Winners and losers explain biodiversity response to logging!!! Muitos Caveats; estudos baseados em espécies vagabundas!!

Logging as promise for the sustainable use of tropical forests: Evidence and way forward (foco nas concessões a despeito da falha do modelo)

Introduction (low-impact logging vs. conventional logging)

A convincing body of evidence shows that as it is presently codified, sustainable forest-management (SFM) logging implemented at an industrial scale guarantees commercial and biological depletion of high-value timber species within three harvests in all three major tropical forest regions. The minimum technical standards necessary for approaching ecological sustainability directly contravene the prospects for financial profitability. Therefore, industrial-scale SFM is likely to lead to the degradation and devaluation of primary tropical forests as surely as widespread conventional unmanaged logging does today. Recent studies also show that logging in the tropics, even using SFM techniques, releases significant carbon dioxide and that carbon stocks once stored in logged timber and slash takes decades to rebuild. These results beg for a reevaluation of the United Nations Framework Convention on Climate Change proposals to apply a Reducing Emissions from Deforestation and Forest Degradation subsidy for the widespread implementation of SFM logging in tropical forests. However, encouraging models of the successful sustainable management of tropical forests for timber and nontimber products exist at local-community scales. Zimmermann & Kormos 2012.

The recent fres in southern Australia were unprecedented in scale and severity. Much commentary has rightly focused on the role of climate change in exacerbating the risk of fre. Here, we contend that policy makers must recognize that historical and contemporary logging of forests has had profound efects on these fres’ severity and frequency. Lindenmayer et al. 2020

“Logging in Muddy Waters” analyzes the boom in forest exploitation that characterized the 1990s in Cambodia, focusing on the instrumentalization of disor- der and violence as a mode of control of forest access and timber-trading channels. The article examines tensions existing between the aspirations of Cambodians for a better life, the power politics of elites, and the hope of some in the international community for a green and democratic peace. These tensions have produced both an interlocking pattern of “illegal logging” from the highest levels of the state to self-demobilized soldiers and peasants and sustained criticism that was only temporarily resolved through a legalization of the forest sector that benefited large-scal e companies to the prejudice of the poor. Le Bilon 2002

Does logging affect the fire proneness of forests? This question often arises after major wildfires, but data suggest that answers differ substantially among different types of forest. Logging can alter key attributes of forests by changing microclimates, stand structure and species composition, fuel characteristics, the prevalence of ignition points, and patterns of landscape cover. Lindenmayer et al 2009

Tropical forests, ecosystem services and global sustainability

Logging: geographic coverage, economic and social relevance

Logging: impacts at multiple levels of ecological organization

Logging: knowledge gaps and scientometrics

Paper objectives, hypotheses and potential contribution

Population-level effects

Number and taxonomically profile of tree species exploited commercially; patterns of species rarity and potential populational collapse; changes on market offer as proxy for collapse

Changes on forest structure with cascades on tree community level and patterns of ecosystem functioning

Changes on forest cover, biomass, and carbon (i.e. the collapse of the forest emergent layer and the big tree stands)

Up to 0.5 km/100 ha of primary surfaced roads are built to provide truck access. All cut boles are dragged to these roads. Roads are 5 m wide, and at least 20 m on either side is deforested. Within the 50 1-ha sample quadrats of recently logged Figure 1. Distribution of logging tracks and gaps (unshaded) and intact forest (dotted) actually mapped on a representative area, one year after log- ging. The hatched squares are the 50 x 50 m bird sample plots. They give the scale. forest, the vegetation at ground level was almost com- pletely destroyed or heavily damaged on an average of 38% of the area (extremes = 15-62%), divided into 30% (10-50% ) for tree fall gaps and associated distur- bances and 8% (5-12% ) for tracks between them. The damaged areas were either devoid of any standing trees or were covered with a thick layer of slash. When mod- erately disturbed forest and logging roads were in- cluded, the proportion of forest area substantially af- fected on the 336 0.25-ha sample plots amounted to 51.4 + 30.3% of the understory under 10 m and 63.2 + 37.8% of the canopy above 20 m. Similar levels of dam- age (33-54% of the total area) were measured by for- esters on nearby experimental sites under two different logging intensities (Schmitt & Bariteau 1990). The mean gap size was markedly larger in logged than in primary forest because artificial gaps all came from the largest, fully crowned trees and were often enlarged by dragging maneuvers, whereas many natural gaps were created by the fall of tree parts or dead trunks. For each tree felled on the 1-ha samples, on average 57 (range = 14-76) smaller trees were killed or damaged. Thiolly et al. 1992.

Microclimatic changes and forest response (pioneer vs. shade tolerant species)

Bushmeat

Biological invasion and the proliferation of winner-plant species

Logging, fire and climate changes

Regulations and the way forward

Current regulations in the light of potential impacts

Required best practices, economic viability and ecological sustainability

Sustainable logging is not able to curb both deforestation or illegal logging; Little to show for our efforts? After nearly two decades and hundreds of millions of dol- lars in private and public investment, only a tiny percentage of tropical forests are man- aged sustainably (Bowles et al. 1998). Sustainable/managed logging as tool for: (1) disciplinate timber market, (2) curb deforestation and forest degradation, (3) protect biodiversity and share logging benefits among locals.

Different logging scenarios for an initially undisturbed forest stand at Deramakot (Malaysia) were simulated. Two different logging methods (conventional and low impact logging) in combination with different cutting cycles (from 20 to 100 years) were investigated for their long-term impact on the forest. We characterize the impacts with four indicators: total yield, yield per cut, species composition, and canopy opening. Our simulation results indicate that too short logging cycles (<40 years) overuse this forest. The corresponding yields are low and the species composition is highly disturbed. For conservation of species composition and minimizing erosion risk, logging cycles of 80 or 100 years in combination with low impact methods are recommended (Huth & Ditzer 2001).

Harvesting practice in the highly productive Bornean forests may exceed 10 trees ha)1 , damaging more than 50% of the original stand (Sist et al., 2003).

Selective logging resulted in a significant decrease in canopy cover and an increase in the density of understorey vegetation. As logged plots grew older, canopy cover increased, whereas understorey stem density decreased. Plots logged in 1993 (4 years old) had, on average, 49.3% of their surface areas occupied by gaps, whereas those logged in 1987 (10 years old) had only 27%. The average surface area occupied by gaps in the control plots was 15.2%. Vasconcelos et al 2000.

Although this process generally involves the removal of only a small proportion of trees (c. 14 trees ha-' in Malaysia; Whitmore 1984), associated damage levels are often severe (between 62% and 80% of trees greater than 30 cm girth were lost as a result of selective logging in Sabah, East Malaysia; Lambert 1992).

Some conservation scientists propose combining protected areas with natural forest timber concessions to sustain larger forest landscapes than otherwise possible via protected areas alone [3,7,8,9,10,11,12] Gaveau et al. 2013

Natural forest timber concessions are parcels of natural forest leased out to companies or to communities to harvest timber on a long term basis. When natural forest timber concessions are additional to more strictly protected areas they bring an opportunity to maintain larger and better connected forest landscapes with a greater capacity to maintain low density, large range and high mobility species [16]. Indeed, timber concessions are de facto a kind of protected area in most tropical countries, as also indicated by their inclusion in the IUCN protected area categories (as Category VI). Gaveau et al. 2013.

Not only does selective logging maintain a forest structure, a recent global meta-analysis of .100 scientific studies concluded that timber extraction in tropical forests has relatively benign impacts on biodiversity, because 85–100% of mammal, bird, invertebrate, and plant species richness remains in forests that have been harvested once [17]. Thus, a logged tropical forest can remain a biologically rich forest [12]. Gaveau et al. 2013

Not everyone is convinced that natural forest timber concessions should play a major role in tropical forest conservation [20]. Many equate timber harvesting (logging) with forest destruction and loggers with forest destroyers [21,22]. Many concerns relate to the apparently increased likelihood of a forest harvested for timber being further degraded by wildfires or converted to agriculture. Harvested forests appear to have increased vulnerability to fire [23,24,25]. Some governments equate ‘logged forests’ with ‘degraded lands’’ or ‘‘wastelands’, and reclassify these forests for conversion to industrial crops such as oil palm [26]. Gaveau et al. 2013

Prim ary forests throughout Asia are rapidly being logged. In M alaysia, of an estim ated 19.8 million hectares (ha) f of forest rem aining in 1986, 14.8 million ha (74.7%) were reserved as production forest (Nectoux & K uroda 1989), and subjected to selective logging, which, in theory, is repeatable on a 25-40- year cycle. However, logging practices in East M alaysia are such that it is questionable w hether the process is sustainable (Nectoux & K uroda 1989; Johns 1989£). Lambert 1992

A comparison between unlogged, 6-month and 18-year post-harvest forest stands indicates lasting effects of highly selective, high grade logging. While there was little difference in tree species composition and diversity between treatments, stem densities of both saplings and trees in unlogged forest were significantly higher than those in forest sampled 18 years after logging. Evidence suggests inadequate recruitment of Entandrophragma cylindricum and E. utile, the principal timber species, to justify continued timber extraction. Data indicate a significant shift in canopy dominance from shade intolerant to shade bearing species due to insufficient canopy disturbance. Nevertheless, an abundance of other top quality timber species remains after selective removal of African mahogany and these forests will remain attractive to loggers long after the elimination of Entandrophragma spp. Hall et al 2003

ices. It is concluded that these two primates and C. guereza have benefited from logging in Budongo and that logging has had little effect on the other two primates. Plumptre & Raynolds 1993

The length of logging roads surveyed in C3 was 690 m, or 53 m ha1. For an average total road width(including contributing portions of the cutslopes and fillslopes) of 6 m, this represents 32.% of the catchment. Approximately 2300 m of skid trails branched from the logging road in C3, representing 177 m ha1. For an average skid trial width of 3Ð7 m, this disturbance comprises 6Ð5% of C3. Fourlandings for storing logs in C3 accounted for a total area of 1970 m2 or1.5% of the catchment. The extent of road, skid trail and landing disturbances in C3 was almost twice that estimated in a small selectively logged catchment more than 100 km north of our site (Baharuddin and Abdul Rahim, 1994). Road and skidtrail disturbances in C3 were similar to those measured in a larger study area in Sabah, Malaysia, that wasselectively logged using conventional techniques (e.g. crawler tractors) (Pinardet al., 2000). However, Pinardet al. (2000) found that skid trail disturbances during conventional logging covered an average of 11Ð9%of the area, which is almost twice the skid trail disturbance in C3. Other studies in Malaysian forests have revealed even higher levels (15 to 40% of the area) of tractor skid trails (Chai, 1975; Jusoff, 1991). Sidle et al 2004.

Soil loss rates of 272 š 20 t ha1 year1 from a logging road and 275 š 20 t ha1 year1 from skid trails were estimated from erosion features on and adjacent to these corridors. Approximately 60% of the soil loss from the logging road was generated from the running surface (rills, gullies and sheet wash). In comparison, 22% and 18% of the losses emanated respectively from erosion of disturbed cutslope and fillslope material (discharging onto the surface of the road). Of the total soil loss from the 690 m logging road and landings (1970 m2), an estimated 78% reached the stream system during the 16 month period after initial road construction and logging.

An average of six trees were harvested per hectare in logging operations (n = 3 study areas) and the volume yield averaged 38 m 3 ha- 1. Damages to the forest during logging are significant. Twenty-seven trees greater than or equal to 10 cm dbh are severely damaged for each tree harvested. This damage occurs in the opening of approximately 40 m of logging road and 600 m z of canopy per tree harvested. Vines are favored by these open conditions and forest fires are possible . Verissimo et al. 1992.

Our findings suggest that responsible reduced-impact logging practices in neotropical forests can be considered as a relatively benign form of forest management that can coexist with the requirements of both local economies and biodiversity conservation. However, our study sites experienced comparably low extraction rates, and detrimental effects such as hunting were low. Our results therefore provide an opportunity to scrutinise the effects of best practice logging systems, though do not necessarily represent typical circumstances across tropical forests. Bicknell & Peres 2010

Between the first and second remeasures, average diameter increment decreased from 0.4 to 0.2 cm year-‘, mortality remained relatively constant at 2.5Oh year-‘, while recruitment (at 5 cm diameter at breast height) decreased from 5 to 2W. Total volume production declined from approximately 6 to 4 m3 ha-’ year-‘, while commercial production remained about 0.8 m3 ha-’ year-‘. New commercial species increased the commercial volume in 1992 from 18 to 54 m3 ha-‘, and the increment to 1.8 m3 ha-’ year-‘. Results from this experiment provide the first quantitative information for management planning in the Tapajb Forest, and may guide the choice of cutting cycle and annual allowable cut. Silvicultural treatment to stimulate growth rates in forest areas zoned for timber production should be considered as a viable management option. Extrapolations of these results to an anticipated 30-35 year cutting cycle must be interpreted with caution. Ongoing remeasurement and analysis of these and other plots over the next 30 years or more are necessary to provide a stronger basis for management inferences. Silva et al. 1995 .

. In addition, increased amounts of coarse woody debris and greatly reduced canopy cover in both heavily logged and heavily burned forests have increased the risk of future fires that could further degraded these forests. As more and more logging frontiers in the eastern Brazilian Amazon mature it is likely that repeated logging of previously logged stands will become more common. This combined with the increased flammability of previously burned stands is likely to result in the spread of forest degradation unless incentives can be developed to encourage the use of low impact logging techniques and the development of cost effective fire prevention methods. Gerwing 2002.

It was estimated that logging disturbed 45.8% of the stand including 25% ground area disturbance in the form of skid trails, logging roads, and log landings and an additional 25% in canopy openings due to tree felling. On average, 44 trees were damaged for every tree extracted including 22 trees killed or severely damaged, 6 of them commercial species. The most common types of damage included uprooted stems, stem wounds to the cambial layer, and bark scrapes. Jackson et al 2002

Extensive tracts of tropical rain forest were burnt in Borneo during the El Niiio drought of 1983. Severe droughts have occurred previously but without causing such extensive fires. This extensive burning is a result of forests becoming more fire-prone after disturbance by logging. Rates of tree mortality after drought and fire ranged from 38 to 94 percent in logged forests and from 19 to 71 percent in unlogged forests, while for saplings rates of mortality exceeded 80 percent in both forest types. Secondary succession after logging was truncated by fire with the result that the post-fire condition of a forest logged six years before fire was similar to that of a forest logged two years before fire. The impact of fire differs from other natural or man-made gap forming processes in that most pre-existing seedlings and saplings are killed by fire. This left the regrowth in burnt forest depauperate in species diversity and in regeneration of upper-canopy species. Logged and burnt forests suffered severe canopy loss and the ground cover was dominated by grasses (e.g., Imperata cylindrica) or woody creepers (e.g., Eupatorium odoratum). In burnt primary forest, however, canopy loss was less severe and there was a low density of grasses. The prospects for recovery of forest structure appear to be good in burnt primary forest although species composition may be permanently altered. In forests logged prior to the fire, however, prospects for recovery of forest structure are not good, especially if further burning occurs. The recovery of these forests depends heavily on the ability of the secondary tree species to shade out the vigorous grasses, the continued presence of which may herald the conversion of the forest to unproductive grassland as has occurred widely elsewhere in the tropics after over-intensive shifiting cultivation. Woods 1989

Our analysis shows that species traits, such as feeding group and body mass, and logging practices, such as time since logging and logging intensity, interact to influence a species’ response to logging. Frugivores and insectivores were most adversely affected by logging and declined further with increasing logging intensity. Nectarivores and granivores responded positively to selective logging for the first two decades, after which their abundances decrease below pre-logging levels. Larger species of omnivores and granivores responded more positively to selective logging than smaller species from either feeding group, whereas this effect of body size was reversed for carnivores, herbivores, frugivores and insectivores. Most importantly, species most negatively impacted by selective logging had not recovered approximately 40 years after logging cessation. We conclude that selective timber harvest has the potential to cause large and long-lasting changes in avian biodiversity. However, our results suggest that the impacts can be mitigated to a certain extent through specific forest management strategies such as lengthening the rotation cycle and implementing reduced impact logging. Burivalova et al. 2015

We conclude that at a logged forest site in close proximity to primary forest, low intensities of logging do not necessarily reduce the species richness or abundance of butterflies, although assemblage composition is changed. Willott et al 1999

Among the 256 species recorded, overall bird species richness and abundance were depressed by 27±34 % in the logged areas compared to primary forest. The most vulnerable guilds, which decreased by 37±98 % in abundance, were mature forest understorey species, especially terrestrial ones and mixed ¯ocks of insectivores. Hummingbirds, small gaps, vine tangles and canopy species did not decrease, nor increase signi®cantly after logging. Only species naturally associated with dense second growth, forest edges and large gaps actually increased. Habitat specialization was the major determinant of vulnerability to logging, and, to a lesser degree, size (large) and diet (insectivorous), but foraging behaviour and rarity had little eect. Bird sensitivity to changes in logged forest structure may involve physiological intolerance, reduced food categories, increased exposure to predators, too dense understorey for their speci®c foraging behaviour and/or avoidance of gaps. Suggested improvements of current forest management and logging techniques for the maintenance of a higher proportion of the original biodiversity include minimizing logging damages, long rotations (>50 years) between cuts, and keeping unlogged forest patches within logging concessions. Thiollay 1997

Multivariate analyses indicated that understory bat species composition differed strongly between logged and unlogged sites but provided little evidence of logging effects for the canopy fauna. Different responses among feeding guilds and taxonomic groups appeared to be related to foraging and echolocation strategies and to changes in canopy cover and understory foliage densities. Our results suggest that even low-intensity logging modifies habitat structure, leading to changes in bat species composition. Peters et al 2006

Four specific mechanisms were identified; conflict of use and the indirect impacts of logging being those most commonly implicated in negative effects on livelihood-relevant NTFPs. Eighty two percent of reviewed articles highlighted negative impacts on NTFP availability. Examples of positive impacts were restricted to light demanding species that respond to the opening of forest structure and typically represent a small subset of those of livelihood value. Despite considerable impacts on livelihoods, in all three case studies we found evidence to support the potential for enhanced compatibility between timber extraction and the subsistence use of NTFPs. Drawing on this evidence, and findings from our review, we make specific recommendations for research, policy and management implementation. These findings have significant implications for reconciling timber and non-timber uses of tropical forests. Rist et al 2012

Logging also effected striking, long-term changes in both species and functional composition. In particular, the xylem density of recruits in logging gaps was 6% less than in unlogged forests, leaves were 11% less tough and had 6–13% greater mineral nutrient concentrations. 5. Synthesis and applications. Our results suggest that managers of tropical forests should limit overall surface area converted to logging gaps by creating fewer, larger gaps during selective logging, to reduce impacts on the taxonomic and functional composition of the regenerating stand. Baraloto et al 2012

During the post-logging period (8 years), the mortality of large trees was found to drive the annual net changes and largely overcame the AGB gain in the smaller DBH classes. Indeed, plots with high post-logging mortality of large trees showed negative carbon balance t over the study period (8 years). The over mortality of large trees injured by logging contributed significantly to the annual AGB losses (up to 40%) in the first years after logging. Due to the overwhelming importance of this size class in carbon stocks and dynamic, reducing logging damages and intensity might have great impact in the post-logging biomass dynamics. We estimated that reducing logging intensity from 6 to 3 stems ha1 would save 27.7 Mg C ha1 for a 35 years rotation cycle. To compensate this loss of profits, compensatory payments of avoided CO2 emission should worth US $ 6.5/Mg of CO2. This price falls into the range of prices of the international carbon market. Sustainable forest management aiming at enhancing carbon stocks could therefore promote the preservation of the large trees. At our study site, we recommend the adoption of a maximum diameter cutting limit of 110 cm. 2

. Logging of the extent described above slightly im- pairs habitats for E. telfairi but has no effect on T. ecau- datus. Though the latter species is hunted extensively in the logged areas, logging per se does not threaten the survival of either species. Ghanzhorn et al 1990.

In simulations with moderate logging, typical for French Guiana, with large cutting diameter (>60 cm diameter) and long cutting cycles (65 years), the two species V. americana and Sextonia rubra were not able to recover their initial stock at the end of the rotation period, with a large decrease in the number of individuals and in basal area. Under a more intensive logging system (cutting diameter >45 cm diameter, cutting cycles of 30 years) that is common practice in the Brazilian Amazon, only Symphonia globulifera showed no negative impact. Generally, the differences between the genetic parameters in the control scenarios without logging and the logging scenarios were surprisingly small. The main reasons for this were the overlapping of generations and the effective dispersal ability of gene vectors in all species, which guarantee relative homogeneity of the genetic structure in different age classes. Nevertheless, decreasing the population size by logging reduced the number of genotypes and caused higher genetic distances between the original population and the population at the end of the logging cycles. Sensitivity analysis showed that genetic changes in the logging scenarios were principally determined by the growth, densities and cutting diameter of each species, and only to a very small extent by the reproductive system including factors such as pollen and seed dispersal and flowering phenology Degen et al 2006

We show that the effects of selective logging are greater than those expected simply from the removal of commercial species, and can persist for decades. Selective logging, unless it is practiced at very low harvest intensities, can significantly reduce the biomass of a tropical forest for many decades, seriously diminishing aboveground carbon storage capacity, and create opportunities for weeds and vines to spread and slow down the ecological succession. Gatti et al. 2014.

Amazon deforestation contributes significantly to global carbon (C) emissions. In comparison, the contribution from selective logging to atmospheric CO2 emissions, and its impact on regional C dynamics, is highly uncertain. Using a new geographically based modeling approach in combination with high resolution remote sensing data from 1999 to 2002, we estimate that C emissions were 0.04–0.05 Pg C yr−1 due to selective logging from a ∼2,664,960 km2 region of the Brazilian Amazon. Selective logging was responsible for 15–19% higher carbon emissions than reported from deforestation (clear‐cutting) alone. Our simulations indicated that forest carbon lost via selective logging lasts two to three decades following harvest, and that the original live biomass takes up to a century to recover, if the forests are not subsequently cleared. The two‐to three‐decade loss of carbon results from the biomass damaged by logging activities, including leaves, wood, and roots, estimated to be 89.1 Tg C yr−1 from 1999 to 2002 over the study region, leaving 70.0 Tg C yr−1 and 7.9 Tg C yr−1 to accumulate as coarse woody debris and soil C, respectively. While avoided deforestation is central to crediting rain forest nations for reduced carbon emissions, the extent and intensity of selective logging are also critical to determining carbon emissions in the context of Reduced Emissions from Deforestation and Forest Degradation (REDD). We show that a combination of automated high‐resolution satellite monitoring and detailed forest C modeling can yield spatially explicit estimates of harvest‐related C losses and subsequent recovery in support of REDD and other international carbon market mechanisms. Huang & Asner 2010

All species occurred in disturbed forest, but the overall density of civets in logged forest (6.4 individuals per sq km) was found to be significantly lower than in primary forest (31.5 individuals per sq km). This reflected a marked reduction in the abundance of civets from the predominantly carnivorous subfamilies Viverrinae and Hemigalinae. Heydon & Bulloh 1996

We therefore conclude that selectively logged forests are similar to primary forests in their potential for thermal buffering, and subsequent ability to retain temperature-sensitive species under climate change. Selectively logged forests can play a crucial role in the long-term maintenance of global biodiversity. Senior et al 2017

Depending on local logging intensity, height reductions resulted in 2–13 percent decreases in aboveground tree biomass and 3–17 percent decreases in bole volume. These results highlight the adverse effects of logging at both tree and stand levels. This decrease in height is a further threat to future provision of key environmental services, such as timber production and carbon sequestration. Rutishauser et al 2016.

The selective logging that characterizes most timber extraction operations in the tropics leaves large patches of logging blocks (i.e., areas allocated for harvesting) intact, without evidence of direct impacts. For example, in ∼10,000 ha sampled in 48 forest management enterprises in Africa (Gabon, Republic of Congo, and the Democratic Republic of Congo), Indonesia, Suriname, and Mexico, an average of 69% (range 20– 97%) of the area in logging blocks was not directly affected by timber harvests. The proportion of intact forest within logging blocks decreased very slightly with increases in harvest intensity in the accessed portion of the logging blocks (9–86 m3 ha−1 ) but decreased strongly with harvest intensity in entire logging blocks (0.3–48.2 m3 ha−1 ). More forest was left intact in areas farther from roads, on slopes >40%, and within 25 m of perennial streams, but the effect sizes of each of these variables was small (∼8%). It is less clear how much of the intact forest left after one harvest will remain intact through the next. Conservation benefits without reductions in timber yields will derive from better management planning so that sensitive and ecologically critical areas, such as steep slopes and riparian buffers, constitute large and permanent proportions of the intact forest in selectively logged landscapes in the tropics. Putz et al 2019

.. A 10 year rainfall event of 167 mm occurred during the monitoring period and triggered a debris slide and several log-culvert collapses along the area’s main timber haulage road. The sampling design captured this event’s dynamics and allowed lumped catchment response to be traced to the new landforms. During the 1 day period of the 10 year event, some 33 t of suspended sediment were transported from one debris slide, comprising a significant proportion of the 105 t discharging from the whole catchment, which itself constituted 40% of the annual yield of 592 t km2 . The contributory areas with only ephemeral waterflows, including former haulage roads and tracks, generated relatively little sediment during this 10 year event or in other storms. This work suggests that though some sediment sources recover from the impacts of forest road construction and harvesting, collapse of roadfill materials or more local log-culvert failure persists for several years after harvesting. Sustainable forestry guidelines that do not focus on ameliorating these persistent instabilities may not significantly mitigate the geomorphic impacts of conventional, selective harvesting. Chappel et al 2004

Three major conclusions can be drawn from our results. 1. Scale matters: it is important to monitor different levels of biodiversity in order to reveal its actual loss after anthropogenic disturbance. 2. Time matters: the disturbance history of a site is important in order to detect patterns that otherwise remain unnoticed. 3. Geographic history matters at the local scale: whereas general patterns at higher diversity levels were identical in both eco-regions, species richness, species diversity and turnover patterns differed. Ernest et al 2006.

Selective logging is a major contributor to the social, economic, and ecological dynamics of Brazilian Amazonia. Logging activities have expanded from lowvolume floodplain harvests in past centuries to high-volume operations today that take about 25 million m3 of wood from the forest each year. The most common highimpact conventional and often illegal logging practices result in major collateral forest damage, with cascading effects on ecosystem processes. Initial carbon losses and forest recovery rates following timber harvest are tightly linked to initial logging intensity, which drives changes in forest gap fraction, fragmentation, and the light environment. Other ecological processes affected by selective logging include nutrient cycling, hydrological function, and postharvest disturbance such as fire. This chapter synthesizes the ecological impacts of selective logging, in the context of the recent socioeconomic conditions throughout Brazilian Amazonia, as determined from field-based and remote sensing studies carried out during the Large-Scale Biosphere-Atmosphere Experiment in Amazonia program. Asner et al 2009

These results indicate that the distributional pattern of species abundance of tropical butterflies may be used as an indicator of forest disturbance, and that selective logging of tropical forests in SE Asia may be associated with a significant decrease in biodiversity of butterflies, at least during the first 5 years of forest regeneration. Hill et al. 1995

These results demonstrate that even with low-intensity logging (≤3 trees/ha) a minimum 20-year rotation of logging is required for effective conservation of amphibian assemblages in moist semideciduous forests. Furthermore, remnant patches of intact forests retained in the landscape and the presence of permanent brooks may aid in the effective recovery of amphibian assemblages. Adum et al. 2019.

The proportion of stems damaged by logging is a key parameter for the management of natural productive forests in the three tropical continents (Africa, Neotropics, and southeastern Asia). Based on a review of the literature and on a meta-analysis of published data, we estimated this logging damage rate for conventional logging and compared it across continents. Scaling coefficients were estimated to convert damage rate and logging intensity from one unit to another. Felled trees were smaller in the Neotropics (61 cm diameter at breast height (dbh) on average) than in Africa or Asia (92 cm dbh). A pantropical equation relating the proportion of trees damaged (a, no unit) to logging intensity (Nlog, in ha–1) was fitted: a = 1 – (1 + 0.09135Nlog) –0.70461. A significant residual continent effect was found, with lower damage in the Neotropics than in Africa or Asia for the same level of logging intensity, in agreement with the size of felled trees. The damage rate varied with the size of damaged trees and divided equally between destroyed and injured trees, with injured trees experiencing a threefold mortality rate during 5–10 years. Taking account at least of the relationship between logging damage and logging intensity would improve the accuracy of forecasts in forest management. Picard et al. 2012

For the first time, we present evidence of logging-induced physiological stress responses in tropical invertebrates. Selective logging increased the individual levels of fat storage and reduced the relative abundance of two dung beetle species. França et al 2016

In a diverse tropical songbird community, we found that logging‐induced shifts in occupancy and abundance were positively associated with adjustments in per‐capita song and per‐pair duet rates. Importantly, we demonstrate that species with known preferences for undisturbed habitats were most negatively affected by logging in both population distributions and singing behaviours. Conversely, species known to exploit degraded habitats not only thrived in logged forest but also produced more songs per‐capita and duets per‐pair. Pillay et al 2018

Logged forests showed heavier liana loads disparately affecting slowgrowing tree species, which could exacerbate the loss of timber value and carbon storage already associated with logging. Moreover, simulation scenarios of host tree local species loss indicated that logging might decrease the robustness of liana– tree interaction networks if heavily infested trees (i.e. the most connected ones) were more likely to disappear. This effect is partially mitigated in the short term by the colonization of host trees by a greater diversity of liana species within logged forests, yet this might not compensate for the loss of preferred tree hosts in the long term. As a consequence, species interaction networks may show a lagged response to disturbance, which may trigger sudden collapses in species richness and ecosystem function in response to additional disturbances, representing a new type of ‘extinction debt’. Magrach et al. 2016

The study was conducted in Democracia Project, a forest management operation in Amazonas, Brazil. The conventional approach failed to retain any seed trees at the 100 ha block scale for 7 of 37 commercial species, whereas the alternative approach retained a minimum number of seed trees per 100 ha block for all commercial species. The conventional approach resulted in the retention of relatively high proportions of potential seed trees for common species (e.g., 22% for Eperua oleifera and 36% for Maquira sclerophila) that are shade bearers and recruit readily at the site; alternately, for species with constraints to regeneration, it retained relatively low proportions (e.g., 2% for Dinizia excelsa and Hymenolobium nitidum). The alternative approach effectively retained lower proportions of common species (e.g., 10% for E. oleifera and 13% for M. sclerophila) and relatively high proportions of species with regeneration constraints (e.g., 20% for D. excelsa and 16% for H. nitidum). Our study demonstrates that it is feasible to implement at an operational scale, species-specific retention rules that take into account local abundance when inventory data are digitised and spatially explicit. Monitoring regeneration in the residual stands over time will provide the evidence to assess the ecological benefits of the adoption of our alternative approach. Freitas & Pinard 2008

Fragmentation increased decomposition and reduced antbird predation, while selective logging consistently increased pollination, seed dispersal and army-ant raiding. Fragmentation modified species richness or community composition of five taxa, whereas selective logging did not affect any component of biodiversity. Changes in the abundance of functionally important species were related to lower predation by antbirds and higher decomposition rates in small forest fragments. The positive effects of selective logging on bee pollination, bird seed dispersal and army-ant raiding were direct, i.e. not related to changes in biodiversity, and were probably due to behavioural changes of these highly mobile animal taxa. We conclude that animalmediated ecosystem processes respond in distinct ways to different types of human disturbance in Kakamega Forest. Our findings suggest that forest fragmentation affects ecosystem processes indirectly by changes in biodiversity, whereas selective logging influences processes directly by modifying local environmental conditions and resource distributions. The positive to neutral effects of selective logging on ecosystem processes show that the functionality of tropical forests can be maintained in moderately disturbed forest fragments. Conservation concepts for tropical forests should thus include not only remaining pristine forests but also functionally viable forest remnants. Schleuning et al. 2011.

In terms of species composition, old-growth stands were more similar to selectively-logged stands than to second-growth stands, but across stands, selectively-logged forests were most distinct from the other two forest types. An inventory of the standing woody vegetation in each site showed little representation of the woody taxa found in the seed bank. We discuss these results in the context of the main factors that have been postulated to influence the abundance, life form, and species composition of tropical forest seed banks, and explore the role of the latter during intermediate phases of tropical forest succession and regeneration. Dupuy & Chazdon 1998.

The percentage of tree damage increased in line with the increasing felling intensity at both locations. It was concluded that a higher tree density and felling intensity resulted in a greater residual stand damage both in Central Kalimantan and West Sumatra. Matangaran et al 2019.

At the community level, post-logging change in population abundance was not associated with reduced body size, although between species variation suggests that adverse environmental conditions and different coping strategies underlie body size reductions in logged forest. Our study suggests that body size is a valuable metric to assess how logging impacts forest birds, pointing towards potential functional consequences related to seed dispersal within logged forests and need for improved silvicultural practices. Messina et al 2021

Life Cycle carbon emissions from the management of natural tropical forests for wood production in Costa Rica were 5.09 Mg C ha−1 15 year−1 and were dominated by the damage from harvesting operations (Figure 4). Logging damage was responsible for 80% of all carbon lost, followed by SWDS (9%), pellets (4%), stables, stalls, and nurseries (4%), and fuelwood (3%). However, an important part of the ecosystem carbon (i.e., 3.08 Mg C ha−1) was transferred across pools and remained stored along the system after the 15-year period. Anthropogenic reservoirs hold 58% of carbon, especially SWDS (40%). The remaining carbon can still be found at the forest (38%), where it was transferred from living biomass to necromass. These reservoirs delay carbon emissions and together with forest regrowth determined the balance. As a result, the difference between carbon sequestration via regrowth (i.e., −8.15) and life cycle carbon emissions was −3.06 Mg C ha−1 15 year−1. Alice-Guyer et al 2019

Multitemporal analyses allowed the authors to analyze the annual variations in logging and deforestation, as well as the interaction between them. It is shown that, because of both rapid regrowth and deforestation, evidence of logging activities often disappeared within 1–3 yr. Matricardi et al. 2005

Short and long-term effects of selective logging.

|  |  |  |  |
| --- | --- | --- | --- |
| Tipo de logging | Duração do efeito | Higher mortality among damaged trees | Pinnard & Putz 1996 |
|  |  | Biomas loss | Pinnard & Putz 1996 |
|  |  | Reduced canopy cover | Nepstad et al 1998 |
|  |  | Increased light availability | Nepstad et al 1998 |
|  |  | Increased debris | Nepstad et al 1998 |
| Seletivo (1-4 tree/ha) | Pelo menos 10 anos | Bird assemblages impover. | Thiollay et al. 1992 |
|  | Pelo menos 10 anos | Forest structure/damage | Thiollay et al. 1992 |
|  | Several years following logging | Tree mortality after operations (wind disturbance) | Thiollay et al. 1992 |
|  |  | Soil compaction | Thiollay et al. 1992 |
| Intenso 50-60% adult stems | Mais de 50-100 anos | Invasão biológica plants | Brown & Gurevitch 2004 |
|  | Mais de 50-100 anos | Woody plant assemblages | Brown & Gurevitch 2004 |
|  | Mais de 50-100 anos | DBH stem distribution |  |
| Six trees in average (38 m 3 ha- 1) |  | Tree damage and gaps | Verissimo et al. 1992 |
| Selective logging | 12 year after logging | Bird composition | Aleixo 1999 |
|  |  | Forest structure (several tables) all layers | Aleixo 1999 |
| Not specified |  | Logging expansion | Gaveau et 2014 |
| Selective (10 trees per ha) | 15-25 yr after logging | Rare mammal species | Well et al 2007 |
| Selective (10 trees per ha) | 15-25 yr after logging | Rare mammal species | Well et al 2007 |
| Selective (8 trees per ha, 35 m3) | 4-10 yr after logging | Ant community, forest structure and litter | Vasconcelos et al 2000 |
| Selective logging > 40 cm (140 m3) | 5 years after | Soil microbiology | TRIPATHI et al 2016 |
| 14 trees ha possibly | 5 yr after | Butterfly assemblages | Hill et al 1995 |
|  |  | Forest structure | Hill et al 1995 |
| 14-21 m3; 3-9 trees ha | Logging in 1969 | Forest structure | Struhsacker et al 1996 |
|  |  | Damage by elephants | Struhsacker et al 1996 |
|  | Land use change 2000-2010 | Logging dynamics and the future of concessions | Gaveau et al 2013 |
| 1 stem per ha (highly selective) | 6 mo to 18 years | Forest structure | Hall et 2003 |
|  |  | Tree assemblages | Hall et 2003 |
| 19-80 m3 ha | Since 1942 multiple cicles | Primates | Plumptre & Raynolds 1994 |
| Not informed | During logging operation | Soil sediments | Sidle et al 2004 |
|  |  | Track length | Sidle et al 2004 |
| 18 trees ha | 5-6 yr after | Tree assemblages | John 1985 |
|  | During operatiom | Forest structure | John 1985 |
| Not informed | Not informed | Avian throphic structure | John 1985 |
| 90 m3 | 8 yrs after the first logging | Bird community | Lambert 1992 |
|  | During logging operations | Forest structure | Lambert 1992 |
| Moderate-logging | 17 years | Carnivores | Gerber et al 2012 |
|  | 17 years | Forest structure | Gerber et al 2012 |
| Review | Review | Review | Zimmerman & Kormos 2012 |
| Review | Review | Fire in Australia | Lindenmayer et al. 2020 |
|  |  | Legislation in Camdodja | Le Billon 2002 |
| Review | Review | Fire | Lindenmayer et al 2009 |
| Review | Review | Roads impact | Kleinschroth &Healey 2017 |
|  |  | Loggind and fire dynamics | Matricardi et al 2010 |
| 11-41 trees ha | 24 years | Forest recovery | Gourlet-Fleury et al. 2013 |
|  |  | Stock recovery |  |
| 3.9 m3 ha−1, equivalent to only 1.1 trees ha−1 | 16 months | Vertebrate response | Bicknel & Peres 2010 |
| (2.6–6.4 felled trees ha | 3.5 yrs | Canopy gaps | Asner et al. 2004 |
|  |  | Logging intensity |  |
|  |  | Damage maps |  |
| 75 m3 ha-’16 trees 45 cm dbh. | 13-18 yrs | Forest recovery | Silva et 1995 |
| 35-69 m3 | 5-9 yr | Forest structure | Gerwing 2002 |
|  |  | Debris |  |
|  |  | Biomass |  |
|  |  | Tree species richness |  |
|  |  | Lianas |  |
|  |  | Regeneração |  |
| 38.9 and 37.4 m3 ha1 | 16 yrs | Biomass recovery | West et al 2014 |
| 23 m3 ha | 2-3 yr | Canopy cover | Pereira et al 2002 |
|  |  | Ground damage |  |
|  |  | Logging intensity |  |
| 4.35 trees/ha and 12.1 m3 /ha | During opperation | Forest structure  Road damage | Jackson et al 2002/Review |
|  |  | Tree damage |  |
|  |  | Fire damage in logged forests/forest structure | Woods 1989 |
|  |  | Grass invasion |  |
|  |  | Tree mortality/canopy open |  |
|  | 7-8 yrs | Volume | Magnusoon et al 1999 |
|  |  | Species richness |  |
|  |  | Tree composition |  |
| y 0.82 trees (8.11 m3 ) | Early after | Forest structure | Medjibe et al 2011 |
|  |  | Logging intensity and damage |  |
|  |  | Biomasss |  |
| 17 m3 ha1 (approximately 4 trees ha1 ), | 11 years | Seedling performance | Darrigo et al 2017 |
|  |  | Focal tree species response |  |
|  |  | Canopy opness |  |
|  |  | Soil response (phosporous) |  |
| Review | Review | Bird response | Barlow et al 2006 |
| 18 m3 ha–1 | 20–42 months | Bat response | Presley et al 2008 |
|  |  | Rare species |  |
|  |  | Functional groups |  |
| Review | Review | Logging practices | Putz et al. 2000 |
| Selective logging | 15-25 yr | Tree species composition | Slick et al 2002 |
|  |  | Seedling and saplings |  |
|  |  | Canopy openness |  |
| Mahogany logging |  | Forest damage | Whitman et al 1997 |
| t (18.7 m3/ha), | 2-42 meses | Bird response | Wunderle et al. 2006 |
|  |  | Forest structure |  |
| Review | Review | Biodiversity persistance and forest trajectory | Edwards & Laurance 2013 |
| Intensive logging | 1-2 yrs | Butterflies | Lewis 2001 |
|  |  | Light availability |  |
| g 78 + 7 0m3 ha-' | 2, 5 and 12 years | Deer | Heydon & Bulloh 1997 |
|  |  | Forest structure |  |
|  |  | Fruit availability |  |
|  |  | Ficus density |  |
| Meta-analysis (26 papers) | Meta-analysis across the tropics | Bird response | Burivalova et al 2015 |
|  |  | Functional groups |  |
|  |  | Response models |  |
| 23-50 m3 4-8 trees ha | During logging opperation | Forest structure | Uhl & Vieira 1989 |
|  |  | Forest damage |  |
|  |  | Profitability |  |
| Review & Policy |  | Logging emissions | Elly et al 2019 |
| 3–7% of trees > 60 dbh 70 m3 /ha 12-15 trees per ha | 6-yrs | Moth response | Willott et al 1999 |
| on average 3.04 (1.59) trees ha | 1-12 years | Birds | Thiollay et al. 1997 |
| 43% of pre-cut total basal area | 6 mo, 1- 8 years | Forest structure/damage | Cannon et al 1994 |
|  |  | Tree species abundance |  |
| Mohagani logging (1–4 trees/ha | 10 yr | Bat community | Peters et al 2006 |
| Review | Review | NTFP | Rist et al 2012 |
| 15 % trees > 35 cm DBH | 10-36 meses | Biomass and recovery | Figueira et al 2008 |
|  |  | Tree growth and mortality |  |
|  |  | Forest structure |  |
|  |  | NPP |  |
| 1-15 trees (16–94 m3 ha | 20-yrs | Functional profile of tree | Baraloto et al 2012 |
|  |  |  | Pereira et al |
| Multiple sites | 1-5 yr | Bird response | Marsden 1998 |
| 10–15 m3 ha−1; 5–7% of total standing volume harvested | Early-after | Forest structure via lidar | Andersen et al. 2014 |
|  |  | Biomass |  |
| 21.3 m3 há; 16.3 trees ha1 | 2-8 yrs | Large trees | Sist et al 2014 |
|  |  | Tree density and biomass |  |
|  |  | Structure |  |
|  |  | List of studies |  |
| 22.7 to 270.0 m3 /ha (mean = 92.4, | 18 years | Tree species richness | Berry et al 2008 |
|  |  | Alfa, beta e gama diversity |  |
|  |  | Forest structure |  |
|  |  | Ecological composition |  |
| Meta-analysis | Meta analysis | Biomass and carbon | Martin et al 2015 |
| n 10 m3 per ha | 10-yr | Small rodents | Ganzhorn et al 1990 |
|  |  | Forest structure |  |
| Low and intensive | Response to 1 and two cicles | Exploited species /simulated responses | Degen et al 2006 |
|  |  | Demographic and genetic |  |
| Review 4-38 trees ha | 20-30 yrs | Forest structure and composition | Gatti et al 2015 |
|  |  | Biomass |  |
| 2.9 to 7.3 trees/ha were authorized for harvest, or 5.8 to 14.2 m3/ha | 1-5 yr | Understorey birds | Mason 1999 |
| Simulation |  | Carbon & Biomass | Huang & Asner 2010 |
|  |  | Logging intensity (map) |  |
|  |  | Produtividade |  |
|  |  | Carbono no solo |  |
|  |  | Emissão de gases |  |
| 73 m3 ha | 15 yr | Butterflies | Dunbrell & Hill 2005 |
|  |  | Narrow-range species |  |
|  |  | Forest structure |  |
|  |  | Pioneer tree species |  |
| 118 m3 ha | 2-5-12 yr | Small carnivores (civets) | Heydon & Bulloh 1996 |
| 113 m3 ha | 9-12 yr | Microclimate | Senior et al 2017 |
|  |  | Forest structure |  |
| 32.5 to 53.4 m3 ha | 11 yr | Biomass/forest structure | Rutishauser et al 2016 |
| Review | Review | Forest disturbance/intact forest (%) | Putz et al 2019 |
| 118 m3 ha | 8±9 years ago | Butterfly species | Hill 1999 |
|  |  | Forest structure |  |
| 100 m3 ha | 5 yr | Suspended sediment | Chapell et al 2004 |
| 19.5 trees or approximately 57 m3 ha | 2-6 anos | Amphibians response | Ernest et al 2006 |
|  |  | Functional composition/diversity |  |
| Review | Review | Logg and deforestation | Asner et al 2009 |
| 14 trees per ha | 5-yr after logging | Butterflies | Hill et a. 1995 |
|  |  | Forest structure |  |
| 3 trees ha | 10-20 yr | Amphibians | Adum et al. 2012 |
| review | Review meta-analysis | Forest damage | Picard et al 2012 |
| 16.8 m3 /há m; 0 to 50.31 m3 /ha; 0–7.9 trees/ha | 10 months | Dung beetles/fat content | França et al 2016 |
| Suppl material | Supplement material | Songbirds/bioacustic | Pillay et al 2018 |
| Suppl material | Supplement material | Lianas and tree networking | Magrach et al 2016 |
| 12 trees ha1 | < 5 yr | Seed trees/better practices | Freitas & Pinard 2008 |
| 0-34 tree ha | Rencent vs. past | Ecosystem functions | Schleuning et al 2011 |
| 3-9 trees ha | 12-17 yr | Seed bank | Dupuy & Chazdon 1998 |
| 7.2 - 9.2 trees/ha  104.25-111.9 m3 /ha | Early-after/immediately | Forest structure/stand damage | Matangaran et al 2019 |
|  |  | Tree injury vs. log intensity; modelo matematico | Matangaran et al 2019 |
| 115 m3 ha | 10-15 yr | Birds | Messina et al 2021 |
|  |  | Body size/ecological groups |  |
| 1.4-4.1 trees ha | 3 yrs | Aboveground biomass | Ota el al 2019 |
| 11.08 m3 ha− | Through the whole 15-yr cicle | Carbon flux | Alice-Guyer et al 2019 |
|  |  | Forest damage |  |
|  |  | Forest strucuture |  |
| 11 m3 ha | immediately | Carbon stocks | Brown et al 2005 |
|  |  | Forest damage |  |
| Review | Review | Logging expansion | Matricardi et al. 2005 |
|  |  |  | Hill et al 2003 |
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