THE ROLE OF SEAGRASS MEADOWS IN GAZI BAY, KENYA AS CARBON SINKS

MICHAEL NJOROGE GITHAIGA

A thesis submitted in partial fulfilment of the requirements of Edinburgh Napier University, for award of Doctor of Philosophy

School of Applied Sciences

Edinburgh Napier
UNIVERSITY

April 2017

Supervisors

Director of Studies (DoS)

Professor Mark Huxham
School of Applied Sciences
Edinburgh Napier University

Signed ___________________________________

Supervisor

Dr. Linda Gilpin
School of Applied Sciences
Edinburgh Napier University

Signed ___________________________________

Supervisor

Dr. James G. Kairo
Hydrography and Oceanography Department
Kenya Marine and Fisheries Research Institute

Signed ___________________________________

“*We owe it to ourselves and to the next generation to conserve the environment so that we can bequeath our children a sustainable world that benefits all”*.

Prof. Wangari Maathai (2004).

Front cover: Photo of *Syringodium isoetifolium* taken during low spring tide at Gazi Bay, Kenya. (Photo: Githaiga, 2016).

Dedication

To my family: Marion, Vanessa, Nelly, Diana & Eric and to all those that believed in me.

Acknowledgement

The road to this doctorate is one of the greatest and most fulfilling milestones in my life. It has been momentous, exciting and above all challenging but a lovely journey. I have utmost gratitude to the wonderful team that saw my passion, believed in me, nurtured my dream and above all was there to guide me in this academic endeavour. In particular, I am really grateful to my director of studies, Prof. Mark Huxham of Edinburgh Napier University for being such a tremendous mentor. Your financial support, the inspiration, the leadership and guidance throughout this research study are priceless. Dr. Linda Gilpin of the same university was instrumental in helping me nurture the writing skills through the regular review of my work, guided me and above all constantly reminded me of the basics that I ignored when writing but mattered most. I have special thanks to my local supervisor, Dr James Kairo of Kenya Marine and Fisheries Research Institute who introduced me into this field of marine ecology and was not only my academic mentor but provided me with the brotherly love and foresight during my academic sojourn in Gazi village.

I sincerely thank the committee members at various stages during the development of this dissertation. In particular, the chair, Dr. Charlotte Chalmers of Edinburgh Napier University and the other panelist for the professional and brilliant comments plus all the suggestions that helped to develop a clear focus on the studies that culminated to the development of this thesis.

I was lucky to work and learn from a team of experts: Professor Hillary Kennedy and Dr. Martin Skov, both of Bangor University, UK during our regular meetings and fieldwork activities in Zanzibar.

The development of the thesis that culminated to the award of a doctorate would not have been possible without the dedication and able assistance of my two valuable field assistants, Laitani Suleiman and Tom Kisiengo. You ever remained my foot soldiers

in the field and laboratory work even during the periods I was away at Edinburgh, and for this, I salute you. My sincere thanks to the entire KMFRI fraternity both at Gazi station and at the headquarters for walking with me. Special thanks to Kimathi, Kathengi and Onduso for working tirelessly in processing the sediment samples. There are many of my colleagues that I want to thank in person, among them the UK team: Ankje Frouws, whom I shared a lot, Maria Potouroglou, Sasha Flannigan, Rose Adamson, Lucas Ruzowitsky, Connie McKay and the students from local Universities that I worked with: Reagan Okoth, Susan Gatune, Derrick Omollo and Jackline Mutisya. I am deeply indebted to the hospitable teams of Edinburgh Napier that ensured I enjoyed my stay during my regular visits. In particular, I salute Ann Akcan for the friendly welcome and for the arrangements on the logistics that made Edinburgh my second home.

There are many friends both in social fraternity and in the field of academia who deserve mention and appreciation for the companionship and support, which made the walk enjoyable and fulfilling. My relatives were instrumental in prayers and moral support, which indeed strengthened my spirits and the drive to carry on. Though not mentioned in person, be assured that your support will be ingrained forever in my memory. I would be remiss if I fail to mention heartfully and by name the two extraordinary women in my life for their unequivocal and priceless service. First, my mother, Miriam Wairimu for her nurture as I grew up and role model on hard work, right from my childhood and above all the constant prayers for my success. Secondly, my dear wife, Marion Wambui who has not only remained a lifetime friend but a faithful and dedicated custodian of our home, the many years I was away. Lastly but not least is a big thank you to my offspring: Vanessa Wairimu, Nelius Njeri, Diana Njoki and Eric Githaiga for their love and the incredible amount of perseverance they had to contend with during their formative years; a long period of eight years of my post graduate academic pursuit.

This PhD thesis was funded by Coastal Ecosystem Services in East Africa (CESEA) NE/L001535/1, a research project that was funded by Ecosystem Services for Poverty Alleviation (ESPA) programme. The ESPA programme is funded by the Department for International Development (DFID), the Economic and Social Research Council (ESRC) and the Natural Environment Research Council (NERC). Additional support was made through Edinburgh Napier University and Kenya Marine and Fisheries Research Institute (KMFRI) while a research permit (no NACOSTI/2/14/2443/769) for this study was issued by the National Commission for Science Technology and Innovation (NACOSTI) in Nairobi, Kenya to whom I am grateful.

Preface

Conservation of marine ecosystems is of intrinsic interest as their health integrity underpins their ability to provide the biodiversity and ecosystem services. Despite this value, these ecosystems continue being degraded overtime. The increasing quest at present for understanding their functioning is driven by an urge to develop strategies for their conservation in order to ensure the continued provision of the ecosystem services to the ever-growing global population in the wake of climate change. In line with these noble goals, this study was conceived with an aim of developing a thesis, titled "**The Role of Seagrass Meadows in Gazi Bay, Kenya as Carbon Sinks.**" The scope and structure of the thesis constitutes six chapters that have been developed in line with the objectives of the study.

Chapter I: Introduction: This chapter gives an introduction of the seagrass ecosystem. It opens with a short description of the seagrasses and their habit and an enumeration of the biodiversity and ecosystem services provided by the seagrasses. It links seagrasses to the blue carbon awareness thus underscoring the importance of sound management and conservation of the ecosystem. It also gives a short description of the seagrasses of Kenya, narrowing to Gazi Bay, the site where this study was conducted and winds up by giving overall objectives of the study.

Chapter 2: Biomass and productivity of seagrasses in Africa: This review chapter briefly discusses the seagrass ecosystem, its invaluable role in climate change mitigation and for the provision of ecosystem services. It brings forth the status of knowledge of the biomass and productivity of seagrass in Africa by collating and synthesizing all published and available grey literature on the topic of carbon stocks and flows in African seagrass meadows. It compares this information with the global literature and identifies key gaps.

The findings in this review work underscores the limited knowledge of biomass and productivity of seagrass in Africa with only 32 papers and 8 theses/reports from African sites having been found with the majority of the studies having been done along the E. African coast and with no published reports of sedimentary organic carbon from seagrasses in Africa. A paper has been published from this chapter.

Chapter 3: Carbon storage in the seagrass meadows of Gazi Bay, Kenya: The chapter provides detailed estimates of the carbon storage of the four dominant seagrass species of Gazi Bay: *Thalassia hemprichii*, *Thalassodendron ciliatum*, *Syringodium isoetifolium* and *Enhalus acoroides*. It looks at the variability in C_{org} between the sediment and the species biomass, compares the vegetated and un-vegetated C_{org} and also the interspecies variability in the C_{org}. It also reports on the key above-ground parameters of the seagrasses that include the shoot density, canopy cover and height as well as exploring the relationships between the sediment organic carbon and the above-ground biomass as well as whether there are some above-ground predictors for below-ground carbon stores. Key findings from this study were that sediment organic carbon was highly significantly different between species, range: 160.7 – 233.8 Mg C ha^{-1} compared to the global range of 115.3 to 829.2 Mg C ha^{-1}. Vegetated areas in all species had significantly higher sediment C_{org} compared with un-vegetated areas and revealed a degree of spatial consistency and longevity in relatively small patches of seagrass meadows and bare areas thus demonstrating an exceptionally powerful effect of seagrass on carbon sequestration.

Chapter Four: **Experimental test of the impact of seagrass loss on sediment dynamics and on the benthic faunal communities:** The chapter is an experimentally based research work in which seagrass canopies were artificially removed in randomly selected patches with controls to simulate the natural seagrass loss and hence investigate the impact on sediment dynamics and benthic faunal communities. Sediment – Elevation Tables (SETs) were established and monitored monthly over an eighteen month period. Carbon density, effect of seagrass disturbance on hydrodynamic energy reduction, litter decay rates, grain sizes and impact on the faunal communities were investigated. The study recorded positive elevation change of the sediment in the controls and negative elevation in the treatment (seagrass removed) with significant effect of treatment and time on surface elevation change. Seagrass removal was found to impact significantly on carbon density, hydrodynamic energy, on the litter decomposition and on the faunal communities.

Chapter Five: Productivity of seagrasses in Gazi Bay: The chapter reports on the estimation of productivity of the four dominant seagrasses of Gazi Bay, investigated through a combination of field survey and an experiment. Shoot, productivity was investigated through the leaf plastochrone method while ingrowth trenches were used to estimate the below-ground productivity. In the experimental based approach, shoot regrowth characteristics and biomass increment were monitored monthly following seagrass harvesting. The productivity measurements showed inter-species variability with *E. acoroides* recording the highest shoot productivity at 1944 g DW m^{-2} yr^{-1} and below-ground biomass accumulation following recovery at 197.4±108.7 g DW m^{-2} yr^{-1} respectively.

Chapter Six: General conclusion: The chapter summarize the key findings of the study and the implication of these findings within the context of the “blue carbon” framework and the national and international policies. It also compares the seagrass carbon sequestration potential with that of the contiguous mangrove forest ecosystem of the bay. It brings forth the idea of how the knowledge from this study could have practical application by exploring the potential of restoring and protecting seagrass habitats through Payment for Ecosystem Services (PES) scheme through bundling the ecosystem services of seagrasses with associated blue carbon ecosystems. It winds up by identifying the knowledge gaps and giving recommendations for the appropriate management conservation interventions.

Table of Contents

List of tables

List of figures

Acronyms and abbreviations

AFDW	-	Ash Free Dry Weight
AGB	-	Above Ground Biomass
BGB	-	Below Ground Biomass
C	-	Carbon
C_{org}		Organic carbon
COP		Conference of the Parties
DW	-	Dry Weight
IUCN		International Union of Nature Conservation
KMFRI		Kenya Marine and Fisheries Research Institute
LOI	-	Loss On Ignition
NGO	-	Non- Governmental Organizations
PES		Payment for Ecosystem Services
Pg		Petagram
PPT		Parts Per Thousand
TB		Total Biomass
TC		Tonnage Carbon
Tg		Teragram
UNFCCC		United Nations Framework Convention on Climate Change
WIO		West Indian Ocean
WIOMSA		West Indian Ocean Marine Science Association

Glossary

Accretion	Gradual gain of land elevation, as in the addition of sand to a beach by wind or ocean currents.
Allochthonous Carbon	Carbon produced in one location and deposited in another. This type of carbon results from the hydrodynamic environment in which they are found, where sediments and associated carbon is transported from neighbouring systems (offshore and terrestrial).
Autochthonous Carbon	Carbon produced and deposited *insitu*. The carbon results from the vegetation uptake of CO_2 from the ocean and/or atmosphere and is converted for use by plant tissue and decomposes into the surrounding soil.
Blue Carbon	The carbon stored in mangroves, tidal salt marshes and seagrass meadows within the soil, the living biomass above-ground (leaves, branches and stems), the living biomass below-ground (roots), and the non-ling biomass (litter and deadwood).
Carbon pool	Carbon reservoirs such as soil, vegetation, water and the atmosphere that absorb and release carbon.
Carbon stock	Is the total amount of organic carbon stored in an ecosystem of known size.
Ecosystem resilience	Refers to the ability of an ecosystem to cope with and respond to disturbances and to restore itself. Highly resilient ecosystems can respond to natural disturbances faster than

	ecosystems which have low resilience. Degraded ecosystems tend to have lower resilience and are therefore less able to recover following disturbance.
Mitigation	A general term meaning to lessen or make less severe. In the context of climate change, it means any action that lessens climate change, for example by decreasing the quantity of greenhouse gases in the atmosphere.
Interdisciplinary approach	Drawing appropriately from two or more academic disciplines or field study to redefine problems outside normal boundaries and reach solutions based on new understanding of complex situations.
Ocean acidification	Decrease in the pH of seawater due to the uptake of anthropogenic carbon dioxide.
Sequestration	Uptake and storage of a material, such as carbon.
Sediment Organic Carbon	Is the carbon component of soil organic matter. The amount of Sediment C_{org} depends on soil texture, climate, vegetation and historical and current land use/management.
Soil Organic Matter	Is used to describe the organic constituents in the soil (Un-decayed tissues from dead plants and animals products).
Tidal range	Difference in height between highest and the lowest tidal marks.

Abstract

Ongoing work on the 'blue carbon' has established that vegetated coastal ecosystems – mangroves, seagrasses and tidal marshes – are exceptionally powerful natural carbon sinks. Hence, there are important applied arguments for understanding the role of these ecosystems in climate change mitigation. Many gaps in knowledge remain, with seagrasses in particular being poorly understood. The overall aim of the present study was to understand the role of seagrass meadows of Gazi Bay, Kenya as carbon sinks. The specific objectives of the study were: (i) Carry out a comprehensive review on biomass and productivity studies of seagrasses in Africa. (ii) Estimate the carbon storage in the seagrass meadows of the bay (iii) Investigate the impact of seagrass canopy removal on sediment dynamics and on the benthic faunal communities (iv) Investigate productivity of the dominant seagrass species of the bay. (v) Highlight the implication of the knowledge and how it could be useful in the bundling of the ecosystem services of the seagrass meadows with that of adjacent mangrove ecosystem under the payment for ecosystem services (PES).

The review work encountered 32 papers and 8 theses/reports on seagrass biomass and productivity at African sites, with the majority of the studies having been done along the E. African coast; however, there were no published reports of sedimentary C_{org} from Africa, revealing a major gap in knowledge. Results of carbon storage of the seagrass meadows of Gazi, Bay on biomass and sediment organic carbon (C_{org}) for the four dominant species, *Thalassia hemprichii*, *Thalasodendron ciliatum*, *Syringodium isoetifolium* and *Enhalus acoroides,* indicated that sediment organic carbon was highly significantly different between species, range: 160.7 – 233.8 Mg C ha^{-1} compared to the global range of 115.3 to 829.2 Mg C ha^{-1}. Vegetated areas in all species had significantly higher sediment C_{org} compared with un-vegetated areas and revealed a surprising degree of spatial consistency and longevity in relatively small patches of seagrass meadows and bare areas thus demonstrating an exceptionally powerful effect of seagrass on C sequestration. Through a seagrass removal experiment that simulated the impact of seagrass loss on biodiversity and ecosystem functions the study recorded positive elevation change of the sediment in the controls and negative elevation in the treatments with significant effect of treatment and time on surface elevation change. Similarly, there were significant weight losses on clod cards in seagrass removed areas compared to the controls, an indication of the role of seagrasses in the reduction of the speed of water current. Carbon density in surface sediment was significantly higher in the control areas as compared to the treatments while higher litter decay rates were observed in seagrass removed areas than in the controls. Seagrass removal areas had significant decline on the fauna. Productivity estimates of the dominant seagrass species of the bay through a combination of the leaf plastochrone, and ingrowth cores, showed inter-species variability with *Thalassia hemprichii* recording the highest above and below-ground productivity at 188.6±34.8 g DW m^{-2} yr^{-1} and 197.4±108.7 g DW m^{-2} yr^{-1} respectively. Knowledge on the role of seagrasses of the bay as carbon sinks is likely to open opportunities for bundling seagrass ecosystem services with that of the contiguous mangrove ecosystem as part of Payment for Ecosystem Services (PES); an approach that makes economic and ecological sense, given the strong connections between the two ecosystems, and could provide a useful buffer against environmental shocks such as sea level rise and increasing storm surges.

Key words: Blue carbon, carbon sink, mitigation, productivity, seagrass biomass, seagrass species.

Chapter one

1.0 Introduction

1.1 Role of marine vegetated ecosystems

Marine ecosystems are fundamental blocks to the web of life as they mediate numerous biological processes and ecosystem services that underpin resource availability for all forms of life and human well-being. Though ecosystems have always been valued by man, their role as providers of ecosystem services gained prominence following the Millennium Ecosystem Assessment (MA) of 2005 that categorised ecosystem services into four groups: Provisioning services (Food, water, energy, timber, pharmaceuticals etc), Regulating services (Carbon sequestration, coastal protection, flood prevention, purification of water, crop pollination, disease control etc), Supporting services (Nutrient cycling, primary production, seed dispersal, structural habitat provisioning etc) and cultural services (cultural and religious, spiritual inspiration, recreation etc) (Millenium Ecosystem Assessment, 2005).

These ecosystems have functional interconnections and feedback mechanisms that form ecological pathways through which materials are channelled thus sustaining an ecological balance. Among the significant pathways is the biogeochemical cycling of substances that are fundamental to life processes. However, the ecological stability can be upset at certain points in time due to natural or human induced perturbations. Widespread loss of the vegetated marine ecosystems results from direct human impacts, including, overexploitation, pollution, aquaculture, coastal developments mechanical damage (dredging, fishing, and anchoring), eutrophication, siltation, and food web alterations and indirect human impacts, including negative effects of climate change (sea level rise, increased storms, increased ultraviolet irradiance), as well as from natural causes, such as cyclones and floods (Duarte, 2002).

Of considerable concern and perhaps the greatest challenge to human kind at present is the increasing level of greenhouse gases, mainly CO_2 in the atmosphere, which, apart from contributing to global warming and climate change, is also associated with the acidification of ocean water. This interferes with the fixation of calcium carbonate and hence impedes shell formation to the billions of marine fauna as well as contributing to coral bleaching, thus subjecting their very existence in jeopardy. Climate change phenomenon being experienced at present has put these ecosystems at higher thresholds of vulnerability than at any other time before leading to loss of these critical habitats. Such losses have far reaching ramifications (Scheffran & Battaglini, 2011) which include: loss of important biodiversity, food insecurity, increased poverty levels, loss of human settlements; impacts that predispose human conflicts and loss of lives (Burrows & Kinney, 2016; Taenzler et al., 2012). Mitigation and adaptation to the climate change challenges have therefore become an area of considerable interest in management and research, with growing concerns over the potential risks and how these are likely to impact life on earth. Raging debates at the international arena through the Conference of the Parties (COP) under the United Nations Framework Convention on Climate Change (UNFCCC) have culminated in the formation of 'Accords and Protocols' such as the Rio+, Marrakesh Accords, Kyoto Protocol, Paris agreement and the Copenhagen Accord (IUCN, 2011; UNFCC, 1998, 2010, 2016). These multilateral agreements propose actions and strategies of reducing the emission of greenhouse gases particularly carbon dioxide (CO_2). Several strategies of achieving these ambitious targets have being explored. Among them is the geoengineering idea of harnessing the application of artificial methods of carbon capture and storage in deep reservoirs (He et al., 2011; Russell et al., 2012; Scheneider, 2008; Stilgoe, 2016). Though this approach does not undermine the option of using nature based solutions such as the investment in natural carbon sinks, the artificial measures pose potential risks of leakage of these gases leading to irreparable damages and entailing a lot of costs (He et al., 2011; Resnik & Vallero, 2011). Furthermore natural carbon sinks not only provide an effective method of reducing the CO_2 levels in the atmosphere but is associated with multiple benefits that include: habitats for biodiversity, breeding grounds for diverse fauna species, shoreline stability through the attenuation of the strong wave energies and storm surges (Irving et al., 2011). Despite the awareness, disproportionate focus on the terrestrial ecosystems in combating global warming and climate change in the past overshadowed the significant contribution of marine ecosystems.

Recent studies show that vegetated coastal plant communities could contribute significantly to combating climate change through the capture and storage of CO_2 (Duarte, 2017; Duarte et al., 2013; Howard et al., 2017). However, the realisation that these marine ecosystems contribute significantly to the global carbon budgets acquired strong recognition following the “blue carbon” initiative and forthwith these vegetated marine ecosystems: mangroves, seagrasses and the salt marshes became acknowledged as the “blue carbon” ecosystems, thus distinguishing them from the carbon in terrestrial sinks (Mcleod et al., 2011; Nellemann et al., 2009). Despite this recognition, our understanding of the contribution of these ecosystems to the global carbon budgets is limited by lack of comprehensive knowledge of the relative contribution of each of these marine carbon sinks and their ecological connectivity.

The blue carbon ecosystems occupy less than 2% of the world’s ocean surface but are estimated to contribute in excess of 10% of carbon buried annually in the oceans (Fourqurean et al., 2012) and as such provide a more balanced and ecologically sound method of overcoming the challenges of global warming and climate change. Unlike terrestrial systems that store organic carbon primarily in living biomass and soil organic matter, vegetated coastal systems store the majority of organic carbon in sediment (Duarte et al., 2005; Fourqurean et al., 2012; Mcleod et al., 2011). Past studies have demonstrated that marine macrophytes act as effective carbon sinks, capturing over half of the global carbon annually (Nellemann et al., 2009).

Seagrass meadows are among the most productive and biodiverse ecosystems of the world's oceans (Laffoley & Grimsditch, 2009; Mcleod et al., 2011). The total primary production is contributed equally by the angiosperms themselves (the flowering plant) and the macroalgae/epiphytes they support (Guy, 2010). These meadows are net CO_2 sinks within the biosphere, meaning that the seagrass communities tend to be net autotrophs (Duarte et al., 2010). Comparatively, seagrass meadows have received a small fraction of the attention of mangroves and studies from Africa are particularly scarce (Duarte & Chiscano 1999; Fourqurean et al., 2012; Kennedy et al., 2010). Although seagrass carbon burial rates per unit area are lower than in mangroves, their larger spatial extent estimated at 300,000 to 600,000 (Mcleod et al., 2011; Nellemann et al., 2009), means that their carbon sink capacity may surpass that of mangroves. The amount of organic carbon stored in seagrass meadows is estimated to be roughly equal to the total amount stored in the world's marine tidal salt marsh and mangrove forests (Chmura et al., 2003; Kauffman et al., 2011). However, the uncertainties in the spatial cover and the limited knowledge in relation to the carbon dynamics may be the salient reason why they have been overshadowed in the global carbon budgets. Studies have demonstrated that, per unit area, seagrass sediment can store about twice as much C as soils in the temperate and tropical forests (Duarte et al., 2005; Fourqurean et al., 2012; Mcleod et al., 2011). However, an average of 24.3% of their net primary productivity is thought to be exported to neighbouring ecosystems (Duarte & Cebrián, 1996) which means that the present focus on storage of organic carbon in sediments within seagrass meadows alone underestimates the true magnitude of the carbon sequestration that seagrasses support (Duarte & Krause-jensen, 2017). Above-ground biomass is usually negligible since seagrass shoots have a higher turnover by being harvested and rapidly decomposed and/or rapidly exported from seagrass meadows through the movement of tidal waters. Epiphytes are part of the above-ground carbon pool with their contribution depending on species and growth conditions. On a global scale, the sediment C_{org} forms the bulk of organic carbon found below the surface of seagrass beds as the below-ground biomass only represents 0.3% of the total organic carbon (Fourqurean et al., 2012). Despite the importance of below-ground soil carbon pools, they are the least studied. However, there are increased interest in understanding the below-ground carbon dynamics due to the novelty and recent recognition of the significance of below-ground soil carbon in these systems as an important source of carbon globally (Chmura et al., 2003; Fourqurean et al., 2012).

1.2 Seagrass ecology, distribution and habitat characteristics

Seagrasses are a unique polyphyletic group of aquatic plants that have adapted to living in fully submerged coastal marine waters (Green & Short, 2003). They inhabit the shallow photic zone environments of all continents except Antarctica (Fig. 1), forming extensive meadows, termed ‘seagrass beds’, which cover around 0.2% of the oceans globally (Duarte et al., 2005; Green & Short, 2003). Seagrasses fall into six bioregions based on species assemblages, species distribution range as well as the tropical and temperate influences. The temperate bioregions has four subdivisions, which include the Temperate North Atlantic, the Temperate North Pacific, the Mediterranean, and the Temperate Southern Oceans while the tropical bioregions has two: the Tropical Atlantic and the Tropical Indo-Pacific. The later have the highest species diversity at 24 of the close to 60 known seagrass species globally (Green & Short, 2003; Short et al., 2007).

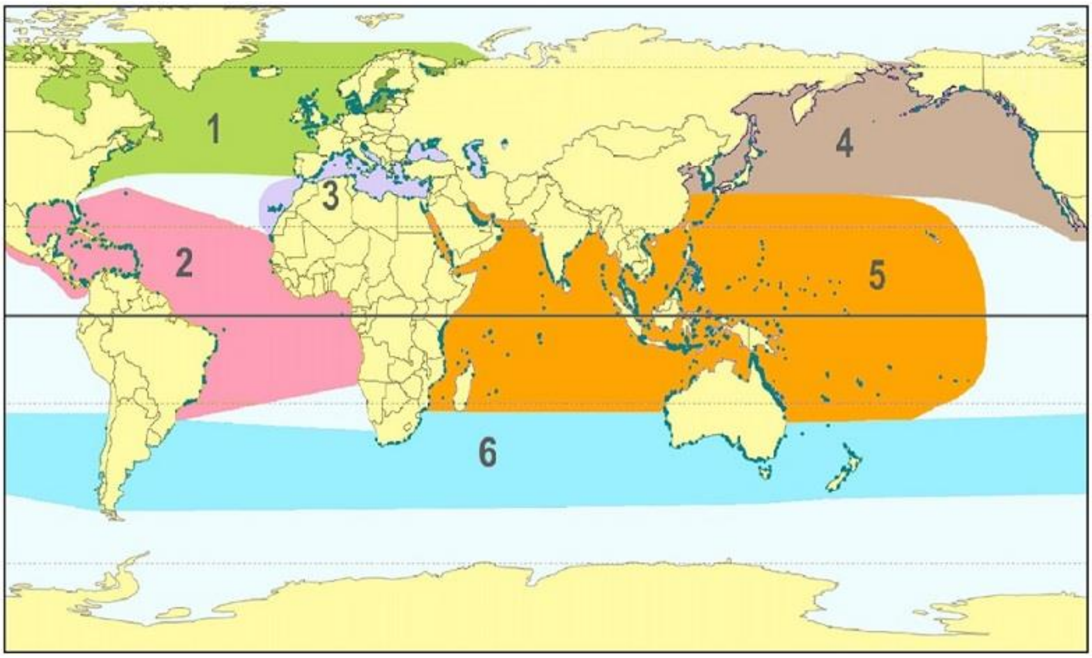

Fig. 1: Global seagrass distribution and geographic bioregions

1. Temperate North Atlantic 2. Tropical Atlantic 3. Mediterranean 4. Temperate North Pacific 5. Tropical Indo-Pacific 6. Temperate Southern Oceans (Source: Short et al., 2007).

Seagrasses are typically distributed from the high intertidal to the low subtidal areas on soft substrates such as sandy bays, estuaries and mudflats where they form either, multi-specific (consisting of two or more species) or mono-specific (containing single species) meadows (Green & Short, 2003). Several environmental parameters that determine their distribution are mainly the biophysical conditions that regulate the physiological activity and morphology of seagrasses, such as temperature, salinity, depth, substrate type, day length, light availability, water currents, wave action, epiphytes, and diseases (Green & Short, 2003; Hemminga & Duarte, 2000). Biologically driven parameters include the availability of seeds and vegetative fragments. Anthropogenic influences often drive conditions that commonly inhibit available plant resources, such as nutrients and high sediment loading (Hemminga &

Duarte, 2000). They also influence the health status of seagrasses either directly through mechanical methods or indirectly by inducing conditions that are detrimental to their survival. Various combinations of these parameters will permit, encourage, or eliminate seagrass from a given locality (Coles et al., 2011). Light is generally the limiting factor for the macrophytes growth and control the depth limit for seagrass meadows (Dennison & Alberte, 1985).

1.3 Ecosystem services of seagrasses

Seagrasses provide a variety of ecosystem services that are estimated to have a net value of \$19,004 ha^{-1} yr^{-1}, making them among the most productive ecosystems on earth (Costanza et al., 1997; Duarte & Sand-Jensen, 1990; Duffy, 2006). Ecologically, they are important in the production of organic carbon which is either stored in the sediment or exported to adjacent ecosystems (Duarte & Cebrián, 1996). Seagrass ecosystem accounts for close to 10 -18% of the total ocean carbon burial with an estimated carbon accumulation rate at 48-112 TC yr^{-1}, (Kennedy et al., 2010; Mcleod et al., 2011). Although, there is no precise estimate of the total burial, it is estimated that close to 50% of seagrass carbon could be from external sources (Kennedy et al., 2010). They are important ecosystem engineers as they alter the local hydrodynamic environment and stabilize sediment, through trapping of the suspended sediment by their canopies and the complex root systems which in turn creates positive feedbacks that are vital for their growth (Carr et al., 2010). By regulating the water quality and nutrient cycle, they contribute to organic carbon, production and trophic transfers to adjacent habitats (McGlathery et al., 2007; Orth et al., 2006). Seagrasses provide refugia and nursery grounds for many other species, including commercially important faunal species. In this way, seagrass directly and indirectly supports important subsistence and commercial fisheries, both within the seagrass and other connected habitats (Mumby, 2005; Nagelkerken et al., 2002) and invertebrate species (Berkström, 2012). Furthermore, these meadows are recognised as extremely important coastal ecosystems for their role in increasing biodiversity by providing a complex habitat for macro-organisms such as fish, birds and invertebrate species (Heck & Valentine, 2006). In addition to the known ecosystem services of seagrasses, a recent study has reported that seagrasses contribute to human health and that of marine organisms by reducing microbiological contamination of the seawater (Lamb et al., 2017). Since seagrasses are contiguous to mangroves, there is an overlap of these functions between the two systems. Studies have reported that mangrove organic carbon is exported to the adjacent seagrass beds (Bouillon et al., 2007; Bouillon et al., 2004; Hemminga et al., 1994). Furthermore both habitats act as an overlapping nursery habitat for juvenile fish (Lugendo et al., 2006; Nagelkerken et al., 2002). In addition to the tangible ecological values, seagrasses provide numerous socio-economic benefits to the local communities. Among them is the provision of fishing and collecting grounds for finfish, shellfish, bait, medicines and fertilizers and “good” substrate for seaweed farming. They are also linked to religious, aesthetic and spiritual values that are important to coastal communities worldwide (Unsworth & Cullen, 2010). Within the WIO region seagrasses provide an important habitat for commercially important fish, thereby indirectly supporting income and food security for coastal communities. Furthermore the meadows uphold a large diversity of animal and plant species, among which turtles and dugongs (UNEP, 2012).

1.4 Threats to seagrasses

Seagrass beds are declining rapidly from the world and almost a third of all seagrass areas have been lost in the last 140 years (Waycott et al., 2009), with a current loss rate of about 1.5% year^{-1} (Pendleton et al., 2012). Major drivers of decline in seagrass vegetation are various types of disturbance, some of which are associated with natural factors such as herbivory (Alcoverro & Mariani, 2002; Eklöf et al., 2009), sedimentation (Cabaço et al., 2008), diseases, strong waves and storm surges. Direct human impacts to seagrasses, in addition to the major indirect impacts threaten the habitat particularly in densely populated areas. These include: (i). fishing and aquaculture, (ii). introduction of exotic species, (iii). boat mooring and anchoring, (iv). habitat alteration (dredging, reclamation and unsustainable coastal developments) which is perhaps the biggest threat at present. Fishing methods such as trawling may significantly affect seagrasses by direct removal (Orth et al., 2006; Waycott et al., 2009). Nutrient over-enrichment of coastal waters and sedimentation upstream associated with effluents discharge and agricultural activities is responsible for increased turbidity and hence decline in water clarity (Larkum et al., 2006). However, quantitative reports on the patterns of decline and possible causes are still insufficient in some regions (UNEP, 2008; Waycott et al., 2009).

In the WIO region, rapid population increase together with the influx of populations to the coastal zones is a major source of pressure to the seagrass meadows. Damage to seagrass meadows can for example lead to increased turbidity in coral reefs due to a decline in the filtering capacity of seagrasses which has both negative aesthetic and environmental consequences (UNEP, 2012). A variety of threats have been recognized within the region. As for example nutrient enrichment through anthropogenic activities along the coast, direct damage from boat propellers, anchoring or dredging. Also imbalance within the seagrass system is feared as overfishing can lead to, among others, increased grazing pressure by sea urchins. The later leads to fragmentation of the meadows, further debilitating the health of the system. Finally, climate change could significantly influence seagrass meadows within the region but due to the large scale and diversity of potential impacts, the specific effect of climate change is difficult to predict. Increased carbon uptake by seagrasses could for example protect adjacent coral reef systems against climate change induced ocean acidification, where the seagrass meadows function as a buffer (UNEP, 2012).

1.5 Coverage, floristic composition, and distribution of the seagrasses of Kenya

The Kenyan coastline stretches to ~600km and has a shallow, moderately-thin continental shelf bordering the Indian Ocean with the substrate comprising of carbonate sands resulting from eroding reefs and thus the seagrass habitats mainly consist of sediments from coral limestone (Gullström et al., 2002; Ochieng & Erftemeijer, 2003).

All the 12 seagrass species described in the Western Indian Ocean region occur in Kenya. These species are widely distributed along the Kenyan coast (Table 1) and most of them occur in either monospecific or mixed stands. Seagrasses in Kenya are represented in three families: *Zostera capensis* belonging to Zosteraceae; *Thalassia hemprichii, Halophila ovalis, Halophila minor, Halophila stipulacea and Enhalus acoroides (*all Hydrocharitaceae) and *Cymodocea rotundata Cymodocea serrulata, Halodule uninervis, Halodule wrightii, Syringodium isoetifolium, Thalassodendron ciliatum (*all Cymodoceaceae). *Zostera capensis* is a rare species in Kenya (Gullström et al., 2002; Ochieng & Erftemeijer, 2003). These species occur in a succession regime with small species such as *Halodule spp*. and *Halophila spp*. being pioneer species and the larger seagrasses such as *T. hemprichii*, *T. ciliatum* and *E. acoroides* forming the more dominant climax communities.

Seagrasses in Kenya are estimated to cover 33,600ha (UNEP, 2009). However, this could be an underestimate as no comprehensive mapping has been done. The most important sites in the country, in terms of extent and diversity of seagrasses, occur between Lamu and Kiunga, Malindi, Mombasa, Gazi Bay Mida Creek and Diani-Chale lagoon (Ochieng & Erftemeijer, 2003).

There has been significant loss of seagrass along the coast, initially attributed to human related activities, but now also to sea urchin herbivory. In Diani-Chale lagoon for instance, preliminary studies indicate that *T. ciliatum* experienced a net loss of more than 50% of cover in the last two decades. These degraded sites were also found to have a density of the sea urchin *Tripneustes gratilla* of more than 37 individuals/m², while healthy sites had a density of four individuals/m² (KWS, 2013). The most affected species has been *T. ciliatum*. Natural recovery of seagrasses has been observed in several areas along the Kenyan coast after reducing anthropogenic stressors (KWS, 2013). Nutrient enrichment is another threat that causes changes in species assemblages and dynamics of nutrient cycling in these areas as well as damage from boat propellers and dredging (Orth et al., 2006).

Table 1: Description of seagrass species along the Kenyan coast

Number of species	**County**						
	Kwale			**Mombasa**	**Kilifi**		**Lamu**
	Funzi-Shimoni-Vanga	**Gazi**	**Diani-Chale**	**Mombasa**	**Bofa**	**Watamu**	**Lamu**
Cymodocea rotundata	x	x	x	x	x	x	
Cymodocea serrulata	x	x	x	x	x	x	x
Enhalus acoroides	x	x					x
Halodule wrightii	x	x	x	x	x	x	x
Halodule uninervis	x	x	x	x	x		
Halophila minor	x	x	x				
Halophila ovalis	x	x	x			x	x
Halophila stipulaceae	x	x	x	x	x		x
Thalassia hemprichii	x	x	x	x	x	x	x
Thalassodendron ciliatum	x	x	x	x	x	x	x
Syringondium isoetifolium	x	x	x	x	x	x	x
Zostera capensis	x	x				x	
Total species	12	12	10	8	8	8	8

(Data source (KWS 2013)

Gazi Bay (4°25’S, 39°30’E), is located in Kwale County, about 55 km south of Mombasa City (Fig. 2). Seagrasses which are found at the centre of the bay cover an area of ~ 7 km^2 (Bouillon et al., 2007; Dahdouh & Speybroeck, 1999). The main climatic seasons are the southeast monsoon (May-September) and the northeast monsoon (November-March). An offshore (shoreward) wind prevails throughout the year. Rains occur in March-May and to a lesser degree in October and November. However, inter-annual shifts in these seasons are common (Schott, 2009). The tide at Gazi Bay is normally mixed semi-diurnal, with a range of 3m at springs and 1.4m at neaps. These tides generate strong reversing currents throughout the tidal creeks and relatively weaker currents in the open regions of the bay. The shoreward wind and the tidal currents combine to vertically mix the water column in the bay, leading to formation of homogeneous water, with a salinity range of 34.5-35.5 PPT (Kitheka, 1996). A significant lateral and vertical salinity gradient develops during the wet seasons as a result of increased river runoff (Kitheka, 1996). The flushing ability of mixed semi-diurnal tides in the bay varies depending on tidal range, tidal prism, and the nature of the tide. Spring tides are characterized by swift flows and tend to rapidly disperse low salinity water. The rates of water exchange are also high during the spring as compared to the neap tides. On the other hand, currents at neaps are sluggish and inhibit flushing of brackish water. This flushing pattern of the tide combined with river runoff has a far-reaching effect in the form of nutrient and material exchanges on the linkage between mangrove, seagrass, and coral reef ecosystems (Haas, 1977).

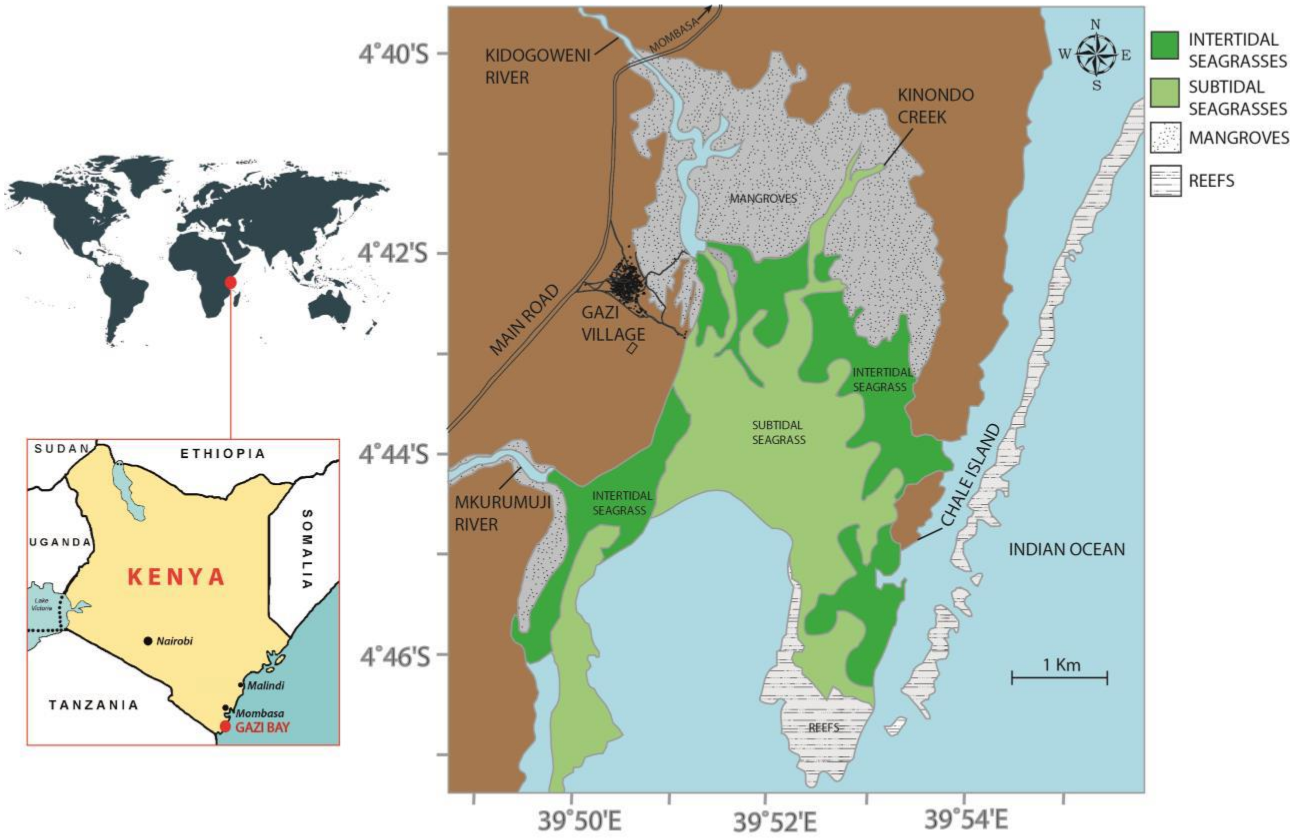

Fig. 2: Map of Gazi Bay, Kenya and the seagrass ecosystem (Ankje & Githaiga 2016)

All the twelve seagrass species recognized along the East African coast have been recorded here, usually attached to both soft and hard substrates in the bay (Bandeira, 2002; Coppejans et al., 1992; Gullström et al., 2002; Isaac & Isaac, 1968). The seagrass community of the bay is comprised of dominant species such as: *Thalassodendron ciliatum* (Forssk.) den Hartog, *Thalassia hemprichii* (Enhrenberg) Aschers., *Enhalus acoroides* (L.f.) Royle, *Syringodium isoetifolium* (Aschers.) Dandy, *Cymodocea rotundata* Ascherson, and *Cymodocea serrulata* (R. Braun) Aschers. & Magnus. These are observed to grow either as monospecific stands (Fig 3) or mixed with other seagrass species with their coverage extending between the intertidal to the subtidal areas in sandy and rocky substrates (Coppejans et al., 1992). Seagrasses show clear zonation patterns with water depth, sediment structure and exposure to air and sunlight during low tide. Species that are tolerant to exposure are found higher up on the intertidal areas, while those that cannot withstand exposure appear in submerged pools. Minor species comprise of *Halodule uninervis* (Forssk.) Aschers., *Halodule wrightii* (Aschers.), *Halophila minor* (Zoll.) den Hartog, *Halophila ovalis* (Braun) Hooker, *Halophila stipulaceae* (Forssk.) Aschers. and *Zostera capensis* (Setch), which was recorded in the past (Isaac & Isaac, 1968), but not in a more recent assessment (Coppejans et al., 1992).

Thalassia hemprichii

Enhalus acoroides

Thalassodendron ciliatum

Syringodium isoetifolium

Photo: Githaiga, 2016

Fig. 3: The four dominant seagrass species of Gazi Bay

The seagrass species mainly appear in multispecific stands in the soft mud, sandy substrates in the intertidal and occasionally in the shallow subtidal areas (Coppejans et al., 1992). The meadows, of the bay form both monospecific and multispecific stands and are usually luxuriant for most parts of the year except during short periods of intense grazing and desiccation which were observed during the present study in the lower part of the bay particularly for *S. isoetifolium* and *T. hemprichii.*

Macroalgae species are also abundant occurring together with the seagrass meadows. Common species include; *Gracilaria cortica*, *Gracilaria saloicornia*, *Halimeda* species, *Cystoseira trinoids*, *Dictyota spp*, *Hyponea cornata*, *Sargassum spp*, *Turinaria decudrens*, *Ulva partusa* and *Ulva reticulata*. The social-ecological significance of the bay is evident as the local communities from the surrounding villages and their environs converge daily at the fish landing beaches to buy fish and other marine products. The seagrass meadows of Gazi Bay fall under Diani-Chale Marine National Reserve which was established in 1994 (Tuda & Omar, 2012). It also forms the northern boundary of the proposed Transboundary Marine Conservation Area between Kenya and Tanzania (TBCA) (MPRUT-KWS, 2016). At present, there is no enforcement of its protection status due to weak governance structures. Use of seine and drag nets by the artisanal fishermen is a daily activity in the shallow waters of the bay and is thus a major cause of degradation of the seagrass together with herbivory especially by the sea urchins. The increased use of seine nets by the local fishermen and the land based activities such as agricultural and mining are perceived by the local community as the most likely causes of the increasing sedimentation that is being observed in the bay. Already the ecosystems are showing signs of degradation, a trend that needs urgent remedy if the integrity of these ecosystems and the services they provide are to be sustained.

1.6 Seagrass morphological structure in relation to the carbon pools

Seagrass anatomy consists of vascular to discrete photosynthetic shoots above-ground coupled with an elaborate below-ground systems (Guy, 2010). These form the basis for the classification of the living biomass into above and below-ground biomass (Fig.

4). The above-ground components comprise mostly the aerial shoots. On the other hand the below-ground components comprise the living roots, rhizomes, the necromass (litter or any detrital materials) and the sediment organic carbon (C_{org}) which comprise of the soil organic matter (Howard et al., 2014). The sediment organic pool forms the bulk of the total organic carbon (Fourqurean et al., 2012). It constitutes the autochthonous and allochthonous materials that have been buried over longer timescales forming the sediment C_{org}. The above-ground part which is visible and easy to monitor constitutes less than a third of the living biomass due to high turnover rates occasioned by destruction (grazing, damage by fishing nets and harvesting by strong waves) as well as decomposition due to the aerobic conditions.

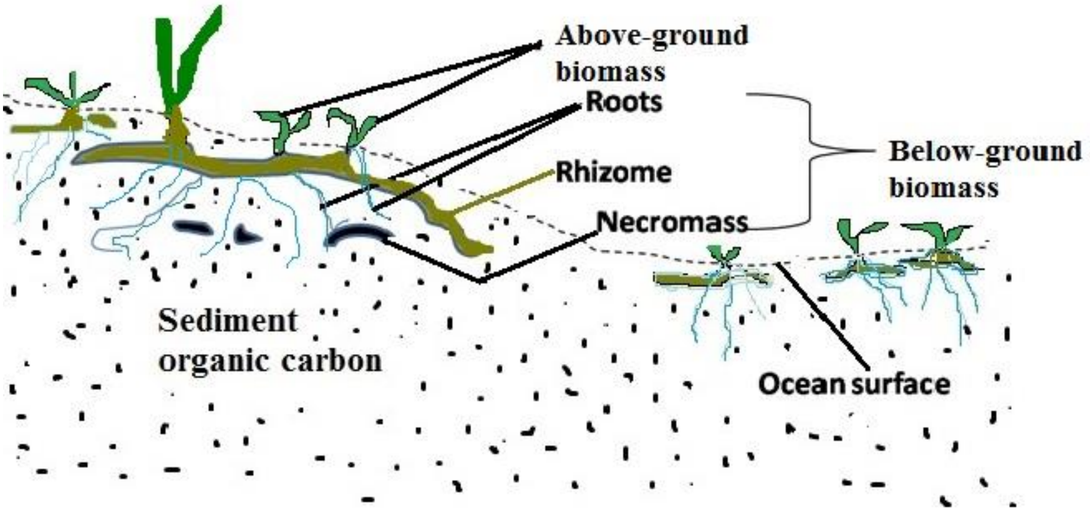

Fig. 4: Diagrammatic illustration of the various carbon pools in the seagrass beds

Below-ground production dynamics are an important aspect of seagrass ecology, since the root and rhizome tissues comprise between 70-80% of the plant's total biomass, functioning as both a storage organ and an anchor for the plant (Duarte, 1991; Kaldy & Dunton, 2000). However, there are some variations depending on the source and the sampling methods. For instance in a review of a global data set, Duarte & Chiscano (1999) observed that the above-ground biomass was almost equal to the below-ground biomass (Mean ± S.E) 223.9 ± 17.5 and 237.4 ± 28 g DW m^{-2} respectively. A review of data from Africa in this study observed that the mean above and below-ground biomass of seagrass were 174.4 and 474.6 g DW m^{-2} respectively. The relative distribution of biomass between above and below-ground components can also be used to deduce the availability of nutrients and light to the plant (Ralph et al., 2007). Studies investigating the biomasses of seagrass species have revealed that the plant's morphology, growth strategy and size can significantly affect the ratio of its above-ground to below-ground biomass (Duarte, 1991). For example (Di Carlo & Kenworthy (2008) found that smaller species such as *H. wrightii* have a reduced canopy cover, and thus are generally shallow rooted with thinner diameter rhizomes and a lower overall biomass compared to their larger canopy-forming associate, *Thalassia testudinum* (*T. testudinum*).

1.7 Seagrasses and the blue carbon link

Blue carbon refers to the carbon stored in mangroves, salt marshes and seagrasses within the sediment, the living biomass above-ground (leaves branches and stems), the living biomass below-ground (roots and rhizomes) and the non-living dead matter (e.g. litter and dead wood) (Mcleod et al., 2011). The carbon sequestered in the coastal soils can be extensive and remain trapped for long period of time (centuries to millennia) resulting in very large carbon stocks (Duarte et al., 2005; Lo Iocano et al., 2008). This is due to the anaerobic conditions that prevent rapid decomposition of the organic matter which would lead to remineralisation plus the high accretion rates (Chmura et al., 2003). Seagrass meadows in particular are estimated to store up to 19.8 (Pg) of organic carbon globally (Greiner et al.,2013): which is comparable to the amount of carbon stored in all the worlds' tidal marshes and mangrove forests combined (Fourqurean et al., 2012). Some of the carbon comes from the trapping of particles in the water column rich in organic matter with results averaging around 4.1%

(Kennedy et al., 2010). This trapping process differs between species as each comprise of varying densities and structural canopies, and that carbon accumulation rates fluctuate in established seagrass meadows depending on the species present, sediment characteristics and the depth range of the seagrass habitats (Greiner et al., 2013; Guy, 2010).

Seagrass sediments contain, on average, twice as much below-ground carbon per hectare as terrestrial forest soil and can be preserved for millennia (Fourqurean et al., 2012). Although terrestrial plant biomass is 95% higher than oceanic plant biomass, the ocean cycles almost the same volume of carbon annually (Nellemann et al., 2009). Current records for the carbon dynamics of seagrasses within Africa are particularly scarce (Fourqurean et al., 2012). This study found that the little research conducted in Kenya, Tanzania and eastern South Africa looks primarily at seagrass ecology and diversity, lacking any empirical based estimates of their potential in carbon capture and storage. With an ever increasing interest in the carbon capture and storage of seagrass, there is a clear need for more extensive research, focusing on the morphology of the different seagrasses. Although the structural differences between species need further investigation, the characteristics of the sediment on which these species reside are equally as significant, particularly in accurately estimating the true C_{org} content of the seagrasses. Past studies of marine environment in Gazi Bay has tended to concentrate on mangroves (Huxham et al., 2010; Kairo et al., 2008; Tamooh et al., 2008) with less attention given to seagrasses. Similar trends can be said of the wider Western Indian Ocean coastline where mangroves and coral reefs have been widely studied compared to the seagrasses (Gullström et al., 2002), hence the rationale for a comprehensive study of the seagrasses. The focus of this thesis therefore was to develop a better understanding of the ecosystem function of the seagrass as a carbon sink. This knowledge is useful in seagrass conservation and management and in particular when exploring the potential for inclusion of seagrasses in carbon offsetting projects.

The following specific questions are addressed in the thesis

i. What is the knowledge status of biomass and productivity of seagrasses in Africa?

ii. What is the carbon storage capacity of the dominant seagrass at Gazi Bay, Kenya?

iii. What is the effect of seagrass removal on surface elevation, sediment accretion and carbon dynamics and how does seagrass loss impact on the life of benthic infauna communities?

iv. What are the productivity levels of the seagrasses of Gazi Bay, Kenya?

v. What is the implication of this study to seagrass conservation, what are the potential areas for future research and how can seagrass ecosystem services be bundled to those of mangroves?

Chapter two

2.0 Biomass and productivity of seagrasses in Africa

Abstract

There is growing interest in carbon stocks and flows in seagrass ecosystems, but recent global reviews suggest a paucity of studies from Africa. This study reviews work on seagrass productivity, biomass and sediment carbon in Africa. Most work was conducted in East Africa with a major geographical gap in West Africa. The mean above-ground, below-ground and total biomasses from all studies were 174.4, 474.6 and 514 g DW m^{-2}, respectively with a global range of 461-738 g DW m^{-2}. The mean annual production rate was 913 g DW m^{-2} yr^{-1} (global range 816 - 1012 g DW m^{-2} yr^{-1}). No studies were found giving sediment organic carbon, demonstrating a major gap in seagrass blue carbon work. Given the small numbers of relevant papers and the large geographical areas left undescribed in Africa, any conclusions remain tentative and much remains to be done on seagrass studies in Africa.

Key words: Africa, blue carbon, productivity, seagrasses

2.1 Introduction

Understanding the role of vegetated coastal ecosystems in global carbon dynamics is a field of growing interest since knowledge of natural carbon sinks and flows can contribute to effective management of human impacts on the climate. Currently, our understanding of the roles of different ecosystems in the global carbon budget is limited by uncertainty about, and ignorance of, both individual ecosystems and their ecological connectivity. Vegetated coastal ecosystems that, in the past, have been relatively neglected have more recently received considerable attention following the 'blue carbon' initiative, which established a clear distinction between the aquatic and terrestrial organic carbon sinks and helped to highlight the high relative efficiency of vegetated coastal sinks (Nellemann et al. 2009, http://thebluecarboninitiative.org/). Of the three key 'blue carbon' habitats – salt marsh, mangrove and seagrass meadows – seagrasses are the most extensive but least studied. Available reviews of seagrass biomass and carbon flows globally (Duarte and Chiscano 1999, Fourqurean et al. 2012) reveal that the majority of studies have been done in Western Europe, the Mediterranean, the Caribbean, Australia and the American coasts. This is an indication of the relative paucity of information about seagrasses in African waters. Globally, seagrass ecosystems are estimated to store as much as 19.9 Pg of organic carbon and the oceans may bury an estimated 27.4 Tg C yr^{-1} in seagrass meadows (Fourqurean et al., 2012). The average standing stock of seagrass is estimated at 460 g DW m^{-2} while the average production is 5.0 g DW m^{-2} d^{-1} (Duarte and Chiscano 1999). Since these figures have been derived without much contribution from seagrass studies in Africa, estimates of the global seagrass carbon budget may change substantially if sequestration and storage rates in African systems are distinctive. Bearing in mind that seagrasses host a high species diversity globally (Short et al., 2007) and the fact that the role of seagrasses in carbon fluxes is acknowledged (Mateo et al., 2006), there is a need to understand variation in biomass and carbon storage across species and sites.

2.1.1 General objective

Carry out a comprehensive assessment of all accessible literature on African seagrass species to determine current knowledge on biomass stocks and productivity.

2.1.2 Specific objectives

i. Establish the number of seagrass biomass and productivity publications, reports and theses
ii. Identify the geographic distribution of these data around Africa
iii. Establish the above-ground, below-ground and total biomass values for 14 seagrass species studied in Africa, pooled across all reported sites
iv. Establish the total biomass values for the seagrass species in different regions of Africa
v. Establish the productivity values of the studied seagrass species in Africa

2.2 Materials and methods

Both the primary and grey literature was used. Four search engines - Google Scholar, Yahoo, Science Direct and ISI Web of Science - were used when looking for any available information on seagrass biomass and productivity studies in Africa up to the end of the year 2015. In addition, manual searches from libraries were done especially for the grey literature. Several researchers thought to have been involved in seagrass biomass and carbon studies in Africa were contacted to provide any available information. The search terms used were ‘seagrass’ in combination with one of the following: “above-ground biomass” ,“below-ground biomass”, “biomass stocks”, “carbon burial”, “productivity”, “Africa”, “target seagrass species” and “names of countries” along the African coasts. Where data on biomass and productivity were given as a range with no means reported, the mid-point was taken as an estimate of the mean from that study. In some cases, relevant information was not given in the text but could be reliably estimated from the figures. Data on biomass and productivity rates for different species at different sites were investigated and summarized.

2.3 Results

2.3.1 Number of publications, reports and theses

Of the over 300 abstracts initially found, 32 papers and eight reports or theses gave information on biomass and/or productivity in Africa. Of these, 25 reported on seagrass biomass stocks alone while 15 reported entirely on productivity or a combination of biomass stocks and productivity. Six reports or theses were on biomass stocks and three on productivity, though one thesis reported on both biomass and productivity (Table 2).

Table 2: Published papers, reports/theses on seagrass biomass and productivity studies around Africa

Country	Biomass stocks		Productivity	
	Papers	Reports/theses	Papers	Reports/theses
Algeria			Semroud, 1990	
Egypt	Gab-Alla, 2001 Mostafa, 1996			
Kenya	Duarte et al., 1998 Ochieng & Erftemeijer 1999 Kamermans et al., 2002 Ochieng & Erftemeijer 2003 Uku & Björk 2005	Gwada, 2004	Duarte et al., 1996 Hemminga et al., 1995 Ochieng & Erftemeijer 1999 Uku & Björk 2005	Ochieng et al.,1995
Libya			Pergent et al., 2002	

Mauritania	Laan & Wolff 2006 Vermaat et al., 1993		Vermaat et al., 1993 Van Lent et al., 1991	
Mauritius	Daby, 2003			
Morocco		Bououraour et al., 2015 Boutahar et al., 2015		
Mozambique	Bandeira, 1997 Bandeira, 2002 de Boer, 2000 Martins & Bandeira, 2001 Paula et al., 2001	Larsson, 2009	Bandeira, 2002 de Boer, 2000	Bandeira, 2000 Larsson, 2009
Seychelles	Aleem, 1984			
South Africa	Adams et al., 1992 Christie, 1981 Hanekom et al., 1988 Talbot et al., 1987	Grindley, 1976		

Tanzania	Eklöf et al., 2005 Gullström et al., 2006 Kamermans et al., 2002 Lugendo et al., 2001 Lyimo et al., 2006 Lyimo et al., 2008	Mvungi, 2011	Lyimo et al., 2006	
Tunisia	Sghaier et al., 2011 Sghaier, 2012		Sghaier, 2012	

These peer reviewed papers, together with the reports and theses, come from studies carried out primarily on the Western Indian Ocean (WIO) coastline, especially in Kenya (Gazi Bay and around Mombasa), Tanzania (sites around Zanzibar Island), Mozambique (Inhaca Island), Aldabra Island in the Seychelles Republic, Mauritius and along the coast of South Africa. Other studies have been conducted at Sharm El-Moyia Bay along the Red Sea coastline of Egypt, Banc d' Arguin in N.W Mauritania and at some bays and lagoons such as Ghar El Melh Lagoon in Northern Tunisia and at Montazah Bay of Egypt on the southern Mediterranean Sea (Fig. 5). Some studies (unpublished) have recently been reported from Marcha Bay, Jbel Moussa Bay and the Atlantic coast of Morocco (Table 3). Data were available for 14 species, with biomass data available for 13 species (Table 3),

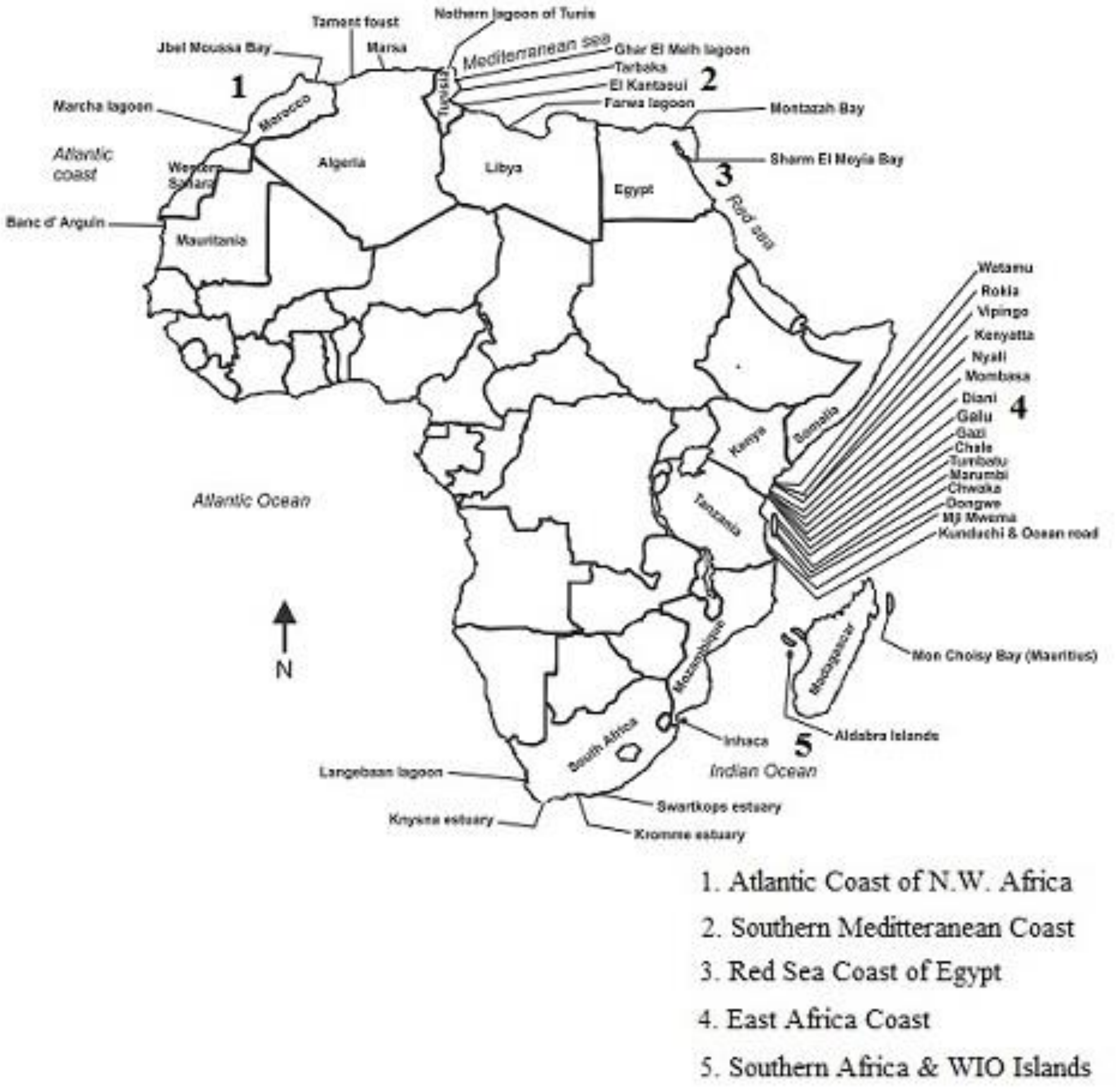

Fig. 5: Sites along the coastline of the African continent where seagrasses have been studied

Table 3: Mean (± S.E) values for above-ground, below-ground and total biomass reported in the cited studies for different seagrass species at sites around Africa. (Value in parenthesis (n) where available represents the sample size).

Country	**Location**	**Latitude & Longitude**	**Species**	**Above-ground biomass (g DW m^{-2})**	**Below-ground biomass (g DW m^{-2})**	**Total biomass (g DW m^{-2})**	**Reference**
Egypt	Montazah Bay	31°12'N, 29°55'E	*Cymodocea nodosa*	287			Mostafa, 1996
	Sharm El Moyia Bay	27° 9'N 34°3'E	*Halophila stipulaceae*			270	Gab-Alla, 2001
Kenya	Galu	4° 18'S, 39°32'E	*Thalassodendron ciliatum*			40.6 ± 16.4	Uku et al., 1996
	Diani	4° 18'S, 10°32'E	*Thalassodendron ciliatum*			279.3 ± 97.6	Uku et al., 1996
	Diani	4° 18'S, 10°32'E	Mixed			430 (33)	Kamermans et al., 2002
	Gazi	4° 25'S, 39°30'E	*Thalassodendron ciliatum*	316.1± 41	368.1 ± 22	725.5 ± 252.5	Ochieng & Erftemeijer 2003

	Chale lagoon	4° 25’S, 39°30’E	*Thalassodendron ciliatum*		243.4		Duarte et al., 1998
	Mombasa	4° 2’S, 39°41’E	Mixed			471.6 ± 66.7	Kamermans et al., 2002
	Roka	1° 36’S, 39°12’E	Mixed			644 (7)	Kamermans et al., 2002
	Mombasa Marine Park	4° 2’S, 39°41’E	Mixed			760 ± 96	Ochieng & Erftemeijer 1999
	Nyali	4° 03’S, 39°43’E	*Thalassodendron ciliatum*–North East monsoon	277.4 ± 36.3	364.9 ± 83.5		Gwada, 2004
			Thalassodendron ciliatum-South East monsoon	269.5 ± 65	312.0 ± 123		
	Nyali	4° 03’S, 39°43’E	Mixed			604 (33)	Kamermans et al., 2002
	Kenyatta	4° 00’S, 39°44’E	Mixed			233 (33)	
	Watamu	3° 23’S, 39°59’E	Mixed			457 (33)	

Mauritania	Banc d’ Arguin	20^{o} 35’N, 16^{o}15’W	Mixed			335	Vermaat et al., 1993
			Mixed		255.0		Laan & Wolff 2006
Mauritius	Mon Choisy Bay	20^{o} 17’S, 5733’W	*Syringodium isoetifolium*			129.3	Daby, 2003
			Halophila ovalis			102.5	
Morocco	Marcha lagoon	40^{o} 39’N, 8^{o}48’W	*Cymodocea nodosa*	8.02-61.2	10.8 -235		Boutahar et al., 2015
	Atlantic coast	23^{o} 30’N, 15^{o}56’W	*Zostera noltii*	32 - 259	21- 314		Bououarour et al., 2015
	Jbel Moussa Bay	30^{o} 8’N, 5^{o}21’W	*Zostera noltii*	3.08 ± 1.12		7.72 ± 1.38	
Mozambique	Inhaca	25^{o} 58’S, 32^{o}55’E	*Thalassodendron ciliatum*	355.2 ± 111.1	792. 4± 342.9	1148 (30)	Bandeira, 1997
			Zostera capensis (Summer)	15.7 ± 4.5	173.4 ± 47.5	190 ± 51.2 (10)	de Boer, 2000
			Cymodocea serrulata (Summer)	34.1 ± 18.6	38.6 ± 14.0	82.0 ± 30.8 (10)	
			Halodule wrightii (Summer)	16.0 ± 22.2	17.1± 14.5	22.2 ± 21.7 (10)	

		Zostera capensis (Winter)	25.7 ± 8.0	198.9 ± 75	219.5 ± 78.1 (10)	
		Cymodocea serrulata (Winter)	17.6 ± 15.2	27.0 ± 14.4	43.1 ± 21.8 (10)	
		Halodule wrightii (Winter)	6.9 ± 5.5	18.1 ± 6.5	22.9 ± 8.2 (10)	
Inhaca (Northern Bay)	25º 58’S, 32º55’E	*Thalassia hemprichii*	154.4 ± 22.7	633.0 ± 163.5	787.4 ± 233.8	Martins & Bandeira 2001
		Halodule wrightii			30.7 ± 11.9	
		Halophila ovalis			0.6 ± 0.4	
		Zostera capensis			4.8 ± 2	
		Cymodocea rotundata			39.9 ± 18.7	
Inhaca (Southern Bay)	25º 58’S, 32º55’E	*Thalassia hemprichii*	147.1 ± 68.65	1729.7± 495.25	1876 ± 389.4	
		Halodule wrightii			0.9 ± 0.7	
		Halophila ovalis			0 ± 0	
		Zostera capensis			0 ± 0	

			Cymodocea rotundata			4.5 ± 4 3	
		25^{o} 58’S, 32^{o}55’E	*Thalassodendron ciliatum*	50.1-170.7	0.04 -1471.1		Paula et al., 2001
			Thalassia hemprichii	14.2-291.1	9.21 – 1307.6		
			Zostera capensis	7.9 – 51.3	66.0 – 195.5		
	Inhaca	25^{o} 58’S, 32^{o}55’E	*Thalassia hemprichii*	49.8 ± 3.1			Larsson, 2009
Seychelles	Aldabra Island	9^{o} 41’S, 46^{o}42’E	*Halodule uninervis*			243	Aleem, 1984
			Halophila ovalis			46.5	
			Mixed species			425	
			Thalassia hemprichii			412.5	
			Thalassodendron ciliatum			468	
			Syringodium isoetifolium			435	
South Africa	Knysna estuary	34^{o} 05’S, 23^{o}21’E	*Zostera capensis*	206			Grindley, 1976

	Langebaan lagoon	33^{o} 01’S, 18^{o}01’E	*Zostera capensis*	217			Christie, 1981
	Swartkops estuary	33^{o} 52’S, 25^{o}38’E	*Zostera capensis*			75.8-124.7	Talbot et al., 1987
	Kromme Estuary	34^{o} 09’S, 24^{o}51’E	*Zostera capensis* (Winter 1979)	105 ± 44			Hanekom et al., 1988
			Zostera capensis (Summer 1980)	55 ± 21			
	Kromme Estuary	34^{o} 09’S, 24^{o}51’E				244	Adams et al., 1992
Tanzania	Chwaka	6^{o} 10’S, 39^{o}26’E	*Thalassia hemprichii*	897.2 ± 754.8	-	-	Kamermans et al., 2002
	Chwaka	6^{o} 10’S, 39^{o}26’E	*Thalassia hemprichii*			85	Eklöf et al., 2005
			Enhalus acoroides			100	
			Thalassodendron ciliatum			90	
	Chwaka	6^{o} 10’S, 39^{o}26’E	Mixed	62 -105			Gullström et al., 2006

	Chwaka	6^{o} 10’S, 39^{o}26’E	*Enhalus acoroides*	76.4-105.1 (20)			Gullström et al., 2008
			Thalassia hemprichii	61.8-99.1(20)			
			Mixed	94.5 (20)			
	Jambiani	6^{o} 6’S, 39^{o}32’E (With Seaweed)	*Thalassia hemprichii*	90.4 ±16.1(5)	185 ± 32.9 (5)	276 ± 48.7 (5)	Lyimo et al., 2006
		(Non Seaweed)	*Thalassia hemprichii*	609 ± 71.5 (5)	2455±726 (5)	3063 ± 715 (5)	
	Chwaka	6^{o} 10’S, 39^{o}26’E (With Seaweed)	*Thalassia hemprichii*	108 ± 23.8 (5)	179±57.9 (5)	286 ± 81.5 (5)	
		(Non-Seaweed)	*Thalassia hemprichii*	175 ± 19.0 (5)	220 ± 3.4 (5)	393 ± 18.7 (5)	
		(With Seaweed)	*Enhalus acoroides*	177 ± 85.5 (8)	563 ± 272 (8)	740 ± 358 (8)	

	(Non-Seaweed)	*Enhalus acoroides*	199 ± 54.5 (8)	415 ± 114 (8)	614 ± 98.9 (8)	
Marumbi	6^{o} 13’S, 39^{o}28’E (With Seaweed)	*Thalassia hemprichii*	465 ± 183(5)	90 4 ± 129 (5)	1369 ± 266 (5)	
	(Non-Seaweed)	*Thalassia hemprichii*	301 ± 42.1(5)	442 ± 66.9 (5)	742 ± 81(5)	
	(With Seaweed	*Enhalus acoroides*	144 ± 63.0 (8)	810 ± 356 (8)	953 ± 418 (8)	
	(Non-Seaweed)	*Enhalus acoroides*	143 ± 57.5 (8)	512 ± 207 (8)	655 ± 264 (8)	
Chwaka	6^{o} 10’S, 39^{o}26’E (With Seaweed)	Mixed			142.4-1652	Lyimo et al., 2008
	(Non-Seaweed)				212.9-1829	
Jambiani	6^{o} 6’S, 39^{o}32’E	mixed			880.4-3467	

		(With Seaweed)					
		(Non-Seaweed)				203.4-3810	
	Kunduchi& Ocean road	6^{o} 40’S, 39^{o}13’E	Mixed			0.25 – 135.29	Lugendo et al., 2001
	Ocean road	6^{o} 45’S, 39^{o} 20’E	*Thalassia hemprichii*	307.0 ±74.9	412.1 ± 93.3		Mvungi, 2011
			Cymodocea serrulata	202.7 ± 69.6	267.7 ± 147.9		
	Mji-mwema	6^{o} 38’S, 39^{o}40’E	*Thalassia hemprichii*	267.0 ± 43.8	1177.4 ± 265.2		
			Cymodocea serrulata	352.2 ± 141.7	737.2 ± 260.8		
	Kiwengwa	5^{o} 60’S, 39^{o}23’E	Mixed			115 (30)	Kamermans et al., 2002
	Dongwe	6^{o} 11’S, 39^{o}32’E	Mixed			224 (21)	
Tunisia	Ghar El Melh	37^{o} 09’N, 10^{o}13’E	*Cymodocea nodosa*	97.3 ± 51.4	264.7 ± 69.2	327.7 ± 86.1	Sghaier, 2012
	Lagoon		*Cymodocea nodosa*	82.5 ± 15.38	333.9 ± 49.4	413.8 ± 46	Sghaier et al., 2011

	Northern lagoon of Tunis	37° 14’N, 09° 56’E	*Zostera noltii*	79.75	Imen et al., 2014

NEM –North East Monsoon, SEM – South East Monsoon. In some studies, the total biomass is not equal to the sum of the above-ground and the below-ground due to differences in the samples sizes but are captured as reported in the studies.

The four families of seagrass and species studied on biomass and productivity in Africa; Hydrocharitaceae (*Enhalus acoroides* (L.F) Royle, *Halophila minor* (Zoll.) den Hartog, *Halophila ovalis* (R.Br.) Hook f., *Halophila stipulaceae* (Forsk.) Aschers and *Thalassia hemprichii* (Enhrenberg) Ascherson) Cymodoceae (*Cymodocea rotundata* Ehrenb. Et Hempx.et Aschers.*Cymodocea serrulata* (R.Br.) Aschers. et Magnus, *Cymodocea nodosa* (Ucria) Aschers., *Halodule uninervis* (Forsk.) Aschers. in Bossier, *Halodule wrightii* Aschers., *Syringodium isoetifolium* (Aschers.) Dandy and *Thalassondendron ciliatium* (Forsk.) den Hartog); Zosteraceae (*Zostera capensis* Setchell, *Zostera noltii*,); Posidonaceae (*Posidonia oceanica* (L.) Delile).

Data on seagrass productivity were available for 10 species (Table 4). Most of the seagrass biomass studies considered mixed stands, but *T. ciliatum* and *T. hemprichii* were the most widely studied individual species, each having been a subject of research in nine out of the 35 locations where biomass studies were reported and in five and six locations, respectively, out of the 18 locations for productivity studies. *Halodule wrightii, C. rotundata, H. stipulaceae* and *H. uninervis* have been studied for biomass stocks in only one location each. Similarly, with the exception of *T. hemprichii* and *T. ciliatum,* a majority of the other species reported in productivity research were studied in only one location (Table 4). *T. ciliatum* was the only species reported to have been studied for all the productivity indices (Table 4).

Table 4: Productivity values expressed as rates of leaf growth, leaf dry weight production, rhizome growth and total dry weight production for different seagrass species at sites around Africa

Country	Location	Latitude & Longitude	Species	Season	Leaf growth (mm shoot^{-1} day^{-1})	Leaf production (g DW shoot^{-1} d^{-1})	Rhizome growth (mm d^{-1})	Total Production (g DW m^{-2} d^{-1})	Reference
Algeria	Marsa	35° 51'N, 10°35'E	*Posidonia oceanica*				0.02		Semroud, 1990
	Tament foust						0.35		
Kenya	Gazi Bay	4° 25'S, 39°30'E	*Thalassodendron ciliatum*					7.5	Hemminga et al., 1995
		4° 25'S, 39°30'E	*Thalassodendron ciliatum*		20.7 ±0.8			4.43 ± 2.7	Ochieng, 1995
		4° 25'S, 39°30'E	Mixed species					2.4 ± 0.6	Ochieng, 1995
	Chale lagoon	4° 25'S, 39°30'E	*Thalassodendron ciliatum*				0.4		Duarte et al., 1996
	Mombasa Marine park	4° 2'S, 39°41'E	Mixed species					8.2 ± 2.8	Ochieng & Erftemeijer 1999

	Nyali	4^o 03’S, 39^{o}43’E	*Thalassia hemprichii*	S.E	17.2 ± 9.5	0.008 ± 0.002	5.5 ± 4.9 (30)	Uku & Björk 2005
				N.E	28.5 ± 4.1	0.008 ± 0.006	5.3 ± 0.5 (30)	
	Vipingo	3^o 45’S, 39^{o}50’E		S.E	17.1 ± 2.6	0.004 ± 0.001	2.4 ± 1.04 (30)	
				N.E	17.1 ± 2.8	0.004 ± 0.002	3.3 ±1.1(30)	
	Nyali	4^o 03’S, 39^{o}43’E	*Thalassodendron ciliatum*	S.E	17.3 ± 1.6	0.005 ± 0.005	3.7 ± 2.4 (30)	
				N.E	18.8 ± 5.9	0.006 ± 0.003	3.1 ± 1.8 (30)	
	Vipingo	3^o 45’S, 39^{o}50’E		S.E	12.4 ± 5.7	0.005 ± 0.002	2.9 ± 2.4 (30)	
				N.E	12.4 ± 5.3	0.004 ± 0.001	1.8 ± 1.6 (30)	
	Nyali	4^o 03’S, 39^{o}43’E	*Cymodocea rotundata*	S.E	12.8 ± 1.6	0.002 ± 0.0005	2.1 ± 0.5 (30)	
				N.E	14.9 ± 1.8	0.002 ± 0.0002	2.3 ± 0.5 (30)	
	Vipingo	3^o 45’S, 39^{o}50’E		S.E	10.0 ± 9.1	0.001 ± 0.0006	2.0 ± 1.1 (30)	

				N.E	11.7 ± 2.0	0.001 ± 0.0005		1.9 ± 1.0 (30)	
Libya	Farwa lagoon	33° 05'N, 11°44'E					0.02- 0.1		Pergent et al., 2002
Mauritania	Banc d Arguin	20° 35'N, 16°15'W	*Cymodocea nodosa*					0.003	Van Lent et al., 1991
			Zostera noltii		0.3				Vermaat et al., 1993
Mozambique	Inhaca Island	25° 58'S, 32°55'E	*Thalassodendron ciliatum*		14.1-18.3				Bandeira, 1997
			Thalassodendron ciliatum		7.5 - 9.5				Bandeira, 2000
			Zostera capensis	Summer	0.7 ± 1.4	0.03		0.18	de Boer, 2000
			Zostera capensis	Winter	0.6 ± 1.1	0.02		0.18	
			Cymodocea serrulata	Summer	2.4 ± 5.3	0.80		0.62	
			Cymodocea serrulata	Winter	1.2 ± 1.5	0.46		0.20	
			Halodule wrightii	Summer	1.5 ± 3.8	0.14		0.20	

				Winter	1.1 ± 2.0	0.08		0.08	
			Thalassia hemprichii		10.4 ± 0.9	0.004		1.08 ± 0.06	Larsson, 2009
South Africa	Kromme estuary	34° 09’S, 24°51’E	*Zostera capensis*					0.93-1.98	Hanekom et al., 1988
Tanzania	Marumbi	6° 13’S, 39°28’E	*Thalassia hemprichii*		13.4 ± 4.7	0.004 ± 0.002		1.97 ± 0.89	Lyimo et al., 2006
	Chwaka	6° 10’S, 39°26’E			17.1 ± 5.2	0.01 ± 0.01		1.86 ± 0.6	
	Jambiani	6° 6’S, 39°32’E			15.8 ± 6.0	0.005 ± 0.002		5.92 ± 2.33	
	Marumbi	6° 13’S, 39°28’E			19.4 ± 7.1	0.02 ± 0.01		2.05 ± 0.9	
	Chwaka	6° 10’S, 39°26’E	*Enhalus acoroides*		24.8 ± 9.4	0.02 ± 0.01		2.77 ± 1.6	
Tunisia	Ghar El Melh Lagoon	37° 09’N, 10°13’E	*Cymodocea nodosa*		3.35 (21)		1.2 ± 1 (21)	1.42 (20)	Sghaier et al., 2011
	Tabarka	36° 57’N, 8°45’E	*Zostera noltii*				0.36		

El Kantaoui	35° 51’N, 10°35’E	*Posidonia oceanica*	0.14	Sghaier et al., 2013

Larger seagrass species such as *T. hemprichii* and *T. ciliatum* recorded the highest per unit area biomass while smaller species, such as *H. wrightii*, recorded the lowest biomass. There was a large range in biomass between the highest and lowest (Fig .6).

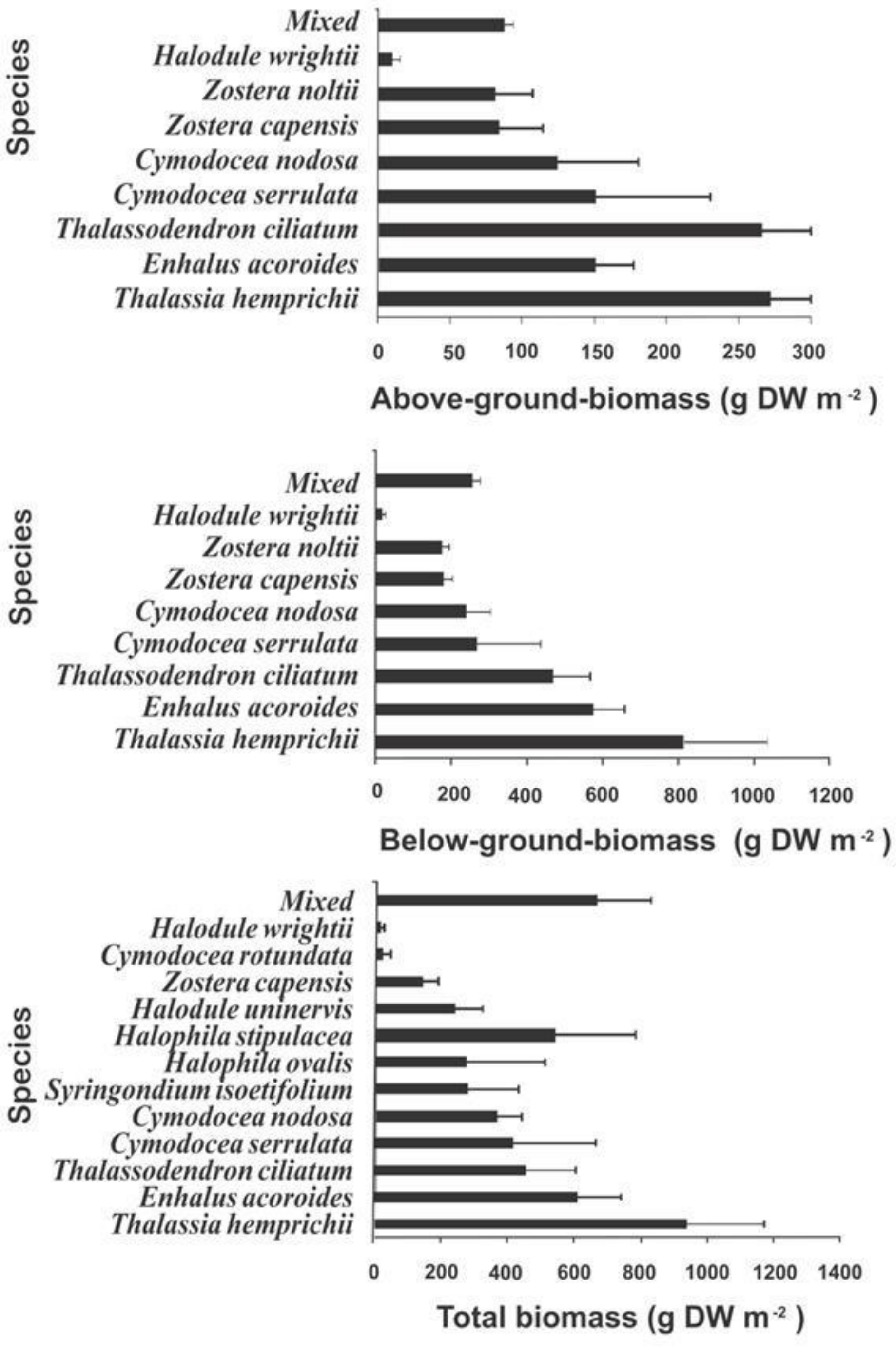

Fig. 6: Mean (±S.E) above-ground, below-ground and total biomass values for seagrass species studied in Africa, pooled across all reported sites

The highest number of published biomass and productivity studies in Africa were carried out between 1996 and 2010 accounting for 65.6% of the total, while 62.5% of theses, reports or articles (unpublished or currently under peer review) have emerged between 2010 and 2015 (Fig. 7).

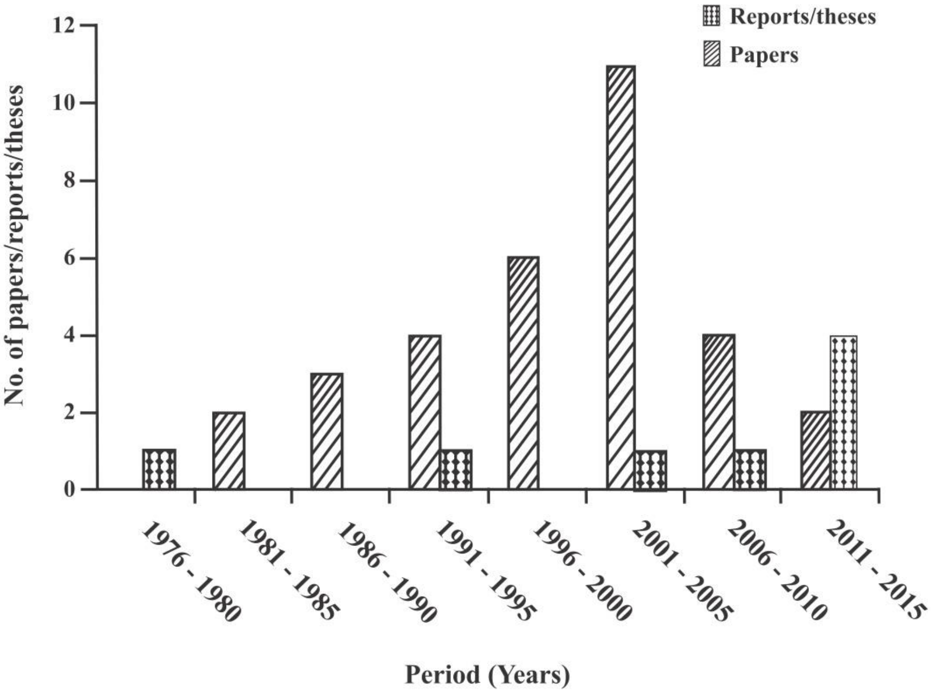

Fig. 7: Number of publications, reports/theses containing information on biomass and productivity of African seagrasses between 1976 and 2015

2.3.2 Biomass of seagrasses in Africa

The study obtained 47 data sets for both the above- and below-ground biomass and 73 for total biomass contained within the 32 papers and eight reports or theses (Table 2). The total and the above-ground biomass data were each reported in 21 of the 40 papers, reports and theses while below-ground biomass was reported in 15 of those papers, theses and reports. The total biomass for all species combined revealed large variation between sites (Table 3). The mean above- and below-ground biomasses for all species and across all sites were 174.4 and 474.6 g DW m^{-2}, respectively, representing an above to below-ground biomass ratio of almost 1:3. The mean total biomass was 514.3 g DW m^{-2}. This was calculated from the data available on total biomass and not necessarily from the sum of above-ground and below-ground biomass as some studies did not record either the above-ground or the below-ground biomass (Table 3). The highest total biomass was recorded for mixed seagrasses in a non-seaweed area at Jambiani in Zanzibar at 3063.3 g DW m^{-2} whilst the lowest total biomass of 0.6 g DW m^{-2} was recorded for *H. ovalis* at Northern Bay on Inhaca Island off Mozambique in the same study (Table 3). In terms of species, the highest biomass was recorded for *T. hemprichii* at 1876 g DW m^{-2} in Southern Bay of Inhaca Island, Mozambique (Table 3). Comparison of the means for the above-ground, below-ground and total biomasses for individual species reveal that the highest mean biomasses were found for *T. hemprichii* at 271.7 g DW m^{-2}, 817.8 g DW m^{-2} and 928.0 g DW m^{-2}, respectively, while the lowest mean biomasses were for *H. wrightii* at 11.5 g DW m^{-2}, 17.6 g DW m^{-2} and 19.2 g DW m^{-2}, respectively. In terms of the five regions where the seagrass data are available (Fig. 5), the East African coast has the highest mean above-ground, below-ground and total biomass at 256.8, 587.1 and 778.1 g DW m^{-2}, respectively. The South Mediterranean seagrasses had above-ground and below-ground biomasses of 155.6 and 299.3 g DW m^{-2}, respectively, while the South Africa and the WIO Islands had means of 95.7and 413.3 g DW m^{-2}, respectively, for the same parameters. Data available from the North West African region show the lowest mean biomass for the three parameters with 61.06 g DW m^{-2} for the above-ground biomass, 145.2 g DW m^{-2} for the below-ground biomass and 159.4 g DW m^{-2} for the total biomass (Fig. 8).

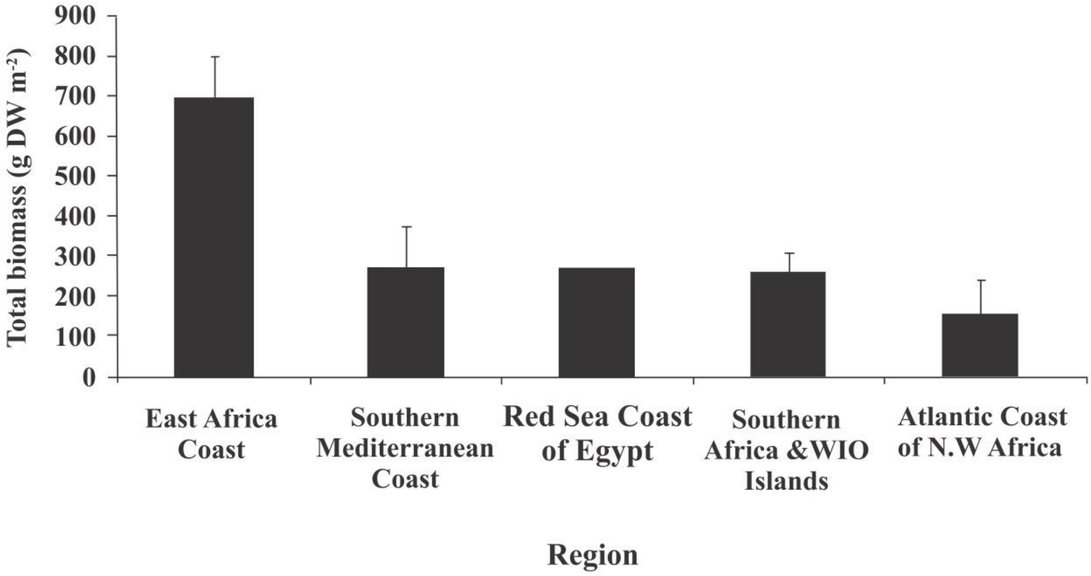

Fig. 8: Mean (±S.E) total biomass values for the seagrass species in different regions of Africa

2.3.3 Productivity rates of seagrasses in Africa

This review obtained 29 data sets on leaf growth rates, 24 on leaf production, seven on rhizome growth rates and 32 on total production (Table 4). The mean leaf growth rate was 12.4 mm shoot^{-1} day^{-1} while the mean leaf production was 0.07 g DW shoot^{-1} d^{-1}. Rhizome growth rates were 0.36 mm d^{-1} while the mean total production was 2.5 g DW shoot^{-1} d^{-1}. Lyimo et al., (2006) studied growth characteristics of *T. hemprichii* and *E. acoroides* at several sites in Zanzibar, where high growth rates in terms of leaf length and dry weight were observed for both species. In another study, Uku & Björk, (2005) recorded higher growth rates for the same parameters for *T. hemprichii* as compared to *C. rotundata* and *T. ciliatum* at Nyali and Vipingo, Mombasa, Kenya. In Gazi Bay, Kenya, Hemminga et al., (1995) reported total productivity for *T. ciliatum* that was much higher than reported from other sites (Table 4). In another study of a monospecific stand of *T. ciliatum* at Gazi Bay, Ochieng, (1995) recorded a mean shoot growth rate of 20.7 mm day^{-1} which was higher than the rate recorded in most of the other studies for the same species. The review for all species, whether growing in multispecific or pure stands, indicated that *Z. capensis* and *C. serrulata* had the lowest shoot growth rates of less than 1mm shoot^{-1} day^{-1} recorded at Inhaca Island, Mozambique (de Boer, 2000). Some seasonality is indicated for *Thalassia hemprichii* with a maximum of 28.5 mm shoot^{-1} day^{-1} during the North East monsoon and 17.2 mm shoot^{-1} day^{-1} during the South East monsoon at Nyali in Mombasa (Uku & Björk. 2005). Daily leaf production also differed between sites and species with a maximum of 0.01 g DW shoot^{-1} d^{-1} for *T. hemprichii* recorded at Chwaka in Zanzibar (Lyimo et al., 2006). Lowest daily leaf production was 0.001 g DW shoot^{-1} d^{-1} for *C. rotundata* recorded at Vipingo in Mombasa (Uku & Björk. 2005). The mean productivity rates for all species, where available, indicated that *T. hemprichii* had the highest total productivity rates while the lowest was in an eelgrass, *Z. capensis* (Table 5). The mean leaf production per day for individual species was highest in *C. serrulata* while the lowest was in *C. rotundata*. Comparison of rhizome growth rates indicated highest rates in *C. nodosa* and lowest in *P. oceanica.* The mean for total production was highest in mixed stands while the lowest was recorded in *H. ovalis* (Table 5).

Table 5: Mean (± S.E) productivity values expressed as rates of leaf growth, leaf dry weight production, rhizome growth and total dry weight production for seagrass species based on all available data around the African coast.

Species	Leaf growth (mm shoot^{-1} day^{-1})	Leaf production (g DW shoot^{-1} d^{-1})	Rhizome growth (mm d^{-1})	Total production (g DW m^{-2} d^{-1})
Cymodocea nodosa	3.35 ± 0		1.2 ± 0	0.71± 0.7
Cymodocea rotundata	12.35 ± 1.0	0.002 ± 0001		2.08 ± 0.1
Cymodocea serrulata	1.8 ± 0.6	0.63 ± 0.17		0.41 ± 0.2
Enhalus acoroides	24.8 ± 0	0.02 ± 0		2.77 ± 0
Halophila ovalis	1.5 ± 0	0.14 ± 0		0.2 ± 0
Posidonia oceanica			0.19 ± 0.1	
Thalassia hemprichii	17.33 ± 1.6	0.007 ± 0.01		3.26 ± 0.6
Thalassodendron ciliatum	15.18 ± 1.6	0.05 ± 0.01	0.4 ± 0	3.90 ± 0.7
Zostera capensis	0.8 ± 0.2	0.04 ± 001		0.47 ± 0.3
Zostera noltii				0.004 ± 0
Mixed				5.3 ± 2.9

2.4 Discussion and Conclusion

This assessment of studies on seagrass biomass stocks and productivity around Africa found a limited number of papers and reports with most of them reporting from countries on the Western Indian Ocean coastline (Kenya, Tanzania, Mozambique, South Africa, Madagascar, Seychelles and Mauritius). A few studies have also been reported from the Red Sea coastline of Egypt, the north eastern part of the Atlantic coastline on the coast of Mauritania and Morocco and more recently some studies (unpublished), have emerged from the Mediterranean coastline of Tunisia. However, the limited number of studies for the seagrasses despite the large seagrass communities demonstrates a paucity of information on the carbon budget and flows in Africa. Similar observations of a geographical bias in research on seagrass biomass stocks, with Africa particularly underrepresented, have been made in other reviews (Duarte and Chiscano 1999, Fourqurean et al., 2012). Some of the seagrass studies in Africa concentrated on one biomass pool (above-ground or below-ground) while others focused on total biomass only (Table 3). An important observation in this review is that seagrass studies in Africa have ignored the sediment organic carbon, the most important part of the putative 'blue carbon' sink provided by seagrasses, revealing a major gap in seagrass blue carbon work. Since the reviewed studies reported on only 14 out of a total of 34 species in the Tropical Atlantic, Tropical Indo-Pacific and South African flora, the current work suggests that the basic ecology, including productivity and standing stock, of many species remains largely unknown. The available data from the seagrass biomass and productivity studies in Africa reveal that seagrasses allocate higher biomass to their below-ground than their above-ground components, with mean estimates for the above and below-ground biomasses of 174.4 g DW m^{-2} and 474.6 g DW m^{-2}, respectively. In a review of seagrass biomass from different studies globally, Duarte and Chiscano (1999) arrived at above- and below-ground mean biomasses of

223.9 g DW m^{-2} and 237.4 g DW m^{-2}, respectively. These findings differ from the results of this study in which the above-ground biomass was only ~37 % of the biomass below-ground. Though these results deviate from our findings, our results are consistent with other observations, such as the most recent review of a global dataset, that the below-ground component of seagrasses forms the largest proportion of the living seagrass biomass and may constitute about two thirds of the total biomass in seagrass meadows (Fourqurean et al., 2012). The similarity of above-ground and below-ground biomass estimates in Duarte and Chiscano (1999) was attributed to the fact that some seagrass biomass studies did not measure the below-ground biomass, which in some cases could account for 15-50 % of the total production as observed in an earlier study (Duarte et al., 1998). Though grazing and mechanical damage inflicted by wave scouring and by human activities may not significantly affect seagrass productivity and biomass storage, it nevertheless impacts on the meadows leading to high turnover rates especially for the above-ground component.

The mean estimate for total seagrass biomass in this review of 514.3 g DW m^{-2} is within the global range. The seagrasses of Abu Dhabi in the United Arab Emirates were estimated to contain a total biomass of 122.3 g DW m^{-2} (Campbell et al., 2014). In a review of global seagrass carbon storage, the *P. oceanica* of the Mediterranean Sea were found to have the highest biomass at 2144 g DW m^{-2} while the mean biomass from the global seagrass data was estimated at 738.4 g DW m^{-2} (Fourqurean et al., 2012). While this global estimate is higher than our total African biomass estimate, this could be explained by the influence of the high biomass of *P. oceanica* in other regions as well as the limited information on seagrass biomass from Africa in previous global estimates. In terms of the five regions along the coasts of Africa where seagrass research has been done, this study observed that the East African seagrasses had the highest biomass at 738.1 g DW m^{-2} compared to 370.8 g DW m^{-2} for the southern

Mediterranean where *C. nodosa* was the dominant species. No study was found from this southern part of the Mediterranean Sea containing information for *P. oceanica*. The review observed that higher biomass values occurred in larger species compared to the smaller species (Fig 6). This may suggest that larger species tend to develop higher below-ground biomass and hence have a higher capacity for biomass storage due to the relatively slow turnover of the below-ground materials (Duarte and Chiscano 1999). The current assessment of available data from Africa on seagrass biomass supports this view.

The current review arrived at a mean total production estimate of 912.5 g DW m^{-2} yr^{-1} against 1012 g DW m^{-2} yr^{-1} obtained in a previous seagrass biomass and production reassessment using a global data set (Duarte and Chiscano 1999) and an earlier one of 816 g DW m^{-2} yr^{-1} (Duarte and Cebrián 1996). Seagrass beds with mixed species were found to have the highest total production, estimated at 1935 g DW m^{-2} yr^{-1}, followed by *T. ciliatum* at 1423 g DW m^{-2} yr^{-1}, suggesting that some species do better when in association with others. Growth patterns for different species and variation in environment between sites could account for the differences in values observed. Some species may have the potential to accumulate biomass but this may be kept low by resource limitation or due to the heavy losses caused by physical disturbance (Duarte and Chiscano 1999). Biomass and productivity for some seagrass species was reported to exhibit seasonality which could be attributed to periodical fluctuations in abiotic factors such as irradiance, temperature and hydrological conditions (de Boer, 2000; Uku & Björk, 2005). The estimates arrived at in this study may involve considerable errors, given the general paucity of studies, particularly for some seagrass species, and a lack of uniformity in the sampling methods used by different researchers. However, with the development of the Blue Carbon sampling manual by the International Blue Carbon Initiatives Scientific Working Group (Howard et al., 2014, http://theblue carboninitiative.org/), and new emphasis on researchers adopting uniform sampling protocols, future research should produce more reliable and comparable estimates. Whilst the research gap revealed here may be similar to many other areas in which Africa is under-represented, seagrasses perhaps present a particular challenge for research in countries with relatively poor infrastructure and resources, since they may require expensive sampling work, utilizing specialized skills such as scuba diving.

Considering that the African coastline is extensive with large areas of seagrass cover, knowledge on the spatial extent of seagrasses is scarce as mapping of seagrasses remain limited. The fact that this review did not find seagrass biomass studies from the West African coast, with the exception of Mauritania which is more to the North West coast despite the presence of seagrass meadows in the region, is another clear indication of the paucity of knowledge on seagrass biomass stocks in Africa. A majority of the studies have been done on the West Indian Ocean coastline mainly through funding by the West Indian Ocean Marine Sciences Association (WIOMSA) in partnership with the well-established research Institutions in the region or through partnership with institutions outside Africa. This signifies the importance of strengthening collaboration between institutions and the need for increased research funding if the knowledge gaps are to be filled. As the first review of seagrass biomass and productivity in Africa, it is hoped the current work will generate interest among the scientific community by identifying an important and missed opportunity for research. By contributing to a better understanding of the role of seagrass ecosystems in carbon budgets in Africa this may help support the protection of these valuable ecosystems.

Chapter three

3.0 Carbon storage in the seagrass meadows of Gazi Bay, Kenya

Abstract

Vegetated marine habitats are globally important carbon sinks, making a significant contribution towards mitigating climate change, and they provide a wide range of other ecosystem services. However, large gaps in knowledge remain, particularly for seagrass meadows in Africa. The present study estimated biomass and sediment organic carbon (C_{org}) stocks of four dominant seagrass species in Gazi Bay, Kenya. It compared sediment C_{org} between seagrass areas in vegetated and un-vegetated 'controls', using the naturally patchy occurrence of seagrass at this site to test the impacts of seagrass growth on sediment $C_{org.}$ It also explored relationships between the sediment and above-ground $C_{org,}$ as well as between the total biomass and above-ground parameters. Sediment C_{org} was significantly different between species, range: 160.7 – 233.8 Mg C ha^{-1} (compared to the global range of 115.3 to 829.2 Mg C ha^{-1}). Vegetated areas in all species had significantly higher sediment C_{org} compared with un-vegetated controls; the presence of seagrass increased C_{org} by 4 - 6 times. Biomass carbon differed significantly between species with means ranging between 4.8 – 7.1 Mg C ha^{-1} compared to the global range of 2.5 – 7.3 Mg C ha^{-1}. These are among the first results on seagrass sediment C_{org} to be reported from African seagrass beds; and contribute towards our understanding of the role of seagrass in global carbon dynamics.

Key words: Allochthonous, autochthonous, carbon storage, Gazi Bay, seagrass, sediment

3.1 Introduction

Carbon sinks in terrestrial ecosystems are better studied than those in marine plant communities. However, the global importance of vegetated coastal habitats as carbon sinks has become appreciated over the last decade (Nellemann et al., 2009). These 'blue carbon' ecosystems (tidal marshes, mangroves and seagrass meadows) have recently been demonstrated to capture and store huge stocks of carbon, and their management and conservation may play an important part in global climate change mitigation strategies (Fourqurean et al., 2012; Kennedy et al., 2010; Lavery et al., 2013; Mcleod et al., 2011). Although they occupy less than 2% of the world's ocean surface area, Duarte & Cebrián (1996), blue carbon ecosystems are estimated to bury nearly 27.4 Tg C yr^{-1} which is about 10% of the yearly estimated organic carbon (C_{org}) burial in the oceans (Duarte et al., 2005). Unlike many terrestrial systems that store C_{org} primarily in living biomass, vegetated coastal ecosystems store much of their C_{org} stock in the sediment, which may produce carbon sinks of hundreds to thousands of years age (Duarte et al., 2005). However, this stored C_{org} risks being released back to the atmosphere when blue carbon ecosystems are degraded (Duarte et al., 2005).

Seagrass meadows are the most extensive of the blue carbon ecosystems, with an estimated global surface area of between 300,000 to 600,000 km^2 (Duarte et al., 2005; Nellemann et al., 2009). Despite this wide spatial coverage, seagrasses are the least well-studied blue carbon ecosystem. They provide important ecosystem services that include: support for commercial fisheries, worth $ 3500 ha^{-1} yr^{-1} (Watson et al., 1993), and subsistence fisheries (De La Torre-Castro & Ronnback, 2004), sediment stabilization (Orth et al., 2006), improved water quality and light availability (McGlathery et al., 2007; Romero et al., 1994) and nutrient cycling (estimated to be worth $ 3.8 trillion yr^{-1} globally (Costanza et al., 1997). In addition, seagrasses are recognized as one of the most efficient carbon sinks in the planet (Fourqurean et al., 2012). Seagrass meadows store about twice as much C_{org} per unit area as soils in productive temperate and upland tropical forests, and provide a global carbon sink of approximately 19.9 Pg (Fourqurean et al., 2012; Macreadie et al., 2014). This is approximately equal to the combined amount of C_{org} stored in the world's tidal marshes and mangrove forests which is estimated at 10 Pg (Chmura et al., 2003). An estimated 50% of the C_{org} buried in seagrass meadows' sediment is thought to be of external origin (Kennedy et al., 2010). The dense canopies of seagrasses reduce water flow velocity, thus promoting the trapping and deposition of sediment and particles from the water column (Gacia, 2001; Marba' et al., 2002). Analysis of a global data set revealed that the mean seagrass biomass C_{org} was 2.52 ± 0.48 Mg C_{org} ha^{-1} (±95% CI), while sediment C_{org} was estimated to account for between 0.002 - 48% of the sediment dry weight (Fourqurean et al., 2012). However, these findings may be inaccurate considering the uneven distribution of research into seagrass carbon budgets globally and the large variation recorded between different sites. Much of the information on seagrasses, especially on sediment $C_{org,}$ is from Mediterranean and Australian seagrass beds (Duarte et al., 2005; Lavery et al., 2013; Nellemann et al., 2009). Past global reviews of seagrass carbon acknowledged the disproportionately low contribution of data from the African continent (Duarte & Chiscano 1999; Fourqurean et al., 2012). A review of seagrass biomass and productivity in Africa in this study found no published estimates of seagrass sediment carbon. Seagrass beds are suffering rapid global decline; almost a third of all seagrass areas are thought to have been lost in the last 140 years (Waycott et al., 2009), and current rates of loss are estimated at around 1.5% $year^{-1}$ (Pendleton et al., 2012). This implies that a significant amount of the stored C_{org} could soon be remineralized and that the potential for future carbon capture is being diminished, undermining efforts to mitigate climate change

(Fourqurean et al., 2012). However, the amount of carbon that could be remineralized at a global scale remains highly uncertain. This is because: first, the vulnerability of this stored carbon to ecosystem change and degradation is little studied (with one recent paper showing surprising persistence of buried carbon following seagrass removal (Macreadie et al., 2014). Secondly, the huge gaps in knowledge on the extent and quantity of the global seagrass carbon sink make global estimates very imprecise. Improving our knowledge of carbon storage and burial rates in seagrass ecosystems, of how these variables differ between sites and of the controls on burial rates and sink sizes is fundamental in achieving a better understanding of how seagrass meadows may contribute to slowing global warming.

Whilst previous seagrass studies in Gazi Bay have focused on species distribution, community composition, growth dynamics, nutrient content and carbon export between the seagrasses and the adjacent ecosystems (Bouillon et al., 2007; Coppejans et al., 1992; Duarte, 1996; Hemminga et al., 1994), no study here (or elsewhere in Africa) has considered the carbon stocks and how these might compare with naturally occurring un-vegetated areas. The current work aimed to fill this gap by estimating the carbon stocks in the seagrass meadows of Gazi Bay, Kenya. At the same time, the work explored whether different seagrass C_{org} pools could be estimated using easy to measure parameters such as the above-ground biomass (AGB) and shoot height. Specifically, the objectives of the study were:

3.1.1 General objective

Determine carbon storage in the seagrass meadows of Gazi Bay

3.1.2 Specific objectives

1) To compare sediment C_{org} between areas with seagrass and un-vegetated control areas for each of the four dominant seagrass species in order to test the impact of these species on sediment carbon storage.
2) To determine the % C_{org} associated with each of the four dominant seagrass species at Gazi Bay and the relative contribution of biomass and sediment to the C_{org} per unit area of the seagrass species.
3) To explore the relationship between sediment C_{org} and the above ground carbon (AGC_{org}).
4) To determine average below and above-ground biomasses of the four dominant seagrass species in Gazi Bay, and compute biomass ratios between these components for each species.
5) To explore relationships between the total biomass and shoot density as well as shoot height for each of the four dominant seagrass species

3.2 Materials and methods

3.2.1 Study site

This study was carried out at Gazi Bay (4°25'S, and 39°30'E), located on the southern coast of Kenya, about 55 km from Mombasa City through permission issued by the National Commission for Science, Technology & Innovation (NACOSTI): Permit no NACOSTI/P/14/2443/769 on 17th February 2014. The bay is a shallow tropical coastal water system (mean depth < 5m), approximately 1.75- 3.5 km wide and 3.25 km long with a surface area of ~ 17 km^2 (Bouillon et al., 2007; Hemminga et al., 1995). It is open to the Indian Ocean through an entrance in the south with depths varying between 3 and 8m in the eastern and western regions respectively (Kitheka, 1996). It is characterized by two creeks, a western creek that extends to a fresh water inflow (R. Kidogoweni) in the north western side of the bay and an eastern creek that lacks such an inflow. On the south western side of the bay is another fresh water inflow from R. Mkurumuji (Fig 9).

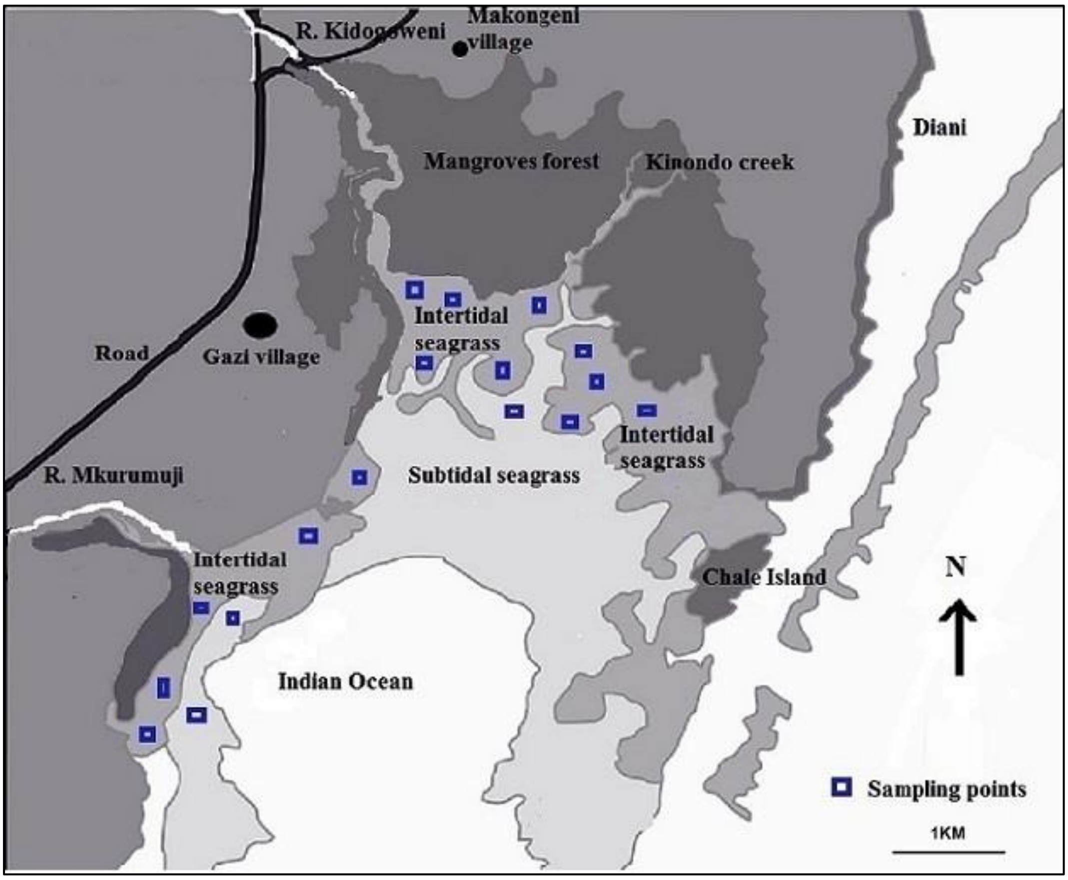

Fig. 9: Distribution of some of the seagrass sampling points in the seagrass meadows of Gazi Bay, Kenya

The main climatic drivers are the southeast monsoons (May-September) and the northeast monsoons (November-March). An offshore wind prevails throughout the year. Long rains occur between March and May while the short rains occur between October and November (Kitheka, 1996; Schott et al., 2009). However, inter-annual shifts in these seasons are common (Haas, 1977). The tides at Gazi Bay are normally mixed semi-diurnal, with a tidal range of 1.4m and 3m at neaps and springs respectively. These tides generate strong reversing currents in the tidal creeks and relatively weaker currents in the open regions of the bay. The shoreward wind and the tidal currents combine to vertically mix the water column in the bay, leading to formation of a more homogeneous water, with a salinity range of 34.5-35.5ppt. A significant lateral and vertical salinity gradient develops during the rainy periods as a result of increased river discharge and the runoff (Kitheka, 1996). The flushing ability of mixed semi-diurnal tides in the bay varies depending on tidal range, tidal elevation, and the nature of the tide. Tides are generally swift during springs and tend to rapidly disperse lower salinity water (Kitheka, 1996; Prrez-llorrns, 1993). The rates of water exchange are also high during spring tides compared to neap tides. On the other hand, currents at neaps are sluggish and inhibit flushing of brackish water. This flushing pattern of the tides combined with river runoff has a far-reaching effect in the form of nutrient and material exchanges on the linkage between mangrove, seagrass, and coral reef ecosystems (Dahdouh & Speybroeck, 1999; Ho & Barrett, 1976; Prrez-llorrns, 1993). Seagrasses, which are found at the centre of the bay, cover an area of ~ 7 km^2 (Hemminga et al., 1995; Pendleton et al., 2012; Schott et al., 2009).

All the twelve seagrass species described along the East African coast have been recorded in the bay, usually attached to both soft and hard substrates (Bandeira, 2002; Erftemeier, 1993; Isaac & Isaac, 1968). The seagrass community in the bay consists of four dominant species: *Thalassodendron ciliatum* (Forssk.) den Hartog, *Thalassia hemprichii* (Enhrenberg) Aschers., *Enhalus acoroides* (L.f.) Royle and *Syringodium isoetifolium* (Aschers.) Dandy. These are observed to grow either as monospecific stands or mixed with other seagrass species, with their coverage extending from the intertidal to the subtidal areas in sandy and rocky substrates (Bouillon et al., 2007). The other, less abundant species, are: *Cymodocea rotundata* Ascherson, *Cymodocea serrulata* (R. Braun) Aschers. & Magnus, *Halodule uninervis* (Forssk.) Aschers., *Halodule wrightii* (Aschers.), *Halophila minor* (Zoll.) den Hartog, *Halophila ovalis* (Braun) Hooker, *Halophila stipulaceae* (Forssk.) Aschers. and *Zostera capensis* (Setch) (Bouillon et al., 2007; Isaac & Isaac, 1968). The meadows are usually luxuriant for most of the year except during short periods of intense grazing and desiccation.

The seagrass meadows of Gazi Bay fall under Diani-Chale Marine National Reserve which was established in 1994. It also forms the northern boundary of the proposed Transboundary Marine Conservation Area between Kenya and Tanzania (TBCA) (MPRUT-KWS, 2016). At present there is no enforcement of its protection status due to weak governance. The use of seine and drag nets by artisanal fishermen is a daily activity in the shallow waters of the bay. Although there are no published data on the effects of this fishing, anecdotal information and personal observation suggests this is a cause of degradation of the seagrass in areas of intense fishing.

3.2.3 Sampling design

This study used sampling procedures specific for coastal blue carbon and the revised Intergovernmental Panel for Climate Change (IPCC) carbon accounting protocols for coastal wetlands (Howard et al., 2014; IPCC, 2013). Intensive studies were made within mono-specific stands of four seagrass species: *T. hemprichii, E. acoroides, T. ciliatum* and *S. isoetifolium*. These species were selected on the basis of their local dominance, as determined by the initial reconnaissance survey carried out by the team. Samples were taken from a wide area following a stratified random design, which extended from plots adjacent to the mangrove forest, all the way to the seagrasses near the mouth of the R. Mkurumuji (Fig 9), in order to sample the spatial variability of biomass and sediment C_{org} .associated with the four target species in the bay. Forty quadrats, each measuring 0.25 m^2, were sampled for the biomass carbon for each of the three species, *T. hemprichii, E. acoroides* and *T. ciliatum* whilst 20 quadrats were sampled for *S. isoetifolium,* owing to its relatively small spatial distribution.

3.2.4 Determination of species type, canopy cover, shoot density and canopy height

Sample plots were established within the identified seagrass strata placing the quadrats of $0.25m^2$ at minimum intervals of 15m in each seagrass strata. This was done during low spring tides since at this time the seagrass beds are exposed and accessible on foot. Species type was determined *in situ* with the help of field manuals (Richmond, 1997). Shoot density and canopy cover were determined by counting all the shoots within quadrats and extrapolating to per m^2 while the % canopy cover was obtained through visual estimates. Canopy height was determined through the measurement of heights of 10% of randomly selected individual shoots from the total within the quadrat and calculating the mean heights.

3.2.5 Estimation of above-ground and below-ground biomass

The above-ground biomass (AGB) was obtained by harvesting all plant materials above-ground within the $0.25m^2$ quadrats. In the laboratory, the seagrass was cleaned with fresh water, sorted and scraped with a razor blade to remove epiphytes. Seagrass fronds were then washed in 10% hydrochloric acid to remove any calcareous material after which they were dried in an oven at 60°C for 72 hours. For below-ground biomass, four cores were taken in each of the four quarters of the $0.25m^2$ quadrats with a Russian peat sampler (50cm long and 5cm diameter). Initial washing was done in the field using a 500µm sieve. In the laboratory further washing and rinsing of the samples was carried out. Upon sieving, the materials were sorted into component parts: roots, rhizomes and necromass (that is dead roots or rhizomes) and dried in an oven at 60°C for 72 hours. The combined below-ground biomass values for the roots, rhizomes and necromass from the four cores per quadrat were summed and then converted to per m^{-2}. The total biomass carbon was obtained by multiplying biomass with a carbon conversion factor of 0.34 (assuming that carbon constitutes 34% of the biomass) and then extrapolated to per hectare following recommended protocols for estimating carbon for marine ecosystems (Howard et al., 2014; IPCC, 2013).

3.2.6 Measurements of bulk density, % organic matter and Sediment C_{org}

Two sediment cores, each extending to a depth of 50cm, were collected in each quadrat, using the peat sampler from the vegetated and the un-vegetated areas, chosen to act as natural ‘controls’ for each species. Vegetated areas were identified as those with seagrass cover while un-vegetated areas were the naturally occurring bare patches, measuring 3 to 6m in diameter, in the midst of the seagrass covered areas.

In the laboratory, the samples were sliced into 5cm sub-sections and were oven dried at 60°C for 72 hours to obtain a constant weight. Similar cores were collected in un-vegetated seagrass areas to serve as ‘controls’.

Dry bulk density (DBD) (the dry weight of sediment per unit volume) was calculated for each of the ten sub-sections per core as follows:

DBD (g/cm^3) = Dry weight/Original volume of the sediment

Organic matter was measured in each of the ten sub-sections per core by Loss on Ignition (LOI) techniques, using a muffle furnace at 450°C for 6 hours. LOI weight was used to calculate the % OM content as follows:

% LOI = ((Initial dry weight – Weight remaining after ignition) /Initial dry weight) × 100. The sediment C_{org} values were arrived at using one of the two equations depending on the organic matter content of the sediment sample:

% LOI < 0.2: % C_{org} = 0.40*% LOI-0.21

% LOI > 0.2: % C_{org} = 0.43*% LOI - 0.33, following recommended protocols for estimating carbon for the marine ecosystems (Fourqurean et al., 2012: Howard et al., 2014). Estimates for the top 50 cm of the sediment were extrapolated to one metre.

3.2.7 Data analysis

Assumptions of normality and homogeneity of variance were tested by examining residuals in all relevant tests; where these were not met, data were transformed to meet parametric assumptions. One way ANOVAs were used to test for the differences in above-ground, below-ground and total biomass between species, with Tukey *post - hoc* analyses used to compare means when significant differences were detected. Linear regression analysis was used to determine possible relationships between above-and below-ground biomass and also between sediment C_{org} and the above-ground biomass for the species areas. Nested two-way ANOVAs were used to compare sediment C_{org} between each species and its un-vegetated control cores. In all statistical tests, the significance level was set at $\alpha = 0.05$..

3.3 Results

3.3.1 Sediment C_{org} in seagrass areas and un-vegetated 'controls'

There was higher sediment C_{org} in the vegetated seagrass areas compared to the un-vegetated areas (Fig. 10). Strikingly these differences persisted down to 50cm depth in all species; initial analyses using depth and 'treatment' (i.e. vegetated vs un-vegetated) as fixed factors in two way ANOVAs showed significant treatment effects but no depth or depth-treatment interaction effects. Hence, a two way fully nested ANOVAs, in which depth was nested within cores and treatment (vegetated/un-vegetated) to recognise non-independence of depth slices from the same cores, were subsequently used. These revealed highly significant effects of treatment on C_{org} density for each of the species: $F_{(1, 180)} = 38.68$, $p < 0.001$ for *T. hemprichii*; $F_{(1, 180)} = 27.89$, $p < 0.001$ for *T. ciliatum*; $F_{(1, 180)} = 32.16$; $p < 0.001$ for *E. acoroides* and $F_{(1, 180)} = 11.55$, $p = 0.003$ for *S. isoetifolium.*

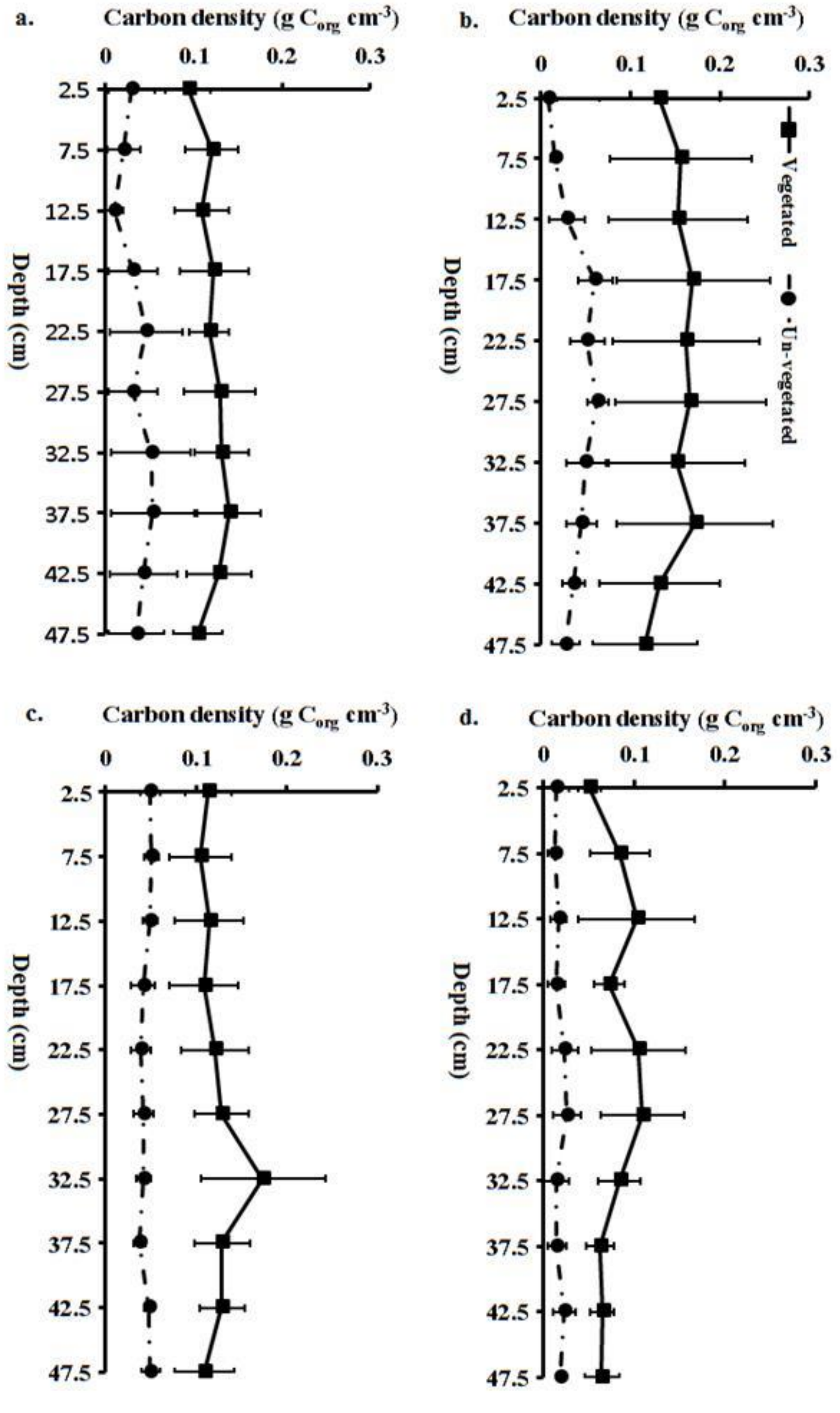

Fig. 10: Carbon density (mean ± 95% C.I.) along depth profiles in the vegetated and un-vegetated areas associated with the dominant seagrass species of Gazi Bay. a. *T. hemprichii* b. *E. acoroides* c. *T. ciliatum* d. *S. isoetifolium*

The sediment C_{org} varied between meadows of different species with the highest being recorded in *E. acoroides* at 295.7 ± 63.6 (mean ± 95% C.I) Mg C ha^{-1} and the lowest in *S. isoetifolium* at 160.7 ± 40.3 Mg C ha^{-1} (Fig. 11). A one way ANOVA showed significant differences in C_{org} among the species ($F_{(3, 56)} = 6.24$, $p = 0.001$).

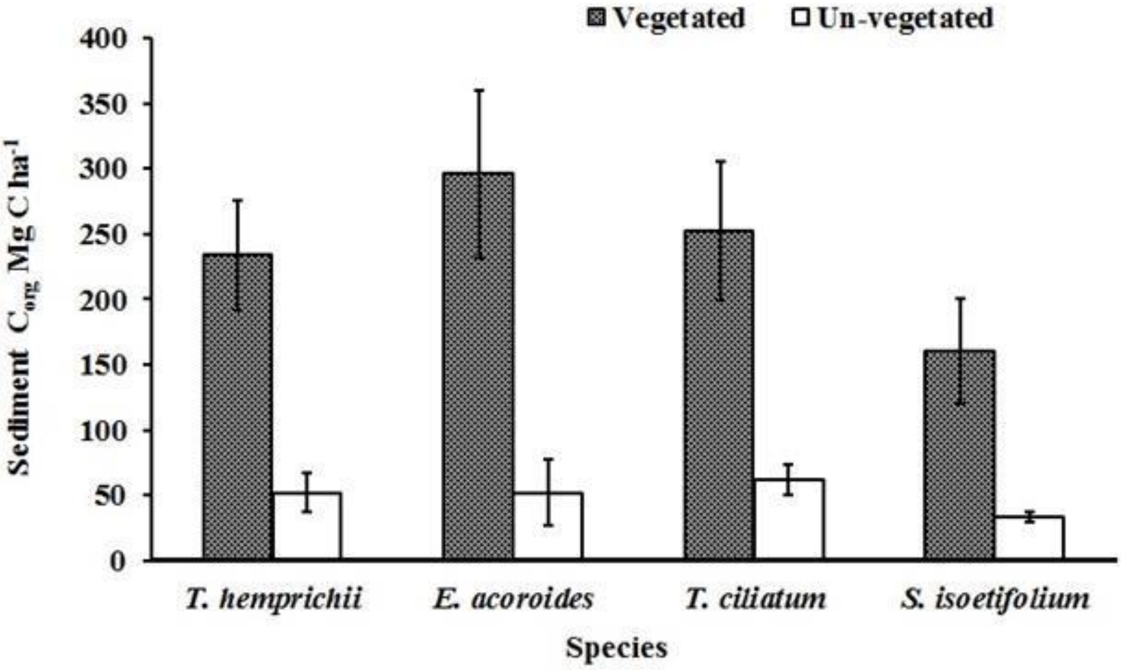

Fig. 11: Variation in sediment Corg between the vegetated and un-vegetated areas for the four seagrass species (means± 95% C.I.)

3.3.2 Comparison of the sediment C_{org} and the above-ground biomass C_{org}

The sediment C_{org} constituted the bulk of the total C_{org}, in all four species areas (Fig. 12). The sediment C_{org} of 295.7 ± 63.6 Mg C ha^{-1} (mean ± 95% C.I) for *T .ciliatum* accounted for the highest proportion at 98.1% of the total C_{org} per unit area dominated by the species while 95.8% was the lowest proportion recorded in *S. isoetifolium* areas. Plotting the sediment C_{org} against the above-ground biomass C_{org} for each of the four seagrass species showed no significant relationships.

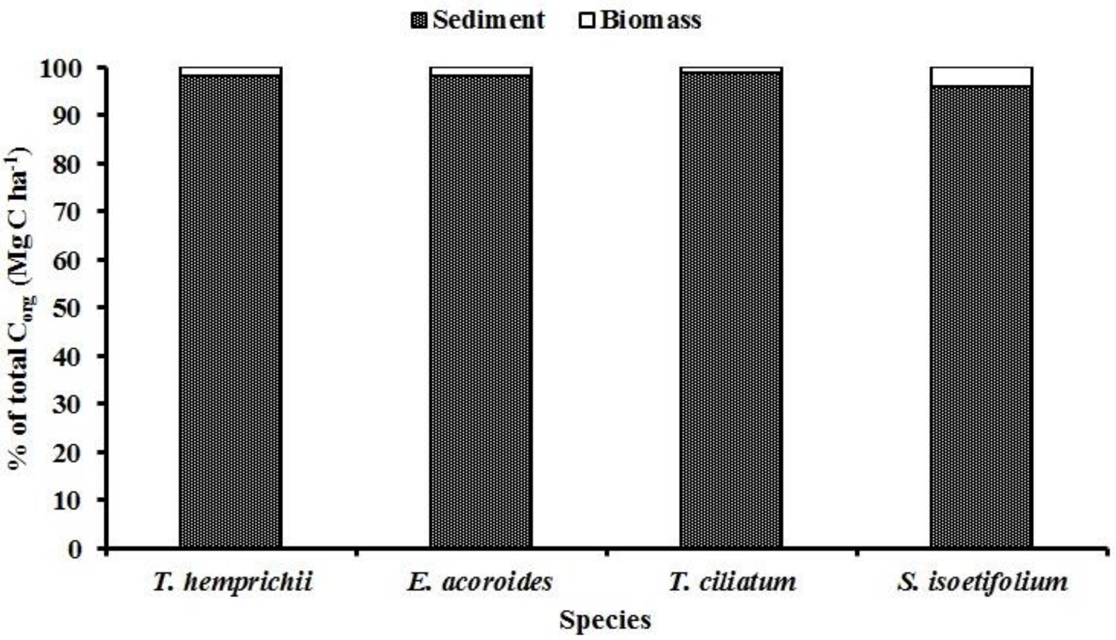

Fig. 12: Relative % of the total Corg (± 95% C.I) for the sediment and the biomass associated with the four dominant seagrass species at Gazi Bay.

3.3.3 Relationships between above-and below-ground biomass

The highest mean AGB was associated with *T. ciliatum* while the lowest was associated with *E. acoroides* (Table 6). The highest mean BGB was recorded in the *S. isoetifolium* while the lowest was recorded in the *T. ciliatum*. Comparison of AGB and BGB relationships in the four species revealed a highly significant relationship in *E. acoroides*, ($F_{(1, 38)} = 25.02$, $p < 0.001$) but there were no significant relationships between the BG and the AGB in the other species. BGB constituted the highest biomass component with an average of 82.2 ± 8% (or ~ 4:1 BG:AG biomass ratio) for all the species, with *E. acoroides* having the highest at 90.9 ± 1% and *T. ciliatum* having the lowest at 71.0 ± 5%.

3.3.4 Relationships between the total biomass and above-ground parameters

Total biomass varied between the species with the highest being recorded in *S. isoetifolium* at 1985 ± 246 g DW m^{-2} while *T. ciliatum* had the lowest at 1405 ± 233 g

DW m^{-2}. The total biomass was significantly different between the species ($F_{(3, 136)} = 4.13$, $p = 0.008$). No significant relationships were found between the total biomass and shoot density for any of the species and neither between the total biomass and the shoot height, except in *T. ciliatum* ($F_{(1,38)} = 9.83$, $p = 0.004$). The positive relationship between shoot density and % canopy cover was apparent across species (Table 6).

Table 6: Mean (± 95% C.I) shoot density, canopy cover (%), canopy height (cm), Above-Ground Biomass (AGB), Below-Ground Biomass (BGB) and Total Biomass (TB) characteristics of the dominant seagrass species at Gazi Bay, Kenya.

Species	Shoot density(m^{-2})	% C. cover	C. ht (cm)	AGB (g. DW $m^{-2)}$	BGB g. DW m^{-2})	TB (g. DW m^{-2})	% BGB
T. hemprichii	996 ± 94	69.3 ± 4.2	18.4 ± 1.4	202.1 ± 29.9	1361.1 ± 281.8	1563.1± 279.2	82.8±3
E. acoroides	248 ± 28	47.5 ± 4.2	55.1 ±4 1	155.9 ± 23.7	1669.2 ± 217.6	1825.8 ± 234.8	90.9±1
T. ciliatum	531 ± 67	61.9 ± 3.7	36.7 ±3.9	308.2 ± 33.5	1096.6 ± 221.4	1404.9 ± 232.5	71.0±5
S. isoetifolium	4351 ± 500	72.0 ± 7.0	23.3 ± 2.7	300.8 ± 42.6	1683.8 ± 242.9	1984.7 ± 245.5	84.0±4

3.4 Discussion

The current study investigated biomass and sediment C_{org} stocks associated with the four dominant seagrass species of Gazi Bay, providing amongst the first data from Africa for seagrass sediment C_{org} (Dahl et al., 2016). It also compared sediment C_{org} in contiguous vegetated and un-vegetated areas for each of the species and demonstrated highly significant differences in carbon density, with the presence of seagrass enhancing sediment carbon stocks by a factor of 4 to 6, depending on the species. The mean sediment C_{org} estimated for the top one metre of the sediment (using IPCC protocols to extrapolate downwards from the 50cm measured depth) from seagrass vegetated areas was 236 ± 24 Mg C ha^{-1}. This is well above the mean of 166 Mg C ha^{-1} derived from a global data set (Fourqurean et al., 2012), (although within the range of 115.5 – 829.2 Mg C ha^{-1}). Across all species, sediment depth did not have a significant effect on sediment C_{org} and the differences observed between vegetated and un-vegetated areas were consistent down to 50cm. This suggests a surprising degree of spatial consistency and longevity in these relatively small (typically 3 to 6m diameter) patches of seagrass meadow and bare areas. Sediment C_{org} dominated the carbon stocks in all species areas, constituting over 97% of the total C_{org} compared to less than 3% contributed by the biomass. Much of this sediment carbon is likely to be allochthonous; globally an average of ~50% of sediment C_{org} associated with seagrass meadows is derived from external sources (Kennedy et al., 2010). In Gazi Bay much of this comes from mangroves; previous studies in Gazi Bay showed that mangrove carbon is exported to the seagrass beds where it is stripped from the water column and settles (Bouillon et al., 2004; Hemminga et al., 1994), with an estimated 21-71% of carbon exported from the fringing mangrove forests captured within the seagrass meadows (Bouillon et al., 2007).

Whilst there is a growing literature exploring sediment carbon in seagrass meadows there are few explicit comparisons of vegetated with un-vegetated areas. A limited number of studies have compared the sediment carbon stocks or both the stocks and burial rates in bare areas and naturally or artificially recolonized meadows (Bouillon et al., 2004; Dahl et al., 2016; Irving et al., 2011; Macreadie et al., 2014; Marba' et al., 2015; Serrano et al., 2016; Serrano etal., 2016). For instance, a study on carbon accumulation in a restored seagrass meadow in Virginia, USA, in which sediment cores were taken to 10-20cm depth, Serrano et al., (2016) reported that after 9 years the meadow had 3 times more carbon than the un-vegetated areas, suggesting powerful effects of seagrass on C sequestration at this site and a rapid recovery following restoration. A synthesis of global data (Mcleod et al., 2011) gave a mean sediment C burial rate of 138 g C m^{-2} yr^{-1} in seagrass beds. A recent study in Oyster Harbour of Western Australia (Marba' et al., 2015) reported that a restored seagrass meadow took 18 years to acquire a carbon accumulation rate of 26.4 ± 0.8 g C m^{-2} yr^{-1} and that a naturally vegetated area which was used as a reference had 63% and 37% more carbon than the un-vegetated and restored areas respectively. To our knowledge, the present study is the first one to compare sediment C_{org} for seagrass in naturally occurring vegetated and un-vegetated areas in Africa and it demonstrates an exceptionally powerful effect of seagrass on C sequestration, with relative C densities being higher in areas under seagrass compared with appropriate un-vegetated control areas and the differences being higher than in any other relevant studies.

A recent study using a global data set (Greiner et al., 2013) reported an average sediment accretion rate of 2mm yr^{-1} for seagrasses. If the sediment at Gazi accumulated at the same rate this suggests a minimum age of 250 years for sediment at 50cm, demonstrating that the current locations of seagrass meadow and bare patches are likely to have persisted for at least decades. However, a robust estimation method requires in-situ dating studies.

The present study did not establish any significant relationships between sediment and biomass C_{org} in any of the species. Lack of strong relationships between either the sediment C_{org} and the AGB or between the AGB and BGB measures means that AGB was not a suitable proxy for the determination of BGB or sediment C_{org} for any of these species and coring is therefore necessary for accurate estimates of these variables.

The biomass estimates obtained in the present study were within the range of published data for the same species in other parts of the world although tended towards the high end of these ranges (Duarte & Chiscano 1999; Duarte et al., 2013; Erftemeier, 1993). The mean for the total biomass C_{org} for the four seagrass species was 5.9 ± 0.9 Mg C ha^{-1}. As was the case for sediment $C_{org,}$ this value is well above the global mean of 2.51 ± 0.49 Mg C ha^{-1} (although *Posidonia oceanica* in the Mediterranean has greater biomass than our species, with a mean of 7.29 ± 1.52 Mg C ha^{-1}) (Fourqurean et al., 2012). Across all species, the BGB was much higher than the AGB, accounting for over 80% of the total. The fact that AGB in all species was substantially lower and slightly more variable than the BGB could be attributed to higher turnover rates for the AGB occasioned by grazing pressure, mechanical removal by tides and human activities such as seine fishing, events that were observed in the bay in the course of this study. Whilst larger species, such as *E. acoroides,* with large fronds and big roots and rhizomes, are likely to accumulate more biomass during growth, which is invested in their below-ground tissues, shoot density is also an important parameter in determining per unit area BGB. *Syringodium isoetifolium*, which had a substantially higher shoot density and % canopy cover than the other species, recorded higher BGB than even the larger species.

Seagrasses cover an estimated area of 7km^2 within Gazi Bay (Bouillon et al., 2007); this is approximately 41% of the bay area considering the entire bay area of 17km^2). Extrapolation of the combined mean biomass and the sediment C_{org} values in the top one metre of the sediment gives an estimated total of 168,642 (range: 117,427 – 211,778) Mg C for seagrass meadow carbon stocks in the bay as a whole. If these findings are representative of seagrass meadows along much of the African coast the current absence of African sites from the global data will result in an underestimate of average carbon storage in seagrass meadows.

There are many sources of uncertainty in this estimate, however; for example the study considered only the four dominant species, which constitute ~ 70% of the seagrasses of the bay, and other species areas may show different C densities. The coring was limited to the top 50cm. Future research at Gazi should not only aim to quantify the C_{org} from all seagrass species but should investigate the C_{org} in deeper sediment and undertake a thorough mapping and estimation of the sedimentary C_{org} of the seagrasses of the entire bay for a better understanding of the carbon storage capacity of seagrasses there.

The present study has established that, as for other species and sites, the sediment C_{org} constitutes by far the major C pool for the seagrass beds of Gazi Bay. The highly significant differences in C_{org} between vegetated and un-vegetated areas underlines the importance of seagrass meadows as shallow marine C sinks, a service that adds to the many other justifications for their conservation. This study provides among the first estimates of sediment C_{org} from seagrasses in the Africa. As such, it contributes to the growing global literature on the importance of seagrass meadows as C sinks. It also provides information of potential relevance to the conservation and management of seagrasses in the area. Gazi Bay hosts a pioneer carbon offset project “Mikoko Pamoja”, the first initiative in the world to restore and protect mangroves through the sale of carbon credits (http://www.planvivo.org/project-network/mikoko-pamoja-kenya). Knowledge of the carbon stocks associated with seagrasses in the bay may open opportunities for bundling seagrass ecosystem services with those of the mangrove ecosystem, an approach that makes ecological sense, given the strong connections between the two ecosystems.

Chapter four

4.0 Experimental test of the impact of seagrass loss on sediment dynamics and on the benthic faunal communities

Abstract

The integrity of vegetated marine ecosystems underpins the provision of ecosystem goods and services. However, habitat degradation could compromise the provision of the services. This study used a removal experiment to test the impacts of seagrass loss on biodiversity and ecosystem functions at Gazi Bay. Sediment elevation tables (SETs) and sediment traps were used to monitor surface elevation and accretion respectively. The effect of seagrass removal on water velocity was investigated indirectly using Plaster of Paris clod cards while sediment carbon concentration was determined at the end of the experiment. Temporal trends in litter decay at varying depths between the treatments and controls were monitored. The mean (± 95% C.I) sediment elevation and erosion in the control and treatment areas was 7.6± 0.4 and -15.8 ± 0.5 mm yr^{-1}.respectively. Repeated measures ANOVA using treatment and time factors revealed significant interaction effect of treatment and time on sediment elevation change $F_{(17,102)}=3.59$. $p < 0.01$. Higher loss in weight of clod cards was recorded in seagrass-removed plots as compared to the controls. There was significant effect of treatment on litter decay rates. Reduced numbers of benthic faunal communities showed dramatic changes in removal as compared to the vegetated controls. These findings demonstrate rapid changes in sediment dynamics and chemistry following seagrass removal, driven in part by changes in the faunal community.

Key words: Accretion, benthic fauna communities, seagrass disturbance, simulation, surface elevation.

4.1 Introduction

Tidal saline wetlands are critical ecosystems of the nearshore environments. Their efficient functioning is governed by several factors that include the hydrology of the area, dynamic forces originating from tidal currents and wind-generated waves, and by the biophysical interactions between these forces as well as the organisms that inhabit the environments. Seagrasses are important for their biodiversity and ecosystem functions. They provide food and shelter to a variety of marine organisms, including commercially important fish species, and thus constitute a valuable component of the near-shore ecosystem (Howard et al., 2014). Their dense canopies help in sediment stabilization and shoreline protection as a result of their effectiveness in wave diminution and the entrapment of suspended particles (Gacia et al., 1999; Gacia et al., 2002; Hansen & Reidenbach, 2013; Hendriks et al., 2014). Dense seagrass meadows are thought to show as much as a threefold difference in re-suspension of fine-grained sediment as compared to un-vegetated areas (Gacia et al., 1999; Gacia, 2001; Hansen & Reidenbach, 2013). Seagrass beds filter particles from the overlying water (Gacia et al., 2003), helping to maintain water transparency and quality and thereby creating a photic zone that is conducive for photosynthesis. Furthermore, they are a major site for carbon burial and sequestration in the biosphere as a result of their capacity to sequester and trap sediment rich in C_{org} (Duarte et al., 2005; Mcleod et al., 2011). The C_{org}, is derived from the sequestered carbon or from the sestonic materials brought in by water (Gacia et al., 2002; Kennedy et al., 2004; Papadimitriou et al., 2005). These sediment particles are often rich in organic matter (OM) which is estimated at 4.1% of the sediment (Kennedy et al., 2010).

Seagrasses are known for supporting rich macro faunal communities (Hemminga & Duarte, 2000). Their vegetated areas uphold a significantly larger faunal diversity compared to un-vegetated patches (Boström et al., 2006; Hemminga & Duarte, 2000).

The complexity of the root system provides shelter and protection for benthic infauna communities (Fonseca & Fisher, 1986; Larkum et al., 2006). Healthy habitats are likely to host a diversity of detrivores that hasten the decomposition of the litter hence aiding in nutrient cycling.

Recent research has identified seagrasses as one of the most threatened ecosystems on earth and it is estimated that close to a third of all seagrass areas have been lost in the last 140 years (Waycott et al., 2009), with a current loss rate of about 1.5% year^{-1} (Pendleton et al., 2012). Major drivers of seagrass loss are land erosion, eutrophication and mechanical damage due to dredging, seining, boat mooring and anchoring (Orth et al., 2006, Short & Wyllie-Echeverria, 1996). Natural factors such as shoot loss during strong waves, occurrence of extreme weather events, diseases such as mass wasting and grazing pressure are also known to change the health status of the seagrass ecosystem (Herkül & Kotta, 2009; IPCC, 2014). Seagrass loss not only leads to habitat loss, thus impacting on the benthic fauna, but also alters the biogeochemical functions of the sediment potentially turning it from carbon sinks to carbon emitters (Pendleton et al., 2012). Despite these challenges, seagrass ecosystems are not passive but demonstrate resilience as a result of the strong bio-geomorphic forces and the eco-engineering activity by benthic organisms leading to modification of the environment and habitat persistence (Cheong et al., 2013). Such mechanisms are important for the stability of wetlands in relation to sea-level rise and are dependent on a feedback relationship between hydro edaphic conditions and plant growth contributing to organic matter accumulation and trapping of particles on transit (Mckee et al., 2007; Mudd et al., 2009). The elevation and accretion rates may be minimal where there is less sediment inflow or where geomorphic forces are large leading to subsidence. Depending on the magnitude, disturbances such as sedimentation, erosion and tectonic adjustments can alter the normal functioning of the systems on a small scale that may not be noticeable but if they cross a threshold can compromise their stability and resilience eventually leading to regime shift of the marine biodiversity. Different systems of seagrasses respond differently to disturbance. These differences may arise from hydrology, which influences the level of scouring and erosion, microbial community composition and activity. This in turn influences rates of carbon remineralisation which may alter sediment biogeochemistry and the contribution of plant material to the below ground carbon pool (Kennedy et al., 2010; Pendleton et al., 2012). Observations of the impacts of seagrass disturbance by both the natural and human induced drivers are a process that requires long term monitoring. Equally, better understanding of the resilience of seagrasses and ability of the system to keep pace with sea level rise require long-term measurements of surface elevation and sediment dynamics. Such long-term data sets do not exist at present, and there are relatively few studies that explore the impacts of seagrass loss on sediment chemistry and biology. Well-designed experiments would allow the exploration of such effects without confounding, and the very large literature on removal experiments in benthic habitats has produced some of our most important insights in coastal ecology. Few of the related experimental based studies globally have investigated the impact of seagrass removal on either the sediment dynamics or on the faunal communities (Dahl et al., 2016; Macreadie et al., 2014, Connolly, 1995; Herkül & Kotta, 2009). A thorough literature search reveal that this study is the first to explore the impact of artificial removal of seagrass on a wide range of sediment dynamics and on the faunal communities. The study used artificial removal of seagrass canopy cover in randomly selected treatment plots and comparisons with intact controls to explore the impacts of seagrass canopy removal on surface elevation and sediment dynamics as well as on the benthic faunal communities on a temporal scale.

4.1.1 General objective

Determine the impacts of seagrass loss on sediment dynamics and on the benthic faunal communities

4.1.2 Specific objectives

i. Determine the effect of seagrass removal on surface elevation
ii. Determine whether seagrass removal has significant effects on the water current speed
iii. Determine the effect of seagrass removal on sediment carbon density
iv. Determine the effect of seagrass removal on litter decay rates
v. Determine the effect of seagrass removal on benthic faunal communities

4.2 Materials and methods

4.2.1 Study site

This study was carried out at Gazi Bay ($4^{o}25$'S, $39^{o}30$'E), of Kwale County, Kenya. The study area was between the western and the eastern creeks, in the shallow part of the lagoon about 150m from the mangrove forest (Fig 13). Although all the twelve seagrass species recognized in the bay are present, the study area is dominated by *Thalassodendron ciliatum* (Forssk.) den Hartog, *Thalassia hemprichii* (Enhrenberg) Aschers., *Enhalus acoroides* (L.f.) Royle, *Syringodium isoetifolium* (Aschers.) Dandy, *Cymodocea rotundata* Ascherson, and *Cymodocea serrulata* (R. Braun) Aschers. & Magnus. These are observed to grow either as monospecific stands or mixed with other seagrass species with their coverage extending between the intertidal to the subtidal areas. Minor species comprise of *Halodule uninervis* (Forssk.) Aschers., *Halodule wrightii* (Aschers.), *Halophila minor* (Zoll.) den Hartog, *Halophila ovalis* (Braun)

Hooker, *Halophila stipulaceae* (Forssk.) Aschers. and *Zostera capensis* (Setch), Macroalgal species are also abundant in these seagrass meadows which include; *Gracilaria cortica*, *Gracilaria saloicornia*, *Halimeda* species, *Cystoseira trinoids*, *Dictyota spp*, *Hyponea cornata*, *Sargassum spp*, *Turinaria decudrens*, *Ulva partusa* and *Ulva reticulata*.

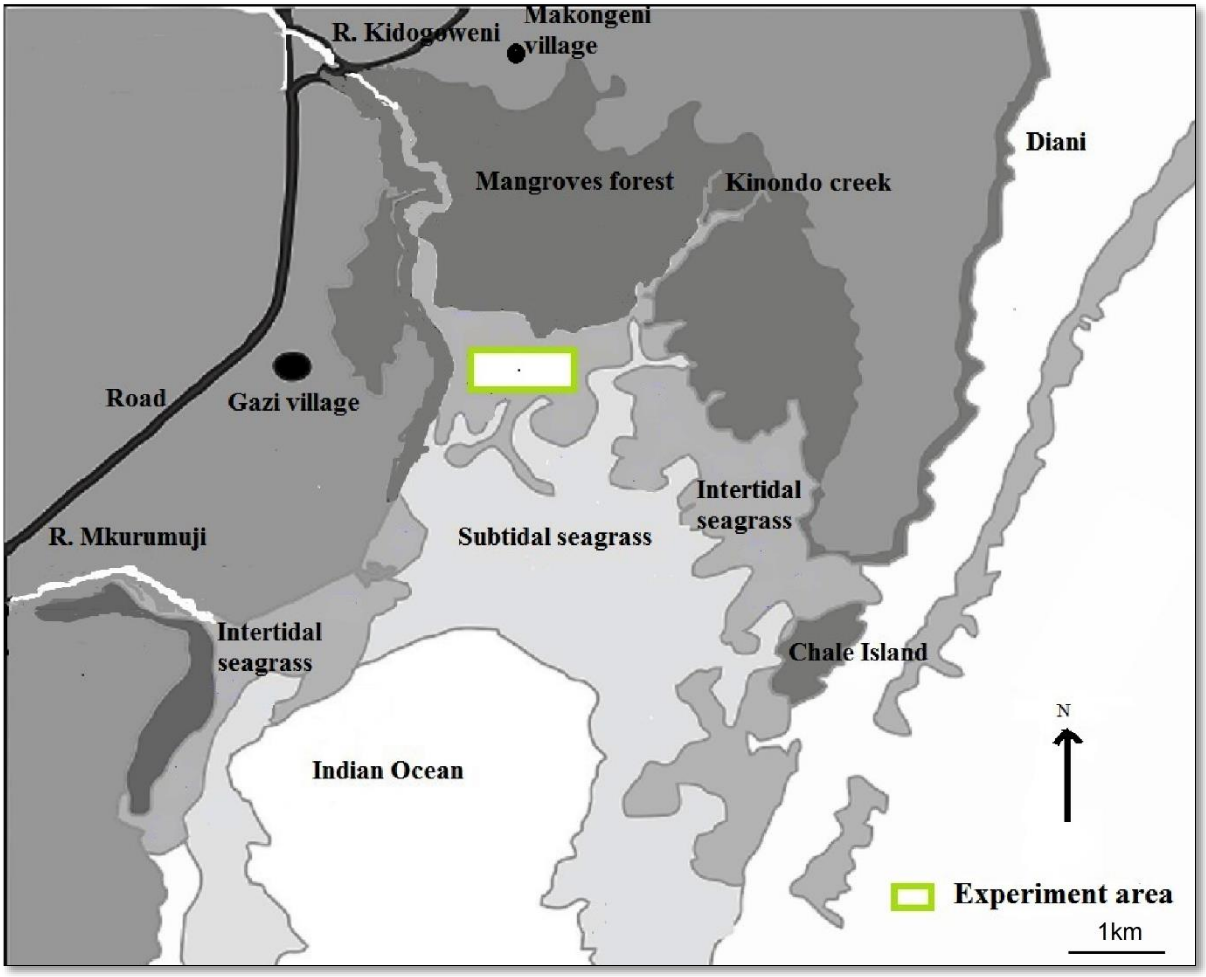

Fig. 13: Study area in the seagrass beds of Gazi Bay used for the removal experiment.

4.2.2 Sampling design

Eight plots measuring 3 by 2m, were selected with four from each of the monospecific stands of *E. acoroides* and *T. hemprichii* and at a minimum distance of 30m apart (Fig 14), in an area of the bay considered to have less human interference. Plots were randomly assigned to removal and control treatments, with the four corners of each plot being marked with PVC pipes. The treatment involved clearing all the seagrasses within the plots at surface level with a pair of garden shears while ensuring minimum sediment_disturbance. Any regrowth was cleared every 30 days within the monitoring period of eighteen months. A timber platform was used every time during clearing and data collection to avoid disturbance or human artefacts.

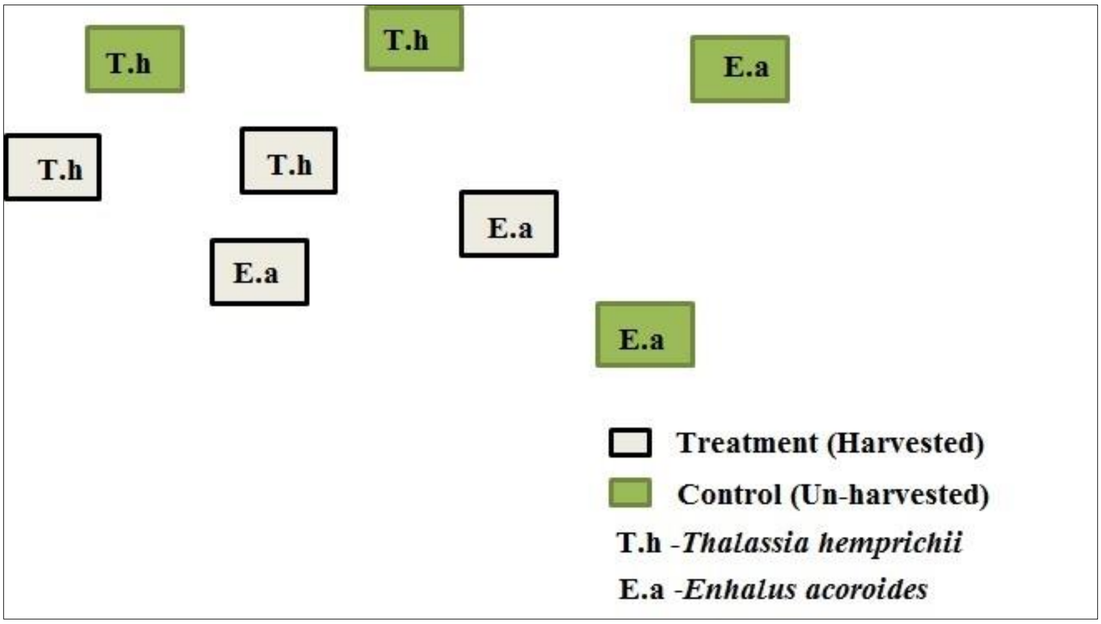

Fig. 14: Sampling design of the treatment and control areas in the removal experiment

4.2.3 Sediment Elevation Tables (SETs)

Sedimentation Elevation Tables (SETs) were established within the 3 by 2m plots. In each SET station, 6 stainless steel rods were hammered to the bedrock at a spacing of 1m ×2m (Fig 15). A spirit level was used to ensure that the rods were inserted vertically and a hack saw was used to cut the top of the rods leaving a projection of 20cm above the sediment surface. In each of the four quadrants of each SET, four transparent plastic tubes were placed by fixing them to sticks projecting above the sediment surface to trap any settling sediment. The trapped sediment was emptied monthly and the tubes returned to the respective positions. Measurements of surface elevation change were done monthly during the low spring tides as this is the time the area is accessible by foot ensuring that the readings were done from one direction throughout. Surface elevation measurements were obtained by measuring the height of the steel rod from the sediment surface. A thin, circular horizontal disc (< 1mm thick and diameter 40mm) was used, placing it at the base to avoid taking measurement from an area with scouring action.

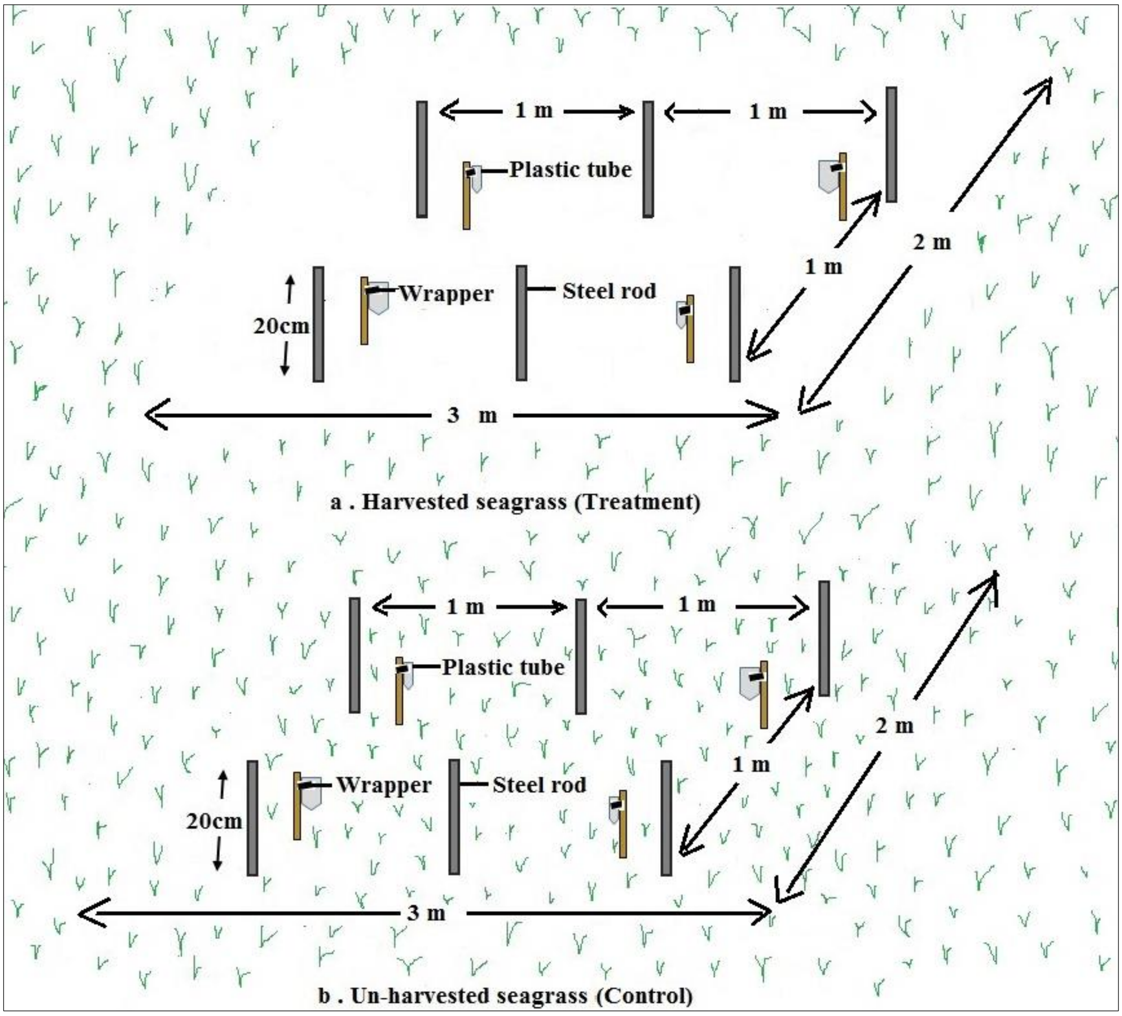

Fig. 15: Illustration of the treatment and control areas of the seagrass removal experiment

4.2.4 Measurements of change in C_{org} density

Measurements of carbon density for surface sediment were intensively done by taking 4 shallow cores of 5cm depth per plot. Between 5cm and 50cm depth, two sediment cores were taken per plot, using a 5cm internal diameter Russian peat sampler corer, one year after the setting of the experiment. In the laboratory, the cores were sliced into 5cm sub-sections and were oven dried for 72 hours at 60°C to obtain a constant weight. Dry Bulk density (DBD) (the dry weight of sediment per unit volume) was calculated for each of the ten sub-sections per core as follows:

DBD (g/cm^3) = Dry weight/Original volume of the sediment

Organic matter was measured in each of the ten sub-sections per core by Loss On Ignition (LOI) techniques, using a muffle furnace at 450°C for 6 hours. The weight loss was used to calculate the % LOI and hence the organic matter (OM) and the organic carbon density.

% LOI = ((Initial dry weight – Weight remaining after ignition)/Initial dry weight) × 100).

C_{org} values were derived using the relations:

% LOI < 0.2: % C_{org} = 0.40*% LOI-0.21

% LOI > 0.2: % C_{org} = 0.43*% LOI - 0.33 following protocols on blue Carbon manual (Howard et al., 2014; IPCC, 2013).

4.2.5 Litter decomposition

Adjacent to the experimental plots, seagrass leaves of the two species *E. acoroides* and *T. hemprichii* were harvested, washed and dried in the oven for 48 hours at 80°C to a constant weight. Known weights (3g) were placed in 5cm by 3cm nylon mesh bags and sealed. These bags had a mesh size of $1mm^2$ and the assumption was that the

$1mm^2$ mesh size was small to prevent any significant loss of small litter particles, but large enough to allow microbial colonization and entry of small benthic invertebrates. In separate points, four litter bags were placed at each of the following depths: 5cm, 10cm and 15cm in each of the four corners of the monitoring plots, such that a total of 12 bags were placed in each of the treatment and control plots. One bag from each depth profile was retrieved every 15 days at spring tides and taken to the laboratory where they were washed on 1mm sieve to remove sediments. The resulting litter after washing was oven dried at 80°C for 48 hours to obtain the dry weight. The decay rates along the depth profiles were then calculated and compared.

4.2.6 Measurements of the effect of seagrass removal on current speed

The effect of seagrass removal on the speed of water current in both the treatment and control plots was measured indirectly by use of plaster of Paris “clod cards” (gypsum blocks) according to Jokiell & Janice (1993). Measurements which took place during spring tides were done five times during the experimental period in the following months; February, March, August and December of 2015 and June of 2016. The clods were prepared by mixing 100ml of fresh water with 80g plaster of paris powder manufactured by Hobby Craft Trading limited, Dorset UK. The powder was slowly added to water and stirred with a spoon after which the slurry was poured into plastic ice cube trays with rectangular troughs measuring 4×2×1.5cm. The trays were tapped vigorously several times to dislodge air bubbles. Clods were allowed to harden for 20-30 minutes before removing them from the trays. They were then sanded at the bottom to attain a uniform weight of 12.5 ± 1.5g. Each of the plaster cubes was glued to a plastic plate measuring 3 cm × 8 cm with silicone cement (No Nonsense Ltd.BA 228RT) and the combined weights recorded. Four plaster clod cards facing the four compass directions were fixed to straight poles at heights of 0cm, 15cm and at 80cm from the sediment surface and one of each pole placed at the centre of each of the 8 plots. They were allowed to remain in the field for 24 hours after which they were removed and taken to the laboratory for oven drying at 40°C for 48 hours (Fig 16). The weight loss of the clod cards after exposure to the field was determined by subtracting the combined dry weight of the clod, plastic plate and the silicon cement before exposure. The assumption was that the weight of the plastic and the silicon cement remained constant being not affected by corrosion by sea water.

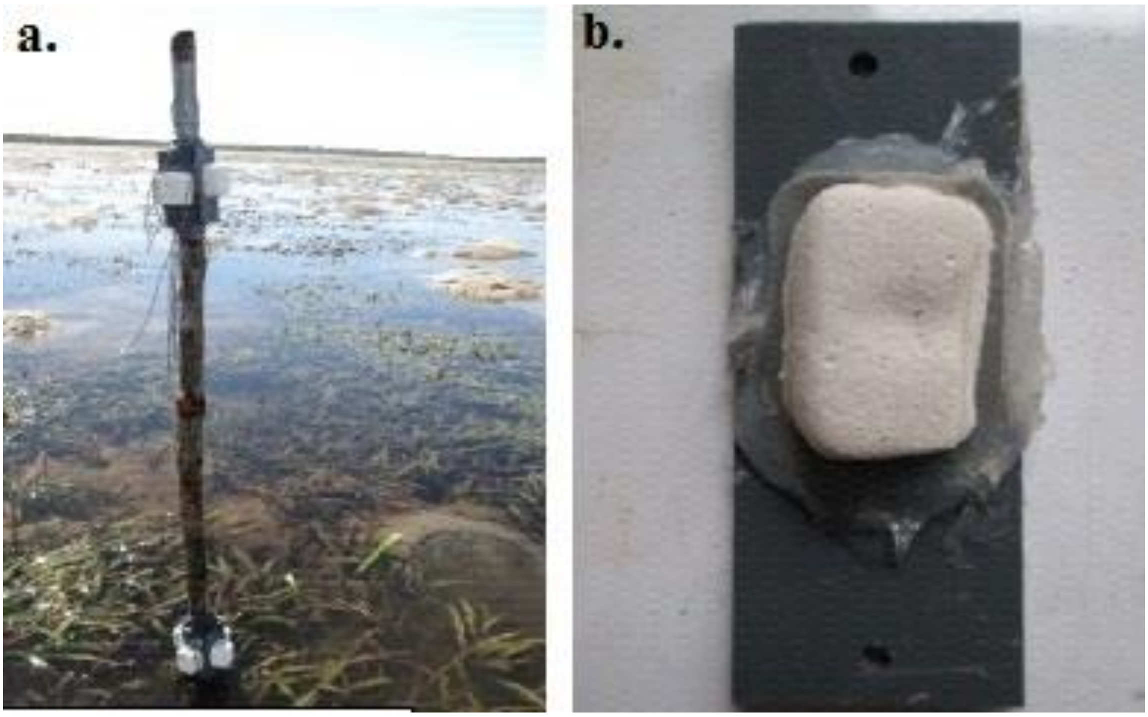

Fig. 16: Plaster of Paris clods for the current speed monitoring

(a) **Clod cards mounted to a stick in field (b) Clod card drying after removal from field**

4.2.7 Percentage moisture and grain size distribution analysis

Surface scrapes of sediment samples approximately 50 grams were taken after the eighteen months from four random locations in each plot for granulometric analysis. In the laboratory, the sediment samples were weighed and oven-dried at 80°C for 24 hours, after which the sediments were reweighed to obtain the percentage moisture content. For grain size analysis, about 25 grams of the dry sediment of each sample was treated with 10ml of aqueous sodium hexametaphosphate and passed through a series of sieves of varying mesh aperture; ranging from 63 to 500µm mesh size to determine the relative proportion of each grain size in the sediment. Four grain size categories based on the particle sizes namely: Silt clay (< 63mn), fine sand (>63-125mn), medium sand (>125-500mn) and coarse sand (>500mn) were identified. The quantity of each size class category was used to calculate the % proportion of each size category in the sediment.

4.2.8 Effect of seagrass removal on faunal communities

Burrows (holes in the sediment) and mounds in the sediment in all the plots were counted on the following months: August 2015, September 2015, December 2015, February 2016 and March/April 2016 to determine the effect of seagrass removal on the faunal communities. These were caused by a range of fauna, including burrowing crabs, fish and shrimp, but the prominent sediment mounds found at the site were caused by the burrowing shrimp, *Callianassa spp*. At the same time the diameter of the burrows were measured in order to have an idea of fauna size and possibly associate its activity to the impact on the sediment. The visual signs of bioturbation were used as an indication of activity of bioturbators following the treatment. Since the signs of increased activity of the bioturbators were not anticipated when designing the experiment, information related to this started to be collected five months after the start of the experiment.

4.2.9 Data analysis

Prior to all analyses, the assumptions of normal distribution (Shapiro–Wilk test) and homogeneity of variances (Levene’s or Bartlett’s tests) were checked followed by a graphical inspection of the data. Repeated measures ANOVA using SPSS was used to compare sediment elevation between the treatment and control areas. A three-way mixed ANOVA in SPSS 20, was used to test the effect of the factors seagrass treatment (removal vs control) and height of gypsum clods on the percentage weight loss of gypsum clods over a 24 hour period, with time as a within-subject factor. In all statistical tests, the significance level was set at $\alpha = 0.05$.

4.3 Results

4.3.1 Surface elevation in vegetated and un-vegetated seagrass beds

The general trend observed from the SETs experiment was cumulative surface elevation gain in the control areas and loss in the seagrass removal areas. While erosion was evident from the start in the removal areas, there was a short phase of seven months when erosion was observed in the removal and the controls coinciding with the period of long rains in the bay but clear differences between the removal and their control appeared eight months from the start of the experiment (Fig 17). The mean (± 95% C.I) sediment elevation over the eighteen-month period in the control areas was 7.6 ± 0.4 mm yr^{-1} while the erosion rates were -15.8 ± 0.5 mm yr^{-1}. Repeated measures ANOVA using treatment and time factors and for the treatment-time interaction, revealed significant interaction effect of treatment and time on sediment elevation change $F_{(17,102)}=3.59$. $p<0.01$.

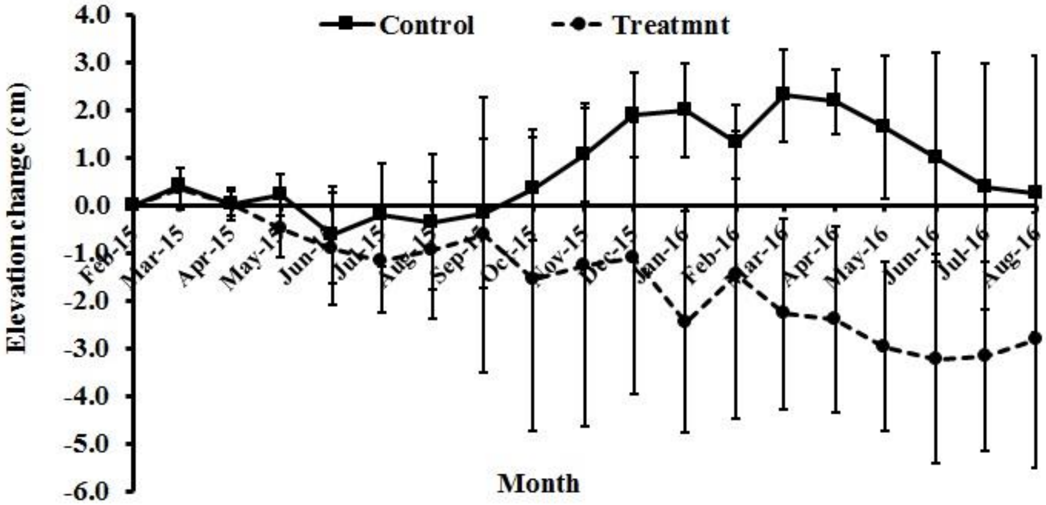

Fig. 17: Mean surface elevation change (± 95% CI) in the seagrass beds of Gazi Bay relative to the initial height of each pin in February 2015

4.3.2 Effects of seagrass removal on water current speed

Clod cards placed in the seagrass removal plots lost more weight than those in the controls within the 24 hour period, suggesting higher current speed in the removal plots. At the same time, less weight loss of the clod cards was recorded for those that were placed close to the sediment surface followed by those at 15cm height while the highest was at 80cm (Fig 18). A two way ANOVA in which treatment and height were used as factors and the treatment and height interaction revealed significant interaction effect on treatment and height on weight loss ($F_{(2,\,12)} = 6.102$, $p = .015$), but there was no significant effect of treatment and time on clod card weight losses.

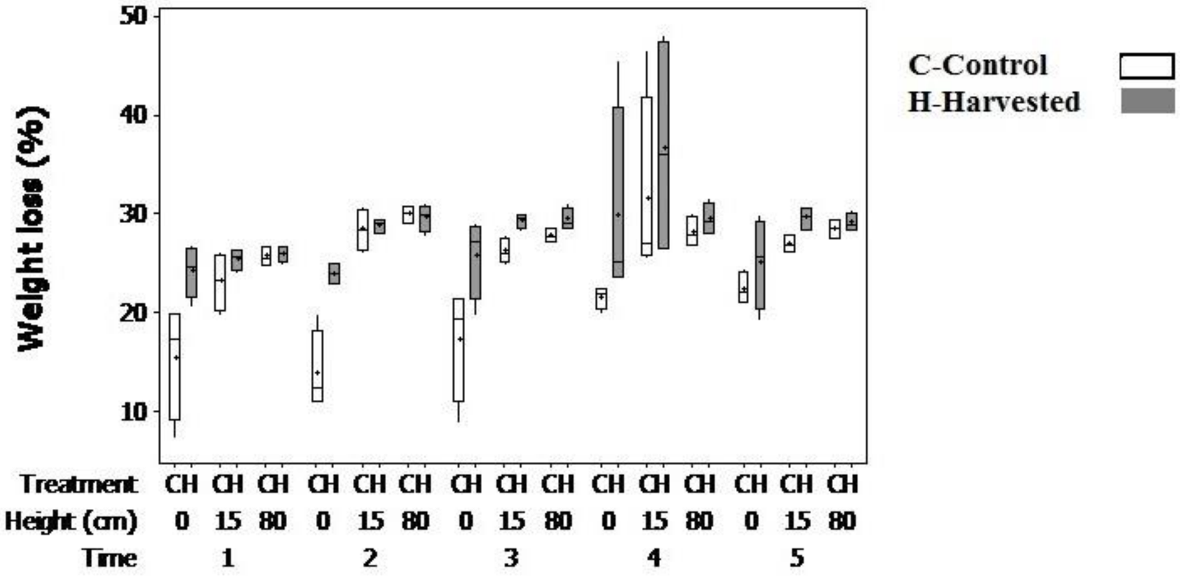

Fig. 18: Effect of seagrass removal treatment on % loss of gypsum clod weights (dots median and quartile range) at different heights in a 24 hour exposure period

4.3.3 C_{org} density between the treatment and their controls

The general observation was that C_{org} density varied between the treatment and the control areas in the surface sediment within the first 5cm for both species at the end of the monitoring period. A one way ANOVA indicated significant effect of treatment on carbon density in the surface sediment: ($F_{(1, 32)} = 16.95$, $p = 0.006$). Down the profile, there was no consistency on C_{org} density between the treatment and their respective controls (Fig 19). A fully nested ANOVA analysis in which depth and treatment were used as fixed variables while the cores were nested in the treatment and depth did not show any significant effect of treatment along the entire depth profile.

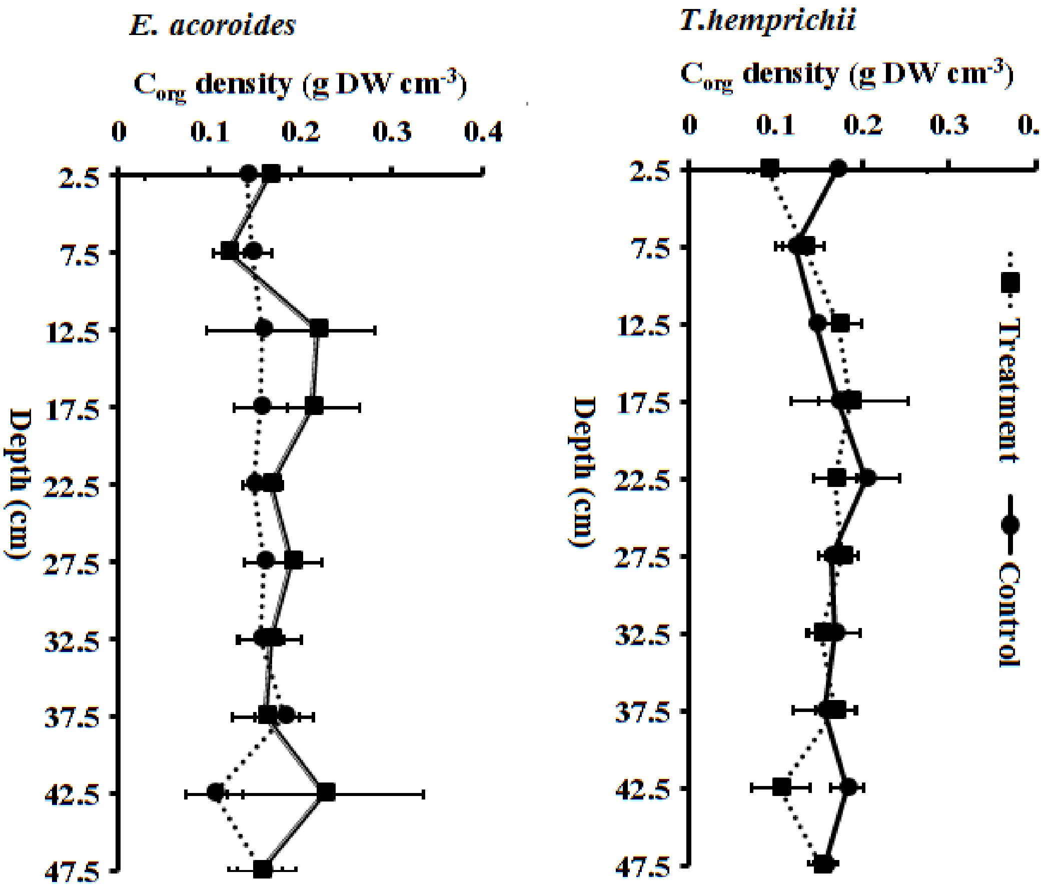

Fig. 19: Sediment C_{org} along a 50cm depth profile in the treatment and the control areas after eighteen months (Mean± 95% C.I)

4.3.4 Sediment trapping in the treatment and control areas

Sediment accumulation in both the treatment and their controls between September 2015 and May 2016 indicated fluctuation in both the treatment and the control areas. More sediment trapping was recorded in the seagrass removal treatment areas (Fig 20) but the variation was high such that the treatment did not show statistically significant effect on sediment trapping.

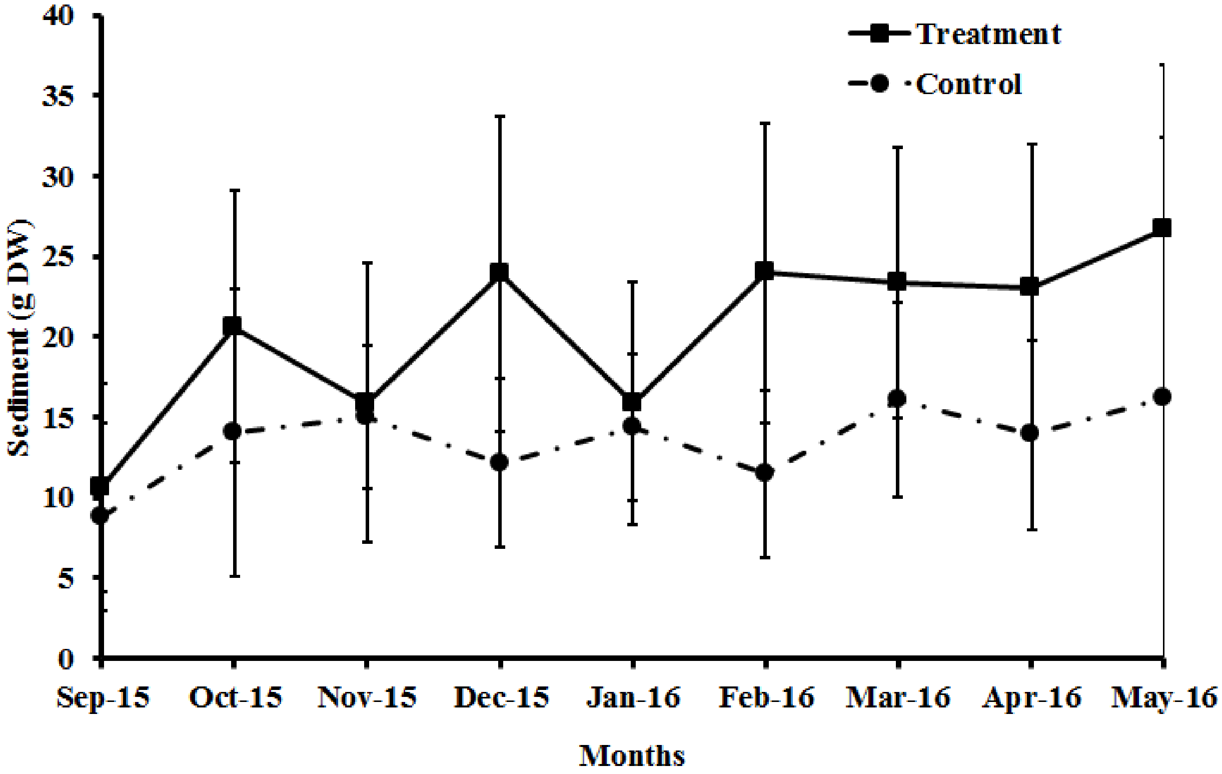

Fig. 20: Trends in quantity of sediment trapped in both the treatment and control areas (Mean ± 95% C.I)

4.3.5 Litter decay at varying depth profiles

The seagrass litter showed exponential decay throughout the two month period of monitoring across all the depths with well-fitting models in all cases. Higher litter decay rates were recorded in treatment areas as compared to the controls (Fig 21). A two way ANOVA with depth and treatment as fixed factors revealed significant effect of treatment on the litter decay rates ($F_{(1, 2)} = 22.50$, $p = 0.042$) while depth did not have significant effect on litter decay rates.

a. 5cm, b. 10cm and c. 15cm depth

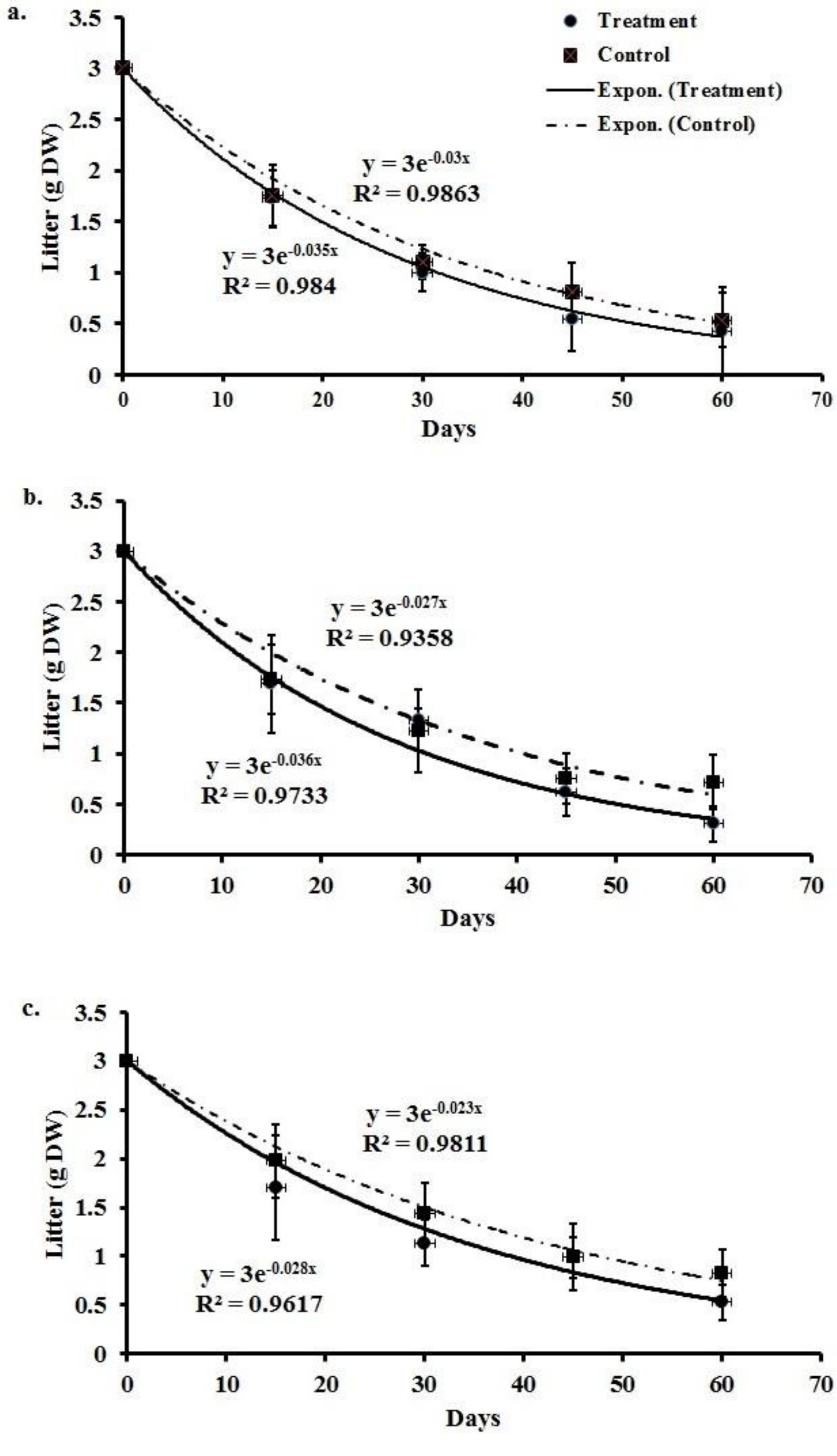

Fig. 21: Litter decomposition over a two month period at (a) 5cm, (b) 10cm and (c) 15cm depth in the seagrass removal and control plots (Mean ± 95% C.I)

4.3.6 Percentage proportion of the grain size fractions in the treatment and control areas

Highest proportion of grain size was recorded for fine sand in both the seagrass treatment and control areas at 43.6 ± 11.9 % and 47.4 ± 6.3% respectively. Coarse sand had the lowest proportion at 6.9 ± 1.1% and 7.9 ± 5.5% in the seagrass treatment and control areas respectively. The proportion of silt was 30.9% and 25.6% in the treatment and control areas respectively (Fig 22). Analysis of the relative proportions of the grain size fractions revealed that treatment did not have significant effects on the proportion of grain sizes.

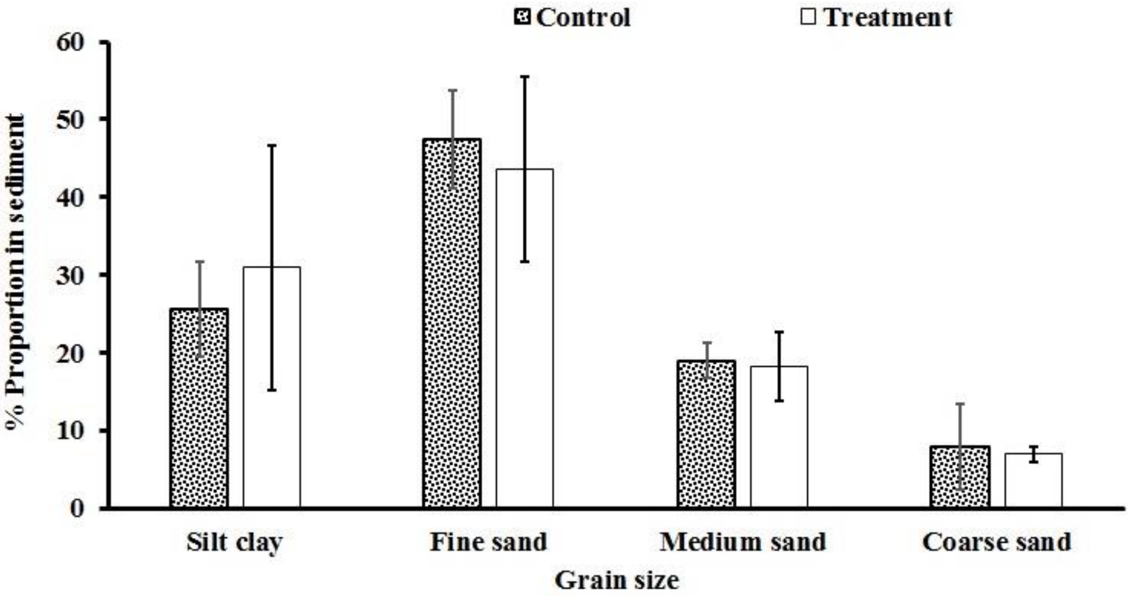

Fig. 22: Mean (± % C.I) proportion of grain sizes in the surface sediment between the treatment and control areas

4.3.7 Effect of seagrass removal on the fauna

Over the first 275 days from the start of the experiment, there was an increasing trend in the number of sand mounds and burrows in the seagrass removal treatment areas while the number remained almost uniform in the controls after which a decline was observed for both (Fig 23). After one year the numbers of burrows in both the treatment and control areas tended to rise while the number of mounds appeared to level (Fig 23). A two way ANOVA using treatment and time factors revealed statistically significant interaction effects of treatment and time on the number of burrows, $F_{(8.48)} = 4.452$, $p < 001$. A two way ANOVA analysis on the effect of treatment and time on the number of mounds, revealed no significant interaction effect but post hoc analysis indicated significant effect of treatment and time on the number of mounds ($F_{(1,6)} = 20,872$, $p = 0.004$) and ($F_{(8,48)} = 8.606$, $p < 0.01$) for the control areas.

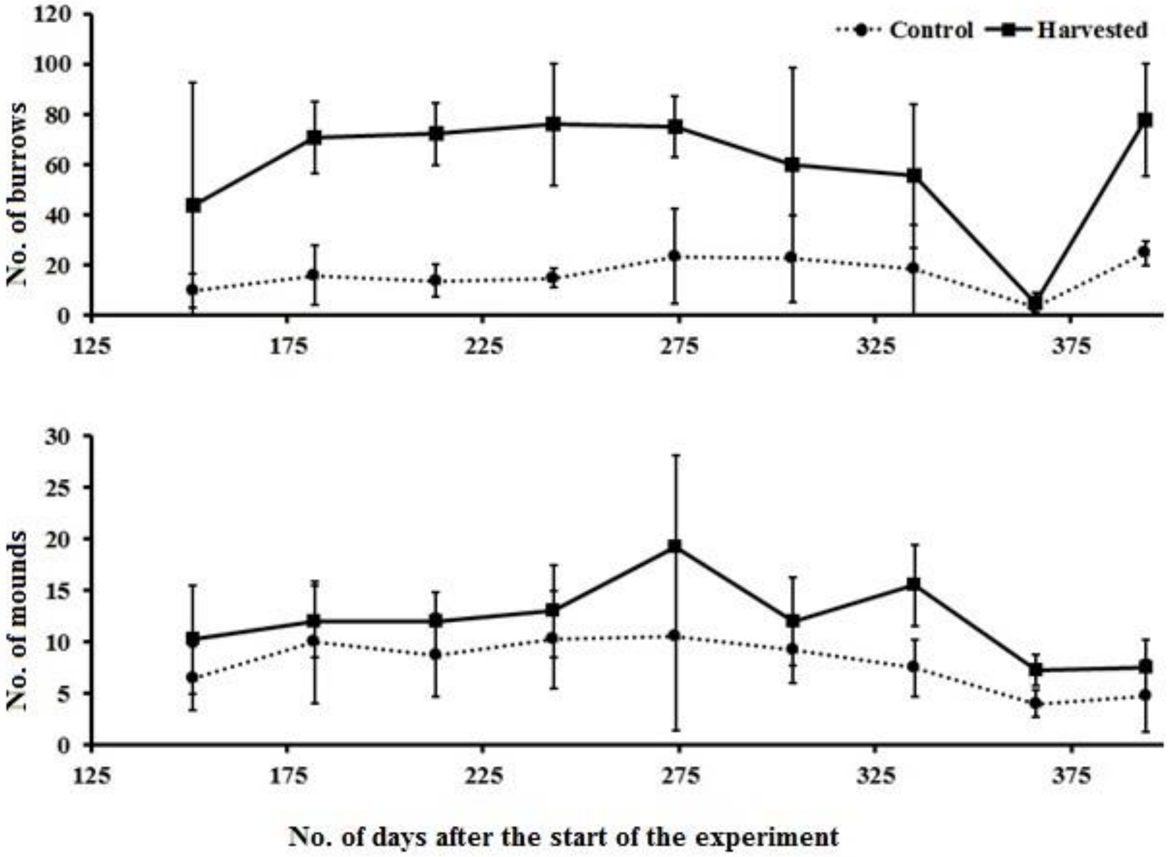

Fig. 23: Trends in the number of burrows and sand mounds between the seagrass removal treatment and control areas over 365 days (Mean ± 95% C.I)

4.4 Discussion

This manipulative experiment is the first of its kind globally and explicitly demonstrates the impact of seagrass canopy removal on the sediment dynamics and on the fauna communities. The study found a significant effect of seagrass canopy removal on sediment elevation change. Over the eighteen-month period, there was cumulative surface elevation change with sediment build up in the control plots while erosion occurred in the removal areas. However, notable difference that could be associated with the treatment started appearing eight months from the treatment implying that the below-ground components exerted influence on sediment elevation change before they disappeared. Differences observed between the removals and the controls could be attributed to the sediment trapping by the seagrasses as well as the burial of organic matter in the seagrass un-harvested areas. The fact that positive elevation change was recorded in the control areas while negative elevation change occurred in the removal areas demonstrates the role of seagrass in influencing upward elevation change and how disturbance or removal of seagrasses from an area may contribute to sediment loss. Sediment elevation is thought to be caused by several processes that occur within the soil profile which include: sedimentation, erosion, compaction, and groundwater discharge/recharge (Cahoon et al., 2011). Past studies on surface elevation have argued that sediment deposition, erosion and compression are processes directly influenced by the hydrologic conditions of each site (Cahoon et al., 2011). There was trapping of sediment by the trap tubes albeit lots of fluctuation, which apart from being linked with the trapping effect of the seagrasses, was also associated with the activity of the bioturbators.

Analysis of the carbon density between the treatment and control areas indicated less organic carbon in the surface sediment of the treatment areas as compared to controls. These results suggest that the seagrass canopy removal led to reduction in the input of organic carbon into the system as the removal of the seagrass meant that photosynthetic tissues of the plants were lost leading to a reduction in the autotrophic synthesis of organic carbon as well as the trapping of the allochthonous carbon by the seagrasses. Despite a short monitoring period, significant effect of the seagrass removal on the near surface sediment C_{org} was recorded further providing evidence that seagrasses are important in the input of organic carbon into the sediment. Lack of significant effect of the seagrasses down the depth profile was an indication that the process of carbon burial is gradual and takes considerable time. In a closely related experiment in the region on effects of shading and simulated grazing in a *T. hemprichii* dominated stand, Dahl et al., (2016) observed no impacts on the sedimentary carbon while the high-intensity disturbances caused a clear depletion of carbon biomass and reduced the seagrass meadow’s capacity to sequester carbon. In another study, on the effect of small scale disturbance, Macreadie et al., (2014), found no detectable effect on sedimentary carbon, while a survey assessing the effect of disturbance as a result of eutrophication and sedimentation at a meadow scale, indicated a reduction in sedimentary carbon as a result of increased erosion of the surface sediment (Marba’ et al., 2015). Some of the carbon input into the seagrass beds is associated with the particles that are trapped and deposited in seagrass-vegetated sediments which have been found to be rich in organic matter (OM), averaging 4.1% (Kennedy et al., 2010). However, this trapping effect is reduced with decreased seagrass density that may be driven by natural and human stresses on seagrass meadows such as storm disturbance or eutrophication (Gacia et al., 2003). The role of seagrasses in trapping of organic carbon was earlier demonstrated by a stable carbon isotope study in the bay, in which the mangrove carbon signal was found to completely fade within the seagrasses, which suggested that this was due to the trapping and absorption by the seagrasses (Hemminga et al., 1994). The capacity of seagrass meadows to trap particles is known to be high; for instance, Barrón et al., (2004) using carbon budgets across seagrass patches of different age, recorded high input of carbon originating elsewhere and trapped into the sediments of *Cymodocea nodosa* patches in a Mediterranean lagoon. Accumulation of sediment within the seagrass beds is further enhanced by the dense network of rhizomes and roots that bind the sediments while the canopies dissipate the wave energy thus preventing its resuspension and instead making it settle at the surface (Fonseca & Fisher 1986).

When the results of surface elevation and accumulation of C_{org} in the sediment are linked with those of seagrass removal on the current speed where significant effect of treatment on the clod cards weights was observed at the sediment-water interface, a more probable conclusion is that the seagrass canopy removal significantly diminishes the sediment trapping ability by the seagrasses and also the diminunition of the water currents. Investigation on the effect of seagrass in lowering current speed measured through the clod cards recorded higher water motion in the treatment areas as compared to their controls as the clod cards placed in the seagrass harvested areas lost more weight than those in the controls within the 24 hour period. Since the clod cards placed at lower heights lost less weight compared to those at the higher heights, this indicates that the water velocity reduction capacity of the seagrasses is between the seagrass canopy height and the sediment surface. The high seagrass canopy cover plus the rhizome and root network attenuate the hydrodynamic energy of the waves at the sediment-water interface, a function that is lost with seagrass removal. Relative weight loss is caused by dissolution of the clod cards and effectively works as a suitable proxy for the indirect measurements of the seagrasses in reducing the hydrodynamic energy from tidal currents and waves (Porter et al., 2000). The water flow velocity obtained in this study does not deviate much from values that have been recorded in other areas. In an experiment on seagrass particle trapping and flow dynamics in Magalluf, Spain, Hendriks et al., (2008) recorded a water flow velocity of 30-40% and argued that these were on the higher side. While monitoring flume flow in the *Zostera marina* bed at San Juan Island of Washington, USA, Gambi et al., (1990) obtained flow velocities ranging between 10-30%. Other studies have demonstrated experimentally that even the low canopy seagrasses are capable of attenuating wave energy (Christianen et al., 2013). This study therefore compliments results of several related studies and thus confirms that seagrass meadows influence hydrodynamic patterns of the water resulting in increased sedimentation, trapping of autochthonous and allochthonous suspended matter, and reducing the degree of sediment re-suspension. It has also been argued that reduction in the water velocity increases water turbulence which reduces the diffusion boundary layer thickness around the blades of the seagrasses, potentially enhancing primary production and photosynthesis (Koch et al., 2002). However, other factors such as the salinity, temperature and the volume of the water discharge from the runoff have an influence on these physical and biological processes (Koch et al., 2007).

Although it was difficult to explicitly establish significant effect of treatment on microbial decomposition of the litter at the different depth profiles, the exponential litter decay rates and the significant treatment effect that were observed in this study confirm that the microbial activity is dependent on oxygen availability and thus rate decreases with oxygen depletion. Higher decay rates in the seagrass removed areas point at how remineralisation of organic carbon occur following seagrass degradation. While the treatment effect were clear on surface elevation, it was not possible to track and obtain reliable estimates of the accretion rates in this experiment as the removal of the seagrass canopy triggered a lot of bioturbation activities by the burrowing fauna such as the shrimps thus making it difficult to quantify the amount of accumulating sediment. Results of this study are consistent with results from previous manipulative experiments that showed a general reduction in sediment C_{org} with seagrass loss. This experiment observed that seagrass disturbance; triggered massive bio-geo-engineering in the sediment as there was an influx of shrimps (*Callianassa spp*) in the treatment areas. Due to the increased numbers, there was an increase in the build-up of more volcano shaped sand mounds and burrows. In a related work in Bolinao, NW Philippines, (Nacorda, 2008) observed that *Alpheid* shrimp made more burrows in sandy patches compared to muddy patches, impacting sediment down to 10 cm depth and this group was associated with high *T. hemprichii* consumption (Palomar et al., 2004; Vonk et al., 2008). One of the significant findings of this study is that there is a lot of ecosystem engineering within the sediment that led to the formation of the many epibenthic structures within the seagrass meadows. Due to the interaction between the integrity of the meadows and the fauna within the sediment, seagrass removal caused more burrowing by the shrimps. This has implications in the biodiversity and ecosystem functioning of the seagrass beds as bioturbation is increased leading to enhanced nutrient cycling, though may contribute to remineralisation and release of the stored carbon. At the same time, the interaction has an influence on the reduction in the hydrodynamic energy and thus reduction in sediment movement.

Though the spatial and temporal cover for the study was limited, the results nevertheless give insight into the effects of natural and human induced activities on the seagrass beds. Although there is an increasing knowledge on the importance of the vegetated coastal ecosystems, they continue to be degraded unabated as a result of the combined natural and human induced perturbations. The resources have become overexploited thus overstretching their resilience and compromising their ability to sustain marine food webs. Many of the problems are confounded by the increased demand for resources as well as a lack of fundamental knowledge to model the resilience of coastal ecosystems to increased pressure from human activities and, therefore, to predict the level of human pressure that is sustainable and predicts pathways and rates of recovery of deteriorated coastal ecosystems. In many areas, there are no explicit policies that govern the sustainable use of the resources and where formulated, there are weak mechanisms of enforcement. These problems are confounded by a lack of restoration plans for the degraded ecosystems.

The findings of this study have implications on the change and stability of the shoreline, survival of marine ecosystems and human settlements. Global mean sea-level rose by almost 0.2 m during the twentieth century (Jevrejeva et al., 2006) while the current sea level rise is estimated at 3.2 ±0.4 mm yr^{-1} (mean ± SE) (Krauss et al., 2014) and is predicted to rise by 0.6 m or more by 2100 (IPCC, 2013; Nicholls & Cazenave, 2010; Parry et al., 2009). Based on available literature, the mean surface elevation in vegetated seagrass areas is 5.0 ± 3.28 mm yr^{-1} (Mean ± SE), whereas the rate of erosion in the un-vegetated areas is 26 ± 12.3 mm yr^{-1} (Mean ± SE) (Bos et al., 2007). However there are regional variability e.g. 7.5mm yr^{-1} for Indonesia and 1.9mm yr^{-1} for the Caribbean as a result of regional variation in ocean warming and other factors and rates are predicted to accelerate (Nicholls & Cazenave, 2010). It is projected that in the 21st century,16–27 million people in Africa are likely to suffer from coastal flooding per year occasioned by sea-level rise and associated effects, with annual damage costing between US$ 5 and US$ 9 billion, if no mitigation mechanism are put in place (Hinkel et al., 2012). Suitable mitigation mechanisms such as building dykes to protect against coastal flooding and nourishing beaches to protect against coastal erosion is likely to reduce the impacts by 11–36% at a cost of US$ 300 billion and a maintenance cost of US$ 3 billion per year. In addition, between US$ 2 and US$ 6 billion per year will be spent on protection against future sea-level rise and socio-economic development (Hinkel et al., 2012). Such projections imply that the vegetated coastal ecosystems are becoming even more vulnerable to submergence and subsequent degradation (Cahoon, 2014; Ellison & Elizabeth, 1997). This does not only threaten the livelihoods of the coastal communities but the loss of the many other ecosystem services that these ecosystems provide.

The role of seagrasses as established in this manipulative experiment of the sediment stabilizing effect of the seagrass canopy have important implications for both coastal protection and ecosystem functioning and thus underline the importance of understanding the importance of conserving the seagrass meadows. The surface elevation rates recorded in this study of 7.6 ± 0.5mm yr^{-1} (mean ± C.I) and erosion rates of -15.8 ± 0.5mm yr^{-1} are therefore within this global mean and when compared to the above mentioned mean sea-level rise imply that the seagrass beds of the bay have the ability to cope with the challenges of sea-level rise by reducing sediment erodibility and maintenance of a higher bed elevation that will help to attenuate waves and overcome the challenges of sea level rise and coastal protection. This will not only provide protection to the coastal inhabitants but will greatly reduce the huge costs of building the artificial walls to mitigate the challenges in the face of the global threat of sea level rise.

Chapter five

5.0 Productivity of four dominant seagrass species at Gazi Bay

Abstract

Biomass accumulation in plants is largely dependent on the plant growth characteristics and its ability to maximize the use of available resources in conditions of resource limitation. Seagrasses are among the most productive ecosystems on earth with a mean annual production of 1012 g DWm^{-2}. This study combined several methods to determine above-ground and below-ground productivity. The leaf plastochrone and regrowth of shoots in seagrass harvested areas was used to determine the above-ground productivity while ingrowth trenches, were used to determine the below-ground biomass accumulation in four dominant seagrass species of Gazi Bay. *Enhalus acoroides* showed the highest shoot production and shoot growth rates of the four species studied at 1944 g DW m^{-2} yr^{-1} while the lowest was in *T. ciliatum* at 438 g DW m^{-2} yr^{-1} with an average of 1037 g DW m^{-2} yr^{-1} across the species studied. The highest below-ground biomass accumulation rate was recorded in *T. hemprichii* at 197.4±108.7 g DW m^{-2} yr^{-1}. Considering that below ground biomass did not appear in some ingrowth trenches after one-year means that seagrasses take considerable time to recover following disturbance and that a longer monitoring period is necessary for reliable estimates. These findings on interspecies variability in productivity rates have implications for biodiversity and ecosystem functions of seagrass beds and are useful in the development of conservation strategies.

Key words: Biomass accumulation, ingrowth trenches, plastochrone, productivity.

5.1 Introduction

Patterns of biomass accumulation in plants are largely dependent on the plant's growth characteristics and its ability to maximize the use of available resources in conditions of resource limitation. Marine plants growing in more labile environments such as in nutrient poor soils or where exposed to frequent storm surges will tend to allocate proportionately more biomass to roots to maximize the acquisition of limiting resources or to ensure that they retain the potential to regenerate when conditions improve (Hemminga & Duarte, 2000). Vegetated wetland ecosystems are often characterised by environmental stressors such as nutrient deficiency, anoxia and hyper salinity. Among the survival strategies for plants in such environments is the high allocation of carbon to below-ground roots, which in some studies have been found to account for over 80% of the biomass fixed by the plants (Kaldy & Dunton, 2000). Hence, these plants will often have higher root: shoot ratios than their terrestrial counterparts. The heavy investment below-ground is not only valuable as a source of food to the detrivores, but the elaborate root network provides an increased surface area for nutrient uptake from the sediment and to trap those on transit. Seagrasses are marine angiosperms that inhabit the shallow photic zone where they make use of the available light for photosynthesis. They are rhizomatous and modular, composed of repeating units (ramets) that exhibit clonal growth (Hemminga & Duarte, 2000). They have true roots and internal gaseous and nutrient transport systems (Paling & McComb, 2000).

The role of seagrasses in regulating water quality and nutrient cycling enable them to contribute to the carbon biogeochemistry and trophic transfers to adjacent habitats (Heck et al., 2008). As a result they are able to support rich biodiversity that include fish, birds and invertebrate species (Heck & Valentine, 2006; Heck et al., 2008). Seagrasses are among the most productive systems on earth with a mean annual production estimated at 1012 g DW m^{-2} yr^{-1} from a review of global data set (Duarte & Chiscano, 1999) and contributing 12% of the net ecosystem production in the ocean (Duarte & Cebrián, 1996). However, inter-species and inter-habitat variability in productivity of seagrasses depends on age, hydrology and the environmental settings in relation to geomorphology, latitude, climate, and management regime (Hemminga & Duarte, 2000). High productivity rates are attributed to adequate supply of the dissolved nutrients which are actively recycled within the seagrass beds (Hemminga & Duarte, 2000). Knowledge on productivity of seagrass species is important as it provides the baseline for understanding the complexity of the habitats and their distinct bioregional characteristics. Additionally, it allows comparisons of geography and evolution and provides a foundation for evaluating past and present human impacts on the global seagrass ecosystem. Furthermore, the above and below-ground productivity measurements are important in generating the root: shoot ratios (RSR) which may reflect the biomass investment patterns or the plant's response to changes in its environmental conditions such as nutrient concentrations. Measurements of seagrass biomass and productivity over time may help identify the influence of seasons on the plant growth and development. Biomass and productivity varies between species, influences carbon fixation as well as energy and nutrient transfers (Duarte & Sand-Jensen, 1990; Heck et al., 2008; Tomasko et al., 2016; Pangallo & Bell, 1988). Studies of productivity on certain seagrass species have revealed faster growth in conditions of light and nutrient availability (Agawin et al., 1996). Usually the influence of a species in a community is directly related to the species spatial cover and biomass accumulation since some seagrass species have faster growth rates and thus are able to grow and spread to vast areas (Hemminga & Duarte, 2000). Values of biomass and productivity of a plant species in a habitat can be used in computing some complexity indices and hence give a better understanding of the dominance of the species in the community (Beckschäfer et al., 2013). Despite this value, the productivity and biomass measurements of seagrasses is often a complex task due to the wide range of species diversity patterns and the challenges of working in the subtidal seagrass habitats. As a result, the biomass characteristics of some seagrass species are yet to be documented (Bouillon et al., 2007; Coppejans et al., 1992; Duarte, 1996; Hemminga & Duarte, 2000). Studies on the relationship between above-ground and below-ground productivity in seagrass ecosystems are of intrinsic interest because of the ecological importance of these processes; they also have practical applications since if above-ground productivity information is available, it can provide an easier way of estimating the below-ground productivity provided the above-ground productivity is a reliable surrogate. Despite the high species diversity in Gazi Bay, productivity studies of seagrass are limited. The few studies carried out at the site have targeted one species, *T. ciliatum* (the dominant seagrass species in the bay) thus ignoring inter-species variability in productivity and also variability between the different seagrass compartments (Duarte, 1996; Ochieng, 1995). This study aimed to investigate the variations in productivity between the four dominant seagrass species of the bay, *T. ciliatum*, *E. acoroides*, *T. hemprichii* and *S. isoetifolium*. For the purpose of obtaining more precise estimates of productivity, the present study used a combination of the plastochrone method, shoot regrowth through monthly shoot harvesting and biomass accumulation through root ingrowth trenches. This was found useful for comparing biomass production for the species of the bay with that of past studies and from other sites using similar methods and to test the efficacy of the root trench method in the biomass accumulation of seagrasses having not been used before in seagrasses.

5.1.1 General objective

Determine productivity of the four dominant seagrass species of Gazi Bay

5.1.2 Specific objectives

i. Determine shoot productivity of three of the four dominant species: *T. hemprichii*, *E. acoroides* and *T. ciliatum* through the leaf plastochrone method.
ii. Determine below-ground biomass accumulation of the four dominant species through the root ingrowth trench method.
iii. Determine the shoot biomass accumulation of two species: *E. acoroides* and *T. hemprichii* following regular shoot harvesting.

5.2 Materials and methods

5.2.1 Study site

The present study was carried out at Gazi Bay (4^o25’S, and 39^o30’E), located approximately 50 km south of Mombasa in Kwale County. Twelve seagrass species are present in the bay and based on the spatial cover, the dominant species are: *Thalassodendron ciliatum* (Forssk.) den Hartog *Thalassia hemprichii* (Enhrenberg) Aschers. *Enhalus acoroides* (L.f.) Royle and *Syringodium isoetifolium* (Aschers.) Dandy. *Thalassodendron ciliatum* has a sympodially branched vegetative system with the leaves being distichously arranged along the upright stems and commonly appear in a few patches in the mid-intertidal region while most are found in the subtidal areas of the bay (Coppejans et al., 1992). *Thalassia hemprichii* with its wide linear leaves and short shoots is more widespread covering most parts of the bay within the upper intertidal and subtidal areas. *Enhalus acoroides* with leathery strap-shaped leaves is restricted to the upper part of the sublittoral zone and inhabit sandy and rocky substrate, being rarely found on mud. The *Syringodium isoetifolium* with long subulate leaves can tolerate some emersion but is usually restricted to the upper sublittoral zone (Coppejans et al., 1992).

5.2.2 Estimation of shoot productivity through the leaf plastochrone method

Data for shoot productivity measurements was collected through the modified leaf plastochrone method in a period of six months with harvesting for growth measurements done after every fortnight. Five quadrats measuring 0.0625m^2 were established randomly in monospecific stands of each of the three dominant seagrass species: *T. hemprichii*, *E. acoroides* and *T. ciliatum* at a minimum distance of 30m from each other. All the shoots in a quadrat were counted and the shoot heights taken. A needle was used to insert a thin nylon thread at the base of the leaf sheath with the marking on the outermost and oldest leaf being the reference (Fig. 24a). Growth measurements were obtained by harvesting the tagged shoots (Fig. 24b) which were classified as “mature” (leaves with hole), “new” (mature leaves without hole) or “young” leaves (shorter than the bundle sheath of mature leaves) and the segments of new growth were removed (including new leaves), rinsed, dried in an oven at 60°C for 48 hours to a constant weight. Leaf growth rate was calculated according to the number of new leaves appearing on the marked shoots during the observation period of two weeks. The leaf materials produced after marking for each shoot divided by the time interval yielded leaf production per shoot (g shoot^{-1} day^{-1}). Aerial production rate was calculated by multiplying the leaf production per shoot by the shoot density and was expressed as g m^{-2} day^{-1}.

a. Tagged shoots in the field for growth rate monitoring

b. Measurement of shoot growth rate after harvesting

Fig 24. Tagged shoots with white nylon thread (24a) growing in field and point of shoot growth measurement of harvested shoots (24b) above the leaf bundle sheath.

5.2.3 Measurement of above-ground productivity through a shoot removal experiment

This was done in experimental plots measuring 3 by 2m in monospecific stands for two species, *E. acoroides* and *T. hemprichii* established at a minimum distance of 30m apart. All the shoots within the 3 ×2m plots were cleared and all the regrowth within the plots were harvested monthly. Every time before harvesting, a 50×50cm quadrat was laid randomly on the ground within the plots. All the re-growing shoots within the quadrat were counted and their heights taken. They were then harvested and taken to the laboratory for drying at 60°C for 48 hours to obtain the biomass.

5.2.4 Below-ground biomass accumulation in root ingrowth trenches

Below-ground biomass accumulation of four dominant species of the bay: *T. ciliatum*, *T. hemprichii*, *E. acoroides* and *S. isoetifolium* was estimated using the root in-growth trench method. In each seagrass species stand, four rectangular trenches measuring

60cm long by 20cm width and 60cm deep were established in randomly selected points at a minimum distance of 40m apart. In each of the trench, all the sediments were removed. The roots present were macerated into tiny pieces, mixed with the sediment and returned to the trench in order to keep the same mix of nutrients present in the trench sediment as in ambient sediment. The positions of the trenches were marked using 60cm long PVC pipes that were pegged at the corners. After one year, the trenches were re-dug and the all the below ground biomass retrieved (Fig 25a). New roots were separated from the sediment by washing the retrieved sediment through a 63μm sieve (Fig. 25b). The roots were oven-dried at 80°C for 48 hours to a constant dry weight. The sum of the dry weights for the roots and rhizomes was used to calculate the below-ground biomass accumulation.

25a. Retrieving roots from the trenches

25b. Washing off sediment from roots

Fig. 25: Retrieving roots from the trenches and washing off sediment from the roots

5.3 Results

5.3.1 Shoot production and leaf growth rates for the seagrass species

The production of new shoots for the three species indicated upward trend overtime apart from the month of June for *T. hemprichii* and *E. acoroides* but with a drop in *E. acoroides* and *T. hemprichii* between the period of June and July and for the three species between the months of July and August (Fig.26a). Shoot biomass production for *T. hemprichii* were relatively stable throughout as compared to that of *T. ciliatum* (26b). Shoot production for *E. acoroides* appeared to decrease between the month of March and May followed by gradual increase up to July after which there was another decline. An almost similar trend was observed in *T. ciliatum*, with the production dropping between March and April and a peak being recorded in July. However, production in *T. hemprichii* was relatively uniform over the monitoring period (Fig 26b). Of the three seagrass species studied, *E. acoroides* showed the highest shoot biomass production at 0.037 ± 0.002 g DW shoot^{-1} d^{-1} followed by that of *T. ciliatum* at 0.006 ± 0.004 g DW shoot^{-1} d^{-1} with *T. hemprichii* having the lowest at 0.005 ± 0.001 g DW shoot^{-1} d^{-1} over the six months monitoring period. However, the production of new shoots appeared to be highest in *T. hemprichii* at 2.3 ± 1.3 shoots d^{-1} followed by that of *T. ciliatum* at 1.4 ± 0.7 shoots m^{2} d^{-1} with *E. acoroides* having the least at 0.6 ± 0.5 shoots m^{2} d^{-1}. Comparison on the mean production of new shoots between the three species, indicated significant difference ($F_{(2,12)} = 5.90$, $p = 0.016$). Similarly the mean shoot biomass production between the three species was significantly different ($F_{(2,12)} = 35.20$, $p < 0.001$). Shoot growth rates for the three species appeared to be stable throughout the monitoring period with a slight peak in the month of May (Fig 26c).

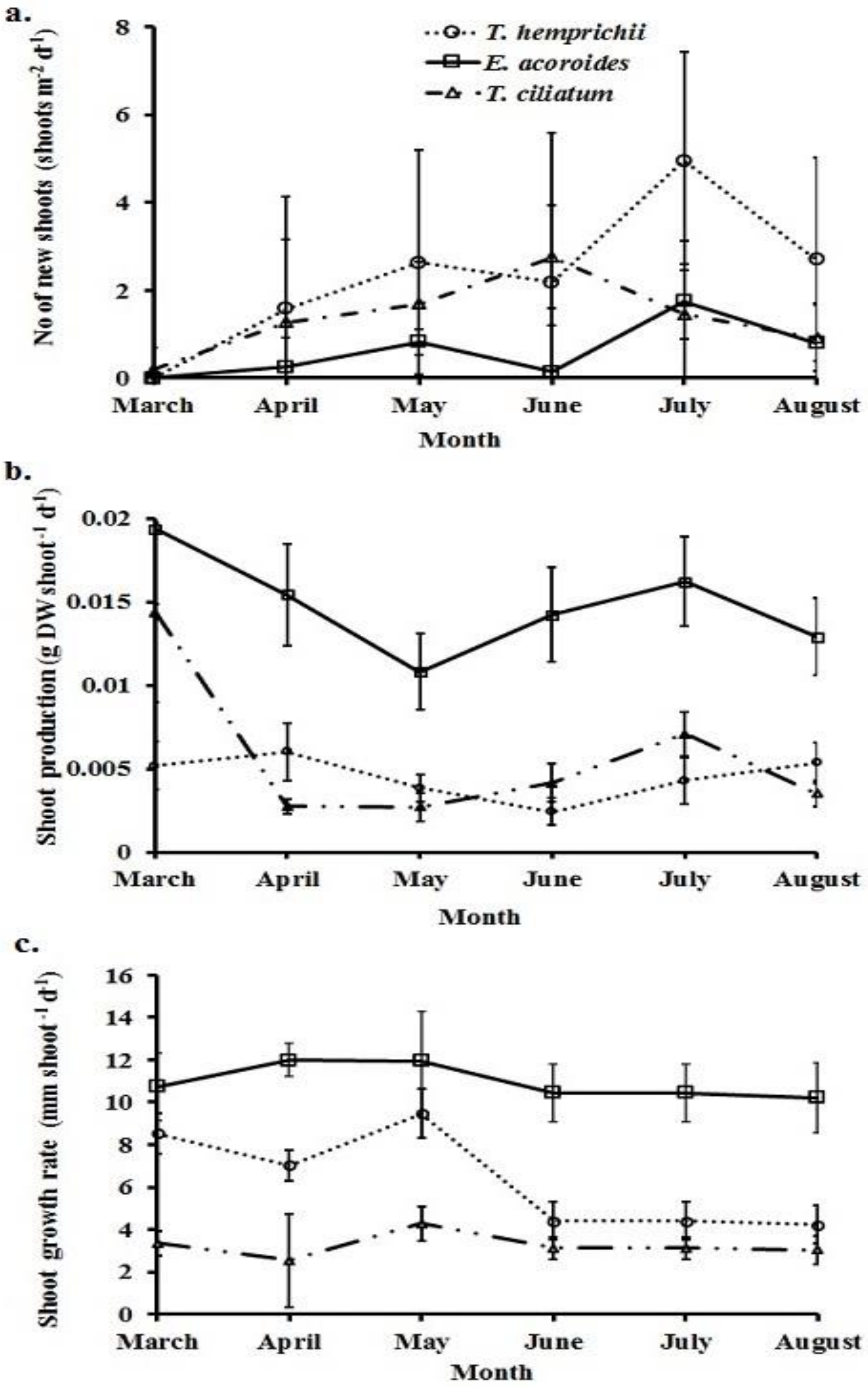

Fig. 26: Measures of shoot productivity (mean ± 95% C.I) for three dominant seagrass species of Gazi, Bay. (a) Number of new shoots, (b) Shoot production and (c) Shoot growth rates

Leaf growth rates and the production of new leaves were highest in *E. acoroides* followed by that of *T. hemprichii* while *T. ciliatum* recorded the least in the six months monitoring period (Fig 27a). The lowest leaf growth rates and production of new leaves were recorded in the month of June for all the three species (Fig 27a). There were tremendous leaf growth rates and production of new leaves in the month of July for the three species followed by a drop for *E. acoroides* and *T. ciliatum* in the month of August (Fig 27b). The mean leaf growth rate between the three species was highly significantly different ($F_{(2,12)} = 160.95$, $p < 0.001$).

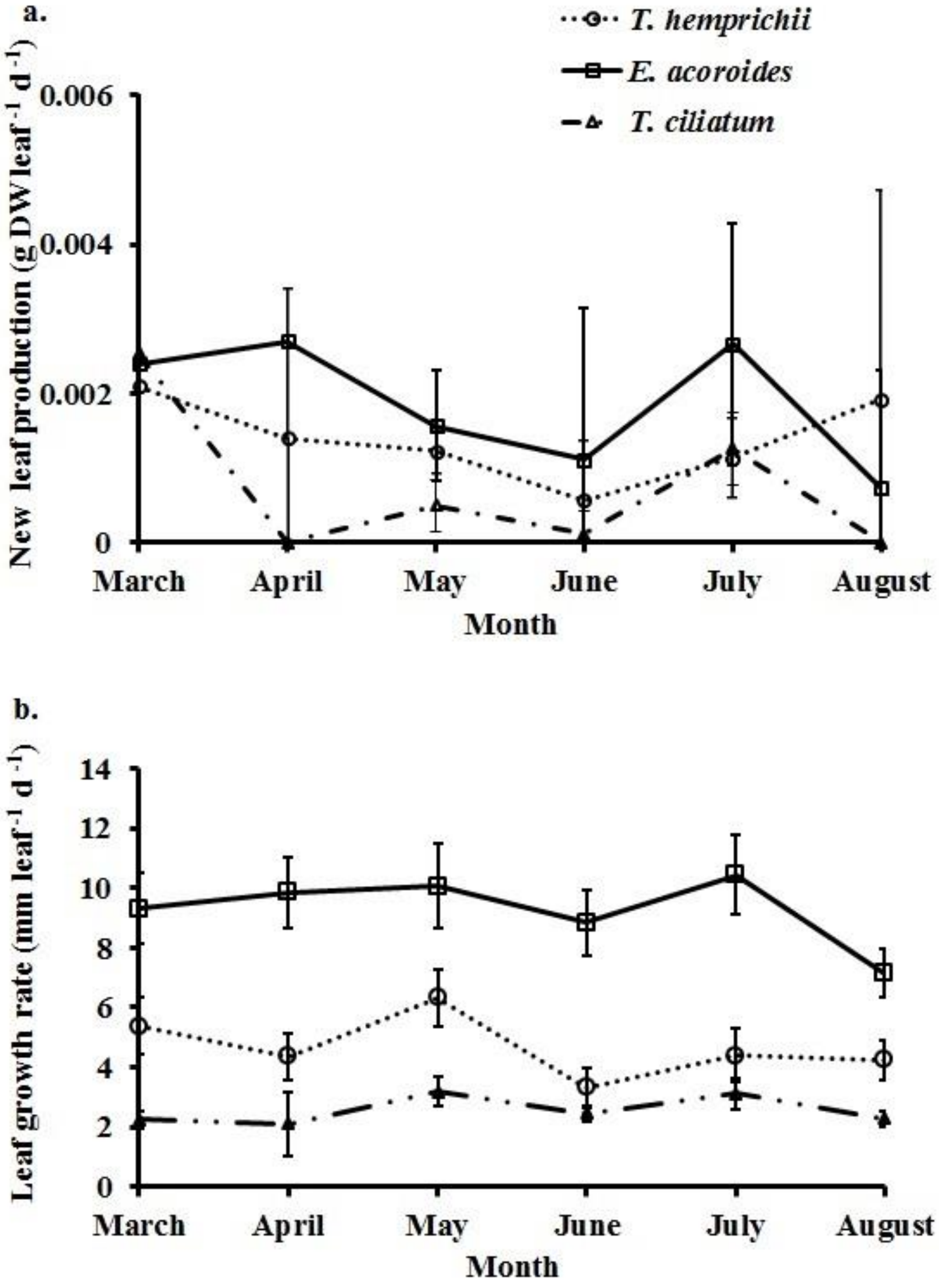

Fig. 27: Leaf productivity (Mean 95% C.I) for three dominant seagrasses of Gazi Bay

5.3.2 Shoot regrowth in experimental plots

The biomass increase (mean± 95% C.I) for the re-growth of *E. acoroides* was 0.52 ± 0.07 g DW m^{-2} d^{-1} fluctuating between 0.40 - 0.74 g DW m^{-2} d^{-1} over the first 15 months of the monitoring period, and finally dropping to a mean of 0.07 ± 0.04 g DW m^{-2} d^{-1} when only a few shoots regrew following the final monthly clearance. The biomass production for the regrowth of *T. hemprichii* was relatively stable; it withstood clearance for 8 months but eventually failed to regrow (Fig 28). The shoot biomass production (mean ± 95 % C.I) for *E. acoroides* and *T. hemprichii* over the 16 month monitoring period was 0.35 ± 0.12 and 0.11 ± 0.05 g DW m^{-2} d^{-1} respectively.

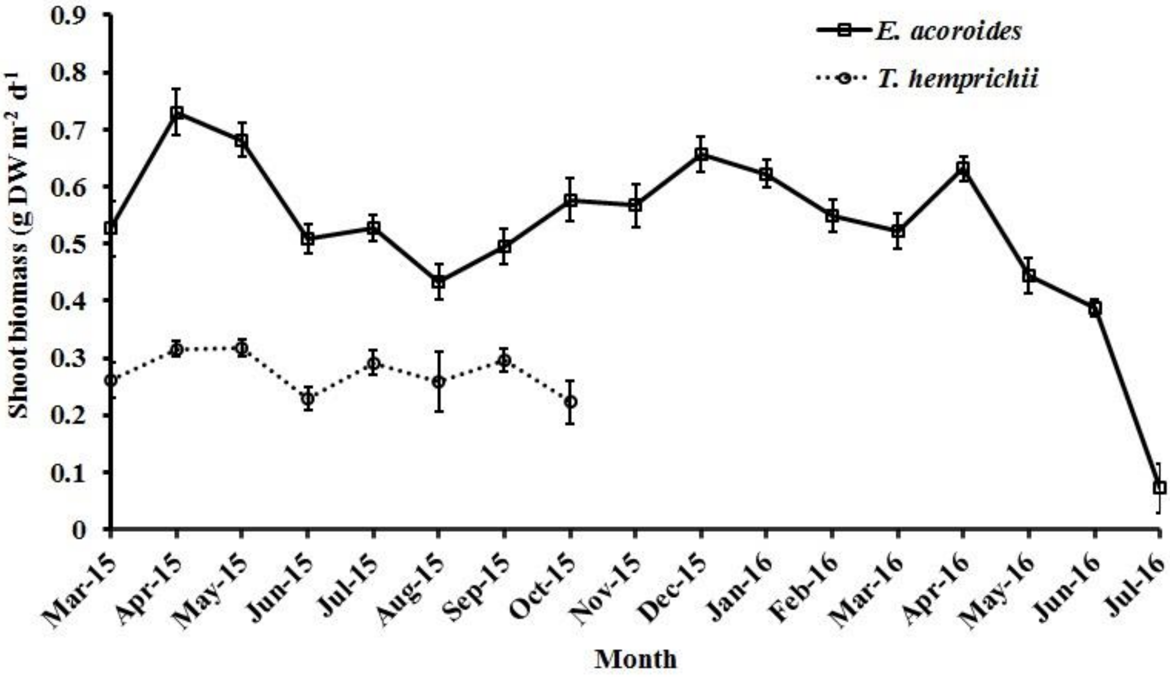

Fig. 28: Biomass accumulation (mean± 95% C.I) for shoot regrowth following monthly harvesting for two species

Six months from the start of the experiment and following the regular clearance of the above-ground shoots, *Halophila ovalis* was observed to colonise the seagrass harvested sites for the *T. hemprichii*. Biomass accumulation (mean ± 95% C.I) was

0.16 ± 0.06 g DW m^{-2}. Their regrowth persisted though with fluctuations over the remainder of the monitoring period (Fig 29).

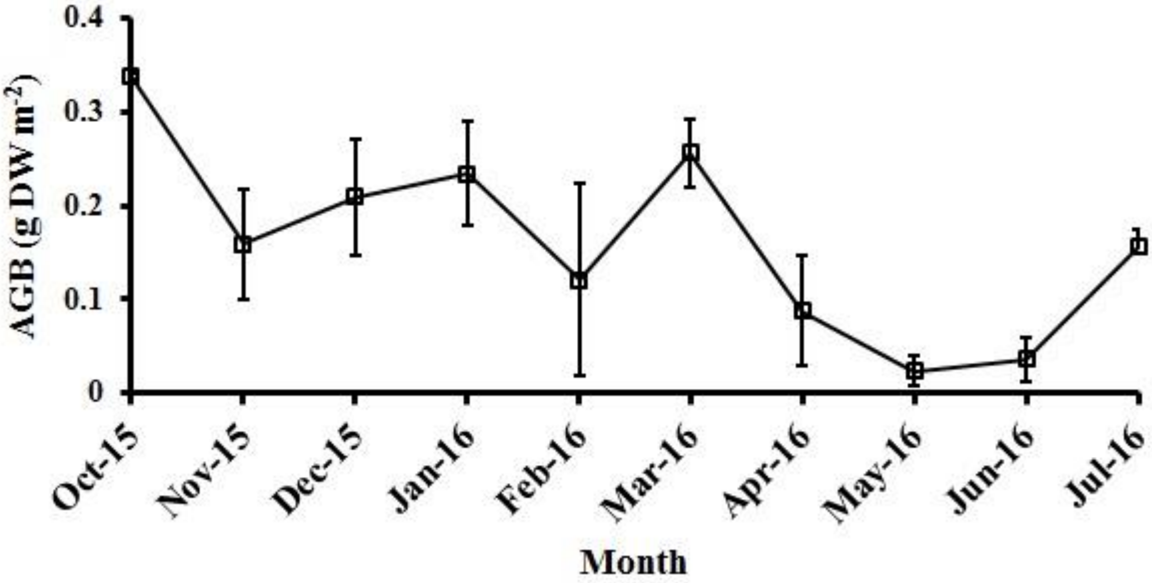

Fig. 29: Shoot biomass accumulation (Mean ± 95% C.I) for *H. ovalis* after colonising the *T. hemprichii* shoots harvested areas

5.3.3 Below-ground biomass accumulation for the in-growth trenches

Comparisons of the biomass accumulation for all the species indicated that *T. hemprichii* accumulated the highest biomass at 197.4 ± 108.7 g DW m^{-2} yr^{-1} while the lowest biomass accumulation was in *E. acoroides* at 12.8 ± 6.8 g DWm^{-2} (Fig 30). The mean below-ground biomass between the four species was statistically significantly different ($F_{(3, 8)} = 5.83$, $p = 0.021$).

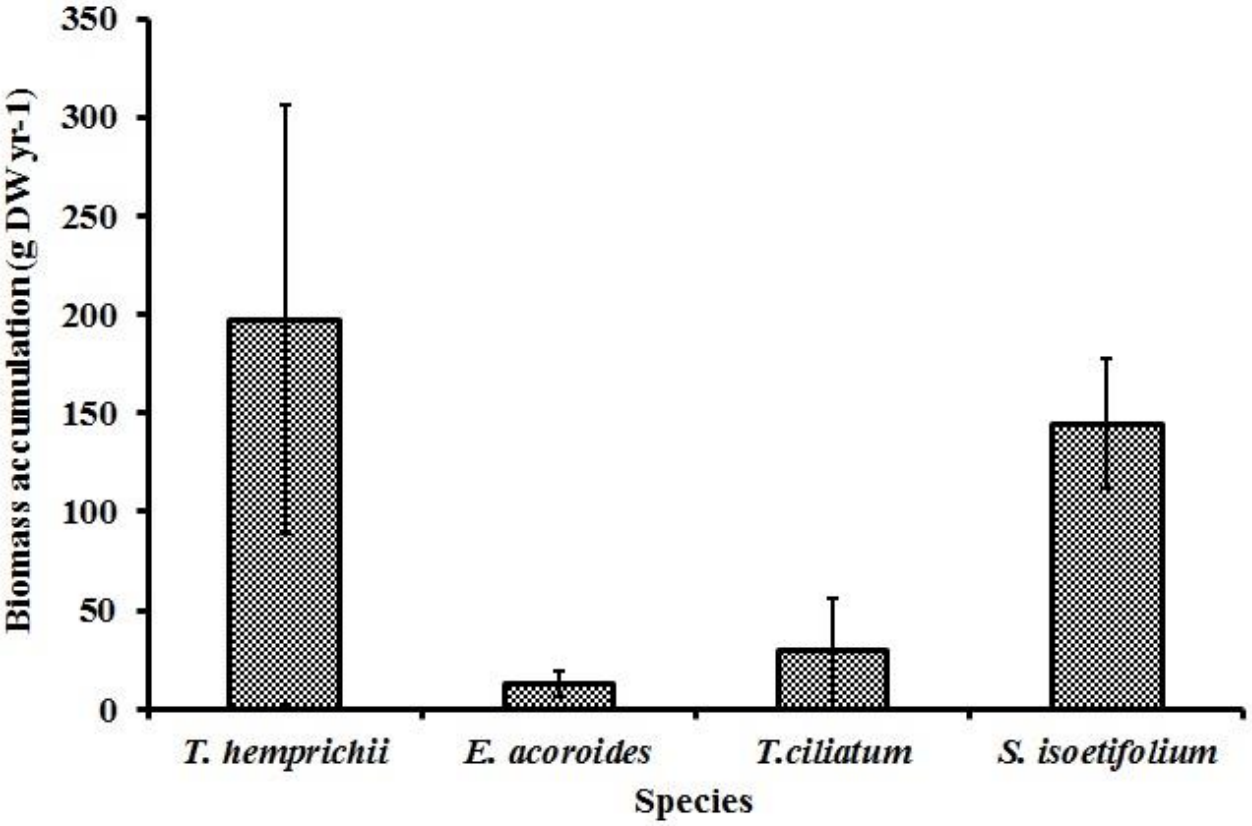

Fig. 30: Below-ground biomass (Mean ± 95%) accumulation for the in-growth trenches of four dominant species of Gazi Bay

5.4 Discussion

Seagrass growth rates vary across habitats and between species. This study investigated the growth rates of the seagrasses through a variety of methods. The above-ground productivity for most of the parameters monitored through the leaf plastochrone method observed that growth rates varied throughout the six month period with notable drops during the month of June. Though the increase in the number of shoots in *E. acoroides* was relatively low compared to both *T. ciliatum* and *T. hemprichii,* this species showed higher production rates which may be attributed to its size. The mean shoot production was highest in *E. acoroides* at 0.037 ± 0.002 g DW shoot^{-1} d^{-1}, followed by that of *T. ciliatum* at 0.006 ± 0.004 g DW shoot^{-1} d^{-1} with *T. hemprichii* having the lowest at 0.005 ± 0.001 g DW shoot^{-1} d^{-1}. However, when these values are extrapolated using the shoot density values, it revealed that *E. acoroides* had the highest annual shoot productivity at 1944 g DW m^{-2} yr^{-1} followed by *T. hemprichii* at 730 g DW m^{-2} while *T. ciliatum* recorded the least at 438 g DW m^{-2} with an average of 1037 g DW m^{-2} for the three species.

Above-ground production through the shoot regrowth in the seagrass removal experiment observed that shoot regrowth persisted for some months, but differed between the two species with *T. hemprichii* dying at the eighth month while *E. acoroides* persisted for a longer period and eventually died after the sixteenth month. The fact that *E. acoroides* persisted for a longer period following the disturbance and had a higher shoot regrowth production of 0.35 ± 0.12 g DW m^{-2} d^{-1} compared to the combined shoot production of *T. hemprichii* and *H. ovalis* of 0.27 ± 0.05 g DW m^{-2} d^{-1} in the *T. hemprichii* monitoring areas suggests that the storage of food reserves in below-ground tissues is important for the survival of the plants and that larger species have greater below-ground reserves stored in the roots and rhizomes which are remobilised for supporting biological processes in the absence of photosynthesis. The establishment of *H. ovalis* in the disturbed stand of *T. hemprichii* was not anticipated at the start of the monitoring activity but indicates a form of succession of seagrass disturbed areas. The fact that the *H. ovalis* persisted despite the regular clearance while *T. hemprichii* died after eight months is an indication of differences in the resilience among the species. Past studies have indicated that *H. ovalis* has high tolerance to irradiance which could be the reason for its ability to survive in the seagrass disturbed areas where the canopy has been removed (Beer et al., 2006). The growth characteristics for the four dominant species of the bay, monitored in this study through the leaf plastochrone method , are comparable to those reported from other studies in the region (Kamermans et al., 2001; Lyimo et al., 2006; Uku & Björk, 2005). The observed range of new leaf production rates in this study of 0.0006 – 0.002 g DW shoot^{-1} d^{-1} for *T. hemprichii* do not show much deviation from the values reported using the leaf marking technique from an earlier study in the country (Uku & Björk, 2005). Leaf production for this species at Nyali was 0.005 ± 0.005 and 0.006 ± 0.003 (g DW shoot^{-1} d^{-1}) during the South East and the North East monsoon winds respectively while in Vipingo the production was 0.005 ± 0.002 and 0.004 ± 0.001 g DW shoot^{-1} d^{-1} during the South East and the North East monsoon winds respectively (Uku & Björk, 2005). Another study in the region at Chwaka Bay, Zanzibar, observed that daily leaf production differed between sites and species with a maximum of 0.01g DW shoot^{-1} d^{-1} for *T hemprichii* (Lyimo et al., 2006). Earlier on, Kamermans et al., (2001), had reported that the average leaf production rates for *T. ciliatum* along different sites of the East African coast, were between 1.4 and 3.0 g DW shoot^{-1} yr^{-1} which translates to 0.004 to 0.008 g DW shoot^{-1} d^{-1}. Bandeira, (2002) recorded leaf production estimates of 0.0043 to 0.0106 g DW shoot^{-1} d^{-1} for *T. ciliatum* beds in rocky and sandy substrates in Mozambique (Ochieng, 1995).

The shoot growth rate of 4.3 mm day^{-1} for *T. ciliatum* recorded in this study is low when compared to the 20.7 mm day^{-1} recorded in an earlier study at the bay for the same species (Ochieng, 1995). Similar studies in the bay for *T. ciliatum* reported a production rate of 7.5 g DW m^{-2} day^{-1} while the annual production ranged between 710 and 1365 g DW m^{-2} yr^{-1} (Hemminga et al., 1995). However, this is likely to vary with seasons. The relatively high areal productivity of the intertidal seagrass during the S.E. monsoon (April-May) can be attributed to the relatively elevated nitrogen concentrations in the bay (Hemminga et al., 1995).

Comparison of the productivity for the seagrasses from the in-growth trenches is difficult and could be the reason for this approach having not been tried elsewhere. Nevertheless, it gives very important observations as results indicate that the complete removal of seagrasses causes disturbance and probably "shock" to the plants and as a result they take considerable time to recover and re-establish in disturbed areas. The below-ground biomass accumulation from the in-growth trenches of this study is comparable to that of other studies that have used the cores. This study recorded the highest below-ground biomass accumulation of 197.4 ± 108.7 g DW m^{-2} yr^{-1} by *T. hemprichii.* In a study of growth and production of *C. nodosa* in a coastal lagoon of the Mediterranean seagrasses, Agostini et al. (2003) recorded a below ground biomass accumulation of 412 g DW m^{-2} yr^{-1}. It is argued that the below-ground production is generally underestimated in the literature because the estimates available often exclude the root production, which accounts for 15 - 50% of total production on average (Duarte et al., 1998). The mean annual production from a global data set of seagrasses of 816 - 1012 g DW m^{-2} yr^{-1} (Duarte & Chiscano 1999) and the 913 g DW m^{-2} yr^{-1} arrived at in this study from a review of seagrass productivity data from African sites. The low values for the below-ground productivity recorded in this study could be as a result of the impact of disturbance and the complete removal of the grasses from the ingrowth trenches. This is supported by the fact that in some of the in-growth trenches, the below-ground biomass was absent even after one year period. These findings attests to the variability in growth forms and the relative differences in responses to perturbations with some seagrasses taking considerable time to re-establish following disturbance. It takes time for the roots and rhizomes of the surrounding plants to extend to the seagrass disturbed areas and upon establishment, shoots are able to sprout from the rhizomes forming the above-ground component. However, there is need for further research. The above-ground compartment may not be a long term investment owing to the high turn-over rates occasioned by grazing and defoliation by natural and human induced perturbations. However, it exhibits the rigorous growth which is an adaptation of replacing the tissues that are lost. These findings on interspecies variability in productivity rates and succession in seagrass degraded areas have useful practical applications in seagrass conservation as well as in the restoration of degraded areas. The fact that *H. ovalis* easily establishes in seagrass disturbed areas naturally imply that the species could be useful for human intervention in restoring seagrass degraded areas. Equally, the low resilience observed in *T. hemprichii* and the relatively low growth rates observed in *T. ciliatum* indicate why the species need better protection if their biodiversity and ecosystem functions are to be sustained.

Chapter six

6.0 General discussion and conclusion

The central focus of this thesis was to elucidate the role of the seagrasses as carbon sinks. This study undertook a comprehensive review of the current knowledge of biomass and productivity of seagrasses in Africa whereby only 32 published papers and 8 theses/reports were encountered from Africa, majority having been done along the E. African coast. The study found no published reports of sedimentary organic carbon from seagrasses in Africa. This suggests a major gap in knowledge on seagrass carbon in Africa despite its coastline having among the highest seagrass species diversity. These results compliments earlier findings that underscored the geographically uneven knowledge of seagrass carbon across the globe (Duarte and Chiscano 1999; Fourqurean et al. 2012). A more explicit finding from this study was that seagrass studies in many regions across the globe have ignored the sediment C_{org}, the most important part of the putative ‘blue carbon’ sink provided by seagrasses, which in itself is a major gap in seagrass blue carbon work. Uncertainties on both the quantities of seagrass C_{org} across the pools and the relative contribution of African seagrasses to the carbon budgets have several implications. Firstly, the role of seagrasses in national and global carbon budgets is likely to be underestimated. Secondly, the value of seagrass habitats is likely to be underappreciated and as such seagrass habitats may be ignored while formulating legislation and policies for the management and conservation of marine ecosystems, leading to further degradation and ultimately increased carbon emissions and loss of other ecosystem services that seagrass beds provide.

This study was the first to compare sediment C_{org} for vegetated seagrass and the un-vegetated areas at an African site and demonstrated an exceptionally powerful effect of seagrass on C sequestration, with relative C densities being much higher in areas under seagrass compared with the respective un-vegetated areas. The sediment C_{org} of the seagrasses of the bay ranged between 160.7- 233.8 Mg C ha^{-1}. The combined mean biomass and sediment C_{org} values that were arrived at in this study, of 240.9±24.2 Mg C ha^{-1}, when extrapolated translate to an estimate of 168,642 Mg C for carbon stocks in the top one metre of the seagrass meadows of Gazi Bay. As a comparison, the ~600 ha of mangrove forest at the site are estimated to hold around 254,725 – 452,725 tonnes of combined biomass and the top one metre of the sediment C_{org} (Cohen, 2014; Gress, Huxham, Kairo, Mugi, & Briers, 2016; Tamooh et al., 2008). This means that mangroves could hold about twice the C_{org} ha^{-1} of the seagrasses. The findings that seagrasses hold significant amount of C_{org} reinforce our understanding of the value of conserving the seagrass beds since intact meadows have the capacity to continue sequestering more carbon from the atmosphere and at the same time, their conservation prevents the remineralization of the stored carbon as well as the continued provision of other ecosystem services.

Like other seagrass areas across the globe, the seagrass beds of the bay have been facing degradation overtime occasioned by both anthropogenic induced and natural factors. The seagrass removal experimental study (Chapter 4) simulated the impacts of small scale seagrass loss/disturbance on sediment dynamics and on the faunal communities in the area. The experiment recorded an average surface elevation rate of 7.6 ± 0.5mm yr^{-1} (mean ± C.I) and erosion rate of -15.8 ± 0.4mm yr^{-1}. At the same time, the experiment showed a significant difference in the carbon density and faunal communities between the seagrass harvested and un-harvested areas. Although outside the scope of the present study, observation of seagrass degradation occasioned by overgrazing by sea-urchins was made. However, of concern to the degradation of the seagrasses and the adjacent mangroves in the southern part of the bay close to the mouth of the Mkurumuji River is the sedimentation that has been observed in the last two years and anecdotal evidence from the local community are that the huge sediment loads are associated with sand mining and poor agricultural activities in the neighbourhood. The gradual sedimentation in the area sends a grim picture on the survival of these blue carbon ecosystems as well as the shoreline stability. These observations confirm the existing knowledge that in the tropics, sediment loading and suspended sediments, associated with human activities upstream cause turbidity, and may have the greatest impact on seagrasses (Duarte et al., 2009; Terrados et al., 1998). At the same time, the increased overgrazing of seagrasses by sea urchins that is being observed in the bay may be triggered by reduced predation of the urchins as a result of overfishing which in severe cases can lead to a big loss of seagrass meadows (Eklöf, 2008). Considering that at present, the blue carbon ecosystems are acknowledged as hot spots for carbon burial in the ocean, where they play a globally significant role in climate change mitigation and adaptation (Duarte et al., 2005; Mcleod et al., 2011), the continued degradation or loss of these ecosystems imply more carbon emissions that further worsen global climate change.

Chapter 5 of this study involved the estimation of productivity of the four dominant seagrasses of Gazi Bay, investigated through a combination of field survey and an experiment. In the survey, shoot productivity was investigated through the leaf plastochrone method while ingrowth trenches were used to estimate the below-ground productivity. In the experimental based approach, shoot regrowth characteristics and biomass increment were monitored monthly following seagrass harvesting. The findings were that seagrass productivity varies across species with larger species accumulating more biomass than the smaller species. The study also observed that disturbed seagrass areas take time to recover and may be vulnerable to other species impacting the productivity patterns. The productivity measurements in this study encountered limited data for comparative purposes especially for some species. Only *T. ciliatum* appear to have been given more focus which may be attributed to its relatively higher dominance in the bay. Similar findings of limited information on productivity have been reported elsewhere in this study. Much of the productivity measurements are for the above-ground parameters ignoring the below-ground component which is more of a longer term investments. Differences in productivity measurements by various methods for the same species and parameters are an indication of the need for harmonized protocols on global carbon models. Otherwise without them, regional or site variations may be inadvertently reported.

6.1 Seagrass management and conservation in Kenya and environmental policies

Seagrass meadows are an underappreciated resource. They are often faced with a myriad of challenges which have contributed to their degradation. The seagrasses are widespread along the Kenya's over 600km coastline and within the internationally recognised territorial waters offshore in the West Indian Ocean. Responsibility to manage seagrasses and associated biodiversity in Kenya is vested with Kenya Wildlife Service either singly when they occur within the marine protected areas (MPA), or jointly with State Department of Fisheries and other agencies when it occurs outside MPA's. Communities can apply for tenureship to manage designated marine areas with seagrasses through Beach Management Units (BMU). A total of 24 locally managed marine areas LMMAs had been established by the end of the year 2015 (Kawaka et al., 2016). Kenya is a signatory to a number of multilateral treaties and agreements that advocate the establishment of Marine Protected Areas (MPAs) as a tool for the fisheries regulation and conservation of biodiversity. Among them are: the Environmental Management and Co-ordination Act (EMCA 1999) which seeks to improve the quality of life of local communities through sustainable development approaches while safeguarding and maintaining the environmental and associated resources; the United Nations Convention on Biodiversity (CBD) that guides the identification of Ecologically and Biologically Significant Areas (EBSAs) in various bio-regions of the world; and the United Nations Convention on Climate Change (UNFCCC), the International Law of the Sea (ILS) and the Jakarta Mandate of 1995 which outlines the program of action for marine and coastal biodiversity within the CBD. These multilateral agreements recognise the unique biodiversity as well as the numerous goods and services offered by marine and coastal ecosystems. To date, good progress has been made in the establishment and implementation of management programs for the MPAs in Kenya with a total of nine MPAs having been established along the coast (MPRUT-KWS, 2016; Tuda & Omar, 2012). Such initiatives not only contribute to the conservation of the biodiversity but are increasingly becoming recognised as major contributors to a growing blue economy and leading to the economic empowerment of the people in line with the national development blue print, vision 2030. The seagrass meadows of Gazi Bay fall under Diani-Chale Marine National Reserve which was established in 1995 by the Kenya government with an spatial area of 75km^2. It also forms the northern boundary of the proposed Transboundary Marine Conservation Area between Kenya and Tanzania (TBCA). As at present, there is no enforcement of its protection status due to weak governance structures. The Integrated Coastal Zone Management (ICZM) Policy is likely to provide positive impacts to the conservation efforts (GoK, 2014). Apart from conforming to requirements of these policies, a more promising strategy is to develop a comprehensive focus on the issues affecting these critical ecosystems by building linkages and synergies amongst public and private sectors, local communities, experts and general stakeholders.

6.2 Application of the study to the conservation and management of the seagrass meadows of Gazi Bay

This study provides useful information that can guide the formulating of policies and strategies for effective management of the marine resources. Better management of the seagrasses and associated ecosystems is important for the livelihood improvement of the local people. The Gazi Bay marine ecosystem supports a large population with rapid growth rates. The population of the two villages of Gazi Bay (Gazi and Makongeni), were 2000 and 700 respectively according to the 2009 Kenya population census and was projected to reach 5000 people in five years' time (GoK, 2010). The bay also supports a wider population of Kwale County. A large proportion of the local population is predominantly artisanal fishermen relying on fish caught in the mangrove/sea grass/coral reef system of the bay while further away are peasant farmers relying on small scale subsistence farming activities. Considering the high poverty levels in the area and the population growth rate, there is need for tapping more wealth from these ecosystems through climate adaptation projects in order to improve their livelihoods and at the same time helping to conserve the environment. The government has formulated a conservation strategy for seagrass and coral ecosystems (KWS, 2013). The strategy discusses threats facing corals and seagrass ecosystems, and proposes appropriate action plans such as reviewing the existing legislation and policies as well as increasing research, monitoring and information management with the aim of enhancing their conservation. With appropriate knowledge, the marine ecosystems have the potential to support the livelihoods of coastal communities while at the same time ensuring environmental sustainability under the Payment for Ecosystem Service (PES) schemes. However, for the PES to succeed, it must be underpinned with robust research, clear property rights and equitable sharing of benefits. There is considerable interest on the use of the PES scheme to restore and protect blue carbon ecosystems in line with Kyoto Protocol. Sequestered carbon could be traded in both compliant and voluntary market. Additionally, specific methodologies for estimating carbon stocks and sequestration potentials have been developed for coastal wetlands (Crooks et al., 2011; Howard et al., 2014). Already, a demonstrative blue carbon project involving mangroves has successfully been implemented at Gazi Bay. Named, Mikoko Pamoja, this is the first community type project of its kind to benefits from sale of mangrove carbon credits (http://www.planvivo.org/project-network/mikoko-pamoja-kenya). The project earns close to $13,000 per year through offsetting about 2125 tonnes of CO_2 equivalent per annum. Similar to mangroves, seagrass ecosystems are important carbon sinks and this study provides carbon storage estimates for seagrasses thus adding to the published information on mangroves and carbon storage in the bay. Due to the mangrove and seagrass nexus, knowledge of the carbon stocks associated with seagrasses in the bay may open opportunities for bundling seagrass ecosystem services with those of the mangrove ecosystem, which can then be traded under PES. This approach makes ecological sense, given the strong connections between the two ecosystems, and could provide useful buffering against environmental shocks that affect one part of the ecosystem more than others.

6.3 Recommendations for further research

The present study established that most of the available information on seagrass carbon was from areas where research has been funded or where there has been effective collaboration between institutions. The findings of this study underscore the importance of strengthening collaboration between institutions and allocating more funds for research. A more integrated research on marine ecosystems using approved global carbon models is likely to provide better understanding of the ecosystems connectivity and the resources that can be bundled to provide better livelihoods options.

The study found a limited number of papers on seagrass carbon in Africa, which apparently demonstrates a paucity of information on the contribution of African seagrasses to the global carbon budget. These findings compliments similar findings from other reviews (Duarte and Chiscano 1999, Fourqurean et al., 2012). The fact that some of the seagrass studies in Africa have ignored important carbon pools such as the sediment C_{org}, reveal the uncertainties and a major gap in seagrass blue carbon work. Future research work on seagrass carbon in Africa should be broad based targeting a wide range of sites, species and should aim at understanding the quantities of sediment C_{org} which has been found to be neglected despite its contribution to the total carbon stocks.

Despite the fact that there is a large and growing literature exploring sediment carbon in seagrass meadows, the current work found few explicit comparisons of seagrass carbon between vegetated seagrass areas and neighbouring un-vegetated areas. Though the study determined C_{org} storage capacity for the seagrasses of Gazi Bay, it is difficult to accurately estimate the carbon burial rates without reliable information on stable carbon isotopes. The few studies carried out in the bay (Bouillon et al., 2007; Hemminga, 1996) have either focused on one seagrass species or carbon export between the mangrove ecosystem and the seagrasses. These inadequacies suggest the need for further research in order to improve our understanding of the value of management and conservation of seagrass meadows.

There are uncertainties on the seagrass coverage across sites in Kenya as seagrass mapping in the past (Dahdouh et al., 1999), has been limited to few sites. With these uncertainties, it is difficult to arrive at precise estimates of the carbon stocks and the relative contribution to the carbon budget. This study has identified several areas of interests but with insufficient information and therefore recommends the following research initiatives:

- There is a critical need to determine the seagrass coverage and species composition in Kenya by combining ground truthing with more advanced and accurate methods such as remote sensing and aerial photography followed by digital imagery analysis using most recent software programmes such as Matlab.
- The C_{org} estimates in this study concentrated on the top one metre of the sediment. Since carbon density across species could vary with depth, future research should focus on the deeper sediment in order to arrive at more precise estimates of the C_{org}.
- The seagrass removal experiment in this study revealed that the degradation of seagrasses has impacts on their biodiversity and ecosystem functions by reducing the number and fauna diversity, reducing the sediment trapping and carbon input. Due to the limited spatial and temporal cover of the study, there is need for comprehensive studies on the causes, the level and impacts of seagrasses degradation in the bay.
- Despite the observations made on seagrass degradation associated with the increased grazing due to overfishing of predators for seagrass herbivores, plus the sediment loads, no quantification of the impacts of these activities on the seagrasses has been made. There is need for research to determine the level of degradation and advice on management and conservation. Globally, similar activities may be occurring and thus there is need to accurately document the status of seagrass threats and accord them proper protection.
- The seagrass removal experiment observed varying resilience rates between the species monitored and some attempts of establishment of other species

following the disturbance. There is need for a more comprehensive research on seagrass resilience following disturbance and identification of suitable species for restoring degraded areas.

- The study on carbon storage of the seagrasses of the bay concentrated on monospecific stands. It is of intrinsic interests to study the carbon storage in multispecific or in a macroalgae heavily invested stands for more precise knowledge of carbon storage of the seagrass meadows.

6.4 Recommendations for management interventions

Seagrass meadows are connected to other marine habitats and thus play a critical role in supporting life to the multitude of organisms within the global marine ecosystems. Sustainability of this nexus require sound management and conservation measures that are driven by innovative approaches to enhance their resilience. However, at an ecosystem level, there appears to be significant misalignment between conservation policies on biodiversity, ecosystem services, and the role of seagrass meadows (Unsworth & Cullen, 2010). The importance of the seagrass ecosystems as a service provider remains marginalized within conservation agendas, principally because marine conservation priorities do not recognize the full socio-economic and ecological value of goods and services that seagrasses provide. The links between seagrass and food security are yet to be understood across scales, although correlations between a health integrity of seagrass beds and fisheries productivity are emerging which may support calls for renewed management initiatives that recognize the need for seagrass conservation (Unsworth & Cullen, 2010). Current recognition of seagrasses within climate mitigation strategies as significant carbon sinks (Duarte et al., 2013) may enhance their conservation and thus add to their global value as a source of food security in a rapidly changing global environment. As such, concern for the conservation of the seagrass meadows is essential for the development of a comprehensive approach to integrated marine and coastal management.

In order to ensure resilience of the seagrasses and to maintain a health integrity and productive seagrass ecosystem, the following management interventions are worthwhile:

- Human-induced factors are major causes of degradation of seagrasses either directly through dredging, damage through the seine nets or indirectly overfishing as well as through sediment loads associated with activities on adjacent land. Though not well documented across many sites globally, there is need to monitor their impacts and develop strategies for arresting them.
- Since the establishment and operationalization of Marine Protected Areas and the Community Conservation Areas have been found to contribute significantly to the conservation of seagrasses and other marine resources in many parts of the world, (Kawaka et al., 2016), there is need to increase the number and size of these areas and to enforce their operation by providing the necessary infrastructure.
- Better understanding of the role of marine ecosystem in the biodiversity and ecosystem functions is central to successful conservation efforts. It is therefore important to communicate the research findings to the end consumers who are mainly the communities that live adjacent to these ecosystems and on which their livelihoods are dependent and whose activities have direct impacts on these ecosystems. There is also a need to create more awareness to the local people on the importance of conserving these habitats for the purpose of environmental benefits and for sustainable income and food security.
- A better management option entails an interdisciplinary approach within the local, regional and international research agenda to provide evidence in

support of a new marine management regime. Not only is it important for conservation managers to understand how habitat variability and fragmentation impact biodiversity, but also on the resources within the habitat that provide support to the biodiversity.

- Though trade-offs between livelihood needs and ecosystem management are sometimes unavoidable, there is a need to apply strategies that can safeguard the overexploitation of the resources by harnessing synergies across a wide spectrum that will help achieve livelihood options and ecosystem management. Such synergies include biodiversity conservation, climate regulation and maintenance of water quality and improvement on quality of people's lives. The provision of alternative livelihood sources can help minimize problems of resource overexploitation. However, in order to achieve better management outputs, there is a need to align management objectives with people's needs.
- Better conservation and management actions can also be strengthened by further developing our understanding of the complex socio-cultural linkages of the communities to these ecosystems as well as deepening their understanding of the ecological value to their survival and for posterity.

References

Adams, J. B., & Talbot, M. M. B. (1992). The influence of river impoundment on the estuarine seagrass *Zostera capensis* Setchell. *Botanica Marina*, *35*(1), 69–76.

Agawin, N. S. R., Duarte, C. M., & Fortes, M. D. (1996). Nutrient limitation of Philippine seagrasses (Cape Bolinao, NW Philippines): In situ experimental evidence. *Marine Ecology Progress Series*, *138*(1–3), 233–243. http://doi.org/10.3354/meps138233.

Agostini, S., Pergent, G., & Marchand, B. (2003). Growth and primary production of *Cymodocea nodosa* in a coastal lagoon. *Aquatic Botany*, *76*, 185–193. http://doi.org/10.1016/S0304-3770(03)00049-4.

Alcoverro, T., & Mariani, S. (2002). Effects of sea urchin grazing on seagrass (*Thalassodendron ciliatum*) beds of a Kenyan lagoon. *Marine Ecology Progress Series*, *226*, 255–263.

Aleem, A. A. (1984). Distribution and ecology of seagrass communities in the West Indian Ocean. *Deep Sea Research*, *31*, 919–933.

Bandeira, S. O., (2000). *Diversity and Ecology of Seagrasses in Mozambique: Emphasis on Thalassodendron ciliatum:* Structure,*Dynamics, Nutrients and Genetic Variability*. (PhD. Thesis). Göteborg University, Göteborg, Sweden.

Bandeira, S. O. (1997). Dynamics, biomass and total rhizome length of the seagrass *Thalassodendron ciliatum* at Inhaca Island, Mozambique. *Plant Ecology*, *130*(2), 133–141. http://doi.org/10.1023/A:1009724308545.

Bandeira, S. O. (2002a). Diversity and distribution of seagrasses around Inhaca Island, southern Mozambique. *South African Journal of Botany*, *68*, 191–198.

Bandeira, S. O. (2002b). Leaf production rates of *Thalassodendron ciliatum* from rocky and sandy habitats. *Aquatic Botany*, *72*(1), 13–24 http://doi.org/10.1016/S0304-3770(01)00192-9.

Barbara, Ondiviela, Losada, J., Lara, L., J.,van Belzen, J. (2014). The role of seagrasses in coastal protection in a changing climate. *Coastal Engineering, 87,* 158–168. http://doi.org/10.1016/j.coastaleng.2013.11.005.

Barrón, C., Marbà, N., Terrados, J., Kennedy, H., & Duarte, C. M. (2004). Community metabolism and carbon budgets along a gradient of seagrass (*Cymodocea nodosa*) colonization. *Limnology and Oceanography*, *49*(5), 1642–1651. http://doi.org/10.4319/lo.2004.49.5.1642.

Beckschäfer, P., Mundhenk, P., Kleinn, C., Ji, Y., Yu, D. W., & Harrison, R. D. (2013). Enhanced Structural Complexity Index : An Improved Index for Describing Forest Structural Complexity, *3*(1), 23–29.

Beer, S., Mtolera, M., Lyimo, T., & Björk, M. (2006). The photosynthetic performance of the tropical seagrass *Halophila ovalis* in the upper intertidal. *Aquatic Botany*, *84*(4), 367–371. http://doi.org/10.1016/j.aquabot.2005.11.007.

Berkström, C. (2012). *Ecological connectivity in East African seascapes*. (Master's Thesis, Department of Zoology), Stockholm University, Stockholm, Sweden. pp 15-29.

Bos, A. R., Bouma, T. J., de Kort, G. L. J., & van Katwijk, M. (2007). Ecosystem engineering by annual intertidal seagrass beds: Sediment accretion and modification. *Estuarine, Coastal and Shelf Science, 74*(1–2), 344–348. http://doi.org/10.1016/j.ecss.2007.04.006.

Boström, C., Jackson, E. L., & Simenstad, C. A. (2006). Seagrass landscapes and their effects on associated fauna: A review. *Estuarine, Coastal and Shelf Science, 68*(3–4), 383–403. http://doi.org/10.1016/j.ecss.2006.01.026.

Bouillon, S., Dehairs, F., Velimirov, B., Abril, G., & Borges, A. (2007). Dynamics of organic and inorganic carbon across contiguous mangrove and seagrass systems (Gazi Bay, Kenya). *Journal of Geophysical Research*, *112*(G2), G02018. http://doi.org/10.1029/2006JG000325.

Bouillon, S., Moens, T., & Dehairs, F. (2004). Carbon sources supporting benthic mineralization in mangrove and adjacent seagrass sediments (Gazi Bay, Kenya). *Biogeosciences*, *1*(1), 71–78. http://doi.org/10.5194/bg-1-71-2004.

Bououarour, O., Kamcha EL, R., Boutahar, L., Tnoumi, A., Bendahhou, Z., Benhoussa, A., & H., B. (2015). Spatial patterns of the *Zostera noltei* meadows across the Atlantic coast of Morocco: Is there a latitudinal gradient? In Guala., & Fais., (Ed.). *Mediterranean Seagrass Workshop book of abstract* (p. 81).- Sardinia, Oristano, Italy, Fondazione Banco di Sardegna.

Boutahar, L., Ramzi, Y. S., Atef, O., EL, K. R., Benhoussa, A., Ostale, E., Bazairi, H. (2015). First data on biomass and abundance of two eelgrass (*Zostera marina*) meadows south to the strait of Gilbraltar (Jbel Moussa, Morocco). In Guala I & Fais M., *Mediterranean Seagrass Workshop book of abstract* (P 109). Sardinia, Oristano, Italy Fondazione Banco di Sardegma.

Browne, C. M., Maneveldt, G. W., Bolton, J. J., & Anderson, R. J. (2013). Abundance and species composition of non-geniculate coralline red algae epiphytic on the South African populations of the rocky shore seagrass *Thalassodendron leptocaule, South Africa Journal of Botany*, 86, 101–110.

Burrows, K., & Kinney, P. L. (2016). Exploring the Climate Change, Migration and Conflict Nexus. *International Journal of Environmental Research and Public Health*, *13*(443), 1–17. http://doi.org/10.3390/ijerph13040443.

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. http://doi.org/10.1016/j.ecss.2008.04.021.

Cahoon, D. R. (2014). Estimating Relative Sea-Level Rise and Submergence Potential at a Coastal Wetland. *Estuaries and Coasts*, http://doi.org/10.1007/s12237-014-9872-8.

Cahoon, D. R., White, D. a., & Lynch, J. C. (2011). Sediment infilling and wetland formation dynamics in an active crevasse splay of the Mississippi River delta. *Geomorphology*, *131*(3–4), 57–68 http://doi.org/10.1016/j.geomorph.2010.12.002.

Campbell, J. E., Lacey, E. A., Decker, R. A., Crooks, S., & Fourqurean, J. W. (2014). Carbon Storage in Seagrass Beds of Abu Dhabi, United Arab Emirates. *Estuaries and Coasts, 160*, 1465–1475. http://doi.org/10.1007/s12237-014-9802-9.

Carr, J., Odorico, P. D., Mcglathery, K., & Wiberg, P. (2010). Stability and bistability of seagrass ecosystems in shallow coastal lagoons : Role of feedbacks with sediment resuspension and light attenuation, *Journal of Geophysical Research*, *115*, 1–14. http://doi.org/10.1029/2009JG001103.

Cheong, S. M., Silliman, B., Wong, P. P., van Wesenbeeck, B., Kim, C.-K., & Guannel, G. (2013). Coastal adaptation with ecological engineering. *Nature Climate Change*, *3*(9), 787–791. http://doi.org/10.1038/nclimate1854.

Chmura, G. L., Anisfeld, S. C., Cahoon, D. R., & Lynch, J. C. (2003). Global carbon sequestration in tidal, saline wetland soils. *Global Biogeochemical Cycles*, *17*(4), 12. http://doi.org/1111 10.1029/2002gb001917.

Christianen, M. J. A., van Belzen, J., Herman, P. M. J., van Katwijk, M. M., Lamers, L. P. M., van Leent, P. J. M., & Bouma, T. J. (2013). Low-canopy seagrass beds still provide important coastal protection services. *PloS ONE*, *8*(5), e62413. http://doi.org/10.1371/journal.pone.0062413.

Christie, N. D. (1981). Primary production in the Langebaan Lagoon with Particular

Reference to Southern Africa. *Estuarine Ecology*. Cape Town; Balkema South Africa pp.101-115.

Cohen, R. (2014). *Estimating the above-ground biomass of mangrove forests in Kenya*. (PhD. Thesis). p 36-49, University of Edinburgh, Edinburgh, UK.

Coles, R., Grech, A., Rasheed, M., Mckenzie, L., Unsworth, R., & Short, F. (2011). Seagrass Ecology and Threats in the Tropical Indo-Pacific Bioregion. In Pirog, S., R. *Seagrass: Ecology, Uses and Threats* (Ed.) (pp. 225–239). Nova Science Publishers, Inc. Queensland Australia.

Connolly, R. M. (1995). Effects of removal of seagrass canopy on assemblages of small, motile invertebrates. *Marine Ecology Progress Series*, *118*(1980), 129–137.

Coppejans, E., Beeekman, H., Wit, M. De, Morfologie, L., Planten, E. Van De, & Ledeganckstraat, K. L. (1992). The seagrass and associated macroalgal vegetation oí Gazi Bay (Kenya). *Hydrobiologia*, *247*, 59–75.

Costanza, R., Arge, R., Groot, R. De, Farberk, S., Grasso, M., Hannon, B., van den Belt, M. (1997). The value of the world’s ecosystem services and natural capital. *Nature*, *387*, 253–260. http://doi.org/10.1038/387253a0.

Crooks, S., Herr, D., Tamelander, J., Laffoley, D., & Vandever, J. (2011). "*Mitigating Climate Change through Restoration and Management of Coastal Wetlands and Near-shore Marine Ecosystems Challenges and Opportunities*". Environment Department Paper 121, World Bank, Washington, DC.

Daby, D. (2003). Effects of seagrass bed removal for tourism purposes in a Mauritian bay. *Environmental Pollution*, *125*, 313–324. http://doi.org/10.1016/S0269-7491(03)00125-8.

Dahdouh-Guebas, F., & Speybroeck, D. Van. (1999). Remote sensing and zonation of seagrasses and algae along the Kenyan coast. *Hydrobiologia*, *400*, 63–73.

Dahl, M., Deyanova, D., Lyimo, L. D., Johan, N., & Gullstr, M. (2016). Effects of shading and simulated grazing on carbon sequestration in a tropical seagrass meadow. *Journal of Ecology*, *104*, 654–664. http://doi.org/10.1111/1365-2745.12564.

de Boer, W. F. (2000). Biomass dynamics of seagrasses and the role of mangrove and seagrass vegetation as different nutrient sources for an intertidal ecosystem. *Aquatic Botany*, *66*(3), 225–239. http://doi.org/10.1016/S0304-3770(99)00072-8.

De La Torre-Castro, M., & Ronnback, P. (2004). Links between humans and seagrasses - An example from tropical East Africa. In *Ocean and Coastal Management* (Vol. 47, pp. 361–387).http://doi.org/10.1016/j.ocecoaman.2004.07.005.

Dennison, W. C., & Alberte, R. S. (1985). Role of daily light period in the depth distribution of *Zostera marina* (eelgrass). *Marine Ecology Progress Series*, *25*, 51–61. http://doi.org/10.3354/meps025051.

Di, G., & Judson, C., A., W. (2008). Evaluation of above-ground and below-ground biomass recovery in physically disturbed seagrass beds, 285–298. http://doi.org/10.1007/s00442-008-1120-0.

Duarte, C., (1991). Allometric scaling of seagrass form and productivity. *Marine Ecology progress Series 77*, 289–300. http://doi.org/10.3354/meps077289.

Duarte, C., M., & Chiscano, Carina, L. (1999). Seagrass biomass and production: a reassessment. *Aquatic Botany*, *65*, 159–174.

Ecology Progress Series, *77*, 289–300. http://doi.org/10.3354/meps077289.

Duarte, C., M. (1996). Growth and population dynamics of *Thalassodendron ciliatum* in a Kenyan back-reef lagoon. *Aquatic Botany*, *55*, 1–11.

Duarte, C., M. (2002). The future of seagrass meadows. *Environmental*

Conservation, *29*(2), 192–206. http://doi.org/10.1017/S0376892902000127.

Duarte, C. M., & Arabia, S. (2017). Reviews and syntheses : Hidden forests , the role of vegetated coastal habitats in the ocean carbon budget, 301–310. http://doi.org/10.5194/bg-14-301-2017.

Duarte, C. M., & Cebrián, J. (1996). The fate of marine autotrophic production. *Limnology and Oceanography*, *41*(8), 1758–1766. http://doi.org/10.4319/lo.1996.41.8.1758.

Duarte, C. M., & Krause-jensen, D. (2017). Export from Seagrass Meadows Contributes to Marine Carbon Sequestration. *Frontiers in Marine Science*, *4*, 1–7. http://doi.org/10.3389/fmars.2017.00013.

Duarte, C. M., Losada, I. J., Hendriks, I. E., Mazarrasa, I., & Marbà, N. (2013). The role of coastal plant communities for climate change mitigation and adaptation. *Nature Publishing Group*, *3*(11), 961–968. http://doi.org/10.1038/nclimate1970.

Duarte, C. M., Marbà, N., Gacia, E., Fourqurean, J. W., Beggins, J., Barrón, C., & Apostolaki, E. T. (2010). Seagrass community metabolism: Assessing the carbon sink capacity of seagrass meadows. *Global Biogeochemical Cycles*, *24*(4), n/a-n/a. http://doi.org/10.1029/2010GB003793.

Duarte, C. M., Middelburg, J. J., & Caraco, N. (2005). Major role of marine vegetation on the oceanic carbon cycle. *Biogeosciences*, *2*, 1–8. http://doi.org/10.5194/bgd-1-659-2004.

Duarte, C., Merino, M., & Agawin, N. (1998). Root production and belowground seagrass biomass. *Marine Ecology, 171*, 97–108.

Duarte, C. M., Nixon, S., Fulweiler, R. W., Dennison, W. B., Valiela, I., Hughes, T., & Bordalba, N. M. (2009). *Global loss of coastal habitats: Rates, causes and consequences*. Fundación BBVA, Bilboa, Spain.

Duarte, C., & Sand-Jensen, K. (1990). Seagrass colonization: patch formation and patch growth in *Cymodocea nodosa*. *Marine Ecology Progress Series*, *65*, 193–200. http://doi.org/10.3354/meps065193.

Duffy, J. E. (2006). Biodiversity and the functioning of seagrass ecosystems. *Marine Ecology Progress Series*, *311*, 233–250. http://doi.org/10.3354/meps311233.

Eklöf, J. S. (2008). *Anthropogenic Disturbances and Shifts in Tropical Seagrass Ecosystems* (PhD Thesis). Department of Systems Ecology, Stockholm University, Stockholm Sweden.

Eklöf, J. S., de la Torre Castro, M., Adelsköld, L., Jiddawi, N. S., & Kautsky, N. (2005). Differences in macrofaunal and seagrass assemblages in seagrass beds with and without seaweed farms. *Estuarine, Coastal and Shelf Science*, *63*(3), 385–396. http://doi.org/10.1016/j.ecss.2004.11.014.

Eklöf, J. S., McMahon, K., & Lavery, P. S. (2009). Effects of multiple disturbances in seagrass meadows: shading decreases resilience to grazing. *Marine and Freshwater Research*, *60*(12), 1317. http://doi.org/10.1071/MF09008.

Ellison, A. M., & Elizabeth, J. F. (1997). Global conservation status of mangroves. *Oecologia*, *112*, 435–446.

Erftemeier, P. (1993). *Factors limiting growth and production of tropical seagrasses: Nutrient dynamics in Indonesian seagrass beds*. (PhD. Thesis). Catholic University of Nijmegen, The Netherlands.

Fonseca, M., & Fisher, J. (1986). A comparison of canopy friction and sediment movement between four species of seagrass with reference to their ecology and restoration. *Marine Ecology Progress Series*, *29*, 15–22. http://doi.org/10.3354/meps029015.

Fourqurean, J. W., Duarte, C. M., Kennedy, H., Marbà, N., Holmer, M., Mateo, M. A., Krause-jensen, D. (2012). Seagrass ecosystems as a globally significant carbon stock. *Nature Geoscience, 5*: 505-509. http://doi.org/10.1038/ngeo1477.

Gab-Alla, A. (2001). Ecological Status of the Seagrass Community in Sharm El-Moyia Bay (Gulf of Aqaba, Red Sea) after Oil Pollution in 1999. *J. KAU: Marine Sciences*, *12*, 231–239.

Gacia, E. (2001). Sediment Retention by a Mediterranean *Posidonia oceanica* Meadow: The Balance between Deposition and Resuspension. *Estuarine, Coastal and Shelf Science, 52*(4), 505–514. http://doi.org/10.1006/ecss.2000.0753.

Gacia, E., Duarte, C., Marbà, N., Terrados, J., Kennedy, H., Fortes, M., & Tri, N. (2003). Sediment deposition and production in SE-Asia seagrass meadows. *Estuarine, Coastal and Shelf Science*, *56*(5–6), 909–919. http://doi.org/10.1016/S0272-7714(02)00286.

Gacia, E., Duarte, C. M., & Middelburg, J. J. (2002). Carbon and nutrient deposition in a Mediterranean seagrass (*Posidonia oceanica*) meadow. *Limnology and Oceanography*, *47*(1), 23–32. http://doi.org/10.4319/lo.2002.47.1.0023.

Gacia, E., Granata, T., & Duarte, C., (1999). An approach to measurement of particle flux and sediment retention within seagrass (*Posidonia oceanica*) meadows. *Aquatic Botany*, *65*(1–4), 255–268. http://doi.org/10.1016/S0304-3770(99)00044-3.

Gambi, M., Nowell, A., & Jumars, P. (1990). Flume observations on flow dynamics in *Zostera marina* (eelgrass) beds . *Marine Ecology Progress Series, 61*(1984), 159–169. http://doi.org/10.3354/meps061159.

GoK. (2010). *The 2009 Kenya Population and Housing Census: "Counting our people for the implementation of Vision 2030"* Kenya National Bureau of Statistics, (Vol. IC).

GoK. (2014). *Integrated Coastal Zone Management (ICZM) Action Plan Towards Integrated Management of Kenya's Coastal.* National Environment

Management Authority (NEMA), Nairobi, Kenya.

Green, E. P., Short, F. (2003). *World Atlas of Seagrasses*. Berkeley CA: California University press, 310 PP.

Greiner, J. T., McGlathery, K. J., Gunnell, J., & McKee, B. A. (2013). Seagrass Restoration Enhances “Blue Carbon” Sequestration in Coastal Waters. *PLoS One*, *8*(8), e72469. http://doi.org/10.1371/journal.pone.0072469.

Gress, S. K., Huxham, M., Kairo, J. G., Mugi, L. M., & Briers, R. A. (2016). Evaluating, predicting and mapping belowground carbon stores in Kenyan mangroves. *Global Change Biology*. http://doi.org/10.1111/gcb.13438.

Grindley. J. R. 1976. *Report on ecology of Knysna Estuary and proposed Braamekraal Marina*. (Internal Report). School of Environmental Studies, University of Cape Town. South Africa. pp.133.

Gullström, M., Bodin, M., Nilsson, P. G., & Öhman, M. C. (2008). Seagrass structural complexity and landscape configuration as determinants of tropical fish assemblage composition. *Marine Ecology Progress Series*, *363*, 241–255. http://doi.org/10.3354/meps07427.

Gullström, M., de la Torre Castro, M., Bandeira, S. O., Björk, M., Dahlberg, M., Kautsky, N., Öhman, M. C. (2002). Seagrass Ecosystems in the Western Indian Ocean. *AMBIO: A Journal of the Human Environment*, *31*(7), 588. http://doi.org/10.1639/0044-7447(2002)031[0588:SEITWI]2.0.CO2.

Gullström, M., Lundén, B., Bodin, M., Kangwe, J., Öhman, M. C., Mtolera, M. S. P., & Björk, M. (2006). Assessment of changes in the seagrass-dominated submerged vegetation of tropical Chwaka Bay (Zanzibar) using satellite remote sensing. *Estuarine, Coastal and Shelf Science*, *67*(3), 399–408. http://doi.org/10.1016/j.ecss.2005.11.020.

Guy, H. (2010). The microbial role in carbon cycling within seagrass sediments. *The*

Plymouth Student Scientist, *3*(1), 234–244.

Gwada, P. (2004). *An assessment of seagrass survival and functioning in response to manipulations in sediment redox at Nyali Lagoon , Kenya.* Mombasa. WIOMSA report. MARG-1. GRANTEE 2001/2.

Haas, L. W. (1977). The Effect of the Spring-neap Tidal Cycle on the Vertical Salinity Structure of the Analysis of salinity data collected during the period following Tropical. *Estuarine Coastal Marine Science*, *5*, 485–496.

Hanekom, N., & Lagoon, L. (1988). Distribution and variations in seasonal biomass of eelgrass. *Zostera capensis* in the Kromme estuary, St Francis Bay South Africa. *S.Afri. J. mar. Sci*.7: 51–59.

Hansen, J. C. R., & Reidenbach, M. A. (2013). Seasonal growth and senescence of a *Zostera marina* seagrass meadow alters wave-dominated flow and sediment suspension within a coastal bay. *Estuaries and Coasts*, *36*(6), 1099–1114. http://doi.org/10.1007/s12237-013-9620-5.

He, M., Luis, S., Rita, S., Ana, G., Eurípedes, V. J., & Zhang, N. (2011). Risk assessment of CO_2 injection processes and storage in carboniferous formations : a review. *Journal of Rock Mechanics and Geotechnical Enginnering*, *3*(1), 39–56. http://doi.org/10.3724/SP.J.1235.2011.00039.

Heck, K. L., Carruthers, T. J. B., Duarte, C. M., Randall Hughes, A., Kendrick, G., Orth, R. J., & Williams, S. W. (2008). Trophic transfers from seagrass meadows subsidize diverse marine and terrestrial consumers. *Ecosystems*, *11*(7), 1198–1210. http://doi.org/10.1007/s10021-008-9155.

Heck, K. L., & Valentine, J. F. (2006). Plant-herbivore interactions in seagrass meadows. *Journal of Experimental Marine Biology and Ecology*, *330*(1), 420–436. http://doi.org/10.1016/j.jembe.2005.12.044.

Hemminga, M. A., & Duarte, C. M. (2000). *Seagrass Ecology*, An introduction..

Cambridge University Press, Cambridge, United Kingdom, 286 pp. ISBN: 0521661846.

Hemminga, M. A., Gwada, P., Slim, F. J., de Koeyer, P., & Kazungu, J. (1995). Leaf production and nutrient contents of the seagrass *Thalassodendron ciliatum* in the proximity of a mangrove forest (Gazi Bay, Kenya). *Aquatic Botany*, *50*(2), 159–170.

Hemminga, M. A., & Mateo, M. A. (1996). Stable carbon isotopes in seagrasses : variability in ratios and use in ecological studies, *Marine Ecology Progress Series*, *140*, 285–298.

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*, *106*(3), 291–302.

Hendriks, I., Sintes, T., Bouma, T., & Duarte, C. (2008). Experimental assessment and modeling evaluation of the effects of the seagrass *Posidonia oceanica* on flow and particle trapping. *Marine Ecology Progress Series*, *356*, 163–173. http://doi.org/10.3354/meps07316.

Herkül, K., & Kotta, J. (2009). Effects of eelgrass (*Zostera marina*) canopy removal and sediment addition on sediment characteristics and benthic communities in the Northern Baltic Sea. *Marine Ecology*, *30*, 74–82. http://doi.org/10.1111/j.1439-0485.2009.00307.

Hinkel, J., Brown, S., & Exner, L. (2012). Sea-level rise impacts on Africa and the effects of mitigation and adaptation : an application of DIVA. *Regional Environmental Change*, *12*, 207–224. http://doi.org/10.1007/s10113-011-0249-2.

Ho, C. L., & Barrett, B. B. (1976). Distribution of nutrients in Louisiana's Coastal waters influenced by the Mississippi River. *Estuarine Coasts and Marine Science*, *5*, 173–195.

Howard, J., Hoyt, S., Isensee, K., Pidgeon, E., & Telszewski, M. (2014) (Ed.). *Coastal Blue Carbon:Methods for assessing carbon stocks and emmission factors in mangroves, tidal salt marshes and seagrasses*. Conservation International, Intergovernmental Oceanographic Commission of UNESCO, International Union for Conservation of Nature. In *Conservation International* (Vol. 36, pp. 99–107). Arlington, Virginia, USA.

Howard, J., Sutton-grier, A., Herr, D., Kleypas, J., Landis, E., Mcleod, E., Simpson, S. (2017). Clarifying the role of coastal and marine systems in climate mitigation. *Frontiers in Ecology and the Environment*, *51*(1), 42–50. http://doi.org/10.1002/fee.1451.

Huxham, M., Marappullige, P., Loku, P. J., Krauss, K. W., Kairo, J., Langat, J., Kirui, B. (2010). Intra- and interspecific facilitation in mangroves may increase resilience to climate change threats. *Philosophical Transactions*, *365*, 2127–2135. http://doi.org/10.1098/rstb.2010.0094.

Imen, B. S., & Abdessalem, S. (2014). Distribution, biomass and phenology of *Zostera noltei* meadows in the Northern lagoon of Tunis in the summer of 2014. In Guala I & Fais M (Eds.). *Mediterranean Seagrass Workshop book of abstract* (P 80). Sardinia, Oristano, Italy Fondazione Banco di Sardegma.

IPCC. (2014). *Climate Change 2014. Mitigation of Climate Change.* (O. Edenhofer, R. Pichs-Madruga, Y. Sokona, E. Farahani, S. Kadner, K. Seyboth, J. C. Minx, Eds.). Cambridge, UK: Cambridge University Press.

IPCC. (2013). *2014, In supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands,* Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M. and Troxler, T.G. (Ed.). Published:

IPCC, Switzerland. Comprehensive Organic synthesis.

Irving, A. D., Connell, S. D., & Russell, B. D. (2011). Restoring coastal plants to improve global carbon storage: reaping what we sow. *Plos ONE, 6*(3), e18311. http://doi.org/10.1371/journal.pone.0018311.

Isaac, W., & Isaac, F. (1968). Marine botany of the Kenya coast. General account of the environment, flora and vegetation. *JE Afr. Nat. Hist. Soc, 27*(116), 7–27.

IUCN. (2011). *Rio+20 and the importance of achieving the Strategic Plan for Biodiversity* (2011-.2020) and the Aichi Biodiversity Targets. The Strategic Plan for Biodiversity.

Jevrejeva, S., Grinsted, A., Moore, J. C., & Holgate, S. (2006). Nonlinear trends and multiyear cycles in sea level records. *Journal of Geophysical Research, 111*, 1–11. http://doi.org/10.1029/2005JC003229.

Jokiell, P. L., & Janice, I. (1993). Water motion on coral reefs : evaluation of the â€™ clod card â€™ technique. *Marine Ecology Progress Series*, *93*, 175–181.

Kairo, J. G., Lang, J. K. S., Dahdouh-guebas, F., Bosire, J., & Karachi, M. (2008). Structural development and productivity of replanted mangrove plantations in Kenya. *Forest Ecology and Management*, *255*, 2670–2677. http://doi.org/10.1016/j.foreco.2008.01.031.

Kaldy, J., & Dunton, K. (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. http://doi.org/10.3354/meps193271.

Kamermans, P., Hemminga, M. A., Tack, J. F., Mateo, M. A., Marbà, N., Mtolera, M., Daele, T. Van. (2002). Groundwater effects on diversity and abundance of lagoonal seagrasses in Kenya and on Zanzibar Island (East Africa). *Marine Ecology Progress Series*, *231*, 75–83.

Kamermans, P., Hemminga, M. A, Marbà, N., Mateo, M. A, Mtolera, M., & Stapel, J. (2001). Leaf production, shoot demography, and flowering of *Thalassodendron ciliatum* along the east African coast. *Aquatic Botany*, *70*(3), 243–258. http://doi.org/10.1016/S0304-3770(01)00156-5.

Kauffman, J. B., Heider, C., Cole, T. G., Dwire, K. A., & Donato, D. C. (2011). Ecosystem Carbon Stocks of Micronesian Mangrove Forests. *Wetlands*, *31*, 343–352. http://doi.org/10.1007/s13157-011-0148-9.

Kawaka, J. A., Samoilys, M. A., Murunga, M., Church, J., Abunge, C., & Waweru, G. (2016). Ocean & Coastal Management Developing locally managed marine areas : Lessons learnt from Kenya. *Ocean and Coastal Management*, *135*, 1–10. http://doi.org/10.1016/j.ocecoaman.2016.10.013.

Kennedy, H., Beggins, J., Duarte, C. M., Fourqurean, J. W., Holmer, M., Marbá, N., & Middelburg, J. J. (2010). Seagrass sediments as a global carbon sink: Isotopic constraints. *Global Biogeochemical Cycles*, *24*(4). http://doi.org/10.1029/2010GB003848.

Kennedy, H., Gacia, E., Kennedy, D. P., Papadimitriou, S., & Duarte, C. M. (2004). Organic carbon sources to SE Asian coastal sediments. *Estuarine, Coastal and Shelf Science*, *60*(1), 59–68. http://doi.org/10.1016/j.ecss.2003.11.019.

Kirui, B., Kairo, J. G., & Karachi, M. (2006). Allometric equations for estimating above ground biomass of *Rhizophora mucronata* Lamk. (Rhizophoraceae) mangroves at Gazi Bay, Kenya. *Western Indian Ocean J. Mar. Sci.*, 27–34.

Kitheka, J. U. (1996). Water circulation and coastal trapping of brackish water in a tropical mangrove-dominated bay in Kenya. *Limnology and Oceanography*, *41*(l), 169–176.

Koch, E. W., Ackerman, J. D., Verduin, J., & Keulen, M. Van. (2002). Fluid Dynamics in Seagrass Ecology — from Molecules to Ecosystems, In Seagrass

Ecology and Conseravation (Ed.) AWD larkum, RJ. Orth and C. Duarte, p193–225.Amsterdam, Springer.

Koch, M. S., Schopmeyer, S. A., Kyhn-Hansen, C., Madden, C. J., & Peters, J. S. (2007). Tropical seagrass species tolerance to hypersalinity stress. *Aquatic Botany, 86*(1), 14–24. http://doi.org/10.1016/j.aquabot.2006.08.003.

Krauss, K. W., Mckee, K. L., Lovelock, C. E., Cahoon, D. R., Saintilan, N., Reef, R., & Chen, L. (2014). Tansley review How mangrove forests adjust to rising sea level. *New Phytologist*, *202*(1), 19–34. http://doi.org/10.1111/nph.12605.

KWS. (2013). *Coral reefs and seagrass ecosystems conservation strategy 2014-2018*. P 8-21, Nairobi, Kenya.

Laan, B. B. P. A. Van Der, & Wolff, W. J. (2006). Circular pools in the seagrass beds of the Banc d ' Arguin , Mauritania, and their possible origin. *Aquatic Botany*, *84*, 93–100. http://doi.org/10.1016/j.aquabot.2005.07.009.

Laffoley, D., & Grimsditch, G.(Eds) (2009). *The management of natural coastal carbon sinks*. Gland, Switzerland, IUCN.

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.

Landry, J. B., Kenworthy, W. J., & Carlo, G. Di. (2008). *The Effects of Docks on Seagrasses , With Particular Emphasis on the Threatened Seagrass, Halophila johnsonii*. Protected Coastal division, Beaufort, South Carolina USA.

Larkum, A. W. D., Orth, R. J., & Duarte, C. M. (2006). Photosynthesis and metabolism of seagrasses at a cellular level. In *Seagrasses: Biology, Ecology and Conservation,* (Ed.) W.D. larkum, R.J. Orth and C.M. Duarte, 323-345. Dordrecht, The Netherlands: Springer. http://doi.org/10.1007/1-4020-2983-7.

Larsson, S. (2009). *The Production of the Seagrass Thalassia hemprichii in Relation to Epiphytic* (M.sc. Thesis, unpublished). Gothenburg University, Sweden, pp. 12.

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. http://doi.org/10.1371/journal.pone.0073748.

Lo Iocano, C., Mateo, M. A., Gracia, E., Guasch, L., Carbonell, R., Serrano, L., Danobeitia, J. (2008). Very high-resolution seismo-acoustic imaging of seagrass meadows (Mediterranean Sea): Implications for carbon sink estimates. *Geophysical Research Letters*, *35*(18), 1–5. http://doi.org/10.1029/2008GL034773.

Lugendo, B.R Mgaya, Y., & Semesi, A. (2001). The seagrass and associated macroalgae at selected beaches along Dar es Salaam coast. *Science in Tanzania, 1*, 359–374.

Lugendo, B. R., Nagelkerken, I., Van Der Velde, G., & Mgaya, Y. D. (2006). The importance of mangroves, mud and sand flats, and seagrass beds as feeding areas for juvenile fishes in Chwaka Bay, Zanzibar: Gut content and stable isotope analyses. *Journal of Fish Biology*, *69*(6), 1639–1661. http://doi.org/10.1111/j.1095-8649.2006.01231.x.

Lyimo, T., Mvungi, E., & Mgaya, Y. (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 J. Mar. Sci.*, *5*(2), 141–152.

Macreadie, P. I., York, P. H., Sherman, C. D. H., Keough, M. J., Ross, D. J., Ricart, A. M., & Smith, T. M. (2014). No detectable impact of small-scale disturbances on “blue carbon” within seagrass beds. *Marine Biology*, *161*(12), 2939–2944. http://doi.org/10.1007/s00227-014-2558-8.

Marba’, N., Arias-Ortiz, A., Masque, 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. http://doi.org/10.1111/1365-2745.12370.

Marba’, N., Hemminga, M. A., Mateo, M. A., Duarte, C. M., Mass, Y. E. M., Terrados, J., & Gacia, E. (2002). Carbon and nitrogen translocation between seagrass ramets. *Marine Ecology Progress Series*, *226*, 287–300. http://doi.org/10.3354/meps226287.

Martins, A., & Bandeira, S. (2001). Biomass distribution and leaf nutrient concentrations and resorption of *Thalassia hemprichii* at Inhaca Island, Mozambique. *South African Journal of Botany*, *67*, 439–442.

McGlathery, Karenck, K., & Anderson, I. C. (2007). Eutrophication in shallow coastal bays and lagoons: The role of plants in the coastal filter. *Marine Ecology Progress Series*. http://doi.org/10.3354/meps07132.

Mckee, K. L., Cahoon, D. R., & Feller, I. C. (2007). Caribbean mangroves adjust to rising sea level through biotic controls on change in soil elevation, 545–556. http://doi.org/10.1111/j.1466-8238.2007.00317.x

Mcleod, E., Chmura, G. L., Bouillon, S., Salm, R., Björk, M., Duarte, C. M., Silliman, B. R. (2011). A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO_2 *Frontiers in Ecology and the Environment*, *9*(10), 552–560. http://doi.org/10.1890/110004.

Millenium Ecosystem Assessment. (2005). Ecosystems and Human Well-being: A Framework for Assessment. *Island Press, Washington, DC.*, 1–25.

Mostafa H.M. 1996. Preliminary observations of the seagrass *Cymodocea nodosa* (Ucria) Ascherson in the Mediterranean waters of Alexandria, Egypt. *Bull. of Natl Inst. Ocea. Fish.* 22: 19-28.

MPRUT-KWS. (2016). *A proposed marine Transboundary Conservation Area between Kenya and Tanzania*, 69PP.

Mudd, S. M., Howell, S. M., & Morris, J. T. (2009). Impact of dynamic feedbacks between sedimentation, sea-level rise, and biomass production on near-surface marsh stratigraphy and carbon accumulation. *Estuarine, Coastal and Shelf Science*, *82*(3), 377–389. http://doi.org/10.1016/j.ecss.2009.01.028.

Mumby, P. J. (2005). Connectivity of reef fish between mangroves and coral reefs : Algorithms for the design of marine reserves at seascape scales, *8*. http://doi.org/10.1016/j.biocon.2005.09.042.

Mvungi, E. F. (2011). *Seagrasses and eutrophication Interactions between seagrass photosynthesis* ,. Stockholm University, Sweden. http://doi.org/ISBN 978-91-7447-250-9.

Nacorda, H. M. (2008). *Burrowing shrimps and seagrass dynamics in shallow-water meadows off Bolinao (NW Philippines),* (Phd thesis). Wageningen University, Wageningen, The Netherlands.

Nagelkerken, I., Roberts, C. M., Velde, G. Van Der, Dorenbosch, M., & Riel, M. C. Van. (2002). How important are mangroves and seagrass beds for coral-reef fish ? The nursery hypothesis tested on an island scale, *Mar Ecol Prog. Ser. 244*, 299–305.

Nellemann, C., Corcoran, E., Duarte, C., Valdes, L., De Young, C., Fonseca, L., & Grimsditch, G. (2009). *Blue carbon. A Rapid Response Assessment (78) United*

Nations Environmental Programme, GRID-ARENDAL.

Nicholls, R. J., & Cazenave, A. (2010). Sea-Level Rise and Its Impact on Coastal Zones. *Science, 328.*

Ochieng, C. A. (1995). *Productivity of seagrasses with respect to intersystem fluxes Gazi Bay (Kenya).* Mombasa. http://doi.org/Final report. Contract no TS3* ct92-0114.

Ochieng, C. A., & Erftemeijer, P. L. (1999). Accumulation of seagrass beach cast along the Kenyan coast: a quantitative assessment. *Aquatic Botany*, *65*(1–4), 221–238. http://doi.org/10.1016/S0304-3770(99)00042-X.

Ochieng, C. A., & Erftemeijer, P. L. (2003). Kenya and Tanzania. In Green E., and Short F. (Ed.). The seagrasses of Kenya and Tanzania, World Atlas of Seagrasses World Conservation Monitoring Centre (Vol. 7, pp. 82–92). University of California Press, PP 83-100.

Orth, R. J., Carruthers, T. J. B., Dennison, W. C., Duarte, C. M., Fourqurean, J. W., Heck, K. L., Williams, S. L. (2006). A Global Crisis for Seagrass Ecosystems. *BioScience*, *56*(12), 987–997.

Paling, E. I., & McComb, A. J. (2000). Autumn biomass, below-ground productivity, rhizome growth at bed edge and nitrogen content in seagrasses from Western Australia. *Aquatic Botany*, *67*(3), 207–219. http://doi.org/10.1016/S0304-3770(00)00092-9.

Palomar, N. E., Juinio-Meñez, M. a., & Karplus, I. (2004). Feeding habits of the burrowing shrimp *Alpheus macellarius*\n. *Journal of the Marine Biological Association of the UK*, *84*, 1199–1202. http://doi.org/10.1017/S0025315404010641h.

Pangallo, R. A., & Bell, S. S. (1988). Dynamics of the aboveground and belowground structure of the seagrass *Halodule wrightii*, *43*(Dawes 1974), 297–301.

Papadimitriou, S., Kennedy, H., Kennedy, D. P., & Borum, J. (2005). Seasonal and spatial variation in the organic carbon and nitrogen concentration and their stable isotopic composition in *Zostera marina* (Denmark). *Limnology and Oceanography, 50*(4), 1084–1095. http://doi.org/10.4319/lo.2005.50.4.1084.

Parry, M., Arnell, N., Berry, P., Dodman, D., Fankhauser, S., Hope, C., Wheeler, T. (2009). *Assessing the costs of Adaptation to Climate Change . A review of the UNFCCC and other recent estimates*. http://doi.org/10.1641/0006-3568(2001)051[0723:CCAFD]2.0.CO;2

Paula, J., Ecosta, P. F., Martins, A., & Gove, D. (2001). Patterns of abundance of seagrasses and associated infaunal communities at Inhaca Island, Mozambique. *Estuarine, Coastal and Shelf*, *53*, 307–318. http://doi.org/10.1006/ecss.2001.0809.

Pendleton, L., Donato, D. C., Murray, B. C., Crooks, S., Jenkins, W. A., Sifleet, S., Baldera, A. (2012). Estimating Global “Blue Carbon” Emissions from Conversion and Degradation of Vegetated Coastal Ecosystems. *PLoS ONE*, *7*(9). http://doi.org/10.1371/journal.pone.0043542.

Pergent, G., Djellouli, A, Hamza, A. A., Ettayeb, K. S., El Mansouri, A. A., Talha, F. M., Platini, F. (2002). Characterization of the benthic vegetation in the Farwà Lagoon (Libya). *Journal of Coastal Conservation, 8*(2), 119. http://doi.org/10.1652/1400-0350(2002)008[0119:COTBVI]2.0.CO;2

Porter, E. T., Sanford, L. P., & Suttles, S. E. (2000). Gypsum dissolution is not a universal integrator of “ water motion ,” *Limnol. Oceanogr. 45*(1), 145–158.

Prrez-llorrns, J. L. (1993). Seasonal dynamics ofbiomass and nutrient content in the intertidal seagrass *Zostera noltii* Hornem from Palmones River estuary, Spain, *Aquatic botany*. *46*, 49-66.

Ralph, P., Durako, M., & Enriquez, S. (2007). Impact of light limitation on seagrasses. *Journal of Experimental*, *350*, 176–193. http://doi.org/10.1016/j.jembe.2007.06.017.

Resnik, D. B., & Vallero, D. A. (2011). Geoenginnering: An Idea Whose Time Has Come? *J. Earth Sci Clim Change*, *1*(919), 1–18.

Richmond, M. D. (Ed.) (1997). *A Guide to the Seashores of Eastern Africa and the Western Indian Ocean Islands. Sweden: Sida/Department for Research Cooperation, SAREC. 448pp*.

Romero, J., Pe'rez, M., Mateo, M. A., & Sala, E. (1994). The belowground organs of the Mediterranean seagrass *Posidonia oceanica* as a biogeochemical sink. *Aquatic Botany*, *47*(1), 13–19. http://doi.org/10.1016/0304-3770(94)90044-2.

Russell, L. M., Rasch, P. J., Mace, G. M., Jackson, R. B., Shepherd, J., Liss, P., Morgan, M. G. (2012). Ecosystem Impacts of Geoengineering : A Review for Developing a Science Plan. *Ambio*, *41*, 350–369. http://doi.org/10.1007/s13280-012-0258-5.

Scheffran, J., & Battaglini, A. (2011). Climate and conflicts : the security risks of global warming. *Regional Environmental Change*, *11*, 26–39. http://doi.org/10.1007/s10113-010-0175-8.

Scheneider, S. H. (2008). Geoengineering : could we or should we make it work ? *Philosophical Transactions of the Royal Society*, *366*, 3843–3862. http://doi.org/10.1098/rsta.2008.0145.

Schott, F. A., Xie, S.-P., & McCreary, J. P. (2009). Indian Ocean Circulation and Climate Variability. *Reviews of Geophysics*, *47*, 1–46. http://doi.org/10.1029/2007RG000245.1.

Semroud, R., S.Mezegrane, L.Soltane. 1990. Etude lépidochronologique de *Posidonia oceanica* dans la régiond'Alger (Algérie): donnéspréliminaires.

Rapp. P.V.Réun. Comm.*Int. Explor. Sci. Médit*. 31: 10.

Serrano, O., Lavery, P. S., López-Merino, L., Ballesteros, E., & Mateo, M. A. (2016). Location and Associated Carbon Storage of Erosional Escarpments of Seagrass *Posidonia Mats*. *Frontiers in Marine Science*, *3*, 42. http://doi.org/10.3389/fmars.2016.00042.

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. http://doi.org/10.1038/srep23193.

Sghaier, Y. R. (2012). Seasonal variation of *Cymodocea nodosa* in the Ghar El Melh lagoon (Tunisia), with reference to insolation, temperature and salinity effects. *Bull. Inst. Natn. Scien. Tech.Mede. Salammbõ*, *39*, 117–125.

Sghaier, Y. R., Zakhama-Sraieb, R., & Charfi-Cheikhrouh, F. (2011). Primary production and biomass in a *Cymodocea nodosa* meadow in the Ghar El Melh lagoon, Tunisia. *Botanica Marina*, *54*(4), 411–418. http://doi.org/10.1515/BOT.2011.045.

Sghaier, Y. R., Zakhama-Sraieb, R. Y. M., & Charfi-Cheikhrouha, F. (2013). Patterns of shallow seagrass (*Posidonia oceanica*) growth and flowering along the Tunisian coast. *Aquatic Botany*, *104*, 185–192. http://doi.org/10.1016/j.aquabot.2011.09.006.

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*(1–2), 3–20. http://doi.org/10.1016/j.jembe.2007.06.012.

Stilgoe, J. (2016). Geoengineering as Collective Experimentation. *Science and Engineering Ethics*, *22*, 851–869. http://doi.org/10.1007/s11948-015-9646-0

Taenzler, D., Vivekananda, J., Kolarova, D., & Dokos, T. (2012). *Climate Change* *and Conflict, Synthesis Report, IfP-EW:* Brussels, Belgium.

Talbot, M. M. B., & Bate, G. C. (1987). The distribution and biomass of the seagrass *Zostera capensis* in a warm-temperate estuary. *Botanica Marina*, *30*(1), 91–99.

Tamooh, F., Huxham, M., Karachi, M., Mencuccini, M., Kairo, J. G., & Kirui, B. (2008). Below-ground root yield and distribution in natural and replanted mangrove forests at Gazi bay , Kenya. *Forest Ecology and Management , 256*, 1290–1297. http://doi.org/10.1016/j.foreco.2008.06.026.

Terrados, J., Duarte, C. M., Fortes, M. D., Borum, J., Agawin, N. S. R., Bach, S., Vermaat, J. (1998). Changes in Community Structure and Biomass of Seagrass Communities along Gradients of Siltation in SE Asia. *Estuarine, Coastal and Shelf Science*, *46*(5), 757–768. http://doi.org/10.1006/ecss.1997.0304.

Tomasko, D. A., Hall, M. O., & Tomasko, D. A. (2016). Productivity and Biomass of the Seagrass *Thalassia testudinum* along a Gradient of Freshwater Influence , *Estuaries and Coasts*, *22*(3), 592–602.

Tuda, A., & Omar, M. (2012). *Protection of Marine Areas in Kenya*. *The George Wright Forum* (Vol. 29).

Uku, J., & Björk, M. (2005). Productivity aspects of three tropical seagrass species in areas of different nutrient levels in Kenya. *Estuarine, Coastal and Shelf Science*, *63*(3), 407–420. http://doi.org/10.1016/j.ecss.2004.12.007.

Uku, J. N., Martens, E. E., & Mavuti, K. M. (1996). *An Ecological Assessment of Littoral Seagrass Communities in Diani and Galu Coastal Beaches , Kenya*.

UNEP. (2008). *Ecosystem Management*.UNEP, annual report, Nairobi, Kenya.

UNEP. (2009). *THE Natural Fix ?: The role of Ecosystems in climate mitigation, A UNEP Rapid Response Assessment-United Nations Environment Programme, UNEP-WCMC, Cambridge, UK.*

UNEP. (2012). Seagrasses in the Western Indian Ocean region. In United Nations

Environment Programme (Ed.), *UNEP (DEPI)/EAF/CP.7/Inf17* (pp. 1–3). Maputo, Mozambique.

UNFCC. (1998). *The Kyoto Protocol to the United Nations Framework Convention on Climate Change*. (UNFCCC). 2303 UNTS 148 / [2008] ATS 2 / 37 ILM 22 (1998).

UNFCC. (2010). *Report of the Conference of the Parties on its fifteenth session , held in Copenhagen from 7 to 19 December 2009 Part Two : Action taken by the Conference of the Parties at its fifteenth session Decisions adopted by the Conference of the Parties*.

UNFCC. (2015). Report of the Conference of the Parties on its twenty-first session, held in Paris from 30 November to 13 December 2015. In *Addendum. Part Two: Action taken by the Conference of the Parties at its twenty-first session.*

UNFCC. (2016). Report of the conference of parties on its twenty first session, held in Paris from 30th November to 13th December 2015.addendum.Part two: action taken by the conference of the parties at its twenty first session, FCCC/CP/2015/10/Add.2,UNFCCC Secretariat, Bonn.

Unsworth, R. K. F., & Cullen, L. C. (2010). Recognising the necessity for Indo-Pacific seagrass conservation. *Conservation Letters, 3*(2), 63–73. http://doi.org/10.1111/j.1755-263X.2010.00101.x.

Van Lent, F., Nienhuis, P. H., & Verschuure, J. M. (1991). Production and Biomass of the Seagrasses *Zostera-Noltii* (Hornem) and *Cymodocea-Nodosa* (Ucria) Aschers at the Banc-Darguin (Mauritania, Nw Africa) - A Preliminary Approach. *Aquatic Botany, 41,* 353–367.

Vermaat, J. E., Beijer, J. A., J., Gijlstra, R., Hootsmans, M. J. M., Philippart, C. J. M., van den Brink, N. W., & van Vierssen, W. (1993). Leaf dynamics and standing stocks of intertidal *Zostera noltii* (Hornem). and *Cymodocea nodosa*

(Ucria) Ascherson on the Banc d'Arguin (Mauritania). *Hydrobiologia*, *258*(1–3), 59–72. http://doi.org/10.1007/BF00006186.

Vonk, J. A., Christianen, M. J. A., & Stapel, J. (2008). Redefining the trophic importance of seagrasses for fauna in tropical. *Estauarine, Coastal and Shelf Science*, *79*, 653–660. http://doi.org/10.1016/j.ecss.2008.06.002.

Watson, R. a., Coles, R. G., & Long, W. J. L. (1993). Simulation estimates of annual yield and landed value for commercial penaeid prawns from a tropical seagrass habitat, northern Queensland, Australia. *Australian Journal of Marine and Freshwater Research*, *44*(1990), 211–219. http://doi.org/10.1071/MF9930211.

Waycott, M., Duarte, C. M., Carruthers, T. J. B., Orth, R. J., Dennison, W. C., Olyarnik, S., Williams, S. L. (2009). Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Ecology*, *106*(30), 12377–12381. http://doi.org/10.1073/pnas.0905620106.

Publications

Published article

Githaiga M.N, Gilpin L, Kairo JG, Huxham M. Biomass and productivity of seagrasses in Africa. Bot Mar. 2016;59(2-3):173–86.

Githaiga M.N, Gilpin L, Kairo JG, Carbon storage in the seagrass meadows of Gazi Bay, Kenya (number: PONE-D-16-32585), PLoS ONE journal-manuscript revised and re-submitted.