3: Mangrove forests in a peri-urban setting: the case of Mombasa (Kenya)

Tudor creek mangroves from the rural Tsalu side **-** a *patchy* distribution of mangroves with wide open spaces

Publication

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Summary

The structure and regeneration patterns of the peri-urban mangrove vegetation of Mombasa at Tudor creek were studied along belt transects at 2 forest sites of Kombeni and Tsalu. Based on the species importance values, the dominant mangrove species were *R. mucronata* and *A. marina*. *L. racemosa*, reported in an earlier floristic survey, was not encountered. Tree density varied from 1,264 trees ha^{-1} at Kombeni to 1,301 trees ha^{-1} at Tsalu and mean tree height was higher at the former site compared to the latter. The size-class structure at both localities showed the numerical dominance of small trees over larger trees. The spatial distribution pattern of adults and juveniles varied greatly between sites and showed a close to uniform pattern (Morisita's Index $I_\delta << 1$) for adult trees, but a tendency to clustered distribution ($I_\delta >> 1$) for juveniles. The present paper shows that unmanaged but exploited peri-urban mangroves are structurally degraded, with numerous large canopy gaps that are characterised by spatial and temporal site heterogeneity. These gaps influence regeneration, implying longer periods for canopy closure. Diversifying uses of mangrove products and establishing reserves as no-cut zones with regulated harvesting will minimise canopy gap sizes, and promote conservation practices. The proposed management strategy shall boost the ecosystem resilience to both anthropogenic and natural disturbances expected in the peri-urban setting in the long run.

Keywords: anthropogenic, canopy gaps, peri-urban, regeneration

3.1 Introduction

Mangrove swamps are typical wetland ecosystems found in coastal deposits of mud and silt throughout the tropical and subtropical coasts. These ecosystems are currently estimated to cover 15.2 million hectares of the tropical shorelines of the world (FAO, 2003; 2005), representing a decline from 18.8 million hectares of mangroves in 1980 (Alongi, 2002; FAO, 2005; Duke *et al.*, 2007). As intertidal ecosystems, mangrove forests provide essential functions and services to coastal zones and to their plant, animal and human populations (Cannicci *et al.*, 2008; Kristensen *et al.*, 2008; Nagelkerken *et al.*, 2008; Walters *et al.*, 2008). Mangrove degradation over time has been recurrently documented (Dahdouh-Guebas and Koedam, 2008; Ellison, 2008a, b), but direct and indirect anthropogenic pressure persists. Diversion of freshwater flows, deteriorating water quality caused by pollutants and nutrients, over-harvesting for fuel-wood and timber as well as conversion into other uses like agriculture, aquaculture, mining, salt extraction and infrastructure all contribute to the degradation and deforestation of mangrove ecosystems (Saenger *et al.*, 1983; Terchunian *et al.*, 1986; Primavera, 1995; Twilley *et al.*, 1995; Mandura, 1997; Ellison, 1998; Dahdouh-Guebas *et al.*, 2000; Valiela *et al.*, 2001; Abuodha and Kairo, 2001; Kairo *et al.*, 2001; Taylor *et al.*, 2003; Benfield *et al.*, 2005; Dahdouh-Guebas *et al.*, 2004; Dahdouh-Guebas *et al.*, 2005b; Duke *et al.*, 2007). Climate change poses an additional threat (Gilman *et al.*, 2008).

In Kenya, mangroves are well developed in many areas along the coastline, particularly in creeks, bays and estuaries. It is estimated that mangroves cover 52,980 ha in Kenya (Doute *et al.*, 1981), though the KFS (Forest Department of Kenya, 1983) estimates 64,426 ha. The bulk of these forests occur in Lamu (33,500 ha), with smaller forests in Kwale (8,800 ha), Kilifi (6,600 ha) and Mombasa (2000 ha) (Doute *et al.*, 1981). Traditionally, mangroves in Kenya have been used as a source of building poles and firewood (Kokwaro, 1985; Dahdouh-Guebas *et al.*, 2000). During the colonial period (before 1963), mangrove wood products formed major export commodities, and by 1950's, sale of mangrove products were ranked third in the national revenue generation (Rawlins, 1957). Overexploitation led to a presidential ban on mangrove exports in 1982. However, harvesting for the domestic market intensified in many parts of Kenya, leading to further overexploitation and subsequent degradation (Kairo, 2001; Abuodha and Kairo, 2001; Dahdouh-Guebas *et al.*, 2000; Taylor *et al.*, 2003; Dahdouh-Guebas *et al.*, 2006; 2006a). It is estimated that 20% of the total mangrove forests in Kenya have been lost already (Abuodha and Kairo, 2001). This raises concerns about the sustainability of mangrove resource utilisation especially because human migration into coastal zones is high. Coastal zones are currently estimated to provide living space for 60% of the world's population (World Resources Institute, 1996; Sheppard, 2001), and the dependence on mangroves by coastal fisher folk is large (Walters *et al.*, 2008).

The major problem facing the management of mangrove forests in Kenya is the lack of a management plan (Dahdouh-Guebas *et al.*, 2000; Kairo *et al.*, 2002a), with annual quotas for extraction decided on unspecified basis, and compounded with inadequately supervised extraction operations (Kairo, 2001; Ferguson, 1993). Peri-urban mangroves in particular, are under disturbance due to over-harvesting for domestic firewood and industrial energy, human encroachment for housing and pollution (Gang and Agatsiva, 1990; Munga *et al.*, 1993; Rees *et al.*, 1996; Mwaguni and Munga, 1997; Taylor *et al.*, 2003).

Mombasa, a peri-urban island area, is bounded by two main creeks namely Tudor and Port Reitz, has a population of 917,864, an average population density of 3,990 persons per km², and an annual growth rate of 2.5% (Kenya Bureau of Statistics, Coast Office – pers. comm. Mohamed). Kilindini Harbour, East Africa's principal modern deep-water seaport, is located at the entrance of the Port Reitz creek. In the period 1983-93, the port of Mombasa and its adjacent waters experienced 5 tanker accidents spilling a total of 391,680 tonnes of oil (Abuodha and Kairo, 2001; Taylor *et al.*, 2003). A major spill in 1988 destroyed 10 ha of mangroves in Makupa (Abuodha and Kairo, 2001; FAO, 2005), and in 2005, 200 tons of crude oil were spilled, affecting 234 ha of mangroves in Port Reitz creek (Kairo *et al.*, 2005). In addition, the Mombasa municipal waste contributes about 4,369 ton/year of BOD, 3,964 ton/year of suspended solids, 622 ton/year of nitrates and 94 ton/year of phosphates into the creeks in the form of raw sewage (Mwaguni and Munga, 1997; Mwangi *et al.*, 1999). This is in addition to coliform and *Escherichia coli* levels of 1800+ per 100 ml and up to 550 cfu per 100 ml respectively (Mwaguni and Munga, 1997; Mwangi *et al.*, 1999).

Little information exists on the status of peri-urban mangroves globally, and few studies have been conducted on the structure and regeneration status of peri-urban mangroves. Previous mangrove forest structural studies in Kenya were conducted on mangroves of Kiunga, (Kairo *et al*., 2002a), Tana River (Bundotich, 2007), Mida (Kairo *et al*., 2002a); and Gazi Bay (Kairo, 2001). Despite the importance of the peri-urban mangroves of Mombasa, the forest structure and regeneration status is unknown. Knowledge on vegetation structure and regeneration potential of a specific forest is a prerequisite to designing forestry directives like annual allowable cuts and designating specific harvesting areas (FAO, 1994). Ultimately, structural assessment will contribute to the development of sustainable forest management plans (Holdridge *et al*., 1971; FAO, 1994). This study was designed to assess the structural conditions of mangrove forests in Tudor creek and to compare results with published data on other Kenyan mangrove forests in a different setting.

3.2 Materials and methods

3.2.1 Study area

Tudor creek (Figure 3.1) bounds Mombasa Island on the northwest and extends some 10 km inland. The creek has two main seasonal rivers, Kombeni and Tsalu, draining an area of 55,000 ha (45,000 ha and 10,000 ha respectively) with average freshwater discharge estimated at 0.9 m^3s^{-1} during the inter-monsoon long rains (cited in Nguli, 2006). It has a single narrow sinuous inlet with a mean depth of 20m, that broadens out further inland to a central relatively shallow basin (5m) fringed by a well developed mangrove forest mainly composed of *R. mucronata* (Rhizophoraceae), *A. marina* (Acanthaceae) and *S. alba* Sm. (Sonneratiaceae). The basin has an area of 637 ha at low water spring and 2,235 ha at high water spring. Mangrove forests occupy 1,465 ha of the creek. The floristic composition of mangroves of Tudor creek has been described by SPEK (1992). There is no obvious zonation displayed by the dominant mangrove species in Tudor creek. *A. marina* and *L. racemosa* occupy the landward zone, whereas mostly a *C. tagal* and *R. mucronata* mosaic covers the middle zone. Wherever present, *S. alba* occupies the seaward margin, but is replaced by tall *A. marina* and *R. mucronata* along small creeks.

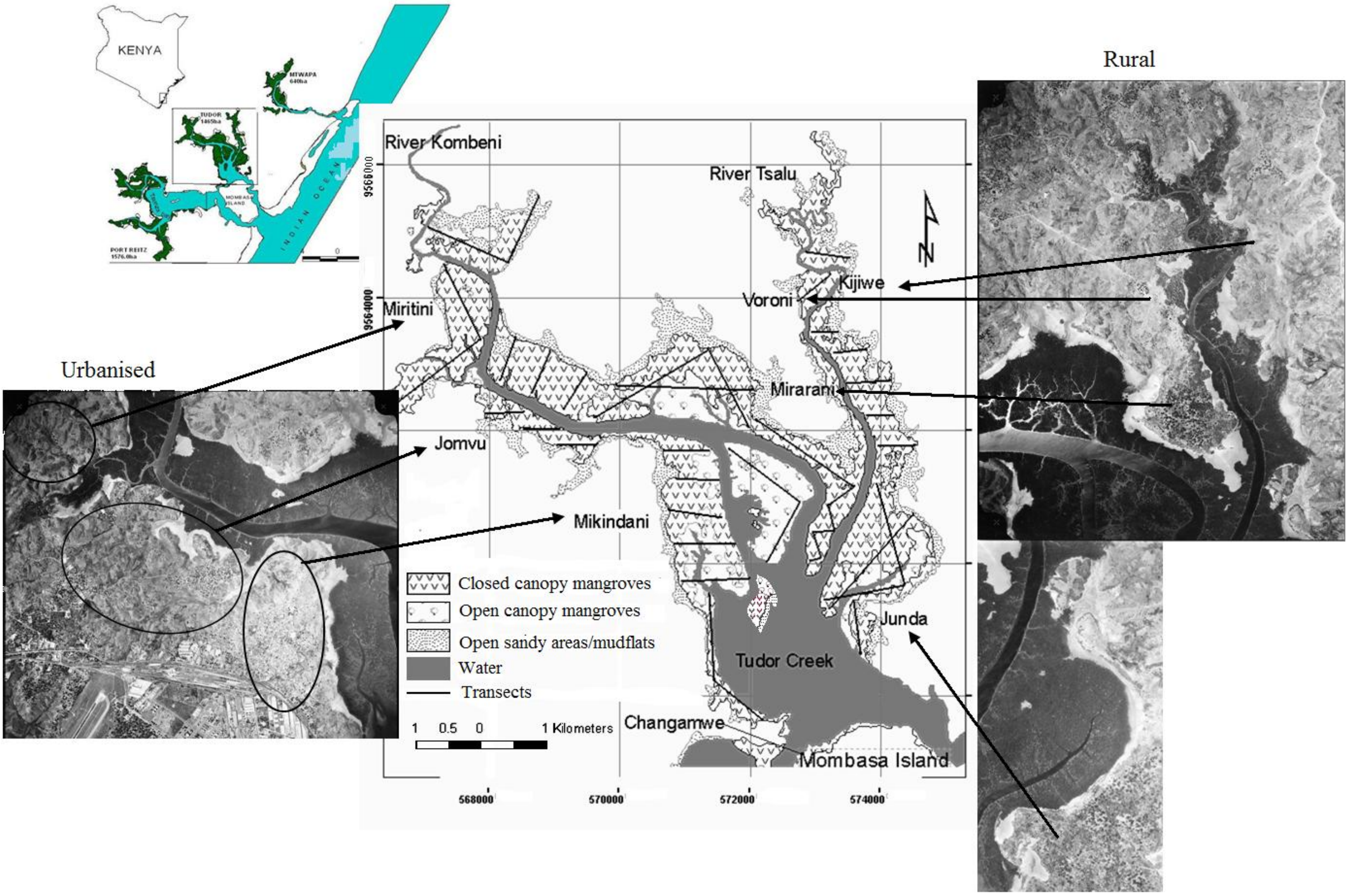

Figure 3.1: Map of Mombasa showing the study area (Tudor creek) and the sampled transects (Source, DRSRS aerial photographs 1992).

The forest resembles the fringing mangroves described by Lugo and Snedaker (1974), with strong inward tidal current during the flood tides which reverses during ebb tides, attaining maximum tide velocities of 0.6-0.7 ms^{-1} (Nguli, 2006), and the dense, well-developed prop roots that accumulate large stocks of debris, with a spring tidal range of 3.5 m and a neap tidal range of 1.1 – 1.3 m.

The mangroves of Tudor creek are separated naturally by two main tidal creeks, Kombeni and Tsalu, 4.5 and 3 km long respectively cutting through the mangroves connecting to the upstream rivers (Figure 3.1). In the framework of this study these two tidal creeks were sampled separately and compared. Tsalu, the eastern tidal creek, includes mangroves near the rural villages of Mirarani, Voroni, Junda and Kijiwe.

Kombeni, the western tidal creek, stretches between the townships of Mikindani, Jomvu and Miritini. The creek is bordered by a steep cliff overlooking a tidal flat that extends to the mudflats occupied by the vast mangrove forest (Figure 3.2).

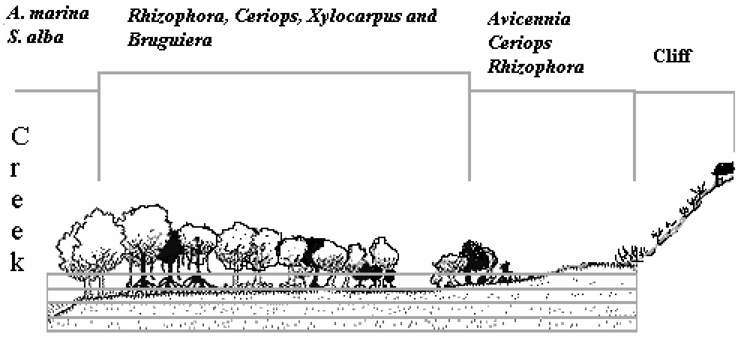

Figure 3.2: Figure illustrating the location of mangroves with respect to the cliff, where settlements and farms are being established

3.2.2 Forest structure and regeneration

A stratified sampling technique was used to sample mangroves of Tudor creek. The location of transect lines were determined by an initial reconnaissance and examination of medium-scale (1:25000) panchromatic aerial photographs of the area. Belt transects of 10 m width were established both perpendicular and parallel to the creek across the entire forest in such a way that they represented as well as possible the general mangrove formation of Tudor creek (Figure 3.1). Vegetation sampling was carried out within 100 m^2 quadrats, established along the transects. A total of 106 and 124 plots were sampled in Kombeni and Tsalu respectively.

Within each quadrat, individual trees with diameter greater than 2.5 cm were identified, counted and position marked. Vegetation measurements included tree height and stem diameter at 130 cm aboveground (D_{130} *sensu* Brokaw and Thompson, 2000, exceptions to this rule are described below). These measurements were used to derive tree basal area, stand density and frequency (Mueller-Dombois, and Ellenberg, 1974; Cintrón and Schaeffer-Novelli, 1984). The ecological importance of each species was calculated by summing its relative density, relative frequency and relative dominance (Cintrón and Schaeffer-Novelli, 1984). The complexity indices (I_c) of the forests were obtained as the product of number of species (s), basal area (m^2/ha) (b), maximum tree height (in meters) (h) and number of stems ha^{-1} (d) x 10^{-5} (Holdridge *et al.*, 1971).

Stems with diameters below 2.5 cm were considered in the category “juveniles” (Kairo *et al*., 2002a, b). Stumps were also counted in each quadrat as an indicator of anthropogenic pressure and tree mortalities.

Tree heights were measured in meters using a Suunto™ hypsometer, whereas stem diameter was measured in centimetres using a forest calliper. For *Rhizophora*, stem diameters were measured 30 cm above the highest prop roots, whereas for *Avicennia*, when the stem forked below 130 cm, individual ‘branches’ in a clump were treated as separate stems. Pole 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 therefore excellent for construction but form 2 stems need slight modification to be used for construction. Poles which are unsuitable for construction were assigned form 3 (Kairo, 2001).

Information on the composition and pattern of natural regeneration was obtained using the method of Linear Regeneration Sampling (Sukardjo, 1987). In 5 x 5 m^2 subplots (within the main 10 x 10 m^2 quadrats), occurrence of juveniles of different species was recorded and grouped according to height classes. Seedlings less than 40 cm in height were classified as regeneration class I (RCI). Saplings between 40 and 150 cm height were classified as RCII, while RCIII was for all small trees with heights greater than 1.5 m but less than 3 m. The analysis of spatial pattern of adults and juveniles in the field was carried out inside 10 x 10 m^2 plots along transects. The measure of dispersion used was Morisita’s dispersion Index (Morisita, 1959), the application of which is described in Greig-Smith (1983). Morisita’s Index (I_δ) is:

$$I_\delta = q\sum_{i=1}^{q} \frac{n(n-1)}{N(N-1)}$$

Where, q is the number of quadrats, n_i is the number of individuals per species in the i^{th} plot, and N is the total number of individuals in all q quadrats. If $I_\delta > 1$, the population is clustered, if $I_\delta = 1$, the population is randomly dispersed and if $I_\delta < 1$, the population is evenly dispersed. Socio-ecological information was obtained primarily from informal interviews conducted with mangrove harvesters, fishermen and Government Forestry Officers.

3.2.3 Data treatment

All data analysis and graphical presentation were obtained with the STATISTICA 8.0 program. One-way ANOVA was performed on stocking densities of different size classes, which we assume as a measure of age. The relative density, dominance and frequency were estimated and the importance values established according to Dahdouh-Guebas and Koedam (2006a).

Stand densities were harmonized using De Liocourt's negative exponential model (Clutter *et al*., 1983). According to the model, the ratio between the numbers of trees in successive diameter classes of uneven-aged stand is roughly constant for a particular forest, but varies from one forest to another. This has been confirmed in a number of uneven-aged forests throughout the world (see e.g. Clutter *et al*., 1983). De Liocourt's model applies particularly in mixed forests where the size classes and recruitment by natural regeneration are continuous. Supposing this constant is q, then the number of trees in successive diameter classes will be represented by a descending geometric sequence of the form $aq^n, aq^{n-1}, aq^{n-2}, aq^{n-3}, \ldots\ldots aq^3, aq^2, aq^1, a$, where a is the number of trees in the largest size class of interest and n is the number of classes.

For such a geometric series, if the logarithm of the frequency in successive classes is plotted against size class, the distribution can be represented as an exponential curve of the form:

$$y = ke^{-ax}$$

where; y is the number of trees in diameter class x; e is the base of natural log (2.718) while k and a are constants. The constants k and a in the equation above vary between forests and with site. The constant k reflects the occurrence of seedling regeneration and tends to be large in forests containing prolific seed-bearing tree species while a determines the relative frequencies of successive diameter classes. A high a is associated with high mortality between classes and is likely to occur in stands comprising light demanding (shade intolerant) tree species (Kairo *et al*., 2002a).

The nature of the future forest was derived from the present forest by fitting exponential models to the size-class structures and comparing the results at a 0.05 significant level. Each class interval was considered to be independent and thus included as within-factor repeated measure variable during the analysis. Regeneration densities and ratios were calculated and the correlation between regeneration densities with canopy gaps analysed using Spearman correlation.

3.3 Results

3.3.1 Floristic composition

The structural characteristics of the mangroves of Tudor creek are given in Table 3.1. Six mangrove species were encountered during the survey. Based on the species' importance values, *R. mucronata* and *A. marina* were the principal species in both sites. *X. granatum* was encountered in Tsalu only, while *L. racemosa* was not encountered despite earlier reports of its occurrence in the landward fringe (SPEK, 1992). *S. alba* was observed to be infested by an unknown insects and/or pathogen. Relative dominance, density, frequency and importance values of these species are shown in Table 3.1. The variation in complexity index between Kombeni and Tsalu is evident, with Kombeni recording a higher index than Tsalu

Table 3.1: Structural attributes of the mangroves of Tudor creek, Kenya

	Species	**Height (m)**	**BA (m^2ha^{-1})**	**Density (ha^{-1})**	**Relative**			**I.V**	**Rank**	**CI**
					Den.	Dom.	Fr.			
Kombeni	*A. marina*	7.98 ± 2.94	7.07	143.40	11.34	43.43	24.53	79.30	2	4.38
	B. gymnorrhiza	4.09 ± 0.86	0.15	16.98	1.34	0.91	5.66	7.91	5	
	C. tagal	2.62 ± 1.05	0.16	54.72	4.33	1.00	10.38	15.70	4	
	R. mucronata	3.61 ± 2.10	5.91	904.72	71.57	36.31	62.26	170.14	1	
	S. alba	5.22 ± 2.41	2.99	144.34	11.42	18.36	12.26	42.04	3	
Tsalu	*A. marina*	6.23 ± 3.30	5.89	167.74	14.60	60.69	25.00	100.29	2	2.60
	B. gymnorrhiza	3.82 ± 1.72	0.17	29.84	2.60	1.71	8.06	12.37	4	
	C. tagal	2.30 ± 0.55	0.58	175.81	15.30	5.99	29.84	51.13	3	
	R. mucronata	3.00 ± 1.06	2.58	750.00	65.26	26.57	83.87	175.70	1	
	S. alba	5.09 ± 2.09	0.39	20.97	1.82	4.00	3.39	9.21	5	
	X. granatum	3.83 ± 0.68	0.10	4.84	0.42	1.05	1.61	3.08	6	

3.3.2 Stocking density

Table 3.2 gives vegetation inventories for Tudor creek mangroves. There were 1,264 mangrove stems ha^{-1} in Kombeni creek, out of which, 71% were *R. mucronata*, 11% *S. alba* and 11% *A. marina*. While there was 1,301 stems ha^{-1} of in Tsalu creek, of which 58% were *R. mucronata*, 14% *C. tagal* and 11% *X. granatum*. Figure 3.3 shows scattergrams of heights against stem diameters. There were significant differences in height (F$_{(1,\ 2,763)}$ = 86.765; p = 0.0001) and stem diameter (F$_{(1,\ 2,763)}$ = 36.727; p = 0.0001) between the tidal creeks. 53% and 58% of the trees in both Kombeni and Tsalu respectively, were in the lower diameter size class (below 6.0 cm). *A. marina* was characterised by low densities of tree in the lower size class (2% ≈ 5%; Table 3.2), and high densities of trees in the medium size class (34% ≈ 50 %). The case of *S. alba* was similar to *A. marina,* being that the effect of selective harvesting has resulted in a relative redistribution of stems to mid-size classes. The observed (bars) and predicted (curve) stem size composition is displayed in figure 3.2. There are significant differences between the observed and predicted size distribution in both sites [Kombeni ($\chi^2 = 210.8989$ df = 5 $p < 0.0001$); Tsalu ($\chi^2 = 217.7398$ df = 5 $p < 0.0001$)]. The observed stem size distributions in both sites display selective harvesting, with over-harvesting of stems sizes 6-13 cm and 6-20 cm in Kombeni and Tsalu respectively. The general quality of the standing wood quantity in the two locations was dominated by the crooked tree form (Table 3.3). No significant differences were found in cutting intensity between sites (*Mann-Whitney U Test*, Z = -1.26192, p = 0.21), but stump densities were slightly higher in Tsalu (4,359 stumps ha^{-1}) than in Kombeni (4,167 stumps ha^{-1}).

Table 3.2: Stand table for the mangrove forest of Tudor creek. Values in parentheses indicate percentage of the total stem density per class per species, and the totals.

		D_{130} Class in cm						***Density***
Site	**Species**	**6**	**6.1-9**	**9.1-13**	**13.1-20**	**20.1-35**	**35**	*(Stems ha^{-1})*
Kombeni	*A. marina*	2 *(1.32)*	11 *(8)*	15 *(11)*	33 *(23)*	68 *(47)*	14 *(10)*	**143 *(11)***
	B. gymnorrhiza	8 *(44)*	4 *(22)*	2 *(11)*	4 *(22)*	_	_	**17 *(1)***
	C. tagal	36 *(66)*	16 *(29)*	2 *(4)*	1 *(2)*	_	_	**55 *(4)***
	R. mucronata	607 *(67)*	131 *(15)*	69 *(8)*	63 *(7)*	28 *(3)*	7 *(1)*	**905 *(72)***
	S. alba	24 *(16)*	25 *(17)*	35 *(24)*	38 *(26)*	20 *(14)*	4 *(3)*	**144 *(11)***
	Total	**675 *(53)***	**187 *(15)***	**123 *(10)***	**139 *(11)***	**116 *(9)***	**25 *(2)***	**1,264**
Tsalu	*A. marina*	15 *(9)*	20 *(12)*	24 *(14)*	39 *(23)*	60 *(36)*	10 *(6)*	**168 *(13)***
	B. gymnorrhiza	17 *(57)*	9 *(30)*	2 *(5)*	2 *(5)*	1 *(3)*	_	**30 *(2)***
	C. tagal	123 *(70)*	34 *(19)*	15 *(9)*	4 *(2)*	_	_	**176 *(14)***
	R. mucronata	590 *(79)*	113 *(15)*	23 *(3)*	16 *(2)*	9 *(1)*	_	**750 *(58)***
	S. alba	3 *(15)*	6 *(31)*	5 *(23)*	3 *(15)*	3 *(15)*	_	**21 *(2)***
	X. granatum	2 *(1)*	42 *(33)*	16 *(13)*	30 *(23)*	34 *(26)*	5 *(4)*	**128 *(10)***
	Total	**748 *(58)***	**224 *(17)***	**85 *(7)***	**94 *(7)***	**107 *(8)***	**43 *(3)***	**1,301**

Table 3.3: Tree form distributions in Tudor creek showing the densities per ha and percentages (in brackets) composition per species.

Form	1	2	3
A. marina	-	16 *(11.05)*	124 *(88.67)*
B. gymnorrhiza	2 *(9.80)*	7 *(33.33)*	11 *(56.86)*
C. tagal	7 *(8.61)*	42 *(52.15)*	32 *(39.23)*
R. mucronata	24 *(3.42)*	283 *(40.23)*	396 *(56.35)*
S. alba	-	28 *(40.57)*	40 *(59.43)*
Total	33 *(3.30)*	375 *(37.05)*	603 *(59.66)*

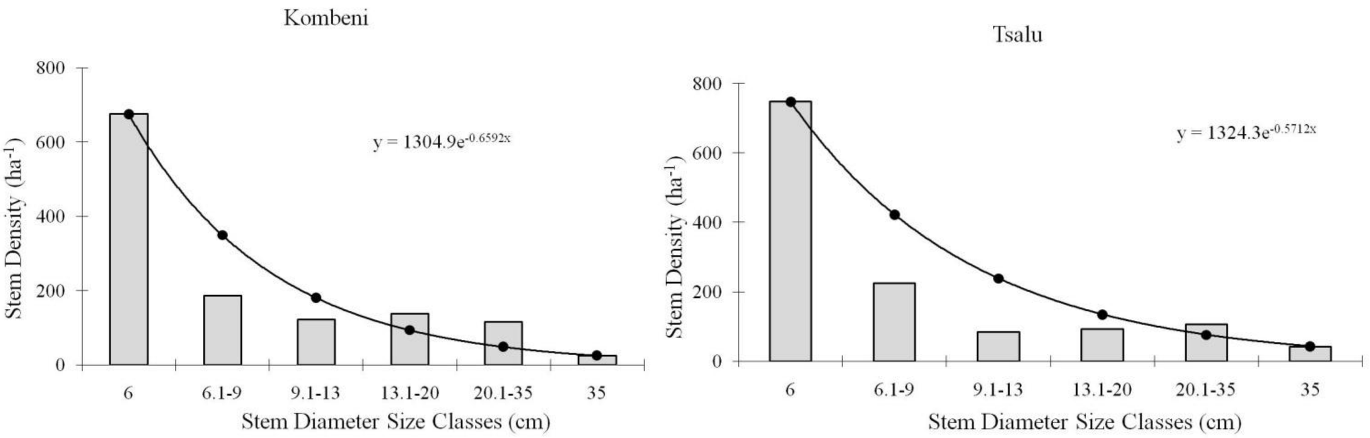

Figure 3.2: Observed (bars) and predicted (curve) size class distribution of mangrove forests of Tudor creek. A high *'k'* value in the stand curve $y = ke^{-ax}$ for Kombeni and Tsalu reflects the occurrence of sporadic natural regeneration in the forest.

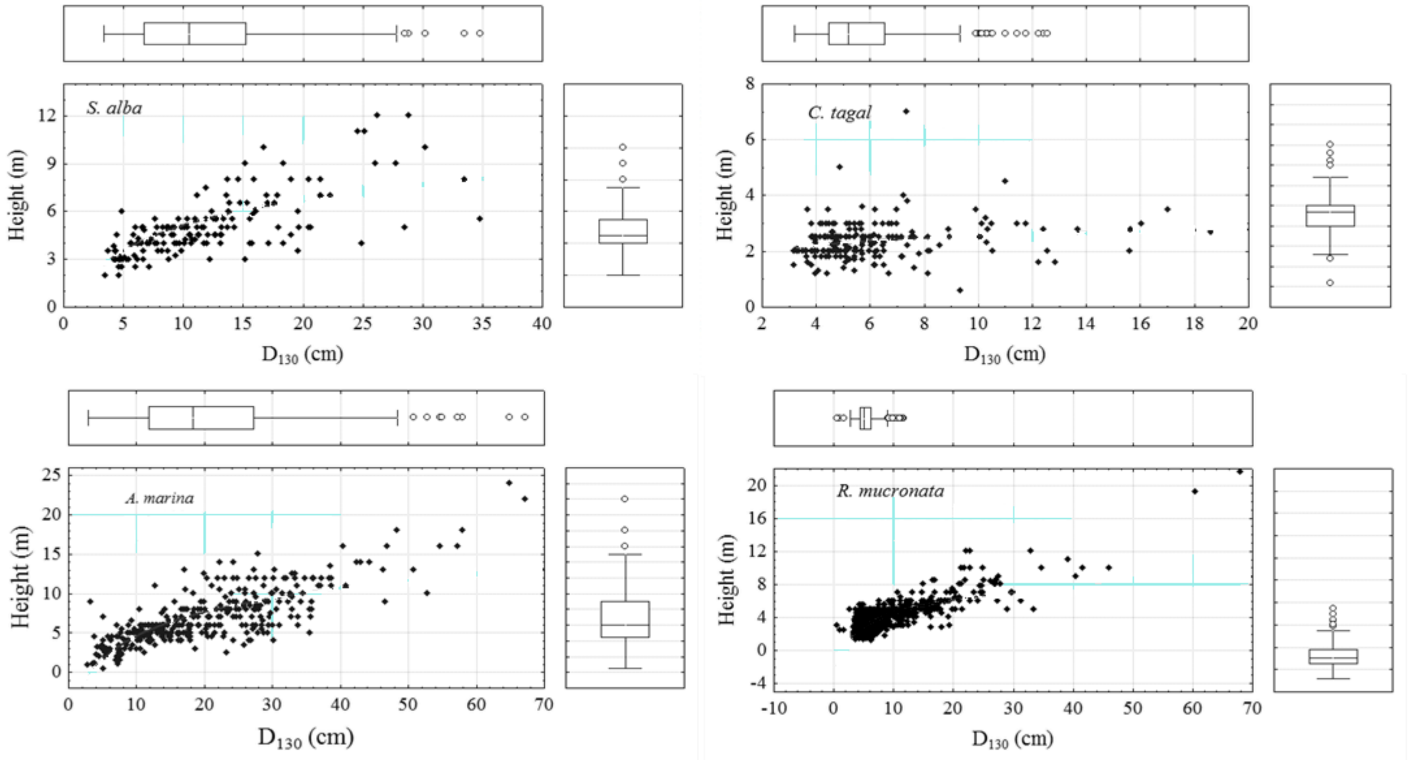

Figure 3.3: Scattergrams of height vs. D_{130} distributions of *R. mucronata*, *A. marina*, *C. tagal* and *S. alba* for Tudor creek mangroves. The box plots display percentile distribution in each case. The extremities of the plot correspond to the maximum and the minimum observations in the data set. The ends of the box are positioned at the 25% and 75% percentiles of the data set.

3.3.3 Regeneration

Significant differences in juvenile densities were observed across the study area. On average the juvenile densities were in the range 21,905 – 33,965 ha^{-1}, with *R. mucronata* representing approximately 45 – 75% of the total juveniles (Table 3.4).

Examination of dispersion pattern showed a tendency towards a random distribution ($I_\delta << 1$) of adult trees, but a clustered distribution for juveniles ($I_\delta > 1$) (Table 3.5). There was a significant negative correlation between open canopy and seedling density per sampled plot (N = 141; Spearman = -0.373086; t (N-2) = -4.74094; p = 0.000005), which together with the proximity to the mother plant and altered environmental gradients in gaps (Clarke, 2004), could have contributed to the clumped distribution of seedlings.

Table 3.4: Juveniles density (saplings ha^{-1}) in Tudor creek. Values in parentheses indicate percentages.

Site	***Species***	***Density ha^{-1}***			***Total Density ha^{-1}***
		RCI 0-40 cm	**RCII** 40.1-150 cm	**RCIII** 150.1-300 cm	
Kombeni	*A. marina*	17,080 (99.70)	37 (0.22)	15 (0.09)	17,132 (50.46)
	B. gymnorrhiza	60 (33.90)	81 (45.76)	37 (20.90)	177 (0.52)
	C. tagal	271 (25.78)	411 (39.11)	368 (35.01)	1,051 (3.10)
	R. mucronata	4,357 (27.94)	4,802 (30.80)	6,432 (41.25)	15,592 (45.92)
	S. alba	_	_	1	1 (0.00)
	X. granatum	_	_	_	_ (0.01)
	Total ha^{-1}	21,768 (64.11)	5,331 (15.70)	6,853 (20.18)	33,953
Tsalu	*A. marina*	2,234 (93.28)	78 (3.26)	82 (3.42)	2,395 (11.09)
	B. gymnorrhiza	318 (81.54)	47 (12.05)	25 (6.41)	390 (1.81)
	C. tagal	1,251 (38.15)	870 (26.53)	1,158 (35.32)	3,279 (15.18)
	R. mucronata	3,713 (23.95)	4,066 (26.22)	7,725 (49.82)	15,505 (71.77)
	S. alba	_	_	_	_
	X. granatum	34 (91.89)	2 (5.41)	_	37 (0.17)
	Total ha^{-1}	7,550 (34.95)	5,064 (23.44)	8,991 (41.62)	21,605

Very few saplings were observed for *S. alba*. The equivalent regeneration ratios i.e. RCI: RCII: RCIII were; 4:1:1 for Kombeni and 1:1:2 for Tsalu. The regeneration ratios are not within the range of effective stocking rate 6:3:1 for saplings described by Chong (1988). However, Tudor creek mangroves can still be considered potentially of good regeneration capacity taking into account the seedling densities.

Table 3.5: Morisita Index in relation to distance from the creek for juvenile *R. mucronata* and *A. marina*. The distribution shows a tendency towards clustering. There is a distinct zonation pattern, with *A. marina* in the landward and seaward sides and *R. mucronata* in the mud flats.

			Distance from creek (m)				
	Site	**Species**	**20**	**50**	**150**	**300**	**700**
Kombeni	Mikindani	*A. marina*	0.0005	-	-	-	-
		R. mucronata	**1.25**	0.52	0.45	-	-
	Jomvu	*A. marina*	**17.4**	-	-	-	-
		R. mucronata	0.01	0.07	0.07	-	-
Tsalu	Junda	*A. marina*	0.066	-	-	-	-
		R. mucronata	**3.32**	**6.41**	**1.62**	0.017	0.016
	Mirarani	*A. marina*	0.0001	**2.60**	-	0.0028	-
		R. mucronata	0.1126	**8.09**	**2.18**	0.056	0.003

3.4 Discussion

3.4.1 Vegetation structure

One notable change includes a decline or loss in *L. racemosa* caused by an observed silted landward fringe. Anthropogenic influences such as indiscriminate and unregulated harvesting, raw domestic sewage pollution and enhanced siltation have had cumulative effects on the structure and regeneration of the forest. As a consequence the forest is characterised by high density of stumps and a dominant crooked tree form. However, the impacts of raw domestic sewage cannot be proven from current observations. Although studies generally indicate sewage enhances mangrove growth rates (Feller *et al*., 2003b; Boonsong *et al*., 2003), with no apparent negative effects (Wong *et al*., 1997; Vaiphasa *et al*., 2007). The complexity index, an indicator of the overall structural development of a forest stand, was low compared to observations along the Kenyan coast (Kairo, 2001; Kairo *et al*., 2002a, b; Bosire *et al*., 2003; Bundotich, 2007). This is an indication of a dominant young vegetation, with low basal area and height.

The stand density, as estimated is lower for large trees, which is to be expected, and diminished for the stem diameter range 6 – 13 cm and 6 – 20 cm for Kombeni and Tsalu respectively. Possibly an implication of higher harvests in a rural setting (Figure 3.2). Theoretically, in an uneven-aged forest there is a normal series of age-gradations, depicted by the reversed J curve [De Liocourt’s negative exponential model (Clutter *et al*., 1983)]. This relation in size classes is not observed for Tudor creek, and when put into size-frequency diagrams it is not possible to obtain a simple correlation between size classes and densities (Figure 3.3). It indicates a forest disturbance regime according to direct needs by the people, lacking a consistent harvesting plan, resulting in a haphazard spatial distribution of different size classes, with a highly selective graphical frequency distribution. Assuming that tree size express age, we can use the density curves obtained in this study (Figure 3.2) to predict the composition of the future managed forest. This can be done by harmonizing the irregularities in the stem size distribution by harvesting ‘excess’ trees in those size-classes observed densities are higher than expected. In the long run, the introduction of multiple uses of mangrove wood will reduce stem density per class.

3.4.2 Regeneration

Natural regeneration was observed all over the creek, with *R. mucronata* seedlings and saplings dominating, while *A. marina* seedlings were abundant, with low density of saplings, implying high mortality of seedlings and/or saplings resulting in low recruitments into successive regeneration classes. *C. tagal*, *B. gymnorrhiza* and *X granatum* had very low regeneration levels, with *S. alba* having particularly low regeneration, with the adults visibly impacted by unknown insects and/or pathogens, and in some areas suffering die backs, an observation also reported in 1992 (SPEK, 1992). Observations indicate regeneration based on the "direct replacement" model, with species replaced by members of the same species as reflected by stand composition. However, establishment and survival is diminished due to site spatial and temporal heterogeneity introduced by canopy gap formation and siltation (Flower and Imbert, 2006).

In this study, we observed high abundance of juveniles in smaller gaps and under canopies than in larger gaps. Clarke and Kerrigan (2000) and Minchinton (2001) reported the impacts of canopy gap size on regeneration. They cited an extremely local dispersal of propagules, coupled with habitat heterogeneity due to disturbance and altered topography as the causes of altered regeneration in large canopy gaps. In Tudor creek, harvesting has enlarged canopy gaps, while siltation has altered edaphic conditions, causing habitat heterogeneity and altered topography, impacting on propagule dispersal, establishment, survival and growth. This scenario is unsustainable, implying inadequate regeneration, altered forest growth and longer times for canopy gap closure as a result of limiting conditions within large canopy gaps (Clarke and Kerrigan, 2000; Clarke, 2004). Dominance of *R. mucronata* can be linked to the larger propagule mass, which enables it to withstand siltation, compared to the smaller seeds of the other species (personal observation). In addition, enhanced predation of dispersed propagules confers advantages to propagules of species that are local canopy members, limiting the range of species available to fill the gaps (Osborne and Smith III, 1990; Clarke and Kerrigan, 2002; Bosire *et al*., 2005b).

3.4.3 Implications

Forest canopy gaps are common in mangroves and usually result after disturbances such as selective harvesting and natural mortality of trees (Duke, 2001). These canopy gaps drive the gap phase regenerative cycles in mangrove forests (Clarke and Kerrigan, 2000; Duke, 2001; Imai *et al*., 2006; López-hoffman *et al*., 2007). The gap creation frequency in Tudor creek is high, stimulating regeneration that approximates selection forest working, favouring *R. mucronata* (FAO, 1994; Ewel *et al*., 1998; Clarke and Kerrigan, 2000; Duke, 2001). However, regenerative turnover might have overwhelmed progress in stand development, causing forest growth reversal resulting in a relatively young forest (Duke, 2001), as older trees have either been selectively harvested or impacted by silt. Siltation, though not systematically studied, causes mortality of both adults and juveniles, through inhibition of gaseous exchange of roots causing root damage, oxygen deficiency and eventually reduced vigour (Ellison, 1998; Thampanya *et al*., 2002).

Gaps are generally characterised by increased light and temperature, high soil evaporation rates with high transpiration rates from trees surrounding the gaps, and high pore-water salinities (Ewel *et al*., 1998; Duke, 2001; Alongi and de Carvalho, 2008). They alter propagule predation by crabs (Osborne and Smith III, 1990; Clarke and Kerrigan, 2002; Bosire *et al*., 2003; Bosire *et al*., 2005b), reduce proportion of reproductive trees (‘propagule limitation’) and diminish soil stability (Ewel *et al*., 1998; Clarke and Kerrigan, 2000; Kairo, 2001). It has been observed that natural gap formation induces less severe physical and chemical changes than gaps formed by human disturbances (Clarke and Kerrigan, 2000; Duke, 2001), resulting in lowered regeneration and development rates under human disturbances (Allen *et al*., 2001; Duke, 2001). In our observation, larger gaps are likely to have more diverse physico-chemical gradients, especially in Mombasa, characterised with a pronounced dry season, manifested by dramatic vegetation responses. This is in contrast to high rainfall areas, lacking distinct wet or dry season, with a buffering effect of freshwater input (Ewel *et al*., 1998; Pinzón *et al*., 2003).

Gaps created by selective harvesting of branches recover over long periods (Ellis and Bell, 2004), with the extent of recovery governed in part, by the regenerative capabilities of the damaged mangrove species and the nature and severity of the causative agent (Snedaker *et al*., 1992; Ellis and Bell, 2004). In contrast to species such as *A. marina*, which hold reserve buds in the stem, mature individuals of the Rhizophoraceae (*Rhizophora* and *Ceriops*) hold reserve buds only in the thin terminal twigs and “conditions severe enough to remove or kill all branches possessing viable reserve buds will potentially eliminate *Rhizophora*” (Hutchings and Saenger, 1987). This implies that specific management principles need to be developed while harvesting different species to ensure that the diversity of the forest is maintained and gap recovery is fast enough to minimise gap sizes.

3.5 Conclusion

The mangroves of Tudor creek, though disturbed, are not irreversibly degraded. However, stand densities and basal areas were lower than for Rhizophoraceae dominated forests along the Kenyan coast (Gazi Bay, 8 – 24 m^2ha^{-1}, 1130 – 2571 stems ha^{-1}; Kairo, 2001; Bosire *et al.*, 2003; Mida creek and Ngomeni, 24.05 - 46.97 m^2ha^{-1}, 2075 - 2142 stems ha^{-1}: Kairo *et al.*, 2002a, b; Bundotich, 2007). Globally, the values are in the middle of the reported range for similar forests (322 - 2470 stems ha^{-1}; Lugo and Snedaker, 1973; Paijmans and Roller, 1977; Jimenez *et al.*, 1985; Smith III, 1988). Within Mombasa, natural disturbances are either relatively small or rare, leaving anthropogenic disturbances the principal threat. Current and future management therefore must focus on regulating the anthropogenic element. Harvesting should be regulated through forest zoning for multiple uses, coupled with a harvesting regime that incorporates replanting and closed periods, ensuring adequate regeneration and forest growth in the long run. This will regulate canopy gap sizes, as intermediate level of gap creation may be optimal for long-term stand stability (Duke, 2001).

To add value to peri-urban mangroves, management for multiple uses as opposed to single (forestry) products is desirable (Rönnbäck, 1999; Barbier, 2000; Ellison, 2008a; Nagelkerken *et al.*, 2008). Thus, in addition to meeting the wood demands of local populations, ‘environmental’ forests as no-cut zones should be established as protection of habitats for migratory birds and other fauna. Introduction of eco-tourism, coupled with an integrated land use plan may regulate pollution and siltation and involve local communities in management. Coupled with an in-depth understanding of the early drivers in mangrove establishment (Krauss *et al.*, 2008), growth and productivity of mature forests (Komiyama *et al.*, 2008), factors driving mangrove dispersion (Di Nitto *et al.*, 2008; Triest, 2008) and spatio-temporal dynamics of mangrove vegetation (Berger *et al.*, 2008) this will help in starting and sustaining recovery programmes (Bosire *et al*., 2008). This will serve to boost the ecosystem resilience in the long-run.