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THE SUCCESS OF ECO-ENGINEERING MANGROVE RESTORATION IN A HIGH ENERGY AREA, AT GAZI BAY, KENYA

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DECLARATION

This thesis is my original work and has not been presented elsewhere for a degree or any other award.

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DEDICATION

I humbly dedicate this work to God for the abundant grace, wisdom, knowledge, and understanding bestowed upon me. I extend this dedication to my mum, Ajelicah Kamotho, and my siblings, Rose Mwari, Stellah Kabeti, Everjoy Kanario, and Doris Karimi, for their unwavering support and inspiration throughout my study. Lastly, to my adorable daughter, Talia Kawira, whose presence fixed my vision and focus throughout this journey.

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LIST OF ABBREVIATIONS AND ACRONYMS

ANOVA	Analysis of Variance
AoI	Area of Interest
CO_2	Carbon (IV) dioxide
DBH	Diameter at Breast Height
FAO	Food and Agriculture Organization
GCP	Ground Control Points
GOK	Government of Kenya
GIS	Geographical Information System
GPS	Global Positioning System
IOT	Indian Ocean Tsunami
IPCC	Intergovernmental Panel for Climate Change
IV	Importance Value Index
KMFRI	Kenya Marine and Fisheries Research Institute
MSL	Mean Sea Level
NGOs	Non-Governmental Organizations
PVC	Polyvinyl Chloride
RCBD	Randomized Complete Block Design
REM	Riley Encasement Method
RMS	Root mean Square
R^2	Correlation coefficient
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
SPSS	Statistical Package for the Social Sciences
UN	United Nations
UNEP	United Nations Environment Program
UTM	Universal Transverse Mercator
USGS	United States Geological Survey
VCM	Voluntary Carbon Market
WIO	Western Indian Ocean

ABSTRACT

Mangroves and their ecosystem offer a range of globally recognized benefits, yet they continue to be lost and degraded. Efforts to restore lost mangroves using conventional methods in high-energy areas result in low success rates due to the removal of seedlings via wave's action. This study assessed the efficacy of using modified Riley Encasement Methods (REMs) in the restoration of mangroves in high-energy areas at Gazi Bay, Kenya. Prior to the mangrove planting experiment, mapping for mangrove cover change, and assessment of vegetation and soil conditions were done. Global Positioning System (GPS) device was used to record the study site coordinates. Landsat images were systematically sampled using these coordinates from the United States Geological Survey (USGS). Vegetation and soil baseline data were collected in 49 square plots of 100 m^2; established along belt transects perpendicular to the shoreline. The mangrove vegetation structural data that was collected included; species composition, tree height (m), and stem diameter (cm); from which the importance value index (IV), basal area (m^2 ha^{-1}) and standing density (stems ha^{-1}) were derived. Sediment cores were made in the center of each square plot for carbon and grain size analysis. Mangrove (*Rhizophora mucronata*) planting was done using a randomized complete block design (RCBD) in which the planting area was divided into three sections. On each section, one block measuring 7 m by 5.5 m was established, resulting to three blocks namely A, B, and C. Within each block, treatments that involved use of bamboo and different-sized PVC pipes were randomly assigned locations. Monitoring involved assessment of survival and growth parameters including shoot growth, number of leaves, number of internodes, number of branches and leaf area. Statistical data analysis was done using SPSS version 26.0, GRADISTAT computer program and Microsoft Excel 2019. The findings of this study were that the mangrove forest was highly degraded recording relatively low proportions of silt and clay (3.03 ± 0.17%), soil organic matter (6.33 ± 0.24) and soil organic carbon (5.52 ± 0.10). Following repeated measures of Analysis of Variance (ANOVA) and a post-hoc Tukey's Honest Significant Difference (HSD) test ($p < 0.05$), the results of the planting experiment revealed significant variations in survival and growth rates among treatments. Seedlings grown within PVC encasements recorded significantly higher survival rates (43%) compared to those in the bamboo (1%) and control groups (4%). These findings suggest that PVC pipes were efficient in supporting and protecting seedlings from external forces. The study highlights the potential of adopting the encasement technique in mangrove restoration. These findings are particularly relevant to environmental conservation policies, climate change mitigation strategies, and coastal community development programs. Current mangrove restoration policies should consider the potential of eco-engineering techniques in addressing challenges facing mangrove restoration in high energy sites.

CHAPTER ONE

INTRODUCTION

1.1 Background of the study

Mangroves are recognized as one of the world's most productive ecosystems, playing crucial roles ecologically, economically, and socially (Triyanti & Chu, 2016; Faridah-Hanum et al., 2019). Globally, millions of people rely on mangroves for various ecosystem goods and services, including coastal protection, fisheries, tourism, and forestry products and carbon sequestration (Mazda et al., 2006; Barbier et al., 2011; Donato et al., 2011). These goods and services contribute to poverty reduction, increased food security, opportunities for tourism and recreation, as well as to the moderation of extreme weather events (Barbier, 2016; Friess et al., 2020). Mangroves support the livelihoods of over 2 million people worldwide through fisheries, forestry, and tourism activities (UNEP, 2014). For example, in the Sundarbans and Bangladesh, approximately 4.3 million people depend directly on mangroves for their livelihoods (Ghosh et al., 2015). In Southeast Asia, around 30% of small-scale fishers depend on mangroves (Barbier et al., 2011). Approximately 80% of coastal communities in Kenya depend on the mangrove ecosystem for their livelihoods (Huxham et al., 2018).

Mangroves provide habitat and breeding ground for marine fauna, including fish (Sanderman et al., 2018). In addition, mangroves sequester about five times more carbon than tropical forests (Pendleton et al., 2012; Sharma et al., 2019). Research shows that mangrove sediments together with the roots are estimated to hold approximately 6.4 billion tons of carbon (Sanderman et al., 2018), and capture about 30 million tons every year (Howard et al., 2017), and therefore are suitable for mitigating climate change (Sidik et al., 2018). Moreover, mangroves and associated ecosystems play a significant role in shoreline and coastal protection (Gedan et al., 2011; Shepard, et al., 2011; Sandilyan & Kathiresan, 2015; Indarsih & Masruri, 2019). This is accomplished through attenuation of the waves’ energy, sediment stabilization and accretion (Lewis, 2005; Alongi, 2008; Ostling et al., 2009; Gedan et al., 2011; Shepard et al., 2011; Steven et al., 2020).

Despite their great value, mangroves throughout the world continue to be lost and degraded as a result of human and natural causes (Carugati et al., 2018; Goldberg et al., 2020). Over the past 50 years, about 35% of the world's mangroves have been lost due to deforestation, aquaculture, conversion to agriculture, overexploitation, and coastal development (Valiela et al., 2001). It is estimated that the world is losing 1-3 % of mangrove areas per year (Hamilton & Casey, 2016). In Kenya, at least 40% of the close to 60,000 ha of mangroves were lost and degraded between 1990 and 2005 (GoK, 2017). Whereas the rate of loss has declined in some areas, the remaining mangrove forests are still threatened by illegal harvesting, land encroachment, pollution, and climate change effects (Bosire et al., 2015). The loss of the outer mangrove fringe has been associated with increased shoreline erosion and decreased elevation (Bandeira & Balidy, 2016; Primavera et al., 2011). This increase the vulnerability to coastal hazards and also alters the optimal conditions for mangrove establishment and growth (Primavera et al., 2011; Brooks & Spencer, 2012). At Gazi Bay in Kenya, mangroves have been historically exploited for wood and non-wood resources (Kairo, 1995; Bosire et al., 2003).

In response, widespread restoration initiatives have been launched to re-establish lost mangrove stands, particularly in recent times when ecosystem restoration has become increasingly popular and essential to address climate change and biodiversity decline (Cadier et al., 2020; Airoldi et al., 2021; Gerona-Daga & Salmo III, 2022). The commonly used approaches include natural regeneration, which relies on the spontaneous dispersal of propagules (Bosire et al., 2003) and artificial regeneration which entails the direct planting of propagules or nursery-raised saplings (Kairo et al., 2001; UNEP, 2020; Gerona-Daga & Salmo III, 2022). Whereas these methods are usually fit for mangrove restoration in low-energy areas, they typically underperform in mangrove-lined ecosystems. Success rates of restoration efforts vary from one project to another (Kodikara et al., 2017; Wodehouse & Rayment, 2019; UNEP, 2020) depending on site history, the choice of the species to be planted, and approach used. Low success rates are often reported in areas exposed to high energy and where habitats have been destroyed and ecological conditions altered (Kairo et al., 2001; Otero et al., 2019).

High-energy sites are characterized by strong wave action, high tidal ranges, and strong currents. These conditions significantly impact the success of mangrove restoration efforts. The force of waves and currents wash away mangrove propagules and young seedlings before they have a chance to establish roots (Balke et al., 2011). The low-energy sites are characterized by calmer waters, lower tidal ranges, and minimal wave action (Murray et al., 2018). These conditions are generally more favorable for mangrove restoration. The calmer conditions in low-energy sites provide a more stable environment for mangrove propagules and seedlings to establish and grow (Lewis, 2005).

In Gazi Bay, where this study was based, community efforts and attempts by researchers to rehabilitate high-energy sites using conventional planting methods failed, recording success rates of less than 10%. This failure is attributed to changes in site conditions and exposure to high wave action (Kairo et al., 2001) that dislodge the newly planted seedlings. Poor mangrove performance has been reported elsewhere, where unshielded coastlines are prone to wave actions (Kamali & Hashim, 2011; Schmitt & Duke, 2015). This study aimed to explore alternative restoration techniques that can be embraced to restore lost mangroves in high energy sites. The study assessed the applicability and efficacy of the modified Riley Encasement Methodology initially proposed by Riley & Kent (Riley & Kent, 1999). The findings of this study are important in highlighting the effectiveness and potential replicability of modified Riley Encasement Method in Kenya.

1.2 Statement of the problem

Changes in ecological site conditions due to mangrove degradation pose significant challenges to restoration efforts. Mangroves in Gazi Bay, have undergone degradation and over-exploitation for a long period of time through anthropogenic activities such as harvesting for building poles and firewood (Kairo, 1995; Bosire et al., 2003). For instance, in the 1970s and 1980s, the industrial exploitation of mangrove wood for energy left large contiguous blank areas, some of which have failed to regenerate naturally to date (Dahdouh-Guebas et al., 2004; Kirui et al., 2013). As a result of the deforestation, the area experiences coastal erosion due to large exposure to waves, winds, currents and tides, to the extent of uprooting and washing away coconut trees in the adjacent agricultural farms. This exposes the Gazi community to the risk of flooding and potential loss of land to the sea. Earlier attempts to rehabilitate these high energy sites through direct planting of propagules, nursery raised saplings and wildlings failed. There is thus a need to explore alternative restoration techniques with the hope that if effective could be embraced in mangrove restoration schemes especially in high energy areas.

1.3 Justification of the study

Owing to the changing climate, coastal regions are at a greater risk of experiencing natural hazards including sea level rise, erosion, flooding, risky storms, tsunamis as well as destruction of coastal infrastructure (Wong et al., 2014; Pörtner et al., 2019; IPCC, 2019). These climate-mediated extreme events are often amplified by loss and degradation of natural capital, including mangroves (Hochard et al., 2019). As such, efforts towards potential mitigation and adaptation options including ecosystems restoration are important in minimizing the impacts of such hazards. On a community level, this study aims to empower local conservation groups like Mikoko Pamoja by providing them with alternative restoration skills, thereby enhancing their capacity for restoration. Consequently, this will lead to an increase in their measurable carbon credits. On a broader scale, the re-establishment of mangroves will yield climate and biodiversity advantages, along with other ecosystem services. Moreover, the community will gain increased protection and resilience against climatic changes. Under REM, the planting materials are installed in polyvinyl chloride (PVC) tubing to protect them from being dislodged by waves (Riley & Kent, 1999). The current study varied the diameter of encasements (7 to 10 cm) as well as the material types (bamboo and PVC pipes), meant to shield and support seedlings during the early developmental stages.

1.4 Objectives

1.4.1 General objective

To determine the efficacy of modified Riley Encasement Methods for mangrove restoration in high-energy areas.

1.4.2 Specific objectives

1. To map mangrove cover changes from 1990 to 2020 in high energy eroding shoreline of Gazi Bay requiring restoration.
2. To determine the vegetation and soil characteristics in the high energy degraded mangrove area.
3. To plant mangrove seedlings using bamboo and polyvinyl chloride (PVC) tubing as encasement tools and assess their survival and growth performance.

1.5 Research questions

1. How has mangrove cover changed from 1990 to 2020 in the high energy eroding shoreline area of Gazi Bay?
2. What are the vegetation and soil characteristics in the high-energy, degraded mangrove area?
3. How does the survival and growth performance of the planted mangroves compare among the encasement tools?

CHAPTER TWO

LITERATURE REVIEW

2.1 Mangrove classification and distribution

Mangroves constitute trees and shrubs that have successfully established along tropical and subtropical coastlines (Duke, 1992; Spalding, 2010). They are classified into several genera and species based on their physiological and ecological characteristics. Major mangrove species include Avicenniacea which includes Avicennia species, such as *Avicennia marina* and *Avicennia officinalis* characterized by pencil like pneumatophores, which facilitate gas exchange in waterlogged soils (Ellison et al., 2005); Rhizophoraceae: constituting Rhizophora species, such as *Rhizophora mangle* and *Rhizophora apiculata* with prop roots, which provide structural support and enhance sediment trapping (Tomlinson, 2016); and Sonneratiaceae comprising *Sonneratia alba*, known for its distinctive, large pneumatophores that help stabilize the shoreline and trap sediment (Nagelkerken et al., 2008).

The global distribution of mangroves spans approximately 137,800 km^2 across tropical and subtropical regions (Giri et al., 2011; Spalding et al., 2010). They are mainly found along coastlines in Southeast Asia, Africa, Central and South America and Australia (Alongi, 2008; Spalding et al., 2010; Basha 2018; Kumar et al,. 2021). Kenya's mangroves, covering an estimated 60,000 hectares (UNEP, 2014), are predominantly located along the Indian Ocean coastline. Dominant species among the Kenyan mangrove forests include *Avicennia marina, Rhizophora mucronata,* and *Ceriops tagal*.

2.2 Mangrove habitat and adaptations

Mangrove distribution is heavily influenced by geomorphological and climatic factors such as moisture contents and temperature ranges (Njiru et al., 2022). They thrive well in areas of low energy and tidal activity, where saltwater occasionally mix with freshwater (Tomlinson, 2016). A unique characteristic of the mangrove ecosystems is exhibited in the horizontal distribution of species, referred to as zonation (MacNae, 1969). Some mangroves species are commonly found along the seaward side, where tidal inundations are frequent, while others inhabit the landward side within intertidal areas. Environmental factors such as inundation frequency, temperatures, salinity, precipitation, and oxygen levels have an important role in influencing species association within the mangrove environment (Snedaker, 1989; Saenger, 2002).

Kenyan mangroves exhibit the typical zonation patterns where *Rhizophora mucronata* and *Sonneratia alba* are typically found in the lowest intertidal zones, followed by *Ceriops tagal* and *Avicennia marina* in the mid-intertidal zones. *Lumnitzera racemosa* and *Heritiera littoralis* inhabit the landward side. *Bruguiera gymnorrhiza* does not display distinct zonation but is scattered within the Rhizophora and Ceriops zones. *Avicennia marina* may display dual zonation, occurring in both the seaward and landward sides (MacNae, 1969;. Ruwa, 1993; Kairo, 2001; Dahdouh-Guebas et al., 2004). Adaptations of mangroves to the challenging environment include; presence of pneumatophores to aid in gaseous exchange, mechanisms like salt excretion, accumulation and ultra-filtration to deal with salinity, and viviparous reproduction (Kathiresan & Bingham, 2001; Alongi, 2002; Sobrado, 2005; Tomlinson, 2016).

2.3 Goods and services derived from mangrove ecosystem

According to the comprehensive categories outlined in the Millennium Ecosystem Assessment (2015), mangroves provide provisioning, regulating, cultural, and supporting services through their direct, indirect, or potential utilization (MEA, 2015).

Provisioning services involve the direct extraction of products or goods from ecosystems. Mangroves supply coastal communities with a diverse array of both wood and non-wood forest products (Duke et al., 2014). Wood products encompass building materials like poles and timber, as well as fuel sources such as firewood and charcoal, utilized in both urban and rural settings (GoK, 2017). Additionally, non-wood products harvested from mangroves comprise fish, crabs, shrimp, dyes, tannins, and traditional medicinal resources.

Regulating services refer to the advantages derived from the regulation of ecosystem processes and the buffering capacity of ecosystem services. In the context of climate change mitigation, mangroves perform the crucial function of capturing and storing carbon in both above and below-ground compartments, as well as in sediment organic carbon (Donato et al., 2011;Hamilton & Friess, 2018). Their role as carbon sinks is integral to the attainment of global sustainability objectives and the goals outlined in the Paris Agreement. Additionally, mangroves assist in attenuating wave energy, thereby stabilizing sediments and preventing shoreline erosion (Costanza et al., 2014).

Supporting services encompass functions essential for generating and delivering of other ecosystem services. These include activities such as biodiversity conservation, primary production, and soil formation. Mangroves serve as a vital global habitat and reserve for juvenile fish and plants, while also serving as breeding grounds for a diverse array of fauna species including mollusks, crustaceans, reptiles, mammals, and birds (Spalding et al., 2010; Salem & Mercer, 2012; Vegh et al., 2014).

Cultural services, on the other hand, pertain to intangible and enriching benefits. For instance, they offer opportunities for ecotourism, recreation, aesthetic appreciation, and spiritual contemplation. In areas where mangroves are honored as shrines, the harvesting of trees is strictly prohibited, thereby preserving the natural condition of the forest (Huxham et al., 2018). Moreover, the mangrove ecosystem supports research activities and environmental education, thereby promoting nature studies among students globally.

2.4 The role of mangroves in shoreline and coastal protection

Mangroves play a valuable role in shoreline and coastal protection. They are capable of minimizing wave energy, stabilizing sediment, and facilitating sediment accretion (Alongi, 2008; Lewis III, 2005; Ostling et al., 2009; Gedan et al., 2011; Shepard et al., 2011;Steven et al., 2020). For instance, a 100-meter-wide stretch of mangroves can significantly attenuate wave energy, reducing the height of incoming waves by approximately 66% (Mclvor et al., 2012). This reduction in wave energy is due to the dense root systems and above-ground structures of mangrove trees, which dissipate wave forces through friction and turbulence. Their root system forms a network that stabilizes sediments, preventing erosion (Shepard et al., 2011). The vegetation cover reduces the speed of the flowing water and wind, promoting sediment accretion, which stimulates the production of the below-ground root system (McKee & Cherry, 2009), further improving soil cohesion. The dense mangrove vegetation together with the reduced wind speed contributes to reduced wave vigor (Wolanski, 2006; Day et al., 2007; McIvor et al., 2012; Sanderman et al., 2018), thus protecting the shoreline from wave damage. In the event of sea level rise, mangroves are able to accrete sediments, thereby adapting to the rising sea levels (McKee et al., 2007; Krauss et al., 2014). The role of mangroves in coastal protection was well illustrated in the event of the Indian Ocean Tsunami (IOT) of 2004 where areas with degraded mangroves experienced high losses of life and property compared to areas with intact ecosystems (Dahdouh et al., 2005; Patel et al., 2014). The loss of life was estimated to exceed 200,000 deaths and the cost of property loss was estimated as more than 9.9 billion dollars (Fehr et al., 2006; Hawkes et al., 2007; Athukorala, 2012), which demonstrates the significance of mangroves in protecting the coast and its people. With the current global fluctuating environmental conditions, shoreline protection has become a major concern (Prasad & Kumar, 2014) since shorelines safeguard coastal communities and other ecosystems.

2.5 Coastal hazards and disaster risk reduction through ecosystem-based interventions

Coastal regions are characterized by high dynamism and susceptibility to both anthropogenic and natural influences (Steven et al., 2020). Hazards including erosion, floods, storm surges and land subsidence are likely to be experienced in coastal regions owing to the changing climate and loss of natural ecosystems including mangroves (Jeschonnek et al., 2016; Hochard et al., 2019; Hamza et al., 2022). A study by Hamza et al., (2022) estimated 16% of Kenya coastline as highly exposed to natural hazards; and projecting a 41% escalation if marine ecosystems continue to be degraded. Global projections indicate that the intensity of coastal hazards including storms and cyclones is likely to increase by 1-10% by 2030 -2080 (Kossin et al., 2020; Steven et al., 2020).

In the past, responses to such hazards included construction of protective structures like seawalls and breakwaters (“grey” engineering). However, these kind of interventions are rigid, challenging to maintain, lack a natural component, and contribute to the depletion of natural habitats as well as “coastal squeeze” (Hsu et al., 2008; Martínez et al., 2014; Steven et al., 2020). Consequently, attention has recently shifted to ecosystem-based interventions in responding to coastal hazards (Kitazato et al., 2018). Ecosystem-based interventions involve the management, conservation, and restoration of natural ecosystems to minimize disaster risks and promote sustainability and resilience (Steven et al., 2020; Sudmeier-Rieux et al., 2021; Wickramasinghe, 2021). Such interventions are capable of minimizing disaster risks, moderating impacts of catastrophic events and are considered sustainable compared to grey measures (Gracia et al., 2018; Sudmeier-Rieux et al., 2021; Seddon, 2022).

Researchers of disaster risk reduction have shown increased interest in the opportunities offered by ecosystem-based approaches (Uy & Shaw, 2012; Sudmeier-Rieux et al 2013; Renaud et al., 2013). Recent research has also emphasized on the efficacy of ecosystem functions and services for reducing disasters (Nel et al., 2014; Spalding et al., 2014). Recently, the importance of managing ecosystem as a measure for disaster risk reduction has been acknowledged in the Sendai Framework on the 2030 Agenda for Sustainable Development (Wahlström, 2015; Sudmeier-Rieux et al., 2021). Ecosystem Based Disaster Risk Reduction measures also largely contribute to effective implementation of recovery interventions after a disaster (Kitazato et al., 2018; Wickramasinghe, 2021). Mangrove ecosystem has been used as a case scenario in describing the ability of natural ecosystems in minimizing the impacts of natural hazards as mangroves provide a natural defense against wave, storms, and surges (Kitazato et al., 2018; Indarsih & Masruri, 2019; Quitain & Parayno, 2021).

2.6 Mangrove losses and implications

Mangroves have been listed among the highly threatened ecosystems (Polidoro et al., 2010). Recent estimates indicate an annual mangrove loss of 1% to 2% (Hamilton & Casey, 2016) occasioned by both natural and anthropogenic factors (Carugati et al., 2018). Approximately, 40% of mangroves in Kenya have been lost in the last four decades (GoK, 2017). The remaining mangroves are further threated by the global climate change which is likely to lead to sea level rise (Bosire et al., 2015).

Human activities such as unsustainable wood harvesting, pollution, sewage release to mangrove environment, conversion of mangrove ecosystem to agricultural land, aquaculture, and unplanned urban development have led to about 35% to 50 % mangrove cover reductions in the last half century (Giri et al., 2011; Thinh & Hens, 2017; Newton et al., 2020). Climate change impacts including rising sea levels, coastal flooding, erosion, sedimentation, fluctuating rainfall patterns, and temperature extremes, storm surges and other associated extreme events further exacerbate the pressure on mangrove forests globally (Gilman et al., 2008; Bosire, 2010; Bosire et al., 2012).

In the Western Indian Ocean (WIO) region, mangrove losses are mainly occasioned by unsustainable wood harvesting for fuel, timber and building (Bosire et al., 2015; Mungai et al., 2019) as well as the conversion of mangrove areas for other uses such as aquaculture, agriculture, and housing (Bosire et al., 2015). Currently, mangroves at Gazi Bay are threatened by the increasing shoreline erosion threatening the integrity of associated ecosystems such as seagrass beds and corals from increased sedimentation (Ndirangu, 2016). According to Zimmer et al (2022), there are about 800,000 ha of potential mangrove restoration sites around the world. Therefore, there is a need to explore alternative approaches for mangrove restoration in such challenging areas.

Mangrove loss risks the release of huge amounts of carbon stored in mangrove ecosystems back into the atmosphere further impacting climate (Murray et al., 2011; Pendleton et al., 2012). Degradation of mangrove forests may also have consequences such as decreased forest cover, changes in forest structure, alteration of species composition, reduction in fisheries production, intensified coastal erosion, unpredictable weather patterns and deprivation of other associated ecosystem goods and services (Abuodha & Kairo, 2001; Kairo et al., 2002; Bosire et al., 2006; Donato et al., 2011; Giri et al., 2011; Zhang et al., 2012).

2.7 Mangroves cover changes

Recently, studies on global mangrove forest cover has reported reductions in spatial extent, as noted by Kirui et al.(2013), Bosire et al.(2014), and Hamilton & Casey (2016). Particularly in Africa, a decrease in mangrove cover has been a documented, with Central and West Africa regions experiencing approximately 20-30% loss over the past 25 years (Feka & Ajonina, 2011). In Kenya, annual mangrove cover loss of about 1% has been attributed to human activities (Kirui et al., 2013). About 1139 ha of mangroves was lost between 1996 and 2016 (Erftemeijer et al., 2022), though, about 578 ha have been recovered between 2016 and 2020. In areas like Tudor and Mwache Creeks in Mombasa, about 80% of mangrove loss reported is linked to land use changes from 1992 to 2009 (Bosire et al., 2014). However, some other areas in Kenya like Vanga, Kilifi, and Ngomeni have experienced significant increase in mangrove cover between 2000 and 2019. This gain is accredited to natural regeneration after sedimentation, active restoration efforts, and the enactment of mangrove conservation measures (Mazi & Kirui, 2021).

Mangrove cover changes directly impact the provision of goods and services associated with their ecosystem (Donato et al., 2011; McIvor et al., 2015; Tran & Fischer, 2017) . Decrease in mangrove cover negatively impact the local and national economy as depicted by the scarcity of wood products such as firewood and building poles (Kairo et al., 2002), reduced fish production (Barbier et al., 2011), and shoreline changes (McIvor et al., 2015; Menéndez et al., 2020). Understanding the extent of changes in mangrove cover and where the changes are occurring at higher rates is important in guiding on where to direct conservation and restoration efforts (Lewis III, 2005).

2.8 Mangrove restoration initiatives

Mangrove restoration projects have been ongoing globally with the realization of the threats posed by climate change and knowledge that mangrove forests help regulate climate by sequestering carbon dioxide (CO_2) (Kauffman & Donato, 2012; Sidik et al., 2018; Sharma et al., 2019). Inclusion of communities through formation of Community Forest Associations (CFAs) has been one of the popular practice embraced to enhance restoration and conservation of mangrove forest (Frank et al., 2017). Other several initiatives including government driven, Non-Government (NGOs), community driven and mixed approaches (UNEP, 2020) have been embraced in different parts of the world. Despite the vast restoration initiatives and resources channeled towards mangrove restoration, success rates are particularly low

(Wibisono & Suryadiputra, 2006; Bosire et al., 2008) and only a few success cases can be quoted. For instance, in Mauritius, government driven initiative for mangrove restoration has worked well with close to 95% success rates (UNEP, 2020). In Madagascar and Mozambique, NGOs and community driven initiative have also been effective in restoration and management of mangroves (UNEP, 2020).

In Kenya, the mixed approach is evident where government agencies, NGOs, funding organizations, and local communities collaborate in the management and restoration of mangroves (UNEP, 2020). Improved community awareness on the importance of mangroves has particularly empowered local communities at Gazi Bay to restore and conserve mangroves through the sale of carbon credits.

2.9 Methods of mangrove restoration

The most popular approaches of mangrove restoration are natural regeneration (Bosire et al., 2003) and artificial regeneration which entails use of propagules or nursery-raised saplings (Kairo et al., 2001; UNEP, 2020). Mangroves ecosystems are very dynamic (Alongi, 2013; Noor et al., 2015; Mahmood et al., 2021) and over-harvesting of wood products and other disturbances leads to habitat change that may impact natural regeneration (Kirui et al., 2008). The success of natural regeneration is influenced by both biological and physical factors including tidal action, condition of the forest, propagule/seed availability and soil stability (UNEP, 2020).

Natural regeneration largely relies on the spontaneous dispersal of propagules, making it suitable for areas with an abundant supply of propagules and unchanged hydrology. This approach is cost-effective, as it requires no labor, and seedlings tend to establish more vigorously with minimal soil disturbance. The resulting forest closely resembles the native mangrove species mixture. However, challenges such as poor establishment due to waves disturbance, scarcity of propagules in the absence of mother trees, and less control over the spacing and composition of seedlings are experienced (UNEP, 2020). In areas where natural regeneration fails, other interventions need to be incorporated.

Artificial regeneration encompasses human interventions such as direct planting in cases where propagules supply is limited and tidal regimes altered. This approach allows control over species mixture, distribution as well as diseases control. It also offers employment opportunities during nursery establishment and out-planting. Additionally, it serves as a platform for training, and enhances community sense of belonging and ownership. However, this approach is sometimes costly especially in areas with altered hydrological regimes. Other possible shortcomings include; probable introduction of inappropriate species, community conflicts, and the risk of pest infestation mostly in single-species plantations (UNEP, 2020).

2.10 Factors influencing mangrove restoration success

A myriad of factors including; changes in site conditions, species mismatch, mangrove dynamism, inappropriate restoration methods, inadequate site assessment, poor coordination among stakeholders and lack of intensive care after planting are among the factors associated with failures of mangrove restoration (UNEP, 2020). Recently, factors such as high wave energy, inappropriate topography and hydrological changes have also been associated with the failures of mangrove restoration (Kodikara et al., 2017; Kibler et al., 2019). Hydrodynamic forces including wave energy, currents and tidal action have of late become high in shorelines owing to the rising sea levels thereby altering shoreline dynamics as evidenced by the increased erosion (Ndirangu, 2016). Such high energies exceeds the establishment threshold of the planted mangrove seedlings and saplings (Balke et al., 2011). In coastal regions dominated by wave activity, mangroves are limited to the protected bays since higher energy near the shore stimulates sediment deposition forming a barrier, responsible for mangrove dieback (Raw et al., 2006). A study by Kairo et al (2001) demonstrated that the survival rates of mangrove seedlings planted in areas exposed to high energy range from 0 - 10%.

2.11 Indicators of success in mangrove restoration projects

Various standards are employed to assess the efficacy of restoration efforts (Le et al., 2012; Wortley et al., 2013). Nevertheless, ongoing discussions aiming to establish a unified measure for evaluating and gauging the success of restoration initiatives continue to provoke debate (Wortley et al., 2013). The Society for Ecological Restoration International (SERI) formulated a set of nine fundamental characteristics, which serve as guidelines and metrics for judging the success of a restored ecosystem in comparison to its natural counterpart (SER, 2004). These attributes are categorized into vegetation structure, ecological processes, and species diversity (Ruiz-Jaén & Aide, 2005).

The evaluation of vegetation structure typically involves assessing indicators such as tree height, stem diameter, stand density, biomass, canopy cover, and natural regeneration, which in turn inform predictions about plant succession. Diversity, on the other hand, encompasses the abundance and variety of floral and faunal species across different trophic levels (Wortley et al., 2013). Ecological processes entail evaluating factors like reproduction or dispersal, soil development, nutrient cycling, and biological interactions (SER, 2004; Ruiz-Jaén & Aide, 2005).

However, a scrutiny of how to gauge restoration success by Ruiz-Jaén & Aide, (2005) indicates that ecological metrics have not received as much attention compared to vegetation structure and diversity. This disparity is attributed to the longer time frame and greater resource demands associated with measuring ecological processes. Moreover, during the initial establishment phase of a restoration program, survival rates and the spatial extent of restored sites emerge as significant factors (Le et al., 2012).

2.12 Impacts of overexploitation on mangrove structure and composition

The overexploitation of mangrove forests has detrimental effects on the species composition and structural complexity of the forest, potentially hindering its functioning and regeneration. Deforestation and excessive exploitation of mangrove forests have been shown to impact the composition of mangrove flora and fauna, as documented in various studies conducted in Kenya and other regions (Abuodha & Kairo, 2001 ;Dahdouh-Guebas et al., 2004; Skilleter & Warren, 2000). Overexploited forests often exhibit stunted growth and reduced forest cover, a phenomenon observed in locations such as the Pacific Island of Kosrae (Allen et al., 2001), as well as in Kenya (Kairo et al., 2002; Dahdouh-Guebas et al., 2000). Degradation of mangrove ecosystems affects forest functioning by diminishing primary productivity (Kihia et al., 2010). Physical disturbances by humans also decrease the prevalence of commercially valuable species such as *Rhizophora mucronata* and *Bruguiera gymnorrhiza*, often leading to their replacement by less valuable species like *Avicennia marina* and *Ceriops tagal* in disturbed areas. Various structural attributes, including stem density, height, basal area, biomass, and carbon stocks, exhibit significant variations, with diminished values typically observed in exploited regions (Rasquinha & Mishra, 2021).

2.13 History of mangrove exploitation and management interventions in Gazi Bay

Historically, mangroves in Gazi Bay have been exploited for wood and non-wood resources (Kairo, 1995; Bosire et al., 2003). Reforestation program to rehabilitate degraded mangrove areas in Gazi was initiated in 1991 by the Kenya Marine and Fisheries Research Institute (KMFRI). In 1994, more than 200,000 seedlings of different mangrove species were planted (Kairo, 1995) and by 2004, about 100 ha of the deforested areas had been replanted. In 2013, a joint management program involving communities, forest agency and donors was initiated through a carbon offset program, Mikoko Pamoja (Huff & Tonui, 2017). Under Mikoko Pamoja, the Gazi community restores and protects mangroves through the sale of carbon credits in the Voluntary Carbon Market (VCM); and is the world's first community-led conservation project funded by carbon credits.

Gazi community's efforts in restoring and conserving mangroves forest have further attracted numerous donors support through coordination by Kenya Marine and Fisheries Research Institute (KMFRI). This has supported the construction of a boardwalk named 'Gazi Women Boardwalk' which is utilized as an ecotourism project, alternative livelihood activity and incentive scheme to mangrove conservation by the local community. Revenue generated from the carbon off-setting scheme and ecotourism services is used to maintain the boardwalk as well as in supporting community development projects and mangrove conservation activities (UNEP, 2020; Runya et al., 2022).

Despite the enormous efforts towards mangrove restoration in Gazi Bay, over the years, some sites failed to recover, with very low success rates being reported. This failure is attributed to changes in site conditions and exposure to high wave action (Kairo et al., 2001). As such, the current study focused on improving restoration success of mangroves replanted in high energy degraded site of the bay.

2.14 Eco-engineering in mangrove restoration

The traditional/conventional methods are sometimes less effective for mangrove restoration in high-energy areas and where ecological conditions have been altered. These methods provide minimal protection against waves and other external stressors (Krumholz & Jadot, 2009). In such cases, eco-engineering approaches may be suitable alternatives. In the context of ecosystems restoration, eco-engineering may be referred to as the incorporation of the ecological principles and engineering skills to restore, improve, or create mangrove ecosystems (Chapman & Underwood, 2011;Elliott et al., 2016). It is a less commonly used approach and is normally applied in the event that conventional mangrove restoration methods prove ineffective. The engineering practices may involve the use of artificial structures to stabilize the coastline and offer protection to the young mangrove saplings. The ecological aspects entail identification of restoration site, assessment of the site, selection of species to be planted, execution of restoration plan, monitoring and subsequent maintenance (Bhakta et al., 2016).

In the past, marine ecosystems management goals have been realized through hard engineering practices which are rigid, challenging to maintain, lack a natural component, and contribute to the depletion of natural habitats" (Hsu, et al., 2008; Martínez et al., 2014). Ecological engineering aims to address such challenges through re-designing the structures to be multifunctional, benefiting both humans and nature (Dafforn et al., 2015). The application for eco- or 'green' engineering is recommended in the scenarios where hard structures are inevitable, to enhance positive ecological impacts (Chapman & Underwood, 2011). In addition, the eco-engineering practices available for marine infrastructure and developments offer a spectrum of options for implementation by managers, dependent upon local conditions and community expectations. For instance, in the event that coastal and shoreline protection are anticipated, it is worthwhile to consider interventions such as restoration of habitats including mangroves and salt marshes that offer natural defense against amplified waves and storms. Such approaches are ecologically sustainable (Gedan et al., 2011; Hoang Tri et al., 1998) as opposed to 'soft' or 'hard' engineering approaches.

2.15 Case studies of eco-engineering in mangrove restoration

Eco-engineering approaches have been increasingly applied in mangrove restoration to enhance ecosystem resilience and provide multiple benefits such as coastal protection, biodiversity conservation, and livelihood support (Whelchel et al., 2018). For instance, in Australia, at the Sydney Harbour, the inclusion of flowerpots to the seawalls was noted to increase species diversity. Such enhancements help maintain water at low tide, and diversify the array of habitats present on seawalls, which positively impact biodiversity (Browne & Chapman, 2011; Browne & Chapman, 2014). In Thailand, the use of sediment trapping devices, such as bamboo fences and brushwood dams, has been effective in mangrove restoration efforts. These structures reduce wave action and enhance sediment accumulation, creating conducive environments for mangrove seedlings to establish and grow (Winterwerp et al., 2013). In the Philippines, one notable project is in the province of Bohol, where permeable bamboo structures were installed along eroded shorelines to reduce wave energy and trap sediments. This facilitated the natural regeneration of mangroves, which provided critical habitat for marine life and protection against storm surges (Primavera & Esteban, 2008).

In Florida, USA, the concept of living shorelines has been adopted to restore mangrove habitats. This approach involves the strategic placement of natural materials such as oyster shells, coir logs, and native vegetation to create stable, erosion-resistant shorelines. These structures not only support mangrove growth but also provide habitats for various marine species and improve water quality (Bilkovic et al., 2017). In the United States, particularly in Louisiana, a combination of green (mangroves) and grey (concrete structures) infrastructure has been used to restore coastal areas affected by erosion and hurricanes. Mangrove planting combined with breakwaters has proven effective in reducing wave energy, trapping sediments, and rebuilding the coastline (Gedan et al., 2011). Some other projects are sought to boost tourism and recreation activities, or reduce the risks of flooding, and offer climate change adaptation options (Whelchel et al., 2018).

2.16 Limitations and opportunities for Eco-Engineering

The application of eco-engineering practices require a higher level of confidence and certainty about natural infrastructure as being able to reduce risk and offer the needed site specific solution (Whelchel et al., 2018). The practice also needs not to conflict with the existing or future land use plans. The utilization of eco-engineering methods is notably limited, with constrained tools (Bouma et al., 2014). Funding opportunities also are mostly available for conventional environmental protection and restoration as opposed to eco-engineering projects (McCreless & Beck, 2016). The absence of established standards and specific guidelines impedes their wider adoption. Comprehensive case studies and sustained long-term monitoring are essential to reinforce confidence and facilitate comparability with conventional engineering approaches for coastal areas and other ecosystems (Whelchel et al., 2018).

2.17 The Riley Encasement Method (REM)

The REM was invented for the purpose of establishing mangroves in high energy shorelines so as to overcome the limitations of conventional planting methods (Riley & Kent, 1999). In its application, REM incorporates the use of encasements to shield seedlings, integrating elements of eco-engineering. It entails the isolation of the individual seedling in an encasing vessel and the subsequent adaptation to the site conditions. Isolation creates an artificial environment favorable for early development of plants protecting them from strong winds, wave activity, and other external disturbances (Riley & Kent, 1999). The REM has been successfully practiced in the coastline of Hawaii and Florida (Walsh, 1967; Kent & Lin, 1999; Riley & Kent, 1999).

Initially, REM was designed to utilize specific materials but over time due to cost of production, availability of the materials and intensity of use, there is increasing modifications of the method to accommodate changing environmental scenarios. For example, Riley and Kent in their experiment used plastic materials but currently researchers have incorporated use of bamboo piping’s as encasement vessels. Appropriate sourcing of the seedlings to be planted, encasement and seedling elevation, depth of encasement, and encasement length are vital to ensure success (Riley & Kent, 1999) and should be considered.

In the current study, PVC pipes and bamboo pipes were utilized as the encasing vessels to provide protection during the early developmental stages (Riley & Kent, 1999). This physical protection serves to mitigate the impacts of wave and wind action. Importantly, the PVC pipes are intended to be removed once the saplings have established (Clarke & Johns, 2002).

2.17.1 Specifications of the Riley Encasement Method

To enhance restoration success, it is imperative to adhere to REM specifications, which encompass various factors such as the source of the seedling, encasement technique, seedling elevation, depth and length of encasement, as well as encasement alleviation (Riley & Kent, 1999). The encasement device should possess vertical rigidity to facilitate easy penetration or anchorage into bottom sediments, while also maintaining lateral flexibility to support unrestricted plant growth, moisture exchange, drainage, and adequate light penetration. The source of seedlings significantly influences their survival post-planting. For optimal survival rates, it is advisable to procure propagules directly from mother trees during the natural abscission period. Additionally, the elevation of the encasement device is crucial, particularly for mangroves, given their intertidal habitat. Aligning the encasement elevation with the natural recruitment zone of the intended mangrove species is essential for successful establishment. In areas with secured shorelines, lower encasement elevations are vital. However, in regions of natural regeneration, seedling elevation should match that of the surrounding sediment surface. When encasement is positioned at a lower elevation, seedlings should be placed accordingly to mimic natural recruitment levels (Riley & Kent, 1999).

The depth at which the encasement device is inserted is primarily determined by the stability of the sediment being utilized. Optimal encasement depth is crucial for providing ongoing support to the developing plant. Furthermore, the length of the encasement determines the level of protection afforded to the seedling against strong waves and external damage. Encasement alleviation plays a vital role in enhancing adaptation rates by promoting the extension of aerial roots into the surrounding surface (Riley & Kent, 1999).

CHAPTER THREE

MATERIALS AND METHODS

3.1 Study site description

The study was carried out in the south coast of Kenya at Gazi Bay, located between 4°24′S to 4°30′S latitude and 39°28′E to 39°32′E longitude, about 55km from Mombasa (Figure 1). The bay occurs close to the Diani-Chale Peninsula, a renowned tourist destination famous for its white sandy beaches. Two villages, Gazi and Makongeni are adjacent to the bay. Gazi village occurring on the northern shore of the bay is the primary settlement in the area and serves as a base for many conservation and research activities. Makongeni village, situated to the west of Gazi Bay forms another community that interacts closely with the bay's resources (Figure 1).

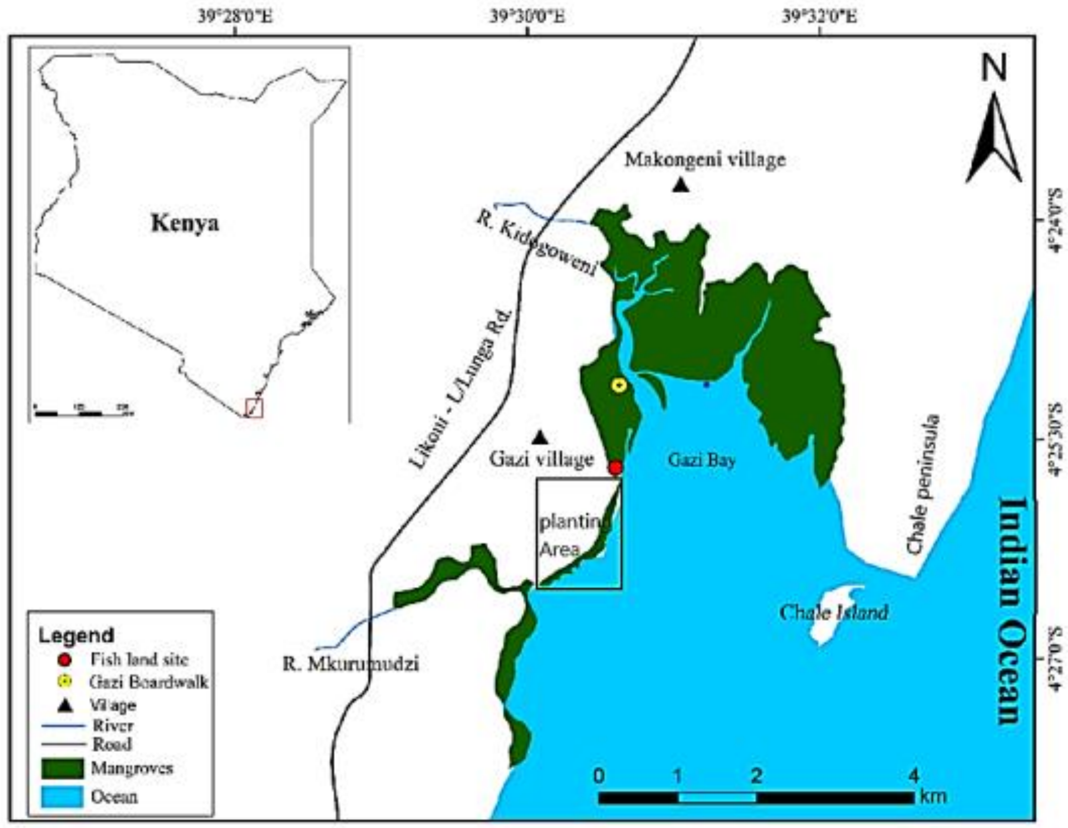

Figure 1: Map of the study site showing planting area (modified from Runya et al., 2022).

The bay covers about 18 km^2 (Coppejans et al., 1992) and is distinguished by the eastern and western creeks, fringing mangroves, seagrass beds, and a long intertidal area. Two seasonal rivers, Kidogoweni and Mkurumudzi, drain into the western creek and provide freshwater input into the bay while groundwater seepage happens in a few points (Tack & Polk, 1999). The bay connects to the Indian Ocean through a relatively broad opening (3500 m), and is usually shallow, with an average depth of 5 m at the entrance (Kitheka, 1996). It is protected to the east by Chale Peninsula and to the south by a fringing coral reef. These reefs that occur in dispersed clusters, serve as habitats for various marine fauna including mollusks, crustaceans, and fish and are thus important in sustaining community livelihoods through tourism and fisheries (Obura, 2012).

3.2 The climate and mangrove of Gazi Bay

The climate of Gazi Bay may be classified as tropical wet/dry according to Koppen classification (Peel et al., 2007). The wet season is associated with the southeast monsoons. It is characterized by heavy rains and rough seas, and usually occurs from March to August. Dry season follows from November to March, associated with the northeast monsoon winds and calm seas (McClanahan, 1988). Annual total rainfall in Gazi Bay ranges from 1000 mm to 1,600 mm, with temperature ranging from 19 to 34^{0}C throughout the year (UNEP,1998). Humidity is high, and averages about 80% all year round (Kitheka, 1996).

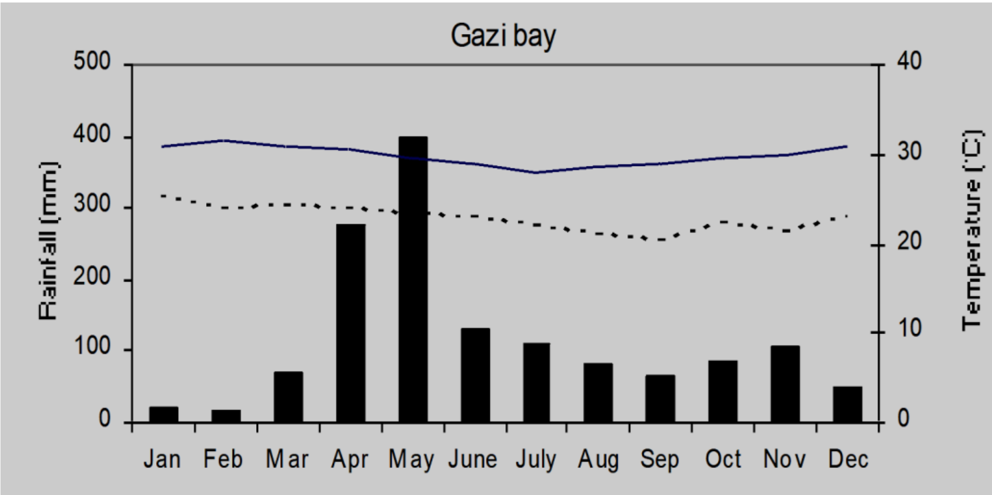

Figure 2: Rainfall and temperature regimes throughout the year. Source: https://weatherandclimate.com.

There are about 700 ha of mangroves in Gazi bay; represented by nine species (Kirui et al., 2006; Neukermans et al., 2008). The most important mangrove species in the bay are *Rhizophora mucronata*, *Ceriops tagal* and *Avicennia marina*, occupying more than 80% of the forest formation (Kairu et al., 2021). Gazi mangroves exhibit a zonation pattern that is similar to other mangrove forests in Kenya. The seaward side is occupied by *Sonneratia alba*. This is followed by *Rhizophora mucronata* - *Bruguiera gymnorrhiza* in the mid-zone and the *Ceriops tagal*, *Avicennia marina* and *Lumnitzera racemosa* on the landward side (Kairo, 1995). The community in the villages benefits from sale of carbon credits from the mangrove forest, which has positively impacted their livelihoods.

3.3 Data collection

Ethical considerations were not applicable in this study because it did not involve human or animal subjects, or sensitive environmental impacts that would typically require ethical review. As such, there were no ethical issues related to consent, welfare, or privacy that needed to be addressed.

3.3.1 Mangrove cover changes in high-energy site of Gazi Bay

The Ground Control Points (GCP) were derived from Global Positioning System (GPS) coordinates obtained during fieldwork. Landsat images from 1990 to 2020, corresponding to the GPS coordinates, were extracted from the United States Geological Survey (USGS) website (https://www.glovis.usgs.gov/) for Landsat satellites 5, 7, and 8 (path 165, rows 61 and 62). These images were carefully selected based on quality and absence of clouds. Geo-referencing of both Landsat images and Ground Control Points (GCP) to the World Geodetic System (WGS) 1984 was performed, followed by registration to the local Universal Transverse Mercator (UTM) coordinate system using ArcGIS geo-referencing tools.

To enhance image quality, the image analysis window in ArcMap was utilized. Geometric correction was executed to refine geo-location to a Root Mean Square (RMS) of 0.5 of a pixel, aligning with the recommendation of Ghosh et al. (2016). The area of interest focused on mangrove cover, and the corrected images were subset and clipped to include regions where mangroves are likely to be found. In the image analysis, the ESRI ArcGIS 10.5 software employed both the Supervised

Maximum Likelihood Classifier (SMLC) and Iso cluster Unsupervised Classification (IUC) algorithms. To identify changes in mangrove forest cover from 1990 to 2020, a post-classification technique (Nearest Neighbor and Boundary Clean) was applied. This method, recognized as a common approach in change detection (FAO, 1994; Giri et al., 2015; Giri, 2016), provided 'from–to' change information by comparing mangrove coverage across different years.

3.3.2 Vegetation surveys

Systematic sampling design was adopted in the study. Square quadrats of $100m^2$ were established along 50 m belt transects established perpendicular to the shoreline at an interval of 20 m. In the quadrats, all mangroves with a stem diameter at breast height (dbh) ≥ 2.5 were identified, counted, and recorded. Vegetation attributes including tree height (m) and dbh (cm) were measured using clinometer and forest caliper respectively. Diameter at breast height measurements for all species were taken at 130 cm above the ground (Cintron & Schaeffer Novelli, 1984), except for *Rhiziphora mucronata* in which they were taken at 30 cm above the highest grounded prop root (Komiyama et al., 2005), as it is complex in structure (Dahdouh-Guebas & Koedam, 2006). From the field data, the following were derived; tree basal area (m^2/ha), stand density (stems/ha), and importance value (IV); following procedures discussed in (Kauffman & Donato, 2012) as well as (Kershaw et al,. 2016).

$$\text{Basal Area } (\text{m}^2\text{ha}^{-1}) = \frac{\text{Sum of Cross-sectional Area}(\frac{\pi \text{DBH}^2}{4})}{\text{Plot Area}(\text{m}^2)} \times 10{,}000 \qquad \textbf{Eqn. 1}$$

$$\text{Stem Density (stems ha}^{-1}) = \frac{\text{Number of Stems in Plot}}{\text{Plot Area (m}^2)} \times 10{,}000 \qquad \textbf{Eqn. 2}$$

$$\text{Relative Density } = \frac{\text{Number of Individuals of a Species}}{\text{Total Number of Individuals}} \times 100 \qquad \textbf{Eqn. 3}$$

$$\text{Relative Frequency} = \frac{\text{Total number of quadrats in which species occurred}}{\text{Total number of quadrats occupied by all species}} \times 100 \qquad \textbf{Eqn. 4}$$

$$\text{Relative Dominance} = \frac{\text{Total Basal Area of a Species}}{\text{Basal Area of all Species}} \times 100 \qquad \textbf{Eqn. 5}$$

$$\text{IV } = \text{ Relative Density } + \text{ Relative Frequency } + \text{ Relative Dominance} \qquad \textbf{Eqn. 6}$$

3.3.3 Sediment sampling and processing

Within the $100m^2$ quadrats established for vegetation assessment, two sediment cores were collected using a 7.0 cm diameter half-arc soil corer. The soil corer was vertically pushed into the soil until a maximum depth was reached and pulled out gently. Each core was partitioned using a ruler into sections representative of various depths (0-15 cm, 15-30 cm, 30-50 cm, and 50-100 cm). Sub-samples measuring 5 cm were collected from midpoints of the sections as follows; 5-10, 20-25, 37.5-42.5 and 72.5-77.5 cm (Kauffman & Donato, 2012). The samples were then put in labeled containers and transported to the laboratory and stored at 4^oC awaiting analysis of particle size, bulk density, and sediment organic matter.

3.3.4. The determination of sediment particle sizes

The dry sieve method was used to determine the distribution of soil grain sizes. A standard weight of 100g of the dried sample was measured, homogenized and then passed through a series of sieves ranging from 2 mm to 38 µm mesh-size. Subsequent statistical analysis was done following (Folk & Ward, 1957) method on GRADISTAT computer program to determine the percentage proportions of silt, clay and sand (Blott & Pye, 2001).

3.3.5 The determination of bulk density and sediment organic matter

Soil bulk density is considered as the soil's dry weight per unit volume and it indicates the soil compaction. To calculate bulk density, samples were placed in pre-weighed aluminum foils and oven dried at 60^oc until a constant dry weight, weighed and recorded. The bulk density was calculated by dividing the mass of the oven-dried sample by the volume of the sample as illustrated in the equation below (Howard et al., 2014).

$$\textbf{Bulk Density (g/cm}^3\textbf{)} = \frac{\text{Weight of oven-dried sample (g)}}{\text{Volume of the sample (cm}^3\text{)}} \qquad \textbf{Eqn. 7}$$

Where,

$$\textbf{Volume} = \text{cross-sectional area of the corer } \left(\pi r^2\right) \times \text{height of the sub-sample} \quad \textbf{Eqn. 8}$$

To determine soil organic matter (SOM), loss-on-ignition (LOI) method was used. It is a semi-quantitative method in which soil organic matter is lost in the process of combustion in a furnace. To maximize the efficacy of the results, samples used in computing for bulky density were used. The dried samples were homogenized and ground using an electronic grinder machine for ten (10) minutes and then sieved to separate the fine sediment from debris. Samples were then combusted at 450^0C in an induction furnace for 8 hours in three replicates of 5.0 g each, after which they were cooled in a desiccator and weighed after combustion. Percentage soil organic matter (SOM) was then calculated using the equation:

$$\textbf{SOM (\%)} = \frac{\text{Mass before combustion (g)- Mass after combustion(g)}}{\text{Mass before combustion (g)}} \times 100 \qquad \textbf{Eqn. 9}$$

Soil organic matter consists of many nutrients including hydrogen, carbon, nitrogen, oxygen and sulfur (Howard et al., 2014). An empirical equation relating SOM% to SOC% in mangroves ecosystem was used to estimate Soil Organic Carbon (SOC) from SOM (Kauffman et al., 2011) as follows:

$$\textbf{\%C}_{\textbf{org}} = 0.415 \times \%\ \text{SOM} + 2.89. \qquad \textbf{Eqn.10}$$

The constant 0.415 is the conversion factor that translates the percentage SOM into an estimate of the percentage of soil organic carbon (% C_{org}). It suggests that for every unit increase in % SOM, the % C_{org} increases by 0.415 units. 2.89 is the baseline adjustment It adjusts the % C_{org} estimate to account for baseline organic carbon content that might not be directly proportional to % SOM.

3.3.6 Experimental mangrove planting

3.3.6.1 Site selection

The choice of the planting site was motivated by both natural and socio-ecological factors. In terms of natural suitability, the site was selected due to the previous presence of mangrove vegetation. Additionally, there were signs of secondary growth and limited vegetation, featuring a few noteworthy species. The location is also characterized by erosion due to exposure to strong waves, currents, and winds. Regarding socio-ecological suitability, the community expects improved protection against coastal hazards, such as flooding, through the rehabilitation of mangroves. Majority of the community in Gazi also supports the idea of replanting mangroves to control shoreline change in the designated area. Mangrove rehabilitation efforts are anticipated to align harmoniously with both current and future land use and development needs in the area. The site was designated for annual planting by communities involved in the carbon offset scheme, Mikoko Pamoja. Based on the technical specifications, the community is expected to plant 4000 new mangrove seedlings each year in the degraded areas of Gazi Bay. Therefore, the current restoration experiment aligns with the community needs for mangrove rehabilitation, aiming to yield climate-related benefits.

3.3.6.2 Encasements and propagules preparation

Using an electric driller, the 3” and 4” PVC piping and bamboo piping were drilled to make holes along the length for allowing water to flush in and out. A vertical slit was then made in the PVC pipes to allow easy removal when mangroves are fully established. The mature propagules of *Rhizophora mucronata* were collected from the forest floor or by shaking the mother plant and picking the falls. To initiate rooting, the propagules were spread on top of moistened sawdust in a dark room and covered with a wet heavy blanket (Figure 3). Propagules which did not root within 7 days were considered non-viable and discarded. The viable propagules were assigned to treatments.

Figure 3: (a) Propagules spread on top of moist sawdust and (b) propagules kept in a dark room.

3.3.6.3 Determining ground elevation

A hose pipe technique was used to establish the topographic level of the reference site for the experimental setup. During the highest tide, the highest watermark was identified as the reference point against which ground elevation was measured using a line level during the lowest tide. Wooden poles were strategically placed into the ground, one at the highest watermark and another at the lowest watermark. A cord was affixed to one end of the pole and stretched to the other end. Using the hose pipe, adjustments were made to the cord's position on the pole until a level was achieved. Subsequently, the height from the ground to the cord was measured and documented at both points, as well as at intermediate points along the way, until the entire area was surveyed. Figure 4 shows the results of this activity.

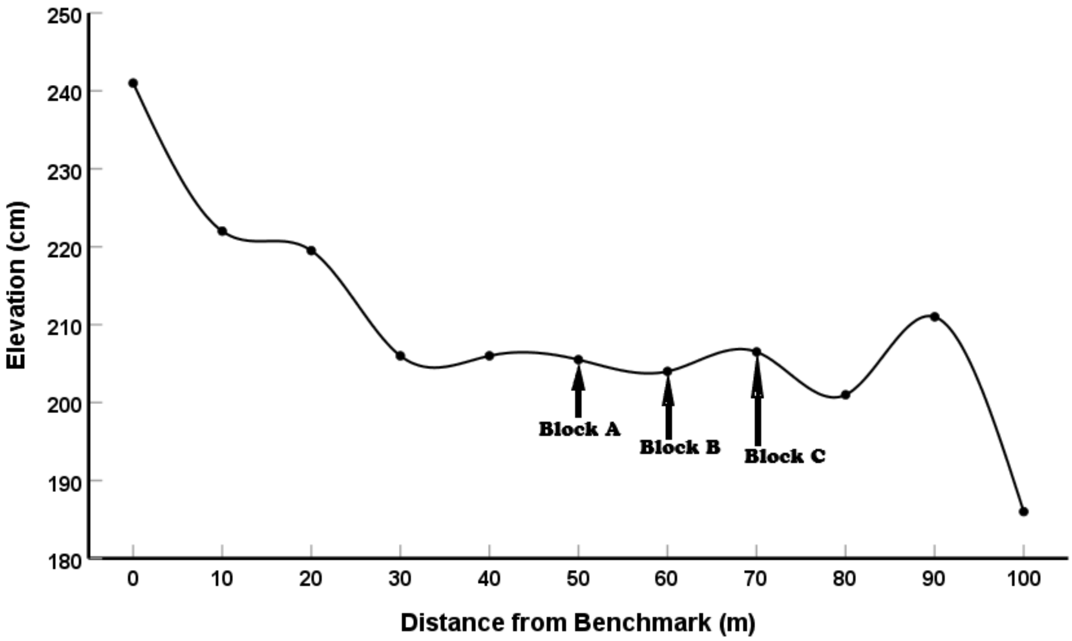

Figure 4: Planting site ground elevation measurements in relation to the mean sea level (MSL).

3.3.6.4 Experimental Design

The randomized complete block design (RCBD) was used in setting up the experiment. Three rectangular blocks measuring 5.5 m by 7 m were established perpendicular to the shore (Figure 6). Treatments that included 3” PVC (7 cm in diameter), 4” PVC (10 cm in diameter), bamboo tubing, and control were randomized separately for each block using the RAND function in the Excel 2016 package. On all block pits were dug at a spacing of 50 cm for encasement installation as the bedrock was shallow, making it impossible to directly push encasements into the ground. The planting spacing was kept at a minimum distance of 50 cm because the site faces strong waves. Closer seedling spacing is recommended for such sites to help withstand the impact of waves (Melana et al., 2000; Change, 2018). After establishment, if all seedlings are growing well, they could be transplanted to other areas with gaps (Change, 2018). After digging pits, the encasements were securely installed in the ground. These encasements were then cut to a height of 50 cm, matching the elevation of the nearest *Rhizophora mucronata* mangroves at the planting site Kairo (1995). This ensures that the newly planted seedlings are at the same elevation as the existing mangroves, promoting optimal growth conditions by mimicking the natural environment. The pipes were then filled with sufficient mangrove sediments to set the seedling at an elevation consistent with the mean-high-water mark (Figure 5). In each encasement, one propagule was planted. Every block contained 15 rows at spacing of 50 cm and every line contained 3 units of each of the treatments, which resulted in 45 units of every treatment in a block. This added up to a total of 180 propagules in every block and a total of 540 propagules in all blocks (Figure 6).

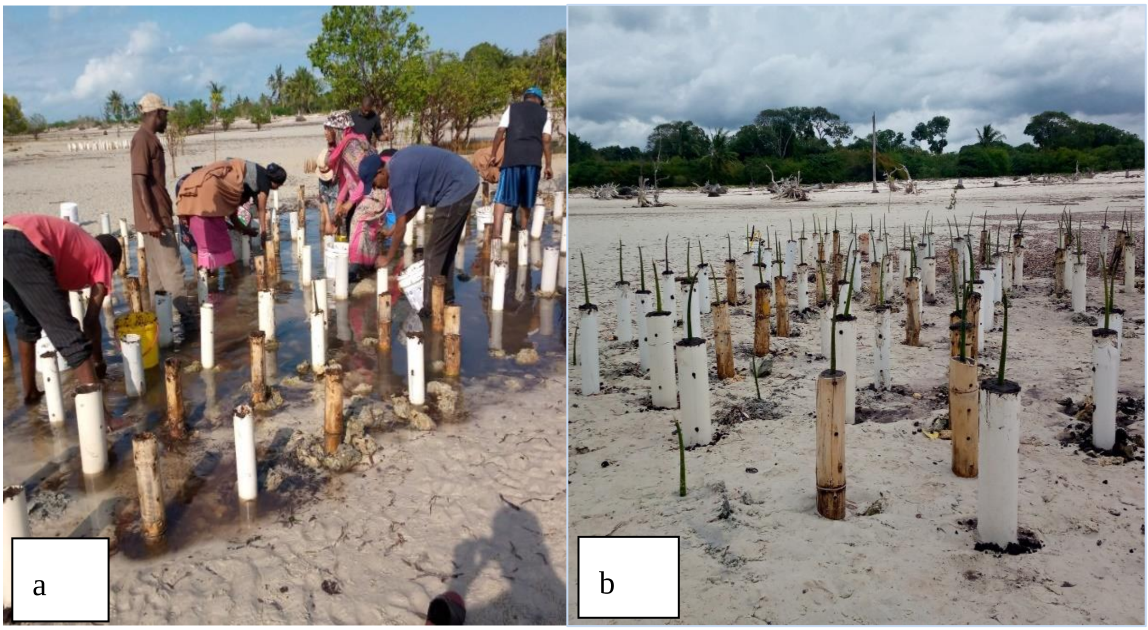

Figure 5: (a) Sediment filling within the installed encasements and (b) newly planted mangrove seedlings.

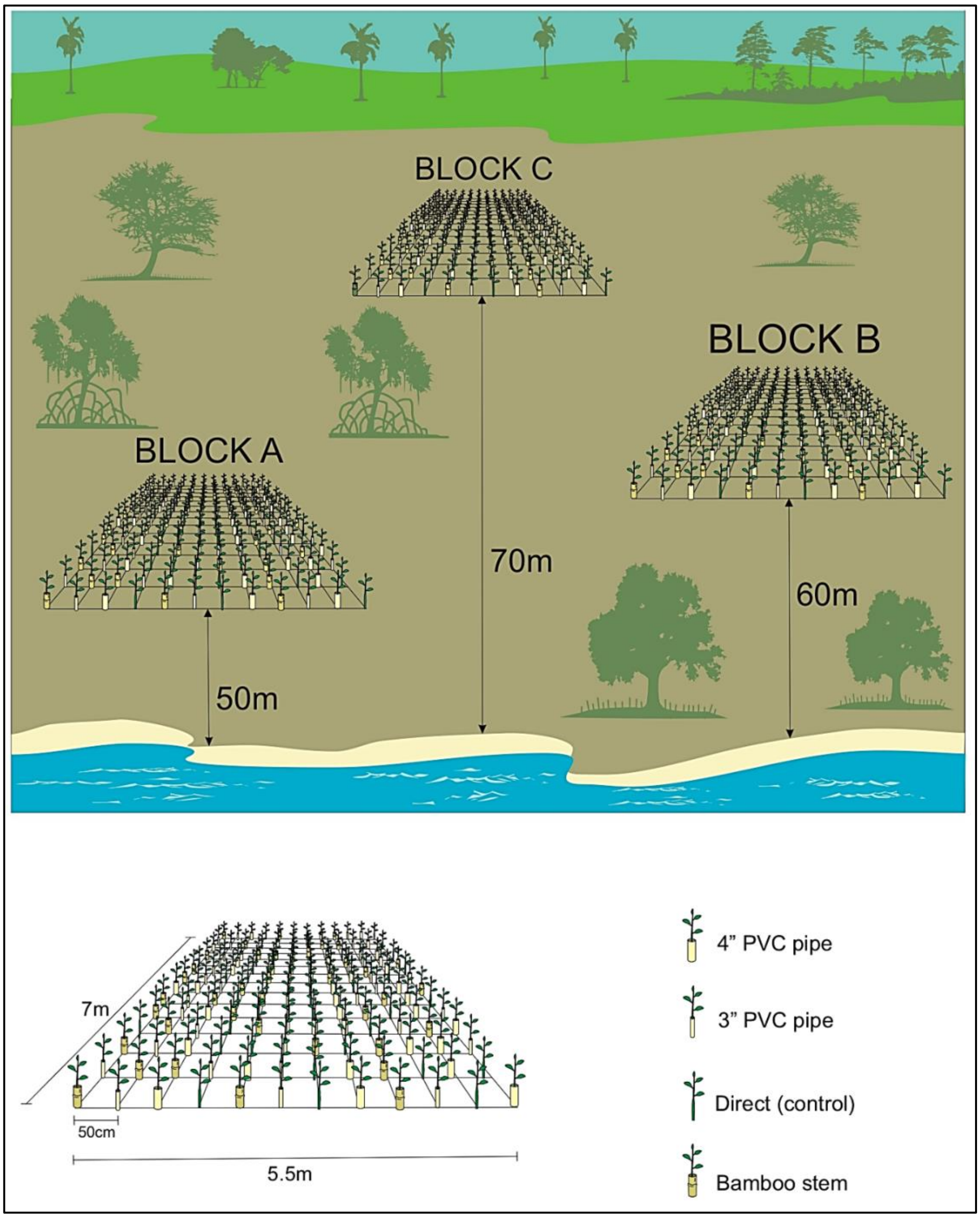

Figure 6: Schematic presentation of the experimental design showing randomized blocks and treatments in the planting area.

3.3.6.5 Monitoring of growth performance

The monitoring of planted mangrove propagules was done monthly for eight months by collecting survival and growth data using the protocol in the UNEP's guide-lines on mangrove ecosystem restoration for the Western Indian Ocean Region (UNEP, 2020). The following vegetation data were generated (Table 1): shoot height (cm), number of leaves, number of internodes, number of branches, and leaf area (Figure 7).

Table 1: Monitoring schedule adopted in assessing survival and growth performance of the planted seedlings

Time after planting	Parameters measured
0 to 3 months	Percentage Survival
4 to 6 months	Percentage Survival Shoot height (from first node to the base of top-most leaves) Number of leaves (for all individuals) Number of internodes (for all individuals)
7 to 8 months	Percentage Survival Shoot Height Number of leaves (for all individuals) Number of internodes (for all individuals)

Leaf area calculations were made following Cain and Castro (1959) equation. Total leaf area per plant was calculated by summing up the area of all leaves in a plant as follows:

$$Leaf\ Area\ (Ai) = \frac{2}{3}\ (L \times W) \qquad \text{Eqn. 11}$$

$$Total\ Leaf\ Area\ (A) = \sum_{i=1}^{n} (Ai) \qquad \text{Eqn. 12}$$

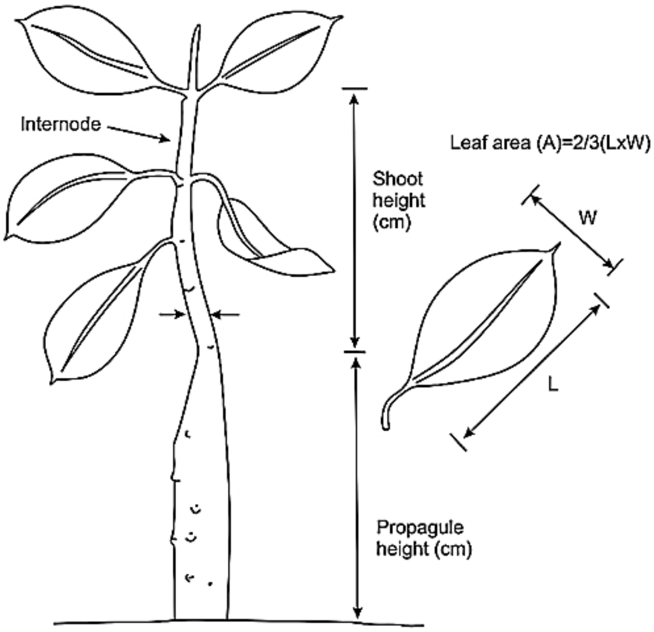

Figure 7: Biometry of mangrove sapling. Modified from (UNEP, 2020).

3.4 Data Analysis

Statistical analysis was done using the SPSS version 26.0, GRADISTAT computer program, and Microsoft Excel 2019. Satellite imageries on mangrove cover were analyzed on ArcGIS 10.5. The tabular and graphic presentations of data allowed ease of visualization. The normality of the data was tested using the Kolmogorov–Smirnov test and Shapiro–Wilk test and data were normalized where necessary to meet parametric assumptions. Sediment physical-chemical measures were analyzed using one-way ANOVA to compare means across different soil depths. When significant variations were found, a Tukey HSD post-hoc test was conducted to identify specific differences. Pearson correlation analysis was used to determine the relationship between stem diameter and height. Seedlings survival rate data were analyzed using repeated measures ANOVA to compare means across different treatments and blocks. Significant variations detected by ANOVA were further examined using a Tukey HSD post-hoc test ($P < 0.05$). Survival rates of the seedlings were represented as percentages. The bamboo and control groups were not included in the growth performance analysis due to the low survival rates (high mortality), as the small sample size would compromise the results of the study.

CHAPTER FOUR

RESULTS

4.1 Mangrove cover change between 1990 and 2020 in high energy site of Gazi Bay

The area of interest (Figure 8) was found to be approximately 1500m in length and 100m wide.

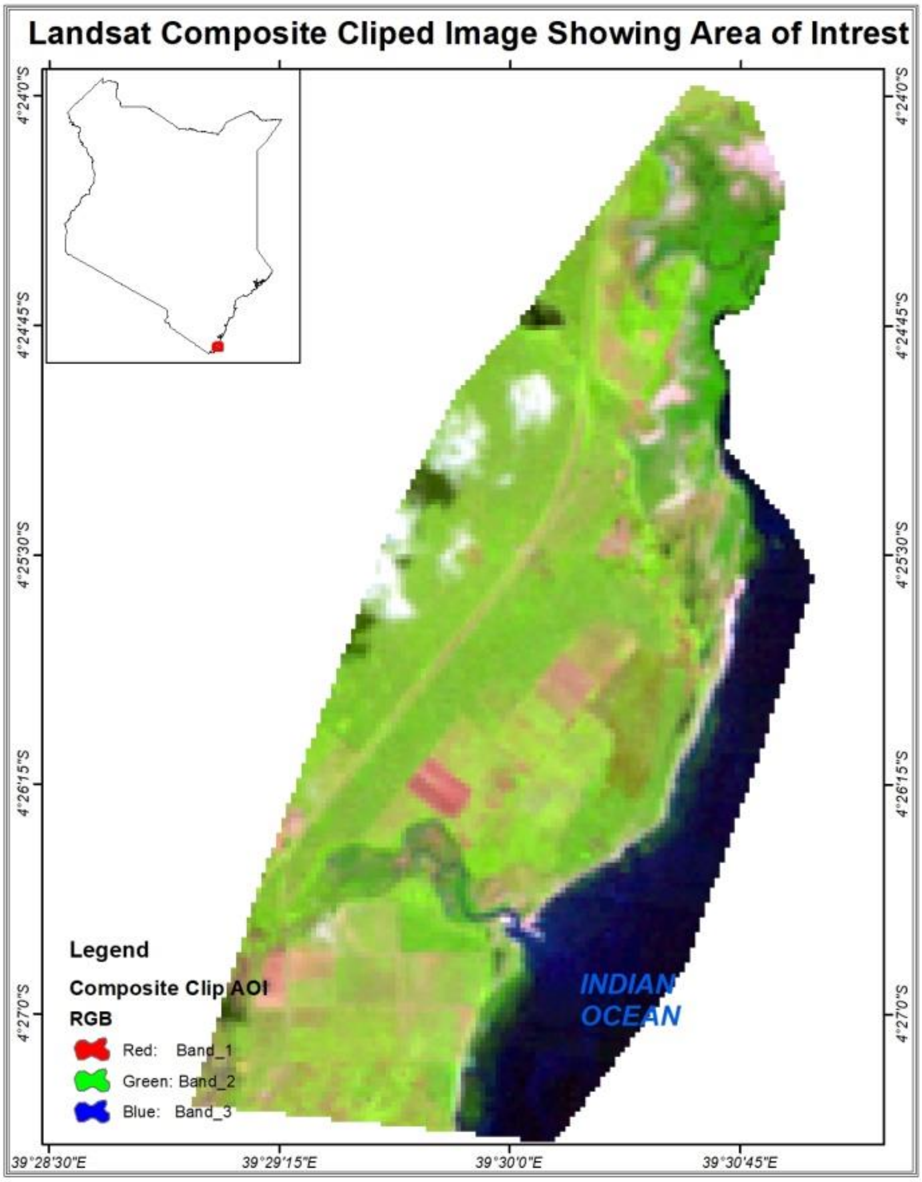

Figure 8: 1990 Landsat Area of Interest (AOI) clipped image in natural color.

Results indicate significant temporal variations in mangrove cover between 1990 and 2020 (Figure 9). In 1990, the study site had a total mangrove cover of 98 hectares. In the year 2000, this cover had decreased to 95 hectares, reflecting a 3.1% loss. By 2010, the mangrove cover had dropped to 78 hectares, and by 2020, it further declined to 52 hectares, marking a substantial 45.3% loss over the two decades (Figure 10). Overall, the mangrove cover loss between 1990 and 2020 was about 46.9%.

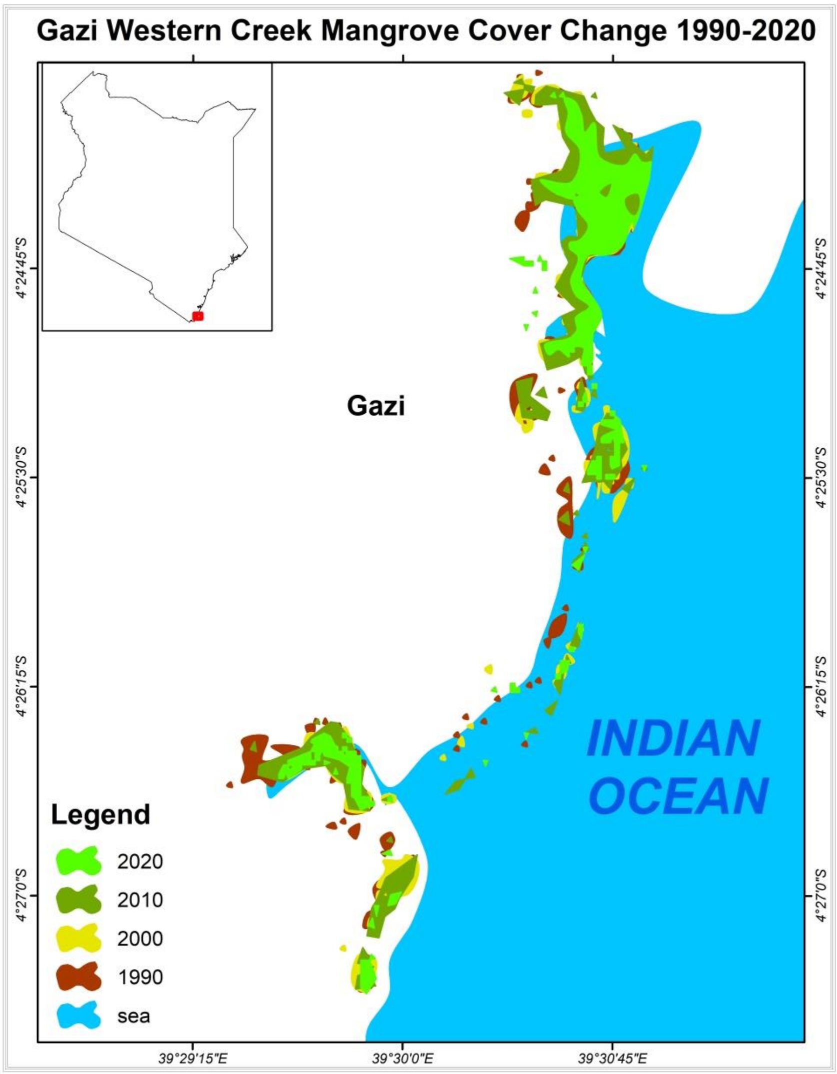

Figure 9: Mangrove cover within the AOI for the selected time period (1990, 2000, 2010, and 2020) in Gazi Bay, eroded site.

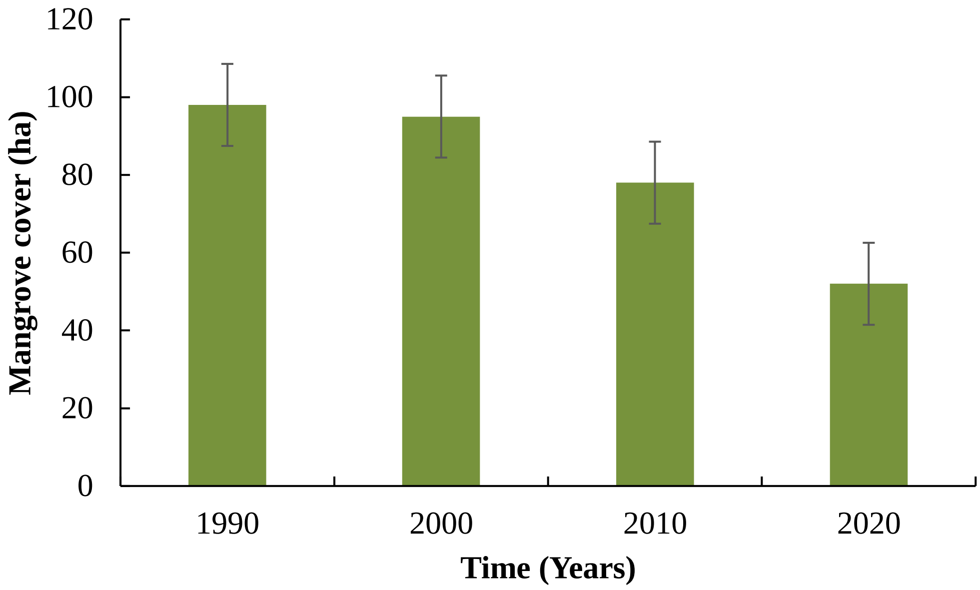

Figure 10: Mangrove cover changes between 1990 and 2020.

4.2 Biophysical characteristics in the degraded study site of Gazi Bay

4.2.1 Sediment properties

Soils in the degraded areas of Gazi Bay are characterized by high proportions of sand accounting for 96.97 ± 0.17 % with very small proportions of silt and clay (3.03 ± 0.17 %). No clear pattern is depicted in percentage sand as well as clay and silt with depth (Table 2). However, the difference in the values across the depth intervals are not statistically significant ($F_{(3,207)} = 1.222$; $p=0.303$). Very low bulk density values of below 1 g/cm³ were recorded. This is unusual especially in soils with high proportions of sand. The average bulk density across the different depth intervals ranged from 0.065 to 0.335 g/cm^3 (mean: 0.255 ± 0.007 g/cm^3) (Table 2). The percentage proportions of soil organic matter (SOM) ranged from 5.948 to 7.315 % (mean: 6.334 ± 0.24 %); while those of soil organic carbon (SOC) ranged from 5.358 to 5.926 % (mean: 5.519 ± 0.10) with no significant differences between the depth intervals ($F_{(3,207)} = 0.944$; $p=0.420$) (Table 2).

Table 2: Sediment physical-chemical properties in the different depth intervals of Gazi Bay eroded site (mean ± s.e)

Sample Depth (cm)	Coarse Sand (%)	Medium Sand (%)	Fine Sand (%)	Silt-clay (%)	Bulk density (g/cm^3)	SOM (%)	SOC (%)
0 – 15	13.54±0.98	27.68±2.30	55.33±2.60	3.45±0.30	0.335±0.01	6.419±0.35	5.554±0.15
15 – 30	19.50±1.41	27.60±1.87	50.05±2.30	2.86±0.24	0.281±0.01	5.948±0.48	5.358±0.20
30 -50	18.74±1.53	24.66±2.08	53.76±2.55	2.84±0.43	0.184±0.01	6.251±0.55	5.484±0.23
50 – 100	19.23±1.73	19.51±1.75	58.65±2.99	2.60±0.34	0.065±0.00	7.315±0.58	5.926±0.24
Average	**17.19±0.70**	**26.10±1.13**	**53.68±1.35**	**3.03±0.17**	**0.255±0.01**	**6.334±0.24**	**5.519±0.10**

The values of bulk density varied significantly with depth ($F_{(3,207)} = 97.181$; $p<0.05$) as they decreased with increasing depth (Figure 11).

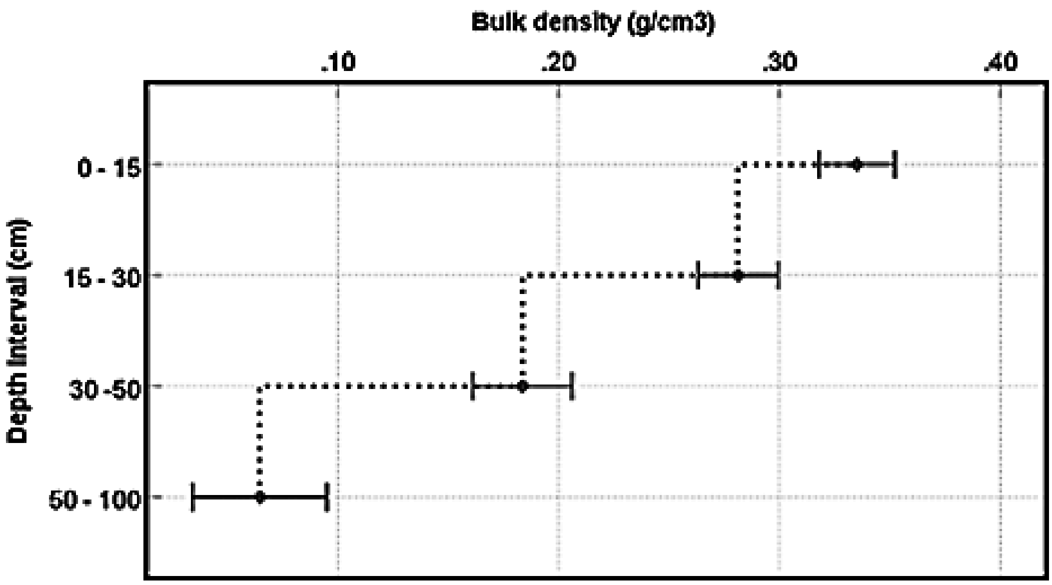

Figure 11: The relationship between depth and bulky density. Error bars represent the standard error of mean.

4.2.2 Mangrove Forest structure

Three species of mangroves were encountered in the project site. Based on their importance value (IV) index, the most dominant species in the site is *Sonneratia alba* (IV = 293.23%); followed by *Rhizophora mucronata* (16.25) *and Avicennia marina* (13.25) (Table 3). The presence of mature established species of *Rhizophora mucronata* in the study area enabled the selection of the species for the restoration experiment (Figure 12).

Table 3: Species composition and importance values of mangroves encountered in the eroding Gazi Bay shoreline

Species	**Relative values (%)**			
	Dominance	**Frequency**	**Density**	**IV (%)**
Avicennia marina	2.45	9.09	1.709	13.25
Rhizophora mucronata	1.25	13.64	1.368	16.25
Sonneratia alba	96.30	100	96.923	**293.23**

Figure 12: A photo taken from the study site showing mature Rhizophora stand establishing in combination with the native species.

The overall stand density of the mangroves in the study site at Gazi Bay was 1659 ± 460.13 stems ha^{-1}, with mean basal area, tree height and DBH of 21.06 ± 2.45 $m^2 ha^{-1}$ (range: 3.28 - 39.96 m^2ha^{-1}), 5.49 ± 0.09 m (range: 1.5-11.2 m) and 8.52 ± 0.23 cm (range: 2.5-31.5 cm) respectively (Table 4).

Table 4: The structural attributes of the mangroves encountered in Gazi Bay degraded site (mean ± s.e)

Attributes	**Values**
Number of plots	22
Number of species	3
Mean height (m)	5.49 ± 0.09
Mean Dbh (cm)	8.52 ± 0.22
Mean Basal Area (m^2ha^{-1})	21.06 ± 2.45
Mean Stem Density (stems ha^{-1})	1659 ± 460.13

Heights against stem diameters scattergram of mangroves in the bay indicated a positive correlation between stem diameter and height. However, regression analysis reveals that it is a low positive correlation (R^2 = 0.255). Some 50% of trees in the study sites had diameters and height range from 4.5 to 11.5 cm and 3.5 to 7.0 m, respectively (Figure 13).

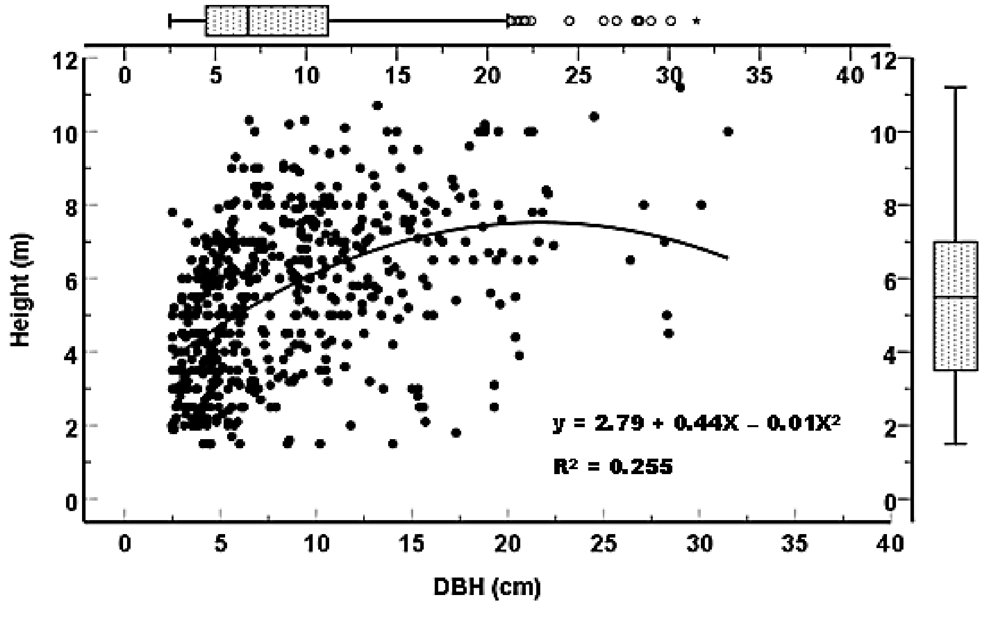

Figure 13: Height-diameter distribution of mangroves in Gazi degraded site. The box plots on the side exhibit the percentile spread of DBH and height.

4.3 Trial mangrove growing experiments in high energy site of Gazi Bay

4.3.1 Survival rates of the replanted mangroves

Three months after planting (the last month at which mangroves encased within bamboo poles were present), Block A recorded survival rates ranging from 4-52%. Among these, those encased in 3” PVC pipes displayed the highest survival rate at 52%, followed by those in 4” PVC at 42%, and bamboo pipes at 9%. The control group, consisting of directly planted seedlings, recorded a 4% survival rate during the same period. At the end of the eighth month, survival rates ranged between 4-38%. Mangroves in 3” PVC achieved the highest survival rate at 38%, followed by those in 4” PVC at 36%, and the directly planted seedlings at 4% (Figure 14).

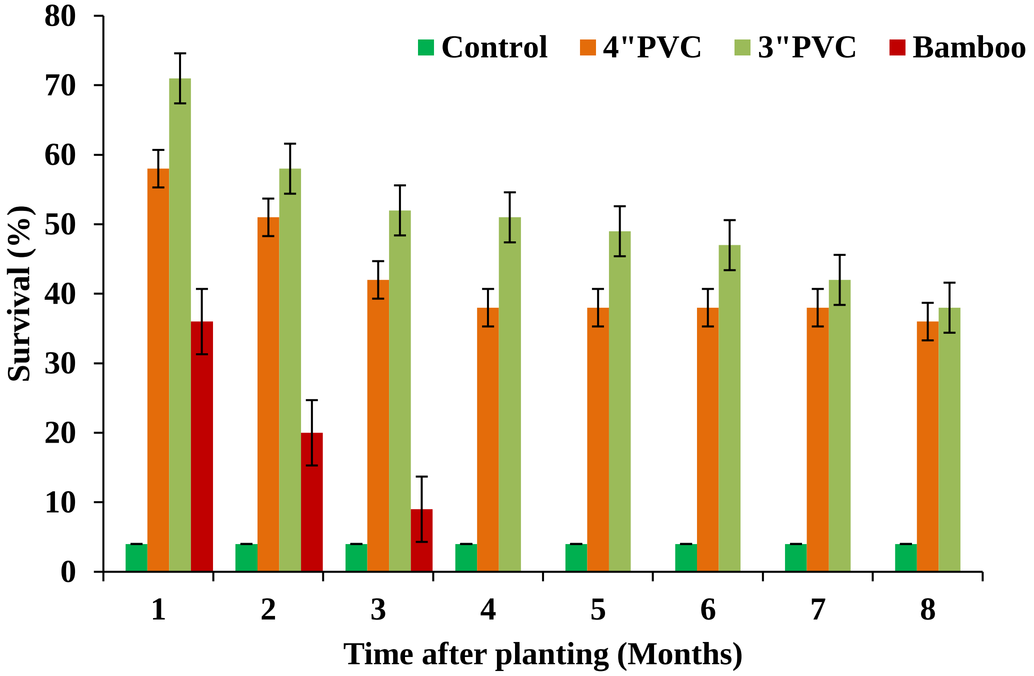

Figure 14: Survival rates of mangroves established in Block A under different treatments over time.

Eight months after planting, mangroves grown in Block B recorded survival rates ranging from 4-58%, with those established in 4” PVC recording the highest survival rate of 58%, followed by those in 3” PVC (51%), and bamboo pipes (4%). Over the same growth period, directly planted propagules had a survival rate of 7% only (Figure 15).

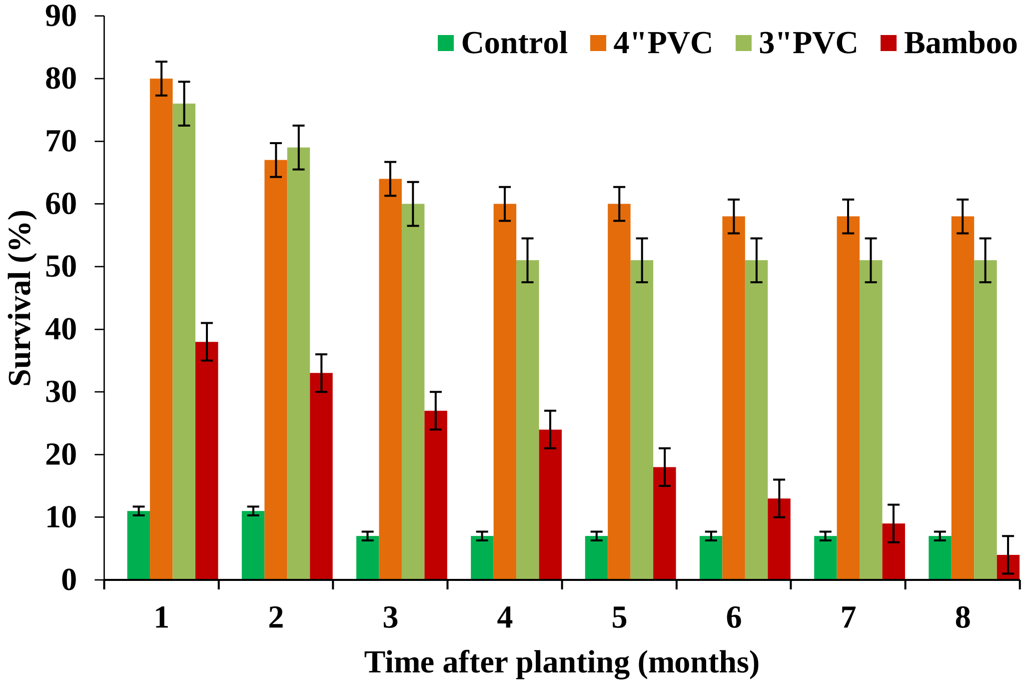

Figure 15: Survival rates of mangroves established in Block B under different treatments over time.

Mangroves sown in Block C had survival rates ranging from 4-60% two months after planting (the last month at which mangroves encased within bamboo poles were present), with seedlings encased in 4” PVC having the highest survival (60%), followed by those encased within 3” PVC (53%), and bamboo pipes (16%). The directly planted propagules had the lowest survival rate of 4%. At the end of the experiment (8 months after planting), survival rates ranged from 2 to 40%, with mangroves planted within 3” PVC recording a higher survival rate of 40% followed by those planted in 4” PVC (36%), and the control with a 2% survival (Figure 16).

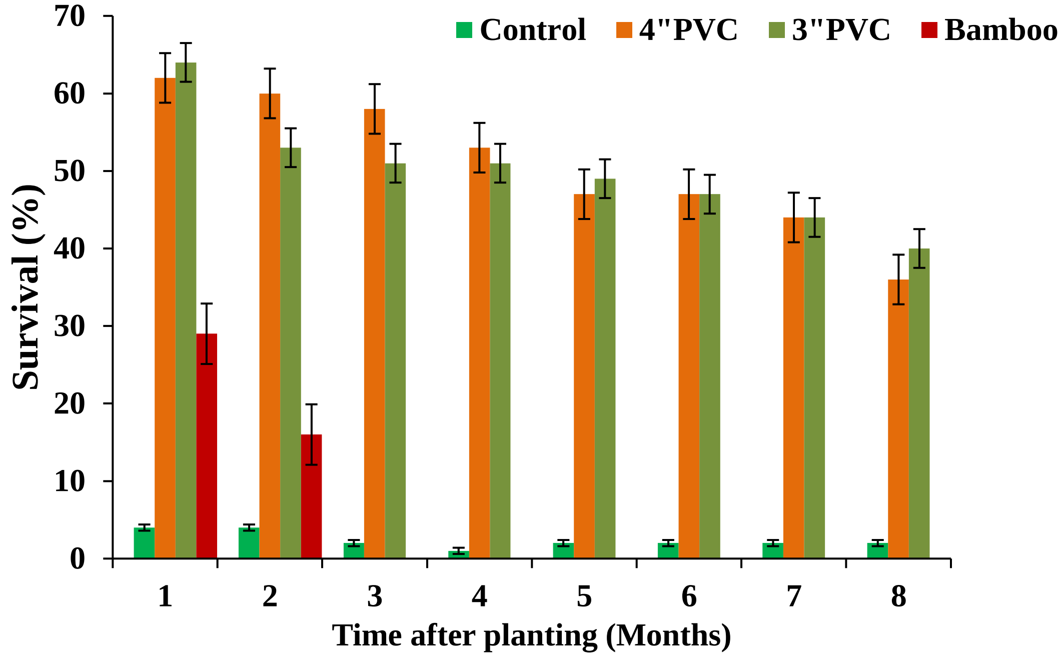

Figure 16: Survival rates of mangroves established in Block C under different treatments over time.

Overall, survival rates varied significantly between treatments ($F_{(3, 81)} = 80.126$; *$p<0.05$*) (Table 5). However, there were no statistically significant differences in survival rates between blocks ($F_{(2, 82)} = 0.500$; *$p=0.378$*) (Table 5). Seedlings planted in Block B recorded a slightly higher mean percentage survival of 39 ± 4.5 %, followed by Block C with 33 ± 4.6 % and lastly Block A with 31 ± 4.1%.

Table 5: ANOVA table on the effect of treatments and blocks on survival rates.

	Treatments					**Blocks**				
	Sum of Squares	**df**	**Mean Square**	**F**	**Sig.**	**Sum of Squares**	**df**	**Mean Square**	**F**	**Sig.**
Between Groups	34975.962	3	11658.654	**80.126**	**.000**	1096.196	2	548.098	**.984**	**.378**
Within Groups	11785.849	81	145.504			45665.615	82	556.898		
Total	**46761.812**	**84**				**46761.812**	**84**			

Following significant differences in survival rates between treatments, a post hoc Tukey's Honest Significant Difference (HSD) test was conducted to determine which specific treatments differed from each other (Table 6). Survival rates for the control group varied significantly from PVC and bamboo grown seedlings *($p<0.05$)* (Table 6). Similarly, survival rates for bamboo grown seedlings varied significantly from PVC grown seedlings *($p<0.05$)*. No significance difference in survival rates was recorded between 3'' and 4'' PVC grown seedlings *($p=0.997$)*. Mangroves encased in 3" and 4" PVC showed significantly higher percentage survival of 43%, whereas those grown in bamboo and the control group exhibited much lower percentage survival of 1% and 4%, respectively.

Table 6: Post hoc test results showing group-wise comparisons in survival rates

Post Hoc-Tukey HSD						
					95% Confidence Interval	
(I) Treatment	**(J) Treatment**	**Mean Difference (I-J)**	**Std. Error**	**Sig.**	**Lower Bound**	**Upper Bound**
Control	BP	-44.417*	3.482	.000	-53.55	-35.28
	SP	-45.083*	3.482	.000	-54.22	-35.95
	Bamboo	-13.522*	4.154	.009	-24.42	-2.63
BP	Control	44.417*	3.482	.000	35.28	53.55
	SP	-.667	3.482	.997	-9.80	8.47
	Bamboo	30.894*	4.154	.000	20.00	41.79
SP	Control	45.083*	3.482	.000	35.95	54.22
	BP	.667	3.482	.997	-8.47	9.80
	Bamboo	31.561*	4.154	.000	20.66	42.46
Bamboo	Control	13.522*	4.154	.009	2.63	24.42
	BP	-30.894*	4.154	.000	-41.79	-20.00
	SP	-31.561*	4.154	.000	-42.46	-20.66

***. The mean difference is significant at the 0.05 level.**

4.3.2 Growth performance of the planted mangroves

Four months after planting, seedlings growing in Block A had attained a shoot length of 7.3-22.2 cm (average: 12.82 ± 0.52), with 4-8 leaves (average: 6.15 ± 0.27) and 1-3 internodes (average: 2.08±0.13). At six months, seedlings had a shoot length ranging from 11.8 to 28 cm (average: 18.83±0.62), with 8-12 (average: 10.31 ± 0.2) leaves and 3-5 (average: 4.5 ± 0.1) internodes. By the eighth month, the seedlings' shoots ranged from 15.5 to 37.2 cm (average: 25.38 ± 0.83), with those encased in 4" PVC and 3" PVC attaining a mean of 25.91± 1.43 cm and 24.88 ± 0.90 cm, respectively. The number of leaves and internodes ranged between 10-14 (average: 12.36 ± 0.2) and 4-6 (average: 5.18 ± 0.1), respectively, with seedlings encased in 4" PVC recording an average of 12.88 ± 0.32 leaves and 5.44 ± 0.16 internodes while those encased in 3'' PVC had an average of 11.88 ± 0.21 leaves and 4.94 ± 0.10 internodes (Figures 17,18, 19). There were no significant statistical differences in vegetation attributes among seedlings grown in 3" and 4" PVC ($p>0.05$).

By the fourth month of growing, seedlings planted in Block B had attained a shoot length ranging from 4.9 to 24.3 cm (average: 16.36 ± 0.63), with 2-8 (average: 5.12 ± 0.21) leaves and 1-3 (average: 1.63 ± 0.10) internodes. In six months, the shoot length was ranging between 12.4 and 33 (average: 21.58 ± 0.71), while the number of leaves and internodes ranged between 4 and 12 (average: 8.36± 0.21) and 1and 5 (average: 3.18±0.11), respectively. On reaching the age of eight months, seedlings had attained shoot length ranging from 15.1-38.1 (average: 26.33 ± 0.67), 8-14 (average: 11.25 ±0.19) leaves, and 3-6 (average: 4.62 ± 0.10) internodes. Seedlings encased in 4" PVC had an average shoot length of 27.49 ± 0.86 cm, 11.54.88 ± 0.28 leaves, and 4.77 ± 0.14 internodes, while those encased in 3" PVC recorded an average of 25.21 ± 1.08 cm in shoot length, 10.91 ± 0.25 leaves, and 4.45 ± 0.13 internodes (Figures 17,18, 19). The differences recorded among seedlings grown in 3'' and 4'' PVC were statistically insignificant ($p>0.05$).

At four months of growing, seedlings in Block C attained shoot length of 4.3-18 cm (average: 13.28 ± 0.42 cm), 2-8 (average: 5.81 ± 0.27) leaves, and 1-3 (average: 1.95 ± 0.13) internodes. After six months, the shoot length ranged between 8-24.5 cm (average: 18.76 ± 0.46), with 4-12 (average: 9.86 ± 0.21) leaves and 1-5 (average:

3.93 ± 0.1) internodes. By the eighth month, the shoot length reached 12.3-35.3 cm (average: 24.73 ± 0.48 cm), with 8-14 (average: 11.45 ± 0.22) leaves and 3-6 (average: 4.73 ± 0.11) internodes. Seedlings grown in the 3" PVC recorded an average shoot length of 21.09 ± 1.17 cm with 11.44 ± 0.27 leaves and 4.72 ± 0.14 internodes. In contrast, those enclosed within the 4" PVC achieved a mean shoot length of 22.39 ± 1.42 cm. These seedlings also displayed 11.47 ± 0.36 leaves and 4.73 ± 0.18 internodes (Figures 17,18, 19). The variances observed between the two treatments were not statistically significant (*p>0.05*).

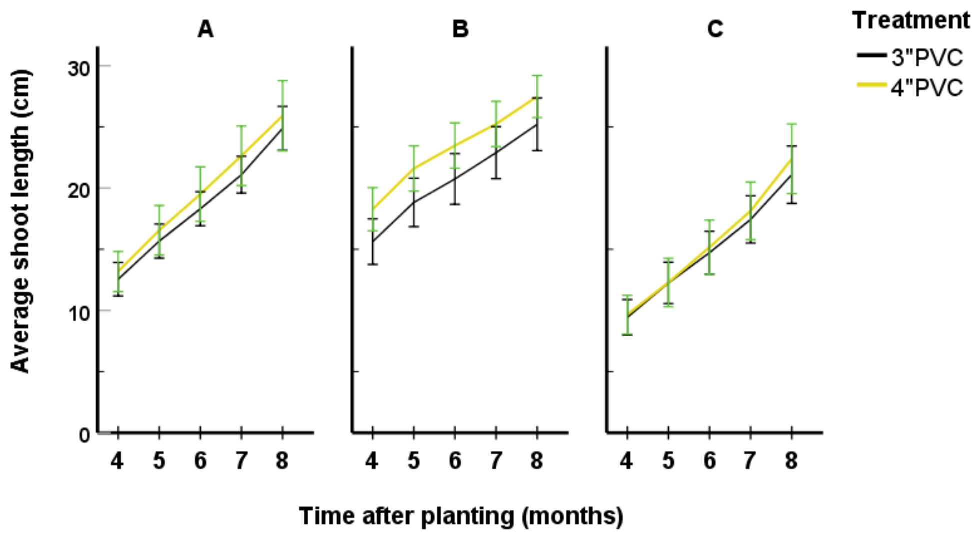

Figure 17: Average shoot growth (in cm) in blocks per treatment.

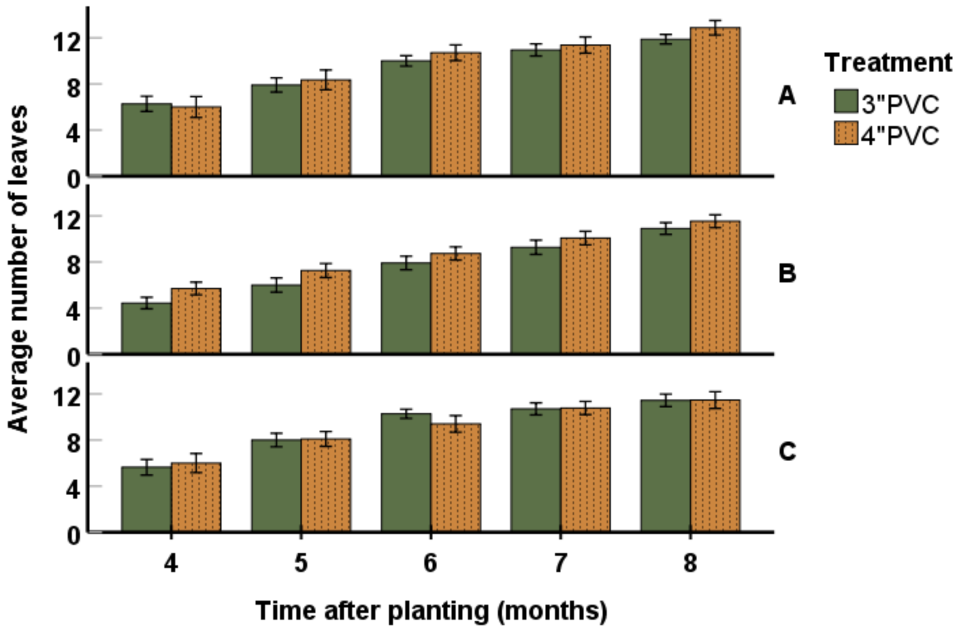

Figure 18: Average number of leaves in blocks per treatment.

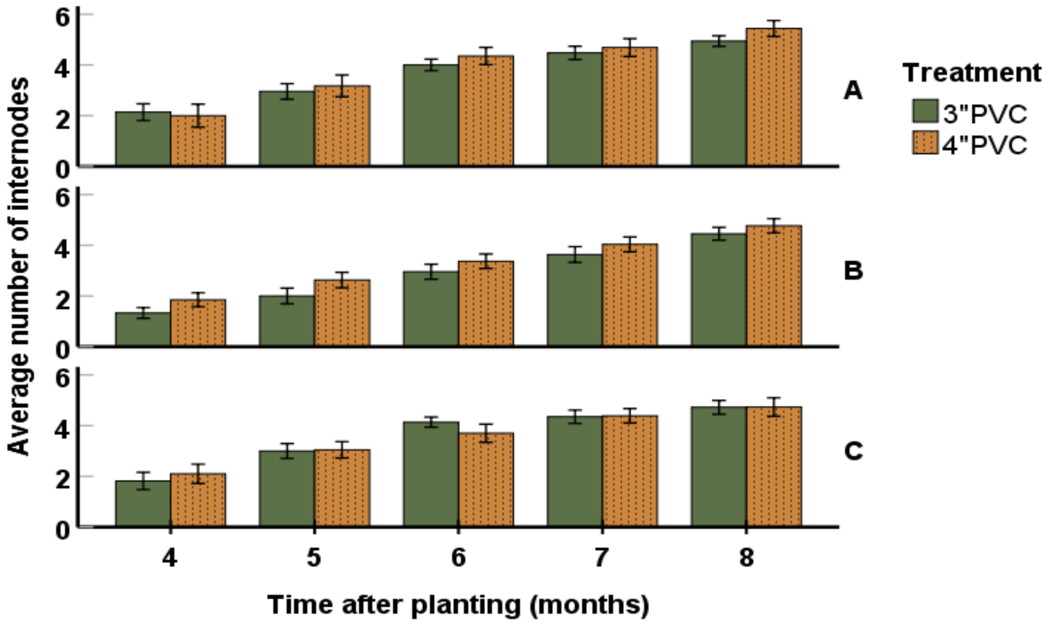

Figure 19: Mean number of internodes in blocks per treatment.

At the conclusion of the experiment, the differences observed in growth performances of mangroves encased in 3" and 4" PVC were not statistically significant (*p>0.05*) as presented in Table 7.

Table 7: Overall growth performances for the replanted mangroves at the end of the experiment (mean ± se)

Treatment	Shoot growth (cm)	Number of leaves	Number of internodes	Number of Branches
3''PVC	23.84 ± 0.66	11.37 ± 0.15	4.68 ± 0.08	1.87 ± 0.13
4''PVC	25.65 ± 0.72	11.89 ± 0.20	4.95 ± 0.10	2.45 ± 0.18
Total	24.74 ± 0.69	11.63 ± 0.13	4.82 ± 0.06	2.27 ± 0.13

The leaf area per plant for seedlings encased in 3" and 4" PVC varied within the ranges of 41.33-149.29 cm^2 and 41.79-146.29 cm^2, respectively, at the six-month mark. In Blocks A, B, and C, leaf area ranged from 47.20-149.29, 48.10-142.91, and 41.33-145.51 respectively. On average, considering all blocks together, leaf area of 100.48 ± 2.05 (with a range of 41.33-149.29) per plant.

CHAPTER FIVE

DISCUSSION, CONCLUSION AND RECOMMENDATIONS

5.1 Mangrove cover changes in Gazi Bay degraded site

Mangrove cover loss has been associated with shoreline changes (McIvor et al., 2015; Menéndez et al., 2020) among other negative impacts including scarcity of wood products (Kairo et al., 2002) and reduced fish production (Barbier et al., 2011). In particular, loss of the outer mangrove fringe has been known to cause increased shoreline erosion and decreased elevation (Bandeira & Balidy, 2016; Primavera et al., 2011). The findings of the study revealed significant mangrove cover loss of 46.9% between 1990 and 2020. These results explain the increased shoreline erosion experienced in the study site. Mangroves are known to protect the shorelines by minimizing wave energy, stabilizing sediment, and facilitating sediment accretion (Gedan et al., 2011; Shepard et al., 2011; Steven et al., 2020).

The degradation of mangroves in the site following the 1970s exploitation of wood for fuel energy in calcium manufacturing industry (Dahdouh-Guebas et al., 2004; Kirui et al., 2013) and subsequent losses of mangroves as depicted by the results justifies the need for restoration in the study site. At the same time, ecosystems restoration has become increasingly important and critical in addressing the impacts of climate change and biodiversity decline (Cadier et al., 2020; Airoldi et al., 2021; Gerona-Daga & Salmo III, 2022). The result also signifies lack of natural regeneration and further losses following overexploitation of mangroves. Loss of the fringing mangroves leads to decreased elevation altering the optimum conditions for mangrove establishment and growth (Primavera et al., 2011; Brooks & Spencer, 2012) thus decreasing mangrove cover. High exposure of the remaining mangrove to environmental stressors such as sedimentation that burry mangroves roots leading to death of the trees as it was apparent in the study site.

5.2 Sediment characteristics

Sediment particle size distribution is among the most critical physical properties that affects soil conditions in regards to erosion (Rahardjo et al., 2008; Zhao et al., 2017). The very low amount of percentage silt-clay (3.03 ± 0.17) in sediment compared to those of forested sites (38.38 %) in Gazi (Kairo et al., 2008) is an indication of a highly degraded and low vegetation cover. The proportions of fine sand (53.68 ± 1.35%) are comparable to those of non-reforested sites in Gazi Bay (58.85± 6.11%) (Kairo et al., 2008) further indicating a situation of a degraded site. The relatively small amounts of SOM (6.33 ± 0.24 %) and SOC (5.52 ± 0.10 %) in the sediment compared to values obtained in Gazi forested sites (38.38, 31.04, 17.38 %) by Kairo et al., (2008) suggests near zero sediment accretion. Mangrove vegetation enhances the build-up of SOM by accreting sediments (Kairo et al., 2008). Mangrove vegetation boosts sediment trapping and settlement by reducing the speed of water movement (McKee & Cherry, 2009; Tue et al., 2012). With the high proportions of sand (96.97 ± 0.17 %) in the sediments, there is a high possibility of decreased soil stability which increases the risk of erosion (Al-Shayea, 2001; Dimitrova & Yanful, 2012) by high tides and other extreme weather events.

Soil bulk density is considered as the soil's dry weight per unit volume and it indicates soil compaction. Normally, as the soil depth increases bulk density is known to increase (Cerón-Bretón et al., 2010) as the weight of the top layer sediments effect soil compaction (Calderón et al., 2011). However, the results of this study deviate from the expectations as bulk density significantly decreases with increase in soil depth (Figure 11). A situation where the top layers of mangrove sediment have experienced physical compaction due to the action of waves, tides and other forms of disturbance was observed. Such a situation could lead to increased bulk density in the shallow layers as the impact of compaction would diminish with depth resulting in lower bulk density in deeper layers. As noted in the results, the bulk density values are quite low (below 1 g/cm^3), which is unusual, especially in soils dominated by sand. The soil samples exhibited a high presence of decomposed plant material and extensive root systems, which likely increased porosity and led to the lower bulk density (McKee & Faulkner, 2000). In some mangrove areas, the sandy soil may be mixed with lighter materials such as silt or fine organic debris, reducing the overall bulk density (Alongi, 2008). The decomposed plant material likely originates from mangroves that were previously present but have since been deforested.

5.3 Mangrove Forest structure in Gazi Bay degraded study site

The experimental mangrove planting area in the bay is dominated by *Sonneratia alba* thus the higher importance value (IV). The species naturally occurs in the low lying intertidal areas, in the inundation Class 1 of Watson (1928). The fringing forest is heavily fragmented due to mangrove harvesting for fuelwood in the 1970's (Dahdouh-Guebas et al., 2004) that left blank contiguous areas (Kirui et al., 2013), leading to the moderately low stand density. Other mangrove species in the zone are *Rhizophora mucronata* and *Avicennia marina*, which exist as both adults and juveniles (Kairu et al., 2021; Kairo et al., 2008).

Adult Rhizophora species have prop roots for anchorage and breathing (Srikanth et al., 2016). The rooting systems forms strong mesh system that enables them to create wave barriers (Kamil, Takaijudin, & Hashim, 2021). As such Rhizophora have high proficiency in wave attenuation compared to other mangroves species. This is due to the complexity of the roots' system, which creates greater friction to incoming waves thus a higher drag coefficient (Tanaka et al.,2007; Ohira et al., 2013; Kamil et al., 2021) (Figure 20). This is critical for the designated planting area as it is highly exposed to wave action. It has been reported that, a 80 m width of Rhizophora species can dissipate incoming wave energy by 80% while a 100 m width *Sonneratia alba* stand can only reduce 50 % of the waves energy (Mazda et al., 2006; Hashim & Catherine, 2013).

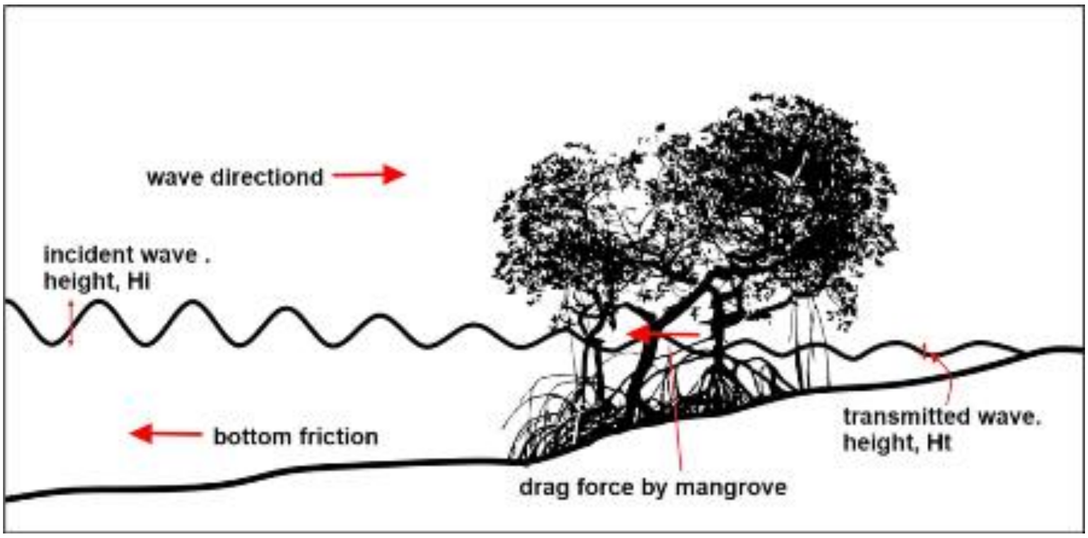

Figure 20: Wave dissipation by mangroves. Adapted from (Kamil et al., 2021).

The rooting system also accretes sediments and facilitates natural recruitment of the indigenous species. The genus Rhizophora is also known to be a land-builder through sediment accretion and stabilizations (Primavera et al., 2016), thus the potential in revitalizing eroded shorelines and keeping pace with rising sea levels (Alongi, 2008; Temmerman et al., 2013; Lovelock et al., 2015). As such, the successful establishment of the species in the site will play a critical role in enhancing the restoration of the Gazi shoreline and facilitate natural recruitment of new mangrove seedlings.

The low positive association (R^2=0.255) between the stem diameters and heights of the mangroves in the degraded Gazi site could probably be a result of the challenging environmental conditions, including sedimentation and exposure to high wave action, that negatively impact the growth patterns of mangrove trees. This might lead to disparities in stem diameter and height thereby weakening the correlation. Such weak correlations have been reported elsewhere, for instance in Mombasa and Mtwapa, Kenya (Njiru et al., 2022), where the mangrove ecosystem is predisposed to human and environmental stressors (Oosterom, 1988; Mohamed et al., 2009).

Overall, conditions along the shoreline of Gazi Bay depict a picture of a highly degraded and exposed mangrove site. This validates the need for alternative restoration approaches to enhance restoration success and ensure that communities are protected as the degraded fringing mangrove belt stands between the land and the sea. In the face of changing climate and sea-level rise, shoreline protection has become a major concern (Prasad & Kumar, 2014) since shorelines safeguard coastal communities as well as coastal ecosystems.

5.4 Survival and mortality of the planted mangroves

In most mangrove restoration projects, survival rates are used as one of the common indicators for success (Kodikara et al., 2017; Wodehouse & Rayment, 2019). The results of this study reveal significant differences in survival rates among treatments. Significant variations were particularly noted between the PVC and bamboo-encased and between the PVC-encased and control, as well as between the bamboo-encased and control. There were no significant differences in survival rates between the different PVC encasement sizes *(p=0.992)*. Major losses that occurred in the first two months following planting was due to tidal washing. This outcome is anticipated in the pre-establishment stage when seedlings have not yet firmly taken root. In the first month, the highest mortality (94%) was recorded for the conventionally planted propagules, which is expected since these propagules had no protection, and thus were highly exposed to strong wave energy (Ndirangu, 2016). High wave energy has been deemed to be a stressor that limits the success of mangrove establishment (Primavera & Esteban, 2008). Similar results were observed in high energy mangrove planting areas of Florida (Teas, 1977). Results also align with a study by (Kairo et al., 2001) that demonstrated that up to 90% mortality for mangrove seedlings exposed to high energy. High mortality for propagules encased in bamboo tubes was also occasioned by washing by tides as a result of the small diameter tube that could not hold enough soil, thus providing a weak support system. This contrasts with the findings of Kent & Lin (1999), where the mortality of seedlings encased in bamboo was occasioned by lack of sufficient light recording 1.5% survival rates. This is so because in the current study seedling were not fully encased, as was the case with Kent and Lin (1999), rather, seedlings were set at an elevation that allowed enough light to get into the seedlings.

Survival rates were highest in propagules installed in 3” and 4” PVC tubes. Despite the differences in PVC diameters, there were no significant differences ($p=0.378$) in survival rates across the blocks. At the end of the experiment, they recorded similar survival rates (43%) while the bamboo tubes and control had survivals of 1% and 4%, respectively. This could be attributed to the PVC’s ability to hold more sediments thus offering adequate support to the seedlings. Kent and Lin (1999), in their experiment, also reported higher survival rates of above 70% for seedlings encased in full-length PVC, exceeding bamboo-grown and conventionally planted seedlings. This highlights the potential of PVC encasement in mangrove restoration within high-energy areas. Modifications to prevent sediment loss from within the pipes could further enhance the efficacy of PVC. In addition, techniques such as crafting physical barriers around the restoration blocks could help in minimizing the impacts of ocean tides, waves, and currents, thereby enhancing success (Hashim et al., 2010; Furukawa et al., 2019). These defenses have the ability to attenuate incoming waves (Shu et al., 2023), thus reducing the energy reaching the planted mangroves.

Other eco-engineering approaches used for mangrove restoration in areas exposed to high wave energy often report higher growth and survival rates compared to the conventional planting techniques. For instance, the use of breakwater in Sungai, Malaysia to minimize the wave energy reaching the planted seedlings was moderately successful with 30% of the originally planted seedlings surviving eight months after planting (Hashim et al., 2010). At Pedada Bay in the Philippines, survival rates higher than 70% were achieved through the use of breakwaters (Furukawa et al., 2019). The effective deployment of breakwaters in Indonesia and Vietnam resulted in decreased wave energy, and thus reducing coastal erosion and sediment accumulation and increasing the rate of mangrove colonization (Le Xuan et al., 2022). In Grand Cayman, the use of anchored armored concrete cultivator pots for mangrove planting in a high energy environment was associated with high survival rates above 70% (Krumholz & Jadot, 2009).

However, the advancement of mangrove restoration techniques in high-energy environments is an ongoing endeavor essential for effective management. This is particularly important due to the various human and climatic pressures on blue carbon ecosystems like mangroves. The ultimate goal surrounds the innovation of cost-effective methods that can be embraced in sites where conventional methods show limited efficacy (Krumholz & Jadot, 2009).

5.5 Growth performance of the planted mangroves

There were no significance differences in sapling shoot length ($p=0.605$) or the numbers of leaves ($p=0.06$), internodes ($p=0.07$), and branches for mangroves encased in 3” and 4” PVC. The average number of leaves per plant by the fourth month of growing (5.67 ± 0.65) is almost similar to that of seedlings planted under field conditions (6.17 ± 3.86) by Kairo (1995) in Gazi Bay. This is an indication that PVC encasements might have the capability of imitating the natural environment under which mangroves establish. However, other parameters including shoot growth and the number of internodes were relatively high under field conditions (35.11 ± 3.86; 7.29 ± 3.78) recorded by Kairo (1995) compared to those in the PVC environment (14.15±0.53; 1.87±0.12) in this study. Additionally, values obtained for the total leaf area per plant at the age of 6 months (100.5±2.1 cm^2 (41.3-149.3 cm^2) are significantly lower compared to those obtained in a study by Kairo (1995) (under field conditions (893.1± 71.3 cm^2 (195.2-1615.5 cm^2). This could be attributed to nutrient scarcity within the PVC environment. Over the course of the current experiment, removal of sediments from the encasements became apparent, potentially resulting in nutrient depletion. This phenomenon may have contributed to slow growth and a decrease in leaf area. The primary factor constraining the growth of mangroves is the availability of nutrients (Almahasheer et al., 2016a; Alongi, 2011). Instances of nutrient deficiencies have been associated with stunted growth in mangroves (Krauss et al., 2008; Almahasheer et al., 2016b). Consequently, there is a need to optimize conditions within the PVC pipes, possibly through improved nutrient management practices.

5.6 Conclusion and recommendations

The conditions observed in the study area revealed a mangrove site with minimal vegetation, significant degradation, and considerable exposure, highlighting the need for restoration. The trend in mangrove cover over the years showed a continued decline, highlighting ongoing loss. Both vegetation and soil characteristics at the site are indicative of severe degradation, consistent with those found in non-forested or heavily disturbed areas, further emphasizing the necessity for concerted restoration efforts. Over the eight months of observation, seedlings encased in 3” and 4” PVC pipes recorded greater survival and growth rates compared to the directly planted and bamboo encased seedlings. However, if the monitoring was done for a longer period, significant differences may have developed between the different-sized PVC encasements. Higher survival rates of 43 % were recorded for seedlings grown within PVC encasements indicating that they were more effective in enhancing the survival of the planted seedlings in high-energy areas of Gazi Bay.

The performance with PVC encasements can be enhanced by devising measures that ensure better nutrient management. Though bamboo encasements were not effective in this study, they have a potential of being useful if those with large diameters are available. A deep substrate would also be suitable for bamboo pipes due to ease of installation. With the proposed revisions, encasement methodology would facilitate survival rates by minimizing seedling washout, which is the key challenge and cause of mortality in high-energy environments (Krumholz & Jadot, 2009). In the long run, established mangroves are likely to return the ecosystem functions by facilitating soil accretion, minimizing wave energy and encouraging biodiversity and thus, return on investment (Beck et al., 2022; Su, Friess, & Gasparatos, 2021). Successful restoration demands commitment by the communities that utilize and are associated with mangroves to ensure the long-term monitoring and maintenance of planted mangroves as the work is quite involving (Ounanian et al., 2018; Gann et al., 2019). While opportunities for further research exist, the findings of this study, if adopted provides practical application of a restoration approach in areas, where strong wave energies have made mangrove restoration futile under the changing climate and in disaster risks reduction in line with the UN's Decade on Ecosystem Restoration (2021 – 2030) (Waltham et al., 2020).

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Publication

1. Kinya, G., Kairo, J. G., Nyoike, R. N., Nguu, J. G., Githinji, B. K., & Githaiga, M. N. (2024). Eco-Engineering Mangrove Restoration at Gazi Bay, Kenya. *Diversity*, *16*(3), 135. https://doi.org/10.3390/d16030135

Conference Presentation

1. Poster presentation at the 12th WIOMSA Scientific Symposium, hosted from October 10th to 15th, 2022, at the Boardwalk Convention Centre in Nelson Mandela Bay, South Africa.

APPENDICES

Appendix 1: Plate (a): Field activity on sediment coring. **Plate (b):** Field activity on ground elevation measurements

Appendix 2: Plate (a) and (b): Field activity on subsequent sediment refilling and plant monitoring


a


b