Managing a tropical rainforest for timber, carbon storage and tree diversity

MARCO BOSCOLO[1] and JOSEPH BUONGIORNO[2]

[1] *Center for Tropical Forest Science, Smithsonian Tropical Research Institute, Harvard Institute for International Development, Harvard University, USA*

[2] *Department of Forestry, University of Wisconsin-Madison, 1630 Linden Drive, Madison, WI 53706, USA*

SUMMARY

Natural tropical forests can provide income, serve as carbon sinks, and be reservoirs of biological diversity. Management for these diverse goals can be guided by optimisation models that seek the best cutting regime, given different constraints and objective functions. The results for a lowland tropical rainforest in peninsular Malaysia show that to maximise a sustainable income one would cut all commercial trees of 30 cm and above every 20 years. All non-commercial trees would remain, and thus contribute to diversity and carbon storage. The best policy to maximise carbon storage was not to cut at all. This also maximised diversity, by leading to a forest in its climax state. However, this policy had a very high opportunity cost. Compromise policies were obtained, for example by maximising carbon storage, or diversity, while getting a competitive rate of return on the capital in trees. Findings suggest that increasing carbon storage and tree diversity can be attained only at significant costs in terms of foregone income. For example, increasing carbon storage beyond the income-maximising amount would cost $47/ton.

Keywords: tropical forest management, soil rent, carbon storage, diversity, linear programming.

INTRODUCTION

A recurrent theme in the tropical forestry literature is the need to manage tropical forests for multiple-uses because they provide many valuable goods and services (Peters *et al.* 1989, Panayotou and Ashton 1992). Of particular concern are values such as timber, often an important source of revenues for governments and local communities, carbon storage, for its relevance for global climate change, and biodiversity, linked to the potential development of markets such as eco-tourism, bioprospecting, etc. (de Montalembert 1991). While the need to account for these values in forestry decisions is widely recognised, its implications for forest management are not clear. How should current management schemes be modified to take other non-timber values into account? What are the characteristics of sustainable forest use?

Previous studies that have looked at the implications of carbon and biodiversity values for forest management have generally considered these ecological functions individually, focussed on temperate forests, or addressed specific management issues. For example, many studies that explored the implications of carbon values for forest management focussed on the optimal rotation age problem (e.g. Binkley and van Kooten 1994, Plantinga and Birdsey 1994, Englin and Callaway 1995)[1]. More relevant to tropical natural forests are studies that explored the effects of alternative optimal cutting policies on the structure and composition of uneven-aged forest stands. Buongiorno *et al.* (1994), for example, developed a matrix model for northern hardwood forests to quantify the trade-offs between income and tree size diversity. They found that the latter can be increased by lengthening the felling cycle and lowering harvest intensities, while maximum diversity was achieved by not cutting the forest at all. Buongiorno *et al.* (1995) developed a density-dependent, multi-species matrix model for the management of uneven-aged forests in the French Jura. Adopting alternative measures of diversity they identified several optimal steady states, depending on the management criteria and index chosen. Ingram and Buongiorno (1996) presented a model for the management of lowland tropical rainforest in peninsular Malaysia and used it to compare optimal cutting policy with existing management schemes in Malaysia and Indonesia. They also evaluated the impact of different felling cycles and harvest intensities on biodiversity, measured by indexes such as Shannon-Wiener[2] and a maxi-min criterion applied to the minimum number of trees in any species-size class. They

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[1] Exceptions are Hoen and Solberg (1994) and Boscolo *et al.* (1997).

[2] Shannon-Wiener (Zar 1984) measures how evenly a population of individuals is distributed among various categories. The more even the distribution the higher the diversity index. The index takes the lowest value when all individuals are concentrated in one category.

found that the solutions that maximised diversity or the net present value had different harvesting intensities but similar felling cycles. Also, the optimal economic cutting policies had much shorter felling cycles than current practices. We suspect that these results are partly due to the fact that the model did not consider the effect of logging damage on the residual stand, thus exaggerating the productive potential of the forest stand.

In this paper we develop an economic model to quantify the effect of carbon storage and tree diversity on the optimal cutting regime of a lowland tropical rainforest in peninsular Malaysia. Our objectives are: (1) to extend and complement previous models by explicitly recognising logging damage, (2) to assess the productivity of the resource in terms of timber, carbon storage, and tree diversity, and (3) to quantify the relevant trade-offs among these products and services. The decision to model logging damage explicitly was motivated by the evidence that, in tropical settings, it is an important determinant of forest productivity (*cf.* Appanah and Weinland 1990). Carbon storage and tree diversity were considered simultaneously to highlight the differences, and similarities, of their effects on tropical forest management.

The level of analysis for this study is the forest stand managed by selective harvest. In order to obtain results that are independent from initial conditions, while embodying the principle of sustainability, our analyses refer to steady-state scenarios. In a steady state, the forest stand yields the same flow of benefits at each cycle, in perpetuity. While steady states are abstracted ideals, their characteristics provide valuable information on the potential productivity of the resource in terms of timber and the other environmental services, and on the relevant trade-offs. They also provide valuable information for policy making which often compares the desirability of alternative 'equilibrium' or 'sustained yield' states.

THE GROWTH MODEL

The growth model adopted for this study is an extension of the transition matrix model developed originally by Buongiorno and Michie (1980). Boscolo *et al.* (1997) modified it to allow for multiple-species (similarly to Buongiorno *et al.* 1995 and Ingram and Buongiorno 1996). In this model, transition probabilities (growth and mortality parameters) depend on species group and tree size. Ingrowth depends on species composition and stand density. In matrix notation, the growth model is:

$$\mathbf{y}_{t+p} = \mathbf{G}^{p}(\mathbf{y}_t - \mathbf{h}_t - \mathbf{D}\mathbf{h}_t) + \mathbf{c}\sum_{n=0}^{p-1}\mathbf{G}^{n} \qquad (1)$$

where $\mathbf{y}_t = [y_{i,j,t}]$ is a vector containing the number of trees of species group i, size class j alive at time t, $\mathbf{h}_t = [h_{i,j,t}]$ is the number of trees cut, p is the felling cycle, and the matrix $\mathbf{G}$ and vector $\mathbf{c}$ contain the one-year growth parameters. A new feature is the recognition of the stand damage caused by harvesting, especially the breaking of small trees caused by the felling of large ones. This effect is modelled with the damage matrix $\mathbf{D}$. Its elements, $d_{i,j}$, are the number of small (10-20 cm diameter) trees of species i killed by the felling of a tree of size j. Thus, at the end of the felling cycle (at time $t + p$), a stand state will depend on the state of the stand at time t, the extent of the removals (expanded by the damage), the length of the cycle p, and the growth parameters in $\mathbf{G}$ and $\mathbf{c}$.

Growth, mortality and recruitment were calibrated with data from a 50-hectare demographic plot located in peninsular Malaysia as described in Boscolo *et al.* (1997)[1]. The model predicts the evolution of a forest stand with trees grouped in 3 species cohorts (dipterocarps, commercial non-dipterocarps, and non-commercial) and seven, 10 cm wide, size classes. The parameters $d_{i,j}$ were assumed constant, proportional to the composition of a virgin stand, and calibrated by positing that the felling of all commercial trees above 60 cm in a virgin stand would kill half the trees in the smallest size class (Griffin and Caprata 1977, cited in Appanah and Weinland 1990). The elements of the **D** matrix are reported in Table 1.

TABLE 1 *Number of trees of 10-20 cm diameter damaged by the felling of one tree (elements of the **D** matrix)*

Diameter of tree felled (cm)	Number of trees damaged, by species group		
	dipterocarps	other commercial	non-commercial
10-20	0.10	0.20	0.85
20-30	0.13	0.26	1.13
30-40	0.25	0.50	2.12
40-50	0.35	0.71	3.02
50-60	0.48	0.96	4.11
60-70	1.05	2.12	9.05
70+	1.45	2.91	12.42

MANAGEMENT CRITERIA

Carbon storage

As scientific evidence of a relation between climate change and anthropogenic emissions of greenhouse gases (carbon dioxide being the most important one) is increasing, so is the recognition of forests' role as carbon sinks. Forests influence carbon concentration in the atmosphere by assimilating CO_2 through biomass build-up and by releasing it through biomass decay (or fire).

In accounting for carbon sequestration and release that occur at different points in time, carbon flows were discounted like timber products. The assumption is that an earlier sequestration, or storage of carbon, is preferred to a later one, with the rate of time preference measured by the same discount rate used to discount monetary values. As a result, the opportunity cost of sequestered carbon, in terms of timber revenues foregone, is expressed as a price analogous to the price of timber.

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[1] The plot is located in a virgin forest that is subject to periodic and localised modifications of the canopy structure due to natural mortality, windthrows, etc. Model calibration rested on the assumption that growth, mortality and recruitment depend on the stand state, independently of how that state was reached.

The carbon accounted for included both the carbon stored in living tree biomass (above and below ground) and the carbon stored in end uses, such as houses. The carbon stored in living trees had the following expression:

$$PVK_s = \mathbf{k}\,[\mathbf{y}_t + (\sum_{n=t}^{t+p-1} \frac{\mathbf{y}_{n+1} - \mathbf{y}_t}{(1+r)^{n-t+1}} - \mathbf{h}_t - \mathbf{Dh}_t)\,(1+\frac{1}{(1+r)^{p}-1})] \qquad (2)$$

where **k** is a vector of parameters giving the carbon stored per tree, *p* is the felling cycle, and *r* is the discount rate. The vector **k** (Table 2) was derived by multiplying tree volumes by the carbon coefficients of Boscolo *et al.* (1997). Equation (2) states that the carbon stored is the sum of the carbon in the stand before harvest, plus the carbon accumulated by the growing stand during the cutting cycle, minus the discounted carbon harvested every cutting cycle. In a steady state the cycles are identical and equation (2) gives the discounted value of carbon over an infinite horizon. For a positive discount rate, PVK_s is finite.

TABLE 2 *Estimates of tree carbon storage*

Diameter (cm)	Carbon storage (tons/tree)		
	dipterocarps	other commercial	non-commercial
10-20	0.24	0.24	0.27
20-30	0.32	0.32	0.35
30-40	0.60	0.60	0.66
40-50	0.86	0.85	0.94
50-60	1.17	1.15	1.28
60-70	2.58	2.54	2.83
70+	3.54	3.49	3.88

Carbon storage in end uses was quantified as

$$PVK_c = [s\,\mathbf{k}\,(\mathbf{h}_t + \mathbf{Dh}_t)(1 - 1/(1+r)^T)\,]\,[1+\frac{1}{(1+r)^{p}-1}\,] \qquad (3)$$

where *s* is the proportion of the carbon harvested that is stored in end uses for T years, after which it is assumed to be released immediately[1].

Then, total carbon storage is

$$PVK = PVK_s + PVK_c \qquad (4)$$

Tree diversity

Left undisturbed, a natural tropical forest tends to reach a near steady state, generally referred to as the 'climax' (Ricklefs and Schluter 1993, Rosenzweig 1995). There is evidence that such forests are extremely diverse in terms of flora and fauna (Whitmore 1990, Primack and Lovejoy 1993). Accordingly, or if for other reasons the natural tropical 'climax' forest is desirable, it seems appropriate to use it as a standard against which to compare the managed forest. In that spirit, the 'closer' the managed forest is to the climax state, the more diverse it is. An operational definition of this measure of diversity is:

$$DEV = \max\,(DEV_{i,j}) \qquad (5)$$

with $DEV_{i,j} = (y^*_{i,j} - y_{i,j})\,/\,y^*_{i,j}$, where $y^*_{i,j}$ is the number of trees of species *i* and size *j* in the climax state, while $y_{i,j}$ is the number of trees of the managed stand after the cut. For management purposes, one ecological goal could be to keep *DEV*, the maximum relative deviation (among all species and sizes) from the climax distribution, as small as possible.

The effect of this min-max approach is to give special attention to the tree categories that are least represented and that, in a managed stand, will deviate the most from those observed in natural climax stands. As suggested by Buongiorno *et al.* (1995), *DEV* can be used either as the objective function (to be minimised), or as a constraint to limit the maximum deviation in a manner consistent with the concept of a 'safe minimum standard of conservation' (Ciriacy-Wantrup 1968).

Although no study has quantified the effect of altering the structure and composition of a primary tropical rainforest on its overall ability to maintain high levels of biodiversity, several studies hint at the important role that both structure and composition play in niche differentiation. Harrison (1962), for example, defines six communities of birds and mammals based on the canopy level occupied and food requirements. Wells (1971) points to a clear vertical stratification of birds in lowland tropical rainforests in peninsular Malaysia. Thus, birds specialised to live at the top of the canopy (e.g. hornbills, barbets, etc.) are the most likely to be affected by the elimination of large trees that results from conventional logging (Whitmore and Burnham 1984)[5]. However, also sub-canopy birds have been found to be reduced by the alteration of understorey conditions that result from conventional logging (S. Kotagama, pers. comm.).

Timber revenues

The soil expectation value (*SEV*) — the present value of future timber revenues, net of all costs (including the cost of the investment in the growing stock) — was chosen to judge the economic desirability of alternative steady states. This is the implicit value of the land used in this kind of silviculture, and it can be compared with the soil rent brought about by any alternative land use, in forestry or otherwise. Thus, in a steady state, the economic harvest ($\mathbf{h}_t$) and the growing stock ($\mathbf{y}_t$) should be chosen so that the maximum SEV is attained. The SEV was defined as:

$$SEV = \frac{\mathbf{v}'\mathbf{h}_t - F}{(1+r)^{p}-1} - \mathbf{v}'(\mathbf{y}_t - \mathbf{h}_t - \mathbf{Dh}_t) \qquad (6)$$

where $\mathbf{v} = [v_{i,j}]$ and $v_{i,j}$ is the 'stumpage value' of a standing tree of species *i* and size *j* (see Table 3), *F* is the harvesting fixed cost per ha, independent of volume cut (infrastructure and

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[4] In this application, *s* was set at 60% (Awang Noor, personal communication), and *T* was assumed to be 80 years.

[5] Of course, biodiversity encompasses all forms of life, not only birds and mammals. The quantification of the effect of harvesting on biodiversity, defined according to alternative indicators (e.g. Magurran 1988, Weitzman 1992), is a research area that certainly deserves more attention. Yet, in tropical settings, it is currently limited by data availability.

administrative costs, estimated at $250 per ha), and *r* is the interest rate. r was set at 6% per year, the rate of return (net of inflation) of alternative investment opportunities in Malaysia (Ingram and Buongiorno 1996).

TABLE 3 *Estimates of tree volume and stumpage value (1991)*

Diameter (cm)	Volume/tree (m^3)	Stumpage value ($/tree)		
		dipterocarps	other commercial	non-commercial
10-20	0.368	-1	-1	-1
20-30	0.487	-1	-1	-1
30-40	0.914	33	17	-2
40-50	1.302	65	38	-3
50-60	1.772	113	69	-4
60-70	3.905	248	153	-8
70+	5.361	340	210	-11

Notes: Estimates of logging costs include variable costs ($28.3 per m^3) and fixed costs (F) = $250 per ha. For the felling of non-merchantable trees we estimated a cost of $2 per m^3 (based on ITTO and FRIM 1994).

NON-TIMBER VALUES AND DESIRABLE STEADY STATES

Maximising carbon storage and tree diversity

The harvest and growing stock that maximise carbon storage in the steady state, without other constraints, was found by solving the following linear programming problem:

$$\max_{\{v_t, h_t, p\}} \ PVK \tag{7}$$

subject to:

$$\mathbf{y}_{t+p} = \mathbf{G}^p (\mathbf{y}_t - \mathbf{h}_t - \mathbf{Dh}_t) + \mathbf{c} \sum_{n=0}^{p-1} \mathbf{G}^n \tag{8}$$

$$\mathbf{h}_t + \mathbf{Dh}_t \le \mathbf{y}_t \tag{9}$$

$$\mathbf{y}_{t+p} = \mathbf{y}_t \tag{10}$$

$$\mathbf{y}_t, \mathbf{h}_t \ge 0 \tag{11}$$

Constraint (8) is the growth equation (1). Constraint (9) recognizes that removals and logging damage cannot exceed the current stand. Constraint (10) is the steady state constraint and ensures that the flows of timber and non-timber goods are maintained in perpetuity. For a given cutting cycle, *p*, equations (7) to (10) give the optimum steady-state harvest **h** and growing stock **y**. Then, solving for different cutting cycle leads to the global optimum.

The results are in Table 4. The maximum amount of carbon the stand could store would be 214 tons ha^{-1} of carbon, with a basal area of 25 m^2ha^{-1}. To attain this objective, no tree would be cut. This was found to be true regardless of the cutting cycle, *p*. Therefore, the steady-state maximising carbon storage per ha is the climax state of the natural tropical forest found by Boscolo *et al.* (1997). Thus, the diversity criterion is $DEV \approx 0$, its optimum value, indicating that the carbon and diversity maximisation (as defined here) are obtained simultaneously by letting nature take its course. However, this complete preservation has a very high cost: $4,127 per ha in terms of timber capital held in the climax stand, and even more in terms of opportunity cost relative to what could be obtained by maximizing SEV, as shown next.

TABLE 4 *Steady state that maximises carbon storage*

Diameter (cm)	Number of trees/ha	Cut	Number of trees/ha	Cut	Number of trees/ha	Cut
	dipterocarps		other commercial		non-commercial	
10-20	30.2	0.0	59.4	0.0	260.8	0.0
20-30	11.6	0.0	17.4	0.0	62.6	0.0
30-40	5.4	0.0	8.9	0.0	19.7	0.0
40-50	3.9	0.0	4.6	0.0	7.4	0.0
50-60	3.5	0.0	3.4	0.0	2.9	0.0
60-70	2.6	0.0	1.7	0.0	1.0	0.0
70+	6.0	0.0	1.6	0.0	1.0	0.0

Felling cycle	-
Total number of trees:	515 trees per ha
Basal area:	24.8 $m^2 ha^{-1}$
Present value of harvest	$0 per ha
Value of residual stock	$4,127 per ha
Carbon storage in end uses (PVK_e):	0 tons per ha
Carbon storage in the stand (PVK_s):	214 tons per ha
Max. deviation from climax state (DEV):	≈ 0%

Note: This species/diameter distribution is almost identical to the climax distribution.

Maximising timber revenues

An alternative steady state, one that maximises the *SEV* generated by timber revenues alone, without any other constraint except sustainability, was obtained by solving the following linear programming problem:

$$\max_{\{v_t, h_t, p\}} \ SEV \tag{12}$$

subject to constraints (8), (9), (10), and (11) above.

Solving for different values of *p* led to the results in Table 5. The maximum *SEV* becomes $702 per ha. This is the implicit value of the land (soil rent) used in this kind of forestry. The carbon storage is 21% lower, at 170 tons ha^{-1}, than that of an unmanaged virgin stand (Table 4). Only a small fraction (< 3%) of this total storage consists of end uses. The cutting policy that maximises soil rent prescribes the felling of all commercial trees (dipterocarp and non-dipterocarps with DBH above 30 cm) and a cutting cycle of 20 years[1]. Instead,

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[1] The length of the felling cycle is affected by the estimated fixed costs (F – see, e.g., Buongiorno and Michie 1980). Should fixed costs exceed the assumed $250 per ha, longer felling cycles may be desirable. Malaysian State governments, for example, levy 'premia', or per unit area charges, in awarding timber concessions. In this regard, our analysis takes the government perspective, not that of a private concessionaire.

none of the non-commercial trees are cut. Indeed, it would cost more to cut them than they are worth. Leaving them standing increases the soil expectation value. The suggestion to fell all commercial trees is almost identical to the old Malayan Uniform System (MUS) (Wyatt-Smith 1963), with the important difference that no non-commercial tree is cut (no poison-girdling or 'liberation thinning'), and the cutting cycle is much shorter. Thus, contrary to intuitive silviculture, it is not profitable to eliminate the non-commercial trees in natural stands, and thus the financial and ecological objectives are not in total conflict. Of course, such selective felling alters the composition of the stand. The basal area of non-commercial species increases from 50% of the total basal area (as found in the virgin stand) to more than 70% under the SEV maximising regime[7].

TABLE 5 *Steady state that maximises soil expectation value*

Diameter (cm)	Number of trees/ha	Cut	Number of trees/ha	Cut	Number of trees/ha	Cut
	dipterocarps		other commercial		non-commercial	
10-20	39.3	0.0	75.7	0.0	290.0	0.0
20-30	14.6	0.0	21.1	0.0	66.2	0.0
30-40	5.4	5.4	6.3	6.3	20.6	0.0
40-50	2.2	2.2	1.3	1.3	7.7	0.0
50-60	0.6	0.6	0.2	0.2	3.0	0.0
60-70	0.1	0.1	0.0	0.0	1.0	0.0
70+	0.0	0.0	0.0	0.0	1.0	0.0

Felling cycle	20 years
Total number of trees (before cut):	556 trees per ha
Total number of trees (after cut):	487 trees per ha
Basal area (before cut):	18.8 m^2 ha^{-1}
Basal area (after cut):	16.0 m^2 ha^{-1}
Residual stand damaged [(a)]:	10%
Extracted volume:	0.85 m^3 ha^{-1} yr^{-1}
Harvest revenues (net):	$338 per ha
Present value of harvests:	$153 per ha
Value of residual stock [(b)]:	-$549 per ha
Soil expectation value (*SEV*):	$702 per ha
Carbon storage in end uses (PVK_e):	5 tons per ha
Carbon storage in the stand (PVK_s):	165 tons per ha
Max. deviation from climax, after cut (*DEV*):	100%

Notes: a. In the steady state logging damage is sensibly smaller than the damage caused during logging of a virgin stand. This is because fewer trees are extracted and of smaller dimensions.
b. The value of the residual stock is negative because felling non-commercial trees costs more than they are worth (Table 2).

The financial cutting cycle of 20 years is also 10 year shorter than that of the new Selective Management System (SMS) (Thang 1987), and the diameter limit is lower. Our results suggest that felling cycles such as those prescribed by the MUS and SMS are economically justified only in presence of fixed costs higher than $250 per ha and very low discount rates.

The *SEV*-maximising regime produces approximately 0.85 m^3 ha^{-1} yr^{-1} and the residual stand basal area is 64% that of a virgin stand. The maximum deviation from the climax distribution is 100% because there are no trees in the largest diameter class of dipterocarps and commercial non-dipterocarps. Instead, the diameter distribution of non-commercial trees is almost the same as in the climax natural forest (Table 4). In this managed forest, logging damage (defined as the ratio of the number of trees damaged relative to all non-harvested trees) is sensibly less (10%) than the damage caused by logging a virgin stand, estimated at more than 30% (Appanah and Weinland 1990, Pinard *et al.* 1995).

Cutting policies under management for multiple uses

The art of multiple-use forest management consists in developing compromise policies that appeal to different constituencies, without necessarily optimising each constituency's preferences. The models and criteria presented above can be useful in designing such compromise policies. As an example, to assess the effects of minimum carbon storage considerations on the economic cutting policy, the following constraint was added to the model (12):

$$PVK \geq PVK_{\min} \qquad (13)$$

This ensures that a minimum carbon storage level ($PVK_{\min}$) is maintained in the forest stand and end uses. Through parametric variations of $PVK_{\min}$ it is possible to derive alternative steady states and measure their ecological and economic performance. The optimal results in terms of soil rent, *SEV*, corresponding to increasing values of $PVK_{\min}$ are in Figure 1. The marginal cost of additional carbon storage increases (the trade-off curve becomes steeper) as the carbon storage constraint is tightened from 170 tons ha^{-1} (in which case it was not binding), to 214 tons ha^{-1} (the maximum possible storage). Storing more carbon requires decreasing the harvest in almost a linear fashion while lengthening the cutting cycle (Figure 2). For example, to increase carbon storage from 170 tons (unconstrained *SEV* maximisation) to 200 tons ha^{-1} requires lowering the annual harvest from 0.85 to 0.25 m^3 ha^{-1} yr^{-1} and lengthening the cutting cycle from 20 to 25 years. At each felling only 6 m^3 ha^{-1} are extracted, almost exclusively of middle-sized dipterocarp species. Since much valuable timber is also left standing, the *SEV* drops from $702 per ha to -$1,788 per ha.

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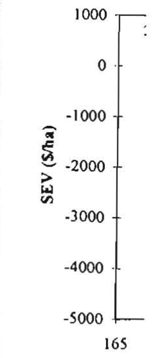
[7] Although only commercial trees are harvested, the continuous presence of commercial trees is insured because only trees that have a diameter of at least 30 cm are taken. With a cutting cycle of 20 years, there is enough time for some of the remaining commercial trees to reach the largest size classes (Table 5). Large trees are more likely to be seed bearing than small ones (Thomas 1996), but the empirical findings underlying the growth model (Boscolo *et al.* 1997) suggest that ingrowth of a species group is positively related to the number of trees of that species present in the stand, regardless of size, plus a constant ingrowth attributable to seeds coming from surrounding stands.



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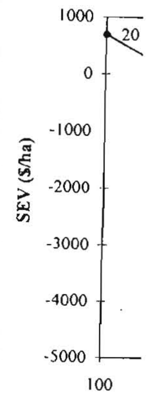

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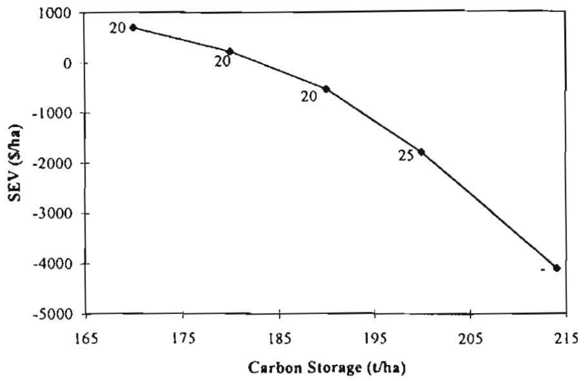

FIGURE 1 *Maximum soil rent (SEV) for a stand storing specific amounts of carbon. Numbers next to the data points are best felling cycles.*

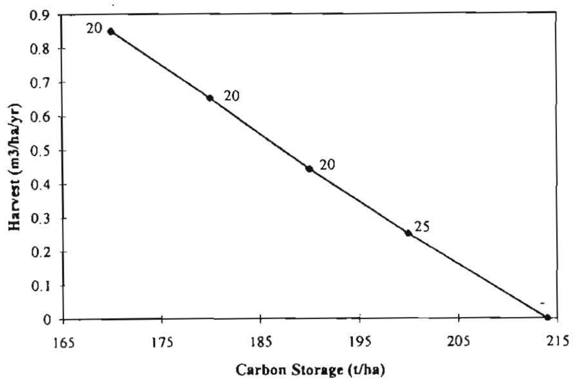

FIGURE 2 *Harvests that maximise SEV while storing specific amounts of carbon. Numbers next to the data points are best cutting cycles*

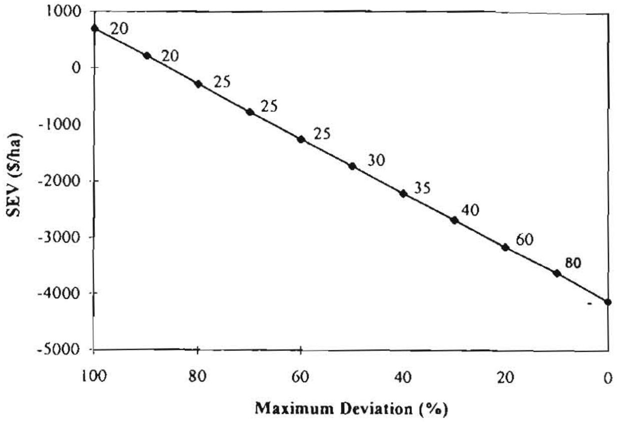

FIGURE 3 *Maximum soil rent (SEV) for a stand within a specified distance from the climax distribution. Numbers next to the data points are best cutting cycles*

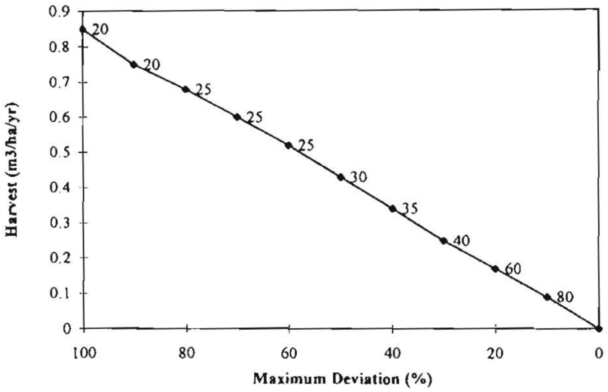

FIGURE 4 *Harvests that maximise SEV for a stand within a specific deviation from the climax distribution. Numbers next to the data points are best cutting cycles*

Alternatively, one could maximise carbon storage, subject to generating a rent at least equal to a prescribed minimum, SEV_{min}. In particular, setting $SEV_{min} = 0$ implies requiring a rate of return on the capital embedded in the stock of trees of 6% per year, competitive with the rest of the economy[8]. The results are in Table 6. This cutting regime would permit a carbon storage 14 tons ha^{-1} higher than the *SEV*-maximising solution. The cut would consist almost exclusively of dipterocarp trees with DBH above 30 cm. Since there would still be some diameter class without trees, *DEV* = 100%.

Similarly, the effect of tree diversity on optimal cutting regimes can be addressed by adding the following constraint to model (12):

$$DEV_{i,j} = \frac{y^*_{i,j} - y_{ij}}{y^*_{i,j}} \times 100 \leq D_{max} \ \forall \ i,j \qquad (14)$$

which forces the relative number of trees of any species and size class to fall short of the climax level by no more than D_{max}, while it can exceed D_{max} by any amount. Setting D_{max} at 100% allows complete harvest, while setting it at zero forces the maintenance of a distribution with at least as many trees in each species-size class as in the climax forest. Reducing the maximum deviation from 100% to 0% leads to a linear decrease in soil rent (Figure 3). Approaching the climax forest would require decreasing the harvest and lengthening the cutting cycle (Figure 4). For example, to keep basal area in every species-size class within at least 50% of that in a virgin stand would require an annual harvest of 0.43 $m^3ha^{-1}yr^{-1}$ and a felling cycle of 30 years (*SEV* would fall from $702 per ha to -$1,730 per ha).

Another possible compromise solution is to maximise diversity subject to a preset target for *SEV*. This is achieved by making D_{max} in equation (14) a variable, and changing the objective function to min (D_{max}), i.e. minimising the largest relative deviation in number of trees with respect to the climax distribution. This formulation, with an *SEV* target of zero, leads to a steady-state where the investment in standing trees yields a return of 6%. The results are in Table 7. Every 20 years a selection of commercial middle-sized (DBH between 30 and 60 cm) trees would be cut. No diameter class would be harvested completely. The 'weakest' tree category would deviate from the climax state by less than 86% which, in this case, means that every 10 hectares there would be at least two trees in every size-class at all times. This steady state increases the carbon storage slightly, from 170 in the case of unconstrained *SEV* maximization to 176 tons per ha.

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[8] Though the return to the land itself would be nil.

TABLE 6 *Steady state that maximises carbon storage while yielding a return of 6% on the tree stock*

Diameter (cm)	Number of trees/ha	Cut	Number of trees/ha	Cut	Number of trees/ha	Cut
	dipterocarps		other commercial		non-commercial	
10-20	33.4	0.0	81.9	0.0	282.2	0.0
20-30	12.5	0.0	23.3	0.0	65.5	0.0
30-40	4.6	4.6	11.9	0.0	20.4	0.0
40-50	1.9	1.9	6.2	0.0	7.7	0.0
50-60	0.5	0.5	2.0	1.9	3.0	0.0
60-70	0.1	0.1	0.7	0.0	1.0	0.0
70+	0.0	0.0	0.6	0.0	1.0	0.0

Felling cycle:	20 years
Total number of trees (before cut):	560 trees per ha
Total number of trees (after cut):	516 trees per ha
Basal area (before cut):	$20.7\ m^2\ ha^{-1}$
Basal area (after cut):	$18.7\ m^2\ ha^{-1}$
Residual stand damaged:	6%
Extracted volume:	$0.57\ m^3\ ha^{-1}\ yr^{-1}$
Harvest revenues (net):	\$241 per ha
Present value of harvests:	\$132 per ha
Value of residual stock:	\$109 per ha
Soil expectation value (*SEV*):	\$0 per ha
Carbon storage in end uses (PVK_c):	3 tons per ha
Carbon storage in the stand (PVK_s):	181 tons per ha
Max deviation from climax, after cut (*DEV*):	100%

TABLE 7 *Steady state that minimises the largest deviation from the climax distribution while yielding a return of 6% on the tree stock*

Diameter (cm)	Number of trees/ha	Cut	Number of trees/ha	Cut	Number of trees/ha	Cut
	dipterocarps		other commercial		non-commercial	
10-20	38.0	0.0	73.4	0.0	285.9	0.0
20-30	14.2	0.0	20.6	0.0	65.7	0.0
30-40	5.4	4.6	6.6	5.4	20.5	0.0
40-50	2.4	1.9	1.8	1.1	7.7	0.0
50-60	1.0	0.5	0.6	0.1	3.0	0.0
60-70	0.5	0.1	0.3	0.0	1.0	0.0
70+	0.9	0.0	0.2	0.0	1.0	0.0

Felling cycle:	20 years
Total number of trees (before cut):	551 trees per ha
Total number of trees (after cut):	491 trees per ha
Basal area (before cut):	$19.7\ m^2\ ha^{-1}$
Basal area (after cut):	$17.2\ m^2\ ha^{-1}$
Residual stand damaged:	9%
Extracted volume:	$0.73\ m^3\ ha^{-1}\ yr^{-1}$
Harvest revenues (net):	\$255 per ha
Present value of harvests:	\$115 per ha
Value of residual stock:	\$115 per ha
Soil expectation value (*SEV*):	\$0 per ha
Carbon storage in end uses (PVK_c):	4 tons per ha
Carbon storage in the stand (PVK_s):	172 tons per ha
Max deviation from climax, after cut (*DEV*)[(a)]:	85.6%

Note: a. This is the objective function being minimised.

CONCLUSION

Growth models coupled with optimisation methods can be very helpful in choosing among alternative management policies in tropical rain forests, while taking into account the non-timber as well as the timber values. Here we gave special consideration to the role of the forest for income generation, carbon sequestration, and preservation of tree diversity.

Starting with a growth model calibrated from a 50-hectare demographic plot in peninsular Malaysia, we explored the features of steady states resulting from different combinations of management criteria. Limiting alternatives to steady states means that sustainability is treated as a fundamental constraint. With the number of possible steady states being infinite, we choose among them by setting different objective functions and constraints regarding timber revenues, carbon storage and tree diversity, judged by the deviation from the climax state.

The solution that maximised land rent consists of felling all commercial trees with diameters over 30 cm. All non-commercial trees are left uncut, thus contributing to diversity and carbon storage. This regime yields a perpetual flow of $17m^3\ ha^{-1}$ and net harvest revenues of \$338 per ha every 20 years.

An advantage of optimisation methods is that they give readily the trade off between objectives governed by constraints, and the one being maximised. For example, the marginal cost of increasing carbon storage starts at \$47 ton^{-1} for the first 10 metric tons, increases to \$77 t^{-1} for additional 10 tons, and increases again up to \$167 t^{-1} in the case of the climax natural forest that is never cut. In this application, marginal cost is measured in terms of foregone land rent. Least cost solutions to increase carbon storage require lengthening the cutting cycle and limiting the harvest almost exclusively to dipterocarp trees.

Overall, the opportunities to modify practices to achieve higher levels of carbon storage appear limited and expensive. These optimisation results are consistent with the simulations of Boscolo *et al.* (1997) which found that, starting from a virgin stand, different cutting cycles or diameter limits were inefficient ways of increasing carbon storage, while a more cost-effective option was to reduce logging damage.

The analysis also gave a measure of the cost of maintaining a diverse stand, in the sense of keeping it as close as possible to its natural climax state. The marginal cost of decreasing the deviation of a managed stand from its climax state was found to be almost constant, at approximately \$50 per percentage point of deviation. For example, to decrease the maximum deviation from 100% (at least one tree category is totally eliminated) to 90% (in each tree category at least 1.6 trees every 10 ha remain standing at all times) would cost approximately \$500 per ha in terms of foregone land rent. The trade-off between land rent and tree diversity is attained through lengthening of the felling cycle and through selective harvesting of mid-sized commercial trees.

With the assumptions adopted in this study, our findings point to solutions that seemingly diverge from cutting policies such as those prescribed by the MUS or the SMS. For example, soil rent maximisation would entail shorter felling cycles and
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cycles and lower diameter limits. Forest management for multiple uses would entail a higher retention of non-dipterocarp trees (e.g., to increase the level of carbon storage) and a more selective harvest of primarily mid-sized commercial trees (e.g., to increase tree diversity).

We recognise that these results must be read in light of the assumptions adopted for the study. At least two words of caution are in order, one regarding the growth model, the other regarding the economic parameters used. With respect to the growth model, we remind the reader that the relationships adopted to predict recruitment levels (embodied in the matrix **G** and the vector **c**) could explain only partially ingrowth variability (see also Boscolo *et al.* 1997, and footnote 7). Clearly, additional research is needed to enhance our ability to predict biological phenomena, such as recruitment, in tropical forests. Second, our economic analyses rest on the assumption that prices are fixed and constant over time. While for commercial tropical species real prices have not changed much over the past few decades (FAO 1990), many species that were once considered non-commercial have recently become economically attractive. We do not know whether this pattern will continue in the future. If so, price increases in any species group will affect the overall results of this analysis. An investigation of such possibility should deserve attention in future research.

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REFERENCES

Appanah, S. and Weinland, G. 1990 Will the management systems for hill dipterocarp forests stand up? *Journal of Tropical Forest Science* **3** (2): 140-158.

Binkley, C.S. and van Kooten, G.C. 1994 Integrating climate change and forests: economic and ecological assessments. *Climatic Change* **28**: 91-110.

Boscolo M., Buongiorno, J. and Panayotou, T. 1997 Simulating options for carbon sequestration through improved management of a lowland tropical rainforest. *Environment and Development Economics* **2**(3): 239-261.

Buongiorno, J. and Michie, B. 1980 A matrix model of uneven-aged forest management. *Forest Science* **26**(4): 609-625.

Buongiorno, J., Dahir, S., Lu, H. and Lin, C. 1994 Tree size diversity and economic returns in Uneven-Aged Forest Stands. *Forest Science* 40(1): 83-103.

Buongiorno, J., Peyron, J.L., Houllier, F. and Bruciamacchie, M. 1995 Growth and management of mixed species, uneven-aged forests in the French Jura : implications for economic returns and tree diversity. *Forest Science* **41**: 397-429.

Ciriacy-Wantrup, S.V. 1968 *Resource conservation, economics and policies*. 3rd edn Univ.of Cal. Div. Ag. Sci. Berkeley.

de Montalembert, M.R. 1991 Key forestry policy issues in the early 1990s. *Unasylva* 166(42): 9-19.

Englin, J. and Callaway, J.M. 1995 Environmental impacts of sequestering carbon through forestation. *Climatic Change* **31**: 67-78.

FAO. 1990 *Forest Products Prices 1969-1988*. FAO Forestry Paper **95**. FAO, Rome, Italy.

Griffin, M. and Caprata, M. 1977 Determination of cutting regimes under the Selective Management System. Paper presented in ASEAN seminar on tropical rain forest management. November 7-10, 1977, Kuantan, Malaysia. Mimeo.

Harrison, J.L. 1962 The distribution of feeding habits among animals in a tropical rain forest. *Journal of Animal Ecology* **31**: 53-63.

Hoen, H.F. and Solberg, B. 1994 Potential and economic efficiency of carbon sequestration in forest biomass through silvicultural management. *Forest Science* **40**: 429-451.

Ingram, C.D. and Buongiorno, J. 1996 Income and diversity tradeoffs from management of mixed lowland Dipterocarps in Malaysia. *Journal of Tropical Forest Science* **9**(2): 242-270.

ITTO (International Tropical Timber Organization) and FRIM (Forest Research Institute of Malaysia). 1994 Economic case for natural forest management.

Magurran, A.E. 1988 *Ecological diversity and its measurement*. Princeton, NJ. Princeton University Press.

Panayotou, T. and Ashton, P. 1992 *Not by timber alone. Economics and ecology for sustaining tropical forests*. Island Press, Washington D.C.

Peters, C.A., Gentry, A. and Mendelsohn, R. 1989 Valuation of an Amazonian rainforest. *Nature* 339 (June): 655-6.

Pinard, M.A., Putz, F.E., Tay, J. and Sullivan, T. 1995 Creating timber harvest guidelines for a reduced-impact logging project in Malaysia. *Journal of Forestry* **93**(10): 41-45.

Plantinga, A.J. and Birdsey, R.A. 1994. Optimal forest stand management when benefits are derived from carbon. *Natural Resource Modeling* **8**(4): 373-387.

Primack, R.B. and Lovejoy, T. E. (eds.) 1995 *Ecology, conservation, and management of Southeast Asian rainforests*. New Haven. Yale University Press.

Ricklefs, R.E. and Schluter, D. (eds.) 1993 *Species diversity in ecological communities*. University of Chicago Press.

Rosenzweig, M.L. 1995 *Species diversity in space and time*. Cambridge University Press.

Thang, H.C. 1987 Forest management systems for tropical high forests, with special reference to Peninsular Malaysia. *Forest Ecology and Management* **21**: 3-20.

Thomas, S.C. 1996 Relative size at onset of maturity in rain forest trees: a comparative analysis of 37 Malaysian species. *Oikos* **76**(1): 145-154.

Wells, D.R. 1971 Survival of the Malaysian bird fauna. *Malay. Nat.* **24**: 248-256.

Weitzman, M. 1992 On diversity. *The Quarterly Journal of Economics* (CVII): 363-406.

Whitmore, T.C. and Burnham, C.P. 1984 *Tropical rain forests of the Far East*. Oxford University Press, New York.

Whitmore, T.C. 1990 *Introduction to tropical rain forests.* Oxford. Clarendon Press.

Wyatt-Smith, J. 1963 *Manual of Malayian silviculture for inland forests.* Malayan Forest Record no. 23. Forestry Department. Peninsular Malaysia.

Zar, J.H. 1984 *Biostatistical analysis.* Engelwood Ciffs, NJ, Prentice-Hall Inc.

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