



Ecological effects of small-scale cutting of Philippine mangrove forests

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Abstract

Small-scale wood harvesting is one of the most ubiquitous forms of resource-use in the tropics, yet ecologists have barely studied it. This paper examines the effects of small-scale woodcutting on forest structure, composition and regeneration of mangrove forests in the Philippines. Information for the study was obtained through the application of extensive bio-ecological assessments of forests and interviews of forest users. Cut mangrove forests were characterized by smaller trees, less basal area and more canopy gaps. At least two-thirds of all canopy gaps were caused by cutting. In spite of these dramatic structural effects, there was little demographic evidence to suggest that significant changes to current species composition are occurring, although this may, in part, reflect that some species have already been eliminated from study areas by past cutting. Among common species, *Rhizophora mucronata* was the only one that appeared to be negatively impacted from cutting in terms of its relative abundance. Although abundance varied, seedlings of all common species measured were taller in canopy and/or expanded gap compared to understory, with *Sonneratia* spp. showing the greatest and *Avicennia marina* the least response. The particular success of *A. marina* in cut forests may be explained by the ability of its seedlings to better persist in the understory and thereby exploit gaps when these are created by cutting. Among common mangrove species, all but *R. mucronata* appear to be regenerating well in cut forests: *Sonneratia* sp., *A. marina* and *R. apiculata* regenerate well by coppice regrowth into the abundant small canopy gaps found in uncut and especially cut forests. Findings from this study highlight the significance of small-scale cutting disturbance and coppice regeneration as biotic factors in mangrove ecology.

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1. Introduction

Anthropogenic disturbance is now regarded by many ecologists as central to understanding the

dynamics of ecosystems (McDonnell and Pickett, 1993; Vitousek et al., 1997). In the case of forests, the extent of human alteration worldwide and especially in the tropics is unrivaled in history (Noble and Dirzo, 1997). A plethora of research has examined deforestation and impacts of timber logging on tropical forests (e.g., Uhl et al., 1981, 1991; Kartawinata et al.,

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1989; Brown and Lugo, 1990; Ter Steege et al., 1995; Chapman and Chapman, 1997; Miller and Kauffman, 1998; Lindenmayer et al., 2002; Parrotta et al., 2002). However, remarkably few studies have examined the ecology of small-scale wood use and its impacts on tropical forests. This is a considerable oversight given that hundreds of millions of rural people living in and adjacent to forests in the tropics exploit them for construction materials, fuel wood and other non-timber products (Nepstad and Schwartzman, 1992; Peters, 1996a; Arnold et al., 2003). In fact, the few studies that have been done suggest that small-scale wood harvesting is pervasive and having substantial, often cumulative effects on forest structure, composition and regeneration (Nyerges, 1989; Smiet, 1992; Murali et al., 1996; Rikhari et al., 1998; Uma Shankar et al., 1998a, 1998b; Ramirez-Marcial et al., 2001; Awasthi et al., 2003; Luaga et al., 2004; Ticktin, 2004).

Like other tropical forests, mangroves have been cleared and degraded on an alarming scale during the past four decades, but they remain an important source of wood and other products for many coastal communities (Christensen, 1982; Hamilton and Snedaker, 1984; Aksornkoae et al., 1992; Diop, 1993; Lacerda, 1993). Trends in mangrove research parallel those for other tropical forests in their near-exclusive focus on large-scale, anthropogenic impacts. For example, mangrove deforestation resulting from expansion of aquaculture and other competing land-uses has been extensively studied (Primavera, 1995; Dewalt et al., 1996; Naylor et al., 1998; Walters, 2003). Likewise, research on cutting in mangroves has focused on state-managed forests subject to industrial/large-scale logging (Christensen, 1983; Putz and Chan, 1986; Aksornkoae et al., 1992; Hussain, 1995; Khoon and Eong, 1995). The few studies that have examined small-scale woodcutting of mangroves suggest significant impacts on forest structure, but provide limited information on how cutting might affect forest composition and regeneration (Eusebio et al., 1986; Smith and Berkes, 1993; Barnes, 2001; but see Pinzon et al., 2003).

Research presented here is part of a larger study of human influences on mangrove forests in the Philippines (Walters, 2000b, 2003). Other papers examine differences between natural and planted forests (Walters, 2000a, 2004). A companion paper to

this one describes patterns of woodcutting and use of mangroves (Walters, 2005). This paper examines the ecological effects of this small-scale cutting on mangrove forests. It seeks to answer the following questions: (1) Is forest structure substantially altered by cutting? (2) Are mangrove forest species impacted differently by cutting? and (3) Is forest regeneration affected by cutting?

2. Study areas and research methods

Fieldwork for this study was conducted in the Philippines between March and December 1997 in North and South Bais Bay and Bindoy, Negros Oriental (9°N/123°E) and on Banacon Island, Bohol (10°N/124°E). Bais Bay is located on the eastern side of Negros Island (Fig. 1). Mean temperatures in Bais vary from 25 to 30 °C and around 1500 mm of rainfalls annually, mostly during a distinct rainy season from July to December. The Bay occupies an area of approximately 5400 ha and is divided into North and South by Daco Island and a constructed causeway that connects Daco to the mainland. The coastal waters of North and South Bais Bay are productive and support a diverse fishery (Luchavez and Abrenica, 1997).

Three-quarters of the nearly 1000 ha of original mangroves in Bais Bay were cleared and developed into fishponds between 1930 and 1980 (Walters, 2003). Fishpond development especially impacted the less frequently flooded mangrove areas since ponds were usually built from the landward side first (Walters, 2003). Today, much of the perimeter of North and South Bais Bay is fringed by narrow bands of frequently flooded and often young forest. Mangrove stands, ranging in area from 3 to 30 ha, are also found at the mouths of each of four rivers that empty into the Bays, and a particularly large and well-developed stand of forest, called Talabong, extends as a peninsula across the seaward front of South Bais Bay. Many coastal residents also plant mangroves in Bais, but the distribution of plantations is patchy. Most plantations in Bais are found immediately adjacent to settlements or along the seaward perimeter of fishponds (Walters, 1997, 2003).

Bindoy is located 20 km north of Bais Bay. An extensive mangrove forest (about 100 ha) is located on

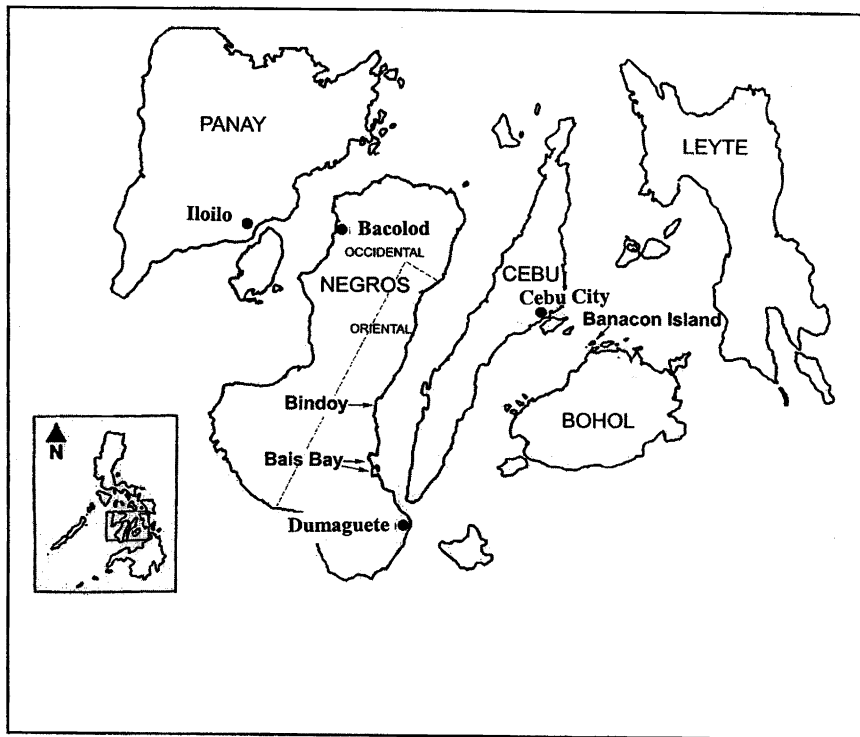


Fig. 1. Location of study sites.

the seaward perimeter of a private estate. These mangroves have been protected by the estate's owner and so are little disturbed by cutting or fishpond development. In contrast to Bais Bay, Bindoy includes extensive mature, interior (landward side) stands of mangrove that are less-frequently flooded by tides.

Banacon is a small, coralline island, located 5 km off the northwest corner of Bohol Province and about 30 km east of Cebu City (Cabahug et al., 1986). Over 95% of Banacon's roughly 500 ha size is mangrove forest. There are currently 550 households crowded onto a 15-ha dryland area on the eastern tip of the island. Virtually all of these households derive their principal income from fishing and related activities (e.g., fish processing, marketing, etc.). Residents have always depended on the harvest of mangrove wood to meet most domestic fuel and construction needs. As well, since the late 1950s they have planted trees so that vast expanses of formerly natural forest are dominated today by monocultures of relatively young planted *Rhizophora stylosa* (Walters, 2000a, 2004). Almost all mangrove wood being cut today on

Banacon comes from the nearly 400 ha of plantations. The mangroves of Banacon received national Wilderness Area designation in 1981 (DENR, 1990), but this has not effectively precluded local people from continuing to plant and cut mangroves there.

To assess forest characteristics, I employed the quadrat/census plot method (Cintron and Schaeffer Novelli, 1984; Peters, 1996b). Each census plot was 10 m × 10 m, with corners and boundaries marked using a 50-m measuring tape. A relatively small plot size was used because trees in most of the stands surveyed were typically small and densely crowded as a result of their being young or having been highly disturbed from cutting. A stratified random sampling approach was used to select plot sites. Approximately equal numbers of plots in both cut and uncut forest stands from each forest type were sampled to evaluate the effects of cutting. I located plots widely in the study areas in an attempt to capture some of the variation due to site-specific differences in ecological conditions and human influences. Slightly greater sampling effort was specifically devoted to plantations

to ensure representation from a wide range of stand ages (Walters, 2000a).

In summary, I surveyed 52 plots: 31 that had been cut (10 in natural forest and 21 in plantations) and 21 that showed little or no evidence of cutting (9 natural and 12 plantations). Unfortunately, cutting was found to be so ubiquitous in the forests under study that it was virtually impossible to find sites with no evidence of past cutting. Nonetheless, plots listed under the “not cut” category were restricted to sites where evidence of cutting was minimal. For natural forests, 13 plots were surveyed across 3 distinct sites in North and South Bais Bay (Dungaun, Dyke and Dauis), and 6 plots were surveyed in Bindoy. For plantations, I surveyed one plot in 5 different sites on Banacon Island, and 28 plots in 24 different plantations across 8 distinct sites in North and South Bais Bay. Plantations from 5 to 60 years of age were surveyed (mean = 30.3 years). Topographic height and salinity were measured at the center of each plot.

To classify forest canopy structure, I walked transects back and forth over the plot, observing the canopy vertically above me every meter so that 100 observations were made across the area of the plot (i.e., every square meter). These observations were subsequently summarized and converted directly to plot percentages. Canopy structure was classified as either “gap”, “expanded gap”, or “understory” following the criteria employed by Runkle (1982) and Lertzman et al. (1996). In particular, expanded gap was the vertical projection to the ground of the boles of the trees defining the boundaries of a canopy gap. It falls vertically beneath canopy foliage, but is nonetheless directly affected by the gap by virtue of its immediate proximity to it.

I numbered, mapped and measured every tree (>1.0 m tall) and seedling (<1.0 m) in each of the 52 forest plots, for a total of 5926 trees and 1999 seedlings. Each tree was identified by species (based on Calumpang and Menez, 1997) and recorded as either “canopy”, “gap-filler”, “gap”, “expanded gap”, or “understory” (after Lertzman et al., 1996). Dead stems >1.0 m tall were classified as “snags”. Evidence of reproduction (i.e., flowers or seeds) was noted, although I found with practice that only *Rhizophora* spp. could be assessed reliably because the seeds and flowers of other species were difficult to observe from the forest floor. I measured the diameter

at breast height (dbh) of each tree stem following the guidelines of Cintron and Schaeffer Novelli (1984). Tree height and height of lowest live branch was measured using a tall bamboo pole marked off at meter intervals. I then recorded whether the tree stem was the original trunk, or whether it appeared to be a secondary stem originating as a sprout from a larger trunk. Finally, I documented evidence of cutting by recording for each tree whether it was a cut stump or had either a cut branch or cut root. Where evidence of cutting was not obvious, I assumed it was not cut. This probably meant that measures of cutting were underestimated since evidence of cutting on older, decayed stumps may well have eroded. Quantitative data were analyzed statistically using SPSS (version 9.0). Plot data were log-transformed for statistical analysis when they did not meet the test for homogeneity of variances (Zar, 1984).

Ethnographic information was obtained primarily from semi-structured interviews that I conducted with 202 residents living in 10 different coastal villages in North and South Bais Bay, 10 residents of Banacon Island and 3 residents of Bindoy. In these interviews, people were asked questions about their knowledge of mangroves, use of mangrove wood and cutting practices, and so on. Interviews were recorded in the field and later transcribed and the texts analyzed.

3. Results

Variations in measured salinity between sampled plots were small, typically ranging between 30 and 35 ppt, with the exception of several plots located near the mouths of rivers, where salinity was measured as low as 16 ppt. However, there was overall no statistical difference in mean salinity (ppt) between cut forest plots ($\chi^2 = 32.2$, S.D. = 4.45, $n = 30$) and uncut forest plots ($\chi^2 = 33.3$, S.D. = 3.28, $n = 21$). Likewise, mean topographic height (cm) did not differ between cut forest ($\chi^2 = 79.5$, S.D. = 31.1, $n = 31$) and uncut forest ($\chi^2 = 78.0$, S.D. = 27.89, $n = 21$).

3.1. Effects of cutting on mangrove forest structure

Cut mangrove forests differed structurally in a number of ways from uncut forests (Table 1). The density of live trees, canopy trees only, and snags was

Table 1

Summary of select ecological characteristics of mangrove forest census plots, comparing mean values (and standard deviations) between uncut and cut forests (plantation and natural combined)

Characteristics	Uncut forest plots ($n = 21$)	Cut forest plots ($n = 31$)	F -values (d.f. = 1,51)
Number of tree species	2.5 (1.4)	2.4 (1.3)	0.02
Live stem density (100 m^{-2})	104.9 (114.0)	93.8 (75.1)	0.18
DBH live stems (cm)	8.2 (6.6)	5.1 (2.8)	5.42*
Canopy tree density (100 m^{-2})	48.8 (59.4)	34.5 (31.2)	0.03
DBH canopy trees (cm)	14.4 (13.5)	8.9 (4.5)	3.33
Live stem basal area (m^2/ha)	36.0 (25.6)	19.2 (9.4)	11.10***
Snag density (100 m^{-2})	4.4 (6.8)	2.8 (3.8)	1.11
DBH snags (cm)	1.5 (0.6)	1.3 (0.5)	1.55
Seedling density (100 m^{-2})	34.0 (42.3)	42.4 (39.4)	0.55
Canopy tree height (m)	9.6 (2.5)	7.5 (1.9)	11.76***
Lowest branch height (m)	2.8 (1.2)	2.0 (1.0)	7.50**
Canopy gap (%)	4.4 (4.9)	12.5 (7.9)	19.39****
Expanded gap (%)	28.0 (25.1)	40.0 (23.4)	3.09
Closed Canopy (%)	67.6 (26.6)	47.6 (28.3)	6.56*
Canopy gap density (100 m^{-2})	2.4 (2.1)	5.7 (3.6)	14.59****
Mean gap size (m^2)	2.0 (2.4)	2.7 (1.4)	1.3

* $P < 0.05$.

** $P < 0.01$.

*** $P < 0.005$.

**** $P < 0.001$.

each lower in cut forests compared to uncut forests, but not significantly. The mean diameter of live trees in cut forests was significantly less than trees in uncut forests. The combined effect of lower tree densities and smaller tree sizes translated into basal areas in cut forests that were, on average, only half as large as in uncut forests. Canopy trees and the lowest live branches were both significantly shorter in height in cut compared to uncut forests. Compared to uncut forests, cut forests were characterized by three times as much canopy gap and almost one-third less closed canopy (Table 1). The mean size of canopy gaps is not significantly different between cut and uncut forests, but the density of such gaps was more than twice as high in cut forests. These data suggest that cutting substantially altered the canopy by creating more, but

not necessarily larger gaps. These differences typically hold if one examines plantations and natural forests separately, with the exception that canopy gaps in plantations are significantly larger in cut forests compared to uncut forests (2.5 m^2 versus 1.1 m^2 , $P < 0.05$, $f = 6.98$, d.f. = 1.32), whereas they were identical in size in cut and uncut natural forests (3.2 m^2 in each).

The gap data discussed thus far are based on survey plot means (Table 1). Data on the size and origins of individual canopy gaps are summarized and presented in Table 2. These show that the density of gaps was higher. Two-thirds of canopy gaps in both natural forests and plantations were caused by cutting. In fact, this is likely a conservative estimate since some of these gaps could have been caused by cutting or some

Table 2

Canopy gaps and their causes, comparing total numbers measured in natural and plantation forest plots

	Natural forest	Plantation forest	Total
Number of canopy gaps	56	172	228
Canopy gap density (100 m^{-2})	3	5.2	4.4
Mean gap sizes (m^2)	2.6 (0.5–20.0)	2.1 (0.5–20.0)	2.2 (0.5–20.0)
Human cause (%)	67.9	64.5	65.4
Natural cause (%)	19.6	10.5	12.7
Unknown cause (%)	12.5	25	21.9

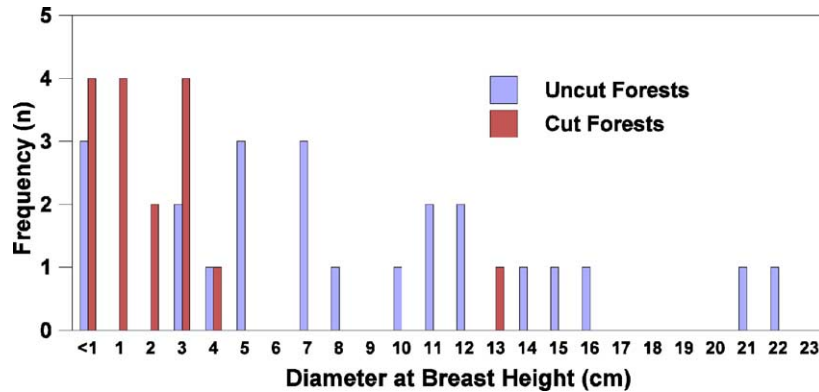


Fig. 2. Size-frequency distribution (dbh) of live *R. mucronata* trees in uncut and cut natural forests.

other form of human disturbance (e.g., deliberate stem breakage without use of a cutlass), even though evidence is lacking.

3.2. Effects of cutting on forest species composition

Size-frequency distributions of common mangrove species in cut and uncut forests are presented in Figs. 2–5. Natural forests only are considered in this particular analysis to eliminate potentially confounding effects of planting. Both *R. mucronata* and *R. apiculata* showed a higher concentration of stems in small size classes in the cut versus uncut forest. Compared to *R. apiculata*, *R. mucronata* was uncommon in the cut forest and, in particular, almost completely absent from all but the smallest (<5 cm) size classes (Figs. 2 and 3). *A. marina* in cut forest is

more abundant in the medium size classes (3–8.9 cm) compared to greater abundance of small and large stems in uncut forests (Fig. 4). The case of *Sonneratia* was similar to *Avicennia* in that the effect of cutting appeared to be a relative redistribution of trees from the outer extremes towards the mid-size classes (Fig. 5). In particular, cut forests had few very large stems, whereas large stems were common in uncut forest.

Where demographic data are presented in terms of cover type, the most striking trend across cut and uncut, and natural and plantation forests was the relative compositional consistency across canopy status classes, with some exceptions. *C. decandra* in uncut forest showed little difference between canopy status classes (Table 3). *R. mucronata* in natural forests were proportionately more abundant in the canopy and in canopy gaps in uncut compared to cut forests,

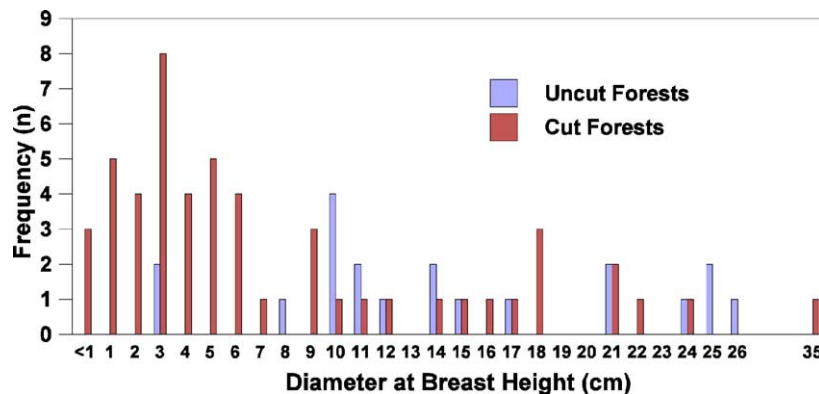


Fig. 3. Size-frequency distribution (dbh) of live *R. apiculata* trees in uncut and cut natural forests.

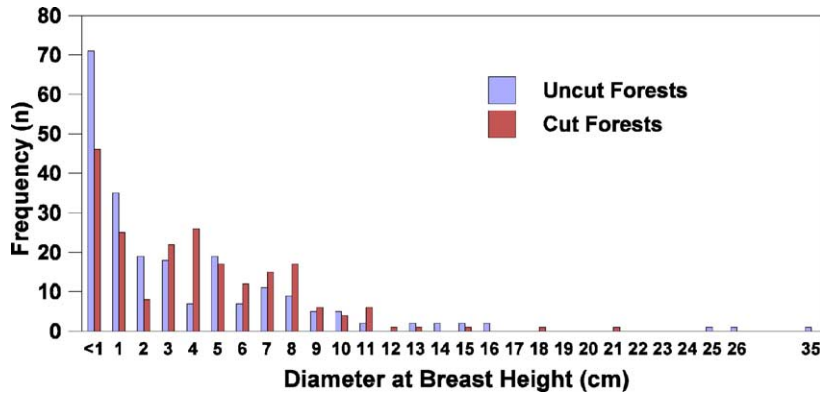


Fig. 4. Size-frequency distribution (dbh) of live *Avicennia* trees in uncut and cut natural forests.

whereas the reverse was observed for *R. apiculata* (Table 3). *A. marina* were abundant in all canopy status classes in both cut and uncut natural forests, but particularly so in understory and expanded gap. *Sonneratia* spp. were similarly common in most canopy status classes in cut and uncut natural forests, with the notable exception of understory where it was nearly absent (Table 3). Uncut natural forests had greater species richness than cut natural forests, with individuals of *O. octodonta*, *B. gymnorhiza*, and *X. graniatum* present in both canopy gaps and understory, albeit in numbers too few to merit comparison.

In plantations, *R. mucronata* dominated all canopy status classes in both cut and uncut forests (Table 4). *A. marina* was proportionately more abundant in most canopy status classes in cut plantations compared to uncut plantations, although especially in understory.

3.3. Effects of cutting on mangrove forest regeneration

Data on seedling abundance are presented by forest type and species in Table 5. Significant differences between cut and uncut forest were not found as sampling variation between plots was very large. Therefore, I examined the relationship between seedlings and canopy gaps as an alternative way to estimate the effects of cutting on mangroves regeneration; the rationale for this being that cutting of mangroves resulted in a substantial increase in the number of canopy gaps (Tables 1 and 2) and it is widely believed that gaps influence forest regeneration. To separate out the confounding effects of planting, the data were analyzed by different forest type (Table 6). Abundance measures were analyzed

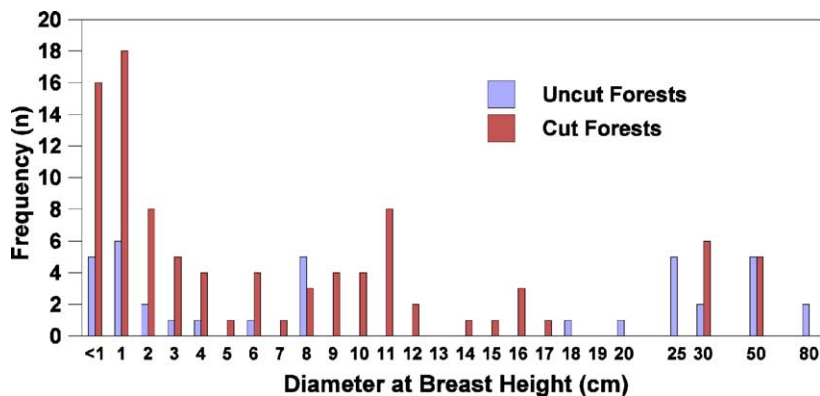


Fig. 5. Size-frequency distribution (dbh) of live *Sonneratia* trees in uncut and cut natural forests.

Table 3

Species composition (% abundance) of mangrove forests by canopy status in uncut and cut natural forests (n = number of trees in sample)

Tree species	Canopy status						Total
	Canopy	Gap-filler	Gap	Expanded gap	Understory	Stump	
Uncut forest (n)	140	29	13	57	179	46	464
<i>R. mucronata</i>	8.6	6.9	15.4	3.5	2.8	2.2	5.2
<i>R. apiculata</i>	7.9	14	0	7	0.6	2.2	4.5
<i>A. marina</i>	35.7	34.4	61.5	57.9	65.4	47.8	51.7
<i>Sonneratia</i> spp.	15	0	7.7	14	3.9	32.6	11.2
<i>C. decandra</i>	12.1	34.4	15.4	10.5	14.5	15.2	14.7
<i>B. gymnorhiza</i>	1.4	0	0	0	2.8	0	1.5
<i>O. octodonta</i>	17.1	10.3	0	5.3	8.4	0	9.7
<i>X. granatum</i>	2.1	0	0	0	1.1	0	1.1
<i>A. officinalis</i>	0	0	0	1.8	0.6	0	0.4
Sub-total	99.9	100	100.1	100	100.1	100	100
Cut forest (n)	131	57	27	80	86	307	688
<i>R. mucronata</i>	1.5	1.8	3.7	6.3	8.1	0.7	2.6
<i>R. apiculata</i>	17.6	21.1	7.4	10	7	10.1	11.9
<i>A. marina</i>	54.2	47.4	29.6	55	76.7	31.3	45.3
<i>Sonneratia</i> spp.	26	29.8	59.2	28.8	8.1	57.6	39.8
<i>A. lanata</i>	0.8	0	0	0	0	0.3	0.3
Sub-total	100.1	100.1	99.9	100.1	99.9	100	99.9

using chi-square to test whether the observed abundance of seedlings found beneath each canopy cover type was expected based on the relative abundance of canopy cover types.

R. mucronata seedlings were significantly more abundant in understory than in canopy gap or expanded gap in natural forests, but the reverse held for plantations (Table 6). *R. apiculata* was more

Table 4

Species composition (% abundance) of mangrove forests by canopy status in uncut and cut plantations (n = number of trees in sample)

Tree species	Canopy Status						Total
	Canopy	Gap-filler	Gap	Expanded gap	Understory	Stump	
Uncut forest (n)	885	116	23	190	570	43	1827
<i>R. mucronata</i>	98.3	99.2	87	85.3	95.8	69.8	95.4
<i>R. apiculata</i>	0.8	0	0	1.1	0.7	2.3	0.8
<i>A. marina</i>	0.8	0.8	13	13.7	2.5	25.6	3.4
<i>Sonneratia</i> spp.	0.1	0	0	0	0.7	2.3	0.4
<i>A. lanata</i>	0	0	0	0	0.4	0	0.1
Sub-total	100	100	100	100.1	100.1	100	100.1
Cut forest (n)	938	286	217	587	498	421	2947
<i>R. mucronata</i>	88.7	96.2	91.7	86.7	62.9	87.2	84.7
<i>R. apiculata</i>	0.6	0	0	0	0.8	0.5	0.4
<i>A. marina</i>	9.3	3.1	8.3	11.2	32.5	10	13
<i>Sonneratia</i> spp.	0.5	0	0	0.2	0	0.5	0.2
<i>C. decandra</i>	0	0	0	1.2	1.4	1.7	0.7
<i>B. cylindrica</i>	0.9	0.7	0	0.5	2.4	0.2	0.9
<i>N. fruticans</i>	0	0	0	0.2	0	0	0
Sub-total	100	100	100	100	100	100.1	99.9

Table 5
Seedling density (100 m^{-2}) in mangrove forest census plots, comparing mean values (and standard deviations) between uncut and cut forests

	Uncut forest plots	Cut forest plots	F-values
Natural forest plots (<i>n</i>)	<i>n</i> = 9	<i>n</i> = 10	d.f. = 1,18
<i>Rhizophora</i> spp.	16.6 (31.0)	3.8 (4.6)	0.39
<i>A. marina</i>	12.6 (23.8)	21.3 (30.9)	0.5
<i>Sonneratia</i> spp.	0.8 (1.7)	16.8 (32.2)	0.33
All seedlings	47.4 (50.8)	42.0 (40.9)	0.8
Plantation plots	<i>n</i> = 12	<i>n</i> = 21	d.f. = 1,32
<i>Rhizophora</i> spp.	14.8 (19.2)	26.2 (39.6)	0.36
<i>A. marina</i>	4.0 (11.4)	12.3 (20.2)	0.2
<i>Sonneratia</i> spp.	0	0	–
All seedlings	23.8 (33.5)	42.6 (39.7)	0.18
All forests plots	<i>n</i> = 21	<i>n</i> = 31	d.f. = 1,51
<i>Rhizophora</i> spp.	15.6 (24.3)	19.0 (34.1)	0.7
<i>A. marina</i>	7.7 (17.8)	15.2 (24.0)	0.23
<i>Sonneratia</i> spp.	0.3 (1.2)	5.4 (19.4)	0.33
All seedlings	34.0 (42.3)	42.4 (39.4)	0.46

abundant in gap and expanded gap in natural forest, although sample sizes were small and the difference only significant if data from all sites were combined ($\chi^2 = 24.2$, $P < 0.001$, $n = 32$). *Sonneratia* seedlings were also more abundant in gaps and expanded gaps

compared to understory in natural forests. By contrast, *A. marina* was more abundant in understory in both natural and plantation forests. *C. decandra* was significantly more abundant in understory in natural forests, but this pattern did not hold in plantations where it was more abundant in expanded gap (Table 6).

I also compared seedling height by canopy cover type with the assumption that it serves as a proxy for growth (Table 7). Unlike abundance, which showed highly varied responses to canopy cover depending on the species, seedling size showed the following, fairly consistent and significant pattern across species: gap > expanded gap > understory. The one exception to this was *A. marina*, for which seedling heights followed the pattern: gap > understory > expanded gap (Table 7).

Data on reproductive rates for *R. maniculata* and *R. apiculata* in different canopy status classes are summarized in Table 8. For both species, reproductive rates were highest in the canopy and declined as one moved from gaps to understory, at which stage reproduction was negligible. These patterns held between cut and uncut forests.

Finally, it became apparent during the study that much of the regeneration in mangroves results, not

Table 6
Seedling abundance of different mangrove species by canopy cover type in natural and plantation forests

Species	Seedling abundance					
	Natural forests			Plantation forests		
	Canopy gap (6.9%)	Expanded gap (43.3%)	Closed canopy (49.8%)	Canopy gap (10.6%)	Expanded gap (30.5%)	Closed canopy (59.0%)
<i>R. mucronata</i>	5	40	111 ^{***} ($\chi^2 = 28.0$)	91	406	231 ^{***} ($\chi^2 = 247.6$)
<i>R. apiculata</i>	3	19	10 ($\chi^2 = 4.34$)	0	0	0
<i>A. marina</i>	17	119	190 ^{**} ($\chi^2 = 9.0$)	15	78	213 ^{***} ($\chi^2 = 17.1$)
<i>Sonneratia</i> spp.	24	147	4 ^{***} ($\chi^2 = 157.2$)	0	0	0
<i>C. decandra</i>	0	9	118 ^{***} ($\chi^2 = 95.3$)	3	18	24 ($\chi^2 = 2.3$)
<i>Bruguiera</i> spp.	3	0	1	1	7	34
<i>X. granatum</i>	0	0	3	0	0	0
<i>O. octodonta</i>	1	0	20	0	0	0
<i>A. lanata</i>	1	1	0	0	0	0
<i>A. officinalis</i>	0	0	1	0	0	0
Total (all species)	54	335	458 [*] ($\chi^2 = 6.1$)	110	509	502 ^{***} ($\chi^2 = 51.7$)

Statistical tests done using χ^2 comparing observed with expected abundance of individual cells based on the actual percentages of forest in each canopy cover type (presented in parentheses).

* $P < 0.05$.

** $P < 0.01$.

*** $P < 0.001$.

Table 7

Seedling sizes (cm) for different mangrove species compared by canopy cover type in natural and plantation forests combined (standard deviations and sample sizes listed in parentheses)

Species	Canopy status			F-values
	Canopy gap	Expanded gap	Understory	
<i>R. mucronata</i>	57.7 (24.2) (n = 96)	56.8 (23.2) (n = 446)	48.0 (20.1) (n = 342)	14.45 ^{***} (d.f. = 2, 881)
<i>R. apiculata</i>	–	81.6 (18.0) (n = 19)	64.0 (20.1) (n = 10)	5.76 [*] (d.f. = 1, 27)
<i>A. marina</i>	65.0 (30.3) (n = 32)	55.7 (30.5) (n = 197)	63.5 (29.0) (n = 403)	4.89 ^{**} (d.f. = 2, 629)
<i>Sonneratia</i> spp.	87.1 (18.1) (n = 24)	59.7 (29.6) (n = 147)	–	19.23 ^{***} (d.f. = 1, 169)
<i>C. decandra</i>	–	74.8 (19.7) (n = 27)	50.1 (26.9) (n = 142)	20.71 ^{***} (d.f. = 1, 167)
Grand mean	62.0 (27.7) (n = 171) ^a	57.4 (26.9) (n = 855) ^a	53.1 (27.4) (n = 973) ^a	10.75 ^{***} (d.f. = 2, 1996)

^a Sample sizes in the bottom row exceed the sum of the column because less common species were not listed separately.

^{*} $P < 0.05$.

^{**} $P < 0.01$.

^{***} $P < 0.001$.

Table 8

Percentage of *R. mucronata* and *R. apiculata* by canopy status showing evidence of reproduction (sample size in parentheses)

Canopy status	Percentage (%) reproductive	
	<i>R. mucronata</i>	<i>R. apiculata</i>
Canopy	82.7 (1710)	93.5 (46)
Canopy gap-filler	21.7 (392)	68.8 (16)
Canopy gap	3.2 (222)	50.0 (2)
Expanded gap	3.4 (678)	0.0 (14)
Understory	0.5 (871)	0.0 (15)
Grand mean	40.9 (3873)	43.0 (128)

Data from cut and uncut forests are combined.

from seeds, but from stump and stem sprouting. While this distinction was made when I measured stems (>1.0 m), I did not record whether seedlings originated from seeds or stem/root sprouts. Nonetheless, I am confident based on personal observation

that virtually all of the *Rhizophora*, *Bruguiera*, and *Ceriops* seedlings I measured were from seeds, whereas most of the *Avicennia* and *Sonneratia* seedlings sprouted from stumps. As for stems, *Sonneratia* spp. were found to have the highest overall rate of coppicing (81.5%), followed by *R. apiculata* (56.6%), *A. marina* (37.6%), *C. decandra* (33.3%), *Bruguiera* spp. (33.3%), and finally *R. mucronata* (2.8%). Overall coppicing rates were also much higher in natural forests (56.1%) compared to plantations (5.6%) (Table 9).

Table 10 shows that the proportion of total basal area resulting from coppice growth was higher in cut forests compared to uncut forests (30.9 and 24.7%, respectively). For individual species, the proportion of coppice in cut and uncut was as follows: 35.2% versus 24.6% for *R. apiculata*, 22.8% versus 10.5% for *A. marina* and 33.3% versus 30.4% for *Sonneratia*. The

Table 9

Cutting and coppicing rates by species for natural and plantation mangrove forests

Species	Natural forests		Plantation forests		Combined	
	Cut ^a (%)	Coppice (%)	Cut (%)	Coppice (%)	Cut (%)	Coppice (%)
<i>R. mucronata</i>	30.9 (42)	11.9 (42)	24.6 (4238)	2.8 (4238)	24.7 (4280)	2.8 (4280)
<i>R. apiculata</i>	44.7 (103)	70.0 (103)	19.2 (26)	15.4 (26)	39.5 (129)	56.6 (129)
<i>A. marina</i>	37.3 (552)	44.2 (552)	22.2 (446)	29.4 (446)	30.6 (998)	37.6 (998)
<i>Sonneratia</i> spp.	72.4 (326)	82.8 (326)	50.0 (14)	50.0 (14)	71.4 (340)	81.5 (340)
<i>C. decandra</i>	13.2 (68)	41.2 (68)	38.1 (21)	9.5 (21)	19.1 (89)	33.7 (89)
<i>Bruguiera</i> spp.	14.3 (7)	0.0 (7)	7.7 (26)	42.3 (26)	36.4 (33)	33.3 (33)
Grand mean	46.5 (1098)	56.1 (1098)	24.4 (4771)	5.6 (4771)	28.5 (5869)	15.1 (5869)

Figures represent percentages of the total number of stems (sample sizes in parentheses).

^a Stems qualify as having been “cut” if either their main stem or at least one significant branch has been cut.

Table 10
Total basal area (m²/ha) of original and coppice stems for different species in uncut and cut natural forests

Species	Basal area (m ² /ha)			
	Uncut forest		Cut forest	
	Original stems	Coppice stems	Original stems	Coppice stems
<i>R. mucronata</i>	2	0.41	0.21	0
<i>R. apiculata</i>	3.53	1.15	2.98	1.62
<i>A. marina</i>	6.39	0.75	4.21	1.24
<i>Sonneratia</i> spp.	17.6	7.68	8.92	4.45
<i>C. decandra</i>	0.73	0.2	0	0
<i>B. gymnorhiza</i>	0.37	0	0	0
<i>O. octodonta</i>	1.57	0.38	0	0
Total	32.19	10.57	16.32	7.31

one exception to this pattern is *R. mucronata* which had no basal area from coppice in the cut forest and 17.0% in uncut forest.

4. Discussion

4.1. Effects of cutting on forest structure

Few studies have examined the ecological effects of selective cutting of mangroves. Nurkin (1994, p. 273) suggests that small-scale harvesting of mangroves in Sulawesi has “negligible” impacts, but others indicate otherwise. Eusebio et al. (1986) found that mangroves in the Philippines subject to cutting were “stunted” and “shrubby” in growth and had only 5–30% of the density of medium- (5–15 cm dbh) and large-size (>15 cm) trees compared to uncut forests (p. 346). Smith and Berkes (1993) similarly found in a Caribbean mangrove that cutting eliminated large trees but greatly increased the density of small stems by stimulating stump sprouting of *Laguncularia racemosa*. They recorded a mean stem density of 7000 ha⁻¹ and basal area of only 4.8 m²/ha in areas with a prior history of repeated, selective cutting.

People living along the coastline in Bais Bay and on Banacon Island have cut mangroves for a variety of reasons, but mostly for fuelwood and posts used for construction of fish corrals (Walters, 2005). Consumption of these wood products was large, and evidence of cutting impacts pervasive in many if not most mangrove stands. Cut natural and plantation forests did not significantly differ in terms of stem density, but cut forests had half the basal area of uncut

forests (Table 1). Cut forests were also characterized by a greater proportion of individual trees in smaller size classes (Figs. 2–5). These data contrast strikingly with data collected earlier this century in the Philippines on undisturbed, “virgin” mangroves, where surveys revealed abundant *Rhizophora* and *Bruguiera* trees 30–60 cm dbh and *Sonneratia* trees up to 100 cm (Brown and Fischer, 1918).

Cutting also increased canopy openness. The amount of canopy gap in uncut mangrove forest in this study was 4.4% (Table 1). This is comparable to the lower measures made in studies of Australian and Indonesian mangroves, which ranged from 4 to 15% (Atmadja and Soerojo, 1991; Smith, 1992). A study of 7-year-old *R. mucronata* plantations in Java likewise found approximately 10% of the forest canopy open (Sukardjo and Yamada, 1992). Studies evaluating the effects of selective logging on canopy structure in various tropical forests have found it to increase canopy openness, roughly in proportion to cutting intensity (Uhl et al., 1991; Ter Steege et al., 1994; Chapman and Chapman, 1997). In this study, cutting increased the amount of mangrove forest in canopy gap three-fold (Table 1). This is consistent with estimates that cutting caused two-thirds or more of all canopy gaps (Table 2) and, more generally, the finding that cutting was the probable cause of 90% of stem mortality in the study areas (Walters, 2005). Although not systematically studied, it would appear that the most common natural causes of canopy gaps in the study areas were wind throw and stem breakage during storm events (pers. observ.).

People harvesting wood in the study areas were typically seeking relatively small-diameter tree stems

and branches used for fish corral posts and cooking fuel (Walters, 2005). Not surprisingly, the canopy gaps created by such wood harvesting were typically small. The largest canopy gaps measured in this study were 20 m², but mean gap size was only 2.6 m² in natural and 2.1 m² in plantation forests, which is small compared to findings from other mangroves. For example, forest gaps in mature, northern Queensland mangroves typically measured 40–120 m² (Smith, 1992) and mean size of gaps in mangroves in Kosrae, Micronesia, were 158 m² (Ewel et al., 1998), although smaller gaps (<10 m²) were deliberately not measured in the latter study and quite possibly overlooked in the former. Pinzon et al. (2003) likewise found that gaps caused by logging in Kosrae were twice as large as those caused naturally (114 m² versus 64 m²), although these authors also did not consider gaps <10 m². By contrast, Feller and McKee (1999) found relatively small canopy gaps dominated *R. mangle* stands in Belize and studies of other forest types have shown that small canopy gaps can have important effects (c.f., Brown and Whitmore, 1992; Kennedy and Swaine, 1992). Such effects may be pronounced in forests with low canopies, as in this study (4–10 m), because proportionately greater sunlight would be expected to reach the forest floor.

4.2. Effects of cutting on forest species composition

Studies of mangroves subject to cutting differ in the extent to which they have detected apparent compositional changes. Eusebio et al. (1986) suggested that *C. tagal* is disproportionately affected by cutting, both because it is more heavily cut and because its regeneration following cutting is relatively poor compared to other species. Both Putz and Chan (1986) and Pinzon et al. (2003) found evidence suggesting that *Bruguiera* spp. increase relative to *R. apiculata* in forests that are subject to logging. By contrast, Ewel et al. (1998) claim that selective cutting of *Rhizophora* and *Bruguiera* in Kosrae, Micronesia, is compensated for by good regeneration of both species and so anticipate little compositional change.

In my study areas, cutting for fuelwood was not species-selective, but preference was given to *Rhizophora* spp. when harvesting for construction wood (Walters, 2005). Nonetheless, I found only limited

evidence to suggest that cutting has changed forest composition. Among common species, *R. mucronata* appeared most negatively impacted by cutting, as evidenced by the fact that its presence in cut natural forest was restricted to very few of the smallest size classes (Fig. 2) and included almost no individuals in the canopy layer (Table 3). This is important because *R. mucronata* was found to be reproductive at small sizes – as small as 0.4 cm dbh and commonly at 1.5 cm dbh and larger – but only if growing in the canopy (Table 8). Poor regeneration in the face of cutting pressure appeared the likely explanation for these demographic patterns (see below).

Evidence suggested that the relative abundance of the other common species, *R. apiculata*, *A. marina* and *Sonneratia* spp., was not decreased by cutting, and regeneration success may increase because of it. Cutting reduced the number of large stems of these species, but each compensated with good regeneration (Figs. 3–5) and each maintained large numbers in the canopy in spite of cutting (Table 3). In particular, *A. marina* had a great abundance of individuals in both the canopy and in gaps in the cut forest, virtually assuring its long-term success in these sites should cutting continue.

I did not sample enough sites where rarer species were present to evaluate the effects of cutting on them. It is possible that the presence of *C. decandra*, *O. octodonta* and *B. gymnorhiza* in a few uncut plots is significant, but most of these and other rarer species typically thrived only in the upper elevation zones of mangroves (Calumpang and Menez, 1997) and these sites have been mostly cleared to make space for fish ponds in Bais Bay (Walters, 2003).

4.3. Effects of cutting on forest regeneration

Mangroves usually recover quickly from disturbance that causes structural damage, including hurricanes and tree cutting (Roth, 1992; Snedaker et al., 1992; Smith and Berkes, 1993; Smith et al., 1994; Wadsworth, 1997). In spite of abundant research, there is little consensus as to factors which influence mangrove seedling establishment (Rabinowitz, 1978; Smith, 1992). In this study, seedling abundance in plots varied so much that differences between cut and uncut forest plots were statistically invisible. Still, patterns of seedling abundance and size

in relation to canopy gaps are suggestive of a relationship between cutting and regeneration.

Levels of photosynthetically active radiation and ground-level temperatures in mangroves are substantially higher beneath canopy gaps compared to neighboring understory (Smith, 1987; Ewel et al., 1998), and these differences increase with gap size. Some have suggested that, since all mangrove species are relatively shade-intolerant, they are likely to depend on the availability of openings in the canopy to grow (Snedaker and Lahmann, 1988; Roth, 1992; Smith, 1992). Several studies have found that seedlings regenerate quickly and in large numbers in canopy openings created by natural disturbance or cutting (Eusebio et al., 1986; Smith, 1987, 1992; Abdulhadi and Suhardjono, 1994; Nurkin, 1994; Ewel et al., 1998). Compared to mangrove seedlings in understory, seedlings in gaps have been found to suffer less predation and herbivory (Osborne and Smith, 1990; Ellison and Fairnsworth, 1993), experience greater survival and grow faster (Ellison and Fairnsworth, 1993; Koch, 1997; Smith, 1987; Feller and McKee, 1999). At the same time, differences due to canopy structure can be “swamped” by other physical factors (Smith, 1987; Smith et al., 1994; Ewel et al., 1998).

Evidence that mangrove species differ in their response to disturbance that creates canopy openings is mixed. Roth (1992) observed high seedling regeneration among all species following catastrophic mortality in Nicaragua. Ewel et al. (1998) similarly detected no differences in gap regeneration of *B. gymnorrhiza* and *R. apiculata* following selective logging, and Clarke and Kerrigan (2000) found little evidence for gap-dependant specialization among mangroves species in Australian forests. By contrast, Pinzon et al. (2003) found greater *B. gymnorrhiza* seedling densities in forest gaps in Kosrae. Tsuda et al. (1995), also working in Micronesia, found that *B. gymnorrhiza* seedlings dominated small gaps whereas equal numbers of *B. gymnorrhiza* and *R. apiculata* were found in large gaps. In Australia, Smith (1987) recorded differences in relative growth rates between species, with *R. stylosa* growing the fastest in gaps and understory. Likewise, *R. apiculata* seedlings were found to dominate canopy openings in Kalimantan mangroves (Abdulhadi and Suhardjono, 1994) and *Rhizophora* seedlings established in greater numbers

than *C. tagal* in Philippine forests where both had been cut (Eusebio et al., 1986).

Data from this study suggest that canopy gaps did not have a strong influence on the total abundance of mangrove seedlings (Table 5). *R. apiculata* were more abundant in gaps and expanded gap, but findings for *R. mucronata* are equivocal: seedlings were more abundant in understory in natural forests, but more abundant in gaps in planted forests (Table 6). The latter were obviously confounded by the effects of planting. However, seedlings of both *Rhizophora* species were taller in gap and expanded gap compared to understory (Table 7). Other researches on *Rhizophora* spp. have observed strong recruitment and/or growth in gaps (Smith, 1987; Ellison and Fairnsworth, 1993; Abdulhadi and Suhardjono, 1994; Koch, 1997; Feller and McKee, 1999; Pinzon et al., 2003). Lab studies found *R. mangle* not to be sensitive to subtle changes in light levels (McKee, 1995), suggesting this species requires sizable canopy openings to elicit strong recruitment and growth responses (see Koch, 1997). The fact that gap sizes in this study were so small may help explain the absence of an abundance response to gaps in *R. mucronata*. In fact, it is common silvicultural knowledge that *Rhizophora* regenerates particularly well in highly open areas and so is well suited for management systems based on clear-felling (Hussain, 1995).

Few studies have examined the influence of canopy structure on regeneration of *Sonneratia* spp. *S. alba* seedlings were equally abundant in gaps and understory in Kosrae mangroves, although sample sizes were small (Pinzon et al., 2003). In this study, *Sonneratia* seedlings showed the greatest abundance and height response to gaps (Tables 6 and 7), suggesting this species is especially light-demanding. Particularly striking in comparison with the other species was the absence of *Sonneratia* seedlings in understory and the large difference in size between gap and expanded gap. In the study areas, *Sonneratia* commonly grew on the seaward margins of forests where the canopy is invariably more open (personal observation). *C. decandra* seedlings were notably more abundant in the understory (Table 6), even though they were taller in expanded gap (Table 7). *C. decandra* has been little studied, but *C. tagal* was found to survive and grow better in gaps compared to understory in Australia (Smith, 1987) and regenerated

well in openings following cutting in the Philippines (Eusebio et al., 1986).

Contrary to Smith (1992), *A. marina* demonstrated a notable affinity to understory conditions in this study: seedlings were both significantly more abundant in the understory and identical in size between understory and gaps (Table 7). *A. marina* trees were also proportionately more abundant in understory in both natural and plantation forests (Tables 3–4). Clarke and Allaway (1993) found no differences in seedling establishment and survival of this species between gaps and non-gaps, whereas Minchinton (2001) found that seedlings of *A. marina* were more abundant in the understory, while saplings were more abundant in gaps. This suggests that gaps may be of little importance for seedling establishment, but of increasing importance as the seedling grows (Clarke and Allaway, 1993; Pinzon et al., 2003). Snedaker and Lahmann (1988) citing Ball and Critchley (1982) have noted among mangroves the exceptional ability of *A. marina* seedlings to persist longer in the understory. Such ability to persist may help explain the relative success of this species in this study because existing seedlings may have the critical “head start” needed to better exploit the small gaps created by cutting (c.f., Putz and Chan, 1986; Brown and Whitmore, 1992).

Regeneration from stump sprouts, or coppice, would likewise offer advantages for exploiting small gaps. Stump sprouting is an important mechanism for regeneration in many tropical forests, especially those subject to frequent structural disturbance (Putz and Brokaw, 1989; Brown and Lugo, 1990; Kauffman, 1991; Foster et al., 1997). Coppicing has been found to confer regeneration advantage following forest cutting (Uhl et al., 1981; Kennedy and Swaine, 1992; De Rouw, 1993; Pinard et al., 1996; Miller and Kauffman, 1998; Kammesheidt, 1999; Luaga et al., 2004) and it has been suggested that repeated cutting will tend to select for species that recover well by coppicing following injury (Kartawinata et al., 1989; Daniels et al., 1995; see also Putz and Brokaw, 1989). In fact, many traditional forest management systems that rely on short-rotation cutting (e.g., for firewood or small-size construction wood) are centered around the management of coppice (Nyerges, 1989; Putz and Brokaw, 1989; Buckley, 1992; De Rouw, 1993; Kirby and Watkins, 1998; Okada, 1999; Luaga et al., 2004; Nygard et al., 2004).

Many mangrove species have the ability to coppice (Hamilton and Snedaker, 1984; Fong, 1992; Smith and Berkes, 1993; Wadsworth, 1997), yet studies of mangroves have focused almost entirely on regeneration from seeds and propagules (Rabinowitz, 1978). Exceptions include Smith et al. (1994), Snedaker et al. (1992) and Roth (1992) who documented high coppicing regeneration in Caribbean mangroves following hurricanes, and Elster and Perdomo (1999) who tested the use of shoot cuttings for restoration of degraded mangrove forests. These studies found that *R. mangle* coppiced and grew from shoots poorly and, as a result, suffered greater mortality than did *L. racemosa* and *A. germinans*. Azariah et al. (1992) believe that the effects of successive cutting on the Muthupet mangrove in India have been a gradual shift in forest composition towards species that coppice well (*A. marina* and *E. agallocha*) and away from species that do not (*Rhizophora* and *Sonneratia*).

I found evidence of coppicing in both cut and uncut forest, although more in the former (Table 10). Good coppicing ability, as was found with *Sonneratia*, *A. marina* and *R. apiculata* (Table 3), almost certainly gives these species an advantage when gaps are created and helps explain their resiliency in the face of cutting. *Sonneratia* and *A. marina* are known to coppice well (Hamilton and Snedaker, 1984), but the high rates of coppicing found in *R. apiculata* were unexpected. Standard botanical and silvicultural wisdom holds that *Rhizophora* spp. are poor at coppicing (Hamilton and Snedaker, 1984; Azariah et al., 1992; Hussain, 1995) and researchers who have done extensive surveys of mangroves in Australia and the Pacific – including sites subject to cutting – claim that coppicing in *Rhizophora* is highly unusual (Norman Duke and Katherine Ewel, personal communication).

Nonetheless, field measurements indicate and I learned from several mangrove planters that *Rhizophora* trees will survive and regenerate well if cut high enough on the stem that live branches (with leaves) are spared. This is consistent with observations in Florida where *R. mangle* trees that had leafy stems below the level of cutting survived artificial canopy removal (Snedaker et al., 1992). Charcoal makers cutting mangroves in Saint Lucia likewise employed such a practice to minimize mortality of *R. mangle* that they

cut (Walters and Burt, 1991). My observations suggest that coppiced *Rhizophora* trees had, in fact, been cut relatively high on their original stems (typically 1–2 m), whereas *Sonneratia* and *Avicennia* often coppiced very close to the ground. Either way, these findings suggest that coppice regeneration in mangroves has been underestimated and warrants further study.

5. Conclusions

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; Lacerda, 1993), it is remarkable that barely a handful of published studies have examined small-scale wood use and its ecological effects on these forests. Study findings presented elsewhere (Walters, 2000a, 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 woodcutting can be a significant form of ecological disturbance in mangroves (see also Walters, 2005). Forest structure was dramatically altered by cutting, but impacts on composition and regeneration were also detectable and appear to reflect varied levels of species resiliency to cutting impacts. Most notably in this respect is the finding that some, but not all mangrove species appeared to respond to small-scale stem and branch cutting by vigorous resprouting of new stems and branches. 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 understanding and effectively managing such small-scale forest cutting.

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