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Theory and Applications

Simulating options for carbon sequestration through improved management of a lowland tropical rainforest*

MARCO BOSCOLO
Center for Tropical Forest Science, Smithsonian Tropical Research Institute, Harvard Institute for International Development (HIID), Harvard University, Cambridge, MA 02138, USA
E-mail: mboscolo@hiid.harvard.edu

JOSEPH BUONGIORNO
Department of Forestry, University of Wisconsin-Madison, 1630 Linden Drive, Madison, WI 53706, USA

THEODORE PANAYOTOU
Harvard Institute for International Development (HIID), Harvard University, One Eliot Street, Cambridge, MA 02138, USA

ABSTRACT. The growing evidence that increased levels of carbon dioxide in the atmosphere are related to global warming has prompted several countries to consider options for reducing and offsetting current carbon dioxide emissions. Opportunities for carbon sequestration with forestry activities have been analysed in detail primarily in industri-

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alized nations, mainly because of data availability. This article presents a model that simulates a tropical forest stand in its role as a source of income and as a carbon store, and quantifies the potential for and cost-effectiveness of carbon sequestration through modifications of management practices. Results suggest that financing modifications of forestry practices may achieve net carbon sequestration in a relatively cost-effective way. Tropical countries with extensive forest resources may be in a position to offer cost-effective net carbon sequestration options.

1. Introduction

The growing evidence that increased levels of carbon dioxide in the atmosphere are related to global warming (IPCC, 1990; Houghton *et al.*, 1996; Santer *et al.*, 1996) and the risks of welfare losses associated with an increase in global temperatures have prompted several countries to consider options for offsetting current carbon dioxide emissions (UNCED, 1992). One such option takes the form of a tradable permits programme where units of carbon sequestered from the atmosphere in one location can be marketed to offset emissions in other locations. Forestry activities are among the initiatives that promise a cost-effective CO_2 abatement, especially where rapid rates of tree growth are combined with high environmental benefits.

Forests influence carbon concentration in the atmosphere by assimilating CO_2 through biomass build-up and by releasing it through biomass decay. In climax forests, where growth, mortality, accumulation and respiration are in long-term balance, harvesting causes biomass, and consequently carbon, to be lost (or sunk into lasting end-uses such as lumber built into houses). Carbon storage is then gradually recovered as the forest grows in the following years (Bormann and Likens, 1979).

Several forestry practices can increase carbon sequestration. They include afforestation, reforestation, preservation of forest land from conversion, adoption of various agroforestry practices, lengthening of the felling cycle, increase in diameter cutting limits to enhance carbon storage, and adoption of low-impact harvesting methods. In tropical countries, research has been done to assess the potential of carbon sequestration through reforestation, protection of forest from land-use conversions, and agroforestry activities (cf. Faeth *et al.*, 1994; Trexler and Haugen, 1995). The possibilities of achieving cost-effective carbon sequestration by modifying current management systems of natural tropical forests have, however, been explored only to a limited extent (Putz and Pinard, 1993), partly because of insufficient knowledge regarding the growth dynamics of tropical forest stands.

Modifying the management of natural tropical forests can improve residual conditions, favouring both future productivity and levels of carbon storage while maintaining high economic returns. Incentives to manage tropical and temperate forests for both timber and carbon may soon appear in several countries (see, e.g., the US Climate Change Action Plan). As a result, the activity of carbon sequestration may be rewarded in monetary terms. This prospect provides an opportunity for economic gains to both managers of forest resources and parties interested in acquiring carbon credits. To measure the potential for such gains, it is important to assess

how the forest resource can be managed for multiple use (timber and carbon storage) and to identify where the cheapest carbon can be 'bought'.

The objective of this article is to develop a matrix growth model that simulates the growth dynamics of a lowland dipterocarp forest in Peninsular Malaysia, and to identify which combinations of timber harvest and carbon storage are technically feasible. Moreover, since the value of timber can easily be assessed, knowledge of the production possibility frontier between timber and carbon allows estimation of the opportunity cost of carbon sequestration in monetary terms.

The literature presents several methods of estimating the costs of carbon sequestration through forestry activities. They include: (a) simple average-cost estimations (e.g. Sedjo and Solomon, 1989; Dudek and LeBlanc, 1990); (b) 'least-cost' approaches, where the marginal cost of carbon sequestration is derived by adopting increasing land rental rates or purchase costs (Moulton and Richards, 1990), or by estimating the cost of removing land from agricultural production (Parks and Hardie, 1995); (c) mathematical programming techniques (e.g. Adams *et al.*, 1993; Hoen and Solberg, 1994); and (d) econometric studies (e.g. Stavins, forthcoming).[1] The approach we adopt can be considered a 'least-cost' study, in the sense that the cost of carbon sequestration is evaluated for alternative management schemes which are then ranked in order of cost-effectiveness.

Our study differs, however, from existing ones in at least two important respects. First, while existing studies on the costs of carbon sequestration through forest activities have focused primarily on the implications of modifying forested areas through afforestation, reforestation or retarded deforestation, in this study we address the potential for carbon sequestration by modification of management schemes on an existing forest stand. Second, our analysis looks at an uneven-aged tropical forest stand where carbon sequestration can be achieved by modifying the felling cycle, the diameter limit, and logging technology. Other analyses that have looked at the implications of considering carbon values for forest management have generally focused on rotation age issues, with almost exclusive reference to even-aged stands (see, e.g., Binkley and Van Kooten, 1994; Plantinga and Birdsey, 1994; Englin and Callaway, 1995). Management options such as continued growth, fertilization, thinning, planting, and clear felling with and without green-tree retention have been considered by Hoen and Solberg (1994) for an even-aged forested area in Norway.

In this article the following questions are addressed:

(1) When carbon sequestration and/or other environmental benefits are ignored, which type of management regime is best in the pursuit of financial interests?
(2) If an interest exists in saving carbon through forest management, how can it be achieved at the least cost? What is the relative cost-effectiveness of alternative management decisions (felling cycle, diameter cutting limit, logging technology)?

[1] This review is by no means comprehensive and is meant only to provide a framework to assess how our work fits into existing methodologies. A more comprehensive review can be found in Richards and Stokes (1995).

(3) What is the supply schedule of carbon savings from a tropical forest stand?

2. A model for the management of a lowland tropical rainforest

Addressing these questions requires knowledge of the forest growth dynamics that determine resource carbon storage and timber-yield capabilities. To mimic such forest dynamics we have developed a model based on transition matrices, of the kind originally developed by Usher (1966) and refined by Buongiorno and Michie (1980) through incorporation of an ingrowth function dependent on stand density.

In this model, the stand state is represented by the vector $y_t = [y_{i,j,t}]$ where $y_{i,j,t}$ is the number of trees per hectare of species i ($i = 1,m$), diameter class j ($j = 1,n$), at time t. The harvest at time t is given by the vector $h_t = [h_{i,j,t}]$. The growth model consists of a system of equations where the stand state at time $t + 1$ is predicted from the stand conditions at time t, after harvest:

$$\begin{aligned} y_{i,1,t+1} &= (y_{i,1,t} - h_{i,1,t})\, a_{i,1} + I_i \\ y_{i,2,t+1} &= (y_{i,1,t} - h_{i,1,t})\, b_{i,2} + (y_{i,2,t} - h_{i,2,t})\, a_{i,2} \\ y_{i,3,t+1} &= (y_{i,2,t} - h_{i,2,t})\, b_{i,3} + (y_{i,3,t} - h_{i,3,t})\, a_{i,3} \\ &\dots \\ y_{i,n,t} &= \qquad\qquad (y_{i,n-1,t} - h_{i,n-1,t})\, b_{i,n} + y_{i,n,t}\, a_{i,n} \end{aligned} \tag{1}$$

where $a_{i,j}$ is the probability that a live tree at time t will still be alive at time $t + 1$ and, in the same diameter class, $b_{i,j}$ is the probability that a live tree at time t and in class $j - 1$ will be in class j at time $t + 1$.

In matrix form the model is formulated as (Buongiorno and Michie, 1980):

$$\mathbf{y}_{t+1} = \mathbf{G}(\mathbf{y}_t - \mathbf{h}_t) + \mathbf{c} \tag{2}$$

where:

$$\mathbf{G} = \begin{bmatrix} \mathbf{G}_1 & & & \\ & \mathbf{G}_2 & & \\ & & \dots & \\ & & & \mathbf{G}_m \end{bmatrix} \qquad \mathbf{c} = \begin{bmatrix} \mathbf{c}_1 \\ \mathbf{c}_2 \\ \dots \\ \mathbf{c}_m \end{bmatrix}$$

and where

$$\mathbf{G}_i = \begin{bmatrix} a_1 & & & \\ b_2 & a_2 & & \\ & \dots & \dots & \\ & & b_n & a_n \end{bmatrix} \qquad \mathbf{c}_i = \begin{bmatrix} I_i \\ 0 \\ \dots \\ 0 \end{bmatrix}$$

The model assumes that transition probabilities (and the underlying growth and mortality rates) are independent of stand state, while ingrowth (the number of trees that enter the smallest diameter class between t and $t + 1$) is expected to be dependent on stand density and composition. This is a modification of the ingrowth model proposed by Buongiorno *et al.* (1995). Ingrowth of a species is a positive linear function

of the number of trees of that species and is a negative function of total stand basal area. At any time t, the ingrowth function is:

$$I_{i,t} = \alpha_i + \beta_i \sum_{i=1}^{m} \sum_{j=1}^{n} B_j (y_{i,j,t} - h_{i,j,t}) + \gamma_i \sum_{j=1}^{n} (y_{i,j,t} - h_{i,j,t}) \quad (3)$$

where B_j is the basal area of a tree belonging to diameter class j.

The model parameters were estimated with data from a large (50 ha) demographic plot located in the Pasoh forest, Negeri Sembilan, Peninsular Malaysia. The plot, established by the Forest Research Institute of Malaysia in 1985 (Manokaran *et al.*, 1990), was censused in 1987 and 1990. During the censuses all free-standing, woody stems with a diameter at breast height (dbh) ≥1 cm were identified, tagged, mapped and their diameter recorded. Previous analyses of census data can be found in Kochummen *et al.* (1990), Manokaran and LaFrankie (1990), Manokaran *et al.* (1992), Appanah and Weinland (1993) and Liu and Ashton (1996).

For the present study, only trees with dbh ≥10 cm were considered. Biomass of these trees accounts for most of the standing biomass and, consequently, carbon storage. The trees of the 50 ha plot have been divided into seven 10 cm diameter classes and three species groups: commercial dipterocarps (D), commercial non-dipterocarps (O), and non-commercial trees (N). The distinction between commercial dipterocarps and non-dipterocarps was made for two reasons: (1) previous studies (e.g. Wan Razali, 1988; Appanah and Weinland, 1991) have shown that dipterocarp species generally exhibit a higher growth rate; and (2) current management requires different diameter cutting limits, generally higher for dipterocarps.

Ingrowth and upgrowth parameters were estimated from the dynamic census data, and the expected mortality rates were further calibrated using knowledge of the steady state (the climax forest). This approach allowed the development of a model able to yield reliable long-term predictions. Full justification of this approach and the procedure used to estimate the model parameters are reported in the Appendix.

3. Simulating the outcome of alternative management decisions

With this growth model (see section 2 and Appendix) we have simulated the impact of alternative management regimes on financial returns and carbon savings at the stand level. For a given forest tract, forty management scenarios were simulated, each with a different combination of the decision variables: felling cycle, diameter cutting limit, and effort to reduce logging damage.[2] Each simulation gave estimates of harvested and residual number of trees, basal area, volume and biomass for each species group. From these data, carbon storage and revenue estimates were derived.

[2] As such, the scenarios considered are not endogenous to economic conditions, i.e. each scenario is not necessarily the most efficient one for a given set of prices for timber and input factors. Selection of the efficient scenarios is carried out subsequently (and 'exogenously'), by ranking them in order of cost-effectiveness.

3.1. Decision variables

Typically, the lowland tropical rainforests of Peninsular Malaysia were formerly harvested according to the Malaysian Uniform System (MUS; Wyatt-Smith, 1963). Under this system all valuable trees are felled, and the residual stand above a certain minimum diameter is poison-girdled. The stand is then left to grow for 60–70 years. The MUS was replaced by the Selective Management System (SMS), in which diameter cutting limits are higher than under the MUS, ranging from 45 to 65 cm, no poison-girdling takes place and felling cycles are shortened to approximately 30 years (Thang, 1987). The simulation scenarios cover the whole range of possible felling cycles and diameter limits:

- *Felling cycles.* Simulations were performed for cuttings every 30, 40, 50, 60 or 70 years.
- *Diameter cutting limits.* Under the MUS, all commercial trees above 45 cm are cut (Thang, 1987). Diameter limits are higher when selective systems are adopted, and generally different for dipterocarps and non-dipterocarps. For simplicity, and to reduce the number of simulations, the same diameter limit was, however, applied to all commercial trees: 30, 40, 50 or 60 cm.
- *Logging damage.* The SMS in Malaysia assumes that harvesting damages about 30 per cent of the residual trees, but these figures may be too conservative (cf. Wyatt-Smith, 1988; Appanah and Weinland, 1991). Logging damage could be decreased by cutting woody climbers before harvesting; mapping the area to be logged; careful planning and building of roads and skid trails; directional felling; and minimal use of bulldozers (Panayotou *et al.*, 1994). Putz and Pinard (1993), referring to a reduced-impact logging project in Sabah, Malaysia, estimated that reduced-impact logging could increase costs by about $135/ha, and reduce damage from 40 per cent to 15 per cent (Pinard *et al.*, 1995). There is, however, little evidence to support this cost figure, which we rounded to $150/ha.

3.2. Management criteria

The performance of each management scenario was judged according to two main criteria: financial returns and carbon sequestration.

Financial returns. Financial returns were measured by the net present value (NPV) of timber extractions over a long time period:

$$\mathrm{NPV} = \sum_{t=0}^{T} (TR_t - TC_t)/(1 + r)^t \tag{4}$$

where TR_t is the total revenue from timber sale in year t, TC_t is the total cost, and r is the yearly interest rate. The real interest rate was set at 6 per cent.[3] The time horizon (T) was 200 years. By that time, and in fact much earlier, a stand cut according to a particular regime has reached a dynamic

[3] The average lending rate in Malaysia from 1976 to 1990 was 5.7 per cent per year, net of inflation (Ingram and Buongiorno, forthcoming).

steady state: the growth just replaces harvest, and the cut is sustainable, for ever. In addition, increasing the time horizon beyond 200 years changes the NPV very little. Therefore, the NPV is near that obtained by following a given scenario over an infinite length of time. The commercial volume harvested was estimated with the following volume equations (FFD, 1987):

$$V = 0.3211 - 0.002175D + 0.0003521D^2 \quad (\text{for } 10 \leq D < 30 \text{ cm})$$
$$V = 0.1991 + 0.006148D + 0.0004081D^2 \quad (\text{for } 30 \leq D < 60 \text{ cm})$$
$$V = 0.8602 - 0.03872D + 0.0013164D^2 \quad (\text{for } D \geq 60 \text{ cm})$$

where V is in m^3 and D is the diameter at breast height, in cm. The same equations are used for the three species groups. Stumpage values were derived by subtracting harvesting costs from log prices (ITTO/FRIM, 1994). To compute net revenues, volume-based charges (estimated at 10 per cent of the log price) and area-based charges ('premiums', estimated at $350/ha, partly based on Awang Noor (1994) and ITTO/FRIM (1994)) have also been deducted. Tree volume and value are given in Table 1. Trees of zero value were not cut in the simulations.

Carbon sequestration. The analysis takes into account variations in carbon stored in living trees (carbon accumulated, in above- and below-ground biomass) and does not consider carbon stored in soil or end-uses. Following Hoen and Solberg (1994), carbon sequestration was measured as 'net carbon accumulation', i.e. the difference between carbon stored and withdrawn over time, per unit of land. Harvesting, ignoring the storage of carbon in lasting end-uses such as buildings, furniture, etc., results in a negative net accumulation (loss). In the following years, the stand grows, accumulates biomass and stores carbon.

To account for the carbon flows at different points in time, at least three different approaches are used in the literature. Richards and Stokes (1995) classify them as flow summation, mean carbon storage, and discounting (or levelization). Flow summation measures the total tons of carbon sequestered, regardless of when sequestration occurs (cf. Faeth *et al.*, 1994). The results are comparable only for scenarios of the same length of time, and they are indifferent to the schedule of carbon sequestration. The mean carbon storage is the average amount of carbon stored per year over the

Table 1. *Estimates of tree volume and stumpage value ($/tree in 1991)*

Diameter class (cm)	*Volume/tree (m^3)*	*Stumpage value[a] ($/tree)*		
		D	*O*	*N*
10–20	0.368	0	0	0
20–30	0.487	0	0	0
30–40	0.914	29	13	0
40–50	1.302	59	32	0
50–60	1.772	104	61	0
60–70	3.905	230	135	0
70+	5.361	316	186	0

[a]Net of taxes.

period analysed. Mean carbon storage reflects the time needed to store carbon, and converges to a limit as the planning horizon increases, but, like flow summation, it assumes indifference between carbon sequestration (or storage) in the near or distant future. Discounting, used here, discounts the amounts of CO_2 sequestered from the atmosphere in each period as if it were money.[4] Net present carbon accumulation (NPCA) was computed as

$$\text{NPCA} = \sum_{t=0}^{T} (C_{t+1} - C_t)/(1 + r)^t \tag{5}$$

where C_t , the amount of carbon stored in a forest stand at time t (in tons/ha), is

$$C_t = TAGB_t \times f_1 \times f_2 \tag{6}$$

and where TAGB is the total above-ground biomass (in tons/ha), f_1 is a correction factor that accounts for root biomass (estimated at 10 per cent of TAGB; Whitmore, 1984), and f_2 is the ratio of carbon to organic biomass (0.5; Kira, 1978).

As suggested by Brown *et al.* (1989), TAGB was estimated at double the commercial biomass which, in turn, was computed as the sum of the products of commercial volume and average wood density (Desch, 1941, 1954) for each species group. The current average biomass in the Pasoh plot was 388 tons/ha,[5] corresponding to a total tree biomass (above and below ground) of 428 tons/ha and a carbon storage of 214 tons/ha (see Table 2). Like costs and timber revenues, carbon flows occurring at different points in time were discounted at 6 per cent per year. Indeed, if a market for carbon storage existed, the value of 1 ton of carbon stored would be computed and discounted exactly like timber revenues.

4. Results

The simulations of management alternatives were done over 200 years. The initial state was the average state per hectare in the Pasoh plot. Since the plot is near a climax state, the simulations show the effect of different managements on a virgin lowland tropical forest stand. An illustration of the biomass flows for two possible management scenarios is presented in Figure 1.

4.1. Best management to maximize financial returns: baseline solution

The results of the simulations are given in Table 3 for conventional logging methods and in Table 4 for reduced-impact methods. The solution that yielded the highest NPV/ha (baseline solution) consisted in cutting every 40 years all commercial trees with diameter greater than 30 cm by conven-

[4] The question whether discounting is appropriate in dealing with carbon flows has been addressed by Van Kooten *et al.* (forthcoming), but see also Hoen and Solberg (1994), Stavins (forthcoming) and Boscolo *et al.* (1996).

[5] Kira (1978) reports another set of allometric relationships obtained from some destructive sampling carried out in the Pasoh forest during 1971 and 1973. By using Kira's relationships we obtained a value for TAGB of 388 tons/ha, identical to the estimate derived following Brown's approach.

Table 2. *Initial stand state* (ha^{-1})

Diameter class (cm)	*Trees*				*Basal area* (m^2)				*Biomass (tons)*			
	D	*O*	*N*	*Total*	*D*	*O*	*N*	*Total*	*D*	*O*	*N*	*Total*
10–20	30.5	61.5	262.3	354.2	0.5	1.1	4.6	6.3	14.8	29.4	140.0	184.2
20–30	11.6	17.5	61.5	90.7	0.6	0.9	3.0	4.5	7.5	11.1	43.4	62.0
30–40	5.4	8.9	19.8	34.1	0.5	0.9	1.9	3.3	6.6	10.6	26.2	43.3
40–50	3.9	4.6	7.2	15.8	0.6	0.7	1.1	2.5	6.8	7.8	13.6	28.2
50–60	3.6	3.1	2.9	9.5	0.8	0.7	0.7	2.3	8.3	7.1	7.4	22.8
60–70	2.5	1.7	0.9	5.2	0.8	0.6	0.3	1.7	13.1	8.8	5.3	27.3
70+	5.9	1.6	0.9	8.4	2.6	0.7	0.4	3.7	41.7	10.9	7.3	59.9
Total	63.5	98.9	355.4	517.8	6.5	5.5	12.1	24.2	98.8	85.7	243.2	427.8

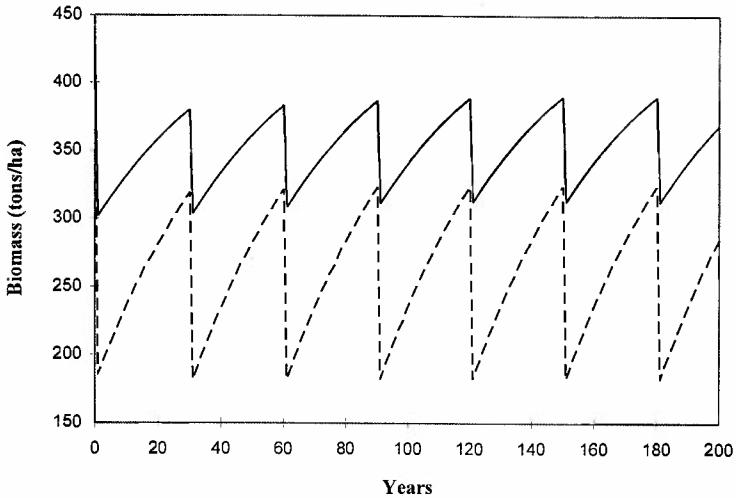

Figure 1. *Biomass flows in a tropical forest stand harvested with alternative logging technologies and diameter cutting limits (30 years felling cycle). Solid line: reduced-impact logging and a diameter cutting limit of 60 cm. Dashed line: conventional logging and a diameter cutting limit of 30 cm.*

Table 3. *Net present value and net present carbon accumulation for various cutting cycles and diameter cutting limits. Harvesting with conventional methods*

	Cutting cycle (years)				
Diameter limit (cm)	*30*	*40*	*50*	*60*	*70*
			NPV ($/ha)		
30	3866	3879	3875	3866	3857
40	3598	3608	3603	3593	3583
50	3225	3231	3223	3212	3203
60	2681	2674	2659	2646	2636
			NPCA (tons/ha)		
30	−88.6	−84.3	−81.9	−80.5	−79.6
40	−85.1	−80.8	−78.3	−76.9	−76.1
50	−82.4	−77.9	−75.5	−74.1	−73.3
60	−78.7	−74.1	−71.6	−70.2	−69.4

tional logging methods. This scenario gave a NPV of $3880/ha and is essentially the management that was practised in the lowlands of Peninsular Malaysia until the mid-1970s.[6] Our simulation differs, however, from the MUS since no poison-girdling or harvesting of non-commercial

[6] As noted above, this management system, the MUS, was later replaced by the SMS, as the core of forestry activities moved from the lowlands to hill areas.

Table 4. *Net present value and net present carbon accumulation for various cutting cycles and diameter cutting limits. Harvesting with reduced-impact methods*

	Cutting cycle (years)				
Diameter limit (cm)	*30*	*40*	*50*	*60*	*70*
			NPV ($/ha)		
30	3713	3731	3727	3717	3707
40	3459	3469	3460	3447	3435
50	3095	3098	3084	3069	3057
60	2564	2547	2524	2505	2491
			NPCA (tons/ha)		
30	−58.2	−56.0	−54.7	−53.9	−53.4
40	−53.2	−51.0	−49.7	−49.0	−48.5
50	−49.2	−46.9	−45.6	−44.9	−44.5
60	−43.8	−41.4	−40.1	−39.4	−38.9

trees has been simulated. The simulation detail showed that most of the NPV was obtained from the first cut that extracted about 90 m^3/ha. The following cuts yielded much less in volume,[7] and even less in monetary terms because of discounting. Therefore, whether the second cut was in 30 or 70 years, the NPV/ha did not change much. This means that, from a financial point of view, the length of the felling cycle matters little as long as all valuable trees are felled in the first cut. Instead, the choice of a diameter limit is critical. NPV/ha was reduced by more than $1,000 by increasing the diameter limit from 30 to 60 cm (Table 3).

Comparing Tables 3 and 4 show that under no combination of felling cycle and diameter cutting limit was the reduction of logging damage financially attractive. While the volume obtained from future cuts was higher, discounted future revenues did not compensate for the additional cost.

4.2. The cost of storing carbon in the forest

The solution that maximized the NPV/ha led to a net present carbon accumulation of −84 tons/ha (it is therefore a loss). This is the discounted sum of carbon removals (from logging) and carbon accumulations (through biomass build-up) over 200 years. When carbon savings are an objective of management, Tables 3 and 4 suggest that carbon losses can be reduced by lengthening the felling cycle, increasing the diameter cutting limit, or reducing logging damage.

[7] Some stands that have been treated in the past according to the MUS now exhibit good stocking levels of commercial species (P. Ashton, personal communication, 1996). This fact apparently contradicts our simulation results. As noted above, however, our simulations did not replicate exactly the MUS, i.e. did not eliminate non-commercial species by either harvesting or poison-girdling. If simulations had considered the harvesting (or poison-girdling) of non-commercial species, the proportion of commercial trees at the second and subsequent harvests would have been significantly higher than predicted.

Each of these adjustments (or a combination of them) represents a divergence from the baseline. Since the baseline was defined as the solution that maximizes financial returns, any divergence from that solution involves a cost in terms of reduced NPV/ha. By comparing the amounts of carbon saved with the corresponding costs in NPV terms, it is possible to investigate which alternatives achieve carbon savings at the least cost.

Starting from the solution with a felling cycle of 40 years and a diameter cutting limit of 30 cm, the most cost-effective strategy for storing carbon in trees is to lengthen the felling cycle. Postponing the second cut from 40 to 50 years increases NPCA by 2.5 tons of carbon/ha at a NPV cost of $3/ha. This translates into a cost of carbon sequestration of only $1.2/ton. At a rate of 2.5 tons of carbon/ha, however, large forest areas are required to store significant amounts of carbon. Lengthening the cutting cycle had, however, practically no effect on the NPV with a discount rate of 10 per cent.

The second cost-effective strategy to save carbon is to reduce logging damage. The effectiveness of this alternative depends on the cost of implementing reduced-impact logging instead of conventional logging methods. For a felling cycle of 50 years and a diameter limit of 30 cm, reduced-impact logging increased NPCA by 27 tons/ha with a sacrifice in NPV of $149/ha. Thus, carbon removals can be reduced at a cost of $5.5/ton. The same results were obtained with a discount rate of 10 per cent.

Carbon savings can also be achieved by increasing the diameter cutting limit. With a felling cycle of 40 years, increasing the diameter limit from 30 to 60 cm increased NPCA by 10 tons/ha. However, this result is obtained at a NPV cost of over $1,200/ha which translates into a cost of carbon sequestration of over $120/ton.

To summarize the opportunity cost of carbon sequestration through natural forest management, we mapped the production possibility frontier between NPV and NPCA (Figure 2). It consists of the combinations of felling cycle, diameter cutting limit and reduced logging damage that were Pareto-efficient, i.e. where one output (NPV or NPCA) could not be improved without diminishing the other. Assuming that all feasible combinations of carbon and income are contained in a convex set, the frontier was drawn by connecting the extreme points. Figure 2 shows that significant carbon savings could be attained at low cost by reducing logging damage. The other alternatives (lengthening the felling cycle and/or increasing the diameter cutting limit) gave little carbon sequestration per hectare or were expensive.

The opportunity cost of increasing carbon storage per hectare of lowland tropical rainforest in Malaysia is shown in Figure 3. The origin is the baseline (NPV-maximizing) alternative. The figure shows that the cost per hectare increases steeply beyond 30 tons/ha of NPCA.

Following Putz and Pinard (1993), we have assumed so far that reduced-impact logging could be achieved at a direct cost of $150/ha. However, this figure is very uncertain. Because reduced-impact logging could achieve carbon savings efficiently, we investigated the sensitivity of the marginal cost of carbon saving (in terms of NPV/ha forgone) to alternative

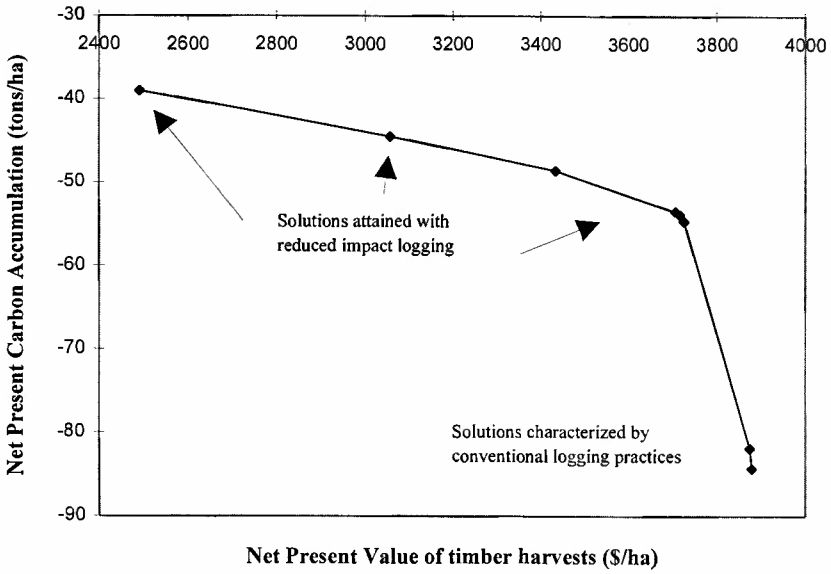

Figure 2. *Production possibility frontier between NPV and NPCA.*

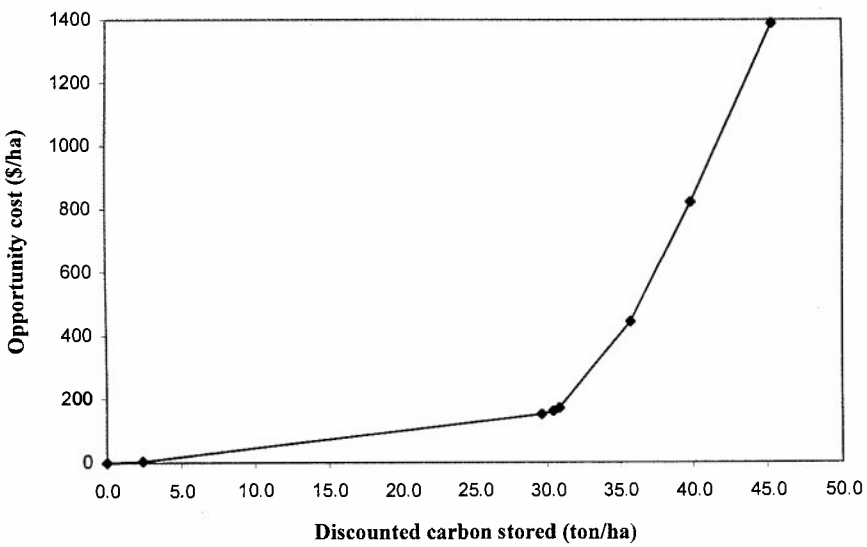

Figure 3. *Opportunity cost (measured in timber revenues forgone) of increasing carbon storage per hectare of lowland tropical rainforest.*

direct costs of implementing reduced-impact logging methods. The marginal cost ($/ton) is the reduction in NPV due to increasing NPCA by 1 ton/ha.

Table 5 shows the marginal cost of storing carbon in the forest for different costs of reducing the logging impact: $75, $150, $300 or $600/ha. As reduced-impact logging was made more expensive it became more cost-

Table 5. *Marginal cost of storing carbon in forest stands for alternative costs of implementing reduced-impact logging (RIL)*

Cumulative carbon savings (tons/ha)	*Marginal cost ($/ton)*	*Solution features*		
		dbh[a] limit (cm)	*Cycle (years)*	*Logging method*
	Incremental cost of RIL = $75/ha			
0.0	0.0	30	40	conventional
2.5	1.2	30	50	conventional
28.4	2.4	30	40	RIL
29.7	6.2	30	50	RIL
30.4	15.7	30	60	RIL
30.9	24.0	30	70	RIL
35.8	55.3	40	70	RIL
39.9	92.4	50	70	RIL
45.4	102.7	60	70	RIL
	Incremental cost of RIL = $150/ha			
0.0	0.0	30	40	conventional
2.5	1.2	30	50	conventional
29.7	5.5	30	50	RIL
30.4	14.3	30	60	RIL
30.9	20.0	30	70	RIL
35.8	55.3	40	70	RIL
39.9	92.4	50	70	RIL
45.4	102.9	60	70	RIL
	Incremental cost of RIL = $300/ha			
0.0	0.0	30	40	conventional
2.5	1.2	30	50	conventional
3.9	7.1	30	60	conventional
30.4	11.2	30	60	RIL
30.9	16.0	30	70	RIL
35.8	55.5	40	70	RIL
39.9	92.4	50	70	RIL
45.4	102.7	60	70	RIL
	Incremental cost of RIL = $600/ha			
0.0	0.0	30	40	conventional
2.5	1.2	30	50	conventional
3.9	7.1	30	60	conventional
4.7	11.3	30	70	conventional
30.9	23.2	30	70	RIL
35.8	55.3	40	70	RIL
39.9	92.4	50	70	RIL
45.4	102.7	60	70	RIL

[a]Diameter at breast height.

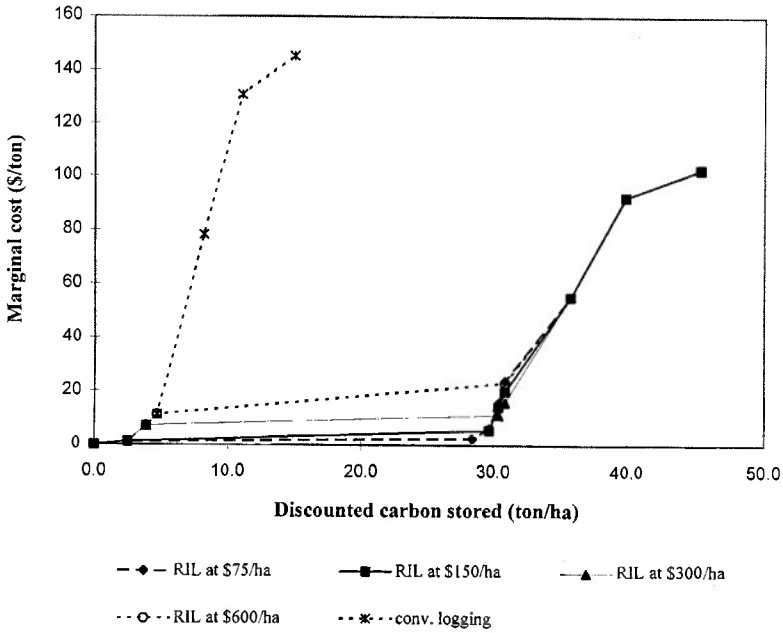

Figure 4. *Marginal costs of storing carbon in a tropical forest stand, for different costs of reduced-impact logging.*

effective to lengthen the felling cycle to 60 or even 70 years before switching from conventional to reduced-impact logging. If reduced-impact logging involves additional expenditures of $75/ha, carbon savings of about 30 tons could be achieved at a marginal cost of $2.4/ton. This figure becomes $5.5/ton at an implementation cost of $150/ha; $11.2/ton at $300/ha; and $23.2/ton at $600/ha. In Figure 4 these results are compared with the marginal cost of carbon savings of using conventional logging.

5. Discussion and conclusion

In this study we have addressed the potential for and cost-effectiveness of carbon sequestration through improved management of lowland tropical forests, by performing a stand-level analysis for a rainforest in Peninsular Malaysia. Though not much forest area in Peninsular Malaysia is left in the lowlands, similar forest types are still abundant in the islands of Sumatra and Borneo (P. Ashton, personal communication, 1996) that are currently being harvested with conventional logging systems. The study suggests that, if logging damage could be reduced from 40 per cent to 15 per cent at a cost of $150/ha, carbon storage in natural tropical forests could be increased at a cost of only $5/ton, in terms of NPV of income. This is significantly lower than comparable[8] cost estimates of carbon sequestration

[8] In this context, comparability refers to the way intertemporal carbon flows have been treated.

achieved through tree plantation (cf. Richards and Stokes, 1995, for a review of studies both in the USA and in developing countries), and compares favourably with carbon taxes currently adopted in Denmark, Finland, The Netherlands, Norway and Sweden (which range from $16/ton to $172/ton; Muller, 1996) or with a recent carbon sale by Costa Rica to Norway at an average price of $10/ton (R. Castro Salazar, personal communication, 1996). The average cost of carbon sequestration through adoption of reduced-impact logging was even lower with higher diameter limits. For a diameter cutting limit of 60 cm, reduced-impact logging would increase carbon storage at a cost of $3.3/ton.[9] This estimate is remarkably close to the figure proposed by Putz and Pinard (1993).

Our analysis has derived the cost of carbon sequestration in natural forests through management, per unit of land. Further research should have a broader scale, with the goal of deriving the *global* supply function for carbon sequestration. To this end, several considerations are in order. First, the results presented in this article are based on a model calibrated with data from a lowland tropical forest in Malaysia. This type of forest is different from hill dipterocarp forests and from lowland forests of other tropical regions. Differences in location, site quality, stand composition, structure, growth, and value of products are likely to affect the cost-effectiveness of carbon sequestration. The sensitivity of the cost of carbon sequestration to these variations could be addressed through comparative studies.

Second, our analysis has considered the trade-offs between financial returns (NPV/ha) and carbon sequestration. However, trade-offs exist also between timber and other forest products and services (watershed protection, biodiversity, recreational opportunities, etc.). For example, current research in Peninsular Malaysia by Mohd Shahwahid and colleagues points out the high cost of dredging induced by logging practices in the catchment area of the Ulu Langat Forest Reserve in the state of Selangor. There, the hill forest helps prevent sedimentation of mini-dams and regulate water flow. Preliminary results suggest that high dredging costs are caused by logging with conventional methods. Reduced-impact logging may reduce dredging costs by about 30 per cent. These examples confirm that environmental impacts must be accounted for and, as we expect them to benefit from more careful harvesting practices, their inclusion will make the cost-effectiveness of carbon sequestration even better.

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[9] This study focused on lowland tropical rainforests. It may be noted, however, that adoption of reduced-impact logging guidelines in hill forests may include slope restrictions (Pinard *et al.*, 1995) leading to losses in the total amount of harvestable timber. Under this circumstance, financial results would need to be adjusted accordingly.

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APPENDIX

Development of a steady-state constrained matrix growth model

Michie and Buongiorno (1984) review four methods of estimating transition probabilities from inventory data. The simplest approach is to compute the a_i and b_i probabilities as the proportions of trees that, during a time interval, stay in the same class or move on to the next one (method I). This is the method that has been most extensively applied (cf., e.g., Mendoza and Setyarso, 1986; Osho, 1991). However, applied to the Pasoh data set, it led to a model that, though accurate for short-term predictions, gave unacceptable steady states (e.g. with a negative number of trees in some classes). Such a model could not be used to analyse sustainable management regimes over long time periods. Similar problems in the development of a matrix model for a lowland tropical forest in Malaysia have been reported by Ingram and Buongiorno (forthcoming). They dealt with this problem by computing the parameters a_i and b_i with Equation (1) by assuming that they knew the steady state (the climax forest) and the mortality rates. A similar approach was applied by Houde and Ledoux (1995) on natural forest stands in Guyana. We also derived the matrix parameters from the steady state (including knowledge of ingrowth), by assuming that the upgrowth parameters, b_i, measured from the two censuses were representative of their true value, while the mortality rates were not. The basis for this assumption is that: (i) diameter growth (directly related to the b_i probabilities) can be estimated with more accuracy than tree mortality; (ii) growth rates were consistent with other growth and yield studies in Peninsular Malaysia; and (iii) anecdotal evidence suggests that windstorms are not uncommon in the area. Since no windstorm occurred between the two censuses, we hypothesized that

mortality between 1987 and 1990 was unusually low and could not be used for long-term predictions. Indeed, to predict the long-term effects of forest management, natural disturbances need to be accounted for, either as random events with a certain probability distribution or as (constant) expectations of long-term mortality rates.

Here, expected mortality rates were derived from the steady-state, or climax, forest. The concept of steady state has often been used in the past to verify long-term model reliability. It can also be used to infer the long-term mortality rates. Since the 50 ha plot is located in a pristine forest (Manokaran *et al.*, 1990), characteristics like basal area, diameter distribution and composition by broad species groups are believed to represent the result of a long evolutionary process. Therefore, the average plot condition per hectare was chosen as an approximation of the forest's steady state, i.e. its ecological climax.

Assuming that the parameters $b_{i,j}$ are independent of stand state (more on this below) and that they are measured accurately by the censuses allows the derivation of the parameters $a_{i,j}$ (and thus of mortality) from the diameter distribution ($y_{i,j}^s$) and ingrowth in steady state (I_i^s). From Equation (1):

$$
\begin{aligned}
a_{i,1} &= -\,(I_i^s - y_{i,1}^s)/y_{i,1}^s \\
a_{i,2} &= (y_{i,2}^s - y_{i,1}^s\, b_{i,2})/y_{i,2}^s \\
&\dots \\
a_{i,n} &= (y_{i,n}^s - y_{i,n-1}^s\, b_{i,n})/y_{i,n}^s
\end{aligned}
\qquad \text{(I.1)}
$$

which implies a mortality $m_{i,j} = 1 - a_{i,j} - b_{i,j+1}$. Table A1 shows all the transition probabilities for the matrix model. The mortality rates obtained in this way are higher than those observed between the censuses, as expected.

While the transition probabilities are independent of stand state in this approach, the ingrowth is not. To estimate the effects of stand density and composition on ingrowth (equation (3)) the 50 ha plot was divided into fifty subplots of 1 ha each. The results of the estimation of the ingrowth equation (3) from the subplot data are shown in Table A2. All coefficients had the expected sign: ingrowth of each species group was influenced negatively by the stand basal area, and positively by the number of trees in that group. The coefficients of determinations varied between 9% and 32%, so that most of the ingrowth variability was not explained by the model. This suggests that little may be lost by modelling ingrowth as a constant (Buongiorno *et al.*, 1995), at least for the O and N species. Much of the variation in ingrowth is probably random and hard to predict (Vanclay, 1992). Still, the effect of the independent variables on ingrowth was statistically significant at the 5% level in all three equations, suggesting a systematic effect of stand state on ingrowth.

The procedure to derive the $a_{i,j}$ coefficients relies on the assumption (shared by all fixed-coefficients matrix models) that mortality and growth rates are independent of stand density. This assumption was tested on the fifty 1 ha subplots. For each subplot, we measured basal area growth, and mortality rates for the three species groups (D, O and N). Mortality rates were then computed for two size classes (1–30 cm, and above 30 cm), while growth rates were averaged in four size classes (10–20, 20–30, 30–40 and

Table A1. *Transition probabilities*

		Dipterocarps (D)			*Commercial non-dipterocarps (O)*			*Non-commercial (N)*		
j	*Diameter class (cm)*	a_j	b_{j+1}	m_j	a_j	b_{j+1}	m_j	a_j	b_{j+1}	m_j
1	10–20	0.945	0.022	0.033	0.961	0.012	0.027	0.955	0.012	0.033
2	20–30	0.943	0.036	0.021	0.959	0.022	0.020	0.950	0.017	0.033
3	30–40	0.923	0.052	0.025	0.957	0.025	0.018	0.946	0.015	0.039
4	40–50	0.928	0.047	0.025	0.952	0.021	0.028	0.960	0.031	0.009
5	50–60	0.948	0.035	0.017	0.969	0.024	0.007	0.921	0.018	0.061
6	60–70	0.952	0.044	0.005	0.957	0.034	0.009	0.945	0.023	0.032
7	70+	0.981	0.000	0.019	0.962	0.000	0.038	0.977	0.000	0.023

Table A2. *Ingrowth equations, by timber group*

Dipterocarps (D)			
		Independent variable	
Statistics	*Constant*	*Basal area (m^2/ha)*	*D trees/ha*
Coefficient	$3.10 = \alpha_1$	$-0.10 = \beta_1$	$0.017 = \gamma_1$
Standard error	0.99†	0.04*	0.004†
R^2_{adj}	0.32		
Commercial non-dipterocarps (O)			
		Independent variable	
Statistics	*Constant*	*Basal area (m^2/ha)*	*O trees/ha*
Coefficient	$3.84 = \alpha_2$	$-0.13 = \beta_2$	$0.017 = \gamma_3$
Standard error	1.02†	0.05*	0.008*
R^2_{adj}	0.09		
Non-commercial (N)			
		Independent variable	
Statistics	*Constant*	*Basal area (m^2/ha)*	*N trees/ha*
Coefficient	$13.27 = \alpha_3$	$-0.30 = \beta_3$	$0.017 = \gamma_3$
Standard error	4.14†	0.11*	0.009
R^2_{adj}	0.15		

† Significant at the 1% level.
* Significant at the 5% level.

40+ cm). There was no statistically significant ($\alpha = 0.05$) relationship between mortality and stand basal area. There was a significant effect of density on growth rates only for a few sizes and species, and the effect was generally quite small.

Other studies of tree growth in tropical forests of South-east Asia also suggest that this assumption is plausible. Though Hutchinson (1979; cited by Panayotou and Ashton, 1992) reported very significant responses of diameter growth to thinning in Sarawak, Malaysia, other studies report that the effect of stand density on diameter growth may be smaller. Wan Razali (1988) found a statistically significant effect of stand basal area on diameter growth in Peninsular Malaysia, but this effect was very small. (Halving basal area from 25 to 12.5 m^2/ha would increase annual diameter growth by a fraction of a millimetre.) Primack *et al.* (1987) comparing logged and unlogged forests in Sarawak could not detect a significant difference in growth rates.

Perhaps the most complete study of tree growth in Peninsular Malaysia was Liu and Ashton's (1996). They developed a new index of inter-tree competition (neighbourhood pressure, *np*) and measured its effect (along with other dimensional, topographical and ecological variables) on tree growth in an unlogged primary forest. Their models explain much more diameter growth variation than previous studies. Yet they found that among over 500 species, less than half showed a statistically significant

effect of *np* on diameter growth, and in these cases this effect was small. For the species affected by competition, a 1% variation in *np* corresponded to a −0.06% variation in diameter growth. Thus, a 50% reduction in *np* would increase diameter growth by only 3%. A 30 cm dbh tree growing at 6 mm/year (or 2%/year) would, after a reduction of 50% of *np*, grow at a rate of 2.06% (= 2% + (3% of 2%)). In 20 years this difference amounts to about 0.5 cm, i.e., by ignoring the effect of competition on diameter growth one would predict a tree size of 44.6 cm instead of 45.1 cm. Therefore, the simplifying assumption of transition probabilities independent of stand state should not affect the overall validity of the model predictions, in this case.