Research Article

Structural Recovery of Logged Forests in the Solomon Islands: Implications for Conservation and Management

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Abstract

Much of the lowland tropical forests in the Solomon Islands have been heavily logged. However, little is known about the recovery status of these forests. We examined factors that influenced the recovery of forest structural attributes within 50 years after selective logging on Kolombangara Island in the western Solomon Islands. Twelve study sites—six logged and six unlogged—were identified across the Island, with two logged sites in each of three recovery-time classes: 10, 30, and 50 years after logging. Within each study site, 12 0.1-ha plots were randomly established, and a series of forest attributes measured in each plot. Our results revealed that local logging intensity and soil attributes have stronger influence on forest-structural recovery than do site attributes such as local topography or tree architecture. Furthermore, half a century of regeneration following logging is insufficient to permit full recovery of forest structure. We conclude that logged forests on Kolombangara and possibly across the Solomon Islands may not fully recover structurally before the next logging cycle, in the absence of a policy on re-entry harvesting. The development of such a policy coupled with robust forest-management measures is pivotal to facilitating sustainable logging while supporting biodiversity conservation in the Solomon Islands. This may be the last best hope for saving lowland forests and their biodiversity on this unique tropical archipelago.

Introduction

Forest structure—the three-dimensional architecture and abiotic elements of a forest—is fundamental to regulating primary productivity, gas exchange, and evapotranspiration because it fosters biophysical interactions and biodiversity at various spatial scales (Arcilla et al., 2015; Gadow et al., 2012; Peña-Claros et al., 2012; Seidler, 2017). In addition to regulating ecosystem functions, forest structure also influences the influx of solar energy to the understory (Ozanne et al., 2003; Wedeux & Coomes, 2015). High radiation influx occurs in canopy gaps which increases the rate of photosynthesis in exposed understory areas, enhancing tree growth and structural development (Katovai & Katovai, 2012). Ecophysiological functions of forest structure also provide goods and services to forest wildlife communities, and contribute to niche specializations that can result in distinct community assemblages along the forest stratum (Brauman et al., 2007; Buchmann, 2002; Layman et al., 2015).

In addition, several studies have suggested that forest canopies alone contain 40% of global biodiversity, 10% of which are canopy-specialist species (Basset et al., 2003; Mendieta-Leiva & Zotz, 2015; Nakamura et al., 2017). More than 25% of herbivorous insects and 10% of vascular epiphytes are canopy specialists (Basset et al., 2003; Wagner et al., 2015). Furthermore, forest-structural attributes are

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important for biodiversity: for example, a tree's canopy size, branching architecture, number and size of cavities, height, and volume are positively correlated with levels of ecosystem provisioning and resident biodiversity (Basset et al., 2003; Maiti et al., 2015; Poorter et al., 2006; Remm & Lõhmus, 2011). At the community level, large old-growth trees contribute extensively to ecosystem functioning, and provide key habitats for biodiversity (Almeida-Gomes et al., 2016; Lindenmayer & Laurance, 2017).

Logging in tropical forests is known to substantially degrade forest structure (Cazzolla Gatti et al., 2015; Fisher et al., 2011; Katovai et al., 2021). For instance, tree felling increases mortality of neighbouring trees via liana networks that drag adjacent trees down, creating considerable gaps within forests (Forshed et al., 2008; Shenkin et al., 2015; Wright et al., 2015). The extraction of felled logs also requires the aid of logging roads, skidding trails, and log-storage areas. These activities usually create large gaps ($>90\,m^2$) in the forest (Cazzolla Gatti et al., 2015; Katovai & Katovai, 2012). The recovery of natural vegetation and therefore forest structure in these gaps can potentially be impeded by topsoil removal and compaction, and a proliferation of invasive plants and animals (Fujinuma & Harrison, 2012; Harrison & Swinfield, 2015; Williamson & Neilsen, 2000).

Although logging in tropical forests tend to have a long-term impact on forest structural recovery (Asase et al., 2014; Shenkin et al., 2015), little is known about factors prompting this recovery (Wedeux & Coomes, 2015), particularly in oceanic islands. Forest recovery on such islands may differ from mainland tropics (e.g. Gillespie et al., 2008; Katovai et al., 2016; Whitmore, 1989), because they often have relatively young soils, a small regional species pool, and higher endemism due to their size and isolation, and are subjected to frequent, severe weather events including cyclones or hurricanes (Burslem et al., 2000; Katovai et al., 2015; Osazuwa-Peters et al., 2015).

Here we examine the pattern and pace of forest structural recovery following selective logging in the Solomon Islands. We (i) assessed factors influencing recovery of forest structure in previously logged forests, and (ii) determined whether a half century was sufficient to allow forest structure to recover to pre-logging conditions. The outcome of this study may aid policy development pertaining to logging practices and forest management on oceanic islands.

Methods

Site Description

This study was conducted on Kolombangara Island ($157°$ E and $5°$ S) in the New Georgia group, Solomon Islands. Kolombangara is a volcanic island having a near-perfect circular shape, and comprising a land area of approximately 80,000 ha (Figure 1). Topsoils on the island are generally Typic Haplorthox—highly weathered brownish red soil—and acidic (pH < 5), and have a high organic matter (Katovai et al., 2012; Wairiu & Lal, 2003). The underlying rocks are predominantly olivine basalt breccias and lavas (Burslem et al., 2000). The exceptionally high annual rainfall on the island ($\sim3000\,mm/yr$) supports wet tropical forests that stretch from the relatively flat coastline to the rim of the volcanic cone (1700 m a.s.l.), which is in the center of the island (Katovai et al., 2016).

The lowland forests on Kolombangara have been logged since 1964 (Katovai et al., 2016). Much of the island's southeast, northeast, and northwest quadrants were either converted to commercial tree plantations or to pastures for grazing (Katovai et al., 2012). These land use activities were marginal in the southwest quadrant of the island thus allowing uninterrupted regeneration in logged forests. Patches of unlogged lowland forest still exist on Kolombangara Island which form a mosaic with logged forests at different stages of recovery. These unlogged forests are not legally protected and are vulnerable to future logging (Cazzolla Gatti et al., 2015; Katovai et al., 2012).

Study Design

Forest survey began in January through to December 2013 where forest structure, proxies of logging impact, topography and soil attributes were surveyed. These were conducted in 144 plots of 0.1 ha (50 m x 20 m) established in both logged and unlogged forests. We evaluated traditional and published information to avoid plot establishment on past human settlements (Katovai et al., 2016). To investigate structural change post-logging recovery, the study was restricted to forests logged 10, 30, and 50 years previously. Old logging records, forestry reports and local knowledge were used to determine areas logged within each aforementioned time category. Using the initial data collected for this study and other comparable studies done on Kolombangara, we used power analyses to determine the sample size for this study (Irvine & Rodhouse, 2010; Katovai et al., 2012; Whitmore,1989). For each time class, two forest coupes logged within the same time frame were selected. In each coupe 12 plots were established, giving a combined total of 72 plots for the three time classes. An additional 72 plots were also established in unlogged forests within the study area, to enable comparison of the structural differences between logged and unlogged areas. Plots were randomly stratified across each logged coupe and unlogged forest patch to ensure the variation between forest interiors and edges are effectively captured.
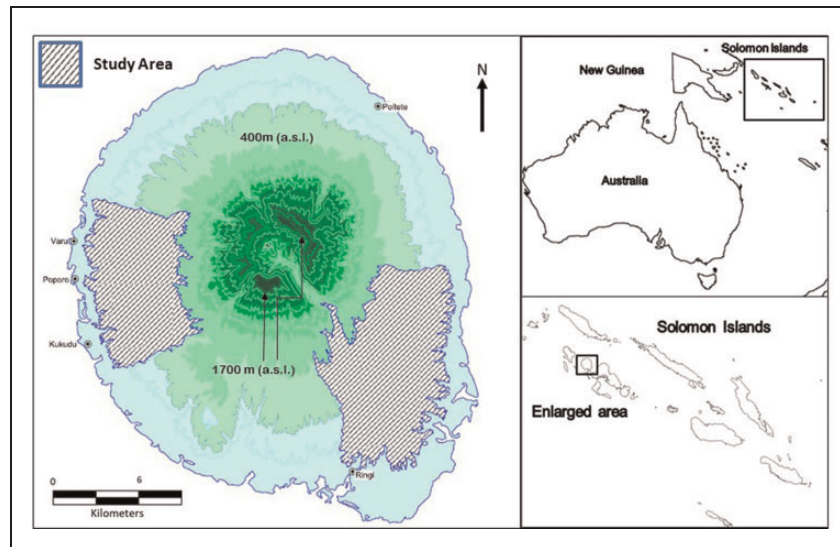
Figure 1. A Map of Kolombangara Island Showing the Study Area.

Forest Structure

Forest-structural attributes recorded in the study included canopy cover, canopy height, and stem abundance for tree size classes–large trees and saplings. Surveys of canopy cover and canopy height, were conducted at 16 random points within each 0.1 ha plot. Visual techniques adopted from Swiecki and Bernhardt (2001) were utilized in estimating canopy cover within a radius of 2 m at each point to determine mean plot canopy cover. Canopy height was measured using a Leica laser (Lieca Camera, Inc., Wetzlar, Germany) where mean values for each plot were then calculated. Large trees (stem DBH > 30 cm) abundance was tallied in each 0.1 ha plots. The abundance of saplings with DBH 1–5 cm and those with DBH >5–10 were tallied in four 5 x 5 m subplots and four 10 x 10 m subplots respectively before the abundance for both size classes were extrapolated to 0.1 ha. Change in each attribute was then modeled as a function of recovery time, using proxies for logging, topography, tree architecture, and soil attributes.

Predictors

Distance to the nearest unlogged forest, harvest intensity and liana abundance were used as proxies for logging. The distance between each logged forest plot and the nearest unlogged forest was determined using a GPS (Garmin 76cx GPS; Garmin International, Inc., Kansas City, USA). Harvest intensity (measured as the loss of stand basal area) was estimated from stumps cut at DBH (~1.3 m). A stem profile model for tropical forests was used to reconstruct cut stumps that were partially decomposed as well as estimate basal area for either cut below or above the DBH (Katovai et al., 2016; Ito et al., 2010). Lianas of all sizes present at DBH were tallied to determine liana abundance in four nested subplots, each of 0.01 ha (10 m x 10 m) in size; liana abundance was then extrapolated to 0.1 ha. Topographical elevation was measured at the center of each plot using a Garmin 76cx GPS (Garmin International, Inc., Kansas City, USA). Slope was estimated in each plot by averaging slope readings from five random points using an ECII D Electronic Clinometer (Haglöf. Inc., Långsele, Sweden). Proxies for tree architecture were determined by height of first major branching and stem abundance of canopy trees—trees with crown constituting the forest canopy layer. The former was measured with a Leica laser distance measurer (Leica Disto D5; Geosystems Inc., Heerbrugg, Switzerland) for 10 randomly-picked trees per plot. All canopy trees within each plot were then counted to determine stem abundance. Soil attributes investigated were leaf litter depth calculated from 10 random points per plot, and soil nitrogen (N). Soil nitrogen in each plot was estimated from 30 cm of top soil samples collected in four randomly selected points per plot using a cylindrical soil extractor. The samples were air-dried and then thoroughly mixed and sieved through a 2 mm mesh (e.g. Asase et al., 2014). The N weight percentage (%N) for each sample were then determined by A Costech Elemental Analyzer (Costech Analytical Technologies, Inc., CA, USA) and Continuous-Flow Isotope Ratio Mass Spectrometry (Bay et al., 2015; Katovai et al., 2016).

Statistical Analyses

Predictors used in this analyses were firstly ordination axes generated by simplifying a much larger set of potential predictors using nonmetric multidimensional scaling

(NMS) on PC-ORD (McCune & Mefford, 2011). We then selected the best predictors for each model-fitting exercise based on sound biological and ecological reasoning (Table 1).

Factors Affecting Structural Recovery. To identify factors influential in the recovery patterns of forest structures, each examined structure was modeled as a function of recovery time, and proxies for logging, topography, tree architecture, and soil attributes (Table 1). The influence of each predictor in the respective models was then assessed, and reported in relation to how it affected the pattern of forest structural recovery in the logged forests.

Linear mixed models (LMMs) were used to investigate the relationships between forest structures and the 11 ecological predictors. Coupe was treated as a random effect in the models, as the plots nested in each coupe were not independent. Before generating a global model and candidate model sets for each selected response, potential variables were investigated by plotting pairs of variables, calculating correlation coefficients for each pair, and examining the variance inflation factors using the package *usdm* in R (Naimi, 2017). Global models were generated, in which each selected forest structure was modeled as a function of all predictor variables (Table 1). Interactions between time since logging and all predictor variables were also included in the models as time–predictor interactions, under assumption that these interactions would be important in explaining forest regeneration after logging (Katovai et al., 2016). A candidate model set having of the 'best simplest model (s)' (delta AICc < 7) was generated from which parameters for all variables included in models were then averaged based on model weights (Burnham & Anderson, 2004; Mazerolle, 2015). Inferences from averaged parameter estimates were drawn based on effect sizes, and whether their 95% confidence intervals (CI) overlapped zero. All analyses were generated using R (R Core Team, 2020).

Forest Structural Recovery. We compared the means of canopy height and cover, ground cover and stem abundances across logging ages using nested analysis of variance (ANOVA) generated in the mixed model function in the *lme4* R package (Bates et al., 2015). Satterthwaite's approximation in the *lmerTest* R package was used to determine the degrees of freedom in the analyses, as 'coupe' and 'time' were treated as random and fixed effects, respectively (Kuznetsova et al., 2015). The *multcomp* package in R was then used to generate post hoc comparison tests for forest structures that showed significant mean differences among forest classes (Hothorn et al., 2015). Graphical visualization of these

Table 1. Forest Structures and the Set of Predictors Included in The Linear Mixed Models.

		Distance to nearest unlogged forests (logging)	Harvest intensity (logging)	Liana abundance (logging)	Site elevation (topography)	Site slope (topography)	Height of major branching (tree architecture)	Stem abundance of canopy trees (tree architecture)	Leaf litter depth (soil)	Soil nitrogen (soil)	Soil pH (soil)	After logging regeneration (time)
Forest structures	Canopy cover	•	•			•			•	•		•
	Canopy height	•	•	•	•	•	•	•	•	•	•	•
	Tree abundance (DBH > 10 cm)	•	•	•	•	•			•	•		•
	Sapling abundance (DBH 1 cm–10 cm)	•	•	•	•	•			•	•		•

Each forest structure was modeled against corresponding predictors (•).

analyses was generated in Statistix 10 (Tallahassee, FL 32317, USA).

Results

Factors Affecting Forest Structure

The relatively low Variance Inflation Factors (< 6.42) and range for paired correlation coefficients (R = −0.42 to 0.44) indicated the absence of significant collinearity among the selected predictors.

Canopy Cover. The average LMM for canopy cover fitted the data well (Pearson's R between observed and model-fitted values = 0.91), and comprised 7 models that included time, height of major branching, leaf litter depth, and soil N (Table A1(a)). Time influenced canopy cover in all three logged forest time classes (Table 2). The time effect for the 10-year post-logging time class was positive (slope = 0.37; 95% CI = 0.2, 0.55), but was negative and of similar scale for the 30-year (slope = −0.32; 95% CI = −0.41, −0.22) and the 50-year (slope = −0.32; 95% CI = −0.41, −0.22) post-logging time classes.

Canopy Height. The averaged model for canopy height had an exceptionally high fit to the data (R = 0.96), and contained three models that included time, liana abundance, and soil N (Table A1(b)). The influence of time was positive for all logged forest classes. The influence of time on canopy height was greatest for the 10-year post-logging time class (slope = 2.56; CI = 0.25, 0.26) and less for the 30- and 50-year post-logging time classes (slope = 0.68; CI = 0.63, 0.73, and slope = 0.74;

CI = 0.69, 0.79, respectively). The influence of liana abundance on canopy height was negative and weak (slope = 0.03; CI = −0.01, 0.0). Although an important component of the averaged model, soil N had no influence on canopy height (Table 2).

Large Tree Abundance. The large tree abundance averaged model showed a strong fit to the data (R = 0.74), and comprised nine models that included time, harvest intensity, distance to nearest unlogged forest, liana abundance, leaf litter depth, slope and soil N (Table A1(c)). However only liana abundance had a marked influenced on large tree abundance (slope = -0.6; CI = -0.47, -0.05) at 10 years after logging (Table 2).

Sapling Abundance. The average LMM for sapling abundance fitted the data reasonably well (R = 0.60), and included nine models. The averaged model included time, harvest intensity, leaf litter depth, and soil N (Table A1(d)). Harvest intensity was the most important predictor in the averaged model, having a positive influence on sapling abundance (slope = 0.5; CI = 0.02, 0.09). Leaf litter depth also had a slightly negative influence on sapling abundance (slope = −0.07; CI = −0.14, −0.01). The influence of time on sapling abundance varied, having a positive influence for the 10-year post-logging time class, but a negative influence for the 30- and 50-year post-logging time classes (slope = 8.18; CI = 7.27, 9.09, and slope = −0.38; CI = −6.42, −7.25, respectively). Although soil N and the interaction between time and soil N were important predictors in the averaged model, they had no influence on sapling abundance (Table 2).

Table 2. Summary of Important Predictors and Their Influence on Forest Structures Among Logged Forests on Kolombangara Island.

Forest structure	Important predictor	Relative importance	Slope	95% Confidence interval	
Canopy cover	Time~10 yrs.	0.61	0.37	0.2	0.55
	Time~30 yrs.	0.61	−0.32	−0.41	−0.22
	Time~50 yrs.	0.61	−0.32	−0.41	−0.22
Canopy height	Liana abundance	0.03	−0.01	−0.01	0
	Soil nitrogen	0.08	−	−	−
	Time~10 yrs.	1	2.56	2.5	2.6
	Time~30 yrs.	1	0.68	0.63	0.73
	Time~50 yrs.	1	0.74	0.69	0.79
Large tree abundance (DBH > 30 cm)	Liana abundance	0.53	−0.6	−0.47	−0.05
	Time~10 yrs.	0.55	−0.41	−1.50	−8.53
Sapling abundance (DBH >1 cm–10 cm)	Basal area harvest	0.51	0.5	0.02	0.09
	Leaf litter depth	0.04	−0.07	−0.14	0
	Soil nitrogen	0.23	−	−	−
	Soil nitrogen:time	0.05	−	−	−
	Time~10 yrs.	0.34	8.18	7.27	9.09
	Time~50 yrs.	0.34	−0.38	−6.42	−7.25

Forest Structural Recovery Across Time

Canopy Cover. Although the mean canopy cover among forest classes was highly significant ($F_{3, 8} = 51.1$; $p < 0.0001$), the post hoc comparisons test revealed only two homogenous groups: (i) the 10-year post-logging time class; and (ii) all other forest classes (Figure 2A). The variation in canopy cover between coupes in each forest class was relatively low (χ^2 (1, $N = 144$) = 0.763; $p = 0.4$).

Canopy Height. The mean canopy height significantly differed among all forest classes ($F_{3, 8} = 471.6$; $p < 0.0001$) with the post hoc test revealing each forest class as a homogenous group. The largest difference occurred between the 10- and 30-year after logging time classes. The differences in means between the 30- and 50-year post-logging forests, and between the latter and unlogged forests, were much smaller but still significantly different (Figure 2B). Canopy height variations among coupes in the same forest class were not significant (χ^2 (1, $N = 144$) = $1.14\mathrm{e}^{-13}$; $p = 1$).

Large Tree Abundance. The mean for large tree abundance differed among the various forest classes ($F_{3, 8} = 14.9$; $p < 0.0001$). However post hoc test revealed only two homogenous groups between which large tree abundance differed (Figure 2C). Coupe variation for large tree abundance within each forest class was relatively higher than mean variation among forest classes (χ^2 (1, $N = 144$) = 2.97; $p = 0.08$).

Sapling Abundance. There were highly significant differences for the mean sapling abundance among the forest classes ($F_{3, 8} = 50.9$; $p > 0.0001$). The post hoc test revealed three homogenous groups among which means differed (Figure 2D). The 10-year after logging time class had the highest mean (6212.5 stems per ha), followed by the 30-year (4291.7 stems per ha) and 50-year (4328.1 stem per ha) after logging time classes, which had similar means. Unlogged forests had the lowest mean (2798.6 stems per ha). Variations in sapling abundance between coupes in the same forest class were fairly low (χ^2 (1, N = 144) = 0.06; p = 0.8).

Discussion

Forest structure of Kolombangara, Solomon Islands have not completely recovered within 50 years of selective logging. It appears that logging-intensity and soil attributes were more important than local topographical and tree architectural factors in influencing the structural recovery (Table 2).

Canopy Cover

Based on our estimates, canopy cover in logged forests on Kolombangara Island appeared to have returned to pre-logging levels of after 10 years of natural regeneration (Figure 2A). Full canopy recovery in our study is comparable with similar studies across insular and mainland tropical forests (Asner et al., 2004; Dalagnol et al., 2019; Milodowski et al., 2021; Pereira et al., 2002). It was argued that forest canopy recovery is strongly inversely correlated to the logging intensity in these forests (Milodowski et al., 2021). Logging in most tropical forests are usually guided by a conventional approach including cut-size limitation [usually >50-60cm DBH] and species preference (Katovai et al., 2015; Putz et al., 2012). In contrast, logging in the Solomon Islands is highly intensive. Any hardwood or softwood tree species of commercial value as small as 30 cm DBH is usually harvested (Bennett, 2000; Katovai et al., 2012). Such intensive logging practice often results in large forest gaps due to non-directional felling of adjacent trees. Nonetheless the quick recovery of canopy cover to pre-logging levels in this study was largely due to high levels of tree recruitment in logging gaps (Shenkin et al., 2015). The proliferation and long-term dominance of large-crown species including *Campnosperma brevipetiolata* contributed strongly to the recovery of the logged forest canopy (Katovai et al., 2016). Lateral branching and crown extension of adjacent trees also contributed to canopy closure (Meng et al., 2006).

There were distinct temporal patterns in canopy recovery after logging. The positive influence on canopy cover at 10 years after logging was largely attributed to the turnover between fast growing short-lived pioneers and mid-succession species (Katovai et al., 2016). The marked negative effect of time in canopy cover across 30 and 50 years after logging suggests that the establishment and growth of long-lived pioneers and late succession species was supplemented by only minimal forest turnover (Katovai et al., 2016; Wedeux & Coomes, 2015). It also suggests that the impact of natural disturbances, including extreme weather, was similar in logged and unlogged forests during the intervening 20-year time period. Cyclonic events more than four decades ago caused major damage to the canopy structure on Kolombangara Island (Burslem et al., 2000). The absence of cyclones affecting the island since 1970 has possibly enabled the canopy structure to recover in logged forests.
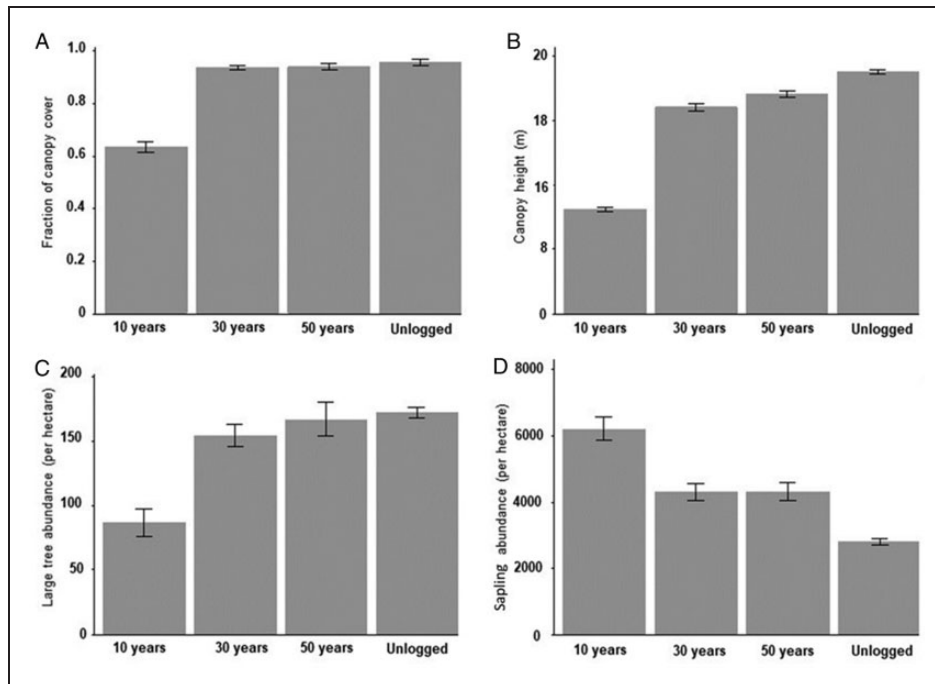
Figure 2. Differences in Forest Structural Attributes Among Forest Classes. Mean canopy cover and large tree abundance (A and C) recovered to pre-logging levels between 10 and 30 years after logging. The letters above each error bar in each class denote homogenous groups as revealed by the post hoc tests. Bars in each class with the same letter do not differ significantly.

Canopy Height

Even after 50 years canopy height has not recovered in the logged forests of Kolombangara Island (Figure 2B). Consistent with studies in Brazil and Malaysia, this indicates that canopy structural development in logged forests is a gradual process (Okuda et al., 2003; Villela et al., 2006). Our findings reveal that the recovery of mean canopy height to pre-logging levels would only be possible if these regenerating forests are protected from further logging. However due to poor governance in the forest sector, re-entry logging is apparently inevitable in the Solomon Islands, indicating that the chances of full recovery is slim (Katovai et al., 2015). Our results also underscore the negative influence of liana abundance on canopy height. Liana abundance is remarkably high in logged forests in the Solomon Islands, particularly in large tree fall gaps. This is consistent with a previous study on Kolombangara Island inferring high levels of canopy loss from high-intensity harvesting can promote a hyper-abundance of *Convolvulus spp.* lianas, which may arrest forest regrowth (Neil, 1984).

Large Tree Abundance

The low mean of large tree abundance at 10 years after logging is largely attributed to the negative influence of liana abundance. The homogeneity in mean of large tree abundance in the 30 years, 50 years and unlogged forest classes suggests that the tree stand density in logged forests reached saturation levels after 10 years of recovery (Figure 2C). The steadily high abundance of large trees at 30 and 50 years of recovery after logging may also indicate more rapid turnover times between pioneers and late succession specialists (Katovai et al., 2016). The high level of recruitment of climax species in the understory of these forests may have facilitated this rapid turnover (Katovai et al., 2012).

Sapling Abundance

Logging has had an important effect on mean sapling abundance and composition on Kolombangara Island. Sapling abundance doubled in recently logged forests and remained at elevated levels in forest 50 years post logging. Because of the high intensity harvesting that occurred on the island (Katovai et al., 2015), numerous large forest gaps were created in the logged landscape, which enabled high light intensities to reach the understory (Katovai & Katovai, 2012). Light enhancement subsequently triggered the proliferation of fast growing, short-lived pioneers during early succession, particularly *Macaranga dioca* (Aoyagi et al., 2013; Katovai et al., 2016). High mortality of this species occurred between

10 and 30 years after logging because of decreasing light levels associated with the progressive recovery of the forest canopy. The decline in sapling abundance was also affected by the low recruitment of climax species (Katovai et al., 2012; Pessoa & Araujo, 2014). There is sufficient evidence from post-logging recovery studies across the tropics suggesting that seeds of climax species are often severely damaged from topsoil compaction by logging machineries resulting in low turnover of climax species (DeArmond et al., 2019; Howlett & Davidson, 2003; Pinard et al., 2000; Van Nieuwstadt et al., 2001).

The similar mean values for sapling abundance in the 30- and 50-year after logging forest time classes may indicate that the natural thinning is very slow. The process may take more than a century to reach a state of dynamic equilibrium (Pessoa & Araujo, 2014). Sapling abundance in these forests was significantly higher than in unlogged forests because of the persistence of long-lived pioneers during succession (Katovai et al., 2016). This pattern of species co-existence usually occurs in forests subject to ongoing disturbances because of geological volatility and extreme weather (Burslem et al., 2000; Whitmore, 1989). The occurrence of regular strong winds, sporadic earthquakes, and landslides on Kolombangara Island may have resulted in a succession whereby long-lived pioneers can persist (Katovai et al., 2012; Whitmore, 1989).

Implications for Conservation and Management

Half a century of regeneration following logging has been insufficient for full structural recovery of logged forests on Kolombangara Island. The same is possibly true for other logged forest estates in the Solomon Islands due to highly unsustainable logging practices since the early 1950s (Bennett, 2000; Laurance, 2000; Wairiu, 2007). Unlike most large island states in the Southeast Asia and Pacific region that have either diversified their export commodities or have transitioned into a mixed or service-based economy, Solomon Islands remains dependent on round-log exports as it contributes 50–70% of annual export revenue since gaining its independence in 1978 (Katovai et al., 2015). Following the collapse in oil palm and rice industries (1986) due to cyclone Namu and the massive decline in other export commodities during ethnic civil unrest (1990s), the Solomon Island government forestry sector facilitated a rise in annual logging exports to assist the ailing economy by increasing logging licenses issued to landowners and permitting them to sub-contract foreign companies to operate their land (Wairiu, 2007). As such, timber-harvesting rates have quadrupled, dramatically exceeding sustainable levels (Katovai et al., 2021).

Despite only partial recovery of forest structure across Kolombangara, such selectively logged forests may still support high levels of biodiversity (Laurance & Edwards, 2014). For example, the recovery of canopy structure to pre-logging levels may have facilitated the restoration of biodiversity to levels similar to that in unlogged forests (Gao et al., 2014). Similarly, the recovery of large tree abundance in logged forests may suggest that these forests have retained ecological functions able to sustain high levels of biodiversity (Almeida-Gomes et al., 2016). The high sapling abundance in regenerating logged forests may provide a refugia for ground-dwelling animals and the recruitment of climax plant species, aiding the increase in understory biodiversity (Katovai et al., 2012). Although the mean canopy height level was lower in logged forests, its large increase with recovery time suggests that biodiversity levels may eventually recover to pre-logging levels (Gillison et al., 2013), if further logging or other land use activities do not occur.

Nonetheless premature re-entry logging and other unsustainable land use activities are still inevitable across Kolombangara due to no policy on re-entry logging coupled with poor logging practices in Solomon Islands (Katovai et al., 2021, 2016, 2015). Such practices can lead to further structural degradation of logged forests on the island, potentially resulting in unprecedented biodiversity loss in the future. It is therefore paramount for forest resource owners across the Solomon Islands to collaborate with relevant government and non-government stakeholders, and develop national policies on re-entry logging and harvesting limitations to drive a transdisciplinary land-use management system (Reed et al., 2020). This system should be based on reduced-impact logging to achieve socioeconomic benefits while sustaining forest resources for future use, and biodiversity conservation outcomes (Edwards et al., 2012; Katovai et al., 2021, 2015). Such an initiative may perhaps be the last resort in protecting key lowland forests on Kolombangara from further degradation and consequently sustaining the biodiversity within these forests.

Appendix

Table A1. Candidate Model Set for Each Forest Structure Variable (only for ΔAICc < 7) Used in the LMM Analyses.

(a) Canopy cover

Model	BAH	Distuf	Brht	Litter	StAb	Slope	Snitro	SpH	Time	BAH:Time	Distuf:Time	Brht:Time	Litter:Time	Time:StAb	Slope:Time	Snitro:Time	SpH:Time	df	AICc	ΔAICc
257	—	—	—	—	—	—	—	—	0.38	—	—	—	—	—	—	—	—	5	−152	0.00
1	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	3	−150.7	1.36
321	—	—	−0.02	—	—	—	−0.13	—	0.43	—	—	—	—	—	—	—	—	6	−149.2	2.86
5	—	—	−0.02	—	—	—	—	—	—	—	—	—	—	—	—	—	—	4	−148.4	3.62
65	—	—	—	—	—	—	−0.14	—	—	—	—	—	—	—	—	—	—	4	−148.3	3.77
265	—	—	—	0.02	—	—	—	—	0.34	—	—	—	—	—	—	—	—	6	−148.1	3.9
9	—	—	—	0.02	—	—	—	—	—	—	—	—	—	—	—	—	—	4	−145.8	6.19

(b) Canopy height

Model	BAH	Distuf	Elev	Liab	Litter	Slope	Snitro	Time	BAH:Time	Distuf:Time	Elev:Time	Liab:Time	Litter:Time	Slope:Time	Snitro:Time	df	AICc	ΔAICc
129	—	—	—	—	—	—	—	2.56	—	—	—	—	—	—	—	5	−125	0.00
193	—	—	—	—	—	—	0.07	2.54	—	—	—	—	—	—	—	6	−120.2	4.76
137	—	—	—	0.01	—	—	—	2.68	—	—	—	—	—	—	—	6	−118.1	6.9

(c) Large tree abundance

Model	BAH	Distuf	Elev	Liab	Litter	Slope	Snitro	Time	BAH:Time	Distuf:Time	Elev:Time	Liab:Time	Litter:Time	Slope:Time	Snitro:Time	df	AICc	ΔAICc
23516	−12.61	—	−0.2	−1.92	−8	—	−52.48	353.7	*	—	*	*	*	—	*	20	683.5	0.00
21468	−11.93	—	−10.18	−2.67	−6.28	—	−58.31	384.3	*	—	*	*	*	—	*	18	684.8	1.35
23548	−12.47	—	−0.44	−1.84	−4.81	0.36	−50.31	341.6	*	—	*	*	*	*	*	21	685.8	2.32
21500	11.63	—	−12.15	−2.69	−1.94	0.46	−57.32	374.5	*	—	*	*	*	*	*	19	686.8	3.33
31740	−12.73	—	−0.15	−1.99	−10.7	−0.31	−53.84	363.9	*	—	*	*	*	*	*	23	687.4	3.96
29692	−12.15	—	−9.77	−2.78	−10.97	−0.53	−60.55	401.3	*	—	*	*	*	*	*	21	687.8	4.28
21460	−14.36	—	25.12	—	−12.39	—	−37.51	275.3	*	—	*	—	*	—	*	17	689.8	4.77
23520	−14.02	—	1.21	—	−17.8	−0.02	−34.12	356.1	*	—	*	—	*	*	*	21	689.8	6.35
21492	−14.12	—	23.71	—	−8.71	0.39	−36.49	266	*	—	*	—	*	*	*	18	690.1	6.64

(d) Sapling abundance

Model	BAH	Distuf	Elev	Liab	Litter	Slope	Snitro	Time	BAH:Time	Distuf:Time	Elev:Time	Liab:Time	Litter:Time	Slope:Time	Snitro:Time	df	AICc	ΔAICc
2	0.1	—	—	—	—	—	—	—	—	—	—	—	—	—	—	4	48.70	0.00
129	—	—	—	—	—	—	—	8.69	—	—	—	—	—	—	—	5	49.70	0.99
1	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	3	50.70	2.00
66	0.1	—	—	—	—	0.05	—	—	—	—	—	—	—	—	—	5	51.40	2.70
16577	—	—	—	—	—	0.06	8.81	—	—	—	—	—	—	—	—	6	52.60	3.81
193	—	—	—	—	—	—	7.99	—	—	—	—	—	—	—	*	8	52.60	3.86
18	0.1	—	—	—	−0.1	—	—	—	—	—	—	—	—	—	—	5	53.00	4.23
65	—	—	—	—	—	—	−0.02	—	—	—	—	—	—	—	—	4	53.20	4.46
130	0	—	—	—	—	—	—	8.20	—	—	—	—	—	—	—	6	54.40	5.65

Each row in the table shows a model with a set of corresponding predictors, degrees of freedom (df), Akaike's information criterion corrected (AICc), and the change in corrected Akaike's information criterion (ΔAICc). An "*" indicate that time and/or predictor–time interactions were included in the corresponding model. "—" implies that predictors and their interactions with time were not included in the corresponding model.

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