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ECOLOGICLA MONITORING OF FORESTED MANGROVE AREAS IN KENYA

BY

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Abstract

The first objective of this present study was to determine the natural regeneration potential of reforested mangrove stands at Gazi Bay, Kenya in terms of successive regeneration classes and assess contribution of previously recruited saplings to the overall structural stand development of the plantations. The second objective was to assess seedling establishment and population structure of seedlings of a known age to determine the average population of these seedlings, which reaches the sapling stage and thus subsequently contributes to stand complexity. Regeneration classes (RCs) were categorized into three classes based on height i.e. *RCI < 40cm, RCII < 40.1 – 150cm* and *RCIII > 150cm* but with dbh of less than 2.5cm. For the first objective, the study was conducted in two reforested monospecific mangrove stands (*Sonneratia alba* and *Rhizophora mucronata*), while for the demographic study, four natural and one reforested *Ceriops tagal* stands were used. All the replanted stands were 8 yr old. For environmental parameters, respective bare and natural sites were used as controls for the first study, while for vegetation structure and seedling/sapling recolonisation, the respective natural sites above were used as controls. Environmental parameters were not measured in the second study, due to lack of an appropriate bare control. The criteria for selection of these control sites was based on physical proximity, site history and inundation class. Interstitial water salinity and temperature were generally similar among the *S. alba* forest sites, while salinity differed significantly among the three sites of the *R. mucronata* forest with the bare site having the highest salinity (56 ± 2), and the natural site had the lowest (36 ± 0.4). Temperature was similar among the reforested and natural sites of the *R. mucronata* forest, which was significantly lower than that in the comparable bare site. pH, NO^-_3 and PO^-_4 concentrations generally didn't differ significantly among sites within the two forests. Moisture content was highest in natural sites and lowest in bare sites within mangrove types. Organic matter content significantly differed among sites within mangroves types with the bare sites having the lowest, while the natural sites had the highest. Clay content was similar among the reforested and natural sites within the *R. mucronata* forest, whereas the bare site had the lowest. For the *S. alba* forest, the natural site had a higher clay content than both the bare and reforested sites which had a similar content. *R. mucronata* was the most dominant species within both its reforested (97%) and natural (84%) sites and this species also had the highest stem density of 3,022 ± 228 stems ha^{-1} and 1,502 ± 191 stems ha^{-1} in the reforested and natural sites respectively. The *R. mucronata* reforested stand had four adult mangrove species (the same species richness as the natural stand), which contributed to a higher complexity index than the *S. alba* reforested stand, which was monospecific for the adult tree species. For juvenile recolonisation, species richness and type were similar among the *R. mucronata* sites (reforested and natural), although the total juvenile density was significantly higher ($p < 0.000$) in the natural stand (7,390 ± 660) as compared to the reforested stand (2,048 ± 667). The converse was true for the *S. alba* stands where the reforested stand had a higher juvenile density (5,704 ± 647) than the respective natural stand (1,008 ± 194). Results therefore suggest that mangrove reforestation has modified site conditions and encouraged recolonisation of new/nonplanted mangrove species. Previously recruited seedlings have also grown into adult trees, which contributed to the higher species diversity and complexity index in the *R. mucronata* reforested stand, which was initially monospecific.

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INTRODUCTION

Increasing awareness of the true value of mangrove ecosystems has led to renewed efforts to protect and conserve them (Field 1996, Macintosh et al. 2002) against a backdrop of widespread degradation (Field 1995, WRI 1996, Spalding et al. 1996, Kairo et al. 2001). Degradation in this context refers to a scenario where forest cover has been adversely affected by unsustainable harvesting of wood (and/or nonwood forest products) and/or conversion to other uses so that its structure, processes, and functions are altered beyond the short-term resilience of the ecosystem (ITTO 2002). As a result, mangrove restoration has been initiated in various parts of the world (Kairo 1995, Aksornkoae 1996, Qureshi 1996, Saenger 1996, Siddiqi and Khan 1996, SFFL 1997). Restoration is defined as the act of bringing an ecosystem back into, as nearly as possible to its original condition, renewing it or bringing it back into use (Field 1996). Chapman and Underwood (1997) and Simberkoff (1990) argue that returning an ecosystem to exactly its original condition is unrealistic. However, an approximation of the original system may be possible and sufficient, given that habitats are subject to a high degree of natural variation. Of most importance however, is that ecosystem functions (e.g. nutrient recycling, soil erosion control, habitat provision, water quality maintenance and storm wave protection among others) should be restored (Kaly and Jones 1998). Forest restoration should thus aim at assisting natural processes of forest recovery in a way that the species composition, stand structure, biodiversity, functions and processes of the reforested forest will match, as closely as feasible, those of the site-specific original forest (ITTO 2002**.** Unfortunately, mangrove reforestation has often been carried out by simply planting mangrove seedlings without adequate site assessment, or subsequent evaluation at the ecosystem level (Field 1996). Moreover for economic reasons, mangrove reforestation efforts are often limited to only one or two species. This raises questions regarding habitat change and reduced ecological function in mangrove plantations compared to natural mixed species mangrove forests (Macintosh 2002). Species diversity is widely regarded as being important in maintaining genetic richness, ecological functioning and the resilience of the ecosystem (Schultz & Money 1993, and Heywood 1995 in Macintosh 2002). Species diversity appears to have two major roles in the self-reorganization of large-scale ecosystems. First, it provides the units through which energy and materials flow, giving the system its functional properties. This increases ecosystem productivity by utilization of more of the possible pathways for energy flow and nutrient cycling. Secondly, diversity provides the ecosystem with the resilience to respond to unpredictable environmental strains e.g. flooding, siltation, disease outbreak etc. Species, which keep the system resilient in the sense of absorbing perturbation, are important in the release and reorganization phases (Costanza et al. 1997)

Vegetational complexity in mangroves is a function of tree species richness among other variables (Holdridge et al. 1971, Bosire 1999, Kairo et al. 2002). This thus suggests that stand complexity is low in reforested monoculture stands, especially if no successful natural regeneration occurs in these stands to encourage colonization by new/nonplanted species. Successful recruitment of new mangrove species into reforested monospecific stands is a function of diaspore availability and dispersal potential (Rabinowitz 1978a, McGuinness 1997, Clarke et al. 2001), predation (Smith 1988, Dahdouh-Guebas et al. 1997, 1998, Lee, 1998, Clarke and Kerigan 2002), physical conditions e.g. local hydrodynamics, salinity, irradiance etc (Ball 2002, Thampanya 2002, Bosire submitted) and competition (Osunkoya and Creese 1997, Sherman et al. 2000, Ball 2002) among other factors. Seedling recruitment is an important factor in the eventual determination of forest structure (Ball 2002). This therefore, demands continuous monitoring of reforested stands to assess their natural regeneration potential and subsequent vegetational complexity.

The objective of this present study was to determine the natural regeneration potential of reforested mangrove stands at Gazi Bay, Kenya in terms of successive regeneration classes and assess contribution of previously recruited saplings (Bosire 1999) to the overall structural stand development of the plantations.

MATERIALS AND METHODS

Site selection

This study was conducted in two reforested monospecific mangrove stands (*Sonneratia alba* and *Rhizophora mucronata*), which were 8 yr old. For environmental parameters, respective bare and natural sites were used as controls. The criteria for selection of these control sites was based on physical proximity, site history and inundation class as described by Bosire et al. (submitted). For vegetation structure, the respective natural sites above were used as controls.

Environmental parameters

The *S. alba* reforested stand seemed to have some vegetational gradient with trees seaward taller than those landward. As a result, stratified sampling was used by making two transects running parallel to the shoreline. Half of the *R. mucronata* plantation was dominated with conspicuous rocks (dead corals), which must have affected spacing during planting and thus subsequent tree density. Based on this, the plantation was divided into two halves running perpendicular to the shoreline and transects (one in each half) made. Three quadrats of 5m x 5m were randomly made per transect. The same experimental protocol was applied to the controls. Environmental parameters were not measured in the *Ceriops tagal* stands, due to lack of an appropriate bare control.

For the above quadrats, interstitial water was collected by digging a hole into the soil of 5 – 15cm (depending on the inundation class and cover type) and an allowance of 5 min given for the water to accumulate. A Universal Multimeter P4 WTW was used to measure salinity, pH and temperature by inserting respective probes into the pore water. For nutrient (NO^-_3 and PO^-_4) analysis, sediment cores (of 6.4cm diameter) were taken to a depth of 10cm and extraction of seawater done in the lab using KCl. Analysis was done using conventional seawater analysis methods by Parsons et al. (1984) with a Technicon Analyser II system. Three subsamples were taken per quadrat for the above environmental parameters (i.e. 3 subsamples x 3 quadrats x 2 transects). An extra sediment core was taken per a quadrat for organic matter and grain size analysis. This core was dried at 80ºC until constant dry weight was obtained. A pre-determined weight of dried sediment was ashed at 450ºC for 24 hr in a muffle furnace to get the organic matter content. About 20g each of the samples dried at 80ºC was weighed and the organic matter in the samples removed by digesting them in an excess of 30% diluted technical H_2O_2 as an oxidizing agent after which the samples were rinsed with demineralized water until a more or less stable suspension was obtained (Wartel et al. 1995). Grain size analysis was done using a combination of dry sieving and sedigraph method as outlined by Wartel et al. (1995). The sedigraph determines the size distribution of particles dispersed in a liquid assuming settling of particles according to Stokes' law (Arnold 1986). For grain size ranges, the unified soil classification system was used (Robert et al. 1997).

Floristic composition and juvenile recolonisation

Phytosociological methods outlined in Cintron and Schaeffer-Novelli (1984) were used in studying vegetation structure. For the *S. alba* stands, based on the gradient described above, three parallel transects were made. Six 5m x 5m quadrat were made per transect (i.e. 6 quadrats x 3 transects) and tree height and dbh for all trees with dbh greater than 2.5cm measured. Nine random quadrats were made per a transect (for the two transects) in the *R. mucronata* stands. From the data generated, absolute tree density, basal area and frequency were calculated. Relative derivatives (density, dominance and frequency) of these absolute values were computed from which the importance values of the stands were calculated. Complexity index was calculated according to Holdridge et al. (1971). This index combines all the measured stand structural attributes (stem density, DBH calculated into basal area, mean height and number of species) to show how complex or structurally developed a stand is. All trees with dbh of less than 2.5cm were classified as juveniles into the following categories (regeneration classes-RC) based on height: RCI < 40cm, RCII 40 – 150cm, RCIII > 150cm (Kairo 2001). The species and abundance of mangrove juveniles in the above quadrats were identified and counted accordingly.

Mangrove demography

Mangrove demography studies have been conducted in various parts of the world to assess the spatial pattern and population structure of seedlings since seedling recruitment determines subsequent forest structure (Mohammed 1991, Osunkoya and Creese 1997, Hegazy 1998, Ball 2002). But such studies have not been conducted along the E. African coast and especially in replanted monospecific stands, where seedling recruitment is so essential to enhance stand complexity. For the purpose of this study, all regeneration class I (RCI) seedlings in the *R. mucronata* reforested and natural stands in ten 25m^2 randomly chosen quadrats for each stand were uprooted so that the new ones, which established afterwards were of a known age. This was also applied in *C. tagal* reforested stand (8 yrs old, 0.6 ha) and in four different *C. tagal* natural stands (five 25 m^2 random quadrats were chosen for each stand). For the latter natural stands, two were at the western creek (1 and 2) and two on the eastern creek (4 and 5) The reforested site was labeled as site 3. This was however, not done in the *S. alba* stands since natural regeneration in the *S. alba* natural was found to be too low (refer results). This experiement was intiated in May 2002 when the two species reach their peak in propagule production. The marked quadrats were given two months up to July for seedlings to settle and establish. By July, no RCI had established in the *R. mucronata* stands, but some had in the *C. tagal* stands. The RCI seedlings in the latter were tagged with dynamo tape tied with fishing line, which are both resistant to salt water corrosion. Growth measurements i.e. height, diameter, number of leaves, number of nodes, density and status of the seedling (eaten by crabs, withered, and not found) were started in August. Even by August, no seedlings had established in the *R. mucronata* stands, which necessitated termination of the experiment in these stands. It is intended that these demography measurements be done for two years in order to determine on average, how many of the RCI seedlings reach the sapling stage and are therefore likely to contribute to the structural development of the stands. As a result, findings from *C. tagal* stands are discussed separately under mangrove demography.

Statistical analysis

Analyses of environmental variables, tree densities, basal areas and pooled regeneration class densities among stands and sites was done using two-way anova (replication with fixed effect), while multiple comparisons within stand were analysed with Tukey's Honest Significant Difference (THSD) test. Individual regeneration classes densities among stands and sites was analysed using the Univariate General linear model three-way anova. Species diversity was calculated using Shannon diversity index (Kent and Coker 1992) and differences tested with one-way anova.

RESULTS

Environmental parameters

Interstitial water salinity was similar ($p > 0.05$) within the *S. alba* forest, except for the reforested and natural sites, which were significantly different ($p < 0.05$), with salinity in the natural site being lower (Table 1). For the *R. mucronata* forest, salinity differed significantly among the three sites with the bare site having the highest salinity (56 ± 2), while the natural site had the lowest (36 ± 0.4). Among the two forests (stands) and cover types (sites), salinity was significantly different ($p < 0.000$), with the interaction term (Table 2) accounting for the highest variation (31%). Salinity was generally higher in the *R. mucronata* stand. pH was similar among sites within the two forests types, except for the *R. mucronata* reforested and natural sites, which were significantly different (Table 1). pH differed significantly ($p < 0.000$) among the two mangrove stands, but was similar among the cover types (Table 2). Interstitial water temperature was similar in the *S. alba* sites. Among the *R. mucronata* sites, the reforested and natural sites had similar temperature, which was significantly lower than in the bare site. Temperature was significantly different among the mangrove stands (Table 2), but similar among cover types, with the stand term accounting for the highest variation (53%). NO^-_3 concentrations were similar in sites within mangrove forests, apart from the reforested and natural sites of the *S. alba* forest, which differed, with the natural site having a higher concentration. PO^-_4 concentration was similar among the *S. alba* sites, while for the *R. mucronata* forest, the reforested and natural sites had a similar concentration, which however, differed significantly from that in the bare site. Moisture content was highest in natural sites and lowest in bare sites within the two mangrove forests. Organic matter content significantly differed among sites within mangroves forests, save for the *S. alba* bare and reforested sites which had similar but lower organic matter content than the corresponding natural site. For the *R. mucronata* forest, the bare site (3 ± 0.6) had the lowest organic matter, while the natural site had the highest (39 ± 2.2). Clay content was similar among the reforested and natural sites within the *R. mucronata* forest, whereas the bare site had the lowest. In the *S. alba* forest, the natural site had a higher clay content (21 ± 4) than both the bare (9 ± 1.9) and reforested (9 ± 0.7) sites, which had a similar content. Moisture, organic matter and clay contents significantly differed among mangrove stands and cover types ($p < 0.000$), with the *R. mucronata* stand having higher contents of each variable.

Floristic composition and juvenile recolonisation

R. mucronata was the most dominant species within both its reforested (97%) and natural (75%) sites (Table 3). This species also had the highest stem density of 3,022 ± 228 stems ha^{-1} (97%) and 1,502 ± 191 stems ha^{-1} (84%) in the reforested and natural sites respectively. As a result, this species had the highest importance values (IV) in both sites (i.e. 279% and 221% for the reforested and natural sites respectively), hence structurally contributing most to the individual stands as compared to the other species. The reforested and natural stands of *R. mucronata* had four species each for the adult trees, while the *S. alba* reforested stand was monospecific and the natural stand had two species (*S. alba* and *A. marina*). A part from the species richness and diversity between the *R. mucronata* stands which were similar, the adult tree species recruited into the reforested stand were the same as those found in the natural stand except for one species which occurred in one stand and not in the other (*X. granatum* and *S. alba*). Stem densities were significantly higher ($p < 0.000$) in both reforested sites as compared to their respective natural sites. The converse was true for the basal areas (Table 3). Among the two mangrove forests (*R. mucronata* and *S. alba*), basal areas significantly differed with the *R. mucronata* having a higher basal area (Table 4). There was however, no significant difference between cover types (sites). Stem density also differed significantly ($p < 0.000$) among the two forests, with the *S. alba* forest having a higher density (Table 4). Due to their higher mean heights and basal areas, the natural stands had higher complexity indices than the reforested stands. The *R. mucronata* reforested stand had a higher complexity index than the *S. alba* reforested stand due to the higher species richness in the former. The same was also true for the natural stands. A graphic relationship of tree density and diameter size class (Fig 1 – 2) showed a reversed “J” distribution, which was strongest in the *S. alba* reforested stand ($R^2 = 0.9973$) and lowest in the *R. mucronata* natural stand ($R^2 = 0.8353$). This relationship wasn’t presented for the *R. mucronata* reforested stand since all the stems in this stand had a dbh of less than 8cm. An exponential relationship between tree height and dbh (Fig 3 – 4) indicated that in terms of the size indicators, trees were more spread out in natural stands, while they were more clustered together in reforested stand.

Juvenile recruitment in terms of species, density and successive regeneration classes (RCs) into the different mangrove stands was varied (Table 5). The *S. alba* reforested site had a significantly higher ($p < 0.000$) juvenile density (5,704 ± 647 ha^{-1}) than its respective natural site (1,008 ± 194 ha^{-1}), while the natural site of *R. mucronata* had a significantly higher density (7,390 ± 660 ha^{-1}) than its reforested site (2,048 ± 667 ha^{-1}). The *S. alba* reforested site had a significantly higher density than the *R. mucronata* reforested site, while the *S. alba* natural site had the lowest density of the four sites. Juvenile species richness and diversity were similar in the *R. mucronata* stands (reforested and natural), while the converse was true for the *S. alba* stands where richness and diversity were significantly higher in the reforested stand. The crown species were the most recruited in their respective stands, with *R. mucronata* being the second most recruited species in the *S. alba* stands. In general, juvenile densities were significantly different ($p < 0.05$) among the two mangrove stands for the pooled and individual regeneration classes (Table 6), not significantly different ($p > 0.05$) among the cover types (sites), while the density of RCI and RCII was similar and significantly higher ($p < 0.000$) than the density of RCIII. Interactions between Stand*Type, stand*RC, Type*RC and Stand*type*RC were significant with the within group term accounting for the greatest variation (50%). No juveniles were however, found in the bare sites, physical proximity, similar site history and inundation class as the respective reforested and natural sites notwithstanding.

Mangrove demography

There was an overall significant difference ($p < 0.05$) in adult tree density, canopy cover and basal area among the *C. tagal* sites (Tables 7 and 8). The difference in adult tree density was contributed by differences between site 1 and 3, and site 3 and 4, while differences among site 1 and 2, 1 and 4, 2 and 3, and 3 and 4 contributed to the highly significant difference ($p = 0.002$) in basal area. The replanted stand (site 3) though only 8 yrs old, had a similar basal area to sites 1 and 5, which may suggest that the stand had a similar wood volume as the two natural stands since even their mean heights were close. There was no significant difference ($p = 0.142$) in juvenile densities among the fives sites (Table 7 and 8), suggesting that they have a similar natural regeneration potential.

RCI seedling establishment by Aug, 2002 was highest in site 1 (5,277 seedlings ha^{-1}), followed by site 3/replanted (3,756 seedlings ha^{-1}), site 5 (2,914 seedlings ha^{-1}), site 4 (1,781 seedlings ha^{-1}) and lastly site 2 (1,230 seedlings ha^{-1}). By March 2003, the trends remained the same but with varying seedling mortalities (Fig 5). Mortality by March was highest in site 2 (47%), followed by site 1 (33%), Site 4 (16%), site 3 (12%) and was lowest in site 5 (10%). Height increment by March was highest in site 3 (4cm), followed by site 5 (3.5cm), site 1 (2cm), site 4 (1.8cm) and lastly site 2 (0.7). Initial height therefore did not seem to positively influence height increment rate. Generally *C. tagal* though the second most dominant species in Gazi bay (after R. mucronata), is normally stunted and with lower growth rates as compared to *S. alba* and *R. mucronata* species for instance, which have growth rates of 1m and 0.8m per yr respectively. It is however, necessary to follow the spatial pattern and population structure of these seedlings for 2 yrs to determine on average the population of RCI seedlings, which is likely to reach the sapling stage. Potential causes of mortality e.g. predation, desiccation and unfavourable hydrodynamics will be investigated over this period.

Discussion and Conclusion

Interpreting relationships between mangrove community structure and environmental variables is necessary for better understanding of the ecology of the ecosystem (Ashton & Macintosh 2002). Studying such variables in areas, which have suffered from degradation and those that have been reforested may help in explaining the changes such sites have undergone and thus their ecological functioning. This information is useful in making management decisions especially in the determination of species to be replanted in degraded sites. Many studies have attempted to explain the influence of biotic and abiotic variables on mangroves regeneration dynamics, which include dispersal potential (Rabinowitz 1987a, McGuinness 1997, Clarke et al. 2001), predation (Smith 1988, Dahdouh-Guebas et al. 1997, 1998, Lee, 1998, Clarke and Kerigan 2002), physical conditions e.g. local hydrodynamics, inundation regimes, geomorphology, salinity, irradiance, soil redox potential, pore water sulphide concentration etc (Watson 1928, McKee 1993, Matthijs et al. 1999, Ball 2002, Thampanya 2002, Bosire et al. submitted) and competition (Clarke and Myerscough 1993, Osunkoya and Creese 1997, Sherman et al. 2000, Ball 2002) among other factors.

Mangrove degradation is thus likely to modify some of the above site conditions in a way that may limit natural regeneration, which may explain the apparent lack of regeneration in the bare sites. Interstitial water salinity and temperature were found to be significantly higher in bare sites than reforested and natural sites within mangrove forests. This was more pronounced in *R. mucronata* sites (which are inundated only during spring high tides) where salinity and temperature were 56 ± 2 PSU and 33 ± 0.3^{0}C in the bare site, 36 ± 0.4 PSU and 30 ± 0.4^{0}C in the natural site respectively. These variables were however, similar among the *S. alba* forest sites probably due to the fact they are in inundation class I, which is normally under water during all high tides. This thus suggests that mangrove degradation leads to exposure of mangrove soils to intense heat (Ashton & Macintosh 2002, Bosire et al. 2003), which leads to elevated soil salinities and temperature. Desiccation/irradiance has been found to be one of the factors limiting mangrove regeneration (Clarke and Myerscough 1993, Ball 2002). pH was similar among sites within both forests. This may be attributed to the buffering effect of seawater. pH was however, significantly different among the two forests which may be explained by the varying inundation regimes. It was overall higher in the *S. alba* forest, which is frequently inundated than in the *R. mucronata* forest, which is inundated only during spring tides. Organic matter content was significantly high in the *R. mucronata* forest than in the *S. alba* forest, which may also explain the relatively higher H^+ concentration. Within the *R. mucronata* forest, organic matter was highest in the natural site (39 ± 2.2%) intermediate in the reforested site (25 ± 1.2%) and lowest in the bare site (3 ± 0.6%). This thus confirms the importance of mangrove litter in contributing to the productivity of the ecosystem. The observations obtained in this study are similar to those obtained by Bosire et al. (submitted) who did a pioneering study on the ecology of these mangrove plantations. Moisture content was highest in natural sites and lowest in bare sites within respective mangrove forests, which seem to emphasize the importance of mangrove cover in reducing sediment water loss, which has a moderation effect on both interstitial water salinity and temperature. Clay content was similar among the reforested and natural sites within the *R. mucronata* forest, whereas the bare site had the lowest. In the *S. alba* forest, the natural site had a higher clay content (21 ± 4) than both the bare (9 ± 1.9) and reforested (9 ± 0.7) sites, which had a similar content. Mangrove deforestation therefore seems to modify local hydrodynamics due to the removal of the inherent root network, which subsequently accelerates soil erosion. The dense mangrove root network normally slows down the energy of the incoming tidal water, hence facilitating settling of resuspended sediments and this most likely has contributed significantly in increasing the clay content in the *R. mucronata* reforested site. Clay, moisture and organic matter contents were positively correlated in all sites.

Mangrove reforestation has modified site conditions, which has most likely led to juvenile recruitment into the monospecific stands. No natural regeneration was found in the bare/clear-felled control sites. A number of mangrove species had however, colonized the reforested sites. The *S. alba* reforested stand had a higher ($p < 0.000$) juvenile species diversity (H = 0.333, J = 0.536) and density (5,704 ± 647 juveniles ha^{-1}) than its natural control (H = 0.109, J = 0.257 and 1,008 ± 194 juveniles ha^{-1} respectively). Of the juvenile species colonizing the reforested stand, *S. alba* was the most dominant (35%), followed by *R. mucronata* species (31) and *C. tagal* (19%). Though the latter two species are the most dominant species in Gazi Bay as a whole, they may not thrive in this inundation class due to long hours of inundation and banacle infestation (Kairo 1995, Kitaya et al. 2002). Bosire et al. (submitted) working in the same area five yrs ago found that the natural site still had a significantly lower density of juveniles and attributed this to the harshness of the habitat for seedling survival due to exposure to stronger wave attack and higher tidal velocities. This was implied from the high sand content (89%) they found in this site, which suggested high erosion rates. In the present study, the reforested site had a lower clay content (9%) than the natural site (21%), suggesting a higher erosion rate in the former, which still had a higher juvenile density. This seems to negate the conclusion by Bosire et al (submitted) based on granulometry, but does not rule out the model due to the relative exposure of this site to the open sea. The natural site also had a dense growth of pneumatophores (pers observations), which may have contributed to the higher clay content in this site as the dense root system slows the energy of the incoming water, hence trapping and facilitating subsequent settling of sediments. The dense root system may however, cause "choking" of seedlings, which may otherwise settle and establish. There was also a dense mat of algae covering this site, which may reduce the surface area on which the seedlings could settle or even hinder actual seedling settlement (Clarke & Myerscough 1993). These proposed models however, require field experimental testing on their role as potential regeneration constraints. The present study confirms that natural regeneration is limited in this *S. alba* natural site, which suggests that the site requires special management approaches. This may include zoning it for protection from further exploitation when the present ban on mangrove exploitation is lifted. The *R. mucronata* natural stand had a higher juvenile density than the reforested stand, but with a similar species diversity ($p = 0.358$). This implies that in terms of species diversity, the reforested stand is functioning as the natural control. But the regeneration potential of the latter is low, though it is adjacent to the natural stand as a source of propagule supply. The reforested stand also reproduced for the first time when this sampling was being done, which rules out propagule supply as a regeneration constraint and so are other factors like unfavourable hydrodynamics and desiccation. The most likely factors may be propagule predation and competition for light as canopy cover was higher in the reforested stand due to the higher stem density. The former factor is normally relatively intense under closed canopies (Osborne & Smith 1990). This is because propagule predators prefer, dense canopies as compared to gaps within forest stands. As a result, seedling performance is better in gaps (Osunkoya and Creese 1997, Ball 2002, Clarke and Kerrigan 2002). This model requires testing in the Kenyan mangrove plantations especially after silvicultural treatment (thinning) in one half of this plantation, which has led to the opening up of the canopy. Because of the higher species diversity in this stand for the adult trees, it had a higher complexity index than the *S. alba* reforested site though they are of the same age. But the latter stand had a higher basal area (8.5 m^2 ha^{-1}) than the former (4.99 m^2 ha^{-1}), which can be attributed to the higher growth of *S. alba* species (Kairo 1995). The higher species diversity in the reforested stand of *R. mucronata* for the adult trees suggests that the juveniles found by Bosire et al. (submitted) five yrs ago, have matured and have been recruited as adult trees leading to the higher complexity index. This was not the case in the *S. alba* reforested stand, implying that survival of the juveniles recorded in this stand during the past study was poor, or their growth rates are low in this inundation regime.

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Table 1. Site averages (Mean ± SE) of sediment characteristics in plots with matched natural © and reforested (Ref. - b) stands as well as bare (a) controls for *S. alba* and *R. mucronata* forests. Presented are also Tukey multiple comparisons within each forest. Underlined sites were similar, while those not underlined were significantly different.

Stand	Site	Salinity (PSU)	pH	Temp (^{0}C)	NO$^-_3$ (µgat/l)	PO$^-_4$ (µgat/l)	Moisture (%)	Organic matter (%)	Clay (%)
S. alba	Bare	35 ± 0.3	6.9 ± 0.1	31 ± 0.2	57 ± 7.1	45 ± 6.4	21 ± 0.8	1 ± 0.5	9 ± 1.9
	Ref.	36 ± 0.2	6.6 ± 0.1	29 ± 0.2	47 ± 7.6	40 ± 3.7	25 ± 0.7	2 ± 0.4	9 ± 0.7
	Natural	34 ± 0.5	6.6 ± 0.1	30 ± 0.1	89 ± 6.9	36 ± 3.7	54 ± 1.8	11 ± 2	21 ± 4
		a b c	a b c	a b c	a b c	a b c	a b c	a b c	a b c
R. mucronata	Bare	56 ± 2	6.6 ± 0.1	33 ± 0.3	28 ± 3.8	89 ± 7.1	24 ± 1.4	3 ± 0.6	15 ± 2.6
	Ref.	38 ± 0.2	6.3 ± 0	30 ± 0.2	39 ± 10	11 ± 1.6	56 ± 1.2	25 ± 1.2	44 ± 2.5
	Natural	36 ± 0.4	6.7 ± 0.1	30 ± 0.4	35 ± 9	8 ± 1.3	70 ± 1.1	39 ± 2.2	43 ± 5.8
		a b c	a b c	a b c	a b c	a b c	a b c	a b c	a b c

Table 2. Two-way anova of the sediment characteristics in *R. mucronata* and *S. alba* forests (stands). The different cover types (bare, reforested and natural) were used as treatments.

Variable	Factor	SS	% variation	df	p
Salinity	Mangrove stand	706	29	2	0.000
	Cover type	667	27	1	0.000
	Interaction	767	31	2	0.000
	Error	318	13	30	
pH	Mangrove stand	2	23	2	0.000
	Cover type	0.25	3	1	0.210
	Interaction	2	23	2	0.000
	Error	4.5	51	30	
Temp	Mangrove stand	30	53	2	0.000
	Cover type	1	2	1	0.100
	Interaction	11	19	2	0.000
	Error	15	26	30	
NO^{-}_{3}	Mangrove stand	2981	8	2	0.140
	Cover type	7862	22	1	0.000
	Interaction	3020	9	2	0.140
	Error	21408	61	30	
PO^{-}_{4}	Mangrove stand	15237	50	2	0.000
	Cover type	160	1	1	0.290
	Interaction	10684	35	2	0.000
	Error	4125	14	30	
Moisture	Mangrove stand	2450	18	1	0.000
	Cover type	9666	69	2	0.000
	Interaction	1250	9	2	0.000
	Error	614	4	30	
Organic matter	Mangrove stand	3069	42	2	0.000
	Cover type	2772	38	1	0.000
	Interaction	1140	16	2	0.000
	Error	331	5	30	
Clay	Mangrove stand	3897	40	1	0.000
	Cover type	2547	26	2	0.000
	Interaction	1234	13	2	0.000
	Error	2008	21	30	

Table 3. . Absolute (and relative) adult tree density, basal area (and derived % dominance) and absolute (as well as relative) frequency of mangrove species in natural and reforested stands of *R. mucronata* and *S. alba* forests. The relative values are expressed as percentage, while averages are given as Means ± SE.

Stand	Site	Species	Abs. density (rel.) ($n\ ha^{-1}$)	Basal area (dom.) ($m^2\ ha^{-1}$)	Abs. frequency (rel.) (%)	IV	Mean stand height (m)	Complexity index
R. mucronata	Ref	*R. mucronata*	3022 ± 228 (97)	4.9 ± 0.4 (97)	100 (85)	279	4.7 ± 0.2	2.91
		B. gymnorrhiza	63 ± 63 (2)	0.1 ± 0.1 (2)	6 (5)	9		
		C. tagal	9 ± 9 (0.3)	0.04 ± 0 (1)	6 (5)	6		
		S. alba	9 ± 9 (0.3)	0.01 ± 0 (0)	6 (5)	6		
		Sum	3102 (100)	4.99 (100)	118 (100)	300		
	Natural	*R. mucronata*	1502 ± 191 (84)	15.8 ± 2.4 (75)	94 (63)	221	7.5 ± 0.5	11.37
		X. granatum	216 ± 138 (12)	2.83 ± 2 (14)	17 (11)	37		
		B. gymnorrhiza	53 ± 29 (3)	2.4 ± 2.4 (11)	22 (15)	29		
		C. tagal	27 ± 15 (2)	0.02 ± 0 (0)	17 (11)	13		
		Sum	1796 (100)	21.1 (100)	150 (100)	300		
S. alba	Ref	*S. alba*	3453 ± 196 (100)	8.5 ± 1.3 (100)	100 (100)	300	4.8 ± 0.3	1.41
	Natural	*S. alba*	2212 ± 193 (99)	21.9 ± 0.2 (99)	100 (94)	292	7.2 ± 0.3	7.13
		A. marina	27 ± 27 (1)	0.24 ± 0.2 (1)	6 (6)	8		
		Sum	2239 (100)	22.1 (100)	106 (100)	300		

Table 4. Two-way anova analysis table of the basal area and adult tree density in the *R. mucronata* and *S. alba* forest stands.

Variable	Factor	SS	% variation	df	p
Basal area	Mangrove stand	24303	47	1	0.000
	Cover type	673	1	1	0.200
	Interaction	197	0	1	0.480
	Error	26868	52	68	
Tree density	Mangrove stand	1.7×10^8	38	1	0.000
	Cover type	1.7×10^7	4	1	0.040
	Interaction	2.2×10^5	0	1	0.810
	Error	2.6×10^8	58	68	

Table 5. Juvenile species and regeneration classes (RCs) in the *S. alba* and *R. mucronata* forest stands.

Stand	Site	Spp	1	2	3	Sum	% Prop.
S. alba	Reforested	*R. mucronata*	791 ± 138	980 ± 201	9 ± 9	1781 ± 348	31
		C. tagal	755 ± 236	306 ± 145	0	1061 ± 381	19
		S. alba	369 ± 95	378 ± 89	1259 ± 176	2006 ± 360	35
		A. marina	531 ± 132	90 ± 42	0	621 ± 174	11
		B. gymnorrhiza	171 ± 61	63 ± 40	0	234 ± 101	4
		Sum	2617 ± 261	1817 ± 370	1268 ± 176	5704 ± 647	100
		% Prop.	46	32	22	100	
	Natural	*R. mucronata*	225 ± 87	162 ± 152	18 ± 18	405 ± 162	40
		A. marina	90 ± 21	18 ± 4	0	108 ± 39	11
		S. alba	18 ± 18	54 ± 45	423 ± 111	496 ± 119	49
		Sum	333 ± 80	234 ± 157	441 ± 110	1008 ± 194	100
		% Prop.	33	23	44	100	
R. mucronata	Reforested	*R. mucronata*	534 ± 173	24 ± 14	372 ± 80	931 ± 128	45
		C. tagal	623 ± 596	0	16 ± 12	639 ± 604	31
		X. granatum	81 ± 46	16 ± 12	0	97 ± 56	5
		B. gymnorrhiza	332 ± 121	16 ± 12	32 ± 16	380 ± 124	19
		Sum	1570 ± 632	57 ± 26	421 ± 85	2048 ± 667	100
		% Prop.	77	3	21	100	
	Natural	*R. mucronata*	4500 ± 614	1246 ± 185	1101 ± 197	6848 ± 649	93
		C. tagal	105 ± 65	24 ± 26	49 ± 22	178 ± 69	2
		X. granatum	97 ± 47	8 ± 9	16 ± 12	121 ± 57	2
		B. gymnorrhiza	154 ± 69	65 ± 45	24 ± 14	243 ± 104	3
		Sum	4856 ± 439	1344 ± 179	1190 ± 210	7390 ± 660	100
		% Prop.	66	18	16	100	

Table 6. Three way-anova analysis for individual regeneration classes (RCI, RCII and RCIII) and two-way anova for pooled regeneration classes for the *S. alba* and *R. mucronata* forest stands

Variable	Factor	SS	% variation	df	p
Individual RCs	Stand	459	2	1	0.002
	Type	15	0	1	0.580
	RC	1369	7	2	0.000
	Stand * Type	5985	30	1	0.000
	Stand * RC	617	3	2	0.002
	Type * RC	668	3	2	0.001
	Stand * Type * RC	961	5	2	0.000
	Error	9980	50	204	
RCs pooled	Stand	1378	4	1	0.020
	Type	45	0	1	0.668
	Stand * Type	17955	50	1	0.000
	Error	16503	46	68	

Table 7. a). Basal area, density, percentage cover and height (Mean ± SE) and b). juvenile regeneration classes in *C. tagal* sites

a). Site	BA $m^2 ha^{-1}$	Density ha^{-1}	Cover (%)	Height (m)
1	2.7 ± 0.5	1,683 ± 268	69	2.1 ± 0.1
2	5.4 ± 0.5	2,396 ± 154	54	2.4 ± 0.1
3	2.7 ± 0.5	3.043 ± 512	75	2.2 ± 0.1
4	5.3 ± 0.7	1,878 ± 259	45	3.8 ± 0.1
5	3.5 ± 0.2	1,586 ± 74	45	3 ± 0.1

b).	RCI	RCII	RCIII	Sum
1	20,429 ± 9,128	14,958 ± 5,979	21,821 ± 18,894	57,208 ± 22,880
2	12,821 ± 7,955	8,256 ± 6,057	2,979 ± 467	24,055 ± 13,983
3	4,241 ± 972	453 ± 140	3,853 ± 524	8547 ± 1,223
4	5,957 ± 1,998	5,018 ± 3,863	3,432 ± 910	14407 ± 4,408
5	13,760 ± 2,840	4,338 ± 895	4,371 ± 899	22,469 ± 3,337

Table 8. Multiple comparisons of adult tree density, percentage cover, basal area and juvenile densities in the *C. tagal* sites.

(I) SITE	(J) SITE	I-J	Std. Error	Sig.
Adult stem density				
1	2	-712.2	426.356	0.473
	3	-1359.8	426.356	0.033
	4	-194.2	426.356	0.990
	5	97	426.356	0.999
2	3	-647.6	426.356	0.563
	4	518	426.356	0.743
	5	809.2	426.356	0.350
3	4	1165.6	426.356	0.084
	5	1456.8	426.356	0.020
4	5	291.2	426.356	0.958
Canopy cover				
1	2	15	3.847	0.007
	3	-6	3.847	0.538
	4	24	3.847	0.000
	5	24	3.847	0.000
2	3	-21	3.847	0.000
	4	9	3.847	0.174
	5	9	3.847	0.174
3	4	30	3.847	0.000
	5	30	3.847	0.000
4	5	0	3.847	1.000
Basal area				
1	2	-2.698	0.778	0.018
	3	-0.04	0.778	1.000
	4	-2.638	0.778	0.022
	5	-0.838	0.778	0.816
2	3	2.658	0.778	0.020
	4	0.06	0.778	1.000
	5	1.86	0.778	0.159
3	4	-2.598	0.778	0.024
	5	-0.798	0.778	0.840
4	5	1.8	0.778	0.181
5	1	0.838	0.778	0.816
Juvenile density				
1	2	-22300	9516.926	0.173
	3	-12652.4	9516.926	0.677
	4	-20713.6	9516.926	0.229
	5	-6792	9516.926	0.951
2	3	9647.6	9516.926	0.846
	4	1586.4	9516.926	1.000
	5	15508	9516.926	0.497
3	4	-8061.2	9516.926	0.912
	5	5860.4	9516.926	0.971
4	1	20713.6	9516.926	0.229
4	5	13921.6	9516.926	0.597
5	1	6792	9516.926	0.951