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Research Article

Impacts of forest management on community assemblage and carbon stock of lianas in a tropical lowland forest, Malaysia

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Abstract

The study determined the impacts of different forest management regimes on liana community assemblages and carbon stocks in the Bukit Panchor Forest Reserve, Malaysia. Based on time span, two forests treated by the Malayan Uniform System (MUS), but with different time of recovery (19 years old: MUS-19 and 42 years old: MUS-42) were selected for this study. The MUS is a silvicultural treatment involving a single harvest of trees of stipulated diameter (≥ 45 cm), followed by other silvicultural operations such as climber cutting. An untreated forest was added as a control. Lianas with diameter ≥ 2 cm were enumerated in ten 40 × 40 m^2 plots within each regime. Liana above-ground carbon stocks were determined using an allometric equation. Observed species richness and Shannon diversity of lianas were significantly lower in the MUS-19 treated forest than in the untreated forest ($p<0.05$), but the values of these attributes were similar in the MUS-42 treated and untreated forests. Rarefied liana species richness was significantly lower in the two treated forests than in the untreated forest ($p<0.05$). Liana stem basal area and carbon stock decreased significantly in the MUS-19 treated forest ($p<0.05$) relative to the untreated forest, whereas the values in the MUS-42 treated forest were similar to those in the untreated forest. In view of the adverse impacts of complete liana cutting on liana diversity, structure and carbon stocks in the treated forests, it is recommended that selective liana cutting be used in controlling lianas.

Keywords: liana carbon stock; liana cutting; liana diversity and community structure; silvicultural treatments

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Introduction

Lianas are woody climbing plants that depend on trees and other plants to ascend to the forest canopy. They enhance the functioning of tropical forest ecosystems due to the many important ecological functions they perform. In terms of species richness, lianas can contribute more than one-third of woody plant species in the tropics [1]. They play a number of roles which help to maintain diversity in tropical forests [2-6]. They also control the structure of tropical and subtropical forest ecosystems [7]. Even though lianas are important in tropical forest ecosystems, they can also have adverse impacts on tree growth and development. For instance, high liana numbers hamper natural regeneration [8,] and growth [9] of trees in tropical forests. They also affect carbon storage capacity of trees [10]. The physical presence of lianas sometimes inflicts damage on trees, and reduces their value as timbers [11]. In view of the negative impacts of lianas on trees, lianas are usually cut in managed forests so as to free trees and enhance their growth and natural regeneration.

Management of lianas is one of the silvicultural practices employed in tropical forests to enhance biodiversity. Over the years, much attention has been given to liana cutting in conjunction with logging in many tropical forests. The main aim of liana cutting is to reduce liana abundance and therefore, their undesirable effects on trees [12]. Lianas are usually cut without being aided artificially to regenerate, because they have the ability to resprout and increase in abundance to appreciable levels with time. A few studies have confirmed the ability of lianas to recover in abundance following liana cutting [8, 13-14], but forest managers who practice liana cutting do not consider the fate of liana species composition and diversity following liana cutting. Although liana cutting can reduce liana abundance and improve tree diversity, it can have adverse consequences on liana species composition and diversity in tropical forests. For instance, Gerwing and Vidal [15] reported that liana species richness was considerably lower in treated plots than in untreated plots within an eastern Amazonian forest, eight years after liana cutting. Due to the significant ecological roles of lianas in tropical forests, liana species loss and composition changes may affect the health of tropical forest ecosystem [15]. However, in another study conducted in a lowland tropical forest in Malaysia, there was no significant difference in liana species richness between treated and untreated forests 40 years after liana cutting [14]. It is not clear whether the difference between the two studies is due to the difference in time span of the silvicultural treatments used. It is therefore important to conduct studies that consider the impacts of time span of forest management regimes on liana species diversity. Such studies could generate information useful for developing effective forest management practices that support liana diversity conservation.

Most studies that examine the influence of silvicultural treatments on liana assemblages have centred on liana abundance [e.g. 8, 16-17], with little or no attention to other important measures of liana community structure such as liana stem basal area and species dominance. For better understanding of the impacts of liana cutting on liana community structure, other structural attributes of lianas should be considered as well. To date, only a limited number of studies have looked at the impacts of liana cutting on liana stem basal area. Gerwing and Vidal [15] and Alvira et al. [12] reported that liana cutting was responsible for 85 and 69 % reductions, respectively, in liana stem basal area in the short term (2 and 8 years, respectively). Comparable silvicultural treatments in the Bobiri forest reserve, Ghana, also resulted in a significant decline in liana stem basal area in some treated forests, even 40 years after the treatments [13].

Although the Bukit Panchor forest reserve has a long history of silvicultural interventions of liana cutting, there is no information on the impacts of the silvicultural activities on liana diversity and community structure as well as liana carbon stocks in the treated forests. The current study was therefore conducted to determine the impacts of different time spans of the Malayan Uniform System (19 years old: MUS-19 and 42 years old: MUS-42) on liana diversity and community structure, and liana carbon stocks in the Bukit Panchor Forest Reserve. The MUS is a silvicultural treatment involving a single harvest of trees of stipulated diameter (≥ 45 cm) followed by other silvicultural operations such as climber cutting. The current study addressed the following research questions: (1) What is the impact of liana cutting on liana diversity and community structure under different time span of treatment? (2) What is the impact of liana cutting on carbon storage of lianas under different time span of treatment?

Methods

Study area, site selection and sampling

The present study was conducted in the Bukit Panchor Forest Reserve in Penang, Malaysia (Fig. 1) (N 5^0 09.631' E 100^0 32.889'). The forest reserve is one of the lowland dipterocarp forests in Malaysia, and the main tree species of the forest vegetation include *Shorea parvilora*, *Shorea curtisii*, *Shorea leprosula* and *Hopea* spp. The average temperature and annual rainfall of the study region are 33°C and 2670 mm, respectively, and the relative humidity for the area is 70-90 %.

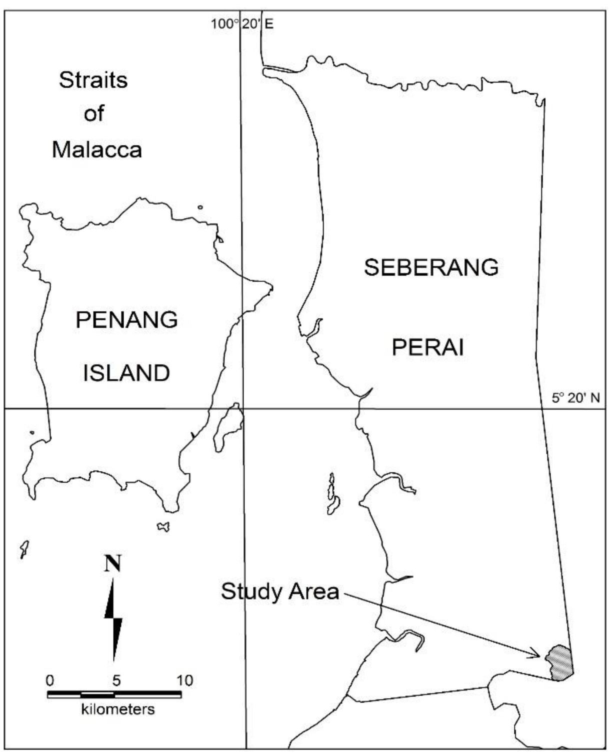

Fig. 1. Map of Penang State showing location of the study area

Based on time span of the Malayan Uniform System (MUS), two different time of recovery were selected for the study. These were the 19 years old MUS (MUS-19) treated forest and 42 years old MUS (MUS-42) treated forest. A series of silvicultural activities characterised the MUS treated forests, which are outlined as follows [18]. Firstly, all trees with dbh ≥ 45 cm were logged, and then five years afterwards all defective and non-commercial trees were poison girdled in a single treatment. In addition, all climbers were cut from the forests, after which enrichment planting was carried out. Enrichment planting was carried out to enhance regeneration of the forests, and it involved planting seeds or seedlings of desirable tree species in the open canopy forests. An untreated primary forest was added as another forest management regime to serve as a control. This made it possible to determine the extent of recovery of liana assemblages and carbon stock since the inception of the silvicultural treatments in the MUS treated forests. Sampling was conducted in two replicate sites within each MUS treated forest and the untreated forest. Each MUS treated forest covered a total area of about 10 ha, and the untreated forest had a total area of about 20 ha.

Five sampling plots (each of dimension 40 × 40 m^2) were randomly located in each of the two sites within each regime. Thus, a total of 10 plots (1.6 ha) were established in each forest management regime (30 plots in all). Lianas with diameter ≥ 2 cm were identified with the help of plant taxonomists. Manuals and Floras were also used in the identification [19-21]. Nomenclature was in accordance with Dransfield [19], Keng and Keng [20], and King [22]. Voucher specimens were kept at the herbarium of the School of Biological Sciences, Universiti Sains Malaysia, Penang, Malaysia.

Total above-ground biomass (TAGB) of lianas was determined using a TAGB liana allometric equation of Addo-Fordjour et al. [23], which was developed in Malaysia for use in both primary and secondary forests. The liana allometric equation is given as follows:

$$\text{Log}_{10}(\text{TAGB}) = 0.49 + 1.09\ (\log_{10}\text{Diameter})$$

Total above-ground carbon stocks of lianas were then determined from the biomass values estimated by the above mentioned liana allometric equation. Carbon content of lianas in the current study was taken to be 50 % of their biomass [24-25]. Rattans were excluded from carbon stock determination in the current study because of their low numbers and the absence of allometric equations for them.

Data analyses

Due to differences in liana abundance in the forest management plots, rarefaction analysis was conducted to correct for the bias associated with the differences by estimating species richness for a standardised abundance among the plots. Rarefaction analysis was run using the software Estimate S [26]. However, sometimes the differences in the number of individuals sampled may be due to real and meaningfully biological patterns in nature [27], and therefore observed species richness values would be real and unbiased. For this reason, observed species richness was also determined for the plots.

Exponential Shannon diversity index (herein after referred to as Shannon diversity) of lianas was computed for each plot using the following formula [28]:

$$Exp(H') = Exp(-\sum_{i=1}^{s} pi \ln pi)$$

where,
Exp = exponent, *pi* = proportion of the ith species; In *pi* = natural log of *pi*

Liana species richness (observed and rarefied species richness) and Shannon diversity were used as the measures of liana community diversity in this study.

Dominance of liana species was determined by computing their importance value index (IVI). Liana species dominance patterns in the forest types were determined by ordinating the forest management regimes and liana species using the IVI of the species in principal component analysis (PCA) [29-31]. The IVI of the species was calculated according to the equation of Cottam and Curtis [32] which is indicated as follows: IVI = RD + RF + RBA
Where RD = relative density; RF = relative frequency; RBA = relative basal area

Similarity of liana species composition between the forest management regimes was calculated using the Sørensen similarity index, *S* indicated as follows [33]:

$$S = 2C/(a+b)$$

where *C* = number of species common to the two forests, *a* = number of species in forest A, *b* = number of species in forest B.

Differences in liana species richness (observed and rarefied) and stem basal area, and liana above-ground carbon stock between the forest management regimes were tested with one-way ANOVA. Tukey HSD comparison tests were conducted to determine differences of means among the forest management regimes. The data were checked for their compliance with the assumptions underlying ANOVA. The basal area data was $\log_{10}$ transformed to meet ANOVA assumptions of homogeneity of variance and normal distribution. All analyses were conducted with the GenStat software (11th edition) (VSN International Ltd, Hemel Hempstead, UK) at a significance level of 5 %.

Results

Liana diversity

A total of 45 liana species belonging to 27 genera and 15 families was identified in the forest management systems (Appendix 1). More liana species were unique to the untreated forest (8 species) in relation to the MUS-19 (5 species) and MUS-42 (4 species) treated forests. Annonaceae and Fabaceae constituted the most species rich families in the untreated forest (21.9 and 12.5 %, respectively), MUS-42 treated forest (18.5 and 29.6 %, respectively) and MUS-19 treated forest (16 and 36 %, respectively).

Mean observed liana species richness and Shannon diversity per plot were significantly higher in the untreated forest than in the MUS-19 treated forest (Table 1; p = 0.020 and 0.015, respectively). However, mean liana species richness and Shannon diversity per plot were similar between the rest of the forest pairs (p > 0.05). Rarefied species richness of lianas was significantly higher in the untreated forest than in the MUS-19 and MUS-42 treated forests (Table 1; p = 0.001), although it was similar in the two treated forests (p > 0.05). Similarity coefficient of liana species composition was highest between the MUS-42 treated and untreated forests (S = 0.64), and least between MUS-19 treated and untreated forests (S = 0.54) (Table 2).

Liana community structure

Mean liana basal area per plot decreased significantly in the MUS-19 treated forest in relation to the untreated and MUS-42 treated forests (p = 0.002).

Table 1 Mean (± SE) values of liana diversity, basal area and carbon stock in the three forest management regimes in the Bukit Panchor Forest Reserve, Malaysia. Mean values with different letters in the same row are significantly different (p < 0.05) according to Tukey HSD comparison test (one-way ANOVA).

Liana parameter	Forest management regime		
	MUS-19	MUS-42	UF
Observed species richness/plot	$6.60^{a} \pm 0.99$	$8.60^{ab} \pm 0.64$	$10.50^{b} \pm 1.02$
Rarefied species richness/plot	$22.40^{a} \pm 2.29$	$24.70^{a} \pm 2.46$	$35.10^{b} \pm 3.27$
Exponential Shannon diversity/plot	$5.92^{a} \pm 0.11$	$7.32^{ab} \pm 0.11$	$9.73^{b} \pm 0.21$
Basal area/plot (cm^2)	$343.20^{a} \pm 35.60$	$770.80^{b} \pm 57.40$	$781.50^{b} \pm 89.60$
Carbon stock/plot (Mg)	$0.12^{a} \pm 0.06$	$0.22^{b} \pm 0.05$	$0.24^{b} \pm 0.05$

19 years old MUS treated forest: MUS-19, 42 years old MUS treated forest: MUS-42, untreated forest: UF

Liana flora in the three management regimes were dominated by *Uncaria sclerophylla* (IVI = 38.24, 37.38 and 42.84 in the MUS-19 treated, MUS-42 treated and untreated forests, respectively) (Appendix 1). Additionally, *Artabotrys crassifolius* (IVI = 16.42, 27.09 and 35.33) and *Bauhinia bidentata* (IVI = 27.23, 16.55 and 14.38) also showed reasonably high dominance in the MUS-19 treated, MUS-42 treated and untreated forests, respectively. *Dalbergia rostrata* (IVI = 19.65, 4.55 and 3.00, respectively), *Tinomiscium petiolare* (24.13, 4.60 and 0, respectively) and *Willughbeia angustifolia* (18.7, 6.05 and 0, respectively) showed a progressive decrease in IVI from the untreated forest through the MUS-42 treated forest to the MUS-19 treated forest. On the other hand, *Spatholobus* sp. exhibited a reverse trend (2.74, 9.06 and 31.63 in the MUS-19 treated, MUS-42 treated and untreated forests, respectively). The five most dominant liana species in the untreated forest were *U*. *sclerophylla* (42.84), *B*. *bidentata* (27.23), *T*. *petiolare* (24.13), *Agelaea borneensis* (22.59) and *D*. *rostrata* (19.65). Together, these species accounted for 45.5 % of the important value index in the untreated forest. In the MUS-42 treated forest the five most dominant species included *U*. *sclerophylla* (37.38), *A*. *crassifolius* (35.33), *Teteracera indica* (23.23), *Mitrella kentii* (22.87) and *Agelaea macrophylla* (20.21), which collectively contributed 46.3 % of the important value index. Finally, *U*. *sclerophylla* (38.24), *M*. *kentii* (33.66), *Spatholobus* sp. (31.63), *A*. *crassifolius* (27.09) and *Caesalpinia parviflora* (25.43) constituted the five most dominant species in the MUS-19 treated forest. Collectively, these species made up 52 % of the IVI in the MUS-19 treated forest.

Table 2 Similarity coefficients of liana species composition among the different forest management regimes

Management regime	MUS-19	MUS-42	UF
MUS-19	-	0.60	0.54
MUS-42		-	0.64
UF			-

19 years old MUS treated forest: MUS-19, 42 years old MUS treated forest: MUS-42, untreated forest: UF

The first two axes of the PCA explained 94 % of the variation in liana species IVI, and clearly differentiated the three forest management regimes (Fig. 2). In each forest management regime, some liana species were grouped into a cluster that was unique to that forest management regime.

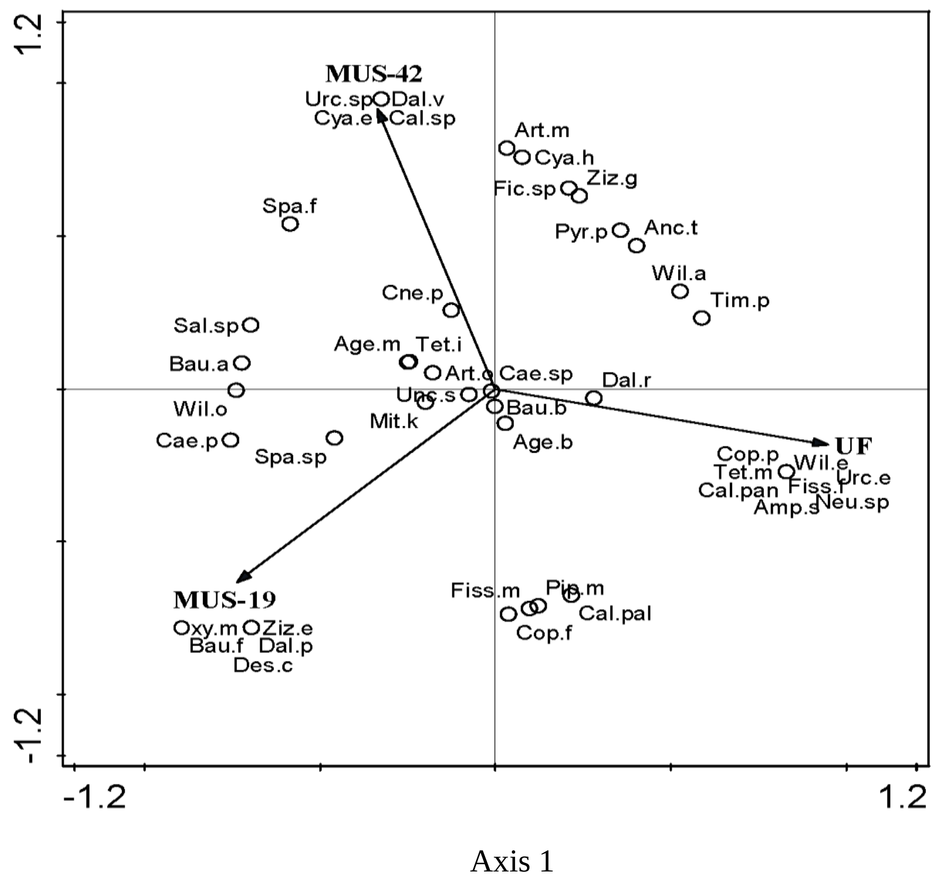

Fig. 2 Projection of forest management regimes (19 years old MUS treated forest: MUS-19, 42 years old MUS treated forest: MUS-42, untreated forest: UF) and liana species on the first two axes defined by the importance value index. The names of the species are represented by their initials.

Liana above-ground carbon stocks

The total amount of above-ground carbon stored by lianas was highest in the untreated forest (1.49 MgC/ha) followed by the MUS-42 treated forest (1.39 MgC/ha) and then the MUS-19 treated forest (0.77 MgC/ha). The mean liana carbon stock per plot decreased significantly in the MUS-19 treated forest compared to the MUS-42 treated and untreated forests (Table 1; $p = 0.004$). However, the mean amount of carbon stock of lianas was comparable in the MUS-42 treated and untreated forests ($p > 0.05$).

The amount of carbon stored by some lianas varied among the forest management regimes (Appendix 2). The carbon stock of *A. borneensis*, *B. bidentata*, *D. rostrata* and *U. sclerophylla* decreased from the MUS-19 forest through to the untreated forest whereas *C. parviflora* and *Spatholobus* sp. showed an opposite trend. The carbon stock of some of the species such as *A. macrophylla*, *A. crassifolius*, *Cnestis pallala*, *Spatholobus ferrugineus* and *T. indica* peaked in the MUS-42 forest. The five species with the greatest carbon storage in the untreated forest were *U. sclerophylla*, *B. bidentata*, *A. borneensis*, *T. petiolare* and *Willughbeia angustifolia*. These species collectively accounted for about 50 % of the above ground carbon stock of lianas in the untreated forest. The most important species in the MUS-42 treated forest in terms of carbon storage were *U. sclerophylla*, *A. crassifolius, M. kentii*, *T. indica* and *Bauhinia bidentata*, which together accounted for 51 % of the total above ground carbon stock of lianas in the MUS-42 forest. In the MUS-19 treated forest, *M. kentii*, *U. sclerophylla*, *Spatholobus* sp., *A. crassifolius* and *C. parviflora* contributed most to total above-ground carbon stock of lianas. The total contribution of these species in the MUS-19 treated forest was 60 %.

Discussion

The patterns of liana diversity observed among the three forest management regimes in the current study indicated that liana cutting had adverse impact on liana diversity, and that the impact was present in the medium term. However, liana diversity was able to recover in relation to the untreated forest in the long term. The liana diversity recovery ability in the long term (i.e. in the MUS-42 treated forest) is consistent with the work of Gardette [14] who reported that liana species diversity in an MUS treated forest was similar to that in an untreated forest in the long term. The findings of the present study thus demonstrate that liana diversity recovery was dependent on the time span of the silvicultural activity. Generally, the similarity of liana species composition among the forest management regimes was moderate, indicating that liana cutting probably had an impact on liana species composition in the study area. Again, liana species composition was more similar between the MUS-42 treated and untreated forests than between the MUS-19 and untreated forests, suggesting that liana composition recovery was somehow dependent on the time span of the treatments. The variation in liana species composition among the different forest management regimes, and different time span of the same regime, suggests the presence of ecological differences among liana species of the forests [34]. Annonaceae and Fabaceae consistently contributed the highest number of species in all the forest management systems. This indicates that these liana families had high recovery ability from liana cutting. This is not surprising because several studies have reported that these families dominate in both human disturbed and primary forests in tropical regions [1, 16, 29].

The current study indicated that the Malayan Uniform System caused a significant adverse impact on liana stem basal area 19 years after liana cutting. This finding is supported by other studies that reported significant reductions in liana basal area after different periods of liana cutting [12-13, 15]. However, in the long term (after four decades), liana basal area was restored to pre-treatment level within the MUS-42 treated forest. This finding differs from that of Foli & Pinard [13] who reported that liana basal area could not be restored to pre-treatment level 40 years after liana cutting. Thus, the findings of the current study and previous ones [12-13, 15] indicate that liana cutting as a silvicultural tool can cause a significant reduction in liana basal area, but this may perhaps be restored to pre-treatment level in the long term.

The present study demonstrated that liana cutting enhanced the dominance of some liana species and vice versa, in the treated forests. For instance, *M*. *kentii* and *A*. *macrophylla* showed higher dominance in the treated forests than in the untreated forest. On the other hand, the dominance of *A*. *borneensis*, *T*. *petiolare* and *D*. *rostrata* was lower in the treated forests than in the untreated forest. In addition, certain liana species were restricted to only one of the forest management regimes, and therefore showed dominance in only those forests. Thus, the application of the Malayan Uniform System in the treated forests created variations in liana species dominance among the forest management regimes. Those variations resulted in a clear separation of the three forest management regimes as depicted in the PCA ordination diagram. Since IVI of the liana species reflects their species composition, abundance, frequency and basal area, the patterns observed in the PCA ordination diagram indicate differences in the above-mentioned liana community properties among the forest management regimes. The PCA ordination further revealed that on the whole, liana species dominance patterns in the untreated forest were more similar to the patterns recorded in the MUS-42 treated forest than to the patterns observed in the MUS-19 treated forest. This implies that the time span of the silvicultural intervention was an important factor in liana species dominance recovery within the treated forests.

This study also revealed a significant impact of the silvicultural intervention on the amounts of total above-ground carbon sequestered by lianas in the various forest management regimes. Although there was about 50 % significant reduction in total above-ground carbon stock of lianas in the MUS-19 treated forest compared to the untreated forest, the amount of total above-ground carbon sequestered by lianas in the MUS-42 treated forest was similar to that in the untreated forest. Therefore, the above-ground carbon storage capacity of lianas, which was completely lost as a result of clear liana cutting in the treated forests, was fully restored after four decades but partially restored in 19 years time. The individual liana species differed in their levels of contribution to total above-ground carbon stocks of lianas in the forests. For instance, the amounts of above-ground carbon stored by species such as *A*. *borneensis*, *B*. *bidentata*, *D*. *rostrata* and *U*. *sclerophylla* decreased with increasing time span of the forest management regimes, showing that the growth of these species was possibly favoured by the silvicultural treatment adopted in the treated forests. On the other hand, other species, such as *C*. *parviflora* and *Spatholobus* sp., exhibited increasing carbon stock with increasing time span of the forest management regimes. Within the same forest type, different liana species responded differently to the silvicultural treatment, resulting in some of the species storing higher amounts of above-ground carbon than others. This explains why the five topmost liana species contributed at least 50 % of total above-ground liana carbon stock in the forest management regimes. Generally, these findings demonstrate that liana cutting had differential influence on the above-ground carbon storage of different liana species.

Implications for conservation

Previous assessments of the impacts of liana cutting on liana assemblages indicated that liana cutting resulted in significant reductions in liana abundance [8, 15] and infestation [8] in short to medium term within some tropical forests. Although liana cutting reduces liana abundance and infestation, concerns have been expressed about the potential effects of this silvicultural activity on liana composition and diversity [8, 15, 35]. The findings of the present study confirmed these concerns, as liana cutting had adverse impacts on liana composition and diversity. Although in the long term the MUS-42 treated forest was able to recover in liana diversity (observed species richness and Shannon diversity), the composition and dominance patterns of its liana species were different from those in the untreated forest. Considering the numerous ecological importance of lianas in tropical forests, liana species loss and composition changes will affect tropical forest functioning and dynamics [15]. For example, changes in liana species diversity and composition in treated forests may affect the diet of tree dispersers, especially in times of scarcity when most trees do not flower or fruit (but lianas do). This phenomenon may have a negative influence on tree seed dispersal, and hence forest regeneration.

In view of the negative impacts of blanket liana cutting on liana diversity and composition in the treated forests, we recommend that selective liana cutting be used in controlling liana numbers in tropical forests, as some authors have suggested in previous studies [e.g. 8, 35]. The use of selective liana cutting would ensure that liana abundance is controlled or reduced while their diversity is maintained for proper functioning of tropical forest ecosystems. Selective liana cutting could be targeted at the most abundant liana species in tropical forests. Controlling the numbers of the most abundant liana species in tropical forests would improve species evenness, and enhance the diversity of lianas in silviculturally treated forests. Furthermore, selective liana cutting may also be targeted at particular trees, whereby lianas are cut from only heavily infested trees.

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Appendix 1 Liana species dominance in the MUS treated forests and untreated forest in the Bukit Panchor Forest Reserve (RD = Relative density, RF = Relative frequency, RBA = Relative basal area, IVI = Importance value index).

Species	MUS-19				MUS-42				UF			
	RD	RF	RBA	IVI	RD	RF	RBA	IVI	RD	RF	RBA	IVI
ANCISTROCLADACEAE												
Ancistrocladus tectorius	-	-	-	-	1.54	1.16	0.84	3.54	1.50	1.96	0.77	4.23
ANNONACEAE												
Artabotrys maingayi	-	-	-	-	4.10	4.65	2.50	11.25	0.50	0.98	0.41	1.89
Artabotrys crassifolius	6.48	9.09	11.52	27.09	10.77	6.98	17.58	35.33	5.50	5.88	5.04	16.42
Cyathostemma excelsum	-	-	-	-	1.54	1.16	0.85	3.55	1.00	0.98	0.38	2.36
Cyathostemma hookeri	-	-	-	-	4.10	5.81	1.37	11.28	-	-	-	-
Desmos chinensis	0.93	1.52	0.91	3.36	-	-	-	-	-	-	-	-
Fissistigma fulgens	-	-	-	-	-	-	-	-	4.00	3.92	6.63	14.55
Fissistigma manubriatum	2.78	1.52	1.60	5.90	-	-	-	-	1.50	2.94	1.25	5.69
Mitrella kentii	12.04	6.06	15.56	33.66	9.74	6.98	6.15	22.87	5.50	5.88	3.59	14.97
Pyramidanthe prismatica	-	-	-	-	2.05	3.49	9.29	14.83	1.50	1.96	10.29	13.75
APOCYNACEAE												
Urceola elastica	-	-	-	-	-	-	-	-	1.00	0.98	0.30	2.28
Urceola sp.	-	-	-	-	2.05	4.65	0.59	7.29	-	-	-	-
Willughbeia angustifolia	-	-	-	-	1.03	2.33	2.69	6.05	5.50	5.88	7.32	18.70
Willughbeia edulis	-	-	-	-	-	-	-	-	0.50	0.98	0.41	1.89
Willughbeia oblonga	1.85	3.03	0.91	5.79	1.03	1.16	1.27	3.46	-	-	-	-
CELASTRACEAE												
Salacia sp.	0.93	1.52	0.19	2.64	1.03	1.16	0.98	3.17	-	-	-	-
CONNARACEAE												
Agelaea borneensis	5.56	6.06	1.37	12.99	2.56	4.65	1.53	8.74	10.50	7.84	4.25	22.59
Agelaea macrophylla	5.56	7.58	1.53	14.67	8.21	9.30	2.70	20.21	1.50	2.94	2.27	6.71
Cnestis palala	0.93	1.52	0.45	2.90	1.54	2.33	4.25	8.12	1.00	1.96	0.46	3.42

19 years old MUS treated forest: MUS-19, 42 years old MUS treated forest: MUS-42, untreated forest: UF

Appendix 1 *Cont'd.*

Species	MUS-19				MUS-42				UF			
	RD	RF	RBA	IVI	RD	RF	RBA	IVI	RD	RF	RBA	IVI
CONVOLVULACEAE												
Neuropeltis sp.	-	-	-	-	-	-	-	-	1.00	0.98	1.45	3.43
DILLENIACEAE												
Tetracera indica	6.48	9.09	1.52	17.09	9.23	9.30	4.70	23.23	2.50	3.92	0.94	7.36
Tetracera macrophylla	-	-	-	-	-	-	-	-	1.00	1.96	0.46	3.42
FABACEAE												
Bauhinia audax	0.93	1.52	10.42	12.87	1.03	2.33	6.41	9.77	-	-	-	-
Bauhinia bidentata	4.63	6.06	5.86	16.55	5.13	4.65	4.60	14.38	8.50	6.86	11.87	27.23
Bauhinia ferruginea	0.93	1.52	3.55	6.00	-	-	-	-	-	-	-	-
Caesalpinia parviflora	4.63	6.06	14.74	25.43	1.03	1.16	3.67	5.86	-	-	-	-
Caesalpinia sp.	2.78	1.52	1.83	6.13	2.56	2.33	2.00	6.89	3.00	2.94	3.24	9.18
Dalbergia parviflora	1.85	3.03	0.88	5.76	-	-	-	-	-	-	-	-
Dalbergia rostrata	0.93	1.52	0.55	3.00	1.03	1.16	2.36	4.55	6.00	6.86	6.79	19.65
Dalbergia velutina	-	-	-	-	0.51	1.16	0.10	1.77	-	-	-	-
Spatholobus ferrugineus	0.93	1.52	0.29	2.74	3.08	4.65	6.74	14.47	-	-	-	-
Spatholobus sp.	14.81	7.58	9.24	31.63	3.59	3.49	1.98	9.06	1.00	0.98	0.76	2.74
MENISPERMACEAE												
Tinomiscium petiolare	-	-	-	-	1.53	2.33	0.74	4.60	9.50	7.84	6.79	24.13
MORACEAE												
Ficus sp.	-	-	-	-	1.03	1.16	0.84	3.03	0.50	0.98	0.11	1.59
PALMAE												
Calamus palustris	0.93	1.52	0.07	2.52	-	-	-	-	1.50	1.96	0.26	3.72
Calamus pandanosmus	-	-	-	-	-	-	-	-	1.50	2.94	0.35	4.79
Calamus sp.	-	-	-	-	1.03	1.16	0.13	2.32	-	-	-	-

19 years old MUS treated forest: MUS-19, 42 years old MUS treated forest: MUS-42, untreated forest: UF

Appendix 1 *Cont'd.*

	MUS-19				MUS-42				UF			
Name	RD	RF	RBA	IVI	RD	RF	RBA	IVI	RD	RF	RBA	IVI
PIPERACEAE												
Piper maingayi	0.93	1.52	0.14	2.59	-	-	-	-	1.00	0.98	0.74	2.72
RHAMNACEAE												
Ziziphus elegans	0.93	1.52	1.28	3.73	-	-	-	-	-	-	-	-
Ziziphus grewioides	-	-	-	-	1.03	1.16	0.82	3.01	0.50	0.98	0.29	1.77
RUBIACEAE												
Coptosapelta flavescens	3.70	4.55	2.06	10.31	-	-	-	-	3.00	2.94	1.56	7.50
Coptosapelta parviflora	-	-	-	-	-	-	-	-	1.00	1.96	0.35	3.31
Oxyceros curtisii	2.78	3.03	0.72	6.53	-	-	-	-	-	-	-	-
Uncaria sclerophylla	14.81	10.60	12.83	38.24	16.92	8.14	12.32	37.38	15.00	8.82	19.02	42.84
VITACEAE												
Ampelocissus spicigera	-	-	-	-	-	-	-	-	2.50	0.98	1.65	5.13

19 years old MUS treated forest: MUS-19, 42 years old MUS treated forest: MUS-42, untreated forest: UF

Appendix 2 Amount of above-ground carbon stored by liana species in the 19 and 42 years old MUS (MUS-19 and MUS-42, respectively) treated forests, and untreated forest (UF) in the Bukit Panchor Forest Reserve, Malaysia

Species	Carbon stock per species (MgC)		
	MUS-19	MUS-42	UF
Agelaea borneensis (Hook. f.) Merr.	0.036	0.046	0.183
Agelaea macrophylla (Zoll.) Leenh.	0.039	0.109	0.044
Ampelocissus spicigera Planch.	-	-	0.052
Ancistrocladus tectorius (Lour.) Merr.	-	0.020	0.028
Artabotrys maingayi Hook.f. & Thomson	-	0.074	0.015
Artabotrys crassifolius Hook.f. & Thomson	0.122	0.340	0.135
Bauhinia audax (de Wit) G. Cusset	0.049	0.068	-
Bauhinia bidentata Jack	0.072	0.114	0.242
Bauhinia ferruginea Roxb.	0.027	-	-
Caesalpinia parviflora (Prain ex King) Prain	0.112	0.047	-
Caesalpinia sp.	0.031	0.056	0.074
Cnestis palala (Lour.) Merr.	0.009	0.064	0.012
Coptosapelta flavescens Korth.	0.037	-	0.054
Coptosapelta parviflora Ridl.	-	-	0.018
Cyathostemma excelsum (Hook.f. & Thomson) J.Sinclair	-	0.028	0.018
Cyathostemma hookeri King	-	0.057	-
Dalbergia parviflora Roxb.	0.017	-	-
Dalbergia rostrata Hassk.	0.010	0.041	0.155
Dalbergia velutina Benth.	-	0.005	-
Desmos chinensis Lour.	0.013	-	-
Ficus sp.	-	0.022	0.014
Fissistigma fulgens (Hook.f. & Thomson) Merr.	-	-	0.121
Fissistigma manubriatum (Hook. f. & Thoms.) Merr.	0.029	-	0.044
Mitrella kentii Miq.	0.193	0.183	0.110
Neuropeltis sp.	-	-	0.033
Oxyceros curtisii (King & Gamble) K.M.Wong	0.018	-	-
Piper maingayi Hook.f.	0.005	-	0.025
Pyramidanthe prismatica Merr.	-	0.098	0.099
Salacia sp.	0.006	0.026	-
Spatholobus ferrugineus (Zoll. & Moritzi) Benth.	0.007	0.113	-
Spatholobus sp.	0.158	0.065	0.025
Tetracera indica (Christm. & Panz.) Merr.	0.041	0.153	0.048
Tetracera macrophylla A. Chev.	-	-	0.020
Tinomiscium petiolare Hook.f. & Thoms.	-	0.025	0.174
Uncaria sclerophylla (Hunter.) Roxb.	0.164	0.347	0.409
Urceola elastica Roxb.	-	-	0.020
Urceola sp.	-	0.025	-
Willughbeia angustifolia (Miq.) Markgr.	-	0.044	0.171
Willughbeia edulis Roxb.	-	-	0.018
Willughbeia oblonga Hook.f.	0.017	0.027	-
Ziziphus elegans Wall.	0.016	-	-
Ziziphus grewioides (Warb.) L.M.Perry	-	0.023	0.030