

Growth-Oriented Logging (GOL): A new concept towards sustainable forest management in Central Amazonian várzea floodplains

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ARTICLE INFO

Article history:

Received 26 July 2007

Received in revised form 13 February 2008

Accepted 19 March 2008

Keywords:

Tropical floodplain forest

Tree-ring analysis

Growth model

Cutting cycle

Minimum logging diameter

Silviculture

ABSTRACT

Against a background of increasing human populations in developing countries, and global climate change, conservation of tropical forests remains one of the most important ecological challenges of our time. One of the biggest difficulties for ecologically sustainable management of tropical forests is obtaining reliable growth data for trees, which is a prerequisite for determining harvesting volumes and cutting cycles. GOL is the first concept for sustainable management of tropical timber resources in Amazonian floodplain forests (várzea) based on species-specific management criteria, such as minimum logging diameters (MLDs) and cutting cycles. From timber species with varying wood densities of different successional stages, volume stocks have been estimated in 1-ha plots and 12 growth models have been developed based on tree rings, which are annually formed as a consequence of the regular, long-term flooding. The MLDs of timber species vary between 47 and 70 cm and the estimated cutting cycles differ the 10-fold, from 3 to 32 years. These enormous differences in the growth rates between tropical timber species are not considered in current management practices, which apply only one diameter cutting limit and one cutting cycle to harvest many tree species. This practice risks the overexploitation of slow-growing timber species, while the fast-growing timber species with low wood densities cannot be efficiently used. Based on the timber stocks and lifetime growth rates, the GOL concept has been created as an aid to improve forest management in the Central Amazonian várzea. The model is unique for tropical silviculture.

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1. Introduction

High productive floodplains (várzea) occur over an area of 200,000 km² along the nutrient-rich and sediment-loaded white-water rivers (Prance, 1979; Junk, 1993). The várzea has been used and settled by a high human population density for centuries conducting agriculture, pasture, fishing, hunting, and the extraction of timber and non-timber products (Junk et al., 2000). Therefore floodplain forests are one of the most threatened ecosystems in the Amazon, endangered by their conversion to agricultural areas and pastures, as well as due to their exploitation by an expanding timber and plywood industry (Higuchi et al., 1994; Uhl et al., 1998). But these unique forest ecosystems perform multiple ecological services, such as slowing down water energy and are therefore indispensable for protection against erosion and consequently for the hydrological cycle of the river system. In addition, the forests deliver timber and non-timber products for

the local population (Kvist et al., 2001) and represent the main food source for many fish species, which in turn are the primary protein base for the local human populations. The forests are habitats for highly adapted plant and animal species and they represent important sinks and sources for biogeochemical cycles (Junk, 1997; Junk et al., 2000). In this context, the sustainable management of floodplain forests is a promising way to protect the multiple functions and services of these forest ecosystems and to guarantee the use of natural resources for a growing human population in Amazonia.

Because of low energy costs for logging, skidding, and transport during the flooded period (Barros and Uhl, 1995; Albernaz and Ayres, 1999) between 60% and 90% of the local and regional markets in the Western Amazon basin of Brazil and Peru are still provided with timber from the várzea (Worbes et al., 2001). In 2000, in the Amazon state of Brazil, about 75% of the timber for the plywood and veneer industry originated from floodplain forests (Lima et al., 2005). Timber species in the várzea are classified using a threshold of about 0.60 g cm⁻³, into low-density and high-density trees (Schöngart, 2003). The latter group is mainly used for house and boat construction, as well as furniture, while the

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low-density species are mainly used for plywood and veneers (Albernaz and Ayres, 1999; Kvist et al., 2001; Worbes et al., 2001). Many tree species in the várzea forests reach high abundances, because species richness in the várzea floodplain forests is much lower than in adjacent upland forests (terra firme) (Wittmann et al., 2006). But due to unsustainable logging practices, lack of information about growth rates and regeneration, many tree species have already disappeared from local and regional markets within a few decades (Schöngart et al., 2007).

The várzea is a landscape patchwork of water bodies, aquatic and terrestrial macrophytes and forests. Almost 1000 tree species compose different várzea forest types covering about 50–75% of the floodplains (Wittmann et al., 2002, 2006). Erosion and sedimentation processes continuously rearrange the floodplains, creating small-scale landscapes corresponding to different successional stages (Salo et al., 1986; Kalliola et al., 1991). Primary succession initiates at sites with high sedimentation rates next to the river. Due to geomorphological and environmental modifications, the young pioneer stands with ages up to 20 years develop to 40 year-old early secondary stages, 50–80 year-old late secondary stages and climax stages at higher elevated sites with relatively long-standing stable environmental conditions and ages up to 300–400 years (Worbes et al., 1992; Schöngart et al., 2003; Wittmann et al., 2004). Along successional gradient, wood densities of tree species increase, while diameter increment rates decrease reflecting replacement of pioneer species by more long-lived pioneer species and climax species (Swaine and Whitmore, 1988; Worbes et al., 1992; Schöngart, 2003). With increasing stand age, tree density declines and species richness increases leading to changes in the canopy architecture characterized by increasing stand height, crown size, and crown area (Terborgh and Petren, 1991; Wittmann et al., 2002; Schöngart et al., 2003).

Trees in the várzea respond to the predictable several months long annual flooding, defined as flood-pulse (Junk et al., 1989) by phenological, morphological, and physiological changes (Parolin et al., 2004). Flooding causes annually recurring anaerobic conditions for the roots, hindering the water-uptake leading to physiological stress and leaf shedding. This ultimately results in a cambial dormancy (Schöngart et al., 2002) and the formation of an annual tree-ring boundary in the wood during the flooded period (Worbes, 1989). The existence of annual tree rings allows the application of dendrochronological methods to determine tree age and growth rates as a basis for growth modelling (Schöngart et al., 2007).

Most studies dealing with sustainable forest management in the várzea so far have based their approach to sustainability on species composition and structure, as for example in the Amazon Estuary (Gama et al., 2005; Fortini et al., 2006). Knowledge on tree ages and increment rates, as well as regeneration processes, in the várzea floodplain is insufficient. In this study, the population structures of timber species in four different successional stages of the Central Amazonian várzea were analysed. Growth parameter models for timber species in diameter, tree height, and volume based on tree-ring analysis were constructed to define species-specific management criteria for harvesting rotation and optimal minimum logging diameter. Based on the results of structure analysis and growth modelling, a specific management concept (GOL) for várzea forests was developed to improve the management practices in the Central Amazonian region.

2. Material and methods

2.1. Study area

The study was located in the Mamirauá Sustainable Development Reserve (MSDR) in the Amazonas state located at the

confluence of the Solimões and the Japurá Rivers, approximately 70 km northwest of the city of Tefé. The MSDR comprises 11,240 km² of várzea floodplains. Since 1992, a variety of community-based management systems have been implemented in the MSDR based on socio-economic and biological-ecological studies, including fisheries, agriculture, agro-forestry, eco-tourism, and forestry (Ayres et al., 1998). The community-based forest management program was established in 2000. Since then, several cooperatives have been founded within the MSDR to conduct controlled timber extractions. This requires a management plan (Forest Code No. 4.711, 15, September 1965) based on an inventory of all timber species >20 cm diameter in the areas selected for harvest. The plan has to be authorized by environmental agencies (Schöngart et al., 2007). The maximum area for total wood extraction is 500 ha per cooperative applying a polycyclic system (Selection Systems), adapted from the CELOS-system in Suriname (de Graaf et al., 2003). This system aims to keep a multi-aged stand through timber cutting at intervals (cutting cycle) by establishing a diameter cutting limit (DCL). The cutting cycle defines the return interval in years between timber harvests in the same area. Due to the harvest of only a few selected trees above the defined DCL, the uneven-aged structure of the forest is maintained by the establishment of seedlings in small gaps and in the understorey. To achieve a more or less constant annual harvest, the total area for the forest management is divided in several blocks with similar size corresponding to the number of years of the cutting cycle. So far the forest management in the MSDR applies a diameter cutting limit of 45 cm, a cutting cycle of 25 years and a maximum yield of 5 trees ha⁻¹ including floats (trunks of low-density trees), which are necessary to skid logs that would sink out of the forest during the flooded period (Schöngart et al., 2007). Ten percent of the trees of a timber species above the DCL must remain in the forest to guarantee the species' regeneration ("seed trees"). These management restrictions are based on laws and normative instructions (IN) established by the Federal Environmental Agency (IBAMA) (Schöngart, 2003). Timber prices per m³ in 2007 varied between R\$ 30.00 and 45.00 (US\$ 15.00–17.50) for low-density logs and R\$ 62.00 (US\$ 31.00) for high-density logs.

The recently established IN no. 5, from 11th December 2006, enables modified management options depending on property, forest type, and area (smallholders, community-based, industrial). Full management plans comprise a minimum cutting cycle of 25–35 years with maximum yields of up to 30 m³ ha⁻¹ year⁻¹ or, alternatively, management plans with low yield intensities (<10 m³ ha⁻¹ year⁻¹) applying a cutting cycle of 10 years (in várzea yields can exceed 10 m³ ha⁻¹ year⁻¹, but must be restricted to three harvested trees ha⁻¹). The IN requires the establishment of species-specific DCLs based on ecological and technical criteria such as tree density distribution of trees ≥ 10 cm DBH, but if this information is not available for a timber species, a common DCL of 50 cm is applied.

2.2. Forest inventories

Forest inventories were performed in 1-ha plots established in four successional stages in the Jarauá sector of the MSDR. The areas are located almost at the same elevation and are annually flooded between 120–150 days, by to average depth of 336–465 cm (Schöngart, 2003). Site age ranged from recently formed sedimentation areas at the margin of the Japurá river (20 year-old early secondary stage and 50 year-old late secondary stage) to sites far away from the river channel (125 year-old intermediate stage and 240 year-old climax stage) (Schöngart, 2003; Wittmann et al., 2006) belonging to the várzea of the Late Holocene not older than 5000 years (Ayres, 1993). Despite the small-scale geomorphological variation of the floodplains, várzea forests are dominated by a high

proportion of generalists, widely distributed tree species (Wittmann et al., 2006) and the forests exhibit high floristic similarities over large geographic distances due to long-distance dispersal by currents and fishes. Thus the studied stands are representative for large areas of várzea floodplains in Central Amazonia.

The plots were divided into 16 quadratics of 25 m × 25 m (625 m²), where diameter at breast height (DBH) of trees ≥10 cm was recorded with a diameter tape (at trees with buttresses and other anomalies the diameter was measured above them). Tree height was measured with an inclinometer (Blume-Leiss). In a 62.5 m² circular subplot established at the centre of each quadrat tree regeneration (saplings >1 m height and <10 cm diameter) was inventoried (10% of the total area). Tree species that are already commercialised and those with a potential use (Martini et al., 1998; Kvist et al., 2001) have been defined as timber species and classified as low-density and high-density timber species by their specific wood density (Schöngart, 2003). The population structure of timber species was analysed by the distribution of abundance using 10-cm diameter classes and considering a common DCL of 50 cm. Total volume for every tree was calculated by the basal area multiplied with the corresponding tree height and a common form factor of 0.6 determined by Cannell (1984) for a number of broadleaf tropical tree species:

$$V_t = \pi \left(\frac{DBH_t}{2} \right)^2 H_t f \quad (1)$$

Where V_t is the volume at age t ; DBH_t is the DBH at age t ; H_t is the tree height at age t , and f is the form factor (the ratio of tree volume to the volume of a cylinder with the same basal diameter and height).

2.3. Growth modelling

From 12 low-density and high-density timber species, 137 stem disks and 133 cores were been collected from emergent individuals for tree-ring analyses. Wood samples were obtained from *Ficus insipida* Willd. (Moraceae) in the 20 year-old early secondary stage and from *Pseudobombax munguba* (Mart. & Zucc.) Dugand, *Luehea cymulosa* Spruce ex Benth. (both Malvaceae), *Ilex inundata* Poepp. ex Reissek (Aquifoliaceae), *Macaranga acaciifolia* (Benth.) Benth., and *Albizia subdimidiata* (Splitg.) Barneby & J.W. Grimes (both Fabaceae) in the 50 year-old late secondary stage. In the 125 year-old intermediate stage and in the 240 year-old climax stage *Tabebuia barbata* (E. Mey.) Sandwith (Bignoniaceae), *Eschweilera albiflora* (DC.) Miers (Lecythidaceae), *Piranhea trifoliata* Baill. (Euphorbiaceae), *Pouteria elegans* (A. DC.) Baehni, *Chrysophyllum argenteum* Jacq. (both Sapotaceae), and *Sloanea terniflora* (Sessé & Moc. ex DC.) Standl. (Elaeocarpaceae) were sampled. Dendrochronology (time series analysis of tree rings) provided data on tree ages by ring counting and measuring of diameter increments (ring width) on cross sections and cores. The key for a successful dendrochronological study is the analysis of tree-ring structure by wood anatomical features (Fig. 1). Therefore, stem disks and cores

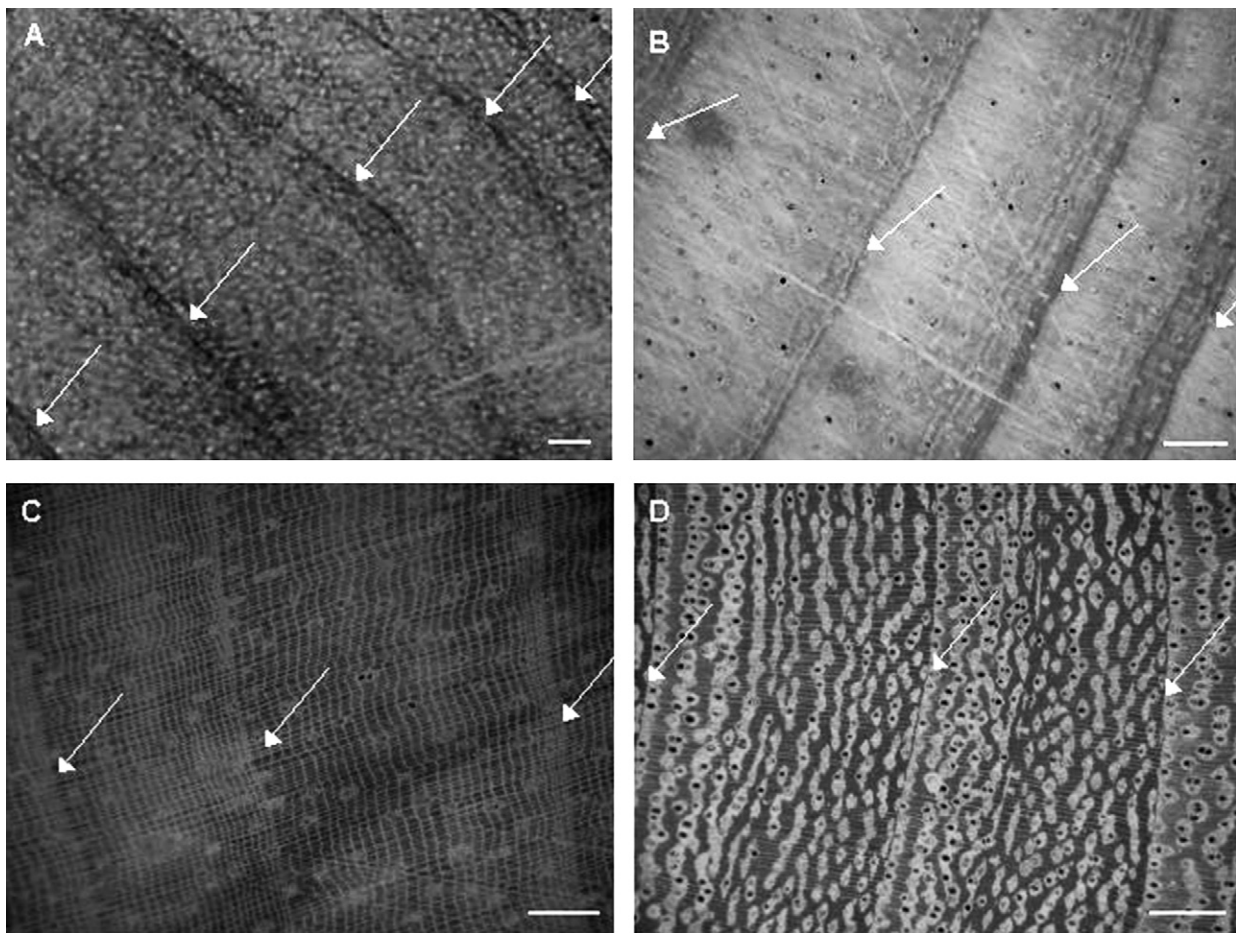


Fig. 1. Macroscopic wood anatomy of várzea tree species: (A) Tree rings characterized by intra-annual wood density variation (*Pseudobombax munguba* (Mart. & Zucc.) Dugand, Malvaceae). (B) Tree rings limited by marginal parenchyma bands (*Macaranga acaciifolia* (Benth.) Benth., Fabaceae). (C) Tree rings characterized by alternating fibre and parenchyma bands (*Eschweilera ovalifolia* (DC.) Nied., Lecythidaceae). (D) Variation in vessel size and distribution (ring porous) (*Tabebuia barbata* (E. Mey.) Sandwith, Bignoniaceae). Arrows indicate the ring boundaries; horizontal bars indicate 1 mm length.

were glued on a wooden support and polished consecutively with sandpaper up to a grain of 600 (Schöngart et al., 2004). The ring width was measured perpendicularly to the rings to the nearest 0.01 mm using a digital measuring device (LINTAB) supported by software for time series analyses and presentation (TSAP-Win) (Schöngart et al., 2005). This program allows the measurement of ring-width and data treatments such as the indexation, cross-dating and correlation analyses of the ring-width curves.

Diameter growth of a tree species was modelled in two ways. First, from different individuals of a tree species, the increment rates measured on the wood samples, from pith to bark, were accumulated to form individual growth curves related to the measured DBH (Brienen and Zuidema, 2006, 2007). The mean cumulative diameter growth curve described the relationship between tree age and diameter of a species (Stahle et al., 1999; Worbes et al., 2003; Schöngart et al., 2007). Second, tree ages determined by ring counting were related to the corresponding DBH from several individuals of a species. Both age–diameter relationships were fitted to a sigmoidal function using diameter as the independent variable (Schöngart et al., 2007):

$$DBH = \frac{a}{(1 + (b/\text{age})^c)} \quad (2)$$

Height growth of a tree species was estimated by combining the age–diameter relationship and the relationship between DBH and

tree height measured in the field fitted to a non-linear regression model (Nebel, 2001; Nebel et al., 2001a; Schöngart et al., 2007):

$$H = \frac{DBH \times d}{(DBH + e)} \quad (3)$$

Thus, for every tree age over the lifespan of a species, the corresponding DBH and tree height can be derived. The cumulative volume growth of a tree species was calculated by Eq. (1) for every tree age. From the volume growth (cumulative volume over the life span of a tree) the current annual volume increment rate CAI_V and mean annual volume increment rate MAI_V were derived for every tree age (Fig. 2) by the following equations (Schöngart et al., 2007):

$$CAI_V = CGW_{V(t+1)} - CGW_{V(t)} \quad (4)$$

$$MAI_V = \frac{CGW_{V(t)}}{t} \quad (5)$$

Where CGW_V is the cumulative volume in different years t over the entire life span. Growth modelling was performed with the software X-Act (SciLab).

From the volume growth model, criteria for a species-specific management were derived (Fig. 2). To achieve an optimal volume production a tree should be harvested between the optimum of the CAI_V ($CAI_{V(max)}$) and the optimum of the MAI_V ($MAI_{V(max)}$). This period was defined as preferred period for harvest (Schöngart, 2003). Harvests before the $CAI_{V(max)}$ and after $MAI_{V(max)}$ would lead to an inefficient use of the growth potential of a tree species, because trees still did not reached or already passed over their optimum growth phase. The minimum logging diameter (MLD) is defined as diameter at $CAI_{V(max)}$ and can be derived by the age–diameter relationship (Nebel, 2001; Schöngart et al., 2007). To indicate growth variation among individuals of a species tree age at the size of the MLD was calculated considering the standard deviation of the specific age–diameter relationship. To estimate the cutting cycle the mean time through 10-cm diameter classes until achieving the specific MLD was calculated (Schöngart et al., 2007) considering the standard deviation of the diameter growth curve. This period represented the average time, which an individual needs to grow from one to the next diameter class (Fig. 2).

$$\text{Cutting cycle} = \frac{\text{Age}_{(MLD)}}{(MLD \times 0.1)} \quad (6)$$

3. Results

3.1. Structure analysis of timber species

Young successional stages (early and late secondary stage) were characterized primarily by low-density tree species (Tables 1 and 2) such as *P. munguba*, *L. cymulosa*, *I. inundata*, *F. insipida*, *M. acaciifolium*, and *A. subdimidiata*. The presence of high-density trees was negligible. In the 20 year-old early secondary stage, 527 trees (63% of the total abundance) with a volume of 185.7 m³ ha⁻¹ (68% of total volume) were found belonging to the low-density tree species group (Table 1), but only seven individuals of *F. insipida* passed over the DCL of 50 cm comprising 25.9 m³ ha⁻¹. In the 50 year-old late secondary stage, 281 trees (58% of total abundance) were recorded belonging to low-density tree species. The most abundant species were *I. inundata*, *P. munguba* and *L. cymulosa*, with a total volume of 479.7 m³ ha⁻¹ (81.3% of the stand volume) (Table 2). More than 67 trees exceeded the DCL of 50 cm, with a total volume of 249.5 m³ ha⁻¹. In the late succession (intermediate and climax stage), high-density tree species dominated, mostly *T.*

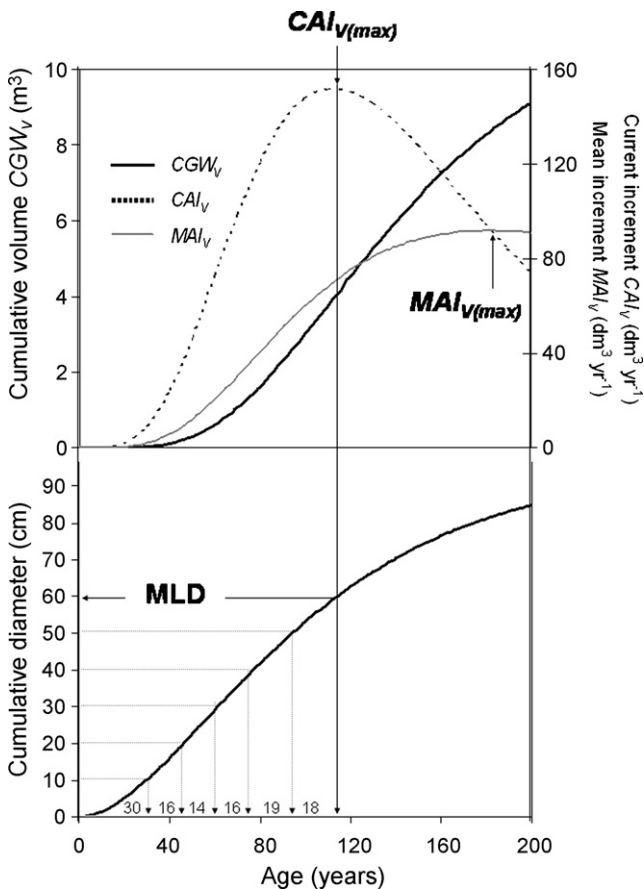


Fig. 2. Volume growth model (CGW_V) of a tree species and estimates of current (CAI_V) and mean (MAI_V) annual volume increment rates. The period between $CAI_{V(max)}$ and $MAI_{V(max)}$ is defined as preferred period for harvest. The diameter at the age of the $CAI_{V(max)}$ is defined as minimum logging diameter (MLD), which is derived from the species-specific age–diameter relationship (below). The cutting cycle is estimated by the mean passage time through 10-cm diameter classes (indicated by the arrows on the X-axis, numbers between the arrows indicate the mean passage time in years) until reaching the defined MLD.

Table 1
Volume stocks ($\text{m}^3 \text{ha}^{-1}$) of low-density and high-density commercial tree species in a 20-year old early secondary stage, shown for 10-cm diameter classes and a diameter cutting limit (DCL) of 50 cm

Early secondary stage	DBH-classes (cm)									DCL (cm)		Sum
	10–20	20–30	30–40	40–50	50–60	60–70	70–80	80–90	≥90	<50	≥50	
Low-density trees												
<i>Ficus insipida</i>	1.4	4.4	21.9	34.5	12.9	7.8	5.2			62.2	25.9	88.1
<i>Luehea cymulosa</i>	25.8	11.1	2.6	1.2						40.7		40.7
<i>Pseudobombax munguba</i>	11.7	29.6	10.2							51.5		51.5
<i>Ilex inundata</i>	2.9	2.1								5.0		5.0
<i>Macrolobium acacifolium</i>	0.1	0.2								0.3		0.3
<i>Albizia subdimidiata</i>	0.1									0.1		0.1
Subtotal	42.0	47.4	34.7	35.7	12.9	7.8	5.2			159.8	25.9	185.7
High-density trees												
<i>Calycohyllum spruceanum</i>	0.1	1.6								1.7		1.7
Subtotal	0.1	1.6								1.7		1.7
Tree species												
Volume of all tree species	67.1	89.7	53.8	35.7	12.9	7.8	5.2			246.3	25.9	272.2
Volume of timber species	42.1	49.0	34.7	35.7	12.9	7.8	5.2			161.5	25.9	187.4
Percentage of timber species	62.7	54.6	64.5	100.0	100.0	100.0	100.0			65.6	100.0	68.8

Table 2
Volume stocks ($\text{m}^3 \text{ha}^{-1}$) of low-density and high-density commercial tree species in a 50-year old late secondary stage, shown for 10-cm diameter classes and a diameter cutting limit (DCL) of 50 cm

Late secondary stage	DBH-classes (cm)									DCL (cm)		Sum
	10–20	20–30	30–40	40–50	50–60	60–70	70–80	80–90	≥90	<50	≥50	
Low-density trees												
<i>Pseudobombax munguba</i>	0.2	4.6	19.1	65.0	30.8	63.4	12.4	8.0	18.3	88.9	132.9	221.8
<i>Ilex inundata</i>	5.4	14.6	26.1	50.4	46.3	27.4	6.4			96.5	80.1	176.6
<i>Luehea cymulosa</i>	0.9	11.7	9.7	15.2	21.5	8.3				37.5	29.8	67.3
<i>Albizia subdimidiata</i>	0.2	0.1	4.2	1.8						6.3	0.0	6.3
<i>Eschweilera ovalifolia</i>	0.1						3.4			0.1	3.4	3.5
<i>Macrolobium acacifolium</i>	0.2	0.7				3.3				0.9	3.3	4.2
Subtotal	7.0	31.7	59.1	132.4	98.6	102.4	22.2	8.0	18.3	230.2	249.5	479.7
High-density trees												
<i>Eschweilera albiflora</i>	0.1									0.1		0.1
Subtotal	0.1									0.1		0.1
Tree species												
Volume of all tree species	24.4	63.3	92.0	150.0	109.3	102.4	22.2	8.0	18.3	329.7	260.2	589.9
Volume of timber species	7.1	31.7	59.1	132.4	98.6	102.4	22.2	8.0	18.3	230.3	249.5	479.8
Percentage of timber species	29.1	50.1	64.2	88.3	90.2	100.0	100.0	100.0	100.0	69.9	95.9	81.3

barbata, *E. albiflora*, *P. trifoliata*, *Eschweilera parvifolia* Mart. ex DC. (Lecythidaceae), *Vatairea guianensis* Aubl. (Fabaceae), *P. elegans*, *C. argenteum* Jacq. (both Sapotaceae), *Macrolobium pendulum* Willd. ex Vogel, *Lecointea amazonica* Ducke (both Fabaceae), and *Calophyllum brasiliense* Cambess. (Clusiaceae) (Tables 3 and 4). Low-density tree species were represented by fewer species and lower volume stocks: *Hevea spruceana* (Benth.) Müll. Arg. (Euphorbiaceae), *S. terniflora*, *Inga* cf. *lateriflora* Miq. (Fabaceae), *Inga pilosula* (Rich.) J.F. Macbr. and *Paramachaerium ormosioides* (Ducke) Ducke (both Fabaceae), *P. munguba*, and *L. cymulosa* (Table 3). All latter tree species had 133–145 individuals ha^{-1} (26–31%), and 158.2–167.4 $\text{m}^3 \text{ha}^{-1}$, but only 20–21 trees, with a total stem volume of 86.5–110.9 $\text{m}^3 \text{ha}^{-1}$, exceeded the DCL of 50 cm. Highest commercial volumes had high-density species such as *T. barbata*, *P. trifoliata*, *E. parvifolia*, *C. argenteum*, *E. albiflora*, and *V. guianensis*.

3.2. Growth models of commercial tree species

Growth models were based on significant relationships between tree age and diameter, and between DBH and tree height

(Table 5). Low-density trees required periods of 15 years (*F. insipida*) to 67 years (*S. terniflora*) to exceed the DCL of 50 cm; high-density species required between 106 years (*P. elegans*) and 151 years (*E. albiflora*) to reach this level (Fig. 3). Because of the higher increment rates of DBH and height, low-density species had a faster volume growth than did high-density species. For selective logging, it is important to know at which age and diameter a tree species reaches its optimal growth. The age at $\text{CAI}_{V(\text{max})}$ and $\text{MAI}_{V(\text{max})}$ of the tree species (Fig. 2) was correlated with their wood density and the relationship can be described by polynomial functions explaining 93–94% of the variance ($p < 0.001$) (Fig. 4). From this model the preferred period for harvest can be also estimated for other várzea tree species, which have not been considered in this study, as a function of wood density. From Fig. 4 MLDs were derived by $\text{CAI}_{V(\text{max})}$ (Table 6). The tree age of a species at the size of MLD was determined by its age–DBH relationships considering the mean growth curve and standard deviation. For most species, the minimum logging diameter defined by the growth models exceeded 50 cm, varying between 47 and 70 cm. Cutting cycles derived from the growth models were 3.3–13.9 years for low-density tree species, while the models predicted

Table 3

Volume stocks ($\text{m}^3 \text{ha}^{-1}$) of low-density and high-density commercial tree species in a 120-year old intermediate stage, shown for 10-cm diameter classes and a diameter cutting limit (DCL) of 50 cm

Intermediate stage	DBH-classes (cm)									DCL (cm)		Sum
	10–20	20–30	30–40	40–50	50–60	60–70	70–80	80–90	≥90	<50	≥50	
Low-density trees												
<i>Hevea spruceana</i>	0.6	0.6	7.0	4.0	6.2					12.2	6.2	18.4
<i>Sloanea terniflora</i>	1.1	1.0	4.7		6.1	7.5				6.8	13.6	20.4
<i>Inga cf. lateriflora</i>	0.4	2.0	2.0		5.5					4.4	5.5	9.9
<i>Pseudobombax munguba</i>	0.3	1.3	2.3	3.4						7.3		7.3
<i>Luehea cymulosa</i>	0.3									0.3		0.3
Subtotal	2.7	4.9	16.0	7.4	17.8	7.5				31.0	25.3	56.3
High-density trees												
<i>Tabebuia barbata</i>	0.2	0.7		5.9	4.8	10.1	5.9			6.8	20.8	27.6
<i>Piranhea trifoliata</i>	0.7	3.7	1.0	5.9		10.4	6.7			11.3	17.1	28.4
<i>Eschweilera parvifolia</i>	0.3	1.9	2.9	3.8		5.9		8.3		8.9	14.2	23.1
<i>Vatairea guianensis</i>	0.6	0.4	3.7		6.2					4.7	6.2	10.9
<i>Pouteria elegans</i>	1.9	2.2			2.9					4.1	2.9	7.0
<i>Chrysophyllum argenteum</i>	0.4	0.9		3.4						4.7	0.0	4.7
<i>Eschweilera albiflora</i>	0.2									0.2	0.0	0.2
Subtotal	4.3	9.8	7.6	19.0	13.9	26.4	12.6	8.3		40.7	61.2	101.9
Tree species												
Volume of all tree species	39.0	53.3	59.2	43.1	48.5	37.6	12.6	8.3		194.6	107.0	301.6
Volume of timber species	7.0	14.7	23.6	26.4	31.7	33.9	12.6	8.3		71.7	86.5	158.2
Percentage of timber species	17.9	27.6	39.9	61.3	65.4	90.2	100.0	100.0		36.8	80.8	52.5

Table 4

Volume stocks ($\text{m}^3 \text{ha}^{-1}$) of low-density and high-density commercial tree species in a 240-year old climax stage, shown for 10-cm diameter classes and a diameter cutting limit (DCL) of 50 cm

Climax stage	DBH-classes (cm)									DCL (cm)		Sum
	10–20	20–30	30–40	40–50	50–60	60–70	70–80	80–90	≥90	<50	≥50	
Low-density trees												
<i>Sloanea terniflora</i>		0.1	0.8					17.8		0.9	17.8	18.7
<i>Inga pilosula</i>		0.8		2.0		5.6	5.9			2.8	11.5	14.3
<i>Luehea cymulosa</i>		0.8						10.4		0.8	10.4	11.2
<i>Paramachaerium ormosioides</i>	0.8	0.6	3.9	1.9						7.2		7.2
<i>Hevea spruceana</i>	1.5	1.5	1.5	1.6						6.1		6.1
<i>Pseudobombax munguba</i>			0.5							0.5		0.5
Subtotal	2.3	3.8	6.7	5.5		5.6	5.9	28.2		18.3	39.7	58.0
High-density trees												
<i>Chrysophyllum argenteum</i>	0.4	1.9	1.1	7.2				8.7	11.6	10.6	20.3	30.9
<i>Eschweilera albiflora</i>	2.0	4.4	5.2		8.7					11.6	8.7	20.3
<i>Piranhea trifoliata</i>	0.1					1.7		4.5	11.0	0.1	17.2	17.3
<i>Eschweilera parvifolia</i>	0.6	2.5		2.2	4.2	4.4				5.3	8.6	13.9
<i>Pouteria elegans</i>			1.5	3.4	3.2					4.9	3.2	8.1
<i>Tabebuia barbata</i>	0.3	0.5	1.0							1.8		1.8
<i>Vatairea guianensis</i>	0.6	1.0		1.7	7.3					3.3	7.3	10.6
<i>Macrobium pendulum</i>	0.1						5.9			0.1	5.9	6.0
<i>Lecointea amazonica</i>		0.4								0.4		0.4
<i>Calophyllum brasiliense</i>	0.1									0.1		0.1
Subtotal	4.2	10.7	8.8	14.5	23.4	6.1	5.9	13.2	22.6	38.2	71.2	109.4
Tree species												
Volume of all tree species	32.3	47.2	41.3	54.1	35.1	27.7	17.8	41.4	22.6	174.9	144.6	319.5
Volume of timber species	6.5	14.5	15.5	20.0	23.4	11.7	11.8	41.4	22.6	56.5	110.9	167.4
Percentage of timber species	20.1	30.7	37.5	37.0	66.7	42.2	66.3	100.0	100.0	32.3	76.7	52.4

cutting cycles of 21.5–32.1 years for high-density timbers. The cutting cycle of timber species in the várzea correlated significantly with specific wood density ($p < 0.01$) (Fig. 5).

4. A model for Growth-Oriented Logging (GOL)

Tropical forest management in general requires data for species-specific and site-specific growth rates, tree ages, and

regeneration processes to enable sustainable forest management. The differences in the diameter and volume increment rates and consequently cutting cycles among commercial tree species and forest ecosystems show clearly that polycyclic systems, operating with only one diameter cutting limit and cutting cycle to harvest various timber species, cannot be considered sustainable. To achieve a higher level of sustainability, two management concepts were developed based on the timber stocks and growth rates of the

Table 5
Number of ring-width measurements and tree height measurements used to model age–diameter and diameter–height relationships of 12 timber species from the várzea for growth modelling

Growth modelling	Number of samples		Number of measurements		Age–diameter relationship			Diameter–height relationship			Age–DBH	
	Ring-width	Tree height	a	b	c	d	e	d	e	Age-DBH	DBH–height	
<i>Albizia subdimitiata</i>	404	63	68.2018 ± 9.2628	34.3210 ± 4.1722	2.4826 ± 0.4428	33.9638 ± 0.6674	15.7564 ± 0.8382	0.83			0.79	
<i>Ficus insipida</i>	277	54	119.5742 ± 13.1913	18.4937 ± 3.3550	1.2997 ± 0.0788	28.6667 ± 0.7424	16.1743 ± 1.3022	0.62			0.55	
<i>Macrobolium acaciifolium</i>	521	18	137.5987 ± 9.2697	72.4934 ± 4.9861	1.8257 ± 0.0511	32.0676 ± 1.8523	26.8365 ± 3.4760	0.69			0.72	
<i>Luehea cymulosa</i>	1,349	338	86.5839 ± 7.9985	37.2305 ± 4.4048	1.5818 ± 0.0827	46.0880 ± 0.7894	38.7428 ± 1.0514	0.67			0.83	
<i>Pseudobombax munguba</i>	683	399	101.8435 ± 2.7307	44.2009 ± 2.0969	1.0507 ± 0.0127	44.2589 ± 0.6661	42.6987 ± 1.1402	0.71			0.92	
<i>Ilex inudata</i>	1,657	230	107.2910 ± 17.2131	55.8769 ± 6.7550	2.3162 ± 0.1282	39.760 ± 0.5244	24.8028 ± 0.7075	0.76			0.90	
<i>Sloanea terniflora</i>	719	49	135.7112 ± 12.1825	93.1059 ± 8.8556	1.9746 ± 0.1156	39.3949 ± 0.9865	28.0887 ± 1.5729	0.83			0.92	
<i>Chrysophyllum argenteum</i>	1,697	60	129.0611 ± 17.0861	154.7658 ± 20.3385	1.7954 ± 0.0876	41.7222 ± 0.9561	29.0724 ± 1.4189	0.94			0.83	
<i>Eschweilera albiflora</i>	2,078	49	156.3565 ± 39.7620	214.0361 ± 51.8604	1.7155 ± 0.1144	40.5848 ± 1.7031	32.4352 ± 2.4191	0.79			0.87	
<i>Tabebuia barbata</i>	1,187	24	125.8597 ± 10.7838	199.1537 ± 19.7898	1.4775 ± 0.0401	32.6267 ± 0.9274	15.6984 ± 1.3845	0.62			0.96	
<i>Pouteria elegans</i>	1,101	43	121.1704 ± 18.3318	128.7794 ± 20.4456	1.8654 ± 0.1631	30.8071 ± 0.8567	16.0890 ± 1.1890	0.90			0.88	
<i>Piranhea trifoliata</i>	4,757	60	129.3670 ± 7.9906	197.8906 ± 16.7651	1.3795 ± 0.0399	40.9838 ± 0.8856	26.0373 ± 1.2283	0.77			0.76	
Total		1,387										

Parameter estimates for age–diameter relationships ($DBH = a((1 + (b/age)^c)^d)$) and diameter–height relationships ($H = DBH \times d/(DBH + e)$) with standard deviations.

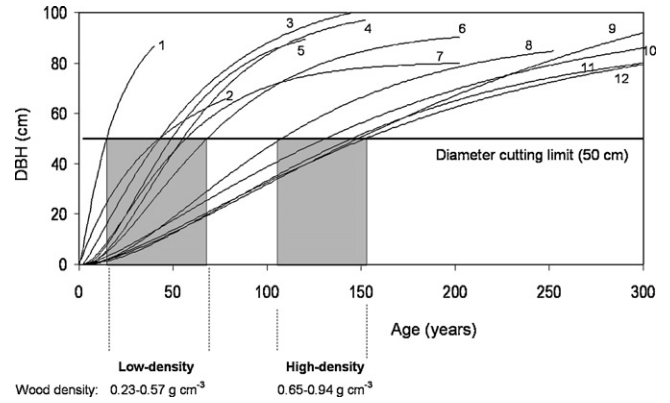


Fig. 3. Mean cumulative diameter growth curves of 12 low-density and high-density tree species from the Central Amazonian várzea floodplain forests (1–*Ficus insipida*, 2–*Pseudobombax munguba*, 3–*Luehea cymulosa*, 4–*Ilex inudata*, 5–*Macrobolium acaciifolium*, 6–*Albizia subdimitiata*, 7–*Sloanea terniflora*, 8–*Pouteria elegans*, 9–*Piranhea trifoliata*, 10–*Chrysophyllum argenteum*, 11–*Tabebuia barbata*, 12–*Eschweilera albiflora*). The diameter cutting limit of 50 cm is indicated.

different successional stages in the várzea called “Growth-Oriented Logging” (GOL).

Forests in early forest succession (early secondary stage and late secondary stage) are almost even-aged and have low tree species richness and more or less homogeneous structure (Schöngart et al., 2003). Due to the similarities in the growth patterns a management concept for this species early succession group is adequate if directed towards producing high quantities of qualitative timber for plywood and veneer (Worbes et al., 2001; Schöngart, 2003). But high volume stocks cannot be managed by a selection harvest system. After a selection logging of 3–5 trees ha⁻¹, the majority of low-density trees will have died following a cutting cycle of 10–25 years, due to the high increment rates and the relatively short life span of these species (Worbes et al., 1992; Schöngart, 2003). It can be assumed that the maximum harvest level of 3–5 trees ha⁻¹ would probably not create favourable light conditions needed on the forest floor for successful regeneration of low-density species, which require high light regimes (Wittmann and Junk, 2003). This can be shown through analysis of the population structure of tree species by 10-cm diameter classes (Hartshorn, 1980; Swaine and Whitmore, 1988) for the four successional stages (Fig. 6), related to

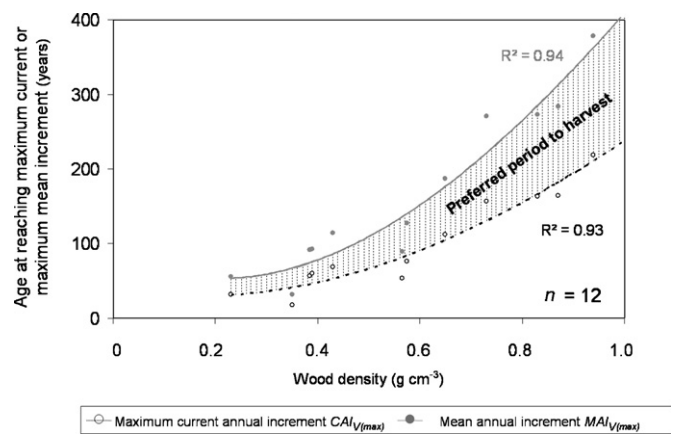


Fig. 4. Estimates by 12 growth models of the respective age at maximum current $CAI_{V(max)}$ and mean annual $MAI_{V(max)}$ volume increment rates (cf. Fig. 2). Both parameters are significantly correlated with specific wood density ($CAI_{V(max)} = -65.147 WD^3 + 389.04 WD^2 - 126.46 WD + 40.372$ and $MAI_{V(max)} = -185.74 WD^3 + 832.03 WD^2 - 322.64 WD + 85.901$). The hatched area indicates the preferred period for harvest defined as the time interval between $CAI_{V(max)}$ and $MAI_{V(max)}$ (cf. Fig. 2).

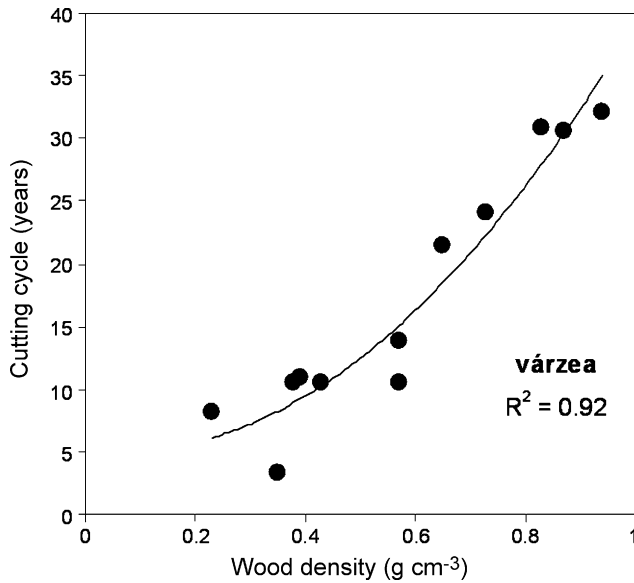


Fig. 5. Estimated cutting cycles of 12 timber species of várzea floodplain forests related to wood density. The relationship is indicated by a polynomial function (cutting cycle = $39.0515 \text{ WD}^2 - 5.0129 \text{ WD} + 5.2157$).

measurements of the relative photosynthetically active radiation (rPAR) at the forest floor for each successional stage (Wittmann and Junk, 2003). The low-density trees *I. inundata*, *L. cymulosa*, and *P. munguba* have, in the 20 year-old early successional stage, with 19.4% rPAR, a J-reverse curve (decreasing abundance with increasing DBH-class) indicating that these tree species had regenerated successfully. In the next 50 year-old early successional stage, the highest abundances of these species were observed in the mid-sized diameter classes and regeneration was absent, when solar radiation on the forest floor had declined to 9.6% rPAR. For *P. munguba*, a high number of regenerating trees were found in gaps, but the abrupt decline to the following diameter class indicated that this species could not grow successfully at this stage. With declining rPAR in the 125 year-old and 240 year-old climax stages, these latter low-density species appear only in low abundances in particular size classes, indicating that they regenerated only sporadically in these stands, probably after the creation of larger gaps (Ziburski, 1991). A polycyclic system is thus economically and ecologically not appropriate for management of these fast-growing low-density timber species.

To manage these shade-intolerant, light-demanding pioneers of early succession, a monocyclic silvicultural system (Shelterwood Systems) is adequate (Whitmore, 1998) (Fig. 7). Monocyclic systems attempt to produce a uniform crop of trees from young regeneration using both heavy harvesting and broad silvicultural treatments. A new even-aged crop is established by applying thinnings to natural regeneration (i.e., at seedling and sapling stages) of the desired trees. At an appropriate time, the overstorey of all marketable stems is removed during one or more harvests. The advantage of this system is high economic efficiency, but it is problematic in terms of removing nutrients and for risk of soil erosion (Lamprecht, 1989). The rotation period of a monocyclic system for low-density timber species during early succession of the várzea floodplains can be defined as the time interval between establishment of seedlings and the harvest of the mature trees ideally when they achieve their maximum current volume increment (cf. Fig. 4, Table 6). For *F. insipida* that dominated the early secondary stage, this period was 17 years. For tree species in the late secondary stage, the rotation period was 40–68 years. Monocyclic systems should, however, not use clear-cuttings, because young successional stages are mostly located near the river, where they play important ecological role by protecting against soil erosion (Wittmann et al., 2004; Schöngart et al., 2007). A modified monocyclic system could be the harvest of all trees with a DBH over the species-specific MLD (Table 6), creating large gaps in the stand to favour regeneration of light-demanding tree species. These gaps could be enlarged by 2–3 yields of the light-density tree species in 3–5 years intervals corresponding to their mean passage time through 10-cm DBH classes, until the original stand is completely removed and replaced (Fig. 7). In this phase, enrichment-plantings could be used to improve regeneration with more commercial tree species, if natural regeneration is absent. A second option is to transform stands of early forest succession directly or following one or more rotations to intermediate and climax stages formed by high-density tree species. This would correspond to natural successional dynamics of low várzea forests (Terborgh and Petren, 1991; Worbes et al., 1992; Wittmann et al., 2002). As forest succession (primary succession) initiates on newly created areas by sedimentation processes of white-water rivers, these newly established areas would compensate for those transformed to late successional forests. Silvicultural treatments for replacements, such as enrichment plantings and control (selection, thinning) of natural forest regeneration of tree species may require more research before being practical.

Table 6

Management criteria (minimum logging diameter-MLD, cutting cycle) derived from growth models for low-density and high-density tree species in the Central Amazonian floodplains

Tree species	Wood density (g cm^{-3})	MLD (cm)	Tree age at MLD (years)	Cutting cycle (years)
Low-density tree species				
<i>Ficus insipida</i>	0.35	55	17.0 ± 3.6	3.3 (2.4–3.7)
<i>Pseudobombax munguba</i>	0.23	47	39.5 ± 2.4	8.2 (7.9–8.9)
<i>Ilex inundata</i>	0.38	59	61.0 ± 9.7	10.5 (8.7–12.0)
<i>Macrobium acaciifolium</i>	0.43	62	67.0 ± 5.6	10.5 (9.9–11.7)
<i>Albizia subdimidiata</i>	0.57	49	53.5 ± 7.1	10.5 (9.5–12.4)
<i>Luehea cymulosa</i>	0.39	61	68.5 ± 11.7	11.0 (9.3–13.1)
<i>Sloanea terniflora</i>	0.57	58	82.0 ± 9.1	13.9 (12.6–15.7)
High-density tree species				
<i>Pouteria elegans</i>	0.65	54	120.0 ± 21.8	21.5 (18.2–26.3)
<i>Chrysophyllum argenteum</i>	0.73	58	144.0 ± 22.4	24.1 (21.0–28.7)
<i>Eschweilera albiflora</i>	0.83	53	164.5 ± 44.6	30.9 (22.6–39.5)
<i>Tabebuia barbata</i>	0.87	54	168.5 ± 19.8	30.6 (27.5–34.9)
<i>Piranhea trifoliata</i>	0.94	70	227.0 ± 23.5	32.1 (29.1–35.8)

Mean tree age and standard deviation at MLD calculated by the age–diameter relationships indicated in Table 5. Cutting cycles are estimated by the mean tree age at MLD, minimum and maximum values in brackets consider standard deviation of tree age at MLD.

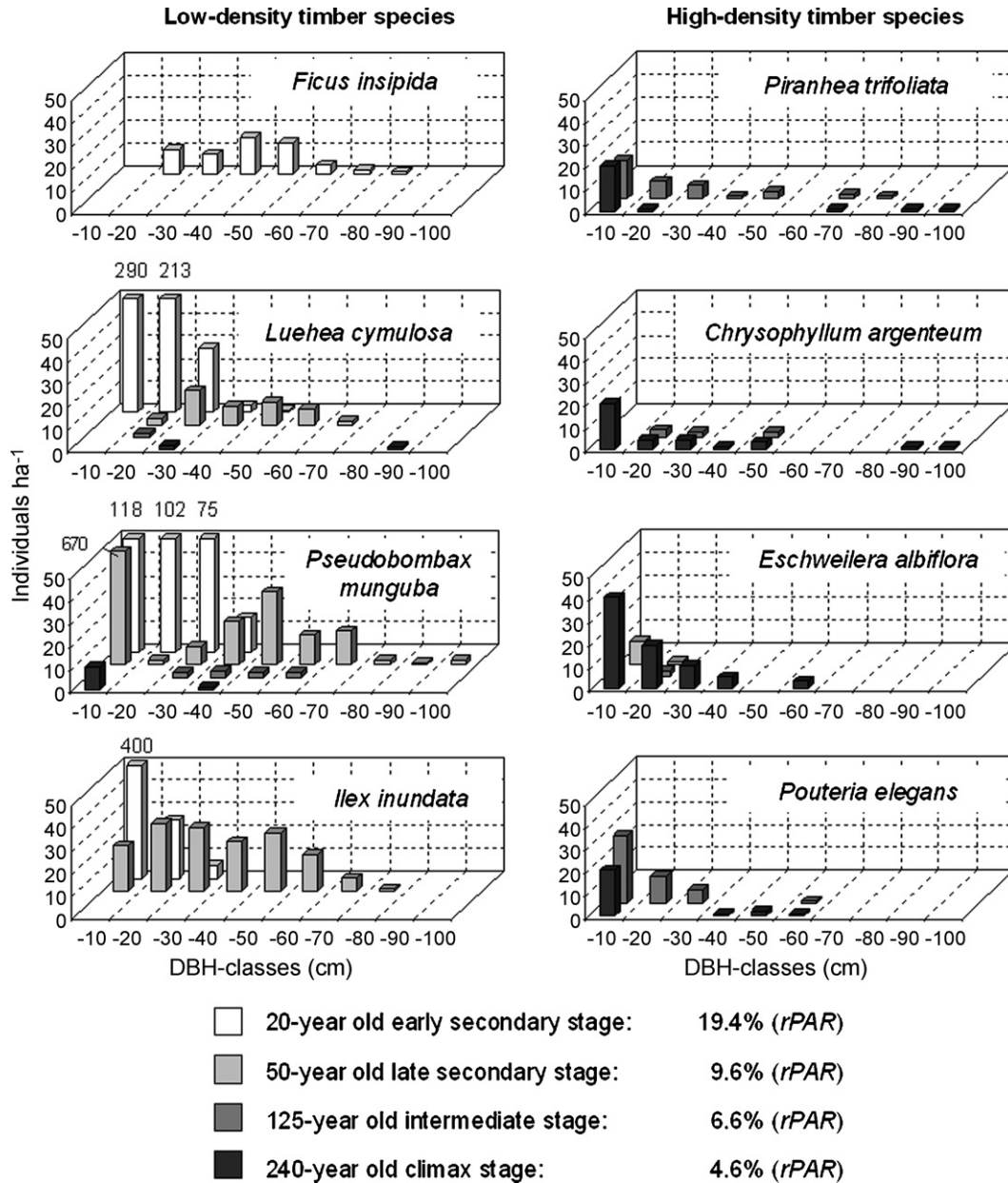


Fig. 6. Population structure (abundance for 10-cm diameter classes) of the low-density timber species (left) and high-density timber species (right) of four different successional stages. Data for mean, minimum and maximum relative photosynthetically active radiation (*rPAR*) are obtained from Wittmann and Junk (2003).

High-density species such as *T. barbata*, *P. trifoliata*, *E. albiflora*, and *C. argenteum* dominated the old-growth forests (intermediate and climax stages) of the várzea, and were characterized by an uneven-age structure (Worbes et al., 1992; Schöngart et al., 2003). This species group achieved volume stockings of 61–71 m³ ha⁻¹ (Tables 3 and 4) concentrated on 13–15 trees ha⁻¹ over the DCL of 50 cm. Therefore, the aim of the forest management in these stands was the production of high-quality timber with high wood densities. The high-density species had similar wood growth. Wood densities were in the range of 0.72–0.94 g cm⁻³ and life spans were 200–400 years (Worbes et al., 1992; Schöngart, 2003). The cumulative diameter growth curves suggested that this species group required periods of 100–150 years to exceed the DCL of 50 cm (Fig. 3). The current volume increments achieved their maximum at an age of about 145–230 years (Fig. 4, Table 5).

In contrast to the low-density species of early successional stages, the high-density species *P. trifoliata*, *C. argenteum*, *E.*

albiflora, and *P. elegans* had mostly *J*-reverse distributions, despite relatively low light intensities of 4.6–6.6 *rPAR* on the forest floor (Fig. 6). This pattern is characteristic for shade-tolerant tree species (Hartshorn, 1980; Swaine and Whitmore, 1988). Therefore, polycyclic selection harvest systems are appropriate to manage these timber resources (Whitmore, 1998) (Fig. 7). Selection logging should concentrate on individuals above species-specific MLDs varying between 53 and 70 cm (Table 6). The application of these MLDs predicted cutting cycles of 22–32 years for the high-density species (Fig. 5). An ideal management regime would be the establishment of species-specific cutting cycles, which is still not viable under the given socio-economic conditions in the várzea, because this implies having forest inventories over large areas. Therefore, a cutting cycle of 30 years common to all species is suggested (Fig. 7) to decrease costs and time for conducting forest inventories, logging, and skidding. The application of low impact management in old-growth várzea forests enabling a yield of

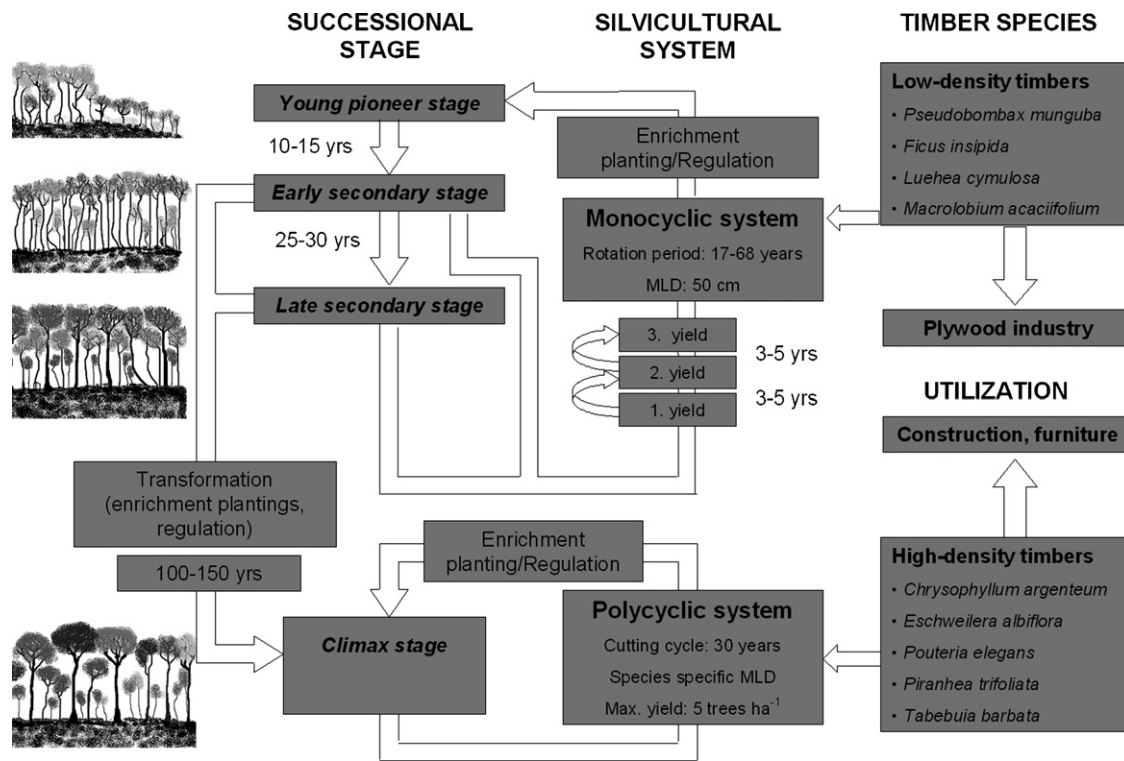


Fig. 7. Forest management concept of Central Amazonian floodplain forests of the low várzea differentiated for low-density (monocyclic system) and high-density (polycyclic system) timber species.

10 m³ ha⁻¹, or possibly more, using a cutting cycle of only 10 years, as it is allowed by the normative instruction (IN) no. 5, should be avoided, because it risks overexploiting the timber stocks of these slow-growing species.

5. Discussion and conclusions

GOL is the first concept for a sustainable timber harvest in Central Amazonian floodplain forests, based on the lifetime growth rates of commercial tree species (Fig. 7). Timber exploitation in the Amazonian várzea has been unsustainable and mostly illegal, over most decades of the 20th century (Ayres, 1993; Higuchi et al., 1994; Hummel, 1994; Albernaz and Ayres, 1999; Anderson et al., 1999). An increased timber extraction from the várzea forests and the competition between forest management and other land-use options (agriculture, pasture, settlement) require strategies and concepts for the long-term conservation of the floodplain forests. In this context, a large number of participatory community-based forest management, and forest management on small scales (private properties <500 ha), have been established in recent years (Amaral and Amaral Neto, 2005), within the frame of large development programs such as PPG7-Projects Pro-Manejo and Pro-Várzea, including projects by other national and international government institutions and NGOs. In the Amazonas state >450 management plans for mainly small-scale private properties and some community-based forest managements have been implemented, mainly in várzea floodplains along the Solimões, Amazonas, Juruá, and Madeira rivers (IEB, 2007). But the long-term success of such forest management planning depends primarily on the ecological sustainability of timber production that requires information on the growth rates of commercial tree species (Boot and Gullison, 1995; Brienen and Zuidema, 2006). So far, most studies have promoted selection harvest combined with reduced impact logging as sustainable forest management

(Dykstra and Heinrich, 1996; Vidal et al., 1997, 2002; Gerwing, 2002). These forest operations are necessary for the conservation of tropical forests to significantly reduce residual stand damage and soil disturbance. Further, reduced impact logging is also less costly and more profitable than conventional systems (Johns et al., 1996; Boltz et al., 2001; Holmes et al., 2002). But these technical and logistical improvements for timber extraction are not sufficient to provide a sustainable management of the timber resources, and there is much scepticism about using growth rates for managing tropical forests.

The definition of species-specific and site-specific management criteria based on wood growth data and structural analysis of the natural stands in the tropics is rare (Nebel et al., 2001a; Schwartz et al., 2002; Sokpon and Biaou, 2002; Schöngart et al., 2007). Growth models for tropical tree species are scarce owing to the high tree diversity and methodological problems in determining tree ages and lifetime growth rates. Permanent sample plot data are the most important source to understand diameter growth of tropical tree species and stands. Most studies have monitored tree growth on permanent plots using repeated diameter measurements (Clark and Clark, 1999; Finegan et al., 1999; Nebel et al., 2001b; Nebel and Meilby, 2005; Valle et al., 2006) or dendrometer bands (Silva et al., 2002), followed by estimating maximum allowable yields using growth models (Vanclay, 1994; Alder and Silva, 2000; Sist et al., 2003) to evaluate volume increment at the stand level (Dauber et al., 2005; Neeff and Santos, 2005; Valle et al., 2006). However, this cannot provide criteria for the species-specific management of timber stocks. Other models consider diameter growth rates of trees from different size classes of a species to project tree growth over the total lifespan (Lieberman and Lieberman, 1985; Korning and Balslev, 1994; Terborgh et al., 1997; Clark and Clark, 1999; Nebel, 2001; Sokpon and Biaou, 2002; Nebel and Meilby, 2005). This latter method is limited due to the relatively short period for monitoring diameter growth (Condit,

1995) and low density of commercial species, especially of larger sizes on permanent sample plots (Clark and Clark, 1996). The modelled growth trajectories are based on simplifications and assumptions, which can result in unrealistic long-term growth data and yield estimations for tree species. Lifetime trajectories of tree species projected by diameter growth rates of different size classes based on permanent sample plot data often underestimate the growth rates of future canopy trees, thus overestimating their tree age (Brienen and Zuidema, 2006). Other studies have evaluated stand growth and development for 240–400 years by process-oriented forest growth models (e.g., for the Southeast Asian dipterocarp lowland forests or tropical forests in Venezuela (Huth and Ditzer, 2000; Kammesheidt et al., 2001; Huth et al., 2005). These models were based on carbon balances of individual trees from different crown layers, modelled using major physiological processes (photosynthesis, respiration) and allometric functions relating above-ground biomass, tree height, stem diameter, and crown parameters. However, estimation of maximum tree ages is based on assumptions and the effects of increasing atmospheric temperature and CO₂-concentrations (IPCC, 2007) and its feedback on photosynthetic and respiration rates have not been considered (Cox et al., 2000), which raises questions about the suitability of these models to predict stand development over centuries (Huth and Ditzer, 2000; Huth et al., 2005). These models are also not able to define species-specific management options.

In contrast to these methods for growth modelling, tree-ring analysis provides a powerful tool (Brienen and Zuidema, 2007) that yields direct values for tree ages and lifetime growth rates of individual trees through retrospective analysis. For tree-ring analysis, emergent trees are sampled that successfully established in the canopy and thus the provided diameter growth data are representative of realistic growth trajectories for harvestable trees (Brienen and Zuidema, 2006). Thus, tree ages determined by dendrochronology yield much lower tree ages than do diameter growth trajectories that were established on short-term permanent sample plot data, which can result in tree ages of more than 1000 years for Central Amazonian tree species (Laurance et al., 2004).

The application of tree-ring data to define management options for tropical timber species is not new discovery. During 1855–1862, in the British colony Burma (today Myanmar), Dietrich Brandis (1824–1906) developed a specific management plan for teak (*Tectona grandis* L., Verbenaceae) based on stand inventories performed along transects (linear taxation) and ring counting of stumps. He estimated a cutting cycle of 24 years (mean time in a size class) and a diameter cutting limit of 4 cubits (~58.8 cm) to promote sustainable use of the teak stocks (Hesmer, 1975; Dawkins and Philip, 1998). But due to controversy about the existence of annual tree rings in the wood of tropical tree species (cf. Worbes and Junk, 1999), the application of dendrochronological methods to model tree growth has been rare for tropical forests. The existence of rings is evident for many tropical tree species from different ecosystems (Worbes, 2002). The growth trajectory for a particular species can be considered as a general representation of growth dynamics for that species under certain site conditions (climate, hydrology, edaphic conditions, successional stage). Growth models can easily be established by tree-ring analysis for different site conditions (Schöngart et al., 2005). The developed GOL concept for the várzea (Fig. 7) was adapted to the natural successional dynamics and is an approach to a higher degree of sustainability in tropical forest management considering the population structure and wood increments of the low-density and high-density tree species. The development of such concepts will provide a powerful tool to promote sustainable management

of timber resources also in other forest ecosystems in Amazonia and in other tropical regions.

However, cutting cycles or rotation periods only guarantee a sustainable use of the timber resources if the harvested species continue to recruit. Little information is available on germination, growth, and establishment of seedlings and saplings as well as their relationship to external abiotic (flooding, light conditions, water and nutrient supply) and biotic factors (seed banks, interspecific and intra-specific competition, herbivory). Further studies should therefore focus on the germination, growth, and mortality rates of seedlings of timber species and their relationship to environmental factors (Wittmann and Junk, 2003). Costs-benefit analyses are necessary to evaluate silvicultural improvements, such as enrichment plantings and thinning to regulate recruitment and competition as well as removing of branches to increase stem quality (Lamprecht, 1989; Mesquita, 2000).

Acknowledgements

I thank the Instituto de Desenvolvimento Sustentável Mamirauá in Tefé for logistical support to the fieldwork. This study was financed by the INPA/Max-Planck Project and by the Federal Ministry of Education, Science and Technology (BMBF) in the frame of the SHIFT-Program ENV 29/2 “Stress physiology and primary production of floodplain forests”, the UNESCO IHPO-Demonstration projects on Ecohydrology “Sustainable timber production and management of Central Amazonian white-water floodplains” as well as the Project 680021/2005-1 of the Brazilian Research Council (CNPq) “Studies to sustain the community-based production in floodplain forests of the Mamirauá and Amanã Reserves”. I thank two referees for valuable comments and linguistic help to improve the manuscript.

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