

Self-sustenance potential of peri-urban mangroves: a case of Mtwapa creek Kenya

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Mtwapa creek mangroves are situated in the vicinity of the populous town of Mtwapa and are surrounded by local communities living in the villages along the creek. Both factors pose potential pressure on these peri-urban mangroves hence compromising their sustainability and ability to support coastal livelihoods. This study therefore evaluated the sustenance potential of Mtwapa creek mangroves based on mangrove cover change, structural status and natural regeneration. This was carried out through assessment of tree abundance, dispersion pattern and natural regeneration status along belt transects in three forest patches (Gung'ombe, Kitumbo and Kidongo) in Mtwapa creek and cover change analysis of selected years. The studied sites were characterised by high stem densities with low basal areas, low utilisation classes and poor forms (poles requiring modification before use or unsuitable for construction accounting for 75.9% of the total stem density). Natural regeneration performance varied among species with *Avicennia marina* having low survival rates while the other species achieved $\approx 100\%$ recruitment to successive regeneration classes. Poles were mainly of form 3 (unsuitable for building) indicating selective harvesting. The results show that Mtwapa creek mangrove forest has sufficient natural regeneration but poor tree development thus hindering the forest's self-sustenance. Additionally and specific to Kidongo, proximity to human settlement contributed greatly to selective harvesting thus compromising the integrity of the forest.

Key words: Forest structure, natural regeneration, disturbances, stress, management

INTRODUCTION

Mangrove forests occupy intertidal areas of tropical and subtropical coasts, between high water and low water mark of spring tide (Tomlinson, 1986). These forests provide timber and non-timber products such as fuel wood, poles, fodder and fisheries resources to millions of people in the tropic (Saenger, 2002). Mangroves also buffer land from storms and provide safe havens for humans (Spalding et al., 1997). Ecologically, they serve as nursery grounds for fish, prawns and crabs as well as habitat for many other organisms. 70% of the population living in the Kenyan coastal villages depend on mangrove for construction (Wass, 1995) and fuel wood (Dahdouh-Guebas et al., 2000; Kokwaro, 1985; Rawlins, 1957). Despite the importance of these unique coastal tropical forests, they are among the most threatened biotopes in the world.

Mangroves form an ecotone between land and sea, and as such undergo constant natural disturbance due to the tidal cycles and its corollaries. Catchment processes (Kitheka 2002), global change and the multiple uses of mangroves set against the background of increasing demographic pressure in most coastal areas, only worsening the situation (Cicin-Sain and Belfiore, 2005). The coastal region of Kenya for instance constitute 8% of Kenya's population (Government of Kenya, 2009) and the growth rate is $>3\%$ which is considered high as per international standards (Government of Kenya, 2009). Overexploitation of mangroves has therefore been cited as a major threat to this resource, leading to a dramatic reduction in their areal coverage worldwide (Giri et al., 2011). This is majorly as a result of increasing demand

for its extractible goods necessitated by rapid population growth and urban expansion (Cicin-Sain and Belfiore, 2005). In particular, peri-urban mangroves are facing further degradation due to lack of planning and uncontrolled development in the shoreline and beach areas intended for tourist attraction (White et al., 2005). In Kenya, peri-urban mangroves also face the problem of pollution from direct discharge of domestic sewage from point sources into the mangrove forest especially at Tudor and Mtwapa creek (Government of Kenya, 2009).

Between 1983 and 1993 Mombasa port and surrounding waters experienced 391 680 tonnes of oil spills that affected mangroves of Port Reitz and Makupa creeks. In addition Conversion of peri-urban mangrove areas for other uses including creation of salt ponds and diversion of fresh water has also contributed to great loss of mangroves (Valiela et al., 2001; Duke et al., 2007). While mangroves of Kenya have had a legal status since Proclamation No. 44 of 1932, clear government policy guidelines are lacking to guide the management of this resource. The result is often non-sustainable exploitation resulting in either overexploitation or cryptic degradation.

To restore the degraded forests, reforestation efforts have been initiated in Kenya (Kairo, 1995; Okello et al., 2012), in Colombia (Elster et al., 1999) Thailand (Aksornkoe, 1996), Pakistan (Qureshi, 1996), Australia (Saenger, 1996) and Bangladesh (Siddiqi and Khan, 1996). However, an assessment of most of the large-scale mangrove restoration efforts illustrates that they seldom succeed (Lewis III, 2011). Since mangrove restoration is costly and human intervention is only recommended where the system has been completely altered (Clements, 1929; Sterner and Elser, 2002), the assessment of the existing mangrove status is crucial to help decide whether the restoration is necessary (Lewis III, 2005).

Forest structural studies have been conducted to characterize and monitor changes in various mangrove ecosystems worldwide (Satyanarayana et al., 2011; Cintrón and Schaeffer-Novelli, 1984; Holdridge et al., 1971; Pool et al., 1977), and in Kenya (Kairo et al., 2002; Kairo, 2001; Dahdouh-Guebas et al., 2004; Mohamed et al., 2009), in order to guide management. However, demand still exists for site specific structural surveys to inform the development and reviews of management plans as well as support silvicultural practices. This is especially important in the current dispensation in Kenya in which mangroves have been divided into blocks being managed by Community Forest Associations (CFAs).

Moreover, forest disturbances tend to be localized rather than a large scale phenomenon and as such vegetation structure may widely vary from one forest to the other (Hubbell, 1979). In addition, even within the same regions, forest areas may display strong differences in zonation (Dahdouh-Guebas et al., 2002). As a result, it would be important to consider each forest area as an independent entity. In this article, we assess the status of both mature trees and natural regeneration of Mtwapa creek mangroves

to determine the sustenance potential of the forest. We present results of tree abundance, dispersion pattern and natural regeneration as well as cover change of the forest. This would provide fact based information important in guiding sustainable management of peri-urban mangroves.

MATERIALS AND METHODS

Study site description

Mtwapa creek lays at the boundary of Mombasa and Kilificountiesto the South and North respectively. These are the most densely populated counties in the coast province with a population growth rate of 3.9%. The creek is adjacent to Mtwapa town and lies about 15 km North from Mombasa city. Like other towns in the two counties, both Mombasa and Mtwapa experiences net immigration of people from upcountry attracted by employment in the tourism sector and other industries including the port (Government of Kenya 2007). The two counties have therefore the highest coastal urban population accounting for 69% of the total individuals living in urban areas in the region (Government of Kenya, 2007; Government of Kenya, 2009). The creek is approximately 13.5km in length and opens to the Indian Ocean through a long narrow channel (Figure 1).

It consists of three forest patches (Gung'ombe, Kitumbo and Kidongo: named after adjacent large villages) which are situated further landward from the mouth (Figure 1) fed by three seasonal rivers (KwaNdovu, Kashani and Kidutani). There are steep slopes (covered with arable lands and a few patches of thickets and occasionally residences) facing the mangroves on both sides of the creek (Figure 1). There are also major hotel developments and tourist settlements along the creek. At the mouth; the creek receives untreated sewage from nearby beach hotels, Shimo la Tewa prison and Mtwapa municipality (personal observation). However, this gets diluted further downstream due to the length of the channel.

Field measurements

In March 2010, a reconnaissance survey and examination of aerial photographs of the area was carried out in order to identify study sites and the location of transects (Kairo et al., 2002). This was followed by a survey done along belt transects laid both perpendicular and parallel to the creek to ensure a good representation of the entire forest. A total of 26, 39 and 13 (10m by 10m) quadrats were surveyed in Gung'ombe, Kitumbo and Kidongo patches, respectively. The number of quadrats was determined based on the size of the forest patch and the ease of access. All mature trees with a butt diameter greater than 2.5cm were marked in each quadrat. Heights (ht) of the identified trees were measured using a Suunto clinometer, while stem diameters at 130 cm above ground (D130) were measured using a forest

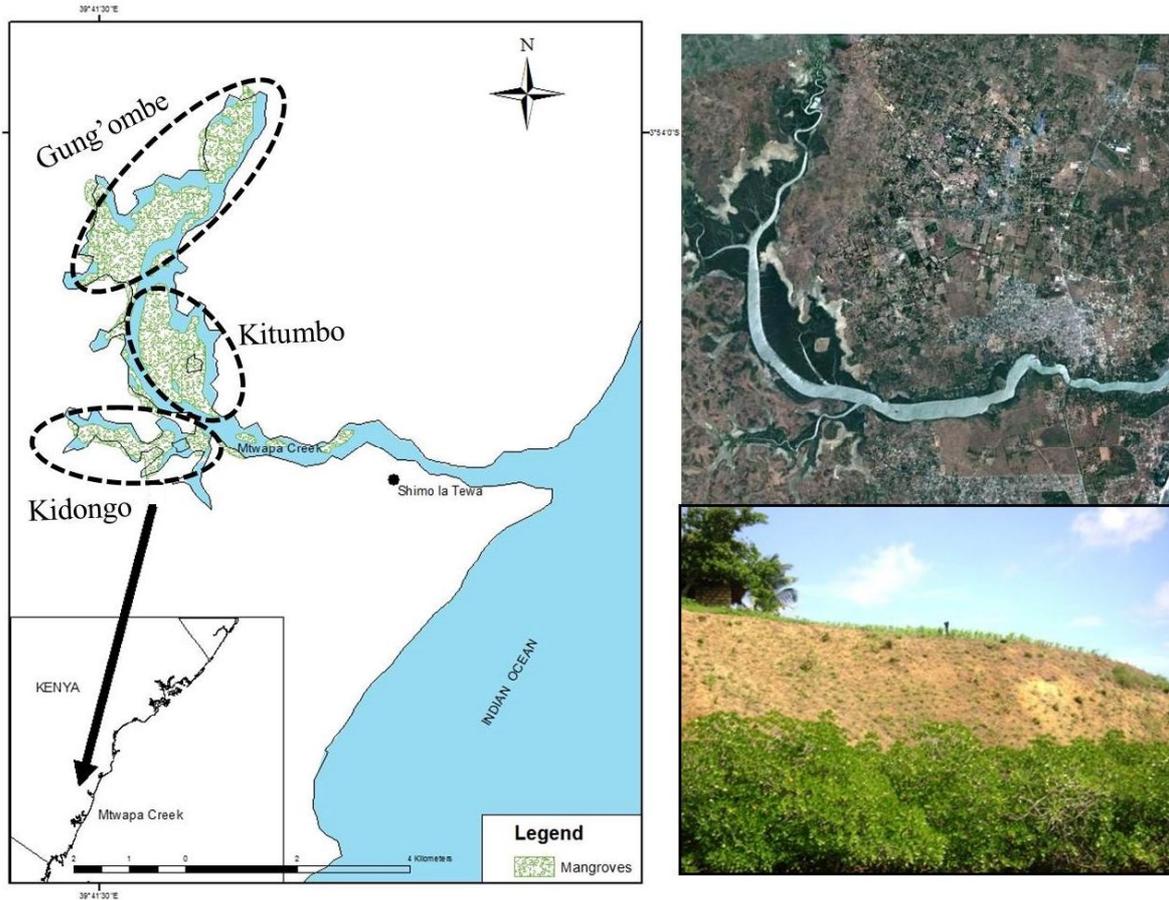


Figure 1. Kenyan coastline (inset), indicating the location of Mtwapa creek. The forest patches Kidongo, Gung'ombe and Kitumbo where the structural survey was conducted in 2010 March are delineated using dotted circles (Source KMFRI database). To the right is Google image showing human settlement around the creek and a picture of agricultural farms on the slopes facing mangroves.

calliper (sensu Brokaw and Thompson, 2000). D130 was measured in all species except for *Rhizophora mucronata* Lamk. (Rhizophoraceae), where stem diameter was measured 30 cm above the highest prop roots due to the architectural complexity of the genus (Dahdouh-Guebas and Koedam, 2006). In cases where the stem forked below 130 cm (for example, *Avicennia marina* (Forssk.) Vierh. (Acanthaceae) and *Sonneratia alba* (Sm.)), the individual 'branches' in a clump were treated as separate stems.

Pole usability in construction (quality) was assessed based on the form of the lead stem and assigned either form 1, 2 or 3. Form 1 stems denote those whose lead stem is straight and excellent for construction. Form 2 stems require slight modification before they are used for construction while the poles that were unsuitable for construction were assigned form 3 (Kairo, 2001). Stumps and standing dead trees were also counted in each quadrat as an indicator of exploitation and tree mortalities. Percentage vegetation cover per quadrat was estimated based on ease of ground visibility (from aerial view) and sky visibility, in quadrats with short trees and tall trees respectively (Zhou et al., 1998).

All trees with D130 below 2.5 cm were put under the category of natural regeneration and termed saplings

(Kairo et al., 2002). The number of different species of saplings was determined in sub-plots of 5m x 5m, within the 10m x 10m quadrats. The saplings were further grouped according to height classes where those less than 40 cm height were classified as regeneration class 1 (RCI). Regeneration class 2 (RCII) had heights between 40 and 150 cm, while all small trees with heights greater than 1.5 m but less than 2.5 m diameter were grouped as Regeneration class 3 (RCIII).

Data treatment

All our analyses are based on assumptions that; (a) natural regeneration represents the future forest and, (b) in the process of development natural regeneration will in turn experience the same level of disturbances as depicted by the status of the current standing mature trees. This is complimented by canopy change analysis to capture past exploitation.

Data analysis and graphical presentation were carried out using Excel and STATISTICA 8.0. After testing for homogeneity of variance using a Shapiro-Wilk's *W* test, a Kruskal-Wallis (a non-parametric test) was performed on stem densities of different size classes.

A Mann-Whitney U Test was used to determine if any correlations existed between vegetation cover per area sampled and stumps left as a result of cutting. Regeneration densities and ratios were calculated and correlation with canopy gaps done using Spearman rank correlation.

Structural attributes

Tree basal area (Ba, m² ha⁻¹) and stand density (De, stems ha⁻¹) (Cintrón and Schaeffer-Novelli, 1984; Mueller-Dombois and Ellenberg, 1974), were derived using equation 1A and 1B. Relative values of density, dominance and frequency were estimated using equations 2A, 2B and 2C (Dahdouh-Guebas and Koedam, 2006). The ecological importance (IV) of each species was calculated by summing its relative density, relative frequency and relative dominance (Curtis, 1956; Cintrón and Schaeffer-Novelli, 1984). The species with the highest IV in the population becomes the principal species.

$$\text{Eq. (1A). Basal area (Ba)} = \sum_{i=1}^m \sum_{j=1}^{n_i} \frac{B_{aij}}{n_i}$$

$$\text{Eq. (1B). Stand density (De)} = \sum_{i=1}^m \frac{1}{\left(\sum_{j=1}^{n_i} \left(\frac{d_{ij}}{n_i}\right)^2\right)}$$

$$\text{Eq. (2A). Relative De (De}_r\text{)} = \frac{100n_i}{\sum_{i=1}^m n_i}$$

$$\text{Eq. (2B). Relative dominance (Do}_i\text{)} = \frac{100B_{ai}}{Ba}$$

$$\text{Eq. (2C). Relative frequency (F}_i\text{)} = \frac{100F_i}{\sum_{i=1}^m F_i}$$

Where:

n_i is the number of trees sampled for species i ;
 j is the tree number, m is the number of species;
 F_i is number of quadrats in which species i is represented, multiplied by 100.

Stems of mature trees were further grouped into utilisation classes with different categories of sizes for inventory purposes. The naming of the classes is in Kiswahili language and refers to different uses of the poles. The classes include; fito, pau, mazio, boriti, nguzo, and vigingi with diameter ranges of 2.5-3.5cm, 4.0-7.5cm, 7.5-11.5cm, 11.5-13.5cm, 14.0-20.0cm, and 20.5-35.0cm respectively (Roberts and Ruara, 1967). An extra class, 'mbao', with D130 greater than 35 cm was also included in this study, to cater for large *Avicennia marina* trees.

Morisita dispersion index

Dispersion was determined using Morisita's dispersion

Index ($I\bar{d}$) (Morisita, 1959). This was based on the analysis of spatial pattern of adults and saplings carried out in the 10 m x 10m quadrats along a single selected transect.

$$\text{Eq. (3A). Morisita index (I)} = q' \times \delta$$

$$\text{Eq. (3B) } \delta = \frac{\sum_{i=1}^{q'} n_i(n_i-1)}{N(N-1)}$$

$$\text{Eq. (3C). } \sum n(n-1) = \sum n^2 - N$$

Where:

q' , is the number of quadrats;

n_i , is the number of individuals per species in the i th quadrat, and

N , is the total number of individuals in all q' quadrats.

If $I\bar{d} > 1$, the population is clustered, if $I\bar{d} < 1$, the population is randomly dispersed and if $I\bar{d} = 1$, the population is evenly dispersed.

Shape of the population distribution

For predictive purposes in forest management, it is often desirable to understand the shape of the underlying distribution of the population. A forest is usually assumed to be stable when a given D130 distributions can be maintained over a period of time. This allows a given harvesting schedule to be maintained repeatedly without compromising stability and as such considered to be sustainable (Sterba, 2004). Most management schemes of uneven-aged forest strive to attain a reverse-J shaped curve as a way of ensuring stability (Ducey, unpublished). De Liocourt's model which was used to harmonise stand densities in this study is one such example. De Liocourt's model is based on the assumption of a constant "q" obtained as a ratio of successive diameter classes (Equation 4).

According to this model, the ratio between the numbers of trees in successive diameter classes of uneven aged stands is approximately constant for a given forest (Clutter et al., 1983; Cancino and Von Gadow, 2002). This model has been popularised in several studies due to its simplistic nature and the fact that it allows comparison of different silvicultural treatments in relatively simple terms (Meyer, 1952; Sammi, 1961; Cancino and Von Gadow, 2002). Though more accurate, descriptions of alternative models are often mathematically complex hence do not allow communication between professionals (Ducey, unpublished).

$$\text{Eq. (4). } y = aq^{n-h}$$

Where:

y , is the predicted density for a given class;

a , is the observed density in the highest class

q , is constant and is the ratio of number of stems between successive classes

n , is the number of classes.

b , varies for each successive class, with its value being one and n in the lowest and highest class respectively.

Canopy change detection

To ascertain past exploitation in addition to stump count, SPOT XS images for 1994 and 2000 (<http://www.planet-action.org/>) were used to assess the cover change. Image classification was done using the unsupervised maximum likelihood classification algorithm in ArcGIS 10.1. Number of classes was specified at ten for both images. The raster pixels were then converted into polygons and those under mangroves class were used to determine forest areal coverage in the two years.

RESULTS

Floristic composition

Five mangrove tree species were encountered in Mtwapa creek with *Rhizophora mucronata* as the principal species in all the sites (Table 1). *Sonneratia alba* and *Xylocarpus granatum* were only encountered in Gung'ombe and Kitumbo with *S. albarecording* low densities in both forest patches. *Rhizophora mucronata* and *Avicennia marina* had higher abundance than the other species in all the forest patches. *Rhizophora mucronata* dominance was majorly as a result of its large stem densities where as that of *A. marina* could have been contributed by large basal area (Table 1). Moreover *R. mucronata* had a high frequency of occurrence of more than 50%.

Vegetation inventory

Rhizophora mucronata accounted for at least 70% of the total stem density in each of the three forest patches. As a result, there was no clear zonation observed. Although *A. marina* showed characteristic large stems close to the water line, the landward *Avicennia* were mostly absent. The other species with major contribution to stem density was *Ceriops tagal*, which however contributed only less than 20% of total stem density in each of the sites (Table 2). In Kidongo stunted this species had trees which were frequently stunted.

All the species encountered (except for *A. marina* and *S. alba*) had characteristic high stem density that was dominated by diameter size classes ranging from 2.5 to 7.4cm, mainly suitable for use as fitos and pau. These low diameter classes, accounted for more than 60% of the total stem densities in all the forest patches. In addition, the stems were mostly of form 3and thus unsuitable for building (Figure 2). Conspicuous

absence of utilisation classes with diameter greater than 11.5cm (in Kidongo) or reduced abundance in Gung'ombe and Kitumbo was also noted. The mean (\pm standard deviation) percentage vegetation cover was 71.53 \pm 8, 56.77 \pm 17.9, 55.71 \pm 13.9 in Gung'ombe, Kitumbo and Kidongo respectively. The percentage vegetation cover per quadrat did not however show any significant correlation with the number of stumps ($Z = 1.06$, $p > 0.05$, Mann-Whitney U Test).

Relationship between height and stem diameter of the most dominant species (*R. mucronata*) in the surveyed patches is shown in Figure 3. All the patches showed similar trends with most trees tending towards an increased height with minimal radial difference. However, difference in heights among forest patches and species were highly significant ($H = 280.9$; $N = 2365$; $p < 0.01$ and $H = 448.6$; $N = 2365$; $p < 0.01$ respectively).

Similarly, differences in stem diameter among forest patches surveyed and between the species encountered were highly significant ($H = 124.4$; $N = 2367$ $p < 0.01$ and $H = 178.2$; $N = 2367$; $p < 0.01$, respectively). Most of the difference in stem diameters and heights among species could be attributed to the exceptionally large D130 recorded in *A. marina* (Table 1).

Regeneration status of the forest

R. mucronata saplings accounted for 60% of the total sapling counts in all the forest patches whereas *X. granatum* was observed in very low numbers while no *S. Alba* were encountered. Limited occurrence of *A. Marina* was also recorded. Such instances were dominated by saplings from RCI with non-representation of samplings from RCII. Both *R. mucronata* and *C. tagal* had equal distribution of saplings among the regeneration classes indicating possible recruitment success in saplings of the two species (Table 4). The sapling densities varied greatly between 42 and 27,673 stems ha^{-1} (Table 3) depending on species and forest patch. Density of already established saplings (RCII and RCIII) was more or less similar in the three patches; 3302 \pm 4681.6, 2280.3 \pm 3361.7 and 2882.3 \pm 3154.9 stems ha^{-1} (mean \pm SD) for Gung'ombe, Kitumbo and Kidongo respectively. Major differences were however observed across species ($p < 0.05$) in all the regeneration classes ($H = 61.08$; $H = 97.68$; $H = 77.65$) for RCI, RCII and RCIII respectively.

Cover change and stumps count

In 1994, the mangrove cover was 526 ha and this had reduced to 524.7 ha in 2000 translating to an annual loss of 0.04% (Figure 4). The red patches in the figure indicate that mangrove loss was more pronounced in Kidongo than the rest of the study site between the two selected years. In addition, the current survey also revealed astumps count higher in Kidongo as compared to the other forest patches (904 \pm 850,

Table 1. Structural attributes of Mtwapa creek mangrove forest: In all the forest patches, *R. mucronata* had the highest values for all attributes.

Forest patch	Species	Height (m)	BA (m ² /ha)	Density (Stems/ha)	Relative			IV ^a
					Density	Dominance	Frequency	
Gung'ombe	<i>A. marina</i>	7,29 ± 2,37	5.42	88	2.44	37.09	8.70	48.23
	<i>C. tagal</i>	2,20 ± 0,65	0.58	388	10.72	3.95	28.26	42.94
	<i>R. mucronata</i>	3,18 ± 1,02	8.43	3119	86.09	57.65	54.35	198.09
	<i>S. alba</i>	5,50 ± 1,00	0.14	12	0.32	0.95	2.17	3.44
	<i>X. granatum</i>	4,50 ± 1,50	0.05	15	0.42	0.36	6.52	7.31
Kitumbo	<i>A. marina</i>	7,21 ± 3,56	4.70	169	4.29	27.67	10.14	42.11
	<i>C. tagal</i>	2,65 ± 1,50	1.43	610	15.48	8.45	24.64	48.57
	<i>R. mucronata</i>	3,80 ± 1,20	8.14	2915	73.98	47.99	50.72	172.69
	<i>S. alba</i>	5,90 ± 0,95	0.43	51	1.30	2.51	1.45	5.26
	<i>X. granatum</i>	4,30 ± 1,36	2.27	195	4.94	13.38	13.04	31.37
Kidongo	<i>A. marina</i>	4,50 ± 3,20	2.54	54	1.86	37.55	14.29	53.69
	<i>C. tagal</i>	2,27 ± 0,84	0.40	438	15.12	5.93	23.81	44.86
	<i>R. mucronata</i>	2,70 ± 0,71	3.82	2408	83.02	56.52	61.90	201.45

^a IV is the ecological importance value of a species. The species with the highest value (bolded) is the principal species.

Table 2. Mtwapa creek mangrove forest stand table. Most poles were of low utilisation class except for *A. marina*.

Forest patch	Species	Utilisation class (butt diameter in cm) ^b						Total density (stems ha ⁻¹)	
		2.5-3.9 (fito)	4.0-7.4 (pau)	7.5-11.4 (boriti)	11.5-13.5 (mazio)	13.6-20.4 (nguzo)	20.5-35.0 (vingingi)		>35.0 (mbao)
Gung'ombe	<i>A. marina</i>	8 (8.7)	4 (4.3)	19 (21.7)	—	8 (8.7)	23 (26.1)	27 (30.4)	88 (2.4)
	<i>C. tagal</i>	215 (55.4)	158 (40.6)	15 (3.9)	—	—	—	—	388 (10.7)
	<i>R. mucronata</i>	1265 (40.5)	1473 (47.2)	238 (7.6)	61 (2)	69 (2.2)	12	—	3119 (86.1)
	<i>S. alba</i>	—	—	4 (33.3)	4 (33.3)	4	—	—	12 (0.3)
	<i>X. granatum</i>	4 (25)	8 (50)	4 (25)	—	—	—	—	15 (0.4)
Total stems ha ⁻¹ by site								3623	
Kitumbo	<i>A. marina</i>	26 (15.2)	44 (25.8)	18 (10.6)	—	36 (21.2)	36 (21.2)	10 (6.1)	169 (4.2)
	<i>C. tagal</i>	200 (33.6)	318 (53.4)	62 (10.3)	10 (1.7)	5 (0.9)	—	—	610 (15.1)
	<i>R. mucronata</i>	1015 (34.8)	1403 (48.1)	364 (12.4)	87 (2.9)	44 (1.5)	3 (0.1)	—	2915 (73.9)
	<i>S. alba</i>	—	13 (25)	31 (60)	3 (5)	5 (10)	—	—	51 (1.3)
	<i>X. granatum</i>	21 (10.5)	85 (43.4)	46 (23.7)	18 (9.2)	13 (6.6)	5 (2.6)	8 (3.9)	195 (4.9)
Total stems ha ⁻¹ by site								5831	
Kidongo	<i>A. marina</i>	9 (16.7)	7 (13)	—	—	—	20 (37)	18 (33.3)	54 (1.9)
	<i>C. tagal</i>	285 (65.1)	112 (25.5)	41 (9.4)	—	—	—	—	438 (15.1)
	<i>R. mucronata</i>	1392 (58)	938 (39)	69 (2.9)	9 (0.4)	—	—	—	2408 (83)
Total stems ha ⁻¹ by site								2900	

^b utilisation classes grouped according to the Kenyan market and apply Swahili names referring to various uses upon which the poles are put into. Values in parenthesis indicate percentage of total stem densities.

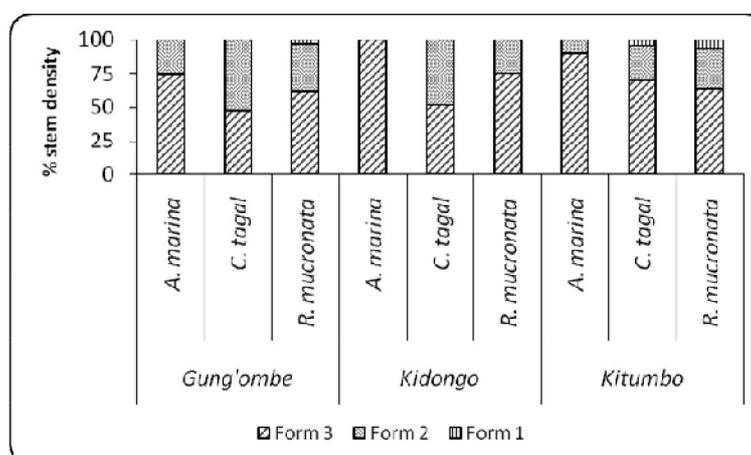
**Figure 2.** Form of lead stems for the three major.

Table 3. Density of saplings encountered within Mtwapa creek mangroves (saplings ha⁻¹).

Forest patch	Species	RCI ^c	RCII ^c	RCIII ^c	Total ha ⁻¹ by species
		<40 cm	40-150 cm	>150 cm	
Gung'ombe	<i>A. marina</i>	672 (99.18)	0	6 (0.82)	678 (2.21)
	<i>C. tagal</i>	905 (38.26)	722 (30.52)	739 (31.22)	2366 (7.70)
	<i>R. mucronata</i>	9328 (33.71)	10856 (39.23)	7489 (27.06)	27673 (90.09)
Totals ha ⁻¹ per RC		10906 (35.50)	11578 (37.69)	8233 (26.80)	
Kitumbo	<i>A. marina</i>	4255 (99.85)	0	6 (0.15)	4261 (12.83)
	<i>C. tagal</i>	2974 (52.69)	1326 (23.49)	1345 (23.83)	5645 (16.99)
	<i>R. mucronata</i>	7687 (33.49)	8171 (35.6)	7094 (30.91)	22952 (69.12)
	<i>X. granatum</i>	50 (14.29)	150 (42.86)	150 (42.86)	350 (1.05)
Totals ha ⁻¹ per RC		14966 (45.07)	9647 (29.05)	8595 (25.88)	
Kidongo	<i>A. marina</i>	42 (100)	0	0	42 (0.2)
	<i>C. tagal</i>	57 (5.26)	229 (21.05)	800 (73.68)	1086 (7.47)
	<i>R. mucronata</i>	2914 (21.73)	3343 (24.92)	7157 (53.35)	13414 (92.24)
Totals ha ⁻¹ per RC		3013 (20.72)	3571 (24.56)	7957 (54.72)	

^csaplings have D₁₃₀ less than 2.5cm and the classes, RCI, RCII and RCIII are grouped according to height. Values in parentheses are percentages of the total stem density per class and species.

Table 4. Sampling ratios of the three major species in Mtwapa creek mangroves.

Site	Tree species	RCI ^c	RCII ^c	RCIII ^c
		<40 cm	40-150 cm	>150 cm
Gung'ombe	<i>A. marina</i>	112	0	1
	<i>C. tagal</i>	1	1	1
	<i>R. mucronata</i>	1	1	1
Kitumbo	<i>A. marina</i>	709	0	1
	<i>C. tagal</i>	2	1	1
	<i>R. mucronata</i>	1	1	1
Kidongo	<i>A. marina</i>	42	0	0
	<i>C. tagal</i>	1	4	14
	<i>R. mucronata</i>	1	1	2

^c Samplings have D₁₃₀ less than 2.5cm and the classes, RCI, RCII and RCIII are grouped according to height

Table 5. Dispersion indices for the two most abundant adult trees species in the mangrove forest of Mtwapa creek.

Site	species	Distance from the creek (m)					
		50	100	200	250	300	350
Gung'ombe	<i>C. tagal</i>	— (—)	3.0 (—)	1.4 (5.2)	— (—)	— (—)	1.6 (10.3)
	<i>R. mucronata</i>	1.4 (0.9)	1.0 (2.7)	0.9 (0.1)	1.3 (0.1)	1.5 (0.8)	2.57 (2.73)
Kitumbo	<i>C. tagal</i>	— (—)	— (—)	0.2 (0.1)	— (—)	0.4 (2.5)	— (—)
	<i>R. mucronata</i>	0.3 (3.3)	1.6 (0.9)	1.2 (1.0)	1.6 (0.4)	1.8 (0.8)	1.0 (4.3)
Kidongo	<i>C. tagal</i>	— (—)	1.7 (—)	— (—)	3.0 (7.2)	2.0 (4.2)	1.2 (3.3)
	<i>R. mucronata</i>	1.0 (5.4)	1.1 (0.7)	1.1 (0.7)	1.3 (0.1)	0.1 (0.3)	1.5 (6.9)

The empty entries indicated by hyphens are sampling points where the specific species were absent. Values in parentheses represent saplings.

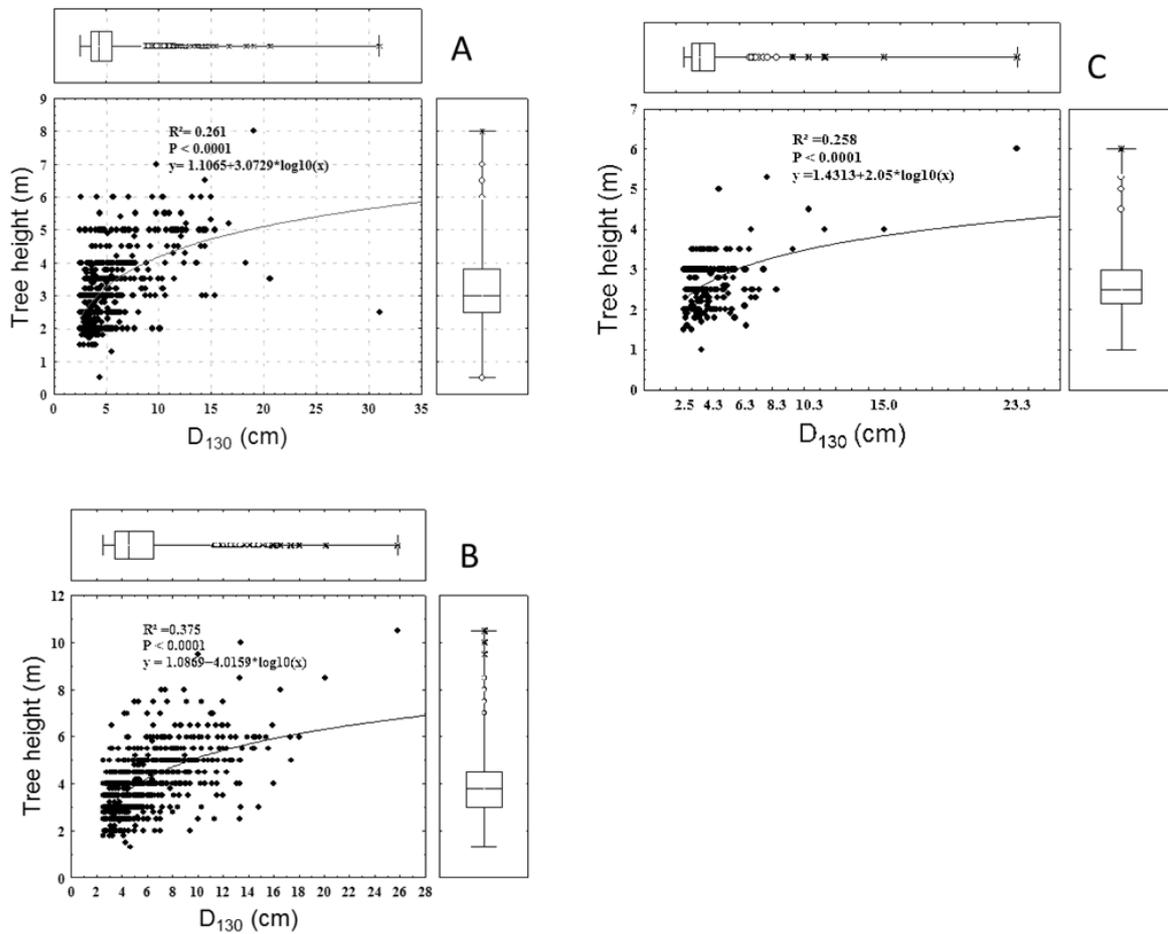


Figure 3. Scattergrams of heights against stem diameters of *R. mucronata* in: A) Gung'ombe, B) Kitumbo and C) Kidongo forest patches in Mtwapa creek. The polynomial equations, correlation coefficients (R^2), and P values are given for all sampling stations. The box plots display percentile distribution with the extremities corresponding to maximum and minimum values in the data set and the ends of box positioned at 25th and 75th percentiles. All the three forest patches have similar trends with most trees tending towards increased height and limited radial growth.

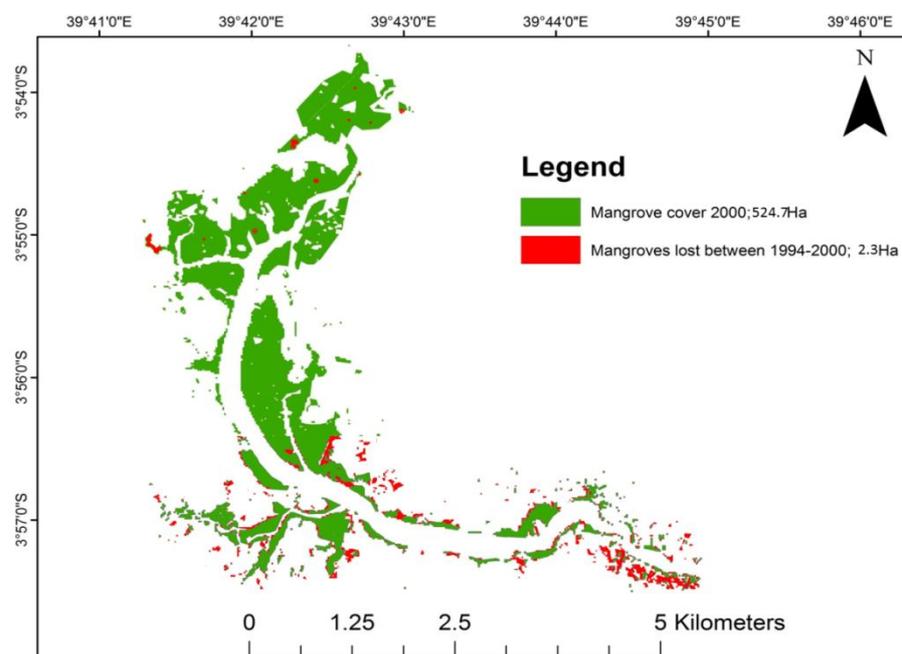


Figure 4. Change in mangrove cover in Mtwapa creek between the years 1994 and 2000

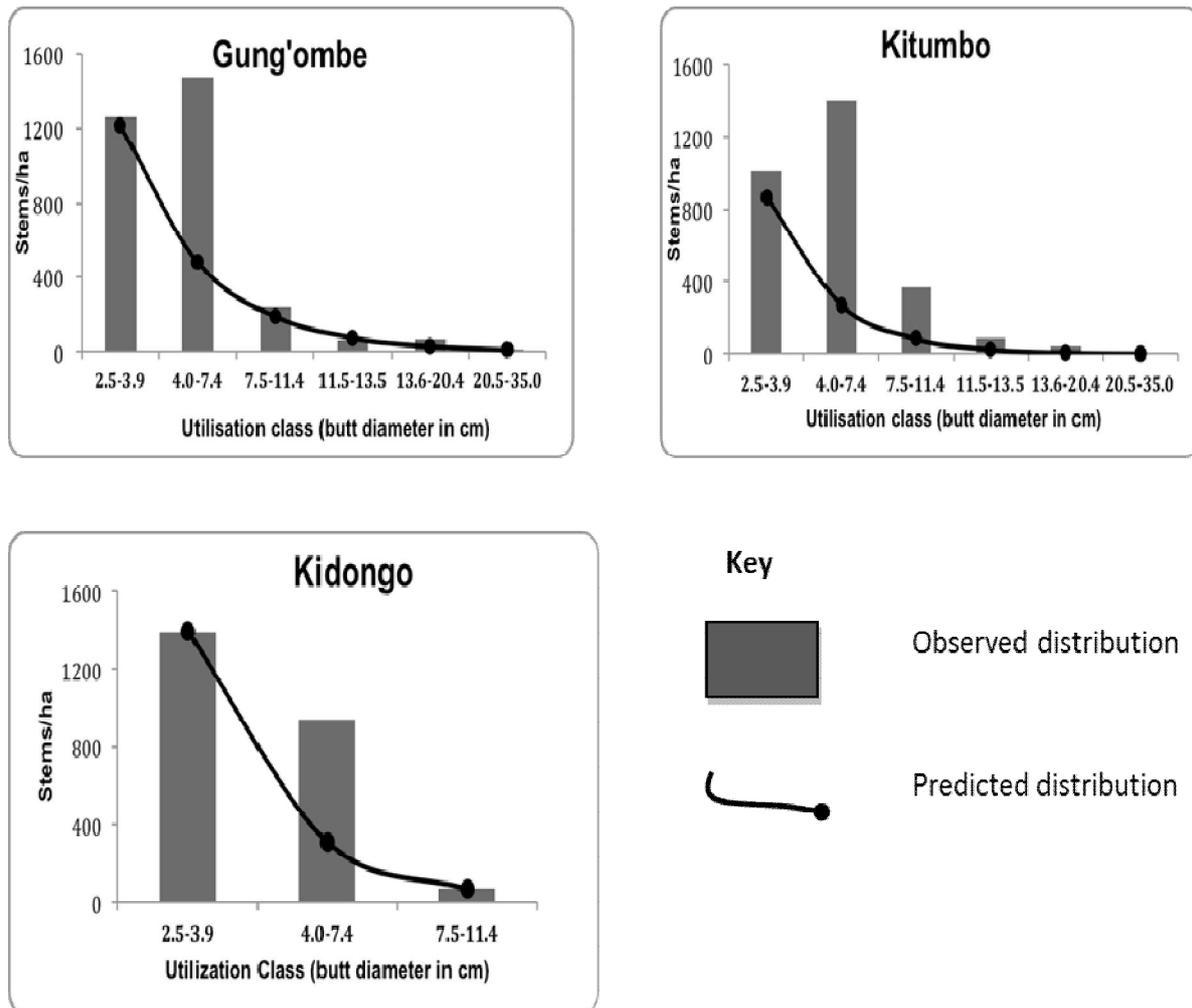


Figure 5. Observed and predicted size class distribution of principal species (*R. mucronata*) in Mtwapa Creek, Kenya. Predictions are based on De Liocourt's model (Clutter *et al.*, 1983) used to harmonize stand densities from 26, 39 and 13, 10m by 10m quadrats sampled in Gung'ombe, Kitumbo and Kidongo respectively. Preferential harvesting of poles is only seen in Kidongo where certain size classes are missing.

750±700.2, and 2557.1±1000.9, Mean stump density per hectare ±SD in Gung'ombe, Kitumbo and Kidongo in that order). The relationship between gaps created by cutting (determined from stumps' count), and the number of saplings was however not significant ($p>0.05$) for all the regeneration classes.

Dispersion and population shape of the principal species

Variation in complexity index among the surveyed sites revealed higher values in Gung'ombe and Kitumbo (12.01 and 15.95 respectively) and a low value of 1.86 at Kidongo. Dispersion according to Morisita index showed a tendency towards uniform distribution for *R. Mucronata* adults as well as saplings ($I^0 < 1$) but a rather clustered pattern ($I^0 \gg 1$) for *C. tagal* (Table 5). The principal species showed a deviation from the De Liocourt's balanced diameter distribution with certain utilisation classes completely missing out in Kidongo (Figure 5). This absence of classes is an important

proxy for determination of selective exploitation of the mangroves by the people.

DISCUSSION

Upper canopy structure

The observed low species representation in Mtwapa 2002), Tudorcreek (Mohamed *et al.*, 2009) and in Gazi bay (Dahdouh-Guebas *et al.*, 2004). *Rhizophora spp.* can take such a position elsewhere, e.g. *R. mangle* in Sittee river - Belize (Kangas, 2002) and *R. mucronata* in Sri Lanka (Mangala and Padma, 1998; Karunathilake, 2003).

Furthermore, *Rhizophora spp.* are known to be the most conspicuous and dominant of the mangrove species in many assemblages worldwide, due to its growth strategies (Duke, 2001). Higher stem densities in Mtwapa creek (2408-3119 stems ha⁻¹) in comparison to levels reported by similar studies in Mida creek of 433-545 stems ha⁻¹ (Kairo *et al.*, 2002) and in creek

could be attributed to geomorphology as proposed by Snedaker et al. (1992) and Bunt and Stieglitz (1999). The steep hills facing the creek limit the landward spread of other mangroves species. As a result, the landward *A. marina* zone was frequently absent with the species only occurring as large trees in the seaward zone. Though it is commonly a landward species, *A. marina* also exhibits double zonation in some mangrove forests as has also been reported in Gazi bay (Dahdouh-Guebas et al., 2004).

As a result of the absence of landward *A. marina* the entire tidal flat was left for *R. mucronata* dominate the entire tidal flat with more than 70% coverage.

The results show *R. mucronata* as the principal species in the study which is a common phenomenon with similar results reported in Mida creek (Kairo et al., 2009) ought to be interpreted with care as the equivalent basal area in this study is relatively low. In Sittee River, Kangas (2002) recorded a basal area of 20.8 m² ha⁻¹ against a density of 775 stems ha⁻¹ for *Rhizophora mangle* (the principal species in that forest).

These stem densities were much lower than those recorded for *R. mucronata* in the current study (between 2408 and 3119). Additionally, even though stem densities of *C. tagalin* in this study were more or less comparable to those recorded by similar studies along the Kenyan coast the basal areas were quite low.

Natural regeneration as the potential future mature stand

Natural regeneration observed in the study was highly linked to the standing mature trees composition since the saplings with successful recruitment were majorly from the highly dominant species (Table 3). Clustering of saplings of the less abundant species and a uniform dispersion of *R. mucronata* observed is similar to findings by McKee (1995), who noted great variation in densities of new recruits (RCI) of mangroves and attributed it to distance from reproductive adults. Similarly, the saplings in the current study followed parental tree distribution.

The observed variability in performance of saplings after the establishment phase could be attributed to factors which affect their survival such as species' differences in sensitivity to mechanical and physicochemical stress arising from; for example, siltation (Mohamed et al., 2009; Terrados et al., 1997), flooding and salinity (Jayatissa et al., 2008), as well as other factors like initial orientation of seedling axis (McKee, 1995).

Siltation in Tudor creek was facilitated by ease of sediment to be carried down the slopes which face the tidal flats (Mohamed et al., 2009). Mtwapa creek has a similar setting and is deemed to face the same problem that might have led to high mortalities of *A. marina* hence low recruitment into successive regeneration classes. High recruitment observed in Kidongo for *C. tagal*, was however attributed to the artificial grouping

criteria of seedlings into regeneration classes, which may have led to inclusion of stunted mature trees in RCIII as saplings.

Implications of structural status of Mtwapa creek mangrove forest on its sustainability and management

Spatial analysis of mangrove coverage in the area reflect an annual loss in coverage of 0.04%, between 1994 and 2000 (Figure 4) which is rather small compared to losses experienced during the same period in other mangrove areas in Kenya (Kirui et al., 2012). Nevertheless, the absence of the highly preferred pole sizes, boriti and pau particularly in Kidongo is indicative of higher consumptive extraction of mangrove wood products in close proximity to human settlements (Kairo et al., 2002) as well as ease of access to the forest area.

Such selective logging could greatly affect pole quality and the stability of future forest (Kairo, 2001), resulting into a forest with poor form trees composed mainly of stunted and shrubby trees and incomplete representation of utilisation classes (Eusebio et al., 1986; Smith and Berkes, 1993). In addition it may lead to fragmentation and alteration of habitats thus affecting the general stability of an ecosystem (Johns, 1989).

Even though it has been reported that selective removal of poles by the cutters (creating small gaps in the forest canopy), stimulates regeneration (Duke, 2001; Kairo et al., 2002) with extremely large gaps inhibiting regeneration (Mohamed et al., 2009), such a relationship was not evident in this study. Lack of any significant link between gaps created by harvesting and regeneration implies that regeneration is sufficient enough to fill the gaps created by exploiting the resource.

In cases where natural regeneration rates are insufficient to fill the gaps created by logging, reduced abundance of mature trees is expected. However, high stem density with extremely low basal area was observed for *C. tagal* and *R. mucronata* (the most common species in the study site).

This is indicative that Mtwapa creek mangroves is a stressed system struggling to sustain itself by producing more saplings (Table 3), that eventually experience reduced growth. It could additionally be attributed to history of repeated selective logging which has been demonstrated elsewhere to result in high stem density and associated low basal areas in the affected forests (Bradley, 2005; Smith and Berkes, 1993).

In spite of the good regeneration and recruitment in Mtwapa creek, the stems develop into a stand of high densities with poor form and low utilisation class. The system could hence be considered unsustainable. In addition, sustainability as described by the De Liocourt's model requires a balanced diameter distribution (Clutter et al., 1983).

The deviation of the Mtwapa creek mangrove stand from the balanced diameter distribution (Figure 5) is

indicative of reduced sustenance capacity of the forest. The current management regime which involves licensing cutters, concentrates solely on quantity harvested by placing and lifting bans randomly without an established mechanism to ensure sufficient turn over (Bosire and Kairo, 2008; Government of Kenya 2009). As a result, the management has failed to take note of cryptic degradation and ensure sustainability.

Even though there are evidences of selective logging in this study and its associated impacts, lack of a clear link between forest poor status and human exploitation suggest that other forms of disturbance could be playing a role as well. Moreover, the local communities through youth and women groups participate in conservation of the forest through replanting and engaging in alternative livelihoods to tame overexploitation, (Okello et al., 2012).

Other human activities in the riparian zone and catchment areas could also impact on self-sustenance potential of the forest. For instance, excessive accretion of mangroves With terrigenous sediment from the hilly land scape should be monitored to determine their impacts on tree growth. Low levels of sediment accumulation (sedimentation) are associated with elevated nutrients and increased growth rate in mangrove forests (Lovelock et al., 2007).

However, in large quantities (greater than 1cm a year) it may lead to stress, dieback and even death (Ellison, 1998). This has been noted to impact negatively on seedling development in adjacent mangrove forest of Tudor creek (Mohamed et al., 2009).

In Mwache creek situated South of Mtwapa creek, poor land use practices in the hinterland have increased sediment loads into the mangrove forest leading to siltation of breathing roots of the trees and eventual death of the trees (Kitheka et al., 2002).

Conclusion

The results of this study showed that Mtwapa creek mangrove forest has sufficient natural regeneration but poor tree development thus hindering the forest's self-sustenance.

Apart from geomorphology which influences the extent of landward spread of mangroves, selective logging particularly in Kidongo has been noted to impact negatively on the integrity of the forest.

Additionally, this study points out that other stress causing forms of disturbances could as well be limiting tree development in Mtwapa creek. Since most mangrove studies have focused on local patterns other than an integrated approach, little or no information is available on distal areas, hence are not reflected in the management units or plans.

It would therefore be prudent to conduct further research in both mangroves and the riparian zones including the catchment areas and look at land use and land use changes. It is envisaged that such approaches will lead to an informed decision in sustainably managing mangrove resources in Kenya.

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