

# Adaptation of a spatially explicit individual tree-based growth and yield model and long-term comparison between reduced-impact and conventional logging in eastern Amazonia, Brazil

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Received 28 August 2006; received in revised form 8 February 2007; accepted 9 February 2007

## Abstract

Timber logging is one of the main land uses in the Brazilian Amazon. Despite its recognized potential as a sustainable activity, logging is generally conducted in an unsustainable or predatory manner, with significant negative environmental impact. There is increasing pressure to adopt more sustainable practices and reduced-impact logging (RIL) is gaining acceptance as a more environmentally benign alternative to unplanned, conventional logging (CL). Comparisons of these two harvest systems have largely focused on differences in efficiency (financial) and immediate impacts on stand structure, residual timber stocks and the physical environment.

Growth and yield simulation models allow long-term predictions about the response of forests to disturbance from harvesting, information that is essential to determine the effects of different management systems and to establish systems for the regulation or control of timber that are compatible with sustainable forest management. We describe the calibration of a spatially explicit individual tree-based ecological model within the SYMFOR/SIMFLORA framework for forests located near Paragominas in eastern Amazonia. Data originated from an experiment comparing RIL to CL within an area of 73.5 ha monitored over a period of 10 years. We evaluated biological realism and accuracy of the model, concluding that forest dynamics are adequately represented.

This model was used to evaluate long-term effects of RIL and CL on the forest. Our results suggest that total and commercial volume recovery following RIL are faster than following CL; it takes 10 and 30–40 years to recover total and commercial volume under RIL while under CL it takes 35–40 and over 60 years, respectively. Despite benefits from RIL, as currently practiced this logging system does not result in long-term sustained timber yields, reinforcing results from previous studies indicating that RIL must be combined with appropriate systems for yield regulation.

In the absence of silvicultural treatments, the model suggests that successive RIL harvests at current intensities will produce stands dominated by pioneer species groups, with few trees belonging to the emergent species group, and an increased proportion of defective trees in commercial size classes. These changes in stand composition present challenges to the long-term financial viability of RIL as currently applied in the Amazon region.

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**Keywords:** Forest stand dynamics; Sustainable forest management; SYMFOR; SIMFLORA; Yield regulation; Brazil; Amazon; Reduced-impact logging

## 1. Introduction

The Amazon Basin contains the largest contiguous extant tropical forest in the world despite well-reported high

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deforestation rates. In the Brazilian Legal Amazon, the cumulative deforestation has been estimated at 650,000 km<sup>2</sup>, with an average of 18,000 km<sup>2</sup> lost each year (INPE, 2004). Logging is one of the main land uses in the region and plays an important role in the regional economy (Veríssimo et al., 1992; Lentini et al., 2003; AIMEX, 2005). Despite the potential for sustainable management of tropical forests, logging in Brazil has historically been a catalyst for land use conversion to extensive cattle ranching and slash-and-burn agricultural systems (Veríssimo et al., 1992, 1995; Schneider et al., 2000; Nepstad et al., 2001). Even when logging is not followed by deforestation, it often results in large-scale forest degradation (Nepstad et al., 1999; Gerwing, 2002; Asner et al., 2005). Considerable effort has been made to change this pattern. Recent examples include the proposed expansion of the National Forests system to create 5,000,000 km<sup>2</sup> of forest concessions in the Brazilian Amazon by 2010, with the stated goal of managing forest resources sustainably (Veríssimo et al., 2002); and market-based incentives such as Forest Stewardship Council (FSC) certification, under which approximately 1.3 million hectares of natural forests are actively managed across the region (FSC-Brasil, 2005).

In this context, timber yield regulation has become a major concern in the Brazilian Amazon. This combined with the desire to reduce the environmental impact of logging has led to the gradual adoption of reduced-impact logging (RIL) techniques as the industry gold standard (Dykstra and Heinrich, 1996; Sist et al., 1998). However, some evidence indicates that, even when current Brazilian forestry regulations are followed and RIL techniques are applied, sustainable timber production is not guaranteed (Schulze, 2003; Phillips et al., 2004; Schulze et al., 2005; van Gardingen et al., 2006). Although some studies have demonstrated the benefits of RIL compared to conventional logging (CL) in terms of lower immediate environmental impacts (Johns et al., 1996; van der Hout, 1999; Armstrong and Inglis, 2000; Asner et al., 2002, 2004a,b; Holmes et al., 2002; Pereira et al., 2002), the long-term effects of RIL on characteristics such as timber yield have so far been inferred rather than demonstrated for Amazonian forests.

Forest growth and yield models have proven to be an indispensable tool for generating long-term predictions of forest dynamics, which in turn are essential for evaluating logging practices and developing silvicultural systems compatible with sustainable forest management (Vanclay, 1994). Currently, published studies relating to forest growth modelling in the Brazilian Amazon are based on data from only three sites: the Tapajós National Forest near Santarém, Pará (Silva, 1989; Alder and Silva, 2000; Cunha et al., 2002; Phillips et al., 2004; van Gardingen et al., 2006); Jari Cellulose in Mazagão, Amapá (Alder and Silva, 2000; Phillips et al., 2004; van Gardingen et al., 2006); and the ZF-2 site near Manaus, Amazonas (Biot et al., 1997; Chambers et al., 2004). Silva (1989) demonstrated that the Tapajós forest cannot completely recover commercial volume following high intensity logging (75 m<sup>3</sup> ha<sup>-1</sup>) in proposed 33-year cutting cycles in the absence of silvicultural treatments and timber market acceptance of new species during future harvests. In the same region, Phillips et al. (2004) found

that current yield regulation (40 m<sup>3</sup> ha<sup>-1</sup> on 30-year cutting cycles) does not provide for long-term sustainable yields even when RIL techniques are adopted. In a related study, van Gardingen et al. (2006) found maximum commercial increments for Tapajós of 0.33 m<sup>3</sup> ha<sup>-1</sup> year<sup>-1</sup>, equivalent to a sustainable harvest of 10 m<sup>3</sup> ha<sup>-1</sup> over a 30-year cutting cycle, when harvests are limited to species currently selected for commercial exploitation. Based on the same dataset and location, Keller et al. (2004) concluded that shifts in species composition are likely as a result of logging and that current forest management practice is unlikely to be sustainable in Tapajós.

At the ZF-2 site near Manaus, Biot et al. (1997) showed that a second harvest of 26 m<sup>3</sup> ha<sup>-1</sup> (8 trees ha<sup>-1</sup>) will not be possible following a 25-year cutting cycle, even applying RIL and silvicultural treatments. Modelling results elsewhere, including tropical forests in other regions, have consistently suggested that RIL needs to be combined with yield regulation to ensure sustainable forest management (Kohler and Huth, 1998; Kammesheidt et al., 2001; van Gardingen et al., 2003).

SYMFOR is a modelling framework that combines a management model, allowing the user to specify silvicultural activities in mixed tropical forest (e.g., harvest, thinning, poisoning, enrichment planting), with an empirical spatially explicit individual tree-based ecological model, which simulates the natural processes of recruitment, growth and mortality. The SYMFOR model, originally designed for use in Indonesia (Phillips et al., 2003), was adapted for use in Guyana (Phillips et al., 2002) and then in the Brazilian Amazon (Phillips et al., 2004). In Brazil the model was further adapted by incorporating new management options and translating it into Portuguese to become the model SIMFLORA. To date, two studies have been published using SYMFOR and SIMFLORA using the Tapajós dataset (Phillips et al., 2004; van Gardingen et al., 2006).

The objective of this paper is to present the calibration and evaluation of SIMFLORA's ecological model for forests of the Paragominas region in eastern Amazonia and to use the model to predict the likely long-term effects of RIL and CL on timber yield and stand composition and volume. Application of the model at a new location, using a new dataset, will greatly enhance the relevance of results from SYMFOR/SIMFLORA enabling a discussion of more generalised conclusions.

## 2. Methods

### 2.1. Study site and data used

A new calibration of SIMFLORA was necessary due to significant differences between forests in the Paragominas region and those in the Santarém, Mazagão and Manaus regions, where previous modelling efforts occurred. Paragominas is located in eastern Amazonia, in Pará State (3°17'S, 47°37'W). Although the four sites are similar topographically (gently rolling) and edaphically (oxisols/yellow latosols), average annual rainfall is lower in Paragominas (1750 mm year<sup>-1</sup> compared with 1900–2110 mm year<sup>-1</sup> in Santarém, 2234 mm year<sup>-1</sup> in Mazagão and 2110–2610 mm year<sup>-1</sup> in Manaus) and the Paragominas region experiences a longer dry period (4–6 months with <50 mm

month<sup>-1</sup> compared with 2–3 months in Santarém, 1–2 months in Mazagão and 0 months in Manaus) (Uhl et al., 1988; Uhl and Kauffman, 1990; Silva et al., 1995; Holdsworth and Uhl, 1997; Alder and Silva, 2000; Chambers et al., 2000, 2001, 2004; Silva et al., 2001; Carvalho et al., 2004; Sombroek, unpublished).

The data for this study were derived from plots covering 73.5 ha divided equally into three treatments: RIL, CL and an unlogged area (control). This area is part of the Pilot Project of Forest Management initiated in 1991 by IMAZON (Instituto do Homem e Meio Ambiente da Amazônia). Logging took place in 1993. Complete inventories and measurements were conducted in 1993 (pre- and post-logging) and in 2003, providing a 10-year period in which stem diameter (at breast height, DBH) increment, mortality and recruitment were assessed. Detailed descriptions of logging damage, logging costs/benefits, and forest monitoring results from this site can be found elsewhere (Johns et al., 1996; Barreto et al., 1998; Vidal, 1998, 2004; Vidal et al., 1998, 2002; Schulze, 2003). In each treatment, two measurement methodologies were adopted:

- In a 5.25-ha (700 m × 75 m) intensive sampling plot (ISP), all trees ≥ 10 cm DBH were measured. This area was reduced to 4.9 ha (700 m × 70 m) for any analysis requiring competition index for trees 10–25 cm DBH since calculation of the competition index required an area divisible by 10 m × 10 m grid squares (see Section 2.2);
- In a 19.25-ha (700 m × 275 m) extensive sampling plot (ESP), measurements were made for trees of commercial species ≥ 10 cm DBH and for trees of non-commercial species ≥ 25 cm DBH.

The ecological model in SIMFLORA is spatially explicit and has three main components: growth, recruitment and mortality. Problems with the spatial coordinates of ca. 400 trees in the unlogged (control) ISP forced us to eliminate this dataset. ISP data from RIL and CL treatments and ESP data from all three treatments were used in model calibration. Data subsets were used to calibrate recruitment, growth and mortality sub-models, as described below. A total of 17,131 trees were recorded, distributed among 295 morpho-species in 47 families. Prior to logging, forests in the two treatment areas (RIL and CL) were similar in structure and species composition (Table 1).

## 2.2. Model description and calibration

The ecological model describes tree-level DBH increment (cm year<sup>-1</sup>), recruitment (annual probability of new trees ≥ 10 cm DBH appearing in the stand), and natural mortality (annual probability of a given tree dying). Each of these three processes was calibrated separately for each of the 10 species groups described below. A competition index allows for interaction between trees and hence among these processes. Only parameters used to calculate the competition index do not vary among species groups. The model simulates all processes using an annual time step. The methodology used in this study closely follows that described by Phillips et al. (2004); hence only modifications to methods employed in that study are described here in detail. All equations and methods to estimate parameters can be found in Table 2.

### 2.2.1. Species groups

The model has 10 species groups, with grouping based on mean DBH increment and 95th percentile of the diameter distribution ( $D_{95}$ ). A linear discriminant analysis was applied to assign species to groups already formed by the cluster analysis results of Phillips et al. (2004) in order to maintain groupings compatible with those in the previous study. This was done for all species with ≥ 10 trees using mean annual DBH increment and  $D_{95}$  as classifying variables. The remaining species were assigned, when possible, according to their scientific name and botanical family, by the following sequence of rules:

- (1) by complete species name (if the same species at Tapajós belonged to a group then the Paragominas species was added to that group);
- (2) by genus (if other species of the same genus at Tapajós or Paragominas belonged to a group then the Paragominas species was added to that group);
- (3) by family (if other species of the same family at Tapajós or Paragominas belonged to a group then the Paragominas species was added to that group);
- (4) for a small percentage of species with < 10 trees, either botanical identification was lacking or no other species belonging to the same family were in the dataset. These were assigned randomly to species groups.

Table 1

Description of ISPs under RIL and CL before logging (1993), based on trees ≥ 10 cm DBH (mean ± standard error)

Stand characteristics	RIL	CL
Total volume (m <sup>3</sup> ha <sup>-1</sup> )	232 ± 23	237 ± 17
Percent volume from pioneer species (groups 7 and 9)	19 ± 4	14 ± 2
Commercial volume (m <sup>3</sup> ha <sup>-1</sup> )	134 ± 19	114 ± 9
Dominant species	<i>Tetragastris altissima</i> , <i>Tachigalia paniculata</i> , <i>Lecythis idatimon</i>	<i>Tetragastris altissima</i> , <i>Lecythis idatimon</i> , <i>Manilkara huberi</i>
Percent species common to both areas (of the 25 most abundant species)	68	

The definition of pioneer species (groups 7 and 9) is given as part of the description of the ecological model.

Table 2  
Equations, variable definitions and methods used to estimate parameters

Description	Equations	Variables	Method to estimate parameters
Competition index calculated with spatially explicit neighbor tree information	$C_i = z_1 \sum_{j=1}^{n_1} \frac{D_j}{D_i} + z_2 \sum_{k=1}^{n_2} \frac{D_k}{D_i} + z_3 \sum_{l=1}^{n_3} \frac{D_l}{D_i}$	$C_i$ is the diameter-dependent competition index of subject tree $i$ ; $z_1, z_2$ and $z_3$ are the coefficients of relative importance of zones 1, 2 and 3, respectively; $n_1, n_2$ and $n_3$ are the total number of over-topping trees (those with DBH larger than tree $i$ ) in zones 1, 2 and 3, respectively; $D_j, D_k$ , and $D_l$ are the DBH of tree $j, k$ , and $l$ (over-topping trees in zones 1, 2 and 3, respectively); $D_i$ is the DBH of subject tree $i$	Parameters $z_1, z_2$ and $z_3$ were set to 9, 4 and 1, respectively, as used by Phillips et al. (2004), to represent decreasing competition between trees as distance increases
Competition index modeled only with the subject tree's DBH	$\hat{C}_i = \frac{c_0}{c_1 + D_i} + c_2$	$\hat{C}_i$ is the competition index estimated as a function of DBH ( $D_i$ ); $c_0, c_1$ and $c_2$ are the model parameters	Non-linear least-square regression
Residual difference between the two competition indexes above	$\tilde{C}_i = C_i - \hat{C}_i$	$\tilde{C}_i$ is the diameter-independent competition index	
Growth model	$I_i = D_i(a_1 + a_2e^{-a_3D_i}) + a_4\tilde{C}_i + a_5$	$I_i$ is the annual diameter increment (cm year <sup>-1</sup> ); $a_1, a_2, a_3, a_4$ and $a_5$ are model parameters	Non-linear least-square regression
Recruitment model	$F = r_1 + r_2I_{10}$	$F$ is the annual recruitment probability; $r_1$ and $r_2$ are model parameters; $I_{10}$ is the estimated annual diameter increment of a hypothetical tree (10 cm of DBH) in the centre of the grid-square	Least-square linear regression where each point was weighted by the number of grid-squares in each class. Bootstrapping was used to determine parameter values and confidence limits
Mortality probability calculation (Sheil et al., 1995)	$m_i = 1 - [1 - (N_0 - N_t)/N_0]^{1/t}$	$m_i$ is the annual natural mortality probability; $N_0$ is the initial number of trees; $N_t$ is the number of remaining live trees in year $t$	
Mortality model	$m_i = \begin{cases} b_1 & \text{if } D_i < b_m + 10 \\ b_2 + b_3D_i & \text{if } D_i \geq b_m + 10 \end{cases}$	$m_i$ is the annual natural mortality probability for the $i$ th tree; $b_1, b_2$ , and $b_3$ are model parameters; $b_m$ is the width of the first diameter class	The lowest diameter class upper limit ( $10 + b_m$ ) was determined as the 5th percentile of the diameter distribution of each species group. The mean mortality rate for this diameter class was $b_1$ . Parameters $b_2$ and $b_3$ were estimated by least-square linear regression where each point was weighted by the number of initial trees in that given size class. Bootstrapping was used to determine these parameter values and confidence limits

Table 3  
Strategies to assign species to groups, species selection criteria, number ( $N$ ) and percentage of species and trees assigned by strategy

Assignment strategy	Selection criteria	Species		Trees	
		$N$	%	$N$	%
Discriminant analysis with mean DBH increment and $D_{95}$	$\geq 10$ trees/species	105	35	15,422	90
By species name	$< 10$ trees/species in Paragominas but $> 50$ trees/species in Tapajós	19	6	120	1
By genus	$< 10$ trees/species in Paragominas but species of the same genus with $> 10$ trees in Paragominas or $> 50$ trees in Tapajós	66	22	1,017	6
By family	$< 10$ trees/species in Paragominas but species of the same family with $> 10$ trees in Paragominas or $> 50$ trees in Tapajós	44	15	156	1
Random	All remaining species	61	21	416	2
Total		295	100	17,131	100

Table 4  
Summary of species group characteristics

	Group name (reference)	Dominant species	$N_t$	$N_s$
1	Slow-growing mid-canopy	<i>Lecythis idatimon</i> , <i>Cordia bicolor</i> , <i>Pouteria lasiocarpa</i>	4130	32
2	Slow-growing understory	<i>Poecilanthe effusa</i> , <i>Theobroma speciosa</i> , <i>Vismia guianensis</i>	586	14
3	Medium-growing mid canopy	<i>Chrysophyllum lucentifolium</i> , <i>Eschweilera blanchetiana</i> , <i>Laetia procera</i>	2926	46
4	Slow-growing lower canopy	<i>Diospyros dukei</i> , <i>Protium tenuifolium</i> , <i>Cordia scabrida</i>	775	24
5	Medium-growing upper canopy	<i>Tetragastris altissima</i> , <i>Pouteria reticulata</i> , <i>Lecythis lurida</i>	1583	25
6	Fast-growing upper canopy	<i>Pseudopiptadenia suaveolens</i> , <i>Copaifera duckei</i> , <i>Carapa grandifolia</i>	609	6
7	Fast-growing pioneers	<i>Inga alba</i> , <i>Cecropia sciadophylla</i> , <i>Guateria schomburgkiana</i>	4342	44
8	Emergents–climax	<i>Manilkara huberi</i> , <i>Hymenaea courbaril</i> , <i>Tabebuia impetiginosa</i>	962	24
9	Fast-growing pioneers	<i>Cedrela odorata</i> , <i>Carapa guianensis</i> , <i>Jacaratia espinosa</i>	133	10
10	Very fast-growing upper canopy	<i>Tachigalia paniculata</i> , <i>Parkia nitida</i> , <i>Tachigalia alba</i>	1085	16

$N_t$  is the number of trees and  $N_s$  is the number of morpho-species in each group

The number of species and trees assigned by each rule and the criteria adopted are presented in Table 3. Although these assumptions are progressively less trustworthy, in general the basic population dynamics of a given species are fairly consistent among sites in eastern Amazonia (Schulze, 2003) and elsewhere (Finegan et al., 1999), and species groupings based on genus or botanical family are not rare in the literature (e.g., Baker et al., 2004; Laurance et al., 2004). Moreover, species that were objectively classified ( $N \geq 10$  trees) represented 90% of the total stem number. The resulting species group characteristics and their associated dominant species are shown in Table 4.

### 2.2.2. Competition index

The competition index was determined using competition intensity zones where Zone 1 has an area of 100 m<sup>2</sup>, Zone 2 has an area of 800 m<sup>2</sup> surrounding Zone 1, and Zone 3 has an area of 1600 m<sup>2</sup> surrounding Zones 1 and 2. The potential competitive effects on trees in Zone 1 decrease from Zone 1 to 3 and competition from trees outside this 2500-m<sup>2</sup> area (Zones 1 + 2 + 3) is considered negligible (for a illustrative figure, see Phillips et al., 2004). A diameter-dependent competition index for the  $i$ th subject tree,  $C_i$ , was calculated as a function of the relative size of over-topping trees (those with DBH larger than the  $i$ th tree) in the three zones (Eq. (1), Table 2). This diameter-dependent competition index was then modelled as a function of the subject tree's DBH ( $D_i$ ), generating the modelled competition index  $\hat{C}_i$  (Eq. (2), Table 2). The competition index effectively used by SIMFLORA ( $\tilde{C}_i$ ) is called the “diameter independent competition index” because it is the portion of the competition index that is unexplained by the fitted inverse function of DBH (Eq. (3), Table 2). Parameters  $c_0$ ,  $c_1$  and  $c_2$ , were determined based on DBH and competition index  $C_i$  of trees in 4.9 ha ISPs and of trees  $\geq 25$  cm DBH in 19.25 ha ESPs.

### 2.2.3. Growth model

The growth model describes annual diameter increment  $I_i$  (cm year<sup>-1</sup>) as a function of tree diameter  $D_i$  and the diameter-independent competition index  $\tilde{C}_i$  (Eq. (4), Table 2). As with the competition model, the dataset used for the growth model came from 4.9-ha ISPs and from trees  $\geq 25$  cm DBH in ESPs.

Outliers were removed through graphical evaluation of increment versus DBH. In order to prevent trees growing to unrealistic sizes, we assumed that diameter increment declined to zero for trees with DBH greater than or equal to the 99th percentile of the species group diameter distribution.

### 2.2.4. Recruitment model

Data used to calibrate the recruitment model came from the 4.9-ha ISPs. Trees were considered false recruits (i.e., incorrectly omitted from the 1993 inventory) if recruitment to the DBH measured in 2003 would have required an annual increment  $>3$  cm year<sup>-1</sup>. This annual diameter increment was considered highly improbable since the 99th percentile of the observed increment for all trees  $>25$  cm DBH was equal to 1.93 cm year<sup>-1</sup>.

For each species group the following procedure was adopted. First, the number of recruited trees was determined along with the competition environment for each 10 m  $\times$  10 m grid square. The estimated annual diameter increment,  $I_{10}$  (determined by the growth model) of a 10-cm DBH hypothetical tree of the same species group in the centre of the grid square was used as a proxy for seedling/sapling growth, reflecting competition environment. Then the grid-squares were grouped into increment classes and the recruitment probability in each class was calculated as the sum of recruited trees divided by the number of grid-squares. This result was divided by 10 years to generate the annual recruitment probability ( $F$ ). The recruitment model relates annual recruitment probability ( $F$ ) to  $I_{10}$  (Eq. (5), Table 2).

In calibrating the recruitment probability function, the distribution of residuals revealed that regression assumptions of normality and homocedasticity were violated. Parameters and confidence limits were therefore estimated through bootstrapping. Briefly, 1000 data were drawn at random with replacement to estimate regression parameters. Mean parameter estimates were determined after repeating this procedure 1000 times, with 25th and 975th ranking estimates taken as 95% confidence limits.

The necessary time in years ( $T_i$ ) for a tree to reach the 10-cm minimum recruitment DBH in a cleared area (skid trails, recent gaps) was estimated based on the predicted increment of a tree with 10 cm DBH and a diameter-independent competition

index value of  $-172$  (low competition environment in unlogged forest).

#### 2.2.5. Natural mortality model

Annual natural mortality probability for the  $i$ th tree ( $m_i$ ) is calculated based on the initial number of trees ( $N_0$ ) and the number of remaining live trees in year  $t$  ( $N_t$ ) (Eq. (6), Table 2; Sheil et al., 1995). This probability was modelled differently in two size classes: a fixed probability for trees with DBH less than  $b_m + 10$  where  $b_m$  is the width of the lowest diameter class; and probability as a linear function of  $D_i$  for trees with DBH greater than  $b_m + 10$  (Eq. (7), Table 2).

For trees with  $D_i$  greater than  $10 + b_m$ , trees were grouped into DBH classes using percentiles of diameter distribution to define size-class widths, in order to avoid having classes without trees or with very few trees. One size-class boundary was set to 25 cm in order to use the different datasets (ISP and ESP) for model calibration.

#### 2.2.6. Damage mortality model

Falling trees create gaps in the forest and gap size depends on tree size and liana abundance. In areas with abundant lianas, gaps were represented by a rectangular damaged area with length equal to tree height plus three times crown radius and width equal to four times the crown radius (Phillips et al., 2004). In areas free of lianas or where lianas were cut, gaps were represented by a smaller trapezium-shaped area with length equal to tree height and maximum width equal to crown radius (Phillips et al., 2003). Among the trees that died naturally, only those  $\geq 80$  cm DBH had the potential to fall and create a gap (Phillips et al., 2004).

#### 2.2.7. Other functions

Other functions from SIMFLORA (e.g., DBH-height relationship, volume equation, and crown radius; Phillips et al., 2004) were not re-calibrated due to the absence of more site-specific data. Since these are secondary functions that do not interfere with the ecological model, we assumed that they are equally valid for Paragominas and Tapajós forests.

### 2.3. Evaluation of the model

The performance of the model was evaluated to determine biological realism and model accuracy. Biological realism was evaluated by comparing stand- and species group-level tree density and basal area during 100-year simulations of undisturbed forest. The model was judged to be biologically sound if the undisturbed forest simulations exhibited a relatively constant stand- and species group-level tree density and basal area. This was defined as a percentage change lower or equal to 100% (either at stand- or species-group levels), calculated as  $[(X_{100} - X_0)/X_0] \times 100$  where  $X_t$  is either tree density or basal area and  $t$  is time.

Model accuracy was evaluated by comparing predicted with observed forest dynamics over the 10-year (1993–2003) period. First we analyzed the accuracy of each sub-model for each species group. Data were extracted to describe the change in

basal area resulting from the growth of live trees (growth sub-model), total basal area of recruited trees (recruitment sub-model), and basal area of trees that died (mortality sub-model). Then we evaluated how well the model predicted overall annual basal area increment for each species group, which is numerically equivalent to subtracting mortality from the sum of growth and recruitment. Fifteen repetitions were done and the mean result over all repetitions was determined.

### 2.4. Evaluating long-term effects of RIL and CL

The impact of RIL and CL treatments were compared by projecting results of the logging experiment using 2003 data to initialize model simulations. In the first scenario we evaluated forest stand dynamics without further harvests. A second set of simulations considered repeated harvests applying the current yield regulation system for Brazil, setting the maximum logging intensity to  $40 \text{ m}^3 \text{ ha}^{-1}$  and cutting cycle to 30 years (Barreto et al., 1998; Phillips et al., 2004; Grogan et al., 2005). Although we recognize that a change to Brazilian forest legislation recently raised the minimum logging diameter to 50 cm DBH for all species (Brasil, 2006), the second set of simulations was conducted with a minimum logging diameter of 45 cm DBH in order to use the same yield regulation under which the original logging in 1993 took place (Johns et al., 1996) and to allow a direct comparison with earlier studies. Also, sensitivity analysis with the SIMFLORA model indicated that this change in minimum logging diameter did not alter the general trends we have identified.

In SIMFLORA a random number between 0 and 1 is assigned to all trees to represent stem quality. Only trees belonging to species listed as commercial in 1993 for the Paragominas region and without defect (minimum quality set to 0.3 to represent 30% of trees being of insufficient quality to be considered commercially viable) qualified for logging in RIL. The same rules were applied in CL except that defective trees were also logged (Holmes et al., 2002). RIL was simulated with 3.1-m wide skid trails and reduced felling damage reflecting liana cutting and directional felling while CL was simulated with 3.9-m wide skid trails and high felling damage (Johns et al., 1996). Also, skid-trail layout differed between logging systems; branched skid-trails were simulated in RIL, forming primary and secondary skid trails, while in CL straight skid-trails were simulated, linking every logged tree to an arbitrarily determined “access point” where skid-trails were designated to leave the plot.

Ten plots, five for each logging system, were simulated for 50 years beyond the 10-year observation period. We restricted simulations to this temporal window because uncertainty increases with the length of simulation, especially considering poor performance of the recruitment model, typical of most forest growth models (Porte and Bartelink, 2002). In addition, 60 years (10 years from experiment plus 50 years of simulation) represents a multiple of the 30-year cutting cycle, allowing evaluation of results from two complete cutting cycles. Finally, the Indonesian model in SYMFOR is known to produce reliable simulations for at least 50 years (van Gardingen et al., 2002).

Fifteen repetitions were done for each plot and averages were calculated by plot. The mean and standard error over all plots were then determined following the approach developed by van Gardingen et al. (2003).

### 3. Results and discussion

#### 3.1. Model calibration

The competition index parameters  $c_0$ ,  $c_1$  and  $c_2$  were estimated as 2832.5,  $-5.49$  and  $-56.38$ , respectively. The initial dataset characteristics as well as parameter estimates for the growth, recruitment and natural mortality models are shown in Table 5. With respect to growth, the species grouping (Table 4) and growth model calibration resulted in an overall  $R^2$  of 35%. A similar percentage of variation explained by the model was determined from analysis of the Tapajós dataset (Phillips et al., 2004) and elsewhere using other models (Gourlet-Fleury and Houllier, 2000).

Predicted felling gaps without liana cutting under CL were originally too large ( $903 \pm 11 \text{ m}^2$ ;  $\pm$  values indicate one standard error throughout the text) compared to those observed

in the field under CL in 1993 ( $355 \pm 32 \text{ m}^2$ ; Johns et al., 1996). The modelled damage was modified, changing the size of the damage rectangle (new dimensions: height = tree height + crown radius, width =  $2 \times$  crown radius), to generate average gap sizes ( $368 \pm 4 \text{ m}^2$ ) similar to those reported by Johns et al. (1996).

#### 3.2. Model evaluation

The biological realism evaluation (Fig. 2) demonstrated that the model exhibited considerable changes in species group composition over time. In particular, there was an increase in tree density and basal area of species groups 6 and 9, resulting in an overall increase in stand-level tree density and basal area as simulation time elapsed ( $0.65 \pm 0.01 \text{ trees ha}^{-1} \text{ year}^{-1}$  and  $0.07 \pm 0.002 \text{ m}^2 \text{ ha}^{-1} \text{ year}^{-1}$ , on average). Also, although stand-level tree density stabilized at approximately 560 trees  $\text{ha}^{-1}$ , basal area steadily increased with simulation length. Based on the assumption that unlogged forests are in dynamic equilibrium (defined here as a relatively constant tree density and basal area at stand and species group levels), the recruitment parameters of species groups 6, 7 and 9 were

Table 5

Number ( $N$ ) and percentage of data and model parameter estimates by species group for growth (Eq. (4), Table 2), recruitment (Eq. (5), Table 2) and natural mortality (Eq. (7), Table 2) models

Species group	Growth model								Recruitment model					Mortality model					
	Trees		Parameter estimates					$I$ (cm year <sup>-1</sup> )	Recruits		Parameter estimates			Deaths		Parameter estimates			
	$N$	%	$a_1$	$a_2$	$a_3$	$a_4$	$a_5$		$N$	%	$r_1$	$r_2$	$T_i$ (years)	$N$	%	$b_m$	$b_1$	$b_2$	$b_3$
1	1913	32	0.420	-0.425	0.000	0.000	0.287	0.22	257	13	0.025	0.005	42	783	29	3.0	3.596	4.103	-0.066
2	231	4	-0.552	0.551	0.000	0.000	0.067	0.09	158	8	0.002	0.198	175	153	6	1.9	5.051	1.578	0.268
3	1163	19	0.005	0.059	0.057	0.000	-0.191	0.28	186	9	0.027	-0.039	52	398	15	3.8	3.508	3.337	-0.047
4	306	5	0.004	0.000	0.000	0.000	0.138	0.20	153	8	0.042	-0.151	56	176	7	2.5	4.057	5.222	-0.075
5	697	12	-0.211	0.210	0.000	0.000	0.259	0.27	75	4	-0.006	0.056	40	137	5	5.0	2.482	1.650	-0.006
6	156	3	-0.006	0.000	0.000	0.000	0.798	0.60	159	8	-0.059	0.101	14	37	1	3.8	1.987	2.404	-0.020
7	919	15	-0.004	0.000	0.000	-0.001	0.642	0.54	939	46	0.019	0.123	13	767	29	3.0	4.471	3.707	0.012
8	352	6	-0.003	0.011	0.011	0.000	0.230	0.38	24	1	0.002	0.000	33	61	2	3.8	1.455	1.719	-0.016
9	35	1	0.073	0.826	0.081	-0.001	-3.888	0.65	35	2	0.004	0.000	15	21	1	3.8	6.003	3.155	0.029
10	195	3	-0.008	-0.423	0.132	-0.001	1.724	0.96	54	3	0.003	0.004	15	137	5	3.0	2.764	3.245	-0.012
Total	5967	100							2040	100				2670	100				

The column  $I$  shows the average growth rate predicted for each group. Table 3 describes each of the groups and their dominant members.

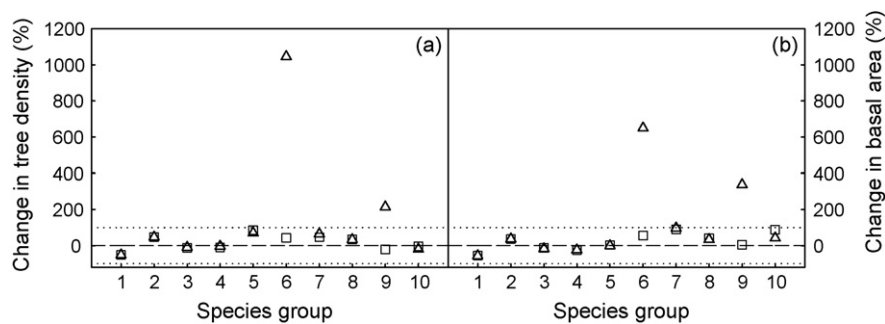


Fig. 1. Biological realism evaluation of the model calibrated for the Paragominas site, showing percentage change in (a) tree density and (b) basal area at the end of a 100-year simulation of undisturbed forest. Results from the original calibrated model ( $\Delta$ ) are compared to those obtained with fine tuned parameters ( $\square$ ). Data are shown by species group. Dotted lines represent the 100% threshold.

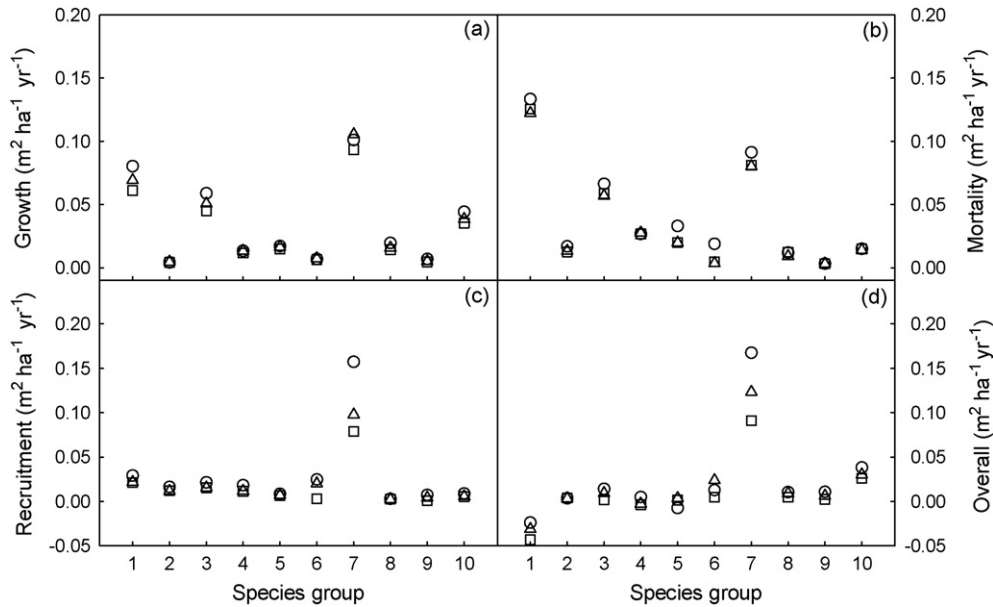


Fig. 2. Sub-model and overall basal area increment accuracy by species groups. Observed results (○) are contrasted to the original calibrated model estimates (△) and with fine tuned parameters (□). (a) Growth sub-model; (b) recruitment sub-model; (c) mortality sub-model; (d) overall basal area increment synthesizing the three sub-models.

adjusted through fine-tuning whilst remaining within their respective 95% confidence limits (new  $r_2$  values equal to 0.082, 0.0741 and  $-0.0075$ , respectively) in order to avoid the increase in their tree density and basal area. As a consequence, stand-level tree density and basal area exhibited only a slight change ( $-0.15 \pm 0.02$  trees  $\text{ha}^{-1} \text{year}^{-1}$  and  $0.02 \pm 0.001 \text{ m}^2 \text{ ha}^{-1} \text{year}^{-1}$ , on average), stabilizing at approximately 480 trees  $\text{ha}^{-1}$  and  $22 \text{ m}^2 \text{ ha}^{-1}$  (Fig. 1).

The evaluation of model accuracy demonstrated that the model accurately predicts growth for each species group (Fig. 2a). However, accuracy of the mortality and recruitment model was somewhat lower, possibly as a consequence of the limited data available for calibration and highly stochastic nature of these ecological processes (Fig. 2b and c). This is an issue in virtually all growth and yield models and is a function of data availability rather than the modelling process (Vanclay, 1994; Porte and Bartelink, 2002). We conclude that the model with the fine-tuned parameters is biologically sound and that the overall dynamics of each species group are adequately represented (Fig. 2d). As a consequence, all subsequent simulations were done with the fine-tuned parameter set (Fig. 2).

### 3.3. Evaluation of long-term effects of RIL and CL

#### 3.3.1. Stand recovery from initial harvest

Fig. 3 shows the projected recovery following a single harvest using RIL and CL. These simulations revealed important differences with respect to stand-level and commercial stock (defined onwards as trees  $\geq 45$  cm DBH from commercial species) recovery. The forest logged using RIL techniques recovered stand-level volume within 10 years (observed data). In the forest subjected to CL, recovery was projected to occur only 35–40 years after initial logging.

Differences between RIL and CL were more pronounced with respect to commercial stock recovery. Commercial stock recovery in RIL was projected to occur after 30–40 years (allowing for the level of uncertainty indicated by the standard

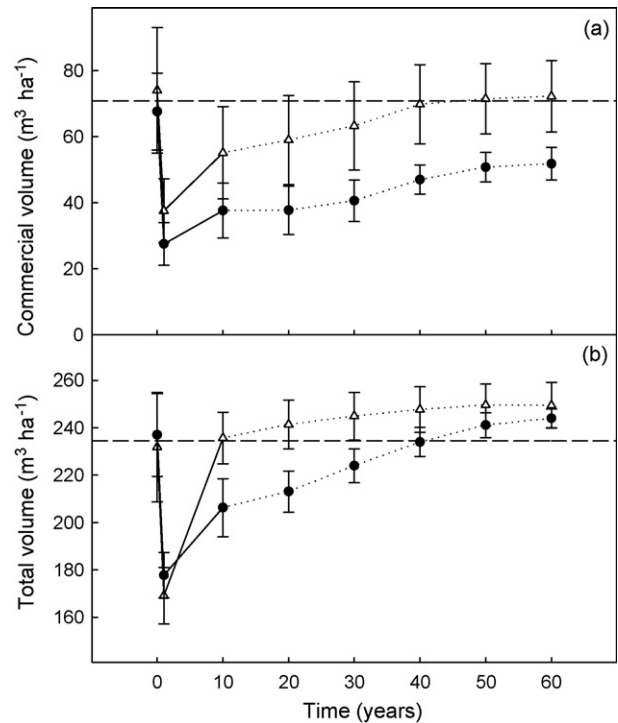


Fig. 3. Stand recovery of total (a) and commercial volume (b) after initial harvest in Year 0 under RIL (△) and CL (●), showing observed (years 0–10, solid lines) + predicted (years 10–60, dotted lines). The horizontal dashed line in each panel represents the initial volume. Error bars represent one standard error of the mean. Total standing volume is for trees  $\geq 10$  cm DBH, commercial volume is for trees  $\geq 45$  cm DBH from commercial species.

errors) while in CL this would take far more than the simulated 60 years despite similar logging intensities ( $37 \pm 7 \text{ m}^3 \text{ ha}^{-1}$  versus  $39 \pm 9 \text{ m}^3 \text{ ha}^{-1}$  for RIL and CL, respectively).

Although the recovery of commercial stock volume was projected to occur 30–40 years after logging in RIL, this does not take into consideration that part of the stock recovered may be composed of defective stems. These can represent up to 50% of the inventoried commercial stock in an unlogged forest (Holmes et al., 2002). Although rarely acknowledged, defect

rate is a key factor determining sustainable cutting cycles and logging intensities. For example, if we assume a defect rate of 0%, projected commercial stock increment during the 60-year simulated period is equal to  $0.58 \pm 0.02$  and  $0.40 \pm 0.09 \text{ m}^3 \text{ ha}^{-1} \text{ year}^{-1}$  for RIL and CL, respectively, while the resulting (harvestable) commercial increment falls to half of that value if the defect rate is 50%. These increment estimates are within the range of values observed from long-term permanent plots and estimated by growth and yield models in

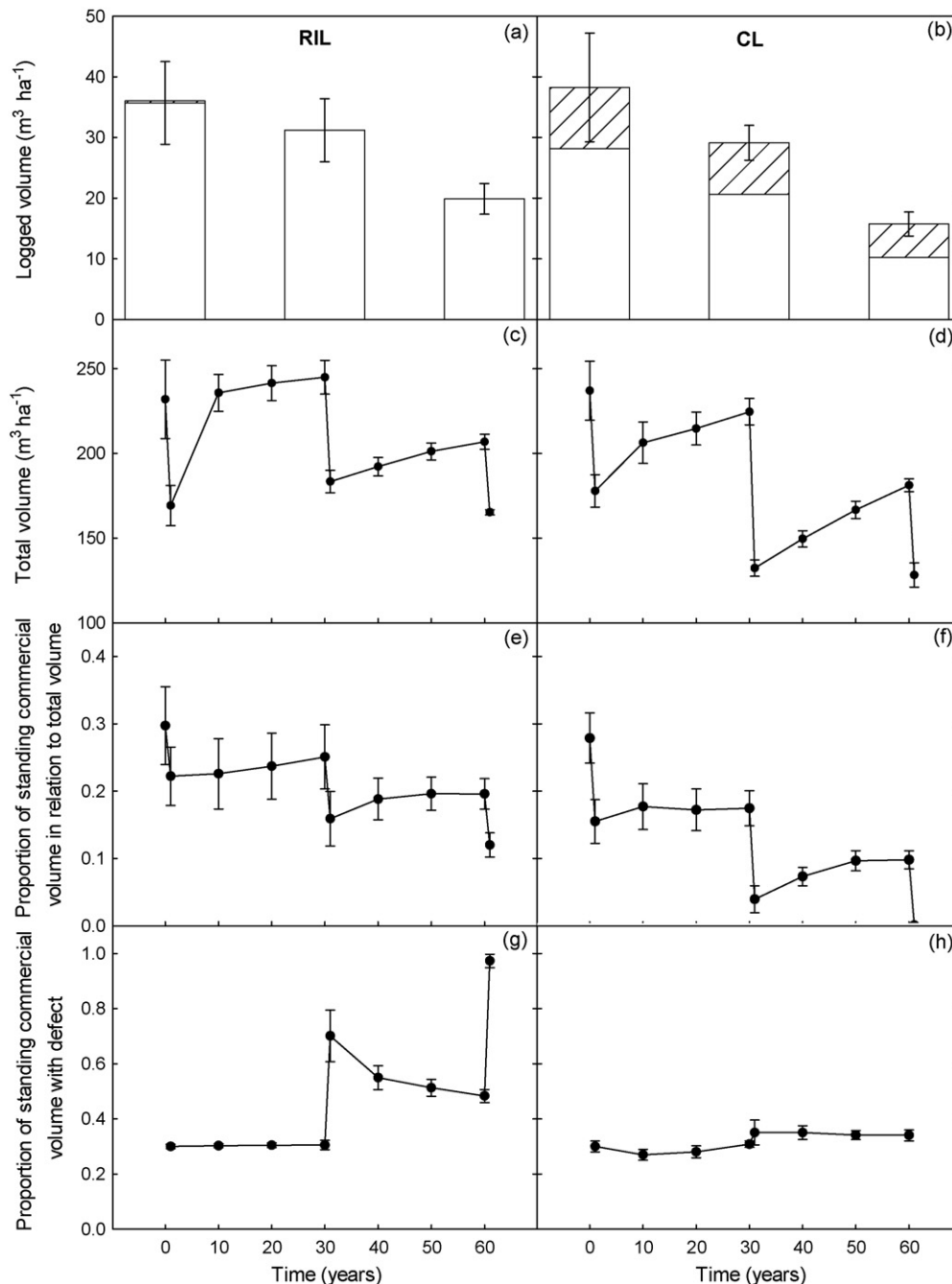


Fig. 4. Multiple harvests with RIL (left panels) and CL (right panels) treatments simulated through two 30-year cutting cycles. (a, b) Logged volume (predicted timber yield) including the volume of defective stems harvested under both treatments (hashed bar columns). (c, d) Total standing volume ( $\text{m}^3 \text{ ha}^{-1}$ ,  $\text{DBH} \geq 10 \text{ cm}$ ). (e, f) Proportion of standing commercial volume relative to total standing volume. (g, h) Proportion of trees from commercial species ( $\text{DBH} > 45 \text{ cm}$ ) that are predicted to have defective stems. Error bars represent one standard error of the mean. Percentage of logged volume represented by defective stems in the first harvest was 1 and 26% for RIL and CL, respectively (Barreto et al., 1998). Results are based on an assumption of 30% of trees with defect in year 10.

the Brazilian Amazon (Silva et al., 1995, 1996; Higuchi et al., 1997; Oliveira, 2000, 2005; Alder and Silva, 2001; Vidal, 2004; van Gardingen et al., 2006).

### 3.3.2. Multiple harvests under current forest legislation

Fig. 4 shows the predicted responses of forests managed under multiple harvests with a 30-year cutting cycle and a maximum logging intensity of  $40 \text{ m}^3 \text{ ha}^{-1}$ , as specified by current Brazilian regulations. RIL resulted in higher commercial yields when compared to CL (Fig. 4a and b) possibly due to the decrease in damage to potential future crop trees and to the fact that defective trees were not logged. However, under both regimes the total yield was predicted to decline with each harvest. Also, RIL appeared to result in a lower long-term impact. For instance, stand volume at the end of the simulation was projected to be higher for RIL than for CL (Fig. 4c and d). Another indicator of reduced impact was a lower projected percentage of stand volume of pioneer species (groups 7 and 9) after three harvests following RIL versus CL ( $35 \pm 2\%$  and  $41 \pm 0.3\%$ , respectively; data not shown), even though initial volumes of pioneer species were higher in the RIL forest (Table 1).

Observed and predicted differences in post-logging recovery between RIL and CL cannot be attributed to intrinsic floristic and or structural differences between areas (Table 1). In CL, higher skidtrail, road, and log landing densities coupled with no liana cutting and careless felling technique resulted in a greater number of damaged and killed trees (Johns et al., 1996; van der Hout, 1999; Armstrong and Inglis, 2000; Asner et al., 2002, 2004a,b; Holmes et al., 2002; Pereira et al., 2002). This greater logging impact in the forest under CL is the most probable reason for its slower stand recovery after logging.

It is clear from these results that the current yield management prescription, even applying RIL techniques, is not sustainable for Paragominas forests, similar to results reported for Tapajós (van Gardingen et al., 2006). In RIL, logged volume decreased from  $36 \pm 7 \text{ m}^3 \text{ ha}^{-1}$  in the first harvest (observed) to  $20 \pm 3 \text{ m}^3 \text{ ha}^{-1}$  in the third harvest (predicted, Fig. 4a) and the proportion of the commercial stock in the stand is progressively depleted during successive cutting cycles, initially representing  $30 \pm 6\%$  of total volume in an unlogged forest and dropping to  $25 \pm 5\%$  and  $20 \pm 2\%$  prior to second and third logging events, respectively (Fig. 4e). This results in a progressive mining of timber resources rather than sustainable forest management. Furthermore, failure to remove low-quality stems (with defect) under RIL means that by the end of the third harvest, the majority of the remaining commercial stems will be of low quality and unsuitable for harvest. Under the assumption that 30% of trees are defective, we project that  $97 \pm 2\%$  of the trees  $\geq 45 \text{ cm}$  DBH of commercial species will be defective after the third harvest in year 60 (Fig. 4g). This trend dwarfs RIL and CL differences in relation to effectively harvestable standing commercial stock after successive harvests. By the end of the simulation period, the stand under CL has an estimated standing commercial stock of  $0 \pm 0.1 \text{ m}^3 \text{ ha}^{-1}$  while the stand under RIL has  $20 \pm 3 \text{ m}^3 \text{ ha}^{-1}$ , from which only 3% of the trees are effectively harvestable (i.e., defect free). The accumulation of defective trees in commercial

size classes by the end of the simulation period is likely to occur even when the defect rate is much lower than that observed in forests in the study region. For instance, under the assumption that 15% of trees are defective and using the same simulation dataset, after the third harvest only half of the trees would be effectively harvestable. This reveals an important shortcoming of current RIL guidelines that must be addressed in order to avoid accumulation of defective trees within managed natural forests.

Finally, significant shifts in species composition were observed in the area subjected to RIL. For instance, even though the projected percentage of volume from trees belonging to pioneer species groups (groups 7 and 9) in RIL is lower than what has been projected for CL, it still represents a nearly two-fold increase in dominance by these species groups relative to the unlogged stand ( $19 \pm 4\%$  in year 0 and  $35 \pm 2\%$  in year 60). Also, our results reveal a decline in the dominance of emergent species (group 8); the percentage of volume in the sequentially logged stand is about half that of the unlogged stand ( $13 \pm 3\%$  in year 0 versus  $7 \pm 1\%$  in year 60). Similar shifts in species composition due to logging, where large slow-growing commercial trees are harvested and replaced by smaller generally non-commercial trees from pioneer species, have been found in other modelling studies (Keller et al., 2004; Phillips et al., 2004; van Gardingen et al., 2006). This may significantly impact the financial viability of future harvests since the pioneer groups are dominated by low-value timber species (sawn wood  $< \text{US\$ } 100 \text{ m}^{-3}$ ) such as *Inga alba*, while emergents in general include high-value timber species (sawn wood  $> \text{US\$ } 200 \text{ m}^{-3}$ ) such as *Tabebuia* spp. and *Dipteryx odorata* (based on a timber market survey conducted in the region in 1998, Lentini et al., 2003).

## 4. Conclusions

The growth and yield simulation model SIMFLORA was calibrated for forests of the Paragominas region in eastern Amazonia and its suitability evaluated with respect to biological realism and accuracy. Although there were some clear data constraints, dynamics of each species group were shown to be adequate for application over medium-term simulations (50 years in the current study).

Projections starting with data describing the stand ten years after logging indicate large differences between RIL and CL with respect to the recovery of commercial and total standing volume. Under RIL, total stand volume may recover within 10 years while 30–40 years are predicted to be necessary for commercial volume recovery. Under CL, the time necessary for volume recovery, both stand-level and commercial, is considerably longer (35–40 and  $>60$  years, respectively). Nevertheless, despite the many benefits of RIL, both systems were found to lead to mining of timber resources and were considered unsustainable when applying current Brazilian regulations of a 30-year cutting cycle and maximum yield of  $40 \text{ m}^3 \text{ ha}^{-1} \text{ year}^{-1}$ .

This study revealed two disturbing trends that must be addressed in order to maintain financial viability of RIL: an increase over time in the proportion of defective or poor quality

stems in the stand and changes in relative importance of species groups in the stand (decrease of high-value emergent species and increase of low-value pioneer species dominance). The results highlight the necessity of improving RIL guidelines concerning harvest intensities, cutting cycles and silvicultural treatments in order to ensure that RIL serves as a foundation for sustainable forest management in Amazonia. It also reinforces the statement that sustainable forest management will require combining guidelines for both RIL and timber yield regulation.

## Acknowledgements

We thank Christina Staudhammer and two anonymous reviewers for their comments. This project had logistic support from the Agrosete Farm owners, Mr. Pérsio Lima and Mr. Thales Lima. This project was supported initially by World Wildlife Fund (WWF Brazil) and currently it integrates *Consórcio Alfa* supported by USAID. Support also came from PPD-PPG7-MCT-FINEP and Projeto Dendrogene (EMBRAPA/DFID). This work would not have been possible without help in the field from many people. Among them, we especially thank Frank Pantoja, Rodney Salomão, Damião Farias, Manuel Farias, Miguel Lopes, Marcelo Galdino de Almeida, Eliana Farias, Manuel Vitorino, Miguel Alves, and Waldemir Ribeiro da Cruz.

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