Development Discussion Papers

Central America Project Series

**A Model to Predict Biomass Recovery
and Economic Potential
of a Tropical Forest**

Marco Boscolo, Joseph Buongiorno,
and Richard Condit

Harvard Institute for International Development

HARVARD UNIVERSITY

DEVELOPMENT DISCUSSION PAPERS
CENTRAL AMERICA PROJECT SERIES

A Model To Predict Biomass Recovery
and Economic Potential of a Tropical Forest

Marco Boscolo, Joseph Buongiorno, and Richard Condit

Abstract

This paper presents a neotropical forest growth model and applies it to estimate biomass loss and recovery given certain initial conditions and planned harvesting activities. The data used to parameterize the model gave a total biomass lower than in other locations in the region, even within the same life zone. This suggests that, even within life zones, different forests exhibit different levels of biomass and biomass growth. Indeed, when the model was applied to simulate biomass trajectories under different initial conditions, it also predicted a lower biomass growth than other studies. The model was then applied to evaluate different strategies to reduce carbon emissions, or increase carbon sequestration, by choosing appropriate harvesting activities. We found that delaying otherwise profitable activities in secondary and intervened forests yielded carbon benefits at low cost: almost $0/tonC in secondary forests and $7/tonC in intervened forest. Instead, modifying practices in primary forests would involve very large reductions in NPV from timber sales. Furthermore, preliminary findings also suggest that reducing logging damage could result in cost-effective reductions in carbon emissions.

Dr. Marco Boscolo is an Associate at the Harvard Institute for International Development.

Dr. Joseph Buongiorno is Professor of Forestry, Department of Forest Ecology and Management, University of Wisconsin, Madison.

Dr. Richard Condit is Director, Latin America Program, Center for Tropical Forest Science of the Smithsonian Tropical Research Institute, Panama.

Table of contents

List of Tables

I. Introduction

In Central America major opportunities to participate in global efforts to mitigate climate change reside in the improved management of their forest resources. Not only is the region generously endowed with forests, but their protection and effective management is consistent with the sustainable development priorities of the region. Forests provide raw material for developing industries (such as the furniture industry in Costa Rica) and fuelwood, satisfy an increasing demand for nature-based tourism, and protect critical watersheds. Most recently, the prospects of a new international market in "carbon offsets" give forests additional opportunities to increase the revenues of farmers, local communities, and Governments (Goldemberg 1998).

Yet, the consideration of forests as viable climate change mitigation tools depends on our ability to assess how much carbon they store and, when disturbed, how much carbon they release and subsequently absorb as biomass builds up. Methods to help in these assessments would be very valuable to estimate carbon emissions and sequestration due to land use changes (deforestation, secondary regeneration), and forest management, and to evaluate the effectiveness of alternative projects and programs.

Current research (Kerr et al., this volume) finds evidence that (1) land use conversions depend on the relative profitability of forest versus alternative land uses; (2) even temporary protection of carbon sinks from deforestation may yield lasting effects, and (3) there is scarcity of information regarding the rate of biomass accumulation in natural forests after varying degrees of intervention. These findings suggest that it is important to understand how forests respond to disturbance, and the economic potential of natural forests (whether virgin or disturbed) to be competitive with other land uses.

The goal of this paper is to present a tool that can help in estimating important data to inform forest managers and policy makers with respect to carbon issues (for example, in the development of emissions baselines or to quantify the impacts of changes in forestry practices and/or forestry returns). We describe a forest growth model based on transition matrices and parameterized from data from a 50-hectare plot located in Barro Colorado Island, in Central Panama. We then illustrate applications of this model to simulate carbon recovery trajectories

for secondary and intervened forest, and to predict management adaptations that can yield incremental carbon, and the trade-offs between biomass accumulation and economic returns.

The specific questions that we address are:

1. What is the biomass recovery of a natural forest after disturbance?

2. What, if any, management would help to get as close as possible to a certain level of biomass within a given time frame?

3. What losses in total above ground biomass[1] (TAGB) are expected from harvesting practices and what impacts (in economic and biomass terms) can be expected from modifying them?

The paper is organized as follows. First, we present a model developed from data from the most intensively monitored plot in the tropics, Barro Colorado Island, in central Panama. We describe the model structure and the data. Second, we apply the model to derive biomass accumulation trajectories depending on three initial conditions: intervened forest, secondary forest, and abandoned pastureland. While our results confirm the view that biomass growth is faster when the forest is far from a mature state, we predict a slower recovery than published data from direct measurements. For forest stands very far away from their natural state (young forest on abandoned pastureland) our results suggest that the speed of recovery depends much on the number of seedlings in the initial state. From this we infer that length of time under clearance (loss of seeds) and proximity to seed sources (tied possibly to the size of the clearing) are likely to affect the speed of recovery. Third, we evaluate which actions, if any, would allow a faster biomass accumulation. We consider this for different initial conditions. Finally, we evaluate whether there are possibilities of light extraction (as opposed to full set asides) that may still yield carbon benefits but also some income.

II. The Forest Model

Our ability to answer the above questions depends on our understanding of how forests respond to varying degrees of disturbance. To mimic the biomass recovery of a forest under different initial conditions we develop a forest growth model based on transition matrices, of the

---

[1] Total above ground biomass is measured as the biomass of living trees above 1 cm dbh. It therefore excludes carbon in roots, soil and litter, and in other components of the forest ecosystem.

kind originally developed by Usher (1966), and then refined by Buongiorno and Michie (1980) through incorporation of an ingrowth function dependent on stand characteristics.

In this model, the stand state is represented by the vector $\mathbf{y}_t = [y_{i,j,t}]$ 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 $\mathbf{h}_t = [h_{i,j,t}]$. 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:

$$y_{i,1,t+1} = (y_{i,1,t} - h_{i,1,t}) \, a_{i,1}(x) \qquad\qquad\qquad + (y_{i,2,t} - h_{i,2,t}) \, d_{i,1}(x) \qquad\qquad + I_i$$

$$y_{i,2,t+1} = (y_{i,1,t} - h_{i,1,t}) \, b_{i,2}(x) + (y_{i,2,t} - h_{i,2,t}) \, a_{i,2}(x) + (y_{i,3,t} - h_{i,3,t}) \, d_{i,2}(x)$$

$$y_{i,3,t+1} = (y_{i,2,t} - h_{i,2,t}) \, b_{i,3}(x) + (y_{i,3,t} - h_{i,3,t}) \, a_{i,3}(x) + (y_{i,4,t} - h_{i,4,t}) \, d_{i,3}(x)$$

....   (1)

$$y_{i,n,t} = \qquad\qquad\qquad\qquad (y_{i,n-1,t} - h_{i,n-1,t}) \, b_{i,n}(x) \qquad + y_{i,n,t} \, a_{i,n}(x)$$

where $a_{i,j}(x)$ is the probability that a live tree at time t will still be alive at time $t+1$ and in the same diameter class, $b_{i,j}(x)$ is the probability that a live tree at time t and in class $j-1$ will be in class j at time $t+1$, x refers to the forest state after harvest, $\mathbf{y}_t - \mathbf{h}_t$.

In this model we also considered $d_{i,j}(x)$, the probability that a live tree at time t and in class $j+1$ will be in class j at time $t+1$, to account for breakage in multiple-stemmed trees and tree "shrinkage". It is known that moisture conditions affect the girth of trees and that some trees have multiple stems. If the main stem dies off, another but smaller stem may be available for measurement at subsequent censuses[2]. In both cases, it appears as if a tree has reduced in diameter.

Similarly to Buongiorno et al. (1995), the parameters *a's*, *b's*, and *d's* are modeled as depending on species, size, and the stand state after harvest *(x),* expressed as either basal area or tree density. This formulation assumes a one-step transition probability, i.e., within one period trees do not growth more than one class. Indeed, within a given census interval, less than 700 trees (less than 0.3%) ever grew more than 10 cm.

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

---

[2] This is because multiple stemmed trees are described in the model as single-stem trees.

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

where:

$$\mathbf{G}_t = \begin{bmatrix} \mathbf{G}_{1,t} & & & \\ & \mathbf{G}_{2,t} & & \\ & & ... & \\ & & & \mathbf{G}_{m,t} \end{bmatrix} \qquad\qquad \mathbf{c}_t = \begin{bmatrix} \mathbf{c}_{1,t} \\ \mathbf{c}_{2,t} \\ ... \\ \mathbf{c}_{m,t} \end{bmatrix}$$

and where

$$\mathbf{G}_{it} = \begin{bmatrix} a_1(x) & d_1(x) & & \\ b_2(x) & a_2(x) & & \\ & ... & ... & \\ & & b_n(x) & a_n(x) \end{bmatrix} \qquad\qquad \mathbf{c}_{i,t} = \begin{bmatrix} I_{i,t}(x) \\ 0 \\ ... \\ 0 \end{bmatrix}$$

In this model, recruitment or ingrowth (the number of trees that enter the smallest diameter class between t and $t+1$) and transition probabilities (and the underlying growth and mortality rates) depend on stand state, so that the model is non linear.

Ingrowth of a species, i.e. the number of trees per unit area that enters the smallest size class between t and t+1 is expressed as a positive linear function of that species' abundance. Growth and mortality rates depend on species, size, and stand characteristics such as basal area or tree density.

Model parameterization

The model parameters were estimated with data from a 50-hectare demographic plot located in Barro Colorado Island (BCI), in central Panama. BCI is perhaps the most inventoried forest plot in the tropics. According to the Holdridge (1971) classification, BCI is in the tropical moist forest life zone, with geographic and climatic conditions quite common in other Central

American countries. For example, about 20% of the area of Costa Rica falls in this life zone (TSC 1993).

The 50-hectare plot in BCI has been censused four times, in 1982-83, 1985, 1990, and 1995. Therefore, three census intervals are available to estimate growth, mortality and recruitment parameters. During each census almost a quarter million 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 (CITE) while various issues related to plot establishment and management can be found in Condit (1998).

For parameter estimation, the over 300 tree species of the 50 ha plot have been divided into nine, 10 cm, diameter classes[3] and two species groups: canopy (c) and understory (u) trees. This distinction was based on maximum tree height according to Grut (1978). Species were classified as canopy if their maximum tree height was at least 20 meters, understory otherwise. This classification was chosen for simplicity. The maximum height of a given species could be estimated from a forest inventory that records, together with species and diameter size, tree height. Thus, it has a certain practicality. Furthermore, these different groups contain different levels of biomass.

To make the parameters from the various census intervals commensurate, the transition rates for 1982-85 were adjusted to five-year probabilities. Annual mortality m_1 was estimated from periodic mortality m_t as $m_1 = 1- (1- m_t)^{1/t}$. Then, five-year mortality was estimated as (but note Sheil and May 1996) $m_5 = 1-(1- m_1)^5$.

The probability of a tree to remain alive and in the same class (*a's*) was computed by dividing the number of survived trees in a class at time t+n by the number of trees at t. For example, the 3-year observed value from 1982-85 $a_{(82-85)i,j}$ was estimated as $a_{(82-85)i,j} = (y_{i,j,1985} | y_{i,j,1982}) / y_{i,j,1982}$. The 5-year estimate for $a_{i,j}$ was then obtained as $a_{i,j} = a_{(82-85)i,j}^{5/3}$. Five-years growth probabilities were then obtained from 1982-85 using $b_{i,j} = 1 - a_{i,j} - m_{i,j}$.

To estimate recruitment, growth and mortality, we divided the plot in 50 one-hectare subplots and regressed the 5-year recruitment, growth and mortality for either canopy or understory trees on stand state indicators (either basal area or tree density), and tree size. Results follow.

Results

Recruitment, mortality and growth equations were estimated for canopy and understory trees separately, by ordinary least squares, with 150 observations, each corresponding to 1 ha subplot in a given census interval.

Recruitment There was no statistically significant relation between recruitment rates and basal area, something that has been noted in previous studies on BCI. Instead, the number of trees in a species group was a strong predictor for the number of recruits, consistently with studies in other tropical forests (e.g., Boscolo, Buongiorno and Panayotou 1997). The recruitment models (Table 1) include year dummies for 1982 and 1985 whose coefficients are positive and significant, indicating that recruitment between the first two census intervals was significantly higher than the one recorded in 1995.

Mortality. As Table 2 shows, tree density was found to positively influence mortality in canopy trees, while basal area had the same effect on understory trees. As expected, mortality decreases with tree size but in a less than proportional manner. Mortality was also significantly higher during the census interval 1982-1985 when the region was affected by a severe drought (Hubbell and Foster 1990). Since the last size class (80+ cm) includes trees of very different sizes, a dummy has been included in the mortality equation. The (80+cm) dummy was significant for the canopy trees.

Growth. The results of the up-growth regressions are in Table 3. Basal area was found to negatively affect growth of both canopy and understory species, but in a less than proportional manner. A positive coefficient for basal area squared, in fact, suggests that basal area changes would have a stronger impact at low rather than at high levels. Growth increases with tree size, but in a less than proportional manner. Similarly to recruitment and mortality, annual growth between 1982-85 was higher than between 1985-95.

---

[3] The only exceptions are the first class, which starts at 1 cm and goes up to 9.9 cm, and the last one, that gathers all

Down-growth results are in Table 4. Although these transition probabilities are very small, they help explain the forest dynamics. Like mortality, the probability of stem breakage or shrinkage diminishes with tree size. Also similar to mortality, its rate is significantly lower for the largest size class.

The growth model based on the equations in Table 1 to 4 gave satisfactory short term and long-term projections of stand growth (see Boscolo, Buongiorno and Condit, in prep.).

III. Simulating Biomass Recovery

We applied the model presented above to simulate the biomass recovery of a forest stand that had been subjected to different degrees of intervention. We assumed four different initial conditions: (1) an intervened forest, (2) a natural secondary forest, (3) an abandoned pastureland with few tree seedlings (estimated at about 1000 seedlings/ha), and (4) abandoned pastureland with many tree seedlings (estimated at 2000 seedlings/ha). The difference between scenario (3) and (4) resides in the assumed intensity and length of the disturbance while the regeneration potential of the two systems is the same.

Initial conditions were adapted from Kleinn and Pelz (1994) which report average inventory estimates for intervened, secondary forests and abandoned pastureland (*potreros*). Kleinn and Pelz describe the forest stock as either "exploitable" or "non-exploitable" species. For our simulation, we assimilated "exploitable" trees to canopy species and the "non-exploitable" to understory.

To translate information about forest structure (number of trees by size class and species) into total above ground biomass, we computed the average biomass of trees in each species group and size class from the trees within 5 ha randomly selected within the 50 ha plot. The total biomass of a tree was computed as: $Y=\exp\{-2.134+2.530*\ln(dbh)\}$ (Brown 1997) where Y is the TAGB in Kg of dry matter per tree and dbh is the diameter at breast height (1.5 m.) in cm. The resulting biomass estimates are in Table 5. Carbon content was then estimated as half of the dry biomass (Brown 1997).

---

trees above 80 cm.

Primary Forest

Guynup, Condit and Michaels (1999) have compared different equations to predict the biomass of a primary forest. The equation we chose gives intermediate values among those reported by Guynup, Condit and Michaels (1999) for BCI. We estimated that the primary forest contains approximately 305 tons/ha. Most of this biomass is in canopy trees (250 tons). Left undisturbed, the model predicts a slight decrease in biomass levels overtime (in 30 years down to 285 tons/ha[4]). At 305 tons/ha, this plot appears to contain a lower biomass level than other forests of the same life zone in other countries. Helmer and Brown (1998), for example, report levels of potential biomass of 518 tons/ha for forest plots in Costa Rica. For forests in Panama and Nicaragua, however, our estimate is very close to the figures for high density-mixed moist forest reported in Brown (1997): 239-366 tons/ha (Panama) and 235-240 tons/ha (Nicaragua).

Intervened Forest

According to our estimates from the forest structure data by Kleinn and Pelz, an intervened forest had a level of biomass of about 186 tons/ha. Three-quarters of total biomass is in canopy trees (Table 5). Biomass recovery in intervened forests occurred slowly, at about 1.3 tons/ha/yr during the first 50 years. The rate of biomass accumulation was higher for canopy trees while, during the first 50 years, biomass in understory trees diminished slightly (see Figure 1).

Secondary Forest

Biomass recovery in secondary forest started from a level of about 90 tons/ha, almost entirely in canopy trees (Table 5). Biomass grew at a rate of approximately 2 tons/ha/yr during the first 100 years, almost twice the rate of intervened forests. Similarly to intervened forests, during the first 15 years, biomass of understory trees diminished slightly (see Figure 2).

---

[4] The model gives a steady state (convergence over an infinite period) with a total biomass of 266 tons/ha.

Abandoned Pastureland

Simulating the rate of biomass recovery in an abandoned pastureland is a long extrapolation for a model that was calibrated with data from an undisturbed forest. It is however, useful to assess the model plausibility for a wide range of initial conditions. Furthermore, the results illustrate the sensitivity of forest recovery to the initial regeneration condition, measured by the density of seedlings. While Kleinn and Pelz did not give information on seedlings (their inventory reports data started at 10 cm dbh) we consider here, for illustration a high and a low initial stock of seedlings. A high stocking of seedlings is likely found where pastoral activities have been conducted only for a short time. Conversely, low seedling stocking can be expected on pastureland where clearing took place long ago, and grazing has been heavy. However, we assume that in both cases, the potential regeneration is the same (there are the same seed sources, tract sizes are the same), and the growth potential is the same (land productivity is the same). As a result, the stand of trees will develop according to the laws of regeneration and growth embedded in the model.

These sole differences in initial conditions, other things being equal, produced very different rates of biomass accumulation. Initial conditions of ~1000 seedlings/ha produced biomass accumulation of approximately 3.8 tons/ha/yr during the first 50 years, almost double the rate in secondary forest (see Figures 3 and 2). But, if the abandoned pastureland had initially 2000 seedlings/ha the biomass accumulation was much faster: about 5 tons/ha/yr (Figure 4). Such figures are in the lower range of observed biomass accumulation in tropical regions (see, for example, Tosi (1997), MINAE (1997), or Weaver (1981)). This may be because the parameters of the model were estimated for conditions far from the simulation setting. Or it may reflect the lower growing potential of this forest in comparison with forests found in other locations in Central America. It should also be recognized that biomass growth can be enhanced or hampered by a host of variables not included in the model, such as soil fertility and/or compaction and distance from forest patches.

Despite those limitations, the model can help clarify whether there are simple management practices on variables fully recognized by the model (i.e. the dynamic density and composition of

the forest stand) that could increase the rate of biomass recovery, and thus the rate of carbon sequestration.

Formally, this problem was formulated as maximizing the stand biomass after a specified amount of time (say 30 years[5]), given a particular initial condition, possibly altered by an initial harvest (which could be nil). The problem is then to select some trees from a given initial set to try to maximize biomass accumulation in a fixed amount of time.

$$\max \text{Biomass}_{30}$$
$$\{h0\}$$
st forest model
$$y0 = \text{initial condition}$$

The results suggest that biomass could be increased slightly above the baseline by removing at year 0 all small understory trees. This action, which of course neglects cost considerations as well as implications for biodiversity, would create favorable conditions for more regeneration and growth of small canopy trees with increases in the objective function. It was found to (negligibly) increase biomass on stands with high abundance of understory seedlings (mature and intervened forest). An illustration for the intervened forest is given in Table 6.

IV. Economic Potential of a Neotropical Forest

Another issue of potential interest is whether opportunities exist to reduce the level of carbon emissions by modifying forest management practices. In this section, we present a simple model that can help in estimating the level of carbon emissions under different cutting scenarios and initial conditions.

All the simulations used a projection period of 30 years and cutting could occur at year 0, year 30, or both. Furthermore, the forest manager could choose to cut any tree whether of commercial value or not, canopy, or understory.

Economic and Environmental Criteria

<u>Timber revenues</u>. The net present value (NPV) from harvesting was defined as:

$$\text{NPV} = \sum_{t=0}^{T} \delta^t \left\{ \sum_{i,j} (P_{ijt} - C_{jt}) h_{ijt} - \text{F}_{\text{Ct}} \right\},\qquad\qquad [\text{X}]$$

where δ^t is the discount factor $1/(1+r)$, with r being the discrete discount rate; h_{ijt} is the number of trees harvested at time t of species group i and size j; P_{itj} is the market value of the logs in a tree of species group i ,diameter class $j,$ in year t; C_{jt} is the variable cost of felling a tree in diameter class j and transporting the logs from that tree to the market point (a mill or port); F_{Ct} is the fixed cost of logging.

The volume of commercial timber was estimated with the method described in Condit et al. (1999). They estimated bole length (B) for about 80 species in BCI with the equation: B = hmax*(1-exp(-a*dbh^b)) where hmax, a and b are estimated from field data. For this study, we used the parameters hmax, a and b developed by Guynup et al. (1999) and estimated bole length for each species and size class. We then obtained values applicable to canopy or understory categories as group averages. Bole volume was derived by multiplying basal area by bole length and by using a tapering factor of 0.6, obtained from direct measurements of a few fallen trees[6]. The results are in Table 6 and show timber volumes varying from 1 to 14 m3/tree for canopy species and 1 to 9 m3/tree for understory.

Stumpage values were derived from the average log price of semi-hardwoods trees which, in many parts of Central America comprise the majority of marketable trees. Based on Gamboa (1996a and 1996b), and CCF (1997), we estimated an average log price at the mill of $78/m3, in 1996. Stumpage values were then obtained by assuming harvesting costs of $32/m3[7] (including felling, skidding, loading transport and unloading) and fixed costs (primarily road construction,

---

[5] While a longer horizon could be chosen, thirty years goes far beyond the Kyoto protocol's first commitment period to demonstrate reduction in the emission of greenhouse gases.
[6] This is a rough estimate since fallen trees may not be representative of all standing ones.

legal, administrative and logistics costs) of \$120/ha, partly based on Gamboa (1996a and 1996b). For a harvest of 11 m3/ha, the above figures result in a stumpage value of \$35/m3, consistent with CCF (1997). The resulting stumpage values are in Table 6. They are positive only for commercial canopy trees of at least 30 cm in diameter[8].

For all the rest, the cost of harvesting exceeds tree value. In the simulation, stumpage values were assumed to remain constant, in real terms, and valued at 1996 prices and costs. The real rate of discount was set at 10% per year. In Costa Rica, higher rates are not uncommon (for example Castro [1999] uses two alternative rates of 5 and 12% to estimate the cost of biological carbon sequestration). We also assumed that harvesting operations, regardless of their intensity, caused a logging damage, killing 15% of the trees in the residual stand (Condit et al 1995).

Carbon emissions. Carbon emissions were defined as variations in the carbon stored in the biomass of living trees from one period to the next. As a result of logging, carbon is released in the atmosphere[9]. Following logging, the forest recovers and sequesters carbon through biomass build-up. The model records this changes as positive and negative emissions respectively. To account for carbon sequestration and release that occur at different points in time, carbon flows were discounted like timber values. The assumption is that an earlier sequestration, or emission prevention, is preferred to a later one with the rate of time preference measured by the same rate used to discount monetary values. The present value of carbon emissions was then defined as:

$$PVCE = \sum_{t=0}^{T} \delta^t \{ k_{ij} * 0.5 * \sum_{i,j} [h_{ijt} + \phi \, y_{ijt} \mid h_{ijt} > 0] + [y_{ijt-1} - y_{ijt}] \} \, , \qquad [2]$$

where k_{ij} is the biomass of a tree of species group i and size j (see table 5) and y_{ijt} is the number of trees of that species and size alive at time t, and ϕ is the logging damage rate.

The problem was formulated as

---

[7] This assumes a distance of 50-80 Km from the site of extraction and the sawmill. For higher distances, lower stumpage values would be obtained.

[8] For the secondary and the intervened forest we assumed that 5% of all canopy trees have commercial value. This is roughly the percentage of canopy trees that are of commercial value at BCI.

Max NPV

{ht}

st   Forest model

        $y0$ = initial conditions

        $PVCE \leq PVCE_0$

By varying the upper bound on the discounted carbon emission, $PVCE_0$ we could quantify the economic trade-off that exist between carbon emissions and timber revenues in the management of alternative forest stands.

Results

The results are summarized in Table 8, for three initial forest conditions, and selected constraints on carbon emission, $PVCE_0$,. The table shows the income obtained at year 0 and 30, the corresponding maximum NPV, and the present value of carbon emissions with a discount rate of 10% (the same rate used to discount monetary values) or 0%, which simply gives physical sequestration of the different alternatives.

The initial conditions for the secondary forest had just enough commercial volume to allow a positive harvest at year 0. NPV was, however, negligible. Postponing the harvest allowed a more profitable cut at year 30 (although in present value terms NPV was also small) but with a significant increase in carbon sequestration. Thus, almost 8 tons of discounted carbon/ha could be sequestered at no cost, in terms of foregone NPV (Table 8 and 9). This was the lowest cost of carbon sequestration, for all the initial conditions and harvests considered.

For the intervened forest, postponing an immediate profitable harvest resulted in larger revenues in 30 years (but smaller NPV). This loss in NPV (equal to $110/ha) was compensated by a change in discounted carbon flows from 11 tons/ha emitted to 4 tons sequestered (Table 8). Delaying harvest in the intervened forest would therefore become desirable for the forest owner

---

[9] It is assumed that the carbon reduction in the forest stand translates directly into carbon emissions (i.e., there is no long-term storage in end uses). This is a simplification that, as Boscolo and Buongiorno (1997) have shown, has minor implications for the final results.

if the modification could be compensated with more than $7.4/tonC (the opportunity cost of switching from immediate to delayed harvest, see Table 9).

In the case of a primary forest, strategies to reduce carbon emissions through modification of harvests gave much higher costs, above $100/tonC. This result is consistent with findings for other areas of the tropics (e.g., peninsular Malaysia, see Boscolo, Buongiorno and Panayotou 1997) where the most cost-effective way to reduce carbon emissions was to adopt better harvesting technologies. Recent research in Brazil suggests that logging damage in neotropical forests could be reduced at a negligible cost, if not at a profit (Holmes et al. 1999). Reducing logging damage from 15% to 5%[10] of the residual stand resulted in a reduction in carbon emissions of 12 tonsC/ha (Table 8). Therefore, if we assume that the cost of improving harvesting practices would be negligible, reducing logging damage could also be an extremely cost effective strategy to reduce carbon emissions.

V. Conclusion

In this paper we present a neotropical forest growth model to estimate biomass loss and recovery given certain initial conditions and planned harvesting activities. The data used to parameterize the model suggested that the estimated total biomass at the BCI plot is consistent with estimates from other forests in Panama and other locations in the region (e.g., Nicaragua, cf. Brown 1997). However, it was lower than other forests in Costa Rica, even within the same life zone. This suggests that, even within life zones (perhaps the most common climatic and ecological classification used in Central America) different forests have different levels of biomass and biomass growth. Indeed, when the model was then applied to simulate biomass trajectories under different initial conditions it also predicted a lower biomass growth than other Costa Rican studies.

The forest growth model was then applied to evaluate different strategies to reduce carbon emissions, or increase carbon sequestration, through modification of harvesting activities. We

---

[10] Unfortunately, we could not find any estimate for reduction of damage thanks to the adoption of reduced impact logging as opposed to conventional logging in Central America. The cited study by Holmes et al. (1999) in the

found that delaying otherwise profitable harvests in secondary and intervened forests yielded carbon benefits at low cost: almost $0/tonC in secondary forests and $7.4/tonC in intervened forest. Instead, reducing current harvests in primary forests would involve very large reductions in timber income. Furthermore, although more research is needed in this area, preliminary findings also suggest that reducing logging damage could be a cost-effective way of reducing carbon emissions.

Since transport costs are an important part of overall harvesting costs, the distance between the forest and the processing mill strongly affect stumpage values. Different stumpage values could easily be accommodated in this model to test the sensitivity of results to different distances to the market. This would allow comparisons of the profitability of forestry in different locations within a country, and to compare forestry with other land uses.

Another possible application of the model is to evaluate ecosystem restoration strategies. For example, one could ask which activities, if any, (such as thinnings or interplanting), would allow a degraded forest to get back to a nearly natural state within a given time. Preliminary results, not reported here, suggest that doing nothing may be the best strategy to achieve such restoration.

Most importantly for the purposes of this paper, the model has potential to develop baselines and scenarios in the design and evaluation of forestry projects. As climate change gains importance in international agreements and debates it allows forest managers and policy makers to incorporate carbon dioxide considerations based on better quantitative information on the opportunity costs of different approaches.

Eastern Amazon, the closest we could find to our area of interest, however, suggests that damage could be more than halved.

References

Boscolo M., J. Buongiorno, and T. Panayotou (1997), "Simulating Options for Carbon Sequestration Through Improved Management of a Lowland Tropical Rainforest", *Environment and Development Economics* 2(3):239-261.

Brown S. (1997), *Estimating biomass and biomass change of tropical forests: A primer*, FAO Forestry Paper No. 134. FAO, Rome.

Camara Costarricense Forestal (CCF) (1997), *Boletin Informativo Bimensual* 5(24).

Castro Salazar R. (1999), "Valuing the Environmental Service of Permanent Forest Stands to the Global Climate: The Case of Costa Rica', Ph.D. Dissertation. Harvard University.

Condit R. (1998), *Tropical Forest Census Plots: Methods and Results from Barro Colorado Island, Panama and Comparison with other Plots*, Springer-Verlag Berlin Heidelberg and R.G. Landes Company Georgetown, Texas, USA.

Condit R., S.P. Hubbell, and R.B. Foster (1995), 'Demography and Harvest Potential of Latin American Timber Species: Data from a Large, Permanent Plot in Panama', *Journal of Tropical Forest Science* 7(4):599-622.

Gersan J., A. Castillo, and G. Garcia Cano (1993) 'La economia de la explotacion maderera en la region oriental de Panama: Darien' Proyecto Regional De la Economia Politica de Desarrollo Sostenible en Centro America CRIES-CEASPA.

Goldemberg José (ed.), (1998), *The Clean Development Mechanism: Issues and Opportunities*, UNDP.

Grut (1978)

Guynup T, R. Condit, and G. Michaels (1999), 'Estimating Total Aboveground Biomass of a Panamanian Tropical Moist Forest' (manuscript).

Helmer E.H. and S. Brown (1998). "Gradient analysis of biomass in Costa Rica and a first estimate of country-wide emissions of greenhouse bases from biomass burning", In press in: C.A.S. Hall (Ed.) *Ecology of tropical development, the myth of sustainable development in Costa Rica*. Academic Press, New York.

Holdridge L.R., W.C. Grenke, W.H. Hatheway, T. Liang, and J.A. Tosi Jr. (1971), *Forest Environments in Tropical Life Zones*, Pergamon Press.

Holmes T.P. et al. (1999), 'Financial Costs and Benefits of Reduced-Impact Logging Relative to Conventional Logging in the Eastern Amazon' Final Report. Tropical Forest Foundation, Fundação Floresta Tropical, USDA Forest Service.

Hubbell S. P., and R.B. Foster (1990), 'Short term population dynamics of trees and shrubs in a neotropical forest: El Niño effects and successional change', *Ecology*.

Kerr S., A. Pfaff, A. Sanchez, and M. Boscolo, (this volume), 'What Role For Tropical Forests In Climate Change Mitigation? The case of Costa Rica'

Kleinn C. and D.R. Pelz (1994), *Inventario Forestal de la Zona Norte de Costa Rica*, COSEFORMA, Project Document No. 40, San Jose, Costa Rica.

Mendez Gamboa J.A. (1996) 'Costos de las Actividades de aprovechamento forestal en el bosque natural de la zona norte de Costa Rica' (manuscript)

Mendez Gamboa J.A. (1996) 'Determinacion de la rentabilidad del manejo del bosque natural en la zona norte de Costa Rica, en fincas propiedad de asociados de Codeforsa' (manuscript)

MINAE (Costa Rican Ministry of Environment and Energy) et al. (1997). "National Proposal for the Territorial and Financial Consolidation of Costa Rican National Parks and Biological Reserves". USIJI Project Proposal. Costa Rica.

Sheil D. and R.M. May (1996), 'Mortality and recruitment rate evaluations in heterogeneous tropical forests' *Journal of Ecology* 84:91-100.

Tropical Science Center (1993), *Mapa Ecologico de Costa Rica*, San Jose, Costa Rica.

Usher, M.B. (1966), 'A matrix approach to the management of renewable resources, with special reference to selection forests', *Journal of Applied Ecology* **3**:355-367.

Weaver, P. (1981) 'Growth and Yield, and sample plot techniques in indigenous forests of the American tropics', Paper prepared for the IUFRO/FAO meeting on Growth and Yield Studies on mixed tropical forests, University of the Philippines at Los Banos.

Table 1. Recruitment Equations, by Species Group (pooled data 1982-1995)

Variables	Coefficients	SE
Canopy trees		
Canopy trees/ha	.0845172	.015109***
Year 1982 dummy	48.25426	10.73535***
Year 1985 dummy	85.31619	10.73402***
Constant	-.981588	21.80391
Adj R2	0.3767	
Understory trees		
Understory trees/ha	.0840227	.0210184***
Year 1982 dummy	201.8628	24.11569***
Year 1985 dummy	267.5824	23.89644***
Constant	14.30978	76.0425
Adj R2	0.4911	

Source: Author's calculations

Table 2. Mortality Equations, by Species Group (pooled data 1982-1995)

Variables	Coefficients	SE
Canopy trees		
No. trees/ha	.0000116	2.42e-06***
Dbh	-.0046818	.0003579***
Dbh squared	.0000632	6.29e-06***
Year 1982 dummy	.0500149	.0024451***
Dbh 80+ cm dummy	-.1746181	.0284703***
Constant	.1102351	.0122195***
Adj R2	0.3259	
Understory trees		
Basal area/ha	.0003485	.0001378**
Dbh	-.0057742	.0005555***
Dbh squared	.0000758	.0000123***
Year 1982 dummy	.0118063	.0019404***
Constant	.178744	.0052715***
Adj R2	0.1935	

Source: Author's calculations

Table 3. Upgrowth Equations, by Species Group (pooled data 1982-1995)

Variables	Coefficients	SE
Canopy trees		
Basal area/ha	-.0045668	.0020442**
Basal area/ha squared	.0000594	.0000302**
Dbh	.0079544	.0003629***
Dbh squared	-.0000605	6.43e-06***
Year 1982 dummy	.0051465	.0021975**
Constant	.0660349	.0340156*
Adj R2	0.6669	
Understory trees		
Basal area/ha	-.0014981	.0008045*
Basal area/ha squared	.0000198	.0000121
Dbh	.0043576	.0003341***
Dbh squared	-.0000419	7.94e-06***
Year 1982 dummy	.00168	.000799*
Constant	.01159	.0132802
Adj R2	0.4113	

Source: Author's calculations

Table 4. Down-Growth Equations, by Species Group (pooled data 1985-1995)

Variables	Coefficients	SE
Canopy trees		
Dbh	-.0010431	.0004216**
Dbh squared	.0000256	5.70e-06***
Year 1985 dummy	.0094705	.0026395***
Dbh 80+ cm dummy	-.1022289	.0179235***
Constant	.0265911	.0062387***
Adj R2	0.1166	
Understory trees		
Dbh	-.0039167	.0012339***
Dbh squared	.0001297	.0000196***
Year 1985 dummy	.0106073	.0051512**
Dbh 80+ cm dummy	-.6650109	.1184652***
Constant	.0458001	.0158119***
Adj R2	0.2851	

Source: Author's calculations

Table 5. Biomass and Structure of Alternative Forest Stands

| Size class | Biomass (Tons/tree) | | Forest structure[2] (trees/ha) | | | | | |
| | | | Intervened forest | | Secondary forest | | *"Potrero"* | |
(cm)	*c*	*u*	*c*	*u*	*c*	*u*	*c*	*u*
1-9.9	0.004	0.002	470.6	219.1	354.1	152.6	-	-
10-19.9	0.222	0.089	69.5	32.3	48.0	20.7	3.4	0.0
20-29.9	0.477	0.395	29.3	13.6	17.4	7.5	2.9	0.0
30-39.9	0.944	0.883	19.2	7.0	8.8	2.5	2.0	0.0
40-49.9	1.767	1.657	13.7	5.0	4.4	1.3	1.2	0.0
50-59.9	2.889	2.946	8.2	3.0	3.2	0.9	0.5	0.0
60-69.9	4.365	4.786	3.8	1.4	2.4	0.7	0.4	0.0
70-79.9	6.643	6.151	1.9	0.7	0.9	0.2	0.5	0.0
80+	10.353	10.353[1]	1.1	0.4	1.1	0.3	0.2	0.0
Total biomass			*139*	*48*	*73*	*20*	*22*	*0*

Source: Adapted from Kleinn and Pelz (1994). We assumed correspondence of canopy and understory with "exploitable" and "non-exploitable" species.

Notes. [1] In the five hectares selected to estimate average biomass, there was no understory tree above 80 cm. Therefore, we assumed the same biomass level as canopy trees. [2] Kleinn and Pelz (1994) do not give the numbers of trees below 10 cm. We estimated them through extrapolation. We also assumed that in the abandoned pastureland there were no understory trees.

Table 6. Accelerating Biomass Recovery in Intervened Forests

| Size class (cm) | Initial state | | Harvest | | State at year 30 w/o harvest | | State at year 30 w/ harvest | |
	c	*u*	*c*	*u*	*c*	*u*	*c*	*u*
1-9.9	470.6	219.1	0.0	219.1	514.9	944.6	518.4	837.0
10-19.9	69.5	32.3	0.0	0.0	95.5	48.0	96.8	39.0
20-29.9	29.3	13.6	0.0	0.0	36.1	10.8	36.6	9.5
30-39.9	19.2	7.0	0.0	0.0	18.7	5.4	19.0	5.2
40-49.9	13.7	5.0	0.0	0.0	11.9	3.0	12.0	3.0
50-59.9	8.2	3.0	0.0	0.0	8.0	1.3	8.1	1.3
60-69.9	3.8	1.4	0.0	0.0	5.1	0.4	5.1	0.4
70-79.9	1.9	0.7	0.0	0.0	2.7	0.1	2.7	0.1
80+	1.1	0.4	0.0	0.0	3.1	0.1	3.2	0.1
Total biomass (tons/ha, c&u)	*186.4*				*202.4*		*202.9*	

Source: Author's calculations

Table 7. Timber Volumes and Stumpage Values (1996)

Size class (cm)	Timber volume (m3/tree)		Log value at mill ($/tree)		Harvesting costs[1] ($/tree)		Stumpage value ($/tree)		
	c	*u*	*c*	*u*	*c*	*u*	*c_com*	*c_ncom*	*u*
1-9.9	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
10-19.9	0.08	0.06	0.00	0.00	0.17	0.12	-0.17	-0.17	-0.12
20-29.9	0.39	0.24	0.00	0.00	0.80	0.50	-0.80	-0.80	-0.50
30-39.9	0.92	0.57	71.39	0.00	29.29	1.18	42.10	-1.89	-1.18
40-49.9	1.69	1.10	131.61	0.00	53.99	2.26	77.62	-3.48	-2.26
50-59.9	2.78	1.76	217.07	0.00	89.05	3.62	128.02	-5.73	-3.62
60-69.9	4.08	2.66	318.04	0.00	130.48	5.49	187.56	-8.40	-5.49
70-79.9	5.65	3.70	440.57	0.00	180.74	7.63	259.82	-11.64	-7.63
80+	14.01	9.00	1093.00	0.00	448.41	18.53	644.59	-28.87	-18.53

Source: Author's calculations

[1] Transport cost, variable in the region between $0.2-$0.3/m3/km (Gamboa 1996a and Gersan 1994), are a significant component of harvesting costs. The data reported here assume a distance of 50-80 Km from the site of extraction and the sawmill.

Table 8. Income and Carbon Emissions from Management

	Income at year 0 ($/ha)	Income at year 30 ($/ha)	NPV[1] ($/ha)	PVCE[2] (tons/ha)
Secondary forest	5	< 0	5	1.7 (-19.8)[3]
	-	95	5	-6 (-16.4)
(do nothing)	-	-	0	-6.8 (-28.6)
Intervened forest	124	<0	124	11 (-6)
	-	244	14	-3.8 (-0.1)
(do nothing)	-	-	0	-5.3 (-20.1)
Primary forest	1848	<0	1848	36.2 (28.9)
	1620	<0	1620	33.9 (27.5)
	1323	<0	1323	31.4 (25.7)
With logging damage reduced to 5%				
	1848	<0	1848	24.2 (21.2)
	1649	<0	1649	22.1 (19.7)
	1416	74	1420	20 (15.3)

Source: Author's calculations

[1] Net present value of timber income. [2] Present value of carbon emissions. A positive value means carbon emissions. A negative value means carbon sequestration. [3] Undiscounted carbon emission or sequestration.

Table 9. Cost Effectiveness of Alternative Activities

Activity	Change in NPV ($/ha)	Carbon emission reduction or increased sequestration as compared to baseline (tonsC/ha)	Cost effectiveness of carbon sequestration ($/tonC)
Delay harvesting activities on secondary forest to year 30	0	7.7	0
Delay harvesting activities on intervened forest to year 30	110	14.8	7.4
Reduce current harvest in primary forest in favor of larger cut at year 30	> 200	> 1	> 100
Adopt quasi-zero cost practices to reduce damage in logging mature forests	(unknown but presumably small)	12	0-5

Source: Author's calculations