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Full Length Research Paper

Assessing ecosystem effects of small–scale cutting of Cameroon mangrove forests

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One of the most universal forms of resource-use in the tropics is small-scale wood exploitation; but ecologists are only starting to study its effects. This paper examines the effects of small-scale wood harvesting on forest structure and composition of mangrove forests. A stratified sampling method was used to select the sample zone. The forest characteristics were assessed by employing the quadrat/census plot method (Cintron and Schaeffer, 1984). To assess canopy structure, plot perimeter was used as a basis for line intercept sampling (Lertzman et al., 1996). Two-thirds of all canopy gaps were caused by human activities and this might have dramatic effects on regeneration because there were significantly more seedlings in canopy gaps compared with closed canopy areas. *Rhizophora* was the dominant species and formed a virtually monospecific stand in the coastline zone with a gradual transition to a mixed forest of *Laguncularia, Avicennia* and *Rhizophora*. Ecological characteristics such as mean tree density, seedling density, mean diameter at breast height, basal areas and gap sizes differed among seaward, middle and landward zones. The findings from the present study highlight that the ecological effect of small scale wood exploitation is a potential threat to mangrove forest ecosystem health.

Key words: Cameroon, mangrove ecology, forest ecosystem health.

INTRODUCTION

Mangrove forests like most other ecosystems provide a full range of goods and services. They play an important role in maintaining a healthy coastal ecosystem by providing far reaching direct and indirect services (Dahdouh-Guebas, 2001; Walter et al., 2008). The social, ecological and economic importance of mangrove forests is enormous. They are among the world's most productive systems, have a high primary production, high rates of recycling and provide a high supply of nutrient source that supports many complex food chains (Clough, 1993; Lefebvre and Poulin, 2000). These features make mangrove systems perfect as breeding and nursery grounds for many marine species including commercially important fishes (Baran, 1999; Alongi, 2002). Mangroves also contribute significantly to the global carbon cycle.

Total global mangrove biomass is approximately 8.7 gigaton dry weight, that is 4.0 gigaton of carbon (Clough, 1993; Twilley et al., 1992). Mangroves forests are reported to have historically provided a variety of renewable products including timber, food, charcoal, firewood and medicine to many local communities worldwide (Primavera, 1995; Dahdouh-Guebas, 2001; Walters, 2003). Mangrove forests are subject to a number of natural and anthropogenic threats. Though there has been considerable attention paid to natural disturbances of mangroves such as hurricanes and climate change (Roth, 1992; Gilman et al., 2008), human activities in these coastal areas such as physical alteration of the habitat, over-exploitation of the resources and pollution cause significant pressure on the environment. These pressures have increased steadily as the human population increases. For several decades, mangrove forests have been cleared and degraded on an alarming

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scale worldwide (Hamilton and Snedaker, 1984; Aksornkoae et al., 1992). Mangroves in many parts of the world are also affected by local-scale exploitation. The negative impact of local-scale exploitation on mangrove forest health varies from place to place and although the potential impacts are huge, formal assessments of the effects are uncommon (Walters, 2005a). The impacts are likely to be complex and may include social, economic and environmental dimensions. Frequent, but low intensity, small-scale mangrove exploitation has significant impact on forest structure, but limited information is available on how mangrove exploitation affects forest composition and regeneration (Eusebio et al., 1986; Smith and Berkes, 1993; Barnes, 2001). Cameroon has a growing coastal population as a result of which increasing use of the country's natural resources is endangering several ecosystems, especially estuarine systems. Mangroves are in decline in Cameroon mainly due to firewood extraction and the cutting of poles for construction (Longonje, 2002). The floristic composition of Cameroon mangrove is characteristic of the Atlantic mangroves of West Africa. It is dominated by *Rhizophora* and comprises mostly of three species *R. mangle, R. harrisonii* and *R. racemosa* (Spalding et al., 1997). Other mangrove species include *Avicennia germinans, Laguncularia reacemosa, Conocarpus erectus, Acrostichum aureum, Pandanus candelabrum* and *Nypa fructicans* (Spalding et al., 1997). The study area provides several ecosystem services such as natural coastal barriers, recreation and fisheries. Local communities in and around the mangroves depend on the forest for their livelihood.

One major socio-economic activity in the mangroves is artisanal fishing; the fish catch is estimated between 76 and 106 tons per year (Gabche, 1997). Some of the densely populated and industrial towns are located at the fringe of mangrove forests, notably Douala (the economic capital), Limbe and Tiko. The Cameroon mangrove is biologically diverse. Apart from the different species of fishes and birds, many endangered species such as marine turtles (*Lepidoshelys olivacea*), dwarf crocodile (*Crocodylus cataphractus*) and West African manatee (*Trichechus senagalensis*) can be found.In this paper we examine disturbance associated with local community forest exploitation. The goal of the present study was to improve understanding of the effects of small-scale wood harvesting on forest structure and composition of mangrove forests for better resource management decisions. Our objectives were to quantify (1) the structure of the mangrove forest in Cameroon estuary, and (2) the changes in the mangrove forest in response to small-scale harvesting.

METHODS

Field work for this study was undertaken from 2008 to 2009. The forest characteristics were assessed by employing the quadrat/ census plot method (Cintron and Schaeffer, 1984).

Study area

This study was carried out in the Cameroon Estuary mangrove (Figure 1) located in the South–Western part of Cameroon between latitude 3° 83' to 4° 10' N and longitude 9° 25' to 10° 00' E. It is a large forest of approximately 1,750 km^2 and is representative of the bigger mangrove areas in Cameroon. The coastal and marine environment of Cameroon forms part of the southern section of the Gulf of Guinea Large Marine Ecosystem (Price et al., 2000). The coastline stretches from the Equatorial Guinea border at latitude 2° 30' to 4° 67' N at the Nigeria border and it is estimated at about 400 km in length (Price et al., 2000).

Plot description

Two different quadrat sizes (50 × 50 m and 10 × 10 m) were used, with the corners and boundaries of plots marked using calibrated measuring ropes and tapes. The larger plot size was used in some of the stands surveyed because trees were typically large and sparsely located. The smaller plot size was used where stands had typically small and densely crowded trees. A stratified sampling method was used to select the sample zones in the Cameroon Estuary. The mangrove forest was divided into 3 zones: seaward (coastline forest), middle (interior forest) and landward (fringe forest). Nine sites were selected randomly to capture representative forest structure and characteristics and these sites were distributed equally between seaward (coastline forest), middle (interior forest) and landward (fringe forest) zones (Figure 1). Thirty-one plots were sampled randomly for floristic composition, stand and canopy structure with almost equal effort in coastline, interior and fringe forest. To assess the floristic composition and stand structure, data were collected on tree species composition, diameter at breast height (dbh), tree height, canopy cover, numbers of seedlings, gaps, gap size, stumps and snag (dead stems). In each plot, every tree was numbered, marked and measured (> 1.0 m tall) and seedlings (< 1.0 m) recorded (Walter, 2005b).

The diameter at breast height (dbh) of each tree stem was measured at 1.3 m or above the highest prop root following Cintron and Schaeffer (1984). Tree height was measured using marked bamboo poles and clinometers.

Canopy assessment

To assess canopy structure, I used the plot perimeter as a basis for line intercept sampling (Lertzman et al., 1996), providing a transect length of 200 m for the 50 × 50 m plots and 40 m for the 10 × 10 m plots. At 2.5 m intervals along each transect, the canopy was viewed vertically upward, and scored as CC (closed canopy) or CG (canopy gap) defined as an area where the canopy is noticeably reduced compared to adjacent areas (Runkle, 1992). All gaps were classified as either natural (that is not caused by human activity; for example gaps caused by fallen tree due to strong wind) or induced (that is caused by humans; for example gaps caused by human clearing or wood exploitation). Gap age was estimated from the stage of decomposition of the gap-making tree, as fresh (recent gap), old (dry trees, but no sign of decomposition) and very old (decomposing trees). To estimate gap size, the distance and angle from the centre of the gap to the gap edge was measured in each of 8 (45°) sectors and the area of the triangles summed (Lertzman et al., 1996).

Statistical analysis

Descriptive statistics (histograms) were used to analyse species composition, distribution and utilization. To test whether or not,

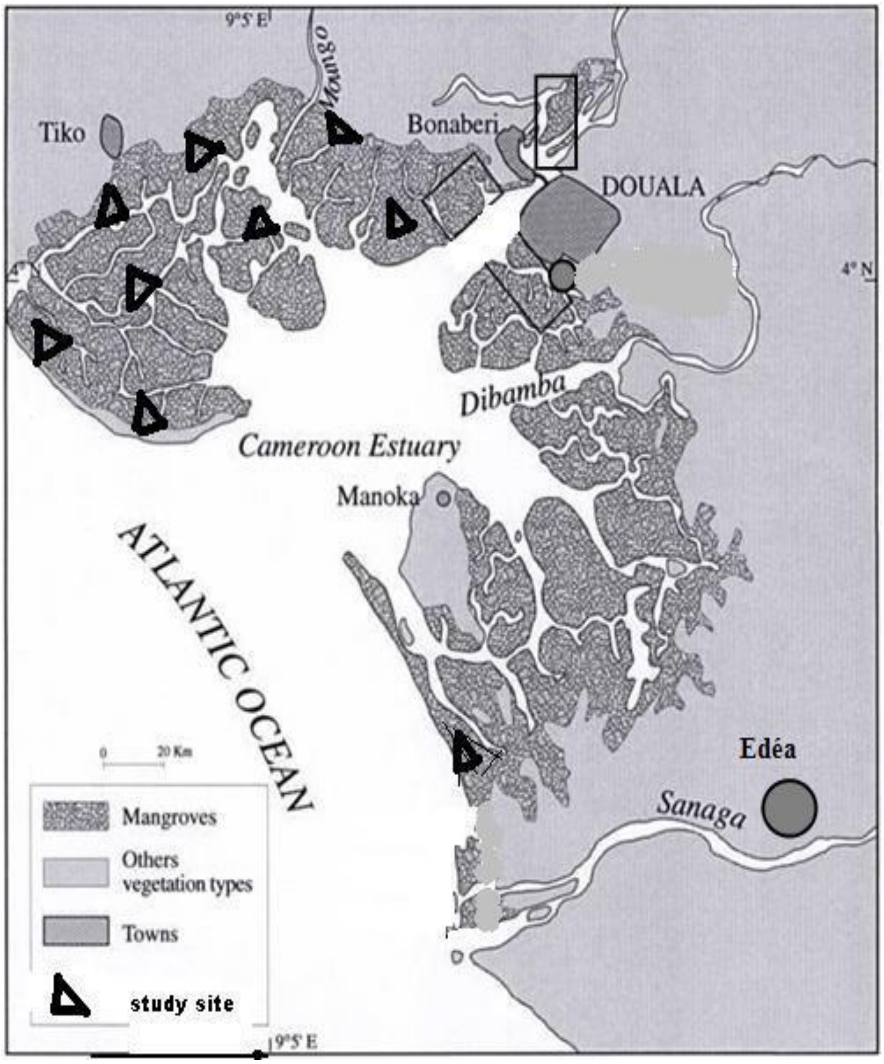

Figure 1. Study site and locations of plots used to evaluate forest structure.

there was a significant difference between selected forest characteristics in coastline, interior and fringe zones, the non parametric Kruskal-Wallis ANOVA was performed. This was because the data were heteroscedatic and transformation (square root and log) did not normalize the data. Nonparametric t- tests were employed to test for differences in seedling abundances in canopy gaps and closed canopy. All of the aforementioned analyses were performed using SPSS and PRISM 5.

RESULTS

Thirty one plots were sampled for forest structure and composition, 12 within the coastline, 8 within the interior forest and 11 within the forest fringe. A total of 3167 individual trees, 423 stumps and 103 snags were recorded. *Rhizophora* (red mangrove) was the dominant species (83.6%) followed by *Avicennia* (black mangrove) at 9.1% and *Laguncularia* (white mangrove) at (7.1%). Virtually, monospecific stand of *Rhizophora* occupied in the coastline zone, with a gradual transition to a mixed forest of *Laguncularia, Avicennia* and *Rhizophora* occupying the interior and fringe zones. Although, *Avicennia* and *Laguncularia* were abundant in interior and fringe zones, they were not the dominant species (Figure 2).

Forest structure

Coastline forest had lower mean tree density and seedling density compared to interior and fringe forest, but the difference was not significant. The mean diameter at breast height (dbh) and basal areas were higher in coastline forest compared to interior and fringe forest, but the difference was not significant (Tables 1 and 2). When comparing the seedling density between the forest zones, the interior had the highest density, but the difference was not significant (Tables 1 and 2).

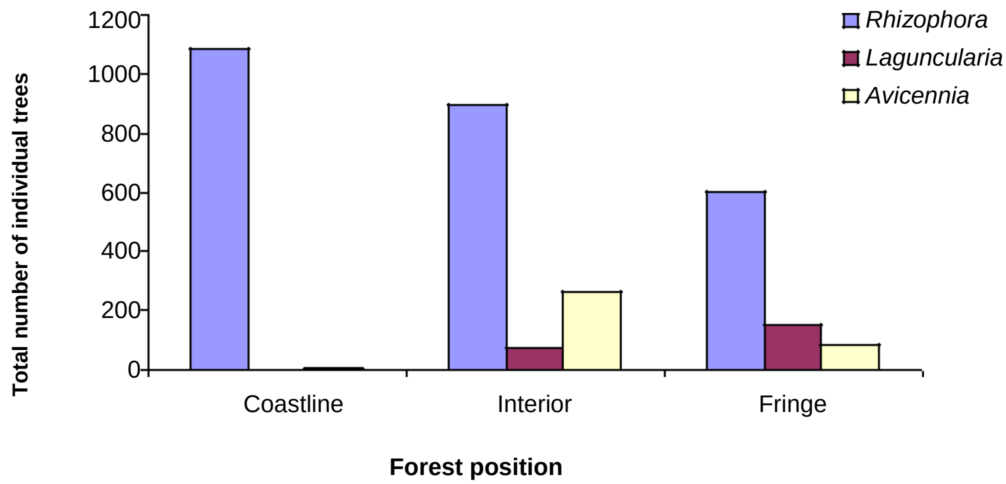

Figure 2. Mangrove tree species distribution within the 3 major zones.

Table 1. Summary of selected ecological characteristics of the mangrove stands studied (means and standar d deviation: for numbers of replicate plots).

Characteristics	Coastline plots	Interior plots	Fringe plots	Average
Tree density (n/100 m^2)	11.9 (19.2)	15.4 (21.1)	20.7 (24.8)	16.0 (20.2)
Diameter at breast height (dbh) of stem (cm)	27.6 (8.7)	24.7 (7.7)	19.2 (7.8)	23.8(19.7)
Stem basal area (m^2/ha)	74.2 (38.8)	59.5 (38.1)	46.6 (5.2)	60.1 (29.8)
Seedling density (n/100 m^2)	24.8 (39.7)	31.4 (45.4)	14.3 (14.6)	23.5(40.1)

Canopy gaps

257 gaps were recorded during the survey, two-thirds (66%) of which were caused by human influence, whilst one-third (34%) was due to natural factors (Table 3). Human influence was responsible for most of the gaps created in fringe, coastline and interior forest (73, 72 and 54% respectively) (Table 3). Average gap size of 3.1m^2 was recorded and gap size differed significantly between zones (ANOVA, P = 0.001) (Table 2). The coastline and fringe gap size were both significantly different from interior (Figure 4). The average gap density of 27.4 was recorded overall (Table 3), but there were significant differences between zones (ANOVA, P = 0.02) (Table 2), with the fringe, interior and coastline canopy gap density differing significantly from one another (Figure 3). Seedling density was not significantly different between zones (Table 2). The relationship between seedlings and canopy was examined as an alternative way to estimate the effect of exploiting forest on mangrove regeneration. Significantly, more seedlings were observed in canopy gaps compared to closed canopy areas (t = 3.5, P = 0.01, Table 4). *Rhizophora* seedlings were more abundant in canopy gap than in closed canopy areas (t = 2.4, P = 0.04), whilst *Avicennia* and *Laguncularia* were not profuse in canopy gap.

Forest species composition

The size-frequency distributions of all mangrove species are represented in Figure 5. All three species showed a higher abundances of stems in small size classes (<25 cm). In contrast to *Rhizophora*, *Avicennia* is completely absent from size classes greater than 95 cm and *Laguncularia* from size classes more than 25 cm. For *Rhizophora*, coastline forest plots had many large stems greater than 25 cm, whilst the interior forest plots had few medium sizes stems, and lack very large stems and fringe forest plots only supported small stems (≤25 cm: Figure 6).

DISCUSSION

According to Walter (2004), some mangrove forests have been dramatically altered through small-scale cutting and deliberate planting of trees by local communities. Few studies have examined the ecological impacts of small–scale exploitation of mangrove with the aim of assessing forest health. According to Smith and Berkes (1993), small-scale cutting of mangrove in the Caribbean reduces the abundance of large trees, but greatly increase the density of smaller trees. Esusebo et al. (1986) found that

Table 2. ANOVA results of selected ecological characteristics in coastline, interior and fringe zones.

Characteristics	Source of variation	SS	DF	MS	F	P-value
Tree density (100 m^2)	Between groups	582	2	291.05	0.69	0.51
	Within groups	11236	27	416.16		Not significant
	Total	11818	29			
Canopy gap density (100 m^2)	Between groups	188	2	94.19	4.72	0.017
	Within groups	558	28	19.96		Significant
	Total	747	30			
Diameter at breast height (dbh) of stem (cm)	Between groups	92563	2	46281.74	78.66	0.58
	Within groups	1863447	3167	588.39		Not significant
	Total	1956010	3169			
Canopy density (100 m^2)	Between groups	144	2	72.27	0.74	0.48
	Within groups	2654	27	98.31		Not significant
	Total	2798	29			
Stem basal area (m^2/ha)	Between groups	82	2	41.24	75.59	0.32
	Within groups	1744	3167	0.55		Not significant
	Total	1826	3169			
Gap size (m^2)	Between groups	1	2	0.48	10.43	0.001
	Within groups	1	27	0.046		Significant
	Total	2	29			
Seedling density (100 m^2)	Between groups	51	2	25.97	0.014	0.9
	Within groups	46779	27	1732.56		Not significant
	Total	46831	29			

Table 3. Canopy gaps and their causes in the 3 forest zones.

	Coastline	Interior	Fringe	Average
Canopy gap density (m^2/100 m^2)	3.7	23.8	54.7	27.4
Gap size (m^2)	3.5	1.1	4.7	3.1
Human cause (%)	72	54	73	66.3
Non-human cause (%)	28	46	27	33.6

cutting of mangroves in the Philippines resulted in stunted and shrubby tree growth, but other studies have shown otherwise. For instance, Nurkin (1994) suggests that small-scale mangrove exploitation has an insigni-ficant effect on mangrove forest structure. In the present study, the canopy gaps created were relatively small, the largest gap size measured was 72.2 m^2, but the mean gap size was much smaller at 3.1 m^2, relatively small when compared to findings from other mangrove studies. For example, Ewel et al. (1998) recorded a mean gap size of 158 m^2 for mangrove in Kosrae Micronesia, though the author deliberately ignored gap sizes less than 10 m^2. Smith (1992) observed gap sizes of mature mangrove forest in Australia of 40 to 120 m^2, but it is possible that he overlooked gaps of less than 10 m^2. By contrast, Walter (2005a) found a smaller mean gap size of 2.6 m^2 for Philippines mangroves and studies of other forest types have shown that such small canopy gaps have an important effect on the forest structure (Feller and Mckee, 1999; Kennedy and Swaine, 1992). Exploita-tion of mangrove wood product was not completely species-selective in this study, but *Rhizophora* was the preferred species. There is evidence that wood exploitation might have changed *Rhizophora* stem-size

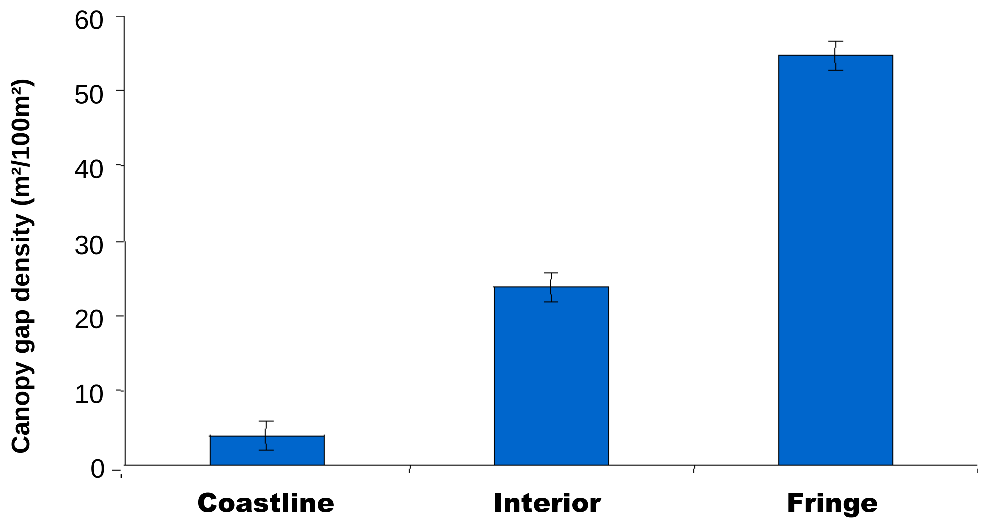

Figure 3. Comparison of canopy gap density within zone.

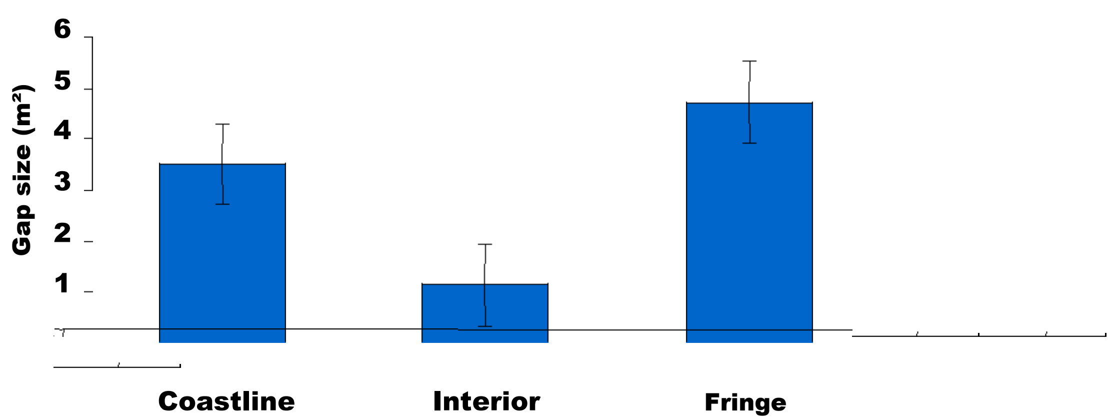

Figure 4. Comparison of gap size within zone.

Table 4. Seedling abundance (total count) of different mangrove species in open and closed canopies.

Species	Canopy gap	Closed canopy
Rhizophora	863	375
Laguncularia	220	59
Avicennia	161	93
Total	1244	527

distribution. Coastline forest (least accessible) is characterised by *Rhizophora* with large stem size, whilst the interior forest has medium range stem sizes and the fringe forest have relatively small stem sizes (Figure 6). This study suggests that mangrove forests differed structurally from the fringe to the coastline, due to a combination of anthropogenic and natural factors (Table 1).

Also, the further one moves away from the residential areas, the bigger the tree sizes, although other factors such as soil salinity and nutrient concentrations are known to influence tree size (Calumpong and Menez, 1997). However, *Avicennia* and *Laguncularia* species did not show clear patterns of size class distribution (Figure5). Mangroves are thought to recover quickly after disturbance (Smith and Berkes, 1993), but the evidence

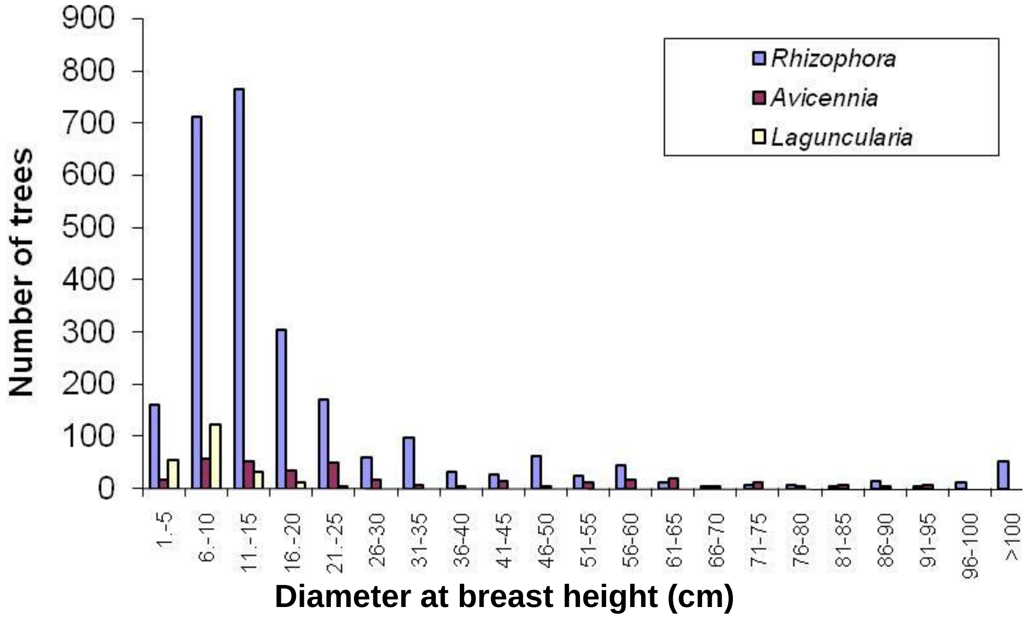

Figure 5. Size-frequency distribution of (dbh) of *Rhizophora*, *Avicennia* and *Laguncularia* species (all 3 zones combined).

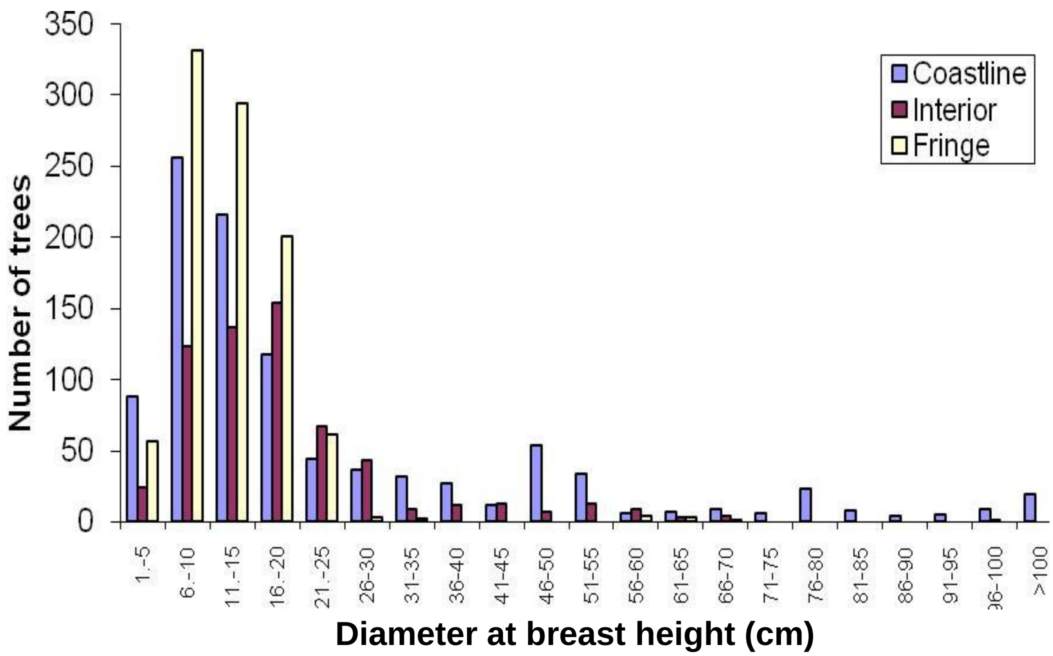

Figure 6. Size-frequency distribution of dbh of *Rhizophora* in coastline, interior and fringe forest plots.

is mixed. Thus, Ewel et al. (1998) found no differences in gap regeneration as a result of selective logging in Kosrae. Clarke and Kerrigan (2000) found that canopy gap had a strong influence on the abundance of mangrove seedlings. In the present study, the most sensitive species in producing the most seedlings was *Rhizophora* which shows a vital difference in gap regeneration (Table 3). Smith (1987) observed significant recruitment of *Rhizophora* species in gaps. According to Feller and Mckee (1999), gap size do not influence *Rhizophora* regeneration. According to Smith (1987), mangrove seedlings regenerate quickly in large numbers in canopy openings. In the present study, the seedling density is relatively low; this might suggest that the

Cameroon mangrove canopy is relatively closed and the forest structure is relatively healthy. Local communities in many tropical coastal regions have exploited mangroves for many years, but studies which examine local-wood utilisation and its ecological effects are uncommon. Policy makers and researchers alike have overlooked local-scale wood harvesting in mangrove forests. Management strategies are thus often developed without regards to either the ecological or economic significance of such attributes. Where such harvesting is significant, conservation efforts may encounter much opposition from the local communities. Forest biodiversity may also be eroded over the long term by continued selective removal of some species more than the others, and by the varied responses of species to cutting distribution (Walters, 2005a). At the same time, understanding patterns of wood use can inform management planning so that it is compatible with existing resource use practices (ITTO, 2002). For example a well- managed mangrove forest plantations provide abundant construction wood that can reduce harvesting pressures on natural forests, so long as the plantations are not permitted to encroach too much into natural forest (Walters, 2004).

A great deal of ecological research has been done on mangroves. Given this, and the fact that we know mangroves are harvested by local people in many tropical, coastal regions (Diop, 1993), it is remarkable that barely a few of published studies have examined local-scale wood use and its ecological effects on these forests. Study findings presented elsewhere (Walters, 2004) show how some mangrove forests have been dramatically altered through deliberate planting of trees by local people. Likewise, findings presented here demonstrate that small-scale, local wood cutting can be a significant form of ecological disturbance in mangroves. Forest structure was dramatically altered by cutting, but impacts on composition and regeneration were also detectable. Most notably in this respect is the finding that most mangrove species appeared to respond to small-scale cutting by significant recruitment of new species. In fact, it is plausible that mangrove forests in many places have already experienced significant changes to species composition as a result of past cutting and other anthropogenic influences (Walters, 2003). Efforts to understand these unique forests and ensure their long term conservation will depend in many cases on under-standing and effectively managing such small scale forest cutting.

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