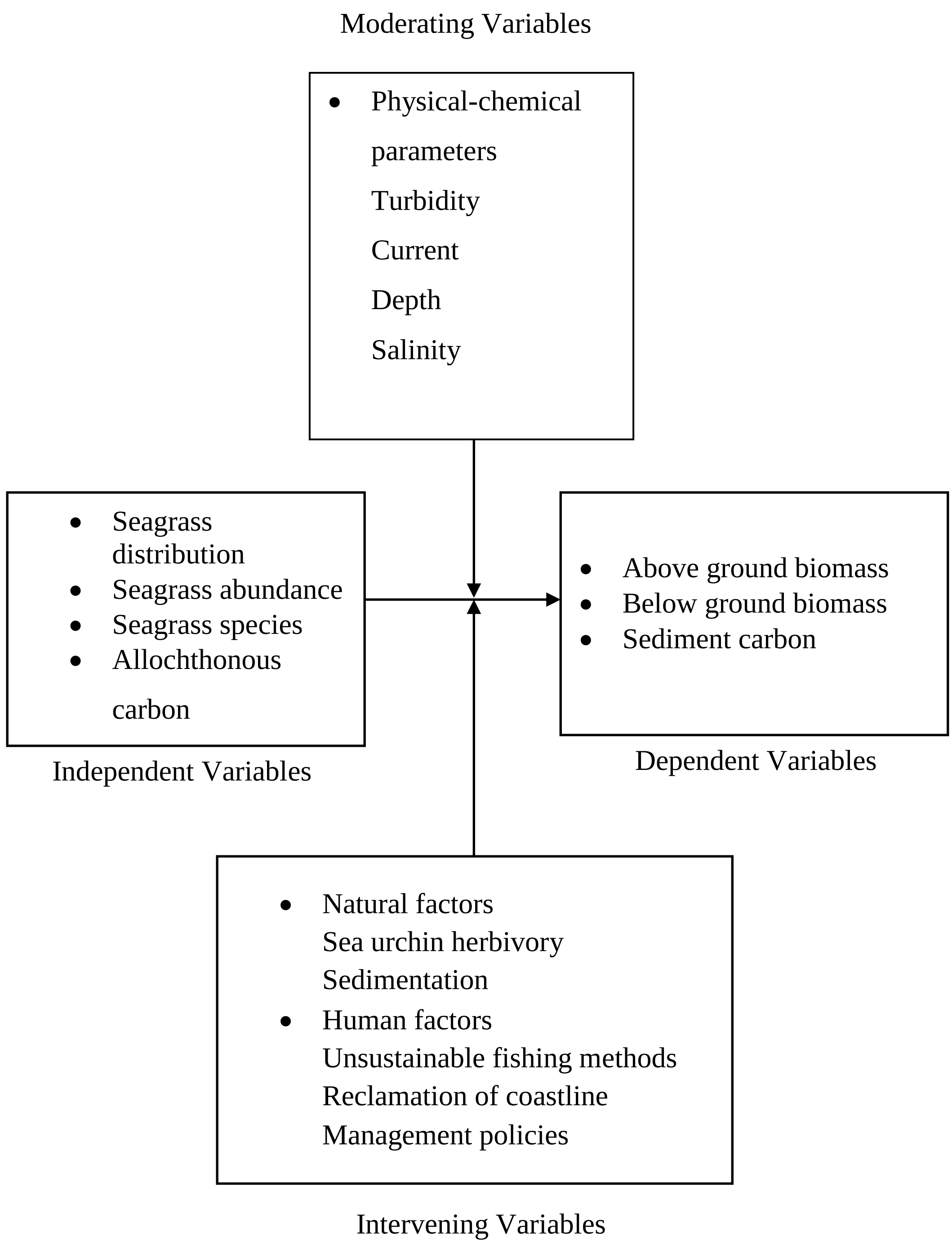

Figure 3: Conceptual Framework

CHAPTER THREE

MATERIALS AND METHODS

3.1 The Study Area

The study was carried out in Eastern and Western Creeks of Gazi Bay, located in Kwale Conty, Kenya (4˚25'S, and 39˚30'E); between the months of May and November 2018. The bay has a surface area of approximately 17 km^2 and is characterized by shallow water system with the mean depth less than 5 m. Mangrove forest cover 615 ha of the bay with seagrasses extending from the intertidal to the subtidal areas (Fig. 4). Coral reefs are also found in the southern part of the bay, occurring in scattered patches. They harbor different species of organisms including molluscs, crustaceans, fish and echinoderms, thereby supporting biodiversity and coastal livelihoods through tourism and fisheries (Obura, 2012). The two tidal creeks originate from the open waters of the bay, penetrating through the mangrove forest.

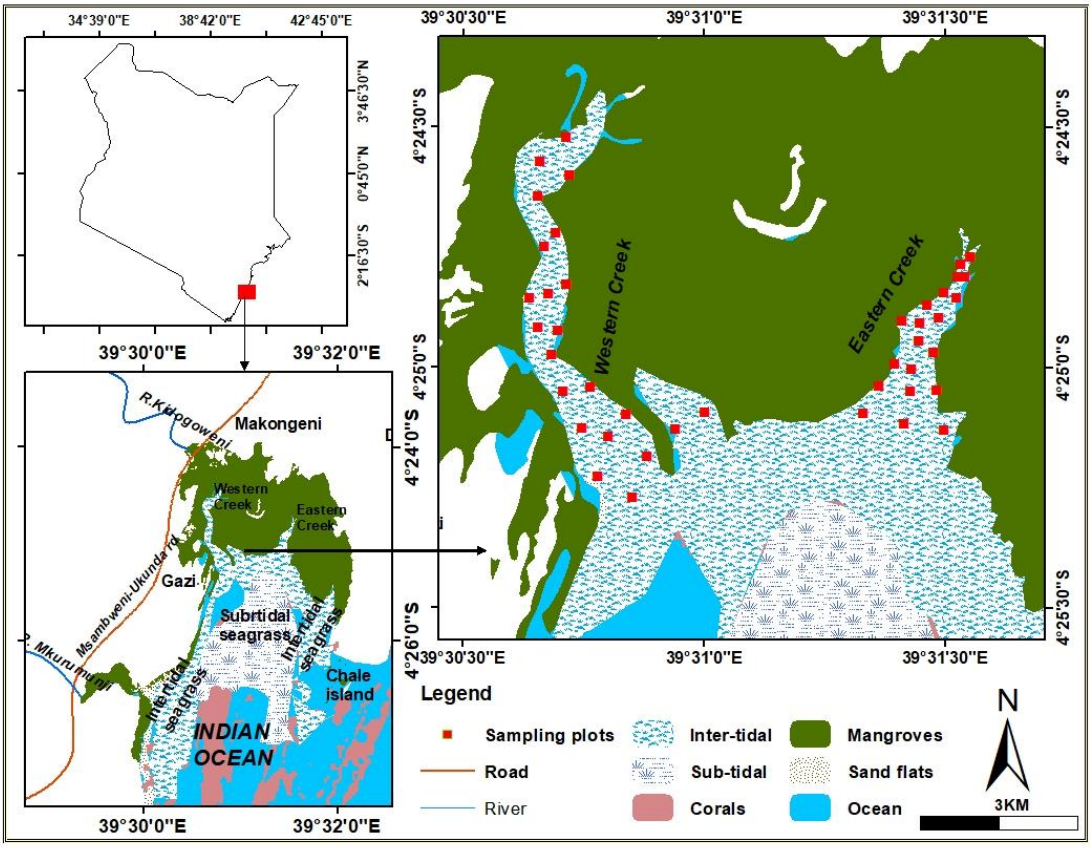

Figure 4: Map of Gazi bay showing Seagrass Meadows within Eastern and Western creeks of Gazi Bay.

3.1.1 Biophysical Conditions

3.1.1.1 Climate

Like in other parts of the coast, the climate of Gazi is influenced by monsoon winds. Short rains occur between October and November coinciding with North East Monsoon winds while the long rains are experienced between March and May during the South East Monsoons period, with inter-annual shifts commonly occurring in these seasons. Offshore prevailing winds similarly occur throughout the year (Schott *et al.,* 2009).

3.1.1.2 Drainage

The western creek opens to River Kidogoweni which is a seasonal fresh water inflow to the Indian Ocean. It, therefore, has a channel connecting through the creek where the fresh water mixes up with the ocean water. On the other hand, there is no freshwater opening to the eastern creek. In the southwestern part of the bay, river Mkurumuji opens to the Indian Ocean, with a higher annual discharge when compared to river Kidogoweni. During the rainy season, fresh water mixes with the sea water at the estuary in the upper water layers (Kitheka *et al.,* 1996).

3.1.1.3 Tidal Regime

The bay has a semi-diurnal tidal regime with the tidal amplitude varying between 70 cm at neap tide and 209 cm during the spring tides. In the bay, the magnitude of the semi-diurnal mixed tides to flush vary and is dependent on the nature of the tide, tidal elevation and the tidal range. During spring periods, the tide disperses salinity water rapidly and is swift. Neap tide, however, flashes brackish water and is generally slower (Kitheka, 1996). During the spring tides, the ebb period measures 7 to 8 hours and is usually longer than flood period which measures 4 to 5 hours. However, this variation reduces towards neap tide making both flood and ebb period to have equal durations (Hemminga *et al.,* 1994). There is a high velocity of current at the entrance of the creeks than in the open waters of the bay. The tides are mixed semi-diurnal and have a tidal range of 3 m at springs and 1.4 m at neaps generating weaker currents in the open bay waters and strong reversing currents within the tidal creeks. As the tides rises, water initially flows through the creek channels before inundating the tidal flats and the surrounding mangroves as the tide rises.

3.1.1.4 Seagrass of Gazi bay

In Gazi bay, the seagrass meadows fall within the Chale-Diani Marine Reserve. The meadows also form part of the Transboundary Marine Conservation Area (TBCA) between Kenya and Tanzania. All the 12 seagrass species described to be found in Kenya are found in Gazi Bay. The dominant species in the bay include the climax, long lived species such as *Thalassodendron ciliatum, Syringodium isoetifolium, Thalasia hemprichii* and *Enhalus acoroides*. Additionally, other pioneering short lived species such as *Halophila minor, Halophila ovalis, Halodule writghtii* and *Halodule uninervis* are also found in the bay. In many cases, these species either occur as single or mixed stands (Ochieng' & Erftemeijer, 2003). The species either mix with other species or grow in monospecific stands with their coverage extending from the subtidal to the intertidal areas in the rocky and sandy substrates.

The seagrass meadows in Gazi bay also occur in close connectivity with mangroves and corals in terms of biophysical characteristics and biodiveristy. As such, human activities and threats on one of the critical ecosystems similarly impact the other. Mangrove species in the bay such as *Sonneratia alba* and *Rhizophora mucronata* inhabit the bay and the fringed creeks thereby allowing for carbon out welling into the adjacent seagrass meadows (Hemminga *et al.*, 1994; Kairo *et al*., 2001).

3.1.2 Socio Economic Activities

The major human activity in the area is fishing. Most of the artisanal fishing activities take place in the intertidal and subtidal parts of the bay where seagrass meadows occur. These include gleaning, Poor fishing activities in the bay, such as beach seining has contributed to loss and degradation of seagrasses in the bay. Other activities carried out by communities living adjacent to the bay include small scale agriculture, mangrove harvesting and eco-tourism (Githaiga e*t al*., 2017).

3.2 Research Design

The study used ecological survey research design. This involved investigating various ecological parameters in-situ without manipulating the environment. Quadrat sampling and the grab and core sampling were used (Henderson & Southwood, 2016).

The sampling procedure adopted stratified random sampling protocols detailed in the Blue Carbon manual (Howard *et al.*, 2014), and the IPCC's supplements on coastal wetlands (IPCC, 2014). In determining the distribution and abundance of seagrass in the creeks, transect method was used where six transects were laid perpendicular to the water line, at intervals of 100 meters. Quadrats measuring 0.5m by 0.5m were then laid along the transect line at intervals of 25 m. In assessing carbon stocks in seagrass meadows within the creeks, stratified random sampling method was used to identify five monospecific stands including *Thalassia hemprichii*, *Cymodocea rotundata, Cymodocea serrulata, Enhalus acoroides, Thalassodendron ciliatum and* mixed based on dominance. Eighty quadrats measuring 0.5 m by 0.5 m were then established within the seagrass meadows and a total of 480 samples obtained for laboratory processing to determine the carbon stocks (IPCC, 2013). The data collection process was done in the months of May and September 2018 during the spring tides when the seagrass beds are accessible.

3.3 The Study Site

The Eastern creek covers an area of 50.0 ha and is fringed by mangrove species, *Sonneratia alba* and *Rhizophora mucronata* that are dominant. It is characterized by shallow subtidal areas and channels that intersect the intertidal flats. Some areas have rocky substrate and macroalgae existing in close association with the seagrass species. The mangrove forest fringing this creek is drained during the ebb tide as the water flows through the channel to the deeper parts of the bay (Figure 5)

Figure 5: Mangrove fringed Eastern creek of Gazi bay with seagrass meadows

The Western creek on the other hand covers about 70.0 ha and is open to the seasonal River Kidogoweni allowing water to flow through the river channel towards deeper parts of the bay during ebb tides. During flood periods of rainy seasons, sea water mixes with incoming fresh water. The creek is more open and is characterized by sand bank usually exposed during the ebb period. This creek is also adjacent to the community residence hence various activities including tourism (Gazi Women Mangrove Boardwalk), aquaculture, fish landing site and bee keeping activities take place here.

3.4 Vegetation Sampling

The sample plots were established within the strata by placing the 0.25 m^2 at minimum intervals 25 m in each of the seagrass strata. This was done during the spring tides when the seagrass beds are accessible and exposed. The different types of seagrass species were identified in-situ using field manuals (Richmond, 1997). The percentage canopy cover was determined through visual estimates. Estimating the canopy height involved measuring heights of 10% of individual shoots randomly selected from the total number within the quadrat and calculating their mean heights.

Canopy cover and shoot density was determined through counting the entire shoots within the quadrats then extrapolated to per m^2 (Githaiga *et al*., 2017).

3.5 Measurement of Physio-chemical Parameters

Measurements of physio-chemical water parameters were done *in-situ*, during low tides. Total dissolved solids (mg/L), conductivity (µs/cm), water temperature (˚C), and p.H were measured using the HANNA Combo PH and EC multi meter Hi 98129. Salinity (ppt) was measured in the creeks using refractometer while depth was measured using a graduated tape.

3.6 Estimation of Above-ground Biomass

For the above-ground biomass, harvesting of all the seagrass materials above ground within the 0.25 m^2 quadrats was done and then cleaned using fresh water. Sorting and scraping then followed to remove the epiphytes. The fronds were then washed with 10% HCl to remove the calcareous materials then drying done in an oven at 60˚C for 72 hours to attain constant weight (Howard *et al.*, 2014).

3.7 Estimation of the Below-ground Biomass

The estimation of the below-ground biomass was done by taking four cores from each of the 0.25 m^2 quadrats using the Russian Peat Sampler. In-situ treatment included washing using a 500 µm sieve. Samples were then sorted into necromass, rhizomes and roots and then dried for 72 hours at 60˚C in the laboratory. The sum of the values obtained for each species was then summed and conversion done to per meter square. Total biomass was then determined through multiplication of biomass with the carbon conversion factor of 0.34 and extrapolation done to per hectare (Howard *et al.*, 2014).

3.8 Determination of Sediment C_{org}

At least two 50 cm sediment cores of 3 – 6 cm in diameter were made in vegetated and un-vegetated quadrats using a peat sampler. The un-vegetated core served as controls. Since the relative content of the sedimentary carbon may be influenced by the sediment compaction, the difference in length from the upper part of the core to the surface of the sediment, outside and inside the corer were assessed when passing it down into the sediment, (IPCC, 2014). The sediment was then homogenized using

500 μm sieve and cleaned of plant material, infauna and larger shells before drying was done. The samples were then sliced into sub-sections of 5 cm and oven-dried at 60˚C for 72 hours to find a constant weight. The top 50 cm estimates of the sediment were then extrapolated to a meter leading to the determination of the sediment C_{org}.
The dry bulk density was then calculated for each of the sub-sections using the formula;

DBD (gcm3) = Dry Weight / Original Volume of sediment …………… (1)

Where **DBD** is Dry Bulk Density

Percentage porosity of the sediment was estimated through calculation of water content in the sediment by subtracting the wet weight from the dry weight of the sediment (Gullstrom *et al.,* 2017).

3.8.1 Measurement of Percentage Organic Matter

The organic matter was determined using the Loss of Ignition Technique. The samples were pre-heated in a muffle furnace for 3 hours at 450˚C. The percentage OM was then calculated using the formula

$$\%LOI = \left(\frac{Initial\ dry\ weight - Weight\ remaining\ after\ ignition}{\text{Initial Dry Weight}}\right) \times 100 \text{ ……………….. (2)}$$

Where **% LOI** is the percentage Loss of Ignition

Depending on the organic matter in each of the sample, the sediment C_{org} values were calculated using one of the two equations below; (Howard *et al.*, 2014).

% LOI < 0.20, % C_{org} = -0.21 + 0.40 (% LOI); …………………………… (3)

% LOI > 0.20, % C_{org} = -0.33 + 0.43 (% LOI). ………………………...…. (4)

Where **% LOI** is the percentage Loss of Ignition and **% C_{org}** is the percentage sediment organic carbon.

3.9 Data Analysis

In calculating the species richness, diversity and abundance, the study used Shannon – Wiener diversity index (H) and the index of evenness. The equitability or evenness is the uniformity of abundance in a defined assemblage of species. The species richness on the other hand is the number of species contained in a given area (Nollan & Callahan, 2006).

The Shannon- wiener index is given by,

$$H = -\sum_{i=1}^{s}(\text{Pi} \times \text{In Pi}) \quad \text{.............................} (5)$$

Where $\boldsymbol{H}$; is the species diversity index, **s**; the number of species and $\mathbf{P_i}$; is the species individual proportion that belong to the i^{th} species of the total number of seagrass individuals.

The Evenness index is given by:

$$E = H/\text{In S} \quad \text{..} (6)$$

Where $\boldsymbol{E}$*;* is the species evenness, **H** is the Shannon wiener Index and **S**; the species richness.

To analyze for the significant variations between creeks and among species, the data was tested for homogeneity and normality of variance. The data was arc sine transformed to meet parametric test, where assumptions of normality was not found. One way ANOVA was used to test for variation in the above ground, below ground and sediment carbon among the species. Where significant variations were detected, the Tukey post hoc test was used to compare the means. Two sample t test was used to test for significant variations in biomass and sediment C_{org} between the creeks and to also test for significant difference in C_{org} between vegetated and un-vegetated areas. In testing for the relationship between above ground, below ground and sediment carbon, correlations analysis was used. In all the statistical tests, the significant level was set at $\alpha = 0.05$ (Kothari, 2004).

3.10 Ethical Considerations

Research clearance was first obtained from the Graduate School of Chuka University. Approval was then sought from Kenya Marine and Fisheries Research Institute, Mombasa. Thereafter, research permit was obtained from National Commission for Science Technology and Innovation (NACOSTI). Upon obtaining the authorization, (Permit No NACOSTI/P/18/52699/24288), the field research activities commenced. High ethical considerations were put in place during the research process to prevent environmental damage on the ecosystems of study or on other ecosystems and organisms. Protective measures were similarly considered for the researchers and the research assistants including availing rain coats, snorkeling gears, a boat and laboratory coats to prevent harm or danger. The information used in the Thesis was also appropriately cited to avoid plagiarism.

CHAPTER FOUR

RESULTS AND DISCUSSION

4.1 Physical–Chemical Parameters in Eastern and Western Creek

Most of the physical-chemical parameters measured were found to vary significantly between the creeks. Depth ranged between 20 cm to 80 cm in the Western creek (mean 0.53 ± 0.31 m) while the values for the Eastern creek ranged between 50 cm and 150 cm (mean 0.93 ± 0.20 m) during the ebb tide when the samples were taken (Table 1). Salinity varied between the two creeks with the Eastern creek recording higher values, mean 34.7 ± 0.65 ppt, (range: 33 – 35 ppt). The Western creek had a mean of 30.0 ± 0.60 ppt, range: 29 – 31 ppt. Two sample t-test revealed a significant variation between the creeks ($t = 18.08, d.f = 20, p < 0.05$) at 95% confidence level.

Turbidity values were higher in the western creek mean, 0.52 ± 0.07 mg/L; range (0.35 – 0.68 mg/L) than in the eastern creek, mean 0.32 ± 0.06 mg/L; range (0.2 – 0.41 mg/L) The means recorded for the two creeks revealed a significant difference ($t = -6.91, d.f = 21, p < 0.05$) at 95% confidence level. Variation was similarly observed in pH values between the creeks with Western creek ranging (7.4 – 8.3) while the pH values for the Eastern creek ranged between 7.8 and 8.0 (Table 2). On the other hand, assessing water temperature between the creeks showed that Western creek had higher values; with the mean at 31.9 ± 0.83 ˚C; range 30.3 – 33.2 ˚C while Eastern creek had 29.7 ± 0.40 ˚C; range 29.0 – 30.2 ˚C. These means revealed a significant variation in temperature ($t = -8.21, d.f = 16, p < 0.05$) at 95% confidence level.

Table 2: Physio-chemical Parameters in Eastern and Western creeks

Parameter	Western creek	Eastern creek
Location (Lat; long)	04.41661˚S; 039.51253˚E	04.41610˚S 039.52610˚E
Depth (m)	0.53 ± 0.31	0.93 ± 0.20
pH (range)	7.4 – 8.3	7.8 – 8.0
Water temperature (˚C)	31.9 ± 0.83	29.7 ± 0.40
Salinity (ppt)	30.0 ± 0.60	34.7 ± 0.65
Turbidity (mg/L)	0.52 ± 0.07	0.32 ± 0.06

4.2 Seagrass Species Distribution and Diversity in the Creeks of Gazi bay

Nine seagrass species out of the 12 species recorded in the entire bay were encountered in the creeks (Table 3). At least six species; *Cymodocea rotundata, Halodule uninervis, Thalassia hemprichii, Cymodocea serrulata, Syringodium isoetifolium* and *Enhalus acoroides* were recorded in both creeks. However, *Halophila stipulacea* and *Halophila ovalis* were only recorded in the Western creek while *Thalassodendron ciliatum* was in the Eastern creek. The most common species in the Eastern creek was *Thalassodendron ciliatum* while *Cymodocea rotundata* was most common in Western creek (Table 3).

Table 3: Seagrass Species Distribution and Frequency in Eastern and Western Creeks

	Western Creek		Eastern Creek	
Species Name	Frequency	%	Frequency	%
Cymodocea rotundata	20	50	3	7.5
Halodule uninervis	9	22.5	4	10
Thalassia hemprichii	6	15	6	15
Cymodocea serrulata	6	15	7	17.5
Syringodium isoetifolium	1	2.5	2	5
Halophila stipulacea	6	15	0	0
Halophila ovalis	1	2.5	0	0
Enhalus acoroides	1	2.5	6	15
Thalassodendron ciliatum	0	0	18	45
Number of species observed	8		7	

The diversity of seagrass species (expressed as Shannon Weinner Index, H) was higher in the Eastern creek (H = 1.71) than in the Western creek had (H = 1.67). Similarly, the Eastern creek had higher Evenness value (E = 0.88) than the Western creek (E = 0.80).

In estimating the relative percentage cover of the species in the two creeks, *Thalassodendron ciliatum* had the highest percentage cover in the Eastern creek at 57.22% followed by *Syringodium isoetifolium* (30%), while *Halodule uninervis* was lowest at 18.75%. In the Western creek, *Syringodium isoetifolium* had the highest cover at 40% followed by *Cymodocea serrulata* at 28.33% and *Cymodocea rotundata* at 26%. The lowest spatial coverage of 15% was recorded by *Enhalus acoroides* and

Halophila ovalis. Despite recording highest percentage cover, *S. isoetifolium* had the lowest frequency in the Western creek.

Overall spatial coverage of seagrass in Eastern creek was higher at 69.17% when compared to 56.43% in the Western creek. The mean seagrass percentage abundance showed a significant variation between the creeks (*$t = 1.97$, $d.f = 35$, $p < 0.01$*) at 95% confidence level.

4.3 Above ground and below ground biomass of seagrass between Creeks and Among Species

Mean values of above-ground biomass was higher in the Eastern creek (1.01 ± 0.15 Mg C ha^{-1}); range: (0.24 – 3.79 Mg C ha^{-1}) than in the Western creek of (0.49 ± 0.03 Mg C ha^{-1}; range, 0.21 – 1.03 Mg C ha^{-1}). In the two creeks, the mean values were significantly different (*$t = -3.422$, $df = 79$, $p < 0.0001$*) at 95% confidence level.

In the comparison of above-ground biomass among the species, the highest biomass values were recorded in *Thalassidendron ciliatum* meadows (2.38 ± 0.28 Mg C ha^{-1}, range 1.22 – 3.79 Mg C ha^{-1}) while the lowest was in *Cymodocea rotundata* of the Western Creek (0.35 ± 0.04 Mg C ha^{-1}, range; 0.21 – 0.56 Mg C ha^{-1}). There was a significant difference in biomass values among the species, ($F_{(7,79)} = 38.35$, $p < 0.0001$), (Table 4).

Table 4: Analysis of Variance (ANOVA) to Test for Differences in Mean above Ground Biomass between Species

Source	df	Sum of Squares	Mean Square	F Value	P
Species	7	0.0629	0.0090	38.35	< .0001
Error	72	0.0169	0.0002		
Corrected Total	79	0.0798			

In comparing the below-ground biomass between the two creeks, higher biomass was recorded in Eastern creek at 9.17 ± 0.67 Mg C ha^{-1}, range (0.76 – 16.72 Mg C ha^{-1}) while in Western creek, mean below ground biomass values was 3.84 ± 0.29 Mg C ha^{-1}, range (0.67 – 8.92 Mg C ha^{-1}). The comparison revealed high significant difference, ($t = -7.25$, $df = 79$, $p < 0.0001$) at 95% confidence interval.

In comparing below ground biomass among the species, the highest means were recorded in *Enhalus acoroides* at 12.35 ± 1.02 Mg C ha^{-1}, range: (8.19 – 16.72 Mg C ha^{-1}) and the lowest in *Cymodocea rotundata* of the Western Creek at 3.07 ± 0.40 Mg C ha^{-1}; range: 1.14 – 4.42 Mg C ha^{-1}). The comparison revealed a significant difference between the means ($F_{(7,79)}$ = 12.98, p < 0.0001), (Table 5).

Table 5: Analysis of Variance (ANOVA) to Test for Difference in Mean Below Ground Biomass Among Species

Source	df	Sum of Squares	Mean Square	F Value	P
Species	7	0.3105	0.0444	12.98	<.0001
Error	72	0.2460	0.0034		
Corrected Total	79	0.5565			

4.4 Total Biomass between the Creeks and Among the Species

Total biomass varied among the species with the highest biomass values being associated with *Enhalus acoroides* while the lowest total biomass was associated with *Cymodocea rotundata* of the Western Creek. Comparing the means revealed a significant difference in total biomass, ($F_{(7,79)}$ = 14.6, p < 0.05) (Table 6).

Table 6: Analysis of Variance (ANOVA) to Test for Significant Difference in Total Biomass among Seagrass Species

Source	df	Type III SS	Mean Square	F Value	P
Species	7	0.30058980	0.04294140	14.60	<.0001

Similarly, there was a significant variation in total biomass between the creeks, (t= -8.44, df. = 79, p < 0.0001) at 95% confidence level with the Eastern creek recording a higher values of 10.18 ± 0.62 Mg C ha^{-1}, range; (3.23 – 17.24 Mg C ha^{-1}), while the Western creek had 4.33 ± 0.29 Mg C ha^{-1}, range: 1.35 – 9.34 Mg C ha^{-1}) (Figure 6).

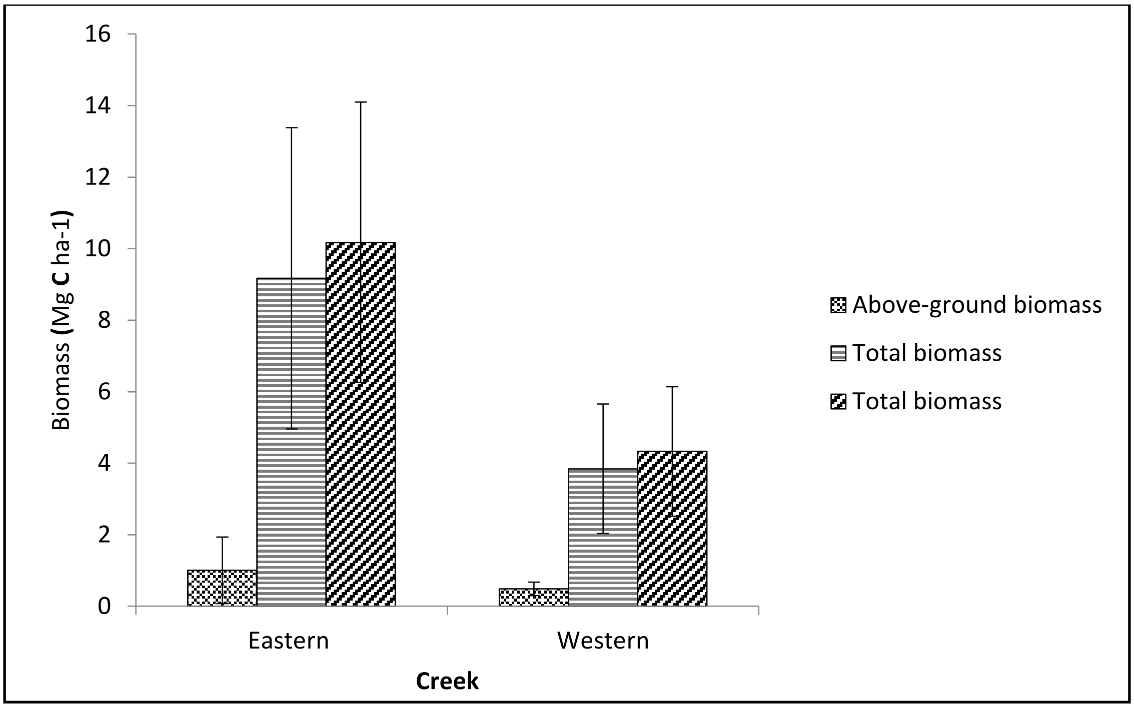

Figure 6: Variation in Biomass between Eastern and Western Creeks (means ± 95% C.I).

4.5 Sediment Organic Carbon (C_{org}) in Seagrass within the Creeks of Gazi Bay

Sedimentary organic carbon (C_{org}) differed between seagrass vegetated areas and the un-vegetated "controls" with higher C_{org} being recorded in vegetated areas (106.65 ± 21.36 Mg C ha^{-1}, range: 67.25 – 160.48 Mg C ha^{-1}), as compared to the un-vegetated areas (47.39 ± 22.53 Mg C ha^{-1}, range: 14.01 – 99.17 Mg C ha^{-1}. Comparing the means revealed a highly significant difference ($t = 12.02$ $p < 0.001$). Among the different species of the Western creek, meadows of mixed species and their adjacent un-vegetated controls recorded the highest values of C_{org} at 111.82 ± 8.40 Mg C ha^{-1}, range: (74.74 – 160.48 Mg C ha^{-1}) and 53.71 ± 7.62 Mg C ha^{-1} range: 18.28 – 99.19 Mg C ha^{-1}, (means ± 95% C.I). *Cymodocea rotundata* had the lowest values of sediment C_{org} in the vegetated areas at 97.57 ± 8.27 Mg C ha^{-1}, range: 67.25 – 133.19 Mg C ha^{-1}, (Figure 7). There was no significant variation in sediment carbon among species of the Western creek in both the vegetated and un-vegetated areas.

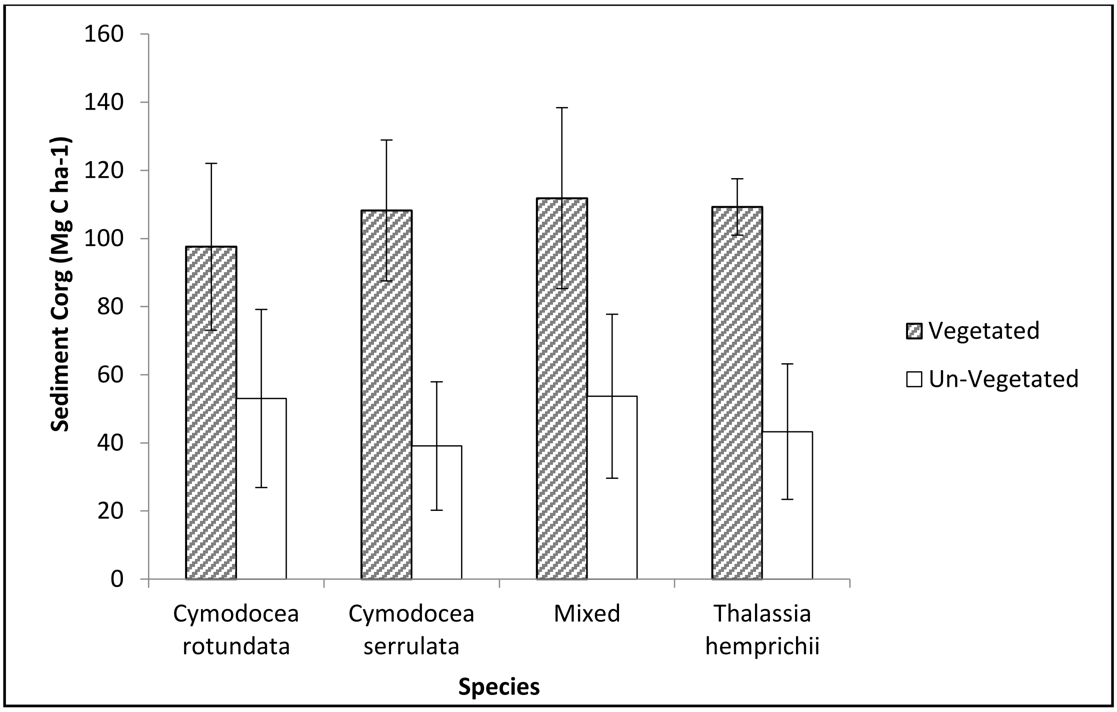

Figure 7: Variation in Sediment Corg between Vegetated and Un-Vegetated Seagrass Areas in the Western Creek (Means ± 95% C.I)

In comparing sediment carbon among the species in both creeks, the means varied with the highest values being recorded in the mixed species of the Eastern creek at 302.45 ± 43.23 Mg C ha^{-1} while the *Cymodocea rotundata* of the Western creek had the lowest means at 97.57 ± 7.74 (mean ± 95% CI) Mg C ha^{-1} (Figure 8). One-way ANOVA revealed a significant difference among the species C_{org}, ($F = 20.28$ $p < 0.0001$). Sediment organic carbon also varied between the species with Eastern creek recording higher values at 258.21 ± 90.12 Mg C ha^{-1}, range: 117.85 – 544.65 Mg C ha^{-1}, while the Western creek recorded 106.66 ± 21.36 Mg C ha^{-1}; range: 67.25 – 160.48 Mg C ha^{-1}. Two sample t-tests revealed a significant difference between the sediment organic carbon in the two creeks, ($t = -10.86$, $df = 44$, $p < 0.001$).

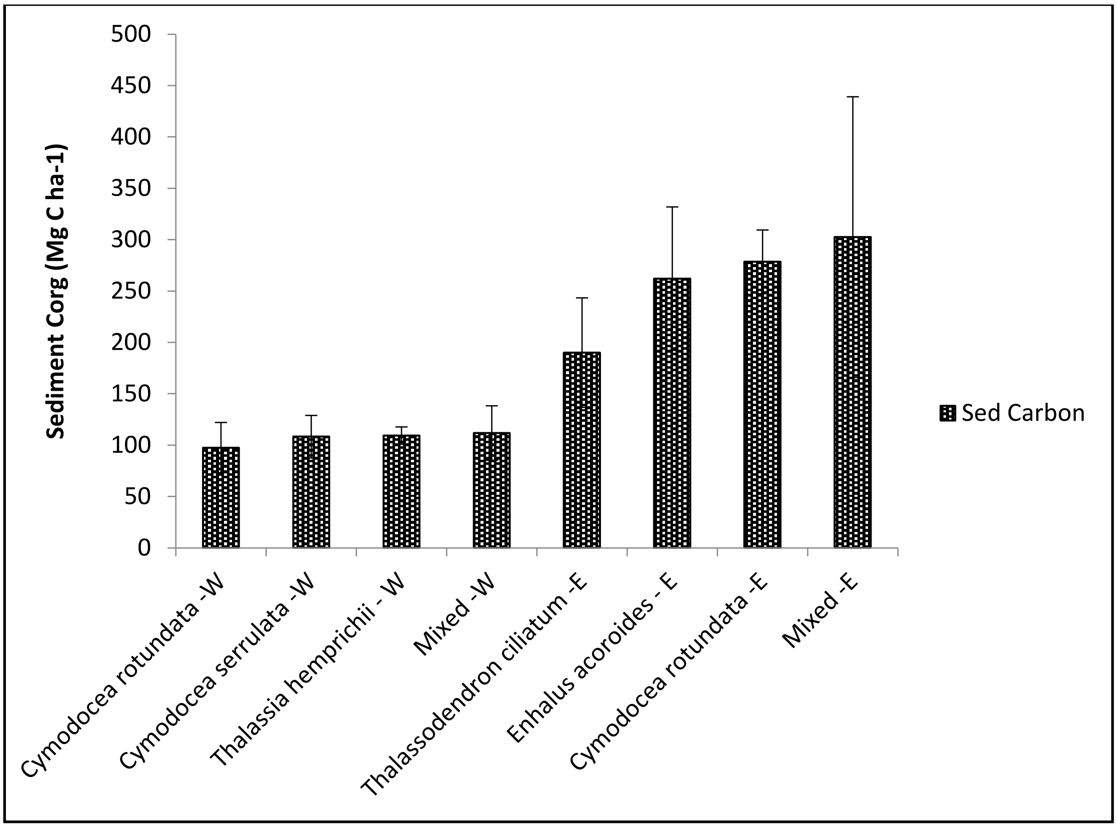

Figure 8: Variation in Sediment C_{org} among Species in Eastern and Western Creeks (Mean ± 95% C.I.) (*Sed Carbon is Sediment organic carbon, E is Eastern, W is Western).*

4.6 Relationship between Above-Ground, Below-Ground Biomass and Sediment Carbon in Eastern and Western Creek

The Eastern creek recorded higher values in all the seagrass biomass and carbon components when compared to the Western creek. In the Western creek, the various carbon pools did not show significant relationships except between below ground biomass and total biomass. In the Eastern creek, there existed significant relationships between above ground and below ground biomass. Similarly, a significant relationship was also observed between below ground biomass and total biomass. Sediment carbon also significantly related with the above-ground biomass in the Eastern creek (Table 7).

Table 7: Correlational Analysis to Test for Relationship between above Ground, below Ground and Sediment Carbon in Eastern and Western Creek

Correlations							
	bg_W	tb_W	sediment_W	ag_E	bg_E	tb_E	Sed_E
ag_W	-0.054	0.05	-0.098	-0.015	0	-0.004	0.084
bg_W		.995*	0.25	-0.105	0.197	0.187	-0.025
tb_W			0.24	-0.106	0.197	0.186	-0.016
Sed_W				0.075	-0.094	-0.084	0.132
ag_E					-.406*	-0.199	-.339*
bg_E						.976*	0.148
tb_E							0.078

*. Correlation is significant at the 0.05 level (2 tailed).

ag – above ground biomass, bg – below ground biomass, tb – total biomass, Sed – Sediment Carbon, W – Western creek, E – Eastern creek

In all the species identified, sediment C_{org} constituted the higher percentage of the total carbon (Figure 9). *C. rotundata* of the Eastern creek recorded sediment C_{org} of 278.41 ± 31.11 (mean ± 95% C.I.) Mg C ha^{-1} and accounted for 96.77% of the total C_{org} for the species per unit area while the lowest proportion was recorded in *E. acoroides* at 95.32% (Figure 9).

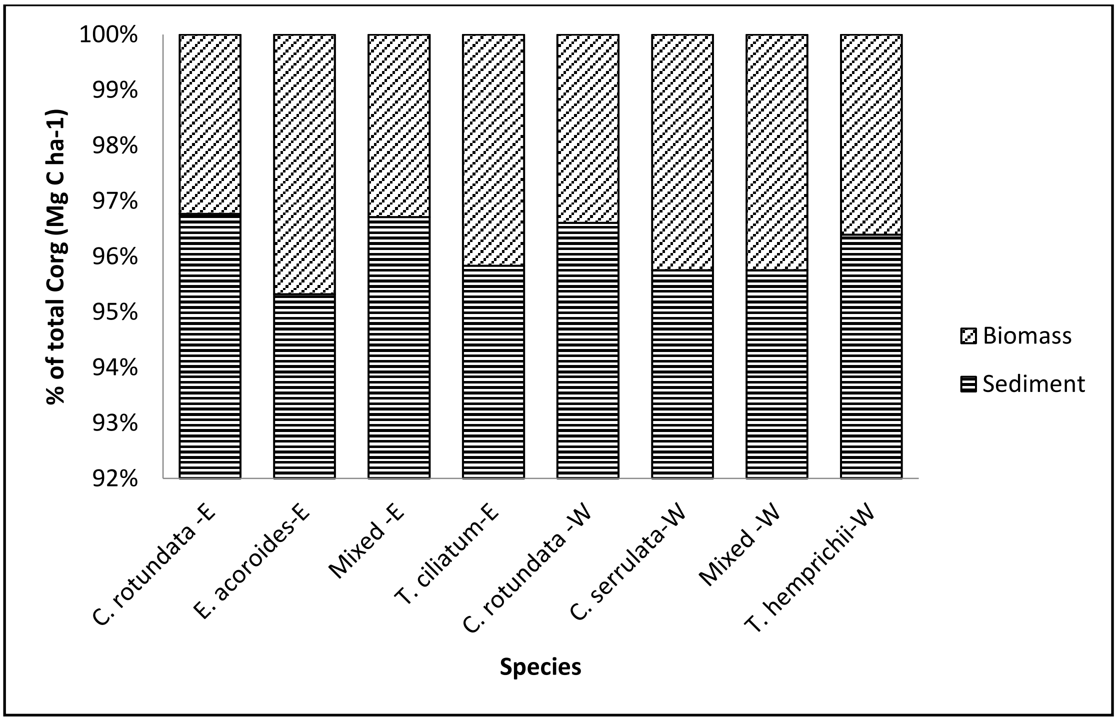

Figure 9: Relative % of the Total C_{org} (± 95% C.I.) for the Biomass and The Sediment C_{org} Associated with the Dominant Species in Eastern and Western Creeks of Gazi Bay.

4.7 Comparison of Seagrass Aboveground Parameters (% Cover, Shoot Height and Shoot Density) between the Creeks and Among the Species

Seagrass above ground parameters including percentage cover, plant shoot height and shoot density varied between the creeks. Eastern creek recorded higher plant height values at 47.51 ± 2.8 cm while the western creek had 25.51 ± 0.7 cm. Comparing the means between the creeks revealed a significant difference ($t = -7.53$ d.f = 44, $p < 0.05$) at 95% confidence interval. Similarly, the variation was observed in shoot density between the creeks with Western creek recording higher values at 643.70 ± 30.39 shoots/ m^2, while Eastern creek recorded 483.00 ± 39.23 shoots/ m^2. Comparing the means revealed a significant difference ($t = 3.24$, d.f = 73, $p < 0.05$) at 95% confidence interval.

In comparing shoot density among the species of the two creeks *Cymodocea rotundata* of the western creek recorded the highest mean shoot density at 780.40 ± 93.79 shoots/m^2 while *Enhalus acoroides* in the eastern creek recorded the lowest value at 204.80 ± 33.25 (means ± 95% C.I) shoots/ m^2. One way ANOVA revealed a significant variation between the species ($F_{(7,79)} = 9.02$, $p < 0.05$) (Table 8).

Table 8: One Way Analysis of Variance to Test for Significant Variation in Mean Shoot Density among Seagrass Species

Source of Variation	*SS*	*Df*	*MS*	*F*	*P-value*
Between Groups	2036441	7	290920.2	9.0225	< .0001
Within Groups	2321557	72	32243.84		
Total	4357998	79			

Shoot height also varied among the species with *Enhalus acoroides* in the eastern creek recording the highest mean value at 62.26 ± 4.21 cm while *Cymodocea serrulata* of the western creek recorded the least mean height at 23.55 ± 0.92 cm (means ± 95% C.I). Testing for variation revealed a significant variation among species ($F_{(7,79)} = 24.94$, $p < 0.05$) (Table 9).

Table 9: Analysis of Variance to Test for Significant Difference in Mean Height among the Seagrass Species

Source of Variation	SS	Df	MS	F	P-value
Between Groups	16282.86	7	2326.12	24.9395	<.0001
Within Groups	6715.478	72	93.27		
Total	22998.34	79			

In comparing the mean percentage cover among the seagrass species under study, *Thalassodendron ciliatum* in the eastern creek recorded the highest mean at 76.50 ± 4.41% while *Enhalus acoroides* of the eastern creek recorded the lowest % cover values at 58.00 ± 5.74%. One way ANOVA did not reveal any significant variation among the species ($F_{(7,79)} = 1.47$, $p = 0.1885$) (Table 10).

Table 10: Analysis of Variance to Test for Significant Variation in % Cover among Seagrass Species

Source of Variation	SS	df	MS	F	P-value
Between Groups	2096.6	7	299.5143	1.4788	0.1885
Within Groups	14582.6	72	202.5361		
Total	16679.2	79			

4.8 Relationship between Biomass and Above Ground Parameters

In the current study, a significant relationship was found to exist between the total biomass, shoot density and shoot height. There was also a significant relationship between above ground biomass, percentage cover and shoot height. However, there was no significant relationship between percentage cover and shoot height (Table 11).

Table 11: Correlational Analysis to Test for Relationship between Biomass and Above Ground Parameters, (% Cover, Shoot Density and Height)

	Cover	Correlations Density	Height	Ag	Bg	Tb
Cover		.365*	0.05	.388*	-0.05	0.016
Density			-.503*	0.015	-.345*	-.339*
Height				.445*	.469*	.540*
Ag					-0.031	0.139
Bg						.986*

*. Correlation is significant at p < 0.05 level (2-tailed).

Ag – above ground biomass, Bg – below ground biomass, Tb – total biomass

CHAPTER FIVE

DISCUSSION

5.1 Assessment of Physical-Chemical Parameters

The current study compared seagrass habitats in the Eastern and Western Creeks of Gazi Bay in terms of their species composition and distribution, and carbon stocks in the above ground, below ground and sediment components. The results were compared against abiotic gradients. The different abiotic factors tested included turbidity, water depth, water pH, water temperature and water salinity to understand their effects on the distribution and stocks of carbon in the creeks.

Turbidity was higher in the Western creek than in the Eastern creek. This difference may be attributed to the seasonal inflow of River Kidogoweni in the Western creek exposing this creek to higher hydrodynamic forces. Higher levels of sedimentation and erosion, are therefore, experienced in this area leading to water murkiness (van Keulen & Borowitzka, 2003; Dimowo, 2013). Temperature values in the two creeks varied significantly with the Western creek recording higher temperatures than in the Eastern creek. These values are within range when compared to the values, (33.4 - 35.1˚C) obtained by Sreenivasulu *et al.,* (2015), in Tupilipalem, Southern India and the range of 8 ˚C and 30 ˚C adapted as normal range within the tropics (23.43684˚N and 23.43684˚S) (Alabaster & Lloyd, 1980). The protected mangroves that fringe the eastern creek form dense canopies thereby lowering irradiance in some parts of this creek. The higher water velocity in Eastern creek may also attribute to the lower temperatures in this creek. On the other hand, western creek is more open and is fringed by more degraded mangroves (Huxham *et al.,* 2018).

Eastern creek recorded higher values in depth than the western creek. This may be attributed to the sand bank and instability in the western creek leading to frequent changes in hydrodynamic conditions (Koch *et al.,* 2006). More accumulation of sand and silt during water in-flow reduces water depth. On the other hand, the eastern creek have stable sediment condition with higher water current hence may explain for the higher values in depth. Higher salinity values were also recorded in the eastern creek than in the western creek. The eastern creek lacks a fresh water inflow hence explains for the higher salinity. Similarly, increased anthropogenic inputs and contaminants in the western creek due to the proximity to human activities may attribute to the lower salinity levels in this creek when compared to the remarkably stable state of the eastern creek (Smith & Elliot, 2016).

5.2 Seagrass Distribution and Abundance in the Creeks of Gazi Bay

Nine seagrass species out of the twelve recorded in Gazi bay were encountered in the two creeks during the study. *Halodule wrightii*, and *Halophila minor* that were not found in the creeks are pioneer species that have been found small patches majorly in degraded areas in other parts of Kenya. Additionally, despite earlier studies by Ochieng' and Erftemeijer (2003), indicating the presence of *Zostera capensis* in the bay, it was not encountered in the creeks. The species found to be dominant within the creeks in this study have been found to be dominant in other parts along the Kenyan coastline including Diani-Chale lagoon, Kiunga Marine Reserve and Mida creek and along the East African Coastline (Gullstrom *et al.*, 2002; Ochieng & Erftemeijer, 2003; McMahon and Waycott, 2006).

The Eastern creek had higher seagrass species diversity and abundance than the Western creek. The dominance of *Thalassodendron ciliatum* in the Eastern creek can be attributed to the stable state of this creek. On the other hand, the pioneer and short lived *Halophila ovalis* and *Halophila stipulacea* were recorded only in Western creek. The availability of an in-flow channel (River Kidogoweni) in the Western creek and the sandy substrates that keep shifting make the area highly disturbed hence explain for the existence of *H. ovalis* and *H. stipulacea* and the higher abundance of *C. rotundata* that usually occupy disturbed areas (Noel *et al.,* 2012). It also explains for the absence of *T. ciliatum* that is usually intolerant of any freshwater input and occupies hard and rocky substrates (Waycott *et al.,* 2004).

The shallow depths observed in the Western creek may make this area to be categorized as shallow intertidal meadow hence get highly affected by tide currents and wave action affecting essential ecological and physiological processes of seagrass in carbon capture (Silva *et al.,* 2015). When seagrass plants are exposed to air for a long time, the amount of water in seagrass tissues leads to desiccation constraining accumulation of CO_2 and inhibiting carbon gains (Silver *et al.,* 2005; Clavier *et al.,* 2011). Similarly, other physical parameters such as turbidity, assessed during the study and found to vary between the creeks may explain for the variation in abundance and diversity. Turbidity was higher in the Western creek. This may reduce light attenuation in seagrass meadows thereby affecting primary productivity, and consequently, growth and shoot density (Laver *et al.*, 2009). This may explain for the higher abundance in eastern than in western creek.

5.3 Seagrass Above-ground and Below-ground Biomass in the Creeks of Gazi Bay

Above ground biomass varied among the species and between the creeks in the current study. These means were within range when compared to a study by Lymo *et al.*, (2008), that reported a mean range of 175 – 609 g DW m^{-2}. However, our values are noticeably higher compared to the means reported by Gullstrom *et al.*, (2006) with a mean range of 62 – 105 g DW m^{-2}. *Thalassodendron. ciliatum* recorded the highest above-ground biomass among the species with *Cymodocea. rotundata* recording the least. This may be attributed to the higher canopy complexity encountered in species such as *T. ciliatum and Posidonia spp* leading to higher biomass. Additionally, large-leaved species usually have the ability to withstand strong waves and current and grow in stable substrates hence having higher shoot biomass (Lavery and Vanderklift, 2002; Peratra *et al.*, 2008). On the other hand, small colonizer species such as *C. rotundata*, *Halodule spp* and *Halophila spp* are small leaved and have higher turnover rates hence lower accumulation of biomass. They are also highly affected by erosion and wave action hence reduced above ground biomass (Lavery *et al.,* 2013).

Mean total biomass of seagrass in the creeks of Gazi bay were within range when compared to the other published data although the present values tended towards upper limits. The current values were slightly lower than biomass values of *Posidonia oceanica* of the Mediterranean region which recorded 7.29 ± 1.52 Mg C ha^{-1}. However, they are above the values recorded by a study in the open waters of Gazi bay at 5.9 ± 0.9 Mg C ha^{-1} and the global mean of 2.51 ± 0.49 Mg C ha^{-1} (Fourqurean *et al.*, 2012; Githaiga *et al.*, 2017). The Eastern creek had significantly higher biomass values of than Western creek. The significant variation in biomass between the creeks can be attributed to the difference in the biophysical parameters leading to a difference in physiological, phenotypic and morphological growth patterns of seagrass.

The estuarine environment of the Western creek due to the effect of River Kidogoweni is likely to subject this seagrass habitat to periodic freshwater inundation leading to low levels of irradiance and burials caused by sediment resuspension, run-off turbidity and nutrient loading (McDonald *et al*., 2016). The high turbidity experienced in the western creek maybe decreasing growth rate and shoot density of seagrass plants (Ruiz and Romero 2003; Lavery *et al*., 2009), in this area hence affecting the capacity of the plants to trap both autochthonous and allocthonous carbon from the water column (Peraltra *et al.*, 2008). Additionally, the proximity of this creek to the human settlement increases anthropogenic influence on seagrass through boat dredging, seine net fishing and ecotourism activities (nearness to the mangrove boardwalk) hence affecting the growth pattern. This was observed during the study and may explain for the lower biomass values recorded in the Western creek.

From the study, the different species recorded varying total biomass. As the case in other prior studies, seagrass species usually encounter high variations in terms of their biomass distribution, size and dynamic properties such as turnover rates, primary productivity and lifespan (Duarte & Chiscano, 1999). Small species such as *Halodule* spp., *Halophila* spp. and *C. rotundata* usually have high turnover and growth rate since they are typically colonizers hence accumulates low amount of biomass than other larger species such as *Thalassia* spp. and *Posidonia* spp. that are persistent and long lived (Kilminster *et al.,* 2015; Serrano *et al.,* 2016). *Thalassodendron ciliatum* recorded the highest above-ground biomass due to its numerous stems and the ability to withstand wave action when compared to the other species. *Enhalus acoroides* that recorded the highest below-ground biomass on the other hand have big rhizomes; big roots and large fronds hence have the ability of accumulating more biomass in the below-ground component during their growth and development (Waycott *et al.,* 2004).

In all the species, higher biomass values were recorded in below-ground biomass than in above-ground biomass. This may be attributed to the characteristic of seagrass to survive in harsh conditions by reducing desiccation exposure at low tides, minimizing anthropogenic disturbances and to enhance stability during the high tides. The high turnover rates in above-ground biomass occurs due to mechanical removal by waves, herbivory pressures and unsustainable fishing activities such as boat dredging and net seining. This affects the shoot density hence contributing to the lower biomass levels (Lymo *et al.*, 2008).

In a study by Kaldy & Dunton (2000), rhizome as a component of below ground biomass in *Thalassia testudinum* accounted for 80 to 90% of the total biomass and 25 – 35% of the total seagrass plant production. The seagrass above ground biomass are considered as short-term carbon sink since they have a higher labile chemical composition than below ground biomass and are exposed to aerobic conditions. They therefore have a lower contribution to the total organic carbon deposits when compared to below ground biomass (Mateo *et al*., 2006; Fourqurean *et al.*, 2012).

5.4 Sediment Organic Carbon in Seagrass Meadows within the Creeks of Gazi Bay

The Eastern creek recorded the highest C_{org} values for the top one meter when compared to the western creek and the values obtained by Githaiga *et al.,* (2017) in the open waters of the bay (236 ± 24 Mg C ha^{-1}). Similarly, the average C_{org} values obtained for the two creeks are above the mean derived from the global seagrass data set at 166 Mg C ha^{-1} although they are within the global range of 115.5 - 829.2 Mg C ha^{-1}, (Fourqurean *et al.,* 2012). The biophysical characteristics and physical-chemical parameters that vary between eastern and western creeks may explain for the variation in sediment carbon observed (Mazarrasa *et al.,* 2018).

Organic carbon storage in seagrass sediment is both autochthonous and allocthonous with an estimated 50% originating from other ecosystems (Kennedy *et al.,* 2010). In the Western creek, organic carbon from the terrestrial ecosystems reaches the blue-carbon ecosystems through the river in-flow (Bouillon *et al.,* 2007). However, much of this carbon is intercepted in the mangroves before reaching the seagrass meadows.

The difference in the structure of mangrove ecosystems adjacent to the two creeks may, therefore, partly explain for the variation in carbon since the mangrove structure and functionality determine the amount of carbon exported to the seagrass meadows. Mangroves in the western creek have been found to be more than in eastern creek as indicated by lower complexity indices (Kairo *et al.,* 2008)

Sediment C_{org} production and accumulation in seagrass meadows are linked to the landscape biomass and nutrient limitation patterns. In the highly productive meadows, the complex above-ground biomass enhances allochthonous carbon trapping and minimizes erosion of the deposited organic carbon (Nordlund *et al.,* 2018). Additionally, rhizomes, roots, necromass and dissolved organic carbon (DOC) contribute substantially to the primary production in seagrass meadows hence there tend to be more C_{org} in complex meadows with high shoot density and below-ground biomass than in sparse meadows (Kaldy *et al*., 2006; Armitage & Fourqurean, 2016).

The lower C_{org} values in the Western creek can therefore similarly be attributed to the less diversity; abundance and low shoot density of the seagrass plants in this creek which leads to lower DOC supply. This in turn affects % OC through reduced decomposition of the refractory organic compounds (Mazarrasa *et al.,* 2018; Giathaiga *et al.,* 2019). On the other hand, the meadows of the Eastern creek are majorly formed by morphologically large species such as *Enhalus acoroides* and *Thalassidendron ciliatum*. These species are more efficient in reduction of erosion and export, enhancement of particle sedimentation from the water column and allocthonous carbon accumulation (Mellors *et al.,* 2002). They also tend to establish more permanent and stable beds than the ephemeral and small species due to their higher resistant ability to the hydrodynamic energy hence high carbon stocking (Ondiviela *et al.,* 2014; Huxham *et al.,* 2018).

Sediment C_{org} constituted the carbon pool with the highest carbon percentage when compared to the other pools. Carbon in the soil compartments are formed from the allocthonous C_{org}, seagrass detritus and the refractory below-ground biomass embedded in the soil matrix. The anoxic properties of the sediment in the blue carbon ecosystems make the C_{org} deposits remain for millennia thereby constituting the long-term carbon pools (Kennedy *et al.,* 2010; Mazarrasa *et al.,* 2018). Additionally, the above ground biomass is exposed to herbivory and aerobic conditions hence more labile when compared to the below-ground component. This explains the above ground component as a short-term sink hence low contribution to the total seagrass carbon.

The current study also compared sediment organic carbon between vegetated and un-vegetated seagrass areas of the Western creek with findings revealing higher carbon values in the vegetated areas (69%), than un-vegetated controls (31%). Restoration activities and naturally growing seagrass beds store up to three times more carbon than their adjacent un-vegetated areas (Serrano *et al.,* 2016). Seagrass beds have low shear stress hence augment the retention and settlement of small organic particles. Deposition of sedimentary C_{org} in seagrass meadows are dependent on three major processes including seagrass biomass accumulation and meadow productivity; burial efficiency of C_{org} in seagrass sediment and allocthonous sediment C_{org} input into the compartment (Miyajima *et al.,* 2017; Mazarrasa *et al.,* 2018). Due to high burial efficiency and seagrass biomass accumulation in the seagrass vegetated areas, these areas have more carbon storage capacity than the adjacent un-vegetated areas.

The seagrass above ground parameters including shoot density, height and percentage cover differed significantly between creeks in our current study. Eastern creek recorded higher heights than the western creek. This may be attributed to the favorable conditions such as nutrient availability from the surrounding protected mangroves thereby enhancing growth within the seagrass meadows. Additionally, the seagrass species composition dominating the two creeks differs hence may account for the difference in shoot heights. The eastern creek is much dominated by *Thalassodendron ciliatum* and *Enhalus acoroides* that usually grow in stable mud or sand substrate hence form complex canopies when compared to other pioneer species such as *H. ovalis* and *C. rotundata* dominating the western creek (Waycott *et al.,* 2004).

On the other hand, western creek recorded higher shoot counts than the eastern creek. This creek is dominated by small species such as *Halodule uninervis, Cymodocea* *rotundata* and *Halophila stipulacea.* Despite having a higher turnover rate due to the unstable substrate, they have higher shoot counts (Serrano *et al.*, 2016), than the other long lived and larger species including *Thalassia spp.* and *Enhalus spp* (Kilminster *et al.,* 2015; Mazarrasa *et al.,* 2018), which are found in the eastern creek.

In general, the values obtained for total biomass and sediment C_{org} for the current study in the two creeks were within range when compared to the values obtained by Githaiga *et al.,* (2017) and other world studies. Variations were also encountered between the creeks due to the difference in biophysical conditions such as temperature, turbidity, salinity, seagrass shoot height and percentage cover that were found to vary between the creeks. These conditions influence seagrass growth and ability to capture and store carbon and were found to be more favorable in the eastern creek hence the higher biomass and C_{org} values in the eastern creek.

CHAPTER SIX

SUMMARY, CONCLUSION AND RECOMMENDATIONS

6.1 Summary of Key Findings

Seagrass diversity and abundance was higher in the Eastern creek at ($H = 1.71$) than in the Western creek ($H = 1.67$). There was also a significant difference ($p < 0.001$) in percentage abundance with the Eastern creek recording 69.17%, and the western creek 56.43%. Large leaved species such as *Thalassodendron ciliatum* and *Enhalus acoroides* were dominant in Eastern creek while small leaved and pioneering species such as *Cymodocea rotundata* and *Halodule uninervis* were dominant to the western creek. This implies that eastern creek is more pristine while the western creek is degraded.

In determining above ground and below ground biomass in the two creeks, the study established a significant variation in carbon biomass between the two creeks and among the species under study. Eastern creek recorded higher total biomass at than the western creek. The average biomass for the two creeks was 7.29 ± 4.23 Mg C ha^{-1}. These values of the current study are within range and higher than the values of the open water of Gazi bay and the global mean of 2.51 ± 0.49 Mg C ha^{-1}.

Sedimentary organic carbon differed significantly between vegetated and un-vegetated areas during the study. The vegetated areas for the sampled species recorded C_{org} of 111.82 ± 8.40 Mg C ha^{-1} while the adjacent un-vegetated perches recorded 53.71 ± 7.62 Mg C ha^{-1} in the western creek. Similarly, a significant variation was observed among the dominant species under study. Sediment carbon also differed significantly between the creeks with the Eastern creek recording higher values than western creek. The average C_{org} for the two creeks was 183.40 ± 100.49 Mg C ha^{-1} which are within the global range of 115.5 - 829.2 Mg C ha^{-1} although tends towards the lower limits.

In comparing seagrass biomass between the different carbon pools, sediment C_{org} constituted the pool with highest percentage. Significant relationship existed between below ground and total biomass in the two creeks and between above ground biomass and sediment C_{org}. This implies that the sediment carbon stores the highest proportion of carbon when compared to other seagrass carbon pools.

6.2 Conclusion

The current study focused on the narrow stretch of the two meadows of Gazi Bay covering 1.2 km^2. It compared the carbon stocks between the Eastern and Western creeks that have varying biophysical features. The total carbon stocks for the two creeks are 21,118.8 Mg C. This value contributes to the seagrass carbon budget of the whole bay and adds to the total C_{org} of seagrass along the Kenyan coast and the African coastline. In the Eastern creek (0.5 km^2), protection of seagrass would ensure approximately 13,419.5 Mg of carbon secured, while 7,769.3 Mg of carbon secured in the western creek (0.7 km^2) from being lost. Based on tier 1 IPCC emission factor of 7.9 tonnes of C ha^{-1}, values for organic soils for wetlands, protecting seagrass in these two creeks will prevent emission of 2,682.13 Mg of CO_2 equivalent yr-^{1} (IPCC, 2014).

From the study, it is established that habitat heterogeneity between Eastern and Western creeks of Gazi Bay cause variation in the distribution, abundance and carbon stocks in seagrass meadows of these two areas with more abundance in the Eastern creek. The sediment organic carbon also constitutes the C pool with the highest proportion in the two creeks. There is a significant difference in C_{org} between vegetated and un-vegetated areas, hence justifying the need for conserving these vital ecosystems as carbon sinks.

Comparing the two creeks shows that Eastern creek has a higher potential of sequestering more carbon hence is recommended for establishing the permanent sampling plots in the process of bundling seagrass carbon into the ongoing Mikoko Pamoja Mangrove Carbon Offset Project. Since this creek boarders the project (protected) area, upscaling the project in this area will ensure accessibility and easy monitoring activities thereby enhancing community participation. Seagrass carbon as a tool for accessing carbon financing will improve conservation and restoration strategies of the seagrass meadows, contribute to the country's GHG emission reduction while also enhancing livelihood improvement among coastal communities.

Enhancing conservation initiatives in seagrass will help in climate change initiatives and facilitate the achievement of the blue economy agenda. The findings also contribute to the available literature on the meadow dynamics and its relation to seagrass carbon stocks.

6.3 Recommendations

The current study prompts the following recommendations:

i. Habitat dynamics and heterogeneity should be considered by coastal managers and communities in the process of implementing seagrass restoration and conservation activities.

ii. Assessment of seagrass species distribution, abundance and carbon stocks should be done in the other parts along the Kenyan coastline to enhance management and implementation of the Coral Reefs and Seagrass Ecosystems Management Strategy.

iii. Seagrass conservation initiatives should be adopted such as bundling of seagrass ecosystem into Mikoko Pamoja Community Project. This will enhance community participation in conservation.

iv. Since degradation increases emission potential, conservation and restoration activities should be encouraged in degraded areas through measures such as avoided boat dredging and use of seine nets that destroy seagrass.

v. Carbon credit project in seagrass can be established in the Eastern creek due to its higher carbon stocks and stable habitat condition when compared to Western creek.

6.4 Suggestion for Further Study

The current study was limited to assessing organic carbon stocks within the creeks of Gazi bay. Further studies can therefore consider inorganic carbon storage within the bay. The study also considered the dominant species only. Future research, can therefore consider phenology of the individual species in Gazi Bay, and their contribution to carbon storage. Assessing carbon stocks in the subtidal parts of the bay can be considered to help in understanding carbon storage in the entire bay and its distribution in the three tidal gradients. Additionally, carbon burial rates and land based activities impacting on seagrass ecosystems can be considered.

REFERENCE

Aboud, S. A., & Kannah, J. F. (2017). Abundance, Distribution and Diversity of Seagrass Species in Lagoonal Reefs on the Kenyan Coast. *American Scientific Research Journal for Engineering, Technology, and Sciences (ASRJETS)*, *37*(1), 52-67.

Agawin, N. S., & Duarte, C. M. (2002). Evidence of Direct Particle Trapping by a Tropical Seagrass Meadow. *Estuaries and Coasts*, *25*(6), 1205-1209.

Alcoverro, T., & Mariani, S. (2004). Patterns of Fish and Sea Urchin Grazing on Tropical Indo-Pacific Seagrass Beds. *Ecography*, *27*(3), 361-365.

Armitage, A. R., & Fourqurean, J. W. (2016). Carbon Storage in Seagrass Soils: Long-term Nutrient History Exceeds the Effects of Near-term Nutrient Enrichment.

Atwood, T. B., Connolly, R. M., Ritchie, E. G., Lovelock, C. E., Heithaus, M. R., Hays, G. C., & Macreadie, P. I. (2015). Predators help protect carbon stocks in blue carbon ecosystems. *Nature Climate Change*, *5*(12), 1038.

Bandeira, S. O., & Björk, M. (2001). Seagrass Research in the Eastern Africa Region: Emphasis on Diversity, Ecology and Ecophysiology. *South African Journal of Botany*, *67*(3), 420-425.

Barbier, E. B., Hacker, S. D., Kennedy, C., Koch, E. W., Stier, A. C., & Silliman, B. R. (2011). The Value of Estuarine and Coastal Ecosystem Services. *Ecological Monographs*, *81*(2), 169-193.

Beck, M. W., Odaya, M., Bachant, J. J., Bergan, J., Keller, B., Martin, R., & Ramseur, G. (2000). Identification of Priority Sites for Conservation in the Northern Gulf of Mexico: an Eco-regional Plan. *The Nature Conservancy, Arlington, VA*.

Beck, M.W., Heck, K.L., Able, K.W., Childers, D.L., Eggleston, D.B., & Gillanders, B.M. (2001). The Identification, Conservation, and Management of Estuarine and Marine Nurseries for Fish and Invertebrates. A Better Understanding of the Habitats that Serve as Nurseries for Marine Species. *Bioscience*, 51.

Bos, A. R., Bouma, T. J., de Kort, G. L., & van Katwijk, M. M. (2007). Ecosystem engineering by annual intertidal seagrass beds: sediment accretion and modification. *Estuarine, Coastal and Shelf Science*, *74*(1-2), 344-348.

Bouillon, S., & Connolly, R. M. (2009). Carbon Exchange Among Tropical Coastal Ecosystems. In *Ecological Connectivity among Tropical Coastal Ecosystems* (Pp. 45-70). Springer Netherlands.

Bouillon, S., Dehairs, F., Velimirov, B., Abril, G., & Borges, A. V. (2007). Dynamics of Organic and Inorganic Carbon Across Contiguous Mangrove and Seagrass Systems (Gazi Bay, Kenya). *Journal of Geophysical Research: Biogeosciences*, *112*(G2).

Bradford, M. A., Veen, G. C., Bonis, A., Bradford, E. M., Classen, A. T., Cornelissen, J. H. C., & Manrubia-Freixa, M. (2017). A test of the hierarchical model of litter decomposition. *Nature ecology & evolution*, *1*(12), 1836.

Burden A, Garbutt R, Evans C, Jones D, Cooper D (2013) Carbon Sequestration and Biogeochemical Cycling in a Saltmarsh Subject to Coastal Managed Realignment. Estuarine, Coastal and Shelf Science 120: 12-20

Burdige, D. J. (2007). Preservation of Organic Matter in Marine Sediments: Controls, Mechanisms, and an Imbalance in Sediment Organic Carbon Budgets?. *Chemical Reviews*, *107*(2), 467-485.

Cabaço, S., Santos, R., & Duarte, C. M. (2008). The Impact of Sediment Burial and Erosion on Seagrasses: A Review. *Estuarine, Coastal and Shelf Science*, *79*(3), 354-366.

Cardoso, P. G., Pardal, M. A., Lillebø, A. I., Ferreira, S. M., Raffaelli, D., & Marques, J. C. (2004). Dynamic Changes in Seagrass Assemblages under Eutrophication and Implications for Recovery. *Journal of Experimental Marine Biology and Ecology*, *302*(2), 233-248.

Ceccherelli, G., & Cinelli, F. (1997). Short-Term Effects of Nutrient Enrichment of the Sediment and Interactions Between the Seagrass Cymodocea nodosa and the introduced green alga Caulerpa taxifolia in a Mediterranean bay. *Journal of Experimental Marine Biology and Ecology*, *217*(2), 165-177.

Clausen, K. K., Krause-Jensen, D., Olesen, B., & Marbà, N. (2014). Seasonality of Eelgrass Biomass across Gradients in Temperature and Latitude. *Marine Ecology Progress Series*, *506*, 71-85.

Clavier, J., Chauvaud, L., Carlier, A., Amice, E., Van der Geest, M., Labrosse, P., & Hily, C. (2011). Aerial and underwater carbon metabolism of a Zostera noltii seagrass bed in the Banc d’Arguin, Mauritania. *Aquatic Botany*, *95*(1), 24-30.

Collier, C. J., Waycott, M., & Ospina, A. G. (2012). Responses of four Indo-West Pacific seagrass species to shading. *Marine Pollution Bulletin*, *65*(4-9), 342-354.

Costanza, R., d'Arge, R., De Groot, R., Farber, S., Grasso, M., Hannon, B., & Raskin, R. G. (1997). The Value of the World's Ecosystem Services and Natural Capital. *Nature*, *387*(6630), 253.

Coppejans, E., Beeckman, H., & De Wit, M. (1992). The Seagrass and Associated Macroalgal Vegetation of Gazi Bay (Kenya). *Hydrobiologia*, *247*(1-3), 59-75.

Cullen-Unsworth, L., & Unsworth, R. (2013). Seagrass Meadows, Ecosystem Services, and Sustainability. *Environment: Science and Policy for Sustainable Development*, *55*(3), 14-28.

Cutter, S. L. (1996). Vulnerability to Environmental Hazards. *Progress in Human Geography*, *20*(4), 529-539.

Dahl, M., Deyanova, D., Lyimo, L. D., Näslund, J., Samuelsson, G., Mtolera, M.S, Björk, M., Gullström, M. (2016). Effects of Shading and Simulated Grazing on Carbon Sequestration in a Tropical Seagrass Meadow. Journal of Ecology 10.1111/1365-2745.12564

De Leeuw, J. W., Versteegh, G. J., & Van Bergen, P. F. (2005). Biomacromolecules of Algae and Plants and their Fossil Analogues in *Plants and Climate Change* (Pp. 209-233). Springer Netherlands.

Dewsbury, B. M., Bhat, M., & Fourqurean, J. W. (2016). A Review of Seagrass Economic Valuations: Gaps and Progress in Valuation Approaches. *Ecosystem Services*, *18*, 68-77.

Di Carlo, G., & Kenworthy, W. J. (2008). Evaluation of Aboveground and Belowground Biomass Recovery in Physically Disturbed Seagrass Beds. *Oecologia*, *158*(2), 285-298.

Dimowo, B. O. (2013). The Physico-chemical Parameters of River Ogun (Abeokuta, Ogun State, Southwestern Nigeria) in Comparison with National and International Standards. *International Journal of Aquaculture, 3*.

Duarte, C. M. (2002). The Future of Seagrass Meadows. *Environmental Conservation*, *29*(2), 192-206.

Duarte, C. M., & Agustí, S. (1998). The CO_2 Balance of Unproductive Aquatic Ecosystems. *Science*, *281*(5374), 234-236.

Duarte, C. M., & Chiscano, C. L. (1999). Seagrass biomass and production: a reassessment. *Aquatic botany*, *65*(1-4), 159-174.

Duarte, C. M., & Prairie, Y. T. (2005). Prevalence of Heterotrophy and Atmospheric CO2 Emissions from Aquatic Ecosystems. *Ecosystems*, *8*(7), 862-870.

Duarte, C. M., & Sand-Jensen, K. (1990). Seagrass Colonization: Patch Formation and Patch Growth in *Cymodocea nodosa*. *Marine Ecology Progress Series*, 193-200.

Duarte, C. M., Sintes, T., & Marbà, N. (2013). Assessing the CO_2 Capture Potential of Seagrass Restoration Projects. *Journal of Applied Ecology*, *50*(6), 1341-1349.

Duffy, J. E. (2006). Biodiversity and the Functioning of Seagrass Ecosystems. *Marine Ecology Progress Series*, *311*, 233-250.

Emmer, I., Needelman, B., Emmett-Mattox, S., Crooks, S., Megonigal, P., Myers, D. (2015). *Methodology for Tidal Wetland and Seagrass Restoration*. Verified Carbon Standard. VM0033 Version 1.0. 2015

Fourqurean, J. W., Duarte, C. M., Kennedy, H., Marba, N., Holmer, M., Mateo, M. A., & Serrano, O. (2012). Seagrass Ecosystems as a Globally Significant Carbon Stock. *Nature Geoscience*, *5*(7), 505.

Gedan, K. B., Silliman, B. R., & Bertness, M. D. (2009). Centuries of human-driven change in salt marsh ecosystems. *Annual review of marine science*, *1*, 117-141.

Githaiga, M. N., Frouws, A. M., Kairo, J. G., & Huxham, M. (2019) Seagrass Removal Leads to Rapid Changes in Fauna and Loss of Carbon. Front. Ecol. Evol. 7:62.doi: 10.3389/fevo.2019.00062

Githaiga, M. N., Gilpin, L., Kairo, J. G., & Huxham, M. (2016). Biomass and Productivity of Seagrasses in Africa. *Botanica Marina*, *59*(2-3), 173-186.

Githaiga, M. N., Kairo, J. G., Gilpin, L., & Huxham, M. (2017). Carbon Storage in the Seagrass Meadows of Gazi Bay, Kenya. *Plos One*, *12*(5), E0177001.

Green, E. P., & Short, F. T. (2003). World Atlas of Seagrasses. Prepared by the UNEP World Conservation Monitoring Centre. *University of California, Press Berkeley, USA.*

Gullström, M., Lyimo, L. D., Dahl, M., Samuelsson, G. S., Eggertsen, M., Anderberg, E., & Nordlund, L. M. (2017). Blue Carbon Storage in Tropical Seagrass Meadows Relates to Carbonate Stock Dynamics, Plant–Sediment Processes, and Landscape Context: Insights from the Western Indian Ocean. *Ecosystems*, 1-16

Harcourt, W. D., Briers, R. A., & Huxham, M. (2018). The Thin (ning) Green Line? Investigating Changes in Kenya's Seagrass Coverage. *Biology letters*, *14*(11), 20180227.

Hemminga, M. A. (1998). The Root/Rhizome System of Seagrasses: An Asset and a Burden. *Journal of Sea Research*, *39*(3), 183-196.

Hemminga, M. A., & Duarte, C. M. (2000). *Seagrass Ecology*. Cambridge University Press.

Hemminga, M. A., Slim, F. J., Kazungu, J., Ganssen, G. M., Nieuwenhuize, J., & Kruyt, N. M. (1994). Carbon Outwelling from a Mangrove Forest with Adjacent Seagrass Beds and Coral Reefs (Gazi Bay, Kenya). *Marine Ecology Progress Series*, 291-301.

Henderson, P. A., & Southwood, T. R. E. (2016). *Ecological Methods*. John Wiley & Sons, Hendriks, I. E., Olsen, Y. S., Ramajo, L., Basso, L., Steckbauer, A., Moore, T. S., & Duarte, C. M. (2014). Photosynthetic Activity Buffers Ocean Acidification in Seagrass Meadows. *Biogeosciences*, *11*(2), 333.

Howard, J. L., Creed, J. C., Aguiar, M. V., & Fouqurean, J. W. (2018). CO2 Released by Carbonate Sediment Production in some Coastal areas may Offset the Benefits of Seagrass "Blue Carbon" Storage. *Limnology and Oceanography*, *63*(1), 160-172.

Howard, J., Hoyt, S., Isensee, K., Telszewski, M., & Pidgeon, E. (2014). Coastal Blue Carbon: Methods for Assessing Carbon Stocks and Emissions Factors in Mangroves, Tidal Salt Marshes, and Seagrasses. Conservation International, Intergovernmental Oceanographic Commission of UNESCO, International Union for Conservation of Nature. In: Conservation International. Arlington, Virginia, USA.; 2014. p. 99 – 107.

Huxham, M., Whitlock, D., Githaiga, M., & Dencer-Brown, A. (2018). Carbon in the Coastal Seascape: How Interactions between Mangrove Forests, Seagrass Meadows and Tidal Marshes Influence Carbon Storage. Curr. For. Rep. 4,101–110. doi: 10.1007/s40725-018-0077-4

IPCC (2013). Summary for Policymakers. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]. *Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA*, 33-109.

IPCC (2014). 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventory, Wetlands. Vol. 2. 2013.

Jackson, E. L., Rowden, A. A., Attrill, M. J., Bossey, S. J., & Jones, M. B. (2001). The Importance of Seagrass Beds as a Habitat for Fishery Species. *Oceanography and Marine Biology*, *39*, 269-304.

Jamaludin, M. R. (2015). Carbon Storage and Preservation in Seagrass Meadows. Retrieved from Http://Ro.Ecu.Edu.Au/Theses/1683

Johnson, R. A., Gulick, A. G., Bolten, A. B., & Bjorndal, K. A. (2017). Blue Carbon Stores in Tropical Seagrass Meadows Maintained under Green Turtle Grazing. *Scientific Reports*, *7*(1), 13545.

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

Kaldy, J. E., & Dunton, K. H. (2000). Above-and Below-ground Production, Biomass and Reproductive Ecology of *Thalassia testudinum* (Turtle Grass) in a Subtropical Coastal Lagoon. *Marine Ecology Progress Series*, *193*, 271-283.

Kaldy, J. E., Eldridge, P. M., Cifuentes, L. A., & Jones, W. B. (2006). Utilization of DOC from Seagrass Rhizomes by Sediment Bacteria: 13C-tracer Experiments and Modeling. *Marine Ecology Progress Series*, *317*, 41-55.

Kendrick, G. A., Orth, R. J., Statton, J., Hovey, R., Ruiz Montoya, L., Lowe, R. J., & Sinclair, E. A. (2017). Demographic and genetic connectivity: the role and consequences of reproduction, dispersal and recruitment in seagrasses. *Biological reviews*, *92*(2), 921-938.

Kennedy, H., Beggins, J., Duarte, C. M., Fourqurean, J. W., Holmer, M., Marbà, N., & Middelburg, J. J. (2010). Seagrass Sediments as a Global Carbon Sink: Isotopic Constraints. *Global Biogeochemical Cycles*, *24*(4).

Kilminster, K., McMahon, K., Waycott, M., Kendrick, G. A., Scanes, P., McKenzie, L., & Glasby, T. (2015). Unravelling Complexity in Seagrass Systems for Management: Australia as a Microcosm. *Science of the Total Environment*, *534,* 97-109.

Kitheka, J. U. (1996). Water Circulation and Coastal Trapping of Brackish Water in a Tropical Mangrove-dominated Bay in Kenya. *Limnology and Oceanography*, *41*(1), 169-176.

Klap, V. A., Hemminga, M. A., & Boon, J. J. (2000). Retention of lignin in seagrasses: angiosperms that returned to the sea. *Marine Ecology Progress Series*, *194*, 1-11.

Kothari, C. R. (2004). *Research methodology: Methods and techniques*. *New Age International.*

Kristensen, P. (2004). The DPSIR Framework. *National Environmental Research Institute*, *Denmark*, *10 pp*.

Lamb, J. B., Jeroine, A.J.M., Water, Van De, Bourne, D. G., Altier, C., Hein, Y. M., Fiorenza, A.E., Abu, N., Jompa, J., & Harvell, D.C. (2017). Seagrass Ecosystems Reduce Exposure to Bacterial Pathogens of Humans, Fishes, and Invertebrates. *Science*, *355,* 731–733

Larkum, A. W., Drew, E. A., & Ralph, P. J. (2007). Photosynthesis and Metabolism in Seagrasses at the Cellular Level in *SEAGRASSES: BIOLOGY, ECOLOGYAND CONSERVATION* (Pp. 323-345). Springer Netherlands.

Lavery, P. S., Mateo, M. Á., Serrano, O., & Rozaimi, M. (2013). Variability in the carbon storage of seagrass habitats and its implications for global estimates of blue carbon ecosystem service. *PloS one*, *8*(9), e73748.

Lavery, P. S., & Vanderklift, M. A. (2002). A Comparison of Spatial and Temporal Patterns in Epiphytic Macroalgal Assemblages of the Seagrasses *Amphibolis griffithii* and *Posidonia coriacea*. *Marine Ecology Progress Series*, *236*, 99-112.

Lipkin, Y., Beer, S., & Zakai, D. (2003). The Eastern Mediterranean and The Red Sea. *World atlas of seagrasses*, 65-73.

Locatelli, T., Binet, T., Kairo, J. G., King, L., Madden, S., Patenaude, G., & Huxham, M. (2014). Turning the Tide: How Blue Carbon and Payments for Ecosystem Services (PES) Might Help Save Mangrove Forests. *Ambio*, *43*(8), 981-995.

Lyimo, T. J., Mvungi, E. F., Lugomela, C., & Björk, M. (2006). Seagrass Biomass and Productivity in Seaweed and Non-Seaweed Farming Areas in the East Coast of Zanzibar. *Western Indian Ocean Journal of Marine Science*, *5*(2), 141-152.

Lyimo, T. J., Mvungi, E. F., & Mgaya, Y. D. (2008). Abundance and Diversity of Seagrass and Macrofauna in the Intertidal Areas with and without Seaweed Farming activities in the East Coast of Zanzibar. *Tanzania Journal of Science*, *34*(1).

Lyimo, T. J., Mvungi, E. F., Lugomela, C., & Björk, M. (2006). Seagrass biomass and productivity in Seaweed and Non-Seaweed Farming areas in the East Coast of Zanzibar. *Western Indian Ocean Journal of Marine Science*, *5*(2), 141-152

Lyimo, L. D. (2016). *Carbon Sequestration Processes In Tropical Seagrass Beds* (Doctoral Dissertation, Department of Ecology, Environment and Plant Sciences, Stockholm University), 21- 46.

Macreadie, P. I., Baird, M. E., Trevathan-Tackett, S. M., Larkum, A. W. D., & Ralph, P. J. (2014). Quantifying and Modelling the Carbon Sequestration Capacity of Seagrass Meadows–A Critical Assessment. *Marine Pollution Bulletin*, *83*(2), 430-439.

Marbà, N., Arias-Ortiz, A., Masqué, P., Kendrick, G. A., Mazarrasa, I., Bastyan, G. R. & Duarte, C. M. (2015). Impact of Seagrass Loss and Subsequent Revegetation on Carbon Sequestration and Stocks. *Journal of Ecology*, *103*(2), 296-302.

Marbà, N., Duarte, C. M., Díaz-Almela, E., Terrados, J., Álvarez, E., Martínez, R., & Grau, A. M. (2005). Direct Evidence of Imbalanced Seagrass (Posidonia Oceanica) Shoot Population Dynamics in the Spanish Mediterranean. *Estuaries and Coasts*, *28*(1), 53-62.

Mateo, M., Cebrián, J., Dunton, K., & Mutchler, T. (2006). Carbon Flux in Seagrass Ecosystems. *Seagrasses: Biology, Ecology and Conservation*. *Springer, Netherlands. (159-192).*

Martins, A. R. O., & Bandeira, S. O. (2001). Biomass Distribution and Leaf Nutrient Concentrations and Resorption of *Thalassia hemprichii* at Inhaca Island, Mozambique. *South African journal of botany*, *67*(3), 439-442.

Mazarrasa, I., Samper-Villarreal, J., Serrano, O., Lavery, P. S., Lovelock, C. E., Marbà, N., & Cortés, J. (2018). Habitat Characteristics Provide Insights of Carbon Storage in Seagrass Meadows. *Marine pollution bulletin*.

McDonald, A. M., Prado, P., Heck Jr, K. L., Fourqurean, J. W., Frankovich, T. A., Dunton, K. H., & Cebrian, J. (2016). Seagrass Growth, Reproductive, and Morphological Plasticity across Environmental Gradients over a large Spatial Scale. *Aquatic Botany*, *134*, 87-96.

Mcglathery, K. J., Reynolds, L. K., Cole, L. W., Orth, R. J., Marion, S. R., & Schwarzschild, A. (2012). Recovery Trajectories During State Change from Bare Sediment to Eelgrass Dominance. *Marine Ecology Progress Series*, *448*, 209-221.

Mellors, J., Marsh, H., Carruthers, T.J.B., Waycott, M., 2002. Testing the Sediment-Trapping Paradigm of Seagrass: Do Seagrass Influence Nutrient Status and Sediment Structure in Tropical Intertidal Environments? Bull. Mar. Sci. 71, 1215–1226.

Miyajima, T., Hori, M., Hamaguchi, M., Shimabukuro, H., & Yoshida, G. (2017). Geophysical Constraints for Organic Carbon Sequestration Capacity of *Zostera marina* Seagrass Meadows and Surrounding Habitats. *Limnology and Oceanography*, *62*(3), 954-972.

Moore, K. A., & Short, F. T. (2007). Zostera: Biology, Ecology, and Management in *Seagrasses: Biology, Ecology and Conservation* (pp. 361-386). Springer, Dordrecht.

Muir, C. E., Sallema, A., Omari, A., De Luca, D., & Davenport, T. R. (2003). The dugong (Dugong dugon) in Tanzania: A National Assessment of Status, Distribution and Threat. *Wildlife Conservation*, *22*.

Musembi, P., Fulanda, B., Kairo, J., & Githaiga, M. (2019). Species Composition, Abundance and Fishing Methods of Small-scale Fisheries in the Seagrass Meadows of Gazi Bay, Kenya. *Journal of the Indian Ocean Region*, 1-18.

Nakaoka, M., Kouchi, N., & Aioi, K. (2003). Seasonal Dynamics of Zostera Caulescens: Relative Importance of Flowering Shoots to Net Production. *Aquatic botany*, *77*(4), 277-293.

Nellemann, C., & Corcoran, E. (Eds.). (2009). *Blue Carbon: The Role of Healthy Oceans in Binding Carbon: A Rapid Response Assessment*. UNEP/Earthprint.

Noel, H. W., Lucero, R. S., Parcasio JR, S. C., Labis, P. Y., & Lucero, M. J. (2012). Productivity and Distribution of Seagrass Communities in Davao Gulf.

Nolan, K. A., & Callahan, J. E. (2006). Beachcomber Biology: The Shannon-weiner Species Diversity Index. In *Proc. Workshop ABLE* (Vol. 27, pp. 334-338).

Nordlund, L. M., Jackson, E. L., Nakaoka, M., Samper-Villarreal, J., Beca-Carretero, P., & Creed, J. C. (2018). Seagrass Ecosystem Services–What's next?. *Marine pollution bulletin*, *134*, 145-151.

Obura, D. (2012). Coral Reefs and Society—Finding a Balance?. *Oryx*, *46*(4), 467-468.

Ochieng, C. A., & Erftemeijer, P. L. A. (2003). Kenya and Tanzania. *World Atlas of Seagrasses*, *82*.

Ondiviela, B., Losada, I. J., Lara, J. L., Maza, M., Galván, C., Bouma, T. J., & van Belzen, J. (2014). The Role of Seagrasses in Coastal Protection in a Changing Climate. *Coastal Engineering*, *87*, 158-168.

Onuf, C. P., Phillips, R. C., Moncreiff, C. A., Raz-Guzman, A., & Herrera-Silveira, J. A. (2003). The seagrasses of the Gulf of Mexico. *World Atlas of Seagrasses. UNEP-World Conservation Monitoring Centre. University of California Press, Berkeley, USA*, 224-233.

Oreska, M. P., Mcglathery, K. J., & Porter, J. H. (2017). Seagrass Blue Carbon Spatial Patterns at the Meadow-Scale. *Plos One*, *12*(4), E0176630.

Orth, R. J., Carruthers, T. J., Dennison, W. C., Duarte, C. M., Fourqurean, J. W., Heck, K. L., ... & Short, F. T. (2006). A Global Crisis for Seagrass Ecosystems. *AIBS Bulletin*, *56*(12), 987-996.

Pendleton, L., Donato, D. C., Murray, B. C., Crooks, S., Jenkins, W. A., Sifleet, S., & Megonigal, P. (2012). Estimating Global "Blue Carbon" Emissions from Conversion and Degradation of Vegetated Coastal Ecosystems. *Plos One*, *7*(9), E43542.

Phillips, R. C., & Menez, E. G. (1988). Seagrass. Smithsonian Contribution to the Marine Science No. 34.

Procaccini, G., Buia, M. C., Gambi, M. C., Perez, M., Pergent, G., Pergent-Martini, C., & Romero, J. (2003).
The Western Mediterranean. *World atlas of seagrasses*, *48*.

Ricart, A. M., Pérez, M., & Romero, J. (2017). Landscape configuration modulates carbon storage in seagrass sediments. *Estuarine, Coastal and Shelf Science*, *185*, 69-76.

Roca, G., Alcoverro, T., Krause-Jensen, D., Balsby, T. J. S., Van Katwijk, M. M., Marbà, N. & Pérez, M. (2016). Response of Seagrass Indicators to Shifts in Environmental Stressors: A Global Review and Management Synthesis. *Ecological Indicators*, *63*, 310-323.

Rothman, D. H., & Forney, D. C. (2007). Physical Model for the Decay and Preservation of Marine Organic Carbon. *Science*, *316*(5829), 1325-1328.

Ruiz, H., & Ballantine, D. L. (2004). Occurrence of the seagrass *Halophila stipulacea* in the Tropical West Atlantic. *Bulletin of Marine Science, 75*(1), 131-135.

Russell, B. D., Connell, S. D., Uthicke, S., Muehllehner, N., Fabricius, K. E., & Hall-Spencer, J. M. (2013). Future seagrass beds: Can increased productivity lead to increased carbon storage?. *Marine Pollution Bulletin*, *73*(2), 463-469.

Saunders, M. I., Leon, J., Phinn, S. R., Callaghan, D. P., O'brien, K. R., Roelfsema, C. M., & Mumby, P. J. (2013). Coastal Retreat and Improved Water Quality Mitigate Losses of Seagrass from Sea Level rise. *Global Change Biology, 19*(8), 2569-2583.

Serrano, O., Lavery, P., Masque, P., Inostroza, K., Bongiovanni, J., & Duarte, C. (2016). Seagrass Sediments Reveal the Long-term Deterioration of an Estuarine Ecosystem. *Global Change Biology*, *22*(4), 1523-1531.

Serrano, O., Ruhon, R., Lavery, P. S., Kendrick, G. A., Hickey, S., Masqué, P., & Duarte, C. M. (2016). Impact of Mooring Activities on Carbon Stocks in Seagrass Meadows. *Scientific Reports, 6*, 23193.

Siebert, T., & Branch, G. M. (2007). Influences of biological interactions on community structure within seagrass beds and sandprawn-dominated sandflats. *Journal of Experimental Marine Biology and Ecology*, *340*(1), 11-24.

Silva, J., Sharon, Y., Santos, R., & Beer, S. (2009). Measuring seagrass photosynthesis: methods and applications. *Aquatic Biology*, *7*(1-2), 127-141.

Short, F. T., & Coles, R. G. (Eds.). (2001). *Global seagrass research methods* (Vol. 33). *Elsevier.*

Short, F., Carruthers, T., Dennison, W., & Waycott, M. (2007). Global Seagrass Distribution and Diversity: A Bioregional Model. *Journal of Experimental Marine Biology and Ecology*, *350*(1), 3-20.

Smyth, K., & Elliott, M. (2016). Effects of Changing Salinity on the Ecology of the Marine Environment. *Stressors in the Marine Environment. Oxford University Press, Oxford*, 161-174.

Southwood, T. R. E., & Henderson, P. A. (2009). *Ecological Methods*. John Wiley & Sons.

Sreenivasulu, G., Jayaraju, N., Sundara Raja Reddy, B. C., & Lakshmi Prasad, T. (2015). Physico-chemical Parameters of Coastal Water from Tupilipalem coast, Southeast Coast of India. *Journal of coastal sciences*, *2*(2), 34-39.

Terrados, J., & Borum, J. (2004). Why are Seagrasses Important? -Goods and Services Provided by Seagrass Meadows. *European Seagrasses: an Introduction to Monitoring and Management*, 8-10.

Trevathan-Tackett, S. M., Macreadie, P. I., Sanderman, J., Baldock, J., Howes, J. M., & Ralph, P. J. (2017). A Global Assessment of the Chemical Recalcitrance of Seagrass Tissues: Implications for Long-Term Carbon Sequestration. *Frontiers in Plant Science*, *8*, 925.

Uchimura, M., Faye, E. J., Shimada, S., Inoue, T., & Nakamura, Y. (2008). A reassessment of Halophila species (Hydrocharitaceae) diversity with special reference to Japanese representatives. *Botanica Marina*, *51*(4), 258-268.

Uku, J., Björk, M., Bergman, B., & Díez, B. (2007). Characterization and Comparison of Prokaryotic Epiphytes Associated with Three East African Seagrasses 1. *Journal of phycology*, *43*(4), 768-779.

UNEP, W. (2001). IPCC Third Assessment Report 'Climate Change 2001'.

Unsworth, R. K., & Cullen, L. C. (2010). Recognising the Necessity for Indo-Pacific Seagrass Conservation. *Conservation Letters*, *3*(2), 63-73.

USGS, E. (2011). Seagrass Habitat in the Northern Gulf of Mexico: Degradation, Conservation and Restoration of a Valuable Resource. *US Geological Survey, and US Environmental Protection Agency*.

Van Der Heide, T., Van Nes, E. H., Geerling, G. W., Smolders, A. J., Bouma, T. J., & Van Katwijk, M. M. (2007). Positive Feedbacks in Seagrass Ecosystems: Implications for Success in Conservation and Restoration. *Ecosystems*, *10*(8), 1311-1322.

Van Keulen, M., & Borowitzka, M. A. (2003). Seasonal Variability in Sediment Distribution along an Exposure Gradient in a Seagrass Meadow in Shoalwater Bay, Western Australia. *Estuarine, Coastal and Shelf Science*, *57*(4), 587-592.

Wabnitz, C. C., Andréfouët, S., Torres-Pulliza, D., Müller-Karger, F. E., & Kramer, P. A. (2008). Regional-scale seagrass habitat mapping in the Wider Caribbean region using Landsat sensors: Applications to conservation and ecology. *Remote Sensing of Environment*, *112*(8), 3455-3467.

Wakeham, S. G., & Canuel, E. A. (2006). Degradation and Preservation of Organic Matter in Marine Sediments. In *Marine Organic Matter: Biomarkers, Isotopes and DNA* (Pp. 295-321). Springer Berlin Heidelberg.

Waycott, M., McMahon, K., Mellors, J., Calladine, A., & Kleine, D. (2004). A guide to Tropical Seagrasses of the Indo-West Pacific.

Waycott, M., Duarte, C. M., Carruthers, T. J., Orth, R. J., Dennison, W. C., Olyarnik, S., ... & Kendrick, G. A. (2009). Accelerating Loss of Seagrasses Across the Globe Threatens Coastal Ecosystems. *Proceedings of the National Academy of Sciences*, *106*(30), 12377-12381.

Willis, D. P. (1983). *Moorland and Shore: their Place in the Human Geography of old Orkney*. Department of Geography, University of Aberdeen.

Zonneveld, K. A. F., Versteegh, G. J. M., Kasten, S., Eglinton, T. I., Emeis, K. C., Huguet, C., & Mollenhauer, G. (2010). Selective Preservation of Organic Matter in Marine Environments; Processes and Impact on the Sedimentary Record. *Biogeosciences*, *7*(2), 483-511.

APPENDICES

APPENDIX I

SITE CHARACTERIZATION SHEET

<table>
<tr><td colspan="3">REGION:</td><td colspan="2">SAMPLE POINT:</td></tr>
<tr><td colspan="3">DATE:</td><td colspan="2">ALTITUDE:</td></tr>
<tr><td colspan="5">Predominant surrounding land use (Specify relative percent in each category):</td></tr>
<tr><td>Mangrove forest</td><td>Landing site</td><td>Seagrass bed</td><td>Fishing area</td><td>Others</td></tr>
<tr><td colspan="5">Seagrass Canopy cover:
[ ] open [ ] lightly shaded (11-45%) [ ] moderately shaded (46-80%) [ ] heavily shaded</td></tr>
<tr><td colspan="5">Tide characteristics:
[ ] Low [ ] Moderate [ ] High</td></tr>
<tr><td colspan="3">Estimated creek width:</td><td colspan="2">Estimated creek depth:</td></tr>
<tr><td colspan="3"></td><td colspan="2"></td></tr>
<tr><td colspan="5">Abiotic characterization of the site:</td></tr>
<tr><td>Temperature</td><td></td><td>Oxygen</td><td>pH</td><td>Conductivity</td></tr>
<tr><td colspan="5">Abundance of dominant seagrass species</td></tr>
<tr><td>Species</td><td>Absent</td><td>Rare</td><td>Common</td><td>Abundant</td></tr>
<tr><td>Thalassia hemprichii</td><td></td><td></td><td></td><td></td></tr>
<tr><td>Cymodocea rotundata</td><td></td><td></td><td></td><td></td></tr>
<tr><td>Enhalus acoroides</td><td></td><td></td><td></td><td></td></tr>
<tr><td>Mixed bed</td><td></td><td></td><td></td><td></td></tr>
<tr><td>Other</td><td></td><td></td><td></td><td></td></tr>
<tr><td colspan="5">Site substrates:</td></tr>
<tr><td>Substrate type</td><td colspan="2">%</td><td>Substrate type</td><td>%</td></tr>
<tr><td></td><td colspan="2"></td><td></td><td></td></tr>
<tr><td colspan="3"></td><td colspan="2"></td></tr>
<tr><td colspan="5">Cluster/strata drawing:</td></tr>
</table>

APPENDIX II

COMMON CARBON ASSESSMENT FORMULAE AND CONVERSION FACTORS

1. Common Carbon Calculations

Total Carbon (MgC/ha) × Area (ha) = Tier 1 total carbon stock for the project site (Mg)

- Where Total Carbon = the mean carbon stock for a given ecosystem
- Area = the area of the ecosystem being investigated

Total potential CO_2 emissions per hectare (Mg CO_2/ha) = Conversion factor for the CO_2 that can be produced from the carbon present in the system × carbon in the system

- Conversion factor = 3.67, the ratio of the molecular weights of CO_2 (44) and carbon (12)
- Carbon in the system = the mean carbon stock for a given ecosystem

Carbon conversion factor for seagrass = 0.34, based on tier 1 IPCC emission factor of 7.9 tonnes of C ha^{-1}

Organic carbon content of a sample = Total carbon content (elemental analyzer or LOI %) – (Inorganic carbon content of ashed subsample × (Weight of subsample after ashing/Dry weight before ashing)

Soil carbon density (g/cm^3) = dry bulk density (g/cm^3) × (% Corg/100)

2. Common Carbon Conversions

1 g of carbon = 0.001 kg of carbon

1 g of carbon = 0.000001 Mg of carbon

1 g of carbon = 10^{-12} Tg of carbon

1 g of carbon = 10^{-15} Pg of carbon

Total core carbon (MgC/hectare) = Summed core carbon (g/cm2) × (1 Mg/1,000,000 g) × (100,000,000 cm2/1 hectare)

APPENDIX III

CHUKA UNIVERSITY RESEARCH AUTHORIZATION

CHUKA **UNIVERSITY**

Telephones: 020 2310512
020 2310518

P.O. Box 109
Chuka

OFFICE OF THE CHAIRMAN
INSTITUTIONAL ETHICS REVIEW COMMITTEE

Our Ref: CU/IERC/NCST/18/40

25th May, 2018

THE CHIEF EXECUTIVE OFFICER
NATIONAL COMMISION FOR SCIENCE, TECHNOLOGY AND INNOVATION
P.O. BOX 30623-00100
NAIROBI

Dear Sir/Madam,

RE: RESEARCH CLEARANCE AND AUTHORIZATION FOR GABRIEL AKOKO JUMA. REG NO NM11/29160/17

The above matter refers:

The Institutional Ethics Review Committee of Chuka University met and reviewed the above MSC Research Proposal titled Assessment of the Distribution, Abundance and Carbon Stocks in Seagrass Meadows within Eastern and Western Creeks of Gazi Bay, Kenya" The Supervisor is Prof. Adiel Magana and Dr. James G. Kairo

The committee recommended that after candidate amends the issues highlighted in the Attached research clearance and authorization check list, the permit be issued.

Attached please find copies of the minutes, research clearance and authorization check list for your perusal. Kindly assist the student get the research permit.

Yours faithfully,

Amuraya

Dr. Moses Muraya
FOR: **CHAIR**
INSTITUTIONAL ETHICS REVIEW COMMITTEE
cc: BPGS

APPENDIX IV

NACOSTI RESEARCH AUTHORIZATION

NATIONAL COMMISSION FOR SCIENCE, TECHNOLOGY AND INNOVATION

Telephone:+254-20-2213471,
2241349,3310571,2219420
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Email: dg@nacosti.go.ke
Website : www.nacosti.go.ke
When replying please quote

NACOSTI, Upper Kabete
Off Waiyaki Way
P.O. Box 30623-00100
NAIROBI-KENYA

Ref: No. **NACOSTI/P/18/52699/24288**

Date: **18th August, 2018**

Gabriel Akoko Juma
Chuka University,
P. O. Box 109-60400
CHUKA.

RE: RESEARCH AUTHORIZATION

Following your application for authority to carry out research on ***"Assessment of the distribution, abundance and carbon stocks in seagrass meadows within Eastern and Western Creeks of Gazi Bay, Kenya,"*** I am pleased to inform you that you have been authorized to undertake research in **Kwale County** for the period ending **17th August, 2019.**

You are advised to report to **the County Commissioner and the County Director of Education, Kwale County** before embarking on the research project.

Kindly note that, as an applicant who has been licensed under the Science, Technology and Innovation Act, 2013 to conduct research in Kenya, you shall deposit **a copy** of the final research report to the Commission within **one year** of completion. The soft copy of the same should be submitted through the Online Research Information System.

BONIFACE WANYAMA
FOR: DIRECTOR-GENERAL/CEO

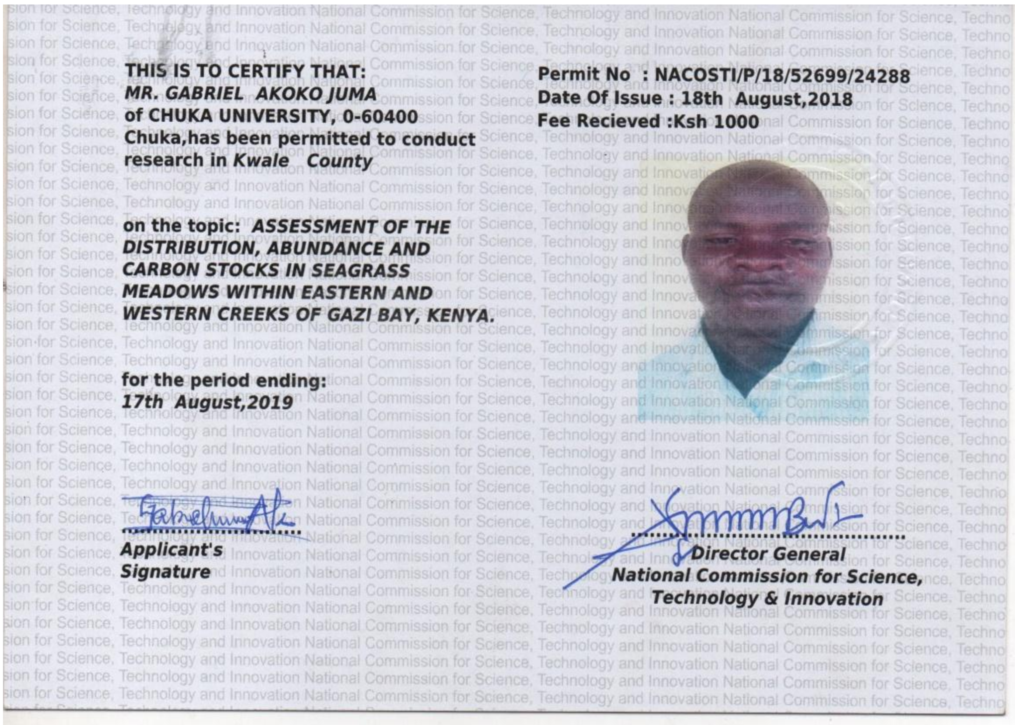
Copy to:

The County Commissioner
Kwale County.

The County Director of Education
Kwale County.

APPENDIX V

NACOSTI RESEARCH CLEARENCE PERMIT



THIS IS TO CERTIFY THAT:
MR. GABRIEL AKOKO JUMA
of CHUKA UNIVERSITY, 0-60400
Chuka,has been permitted to conduct
research in *Kwale County*

on the topic: *ASSESSMENT OF THE DISTRIBUTION, ABUNDANCE AND CARBON STOCKS IN SEAGRASS MEADOWS WITHIN EASTERN AND WESTERN CREEKS OF GAZI BAY, KENYA.*

for the period ending:
17th August,2019

Permit No : NACOSTI/P/18/52699/24288
Date Of Issue : 18th August,2018
Fee Recieved :Ksh 1000

Applicant's Signature

Director General
National Commission for Science, Technology & Innovation