

Long-Term Impacts of Fuelwood Extraction on a Tropical Montane Cloud Forest

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ABSTRACT

Fuelwood extracted from natural forests serves as a principal energy source in rural regions of many tropical countries. Although fuelwood extraction (even low intensities) might strongly impact the structure and species composition of natural forests, long-term studies remain scarce. Here, we estimate the potential long-term impacts (over several hundred years) of such repeated harvesting of single trees on tropical montane cloud forest in central Veracruz, Mexico, by applying a process-based forest growth model. We simulate a wide range of possible harvesting scenarios differing in wood volume harvested and preferred tree species and sizes, and use a set of indicators to compare their impacts on forest size structure and community composition. Results showed that the overall impact on forest structure and community composition increased linearly with the amount of harvested wood volume. Even at low levels of

harvesting, forest size structure became more homogeneous in the long term because large old trees disappeared from the forest, but these changes might take decades or even centuries. Although recruitment of harvested species benefited from harvesting, species composition shifted to tree species that are not used for fuelwood. Our results demonstrate that fuelwood extraction can have marked long-term impacts on tropical montane cloud forests. The results also offer the possibility to support the design of management strategies for the natural species-rich forests that achieve a balance between economic needs and ecological goals of the stakeholders.

Key words: disturbance; ecosystem dynamics; fire wood; forest model; FORMIND; Mexico; logging impacts; selective logging; sustainable forest management; tropical montane cloud forest.

INTRODUCTION

In many rural regions in the tropics, fuelwood is the most important energy source (Broadhead and

others 2001) and many people rely on fuelwood extraction from natural forests for cooking and heating (International Energy Agency 2002). Often the amount of wood harvested for fuelwood greatly exceeds harvested volumes for industrial and other purposes. For example, in Mexico the estimated annual fuelwood consumption is almost twice as high as commercial timber production (Torres-Rojo 2004) and approximately 25 million people depend on fuelwood (Masera and others 2004).

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Vast forest areas are used for fuelwood extraction for domestic use, cooking and heating, local market supply, and charcoal production (Masera and others 2004). However, only in about 15% of the forested area, of which 70–80% is common property of rural communities, formal forest management is applied (SARH 1992; World Bank 1995; Klooster and Masera 2000). Local patterns of fuelwood use are very heterogeneous and an analysis of fuelwood consumption and availability in Mexico has classified 46% of all municipalities as medium to high priority areas, indicating high fuelwood use in these areas (Masera and others 2004). In recent decades, population growth and accelerated forest conversion to agricultural fields or pastures led to increased pressure on the remaining forests to satisfy needs for fuelwood (Echeverría and others 2007).

In contrast to logging for timber, fuelwood harvesting usually occurs continually in the same place and does not rely on heavy machinery for yarding. Although the impacts of timber harvesting on forest biodiversity and wildlife have been subject to extensive research (for example, Fimbel and others 2001; Putz and others 2001; Zarin and others 2004; Meijard and others 2005; Asner and others 2006), long-term consequences of continuous fuelwood harvesting on structure and species composition of tropical forests are largely unknown. Moreover, short-term field research may not detect the effects of low-intensity fuelwood extraction because relatively low use intensities result in slow ecosystem changes which might only be manifested over several decades or even centuries.

Modeling studies, on the other hand, can address the likely long-term impacts on forest dynamics but usually focus on commercial high-intensity selective logging for timber under different management scenarios (for example, Ditzer and others 2000; Huth and Ditzer 2001; Kammesheidt and others 2001, 2002; van Gardingen and others 2003; Huth and others 2005; R ger and others 2007; but see Sist and others 2003; Phillips and others 2004; Gourlet-Fleury and others 2005; Valle and others 2007 for intermediate and low logging intensities). To our knowledge only a few studies have examined the long-term impacts of fuelwood extraction on tropical forest ecosystem dynamics (for example, Sundriyal and Sharma 1996; Holder 2004; McCrary and others 2005).

Here, we apply for the first time an individual-oriented process-based forest growth model (FORMIND; see K hler and Huth 1998; K hler and others 2001) to identify ecological long-term

implications of repeated fuelwood harvesting on tropical montane cloud forest (TMCF) in central Veracruz, Mexico. Although TMCFs in Mexico only cover less than 1% of the territory, they contribute 10% to plant biodiversity (Challenger 1998). In the study area, about 10% of the original TMCF cover remains and in many of the remaining TMCF fragments large living trees are harvested for fuelwood (Williams-Linera and others 2002). In this study, we simulate and analyze various harvesting scenarios including a broad range of harvesting intensities ($5\text{--}100\text{ m}^3\text{ ha}^{-1}$ with a harvesting cycle of 10 years) and harvesting regimes that target different species groups and tree sizes.

Our objectives were (1) to assess changes in forest structure and community composition under different harvesting strategies over several hundred years and (2) to determine the limits of sustained fuelwood yields in terms of the regeneration capacity of the forest. With our study we aim to contribute to an improved understanding of long-term ecosystem dynamics under anthropogenic disturbance and to a sustainable use of native forest resources.

METHODS

Study Area

Tropical montane cloud forest (TMCF) in central Veracruz, Mexico ($19^{\circ}30' \text{ N}$, $96^{\circ}54' \text{ W}$) occurs at an altitude between 1250 and 2050 m. The climate is mild and humid throughout the year with three seasons. A relatively dry-cool season extends from November to March, a dry-warm season from April to May, and a wet-warm season from June to October. Annual precipitation increases with increasing altitude from 1350 to 2200 mm; mean annual temperature decreases from 18 to 12 C  (Williams-Linera 2002). The soil has been classified as Andosol (Rossignol 1987). Characteristic tree species include *Carpinus caroliniana*, *Clethra mexicana*, *Fagus grandifolia*, *Liquidambar styraciflua*, *Quercus germana*, *Q. leiophylla*, *Q. xalapensis*, and *Turpinia insignis*. Inventory data from five old-growth forest fragments (0.1 ha each) indicate that average stem density is $1234 (\pm 329)\text{ ind. ha}^{-1}$ and basal area is $55.2 (\pm 21.3)\text{ m}^2\text{ ha}^{-1}$ (Williams-Linera 2002).

To enable an individual-based simulation of forest dynamics, the 57 native tree species that occurred in the inventory data were grouped into six plant functional types (PFTs) which share similar physiological, morphological, and ecological characteristics (for example, Smith and others 1997; K hler and others 2000). Criteria for classification

Table 1. Tree Species Grouping into PFTs

Plant functional type	PFT	Shade tolerance	Maximum height (m)	Examples
Early successional small trees	1	Intolerant	15	<i>Heliocarpus donnell-smithii</i> , <i>Myrsine coriacea</i>
Mid successional small trees	2	Intermediate	15	<i>Miconia glaberrima</i> , <i>Oreopanax xalapensis</i>
Late successional small trees	3	Tolerant	15	<i>Cinnamomum effusum</i> , <i>Ilex belizensis</i>
Mid successional canopy trees	4	Intermediate	25	Some <i>Quercus</i> spp.
Late successional canopy trees	5	Tolerant	25	<i>Magnolia schiedeana</i> , <i>Beilschmiedia mexicana</i>
Mid successional emergent trees	6	Intermediate	35	<i>Liquidambar styraciflua</i> , <i>Clethra mexicana</i>

into PFTs were shade tolerance and maximum attainable height. Three levels of shade tolerance were distinguished: shade-intolerant, intermediate, and shade-tolerant. Three height groups were considered: small trees (max. height ≤ 15 m, max. dbh ≤ 35 cm), canopy trees (max. height ≤ 25 m, max. dbh ≤ 80 cm), and emergent trees (max. height ≤ 35 m, max. dbh ≤ 100 cm). This classification resulted in six PFTs because some of the combinations are rare (Table 1; see Table S3 in the Supporting Online Material for a list of tree species).

The Forest Growth Model FORMIND

The forest growth model FORMIND simulates the spatial and temporal dynamics of uneven-aged mixed forest stands (for example, Köhler and Huth 1998; Köhler 2000; Köhler and others 2001, 2003; Huth and others 2004, 2005). The model simulates a forest stand (in annual time steps) as a mosaic of interacting grid cells with a size of 20 m \times 20 m, corresponding to the crown size of large mature trees. FORMIND simulates the growth of every individual tree larger than 1 cm dbh. Trees which belong to the same PFT and establish in the same year in the same grid cell are grouped into a cohort and share equal dbh, height, and so on. Trees with dbh greater than 40 cm are usually simulated individually, because all other trees from their cohort have died. The main processes of forest dynamics, that is, recruitment, growth, mortality, gap creation by large falling trees, and competition for light and space, are represented in the model. Thus, FORMIND allows simulating harvesting scenarios targeting different species groups and tree sizes and deriving their impact on forest structure and composition from the interaction of

physiological, morphological, and ecological characteristics of different tree species and mechanisms acting at the individual-tree-level.

It is assumed that light availability is the main driving force for individual tree growth and forest succession. Within each grid cell all trees compete for light and space following the gap model approach (Shugart 1984). For the explicit modeling of competition for light, each grid cell is divided vertically into small height layers (0.5 m). In each height layer the leaf area is summed up and the light climate in the forest interior is calculated from an extinction law. Biomass increment of each individual tree is determined from its carbon balance, that is, gains through photosynthesis and losses due to respiration and litter fall. Photosynthesis and respiration are calculated depending on the available light at the crown of the tree, the photosynthetic capacity of leaves and leaf area, and its biomass, respectively. The growth process formulations are modified from the models FORMIX3 and FORMIX3-Q (Ditzer and others 2000; Huth and Ditzer 2000, 2001).

Allometric functions and geometrical relations are used to calculate the dbh of the tree from aboveground biomass, and tree height, crown diameter, and stem volume from dbh (Köhler and Huth 1998; Köhler 2000). Tree mortality can occur through self-thinning in densely populated grid cells, senescence, or gap formation by large falling trees. Gap formation links neighboring grid cells. The number of recruits of each PFT that exceed the dbh threshold of 1 cm in each year depends on the light level at the forest floor and the shade tolerance of the PFT. Site conditions (for example, soil moisture, nutrient availability) are assumed to be homogeneous and there is no interannual variability of climatic conditions in the model.

A detailed description of FORMIND is given in the Supporting Online Material.

Model Parameterization and Evaluation

Parameters of FORMIND for TMCF were obtained for environmental conditions, tree geometry, tree growth, recruitment, and mortality (Supporting Online Material, Text and Table S1). For some model parameters field measurements from the study site were available (for example, irradiance, wood density). Other parameters were defined based on measurements for the same species (or species of the same genus) from different locations (for example, allometric relationships, maximum rate of photosynthesis). When no direct measurements were available, model parameters were fitted such that related field data from the study site were reproduced. This way, respiration parameters were adjusted such that simulated diameter increment corresponded to field data and recruitment parameters were chosen such that simulated old-growth forest composition matched field observations.

Here we summarize the main results of the model evaluation (more details can be found in the Supporting Online Material). We used inventory data of five fragments of old-growth (that is, late successional) TMCF in central Veracruz which show little evidence of anthropogenic disturbance such as tree cutting or cattle grazing (Williams-Linera 2002) to test the ability of the model to reproduce observed characteristics of old-growth forest. Simulated stem numbers, basal area, and diameter distributions of the different PFTs were consistent with field data. Furthermore, field data from a chronosequence study in the study area covering forest ages of 0.5–80 years (Muñiz-Castro and others 2006) allowed for a model validation with independent data on forest regeneration after the abandonment of cattle grazing. Forest regeneration was also predicted relatively well by our model but a slight overestimation of the speed of forest recovery could be observed. Trees in the model seemed to grow slightly faster than in reality, because stem numbers were predicted correctly, whereas basal area was slightly overestimated (Supporting Online Material, Figure S5).

Fuelwood Harvesting Scenarios

In central Veracruz, fuelwood is harvested both from communal and private forests by individual villagers or small associations of woodcutters without formal training and technical equipment other than chainsaws. Large trees (>40 cm dbh) are felled with chainsaws, *in situ* cut into pieces, and

extracted with the help of pack animals. Preferred tree species for fuelwood include oaks (*Quercus* spp.), hornbeam (*Carpinus caroliniana*), sweetgum (*Liquidambar styraciflua*), and *Clethra mexicana* (Haeckel 2006). These species were mainly classified as PFTs 4 and 6, that is, the canopy and emergent species with intermediate shade tolerance.

To capture these preferences, we simulated four fuelwood harvesting scenarios (Table 2). In the first two scenarios (*S1*, *S2*), only trees of PFTs 4 and 6 were logged. In scenarios *S3* and *S4*, harvesting was applied to PFTs 4, 5, and 6. In scenarios *S1* and *S3*, harvesting concentrated on medium-sized trees with a dbh of 40–60 cm which are preferentially cut in the study area for fuelwood and charcoal production for local market supply (G. Williams-Linera, pers. observation). Scenarios *S2* and *S4* allowed cutting of all trees larger than 40 cm dbh. Within the range of allowed dbh values, the largest trees were logged first.

There are few data available on actual wood extraction. Therefore, we varied the harvesting intensity for each of the scenarios from 5 to 100 m³ ha⁻¹ (with a harvesting cycle of 10 years) to cover a broad spectrum of possible harvesting intensities and to investigate their potential effects. Total standing stem volume of an undisturbed old-growth forest in the study area is approximately 500 m³ ha⁻¹ (we calculated stem volume for each tree in the inventory data based on allometric and geometrical relationships and wood density data; compare Supporting Online Material, Figure S1). Consequently 1–20% of total stem volume is harvested every 10 years by the harvesting scenarios. A harvesting intensity of 100 m³ ha⁻¹ corresponds to 23–47 trees ha⁻¹ depending on the average dbh of logged trees, and this rather high harvesting intensity was simulated to study the potential of the forest for wood supply.

To explore long-term consequences of fuelwood harvesting, we assumed that any external conditions—other than fuelwood harvesting—which influence the species composition of the forests or the processes represented in FORMIND are

Table 2. Fuelwood Harvesting Scenarios

Scenario	Harvested PFTs ¹	Diameter range (cm)
<i>S1</i>	4, 6	40–60
<i>S2</i>	4, 6	>40
<i>S3</i>	4, 5, 6	40–60
<i>S4</i>	4, 5, 6	>40

¹For a description of PFTs see Table 1.

constant. Thus, we assumed that climate change, land-use change, and invasions of exotic species do not take place. We used inventory data from the study area as the initial condition and then simulated forest dynamics over a 100-year period to allow the model to establish a steady state old-growth forest without fuelwood harvesting. A forest is in a steady state when changes in forest structure and species composition do not show any long-term trends. Harvesting scenarios were then applied over a simulation time of 400 years (that is, time steps 100–500 in the model). Preliminary simulation results showed that it might take about 100 years until the forest reaches a new steady state according to the harvesting regime. Thus, from about time step 200 onwards the forest structure and composition fluctuate around average values.

If at a given time step the stem volume of all harvestable trees in the simulation area did not reach the volume value aimed by the harvesting scenario, the respective harvesting operation was omitted, that is, the inter-harvest interval was extended from 10 to 20 years and so on. This was done to keep harvesting scenarios comparable and to clearly reveal the limits of a sustained fuelwood extraction. The percentage of omitted harvesting events was used as an indicator of the discontinuity of fuelwood yields under the given harvesting scenario.

Felled trees were directed to already existing gaps if possible. Apart from trees that were killed by the falling tree, no additional logging damage was considered because wood extraction in the study area is carried out without heavy machinery but with the help of pack animals.

For illustration, Figure 1 shows the development of basal area for a simulation run when harvesting scenario *S3* is applied with a harvest aim of $100 \text{ m}^3 \text{ ha}^{-1}$.

Assessment of Harvesting Scenarios

We evaluated several economic and ecological implications of a range of harvesting scenarios. Mean annual harvest (H ; in $\text{m}^3 \text{ ha}^{-1} \text{ y}^{-1}$) and the percentage of omitted harvesting events (PO ; in %) are economic indicators of amount and continuity of wood harvest. Stem numbers in different diameter classes and importance values (IV , equation 2) describe changes of the size structure of the forest and the dominance of PFTs in terms of relative abundance and basal area, respectively. To aggregate the ecological effects of fuelwood harvesting, we calculated an index of structural change (ISC , equation 1), an index of compositional change

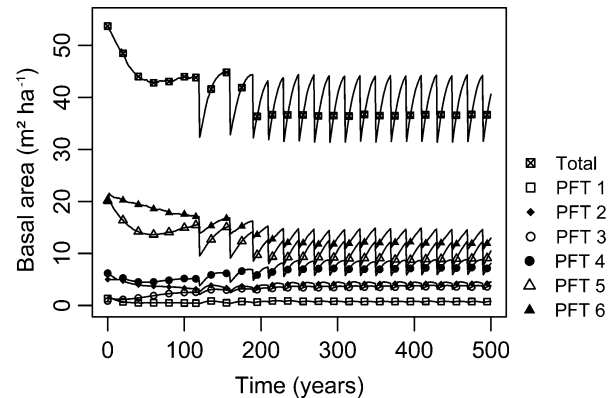


Figure 1. Simulated temporal development of basal area of different PFTs in TCMF in central Veracruz, Mexico, subjected to fuelwood harvesting. Fuelwood harvesting scenario *S3* is applied, that is, trees of PFTs 4, 5, and 6, 40–60 cm dbh are targeted. The harvest aim is $100 \text{ m}^3 \text{ ha}^{-1}$ and harvest cycle is 10 years. Harvesting starts after 100 years. On average, every second harvest has to be omitted because the harvestable volume is lower than the harvest aim.

(ICC , equation 3), and stand leaf area index (LAI). ISC and ICC indicate how similar the size structure and PFT composition of logged forest is to undisturbed forest, and LAI is an indicator of the potential of the forest to protect the soil from erosion. Rainfall interception is positively correlated with LAI (for example, Fleischbein and others 2005), thus the proportion of rain directly hitting the ground decreases with increasing LAI .

H and PO were calculated for each scenario over the simulation period 100–500 years.

To account for the variability of forest structure and composition over time even when in a steady state, ISC , ICC , and LAI represent averages over the entire simulation area and over annual values for the last 100 years of the simulation (that is, simulation time 400–500). ISC was calculated as

$$ISC = \frac{1}{5} \sum_{i=1}^5 \frac{|\bar{x}_{s_i} - \bar{x}_{c_i}|}{\bar{x}_{c_i}}, \quad (1)$$

that is, the differences in mean tree numbers (simulation time 400–500) of five diameter classes ($i = 1$: 5–20 cm, $i = 2$: 20–40 cm, $i = 3$: 40–60 cm, $i = 4$: 60–80 cm, $i = 5$: 80–100 cm dbh) of a simulated logged forest (\bar{x}_{s_i} , $i = 1-5$) in comparison to a simulated old-growth forest (\bar{x}_{c_i} , $i = 1-5$) where no harvesting had been applied. An ISC of zero indicates that the size structure of the logged forest does not differ from an unlogged forest. The higher the ISC score, the larger the difference in size structure of the logged and the unlogged forest.

ICC indicates the change in relative importance of PFTs of the logged forest in relation to an undisturbed old-growth forest based on importance values (*IV*). Importance values of the different PFTs (IV_j , $j = 1-6$, for a description of PFTs see Table 1) were calculated as

$$IV_j = \frac{1}{2} \left(\frac{ba_j}{ba_{total}} + \frac{n_j}{n_{total}} \right), \quad (2)$$

that is, the normalized sum of the relative basal area (ba , $m^2 ha^{-1}$) and relative density (n , trees ha^{-1}) of trees 5 cm or greater dbh of each PFT (j) in relation to all PFTs (total). *ICC* was then calculated as

$$ICC = \frac{1}{6} \sum_{j=1}^6 \frac{|\overline{IV}_{s_j} - \overline{IV}_{c_j}|}{\overline{IV}_{c_j}}, \quad (3)$$

that is, summing the differences between the mean *IV*s of PFT j from the harvesting scenario (\overline{IV}_{s_j}) and the mean *IV*s from the control scenario \overline{IV}_{c_j} (undisturbed old-growth forest), and dividing it by the mean *IV*s of PFT j from the control scenario \overline{IV}_{c_j} for simulation time 400–500. Again, an *ICC* of zero indicates that the *IV*s of the six PFTs of the logged forest do not differ from an unlogged forest and an increasing difference in PFT composition is reflected by increasing *ICC* score.

LAI values were determined from LAIs of individual trees and averaged over the simulation time 400–500.

Comparison with Other Forest Management Strategies

For comparison of harvesting scenarios with other land uses, we also calculated *H*, *ISC*, *ICC*, and *LAI* for simulated intensively managed young secondary forests (only PFTs 4 and 6) and simulated even-

aged plantations of only PFT 6. The secondary forest is dominated by PFTs 4 and 6, and regeneration of other species is prevented. Management consists in harvesting approximately $60 m^3 ha^{-1}$ every 5 years. The plantation is clear-cut every 25 years. *ISC*, *ICC*, and *LAI* were averaged for each scenario for a period of 100 years.

Overall Ecological Impact

To contrast the amount of harvested fuelwood with the overall ecological impact of a harvesting scenario, we calculated an ecological index (*EI*), which measures the structural and compositional similarity of a logged forest to undisturbed old-growth forest. *EI* includes the ecological indicators *ISC*, *ICC*, and *LAI*, which were divided by the maximum value obtained from any of the harvesting scenarios (ISC_{max} , ICC_{max} , LAI_{max}) to scale them to values between 0 and 1, and then summed up weighted equally:

$$EI = \frac{1}{3} \left(\left(1 - \frac{ISC}{ISC_{max}} \right) + \left(1 - \frac{ICC}{ICC_{max}} \right) + \frac{LAI}{LAI_{max}} \right). \quad (4)$$

RESULTS

Amount of Harvested Fuelwood

Mean annual harvest (*H*) and percentage of omitted harvesting events (*PO*) in relation to harvest aim over the 400-year period of harvesting are shown in Figure 2. Maximum *H* ranged from 2 to $12 m^3 ha^{-1} y^{-1}$ for the different harvesting scenarios and intensities, that is, the harvest aims (Figure 2A; results $> 10 m^3 ha^{-1} y^{-1}$ not shown). In scenario *S1* (PFTs 4 and 6; dbh 40–60 cm), volumes higher than $50 m^3 ha^{-1}$ could never be achieved, and consequently no harvesting above

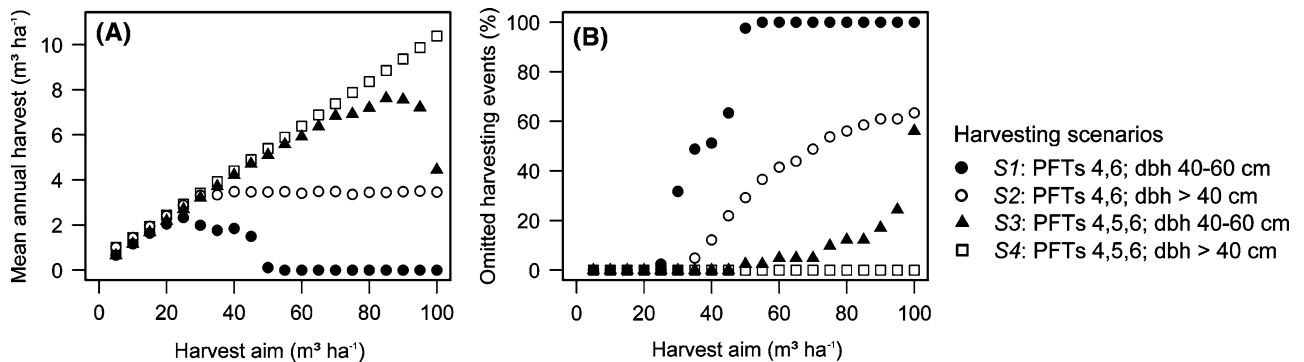


Figure 2. Mean annual harvest (A) and percentage of omitted harvesting events (B) in relation to harvest aim for four fuelwood harvesting scenarios in TMCF in central Veracruz, Mexico. Harvest aim varied from 5 to $100 m^3 ha^{-1}$, harvest cycle is 10 years for a simulation period of 400 years.

50 m³ ha⁻¹ took place in this scenario (Figure 2B). Harvesting volumes above 30 m³ ha⁻¹ under scenario *S2* (PFTs 4 and 6; dbh >40 cm) were only possible with time lags between two harvesting events (that is, omitting harvesting events; Figure 2B). For scenario *S3* (PFTs 4, 5, and 6; dbh 40–60 cm), harvesting events had to be omitted only occasionally when the harvest aim was higher than 45 m³ ha⁻¹ (Figure 2B). Only in the case of scenario *S4* (PFTs 4, 5, and 6; dbh >40 cm), harvesting was never omitted (Figure 2B).

Changes in Stem Numbers

Stem numbers in the two smallest diameter classes (5–20 cm and 20–40 cm dbh) increased with increasing wood extraction for all scenarios indicating that recruitment benefited from harvesting (Figure 3A, B). Recruitment to the 5–20 cm dbh class increased most strongly for PFT 1, followed by PFTs with intermediate shade tolerance (results not shown). The intermediate diameter class (40–60 cm dbh) was the only diameter class from which trees were harvested under scenarios *S1* and *S3*, and, as a consequence, stem numbers declined in these scenarios (Figure 3C). For the other two scenarios (*S2* and *S4*), stem numbers remained relatively constant for low levels of wood extrac-

tion and decreased for higher levels (Figure 3C). Stem numbers in the 60–80 cm diameter class increased for scenarios *S1* and *S2* and decreased for scenarios *S3* and *S4* (Figure 3D). When *S4* was simulated with wood extraction levels greater than 95 m³ ha⁻¹ every 10 years (9.5 m³ ha⁻¹ y⁻¹), no trees larger than 60 cm dbh remained in the forest. The strongest impact of harvesting scenarios was observed for the largest diameter class (Figure 3E) which only contained emergent trees of PFT 6 (80–100 cm dbh). In all cases, even at low levels of wood extraction and in scenarios that did not target the largest trees, stem numbers in this diameter class decreased sharply because in the long term trees were harvested before they reached the largest diameter class and hence no recruitment into this diameter class took place.

The dynamics of forest size structure is illustrated in Figure 4 for scenario *S3* when 45 m³ ha⁻¹ are harvested every 10 years. The sharp decline of trees 40–60 cm dbh due to harvesting causes a decrease of recruitment to larger dbh classes (60–80 cm, 80–100 cm), which show a corresponding delayed decline. Stem numbers in smaller dbh classes increase and higher recruitment into the 40–60 cm dbh class starts to compensate losses through harvesting after 30 years. Again, this recovery of stem numbers propagates to the larger dbh classes with a

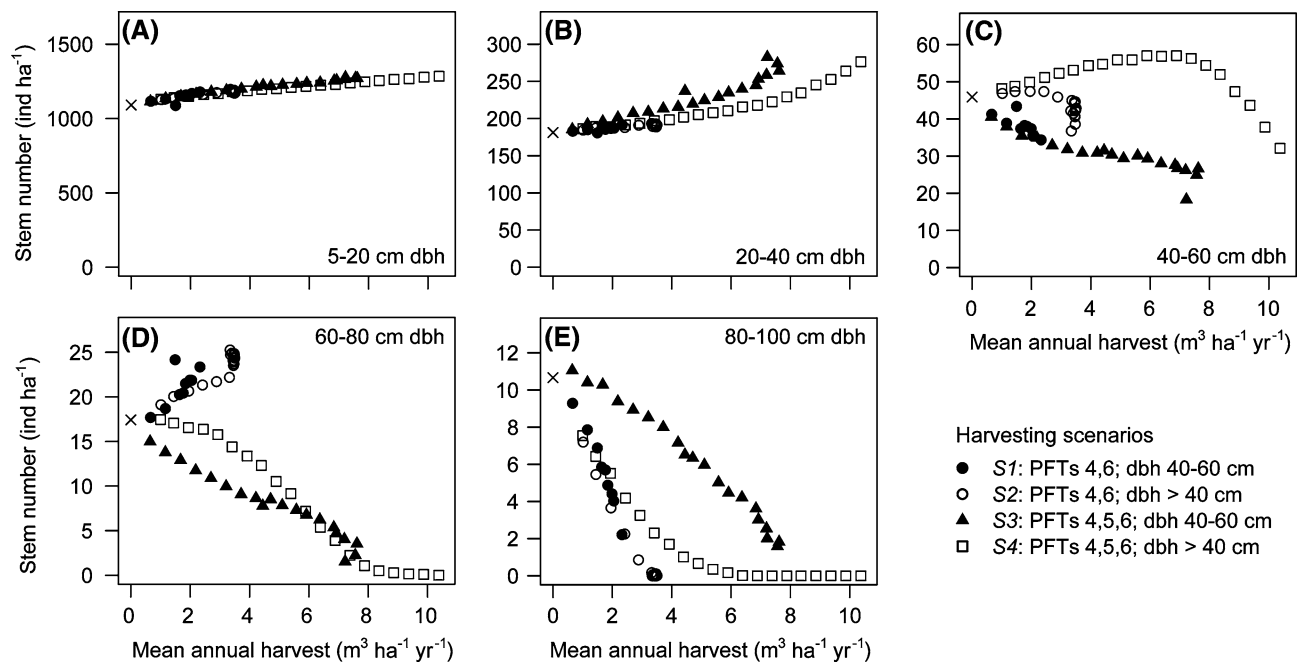


Figure 3. Mean stem numbers in five diameter classes, (A) 5–20 cm dbh, (B) 20–40 cm dbh, (C) 40–60 cm dbh, (D) 60–80 cm dbh, and (E) 80–100 cm dbh, for simulation time 400–500 for four fuelwood harvesting scenarios in TMCf in central Veracruz, Mexico. Mean values for undisturbed old-growth forest are displayed for comparison (x). Mean annual harvest represents the total amount of harvested fuelwood over 400 years of simulated harvesting converted to an annual basis.

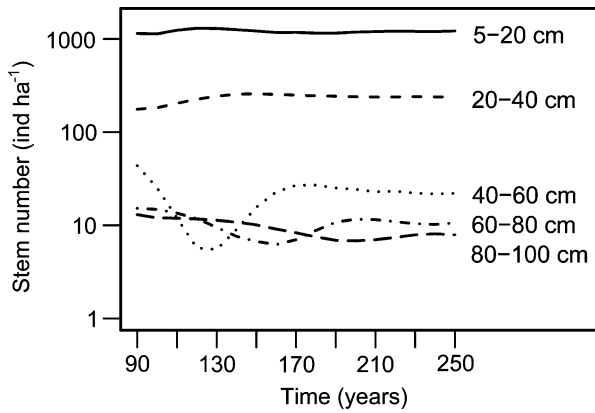


Figure 4. Mean stem numbers in five dbh classes for simulation time 90–250 for scenario *S3* (that is, harvesting PFTs 4, 5, and 6; 40–60 cm dbh) when 45 m³ ha⁻¹ are harvested every 10 years. Harvesting started in year 100.

time lag. Only after 120 years of repeated harvesting, the forest reaches a new steady state.

Changes in PFT Composition

The detailed impact of the harvesting scenarios on PFT composition and dominance can be illustrated

with importance values (*IV*; Figure 5). For all scenarios, *IV*s and thus dominance of pioneer species (PFT 1) remained low. However, mid and late successional small trees (PFTs 2 and 3) as well as mid successional canopy trees (PFT 4) generally increased in dominance with increasing wood extraction (Figure 5). In scenarios *S1* and *S2* (where PFT 5 was not logged), late successional canopy trees (PFT 5) increased in dominance with increasing wood extraction. This increase occurred mainly at the expense of emergent trees (PFT 6) which were subject to harvesting (Figure 5). Scenarios *S3* and *S4* (where also PFT 5 was logged) showed that the dominance of both late successional canopy trees and emergents (PFTs 5 and 6) was slightly reduced when harvesting was extended to all canopy and emergent trees (PFTs 4, 5, and 6).

Overall Impact on Forest Structure and Composition

For all scenarios, the index of structural change (*ISC*) indicated that changes in stem numbers in the five diameter classes increased almost linearly with increasing levels of wood extraction (Figure 6A). At high harvesting intensities, the simulated har-

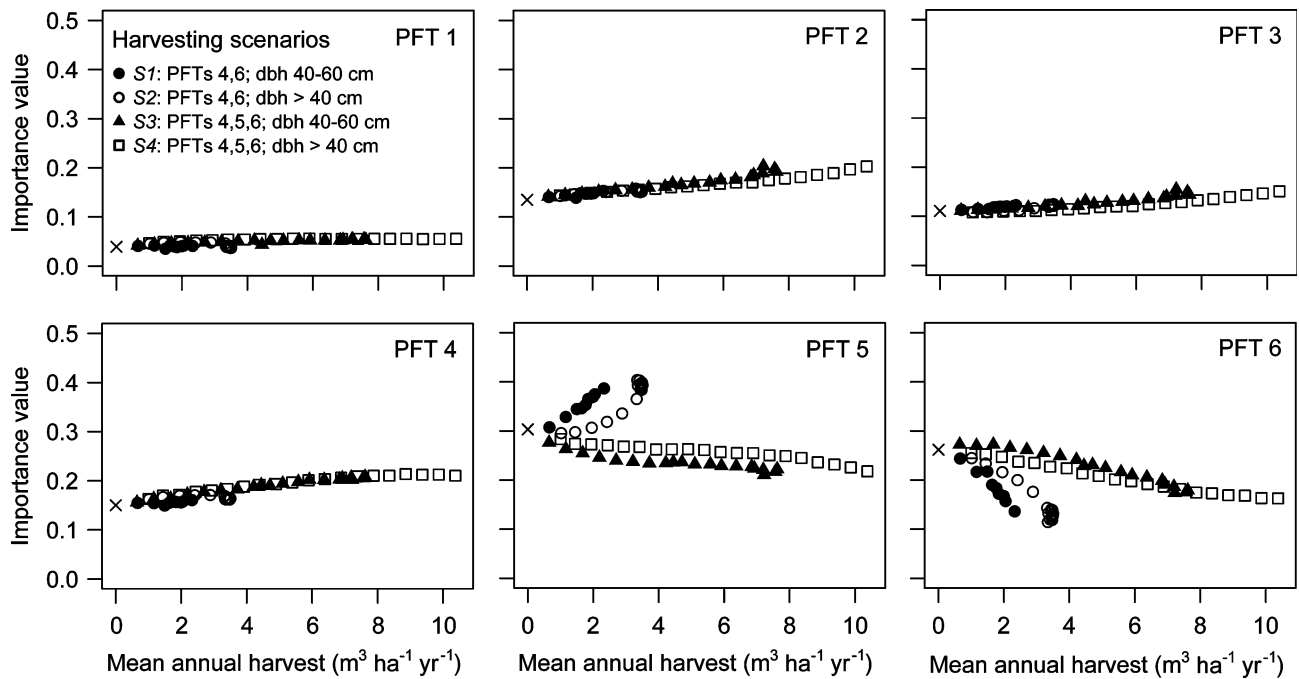


Figure 5. Mean of importance values (see section *Assessment of harvesting scenarios* for details) for simulation time 400–500 as a measure of dominance of six PFTs for four fuelwood harvesting scenarios in TCMF in central Veracruz, Mexico. See Table 1 for a description of PFTs. Importance values of undisturbed old-growth forest are displayed for comparison (×). Mean annual harvest represents the total amount of harvested fuelwood over 400 years of simulated harvesting converted to an annual basis.

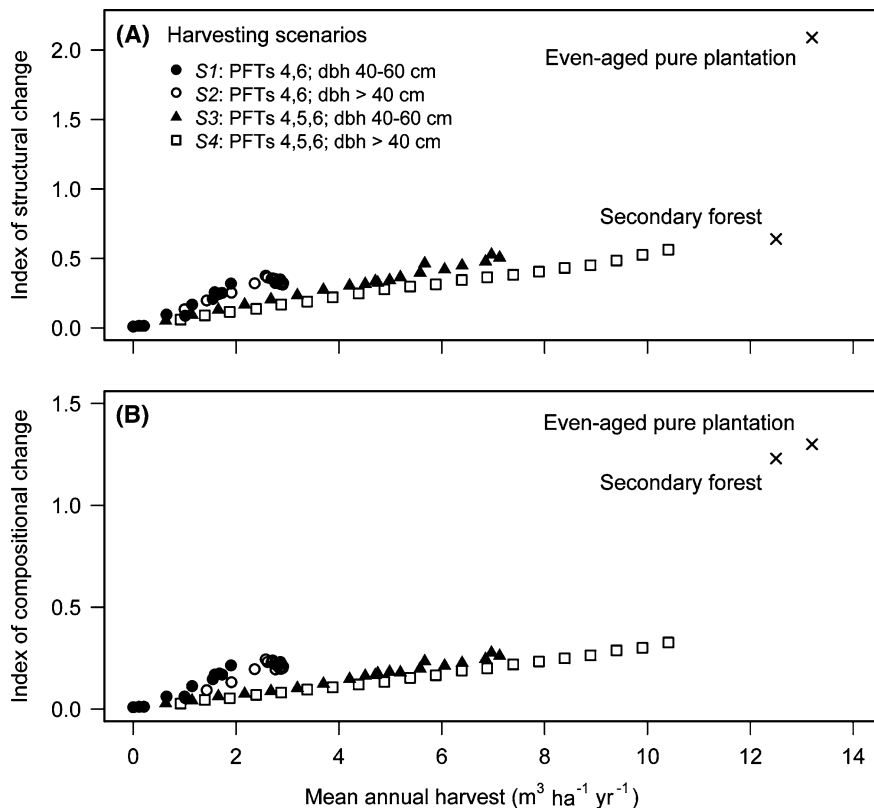


Figure 6. Indices of structural change (*ISC*, **A**) and compositional change (*ICC*, **B**) for four fuelwood harvesting scenarios in TMCF in central Veracruz, Mexico. See section *Assessment of harvesting scenarios* for details on the definition of the indices. *ISC* and *ICC* are shown in the context of other land uses (that is, intensively managed secondary forests dominated by PFTs 4 and 6 and even-aged pure plantations of PFT 6 with clear-cut rotation in 25-year cycles). Mean annual harvest represents the total amount of harvested fuelwood over 400 years of simulated harvesting converted to an annual basis.

vesting scenarios approached the structure of an intensively managed young secondary forest. In comparison, even-aged plantations had a considerably higher *ISC*. The index of compositional change (*ICC*) also increased nearly linearly with extracted wood volume for all harvesting scenarios (Figure 6B). In terms of community composition, the old-growth forests harvested for fuelwood were still much more similar to undisturbed old-growth forest than to intensively managed secondary forests or pure plantations.

LAI of the entire forest showed little variation between scenarios (range 4.3–4.7) and tended to be slightly lower than in simulated undisturbed old-growth forest (4.7). This indicated that leaf area recovered rapidly regardless of the harvesting scenario applied.

Overall, similarity to undisturbed old-growth forest (*EI*) decreased almost linearly for all harvesting scenarios with increasing harvest (Figure 7). Hence, every increase in harvest was accompanied by an increase of ecological impact. For scenarios *S1* and *S2*, the decrease was sharper than for *S3* and *S4*, due to their stronger impact on forest structure and community composition. Compared to other forms of forest management (managed young secondary forests and planta-

tions), however, the ecological impacts of fuelwood harvesting were modest (Figure 7). This was mainly due to the conservation all PFTs in the forest.

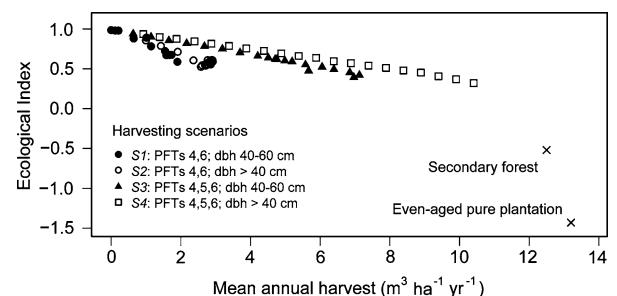


Figure 7. Ecological index (*EI*) which measures ecological similarity to simulated undisturbed old-growth forest for four fuelwood harvesting scenarios in TMCF in central Veracruz, Mexico. See section *Assessment of harvesting scenarios* for details. *EI* is shown in the context of other land uses (intensively managed secondary forests dominated by PFTs 4 and 6 and even-aged pure plantations of PFT 6 with clear-cut rotation in 25-year cycles). Mean annual harvest represents the total amount of harvested fuelwood over 400 years of simulated harvesting converted to an annual basis.

DISCUSSION

In contrast to most studies that focus on the more or less intensive commercial extraction of valuable timber species, we here studied the effects of repeated fuelwood harvesting by the local population. We used the individual-oriented forest growth model FORMIND to simulate long-term impacts of different fuelwood harvesting scenarios. Our simulation results revealed that even at low levels of harvesting forest structure became more homogeneous in the long term. With increasing levels of wood extraction, stem numbers in larger diameter classes decreased whereas they increased for smaller diameter classes. The species composition shifted to PFTs that were not harvested. In some cases, these changes can only be detected after decades or even centuries of repeated disturbance.

Implications of Fuelwood Extraction for Forest Dynamics

The most important impact of simulated repeated fuelwood harvesting on the size structure of the forest was the dramatic decline of large old trees which emerge over the main canopy of the forest up to the point of complete disappearance. Field observations in the study area confirm this result—in some forest fragments trees with a dbh larger than 60 cm are already lacking (G. Williams-Linera, pers. observation). Once these trees are lost, the sizes of gaps created either by natural treefalls or logging also decrease. New recruits of all PFTs as well as already existing advance regeneration of PFTs with intermediate or high shade tolerance could benefit from these smaller gaps. As a consequence, the relative importance of pioneer species did not increase in our simulations. This result is consistent with findings from a study of tree regeneration after selective logging in a tropical rain forest in Suriname where both densities of shade-tolerant and pioneer species increased post-logging (Dekker and de Graaf 2003). In contrast to our results, the density of pioneer species increased after commercial logging of lowland dipterocarp rain forest in South East Asia (Okuda and others 2003; Verburg and van Eijk-Bos 2003). However, in these studies harvesting intensities and associated damages were much higher than in our simulations. The smaller size of natural gaps in regenerating and logged forests compared to old-growth forests has also been reported for tropical montane rain forest in Uganda (Chapman and Chapman 1997) and for tropical lowland moist forest in Costa Rica (Nicotra and others 1999) and

Malaysia (Numata and others 2006). The latter study also identified the lack of large trees in the selectively logged forest to be responsible for smaller gap sizes.

LAI values of the simulated logged forests differed only marginally from undisturbed old-growth forest because extracted leaf area was rapidly replaced by vigorous re-growth. This finding is consistent with observations of LAI in young secondary forests which can be higher than those of old-growth forests (Hölscher and others 2003). Given that protection of soil erosion is related to the available leaf area (for example, Challenger 1998; Fleischbein and others 2005), we expect that the ability of the forest to provide this service does not decline when wood extraction does not exceed the regeneration capacity of the forest. If future studies show that LAI plays an important role in cloud water interception or runoff generation, our results could help to assess the ability of harvested forests to deliver such services.

When modeling TMCF tree recruitment we assumed that the number of saplings of the different PFTs is controlled by light availability. In cloud forest gaps in Costa Rica light availability was found to play an important role in determining densities of shade-tolerant and intolerant tree saplings (Lawton and Putz 1988). Other factors such as nutrient or soil moisture availability and the species composition of mature trees could affect seed availability, seed predation, or establishment probabilities of the different PFTs in a disturbed forest. However, these factors were not modeled explicitly here. A further assumption underlying our modeling approach is that there is only an insignificant impact of fuelwood harvesting on nutrient availability. This might be reasonable for TMCF in the study area because soils are of volcanic origin and relatively nutrient rich (Rossignol 1987).

Overall, our results demonstrate that fuelwood extraction can have severe long-term impacts on the size structure and PFT composition of trees and shrubs. Apart from the tree community, fuelwood harvesting can also have an impact on other taxa that have not been addressed by our modeling approach (for example, Putz and others 2001). There are, for instance, many recent field studies on immediate or short-term effects of selective logging on fauna and flora (for example, Costa and Magnusson 2002; Dunn 2004; Fredericksen and Fredericksen 2004; Borgella and Gavin 2005; Dumbrell and Hill 2005; Lambert and others 2005). In contrast, long-term effects over several logging cycles have rarely been examined because there are

few places where commercial logging for timber has occurred already more than once. A declining number of large old trees and the associated decrease of standing dead trees and coarse woody debris are likely to have negative long-term impacts on many specialist plant and animal species that depend on them as habitat, food sources, or nesting places (see for example, Wolf 2005 for epiphyte diversity).

Recommendations for Sustainable Fuelwood Extraction

The maximum sustainable harvest in our simulations (applying a minimum cutting diameter of 40 cm dbh) was $12 \text{ m}^3 \text{ ha}^{-1} \text{ y}^{-1}$. This rate appears to be high, but field data confirm that TMCF in central Veracruz reaches a basal area of $60 \text{ m}^2 \text{ ha}^{-1}$ within 80 years after the abandonment of cattle pasture (Muñiz-Castro and others 2006). In the model, a basal area of $60 \text{ m}^2 \text{ ha}^{-1}$ corresponds to an overall stem volume of $500\text{--}600 \text{ m}^3 \text{ ha}^{-1}$ (volume calculation is based on the dbh and wood density of each tree, see Supporting Online Material). Thus, the average rate of wood volume accumulation for these 80 years is $6\text{--}7.5 \text{ m}^3 \text{ ha}^{-1} \text{ y}^{-1}$, and as volume accumulation is nonlinear, maximum volume increment must even be higher.

In our study area, fuelwood use is not uniform across municipalities. For example, the municipality of Tlalnelhuayocan has about 9.6 fuelwood users per ha of forest (Masera and Ghilardi unpublished data). Assuming an intermediate annual fuelwood consumption of 675 kg per person (Ramírez-Bamonde 1996; Haeckel 2006), and an intermediate wood density of 0.6 g cm^{-3} , 1.1 m^3 fuelwood would be estimated to be consumed annually per person. This would result in an annual fuelwood need of about 11 m^3 per ha of forest. Additionally, Tlalnelhuayocan borders Xalapa, the capital of the state Veracruz, where large amounts of fuelwood and charcoal are consumed by bakeries, restaurants, and inhabitants. This suggests that here fuelwood extraction is already at the limit of simulated productivity of the forest and that sustainable management strategies are needed to prevent forest degradation in the long term. In four other municipalities in the study area annual fuelwood consumption was estimated to be $3.6 \text{ m}^3 \text{ ha}^{-1}$ (Masera and Ghilardi unpublished data). In these municipalities fuelwood extraction seems to be sustainable in terms of the regeneration capacity of the forest.

The ecological impact of the harvesting scenarios on the forest as measured by indices of structural

and compositional change increased linearly with increasing levels of wood extraction. Hence, every additional amount of harvested wood causes a proportional change of forest size structure and PFT composition. There seem to be no critical thresholds of harvesting intensity where the state of the ecosystem would dramatically change. Of course, this argument only holds true if no other factors were not included in our study, for example, increased fire susceptibility of logged forests (for example, Cochrane 2003).

At the low end of the gradient of harvesting intensity, the main impact is the loss of large old trees, which can be mitigated by explicit retention of some trees that are allowed to grow large and die naturally. This way the structural complexity of used forests could be increased and plant and animal species associated with large trees, senescent trees, or dead woody debris could be conserved (for example, Lindenmayer and others 2006). Harvesting scenarios with high levels of wood extraction markedly alter the size structure and PFT composition of the forest. The forest is artificially held in an intermediate stage of succession where trees are immediately cut when they exceed a dbh of 40 cm.

In cases where only species of intermediate shade tolerance are preferred for fuelwood use, it might be beneficial to cut also some large trees of shade-tolerant canopy species or apply a spatially heterogeneous disturbance regime (for example, small clear-cuts). In this way, larger gaps are created that promote regeneration of less shade-tolerant species and possibly prevent the shift in species composition as observed in the respective harvesting scenarios. Similar recommendations have been given in studies that emphasize the importance of large canopy openings for the regeneration of commercially valuable shade-intolerant tree species (for example, Fredericksen and Putz 2003, and references therein).

From an ecological perspective, a preferable alternative to fuelwood harvesting from the few remaining old-growth forests would be the sustainable management of secondary forests or the establishment of plantations of native tree species in recently degraded areas (Brown and Lugo 1990; McCrary and others 2005). In the study area, forests re-growing after abandonment of agricultural fields or pastures are often dominated by species with intermediate shade tolerance and high growth rates (for example, *C. caroliniana*, *L. styraciflua*) which could be used for plantation cultures as well (McCarter and Hughes 1984).

CONCLUSIONS

Fuelwood harvesting in natural forests will remain an important source of energy and income for the rural population in many tropical areas. Our study showed that native species-rich TMCFs have a high potential to supply fuelwood for local markets. However, simulated repeated tree felling even at low intensities changed forest structure and plant community composition at the level of PFTs in the long term. The magnitude of a combined measure of these changes increased linearly with increasing harvesting intensity. If the ecological impacts of this forest use are to be kept to a minimum, certain precautionary measures should be considered. Harvesting should be extended to all canopy species to prevent major shifts in species composition of the forest. Especially when higher harvesting intensities are applied, a number of large old trees should be maintained in the forest and allowed to grow and die naturally, as those trees provide important habitat for many specialist species. Process-based forest simulation models can support the design of management strategies for natural species-rich forests to achieve a balance between economic needs and ecological goals according to the priorities of the stakeholders. In view of apparent forest exploitation at or even above the regeneration capacity of the forest, rational management options of increasingly abundant secondary forests should be the focus of future research.

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REFERENCES

Asner GP, Broadbent EN, Oliveira PJC, Keller M, Knapp DE, Silva JNM. 2006. Condition and fate of logged forests in the Brazilian Amazon. *Proc Natl Acad Sci USA* 103:12947–50.

Borgella R, Gavin TA. 2005. Avian community dynamics in a fragmented tropical landscape. *Ecol Appl* 15:1062–73.

Broadhead J, Bahdon J, Whiteman A. 2001. Woodfuel consumption modelling and results. Annex 2 in Past trends and future prospects for the utilization of wood for Energy, Working Paper No: GFPOS/WP/05, Global Forest Products Outlook Study. Rome: FAO.

Brown S, Lugo AE. 1990. Tropical secondary forests. *J Trop Ecol* 6:1–32.

Challenger A. 1998. Utilización y conservación de los ecosistemas terrestres de México. Pasado, presente y futuro. México, D.F: CONABIO, UNAM Agrupación Sierra Madre, S.C.

Chapman CA, Chapman LJ. 1997. Forest regeneration in logged and unlogged forests of Kibale National Park, Uganda. *Biotropica* 29:396–412.

Cochrane MA. 2003. Fire science for rainforests. *Science* 421:913–9.

Costa F, Magnusson W. 2002. Selective logging effects on abundance, diversity, and composition of tropical understory herbs. *Ecol Appl* 12:807–19.

Dekker M, de Graaf NR. 2003. Pioneer and climax tree regeneration following selective logging with silviculture in Suriname. *For Ecol Manage* 172:183–90.

Ditzer T, Glauner R, Förster M, Köhler P, Huth A. 2000. The process-based stand growth model FORMIX3-Q applied in a GIS environment for growth and yield analysis in a tropical rain forest. *Tree Physiol* 20:367–81.

Dumbrell AJ, Hill JK. 2005. Impacts of selective logging on canopy and ground assemblages of tropical forest butterflies: implications for sampling. *Biol Conserv* 125:123–31.

Dunn RR. 2004. Managing the tropical landscape: a comparison of the effects of logging and forest conversion to agriculture on ants, birds, and lepidoptera. *For Ecol Manage* 191:215–24.

Echeverría C, Cayuela L, Manson RH, Coomes DA, Lara A, Rey-Benayas JM, Newton AC. 2007. Spatial and temporal patterns of forest loss and fragmentation in Mexico and Chile. In: Newton AC, Ed. Biodiversity loss and conservation in fragmented forest landscapes – The forests of montane Mexico and temperate South America. Oxford: Oxford University Press. pp 14–42.

Fimbel RA, Bennett E, Kremen C. 2001. Programs to assess timber harvesting effects on tropical forest wildlife and their natural habitat. In: Fimbel RA, Grajal A, Robinson JG, Eds. The cutting edge: conserving wildlife in logged tropical forests. New York: Columbia University Press. pp 405–22.

Fleischbein K, Wilcke W, Goller R, Boy J, Valarezo C, Zech W, Knoblich K. 2005. Rainfall interception in a lower montane forest in Ecuador: effects of canopy properties. *Hydrol Process* 19:1355–71.

Fredericksen NJ, Fredericksen TS. 2004. Impacts of selective logging on amphibians in a Bolivian tropical humid forest. *For Ecol Manage* 191:275–82.

Fredericksen TS, Putz FE. 2003. Silvicultural intensification for tropical forest conservation. *Biodivers Conserv* 12:1445–53.

Gourlet-Fleury S, Cornu G, Jéssel S, Dessard H, Jourget J-G, Blanc L, Picard N. 2005. Using models to predict recovery and assess tree species vulnerability in logged tropical forests: a case study from French Guiana. *For Ecol Manage* 209:69–86.

Haeckel I. 2006. Firewood use, supply, and harvesting impact in cloud forests of central Veracruz, Mexico. BSc Thesis, Columbia University, New York.

Holder CD. 2004. Changes in structure and cover of a common property pine forest in Guatemala, 1954–1996. *Environ Conserv* 31:22–9.

- Hölscher D, Köhler L, Leuschner C, Kappelle M. 2003. Nutrient fluxes in stemflow and throughfall in three successional stages of an upper montane rain forest in Costa Rica. *J Trop Ecol* 19:557–65.
- Huth A, Ditzer T. 2000. Simulation of the growth of a lowland Dipterocarp rain forest with FORMIX3. *Ecol Model* 134:1–25.
- Huth A, Ditzer T. 2001. Long-term impacts of logging in a tropical rain forest – a simulation study. *For Ecol Manage* 142:33–51.
- Huth A, Drechsler M, Köhler P. 2004. Multicriteria evaluation of simulated logging scenarios in a tropical rain forest. *J Environ Manage* 71:321–33.
- Huth A, Drechsler M, Köhler P. 2005. Using multicriteria decision analysis and a forest growth model to assess impacts of tree harvesting in Dipterocarp lowland rain forests. *For Ecol Manage* 207:215–32.
- International Energy Agency (IEA). 2002. Energy and poverty. World Energy Outlook 2002. Paris: OECD.
- Kammesheidt L, Köhler P, Huth A. 2001. Sustainable timber harvesting in Venezuela: a modelling approach. *J Appl Ecol* 38:756–70.
- Kammesheidt L, Köhler P, Huth A. 2002. Simulating logging scenarios in secondary forest embedded in a fragmented neotropical landscape. *For Ecol Manage* 170:89–105.
- Klooster D, Masera O. 2000. Community forest management in Mexico: carbon mitigation and biodiversity conservation through rural development. *Global Environ Change* 10:259–72.
- Köhler P. 2000. Modelling anthropogenic impacts on the growth of tropical rain forests. PhD Thesis, University of Kassel, Kassel. Osnabrück: Der Andere Verlag.
- Köhler P, Huth A. 1998. The effect of tree species grouping in tropical rain forest modelling – simulation with the individual based model FORMIND. *Ecol Model* 109:301–21.
- Köhler P, Ditzer T, Huth A. 2000. Concepts for the aggregation of tropical tree species into functional types and the application on Sabah's lowland rain forests. *J Trop Ecol* 16:591–602.
- Köhler P, Ditzer T, Ong RC, Huth A. 2001. Comparison of measured and modelled growth on permanent plots in Sabah rain forests. *For Ecol Manage* 144:101–11.
- Köhler P, Chave J, Riera B, Huth A. 2003. Simulating long-term response of tropical wet forests to fragmentation. *Ecosystems* 6:114–28.
- Lambert TD, Malcolm JR, Zimmerman BL. 2005. Effects of mahogany (*Swietenia macrophylla*) logging on small mammal communities, habitat structure, and seed predation in the southeastern Amazon Basin. *For Ecol Manage* 206:381–98.
- Lawton RO, Putz FE. 1988. Natural disturbance and gap-phase regeneration in a wind-exposed tropical cloud forest. *Ecology* 69:764–77.
- Lindenmayer DB, Franklin JF, Fischer J. 2006. General management principles and a checklist of strategies to guide forest biodiversity conservation. *Biol Conserv* 131:433–45.
- Masera OR, Guerrero G, Ghilardi A, Velázquez A, Mas JF, Ordóñez M, Drigo R, Trossero MA. 2004. Fuelwood “hot spots” in Mexico: a case study using WISDOM – Woodfuel Integrated Supply-Demand Overview Mapping. Rome: FAO, UNAM.
- McCarter PS, Hughes CE. 1984. *Liquidambar styraciflua* L. – a species of potential for the tropics. *Commonwealth Forest Rev* 63:207–16.
- McCrary JK, Walsh B, Hammett AL. 2005. Species, sources, seasonality, and sustainability of fuelwood commercialization in Masaya, Nicaragua. *For Ecol Manage* 205:299–309.
- Meijard E, Sheil D, Nasi R, Augeri D, Rosenbaum B, Iskandar D, Setyawati T, Lammertink M, Rachmatika I, Wong A, Soehartono T, O'Brien T. 2005. Life after logging: reconciling wildlife conservation and production forestry in Indonesian Borneo. Bogor, Indonesia: Center for International Forestry Research.
- Muñiz-Castro MA, Williams-Linera G, Rey-Benayas JM. 2006. Distance effect from cloud forest fragments on plant community structure in abandoned pastures in Veracruz, Mexico. *J Trop Ecol* 22:431–40.
- Nicotra AB, Chazdon RL, Iriarte SVB. 1999. Spatial heterogeneity of light and woody seedling regeneration in tropical wet forests. *Ecology* 80:1908–26.
- Numata S, Yasuda M, Okuda T, Kachi N, Nur Supardi MN. 2006. Canopy gap dynamics of two different forest stands in a Malaysian lowland rain forest. *J Trop For Sci* 18:109–16.
- Okuda T, Suzuki M, Adachi N, Quah ES, Hussein NA, Manokaran N. 2003. Effect of selective logging on canopy and stand structure and tree species composition in a lowland dipterocarp forest in peninsular Malaysia. *For Ecol Manage* 175:297–320.
- Phillips PD, de Azevedo CP, Degen B, Thompson IS, Silva JNM, van Gardingen PR. 2004. An individual-based spatially explicit simulation model for strategic forest management planning in the eastern Amazon. *Ecol Model* 173:335–54.
- Putz FE, Blate GM, Redford KH, Fimbel R, Robinson J. 2001. Tropical forest management and conservation of biodiversity: an overview. *Conserv Biol* 15:7–20.
- Ramírez-Bamonde ES. 1996. Los árboles y arbustos utilizados para leña en la comunidad de Pínohtepec, Municipio de Emiliano Zapata, Veracruz. BSc Thesis, Universidad Veracruzana, Xalapa.
- Rossignol JP. 1987. Los estudios morfoedafológicos en el área Xalapa-Coatepec, Veracruz. In: Geissert D, Rossignol JP, Eds. *La Morfoedafología en la Ordenación de los Paisajes Rurales*. Xalapa: Instituto Nacional de Investigaciones sobre Recursos Bióticos, Instituto Francés de Investigación Científica para el Desarrollo en Cooperación. pp 23–35.
- Rüger N, Gutiérrez AG, Kissling WD, Armesto JJ, Huth A. 2007. Ecological impacts of different harvesting scenarios for temperate evergreen rain forest in southern Chile – a simulation experiment. *For Ecol Manage* 252:52–66.
- SARH. 1992. Reforma de la ley forestal: estado actual de los recursos forestales en México. Mexico City: Secretaría de Agricultura y Recursos Hidráulicos.
- Shugart HH. 1984. A theory of forest dynamics. The ecological implications of forest succession models. New York: Springer-Verlag.
- Sist P, Fimbel R, Sheil D, Nasi R, Chevallier MH. 2003. Towards sustainable management of mixed dipterocarp forests of South-east Asia: moving beyond minimum diameter cutting limits. *Environ Conserv* 30:364–74.
- Smith TM, Shugart HH, Woodward FI, Eds. 1997. Plant functional types: their relevance to ecosystem properties and global change. Cambridge: Cambridge University Press.
- Sundriyal RC. 1996. Anthropogenic pressure on tree structure and biomass in the temperate forest of Mamlay watershed in Sikkim. *For Ecol Manage* 81:113–34.

- Torres-Rojo JM. 2004. Informe nacional – México. Latin American Forestry Sector Outlook Study Working Paper – ESFAL/N/02. Rome: FAO.
- Valle D, Phillips P, Vidal E, Schulze M, Grogan J, Sales M, van Gardingen P. 2007. 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. *For Ecol Manage* 243:187–98.
- van Gardingen PR, McLeish MJ, Phillips PD, Fadilah D, Tyrie G, Yasman I. 2003. Financial and ecological analysis of management options for logged-over Dipterocarp forests in Indonesian Borneo. *For Ecol Manage* 183:1–29.
- Verburg R, van Eijk-Bos C. 2003. Effects of selective logging on tree diversity, composition and plant functional type patterns in a Bornean rain forest. *J Veget Sci* 14:99–110.
- Williams-Linera G. 2002. Tree species richness complementarity, disturbance and fragmentation in a Mexican tropical montane cloud forest. *Biodivers Conserv* 11:1825–43.
- Williams-Linera G, Manson RH, Isunza-Vera E. 2002. La fragmentación del bosque mesófilo de montaña y patrones de uso del suelo en la región oeste de Xalapa, Veracruz, México. *Madera y Bosques* 8:73–89.
- Wolf JHD. 2005. The response of epiphytes to anthropogenic disturbance of pine-oak forests in the highlands of Chiapas, Mexico. *For Ecol Manage* 212:376–93.
- World Bank. 1995. Mexico resource conservation and forest sector review. Washington, DC.
- Zarin DJ, Schmink M, Alavalapati JR, Putz FE, Eds. 2004. Working forests in the neotropics: conservation through sustainable management? New York: Columbia University Press.