

The life cycle carbon balance of selective logging in tropical forests of Costa Rica

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Abstract

The effect of logging on atmospheric carbon concentrations remains highly contested, especially in the tropics where it is associated to forest degradation. To contribute to this discussion, we estimated the carbon balance from logging natural tropical forests in Costa Rica through a life cycle accounting approach. Our system included all major life cycle processes at a regional level during one rotation period (15 years). We used mass flow analysis to trace biogenic carbon. Data were gathered from all logging operations in the Costa Rican NW region (107 management plants), a sample of industries transforming wood into final products (20 sawmills), and national reports. We estimated a surplus of $-3.06 \text{ Mg C ha}^{-1} 15 \text{ year}^{-1}$ stored within the system. When accounting for uncertainty and variability in a Monte Carlo analysis, the average balance shifted to $-2.19 \text{ Mg C ha}^{-1} 15 \text{ year}^{-1}$ with a 95% CI of -5.26 to 1.86 . This confidence interval reveals probabilities of a net increase in atmospheric carbon due to harvesting although these are smaller than those from a system that acts as a reservoir. Our results provide evidence for the carbon neutrality of bio-materials obtained from natural forests. We found that anthropogenic reservoirs play a determinant role in delaying carbon emissions and that these may explain differences with previous carbon balance studies on tropical forest management. Therefore, the climate mitigation potential of forest-derived products is not exclusive to forest management, but measures should be considered throughout the processes of wood transformation, use, and disposal.

KEYWORDS

carbon neutrality, climate change mitigation, harvested wood products, industrial ecology, life cycle assessment, tropical selective logging

1 | INTRODUCTION

Sustainable forest management for wood production is a potential climate mitigation option as wood products may accumulate in anthropogenic reservoirs for relatively long periods, avoiding carbon locked in wood from reaching the atmosphere (Brandão et al., 2012; Smith et al., 2014). Anthropogenic reservoirs consist of products in use, where harvested wood products (HWP) can last for 5–100 years (Brunet-Navarro, Jochheim, & Muys, 2017) and solid waste disposal sites (SWDS) in which some HWPs after their end of life, may remain stored even longer (De la Cruz, Chanton, & Barlaz, 2013; Ximenes, Björndal, Cowie, & Barlaz, 2015). Globally, these reservoirs are known to be growing (Butarbutar, Köhl, & Neupane, 2016; Hashimoto, 2008; Hashimoto, Nose, Obara, & Moriguchi, 2012) and this storage component is an increasingly important part of the land-use related carbon balance.

The climate mitigation potential of forest management depends on storage in anthropogenic reservoirs, but also importantly on the losses associated with timber processing; from harvesting to final application. In the tropics, 1.1 Gt CO₂ emissions have been estimated due to logging

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and it is therefore commonly described as the main cause of forest degradation (Houghton, Byers, & Nassikas, 2015; Pearson, Brown, Murray, & Sidman, 2017). For example, to extract one cubic meter of timber, an associated 1 to 3 Mg C may be lost from the forest due to log extraction, logging damage, and the infrastructure needed for forest operations (Pearson, Brown, & Casarim, 2014). From the extracted timber only a fraction will be transformed into products, out of which an even smaller fraction will remain stored in long-term anthropogenic reservoirs, and the allocation of harvested timber to different product classes determines overall residence time of carbon in wood products.

A third component determining the magnitude of the climate mitigation potential of forest management is the recovery of carbon emissions from the forest. However, these emissions are temporary (as long as no land use change occurs) and will recover through forest regrowth (Houghton, 2012; Houghton, 2013; Houghton et al., 2015; Smith et al., 2014). Recent evidence shows that growth rates after logging tend to increase (Piponiot et al., 2018), although uncertainties on the rates of carbon sequestration due to spatial variation still remain (Baccini et al., 2012). In the case of tropical logged forests, biomass losses have been shown to partly explain the recovery time (Rutishauser et al., 2015) and have been used to estimate the carbon balance.

Estimating the carbon balance is a step closer to the potential climate impact of logging given that it considers all processes leading to carbon emissions, storage, and sequestration, which can take place at different spatial and temporal scales (Newell & Vos, 2012). To integrate these processes, a life cycle carbon accounting approach has been recommended (Geng, Yang, Chen, & Hong, 2017; Hauschild et al., 2018; Klein, Wolf, Schulz, & Weber-Blaschke, 2015; Knauf, 2015; Lippke et al., 2011). This approach accounts for changes in biogenic carbon (BioC; i.e., carbon stored in biomass) from the forest and up until the end of life (EoL) of wood products. That is, it includes the decomposition of biomass in the forest due to logging damage; carbon storage in anthropogenic reservoirs over time, and; depending on the type of EoL, the moment when carbon is released back to the atmosphere via combustion or decomposition. Also, this approach allows the inclusion of carbon sequestered by the forest via regrowth to produce a complete biogenic carbon life cycle balance (BioC-LC).

In the tropics, efforts have been made to integrate the life cycle of timber harvesting and wood use to determine an overall BioC-LC, but attempts so far have been restricted by a lack of empirical data (Murphy, 2004; Numazawa, Numazawa, Pacca, & John, 2017; Piponiot et al., 2016). This situation is not unique for tropical forests since only few life cycle studies include a complete BioC-LC (Cardellini et al., 2018; de Rosa, Schmidt, Brandão, & Pizzol, 2017; Downie, Lau, Cowie, & Munroe, 2014; Helin, Sokka, Soimakallio, Pingoud, & Pajula, 2013; Jasinevičius, Lindner, Cieniala, & Tykkyläinen, 2018; Liu et al., 2017; Newell & Vos, 2012). It is often assumed that the overall integration of these processes results in a carbon neutral outcome, but this assumption has raised a considerable debate (Cardellini et al., 2018; Helin et al., 2013; Johnson, 2009).

For tropical logging, it has been shown that including a detailed biomass life cycle can result in delayed carbon emissions (Piponiot et al., 2016). Although in that study tropical forests producing timber are reported mainly as sources of carbon, zero net emissions (i.e., carbon neutrality) are included within the 95% confidence interval. Therefore, given the conservative assumptions on product use (e.g., HWP being one third of harvest with the rest assumed to be sawdust) and the exclusion of the EoL phase of wood products, it is possible that life cycle processes leading to carbon storage may increase the chance of a carbon neutral outcome.

Here, we present the carbon balance of selectively logged tropical forest in Costa Rica, in which all processes until the end of life of wood products are integrated using a life cycle approach. We do this based on a mass flow analysis (Geng et al., 2017; Jasinevičius et al., 2018) using foreground data collected for natural forest harvest operations, sawmilling industry, and wood product use in Costa Rica. To account for uncertainty and variation, we perform a Monte Carlo analysis (Clavreul, Guyonnet, & Christensen, 2012; European Commission, 2010; Heijungs & Huijbregts, 2004; Heijungs & Lenzen, 2014; Huijbregts, 1998; Lo, Ma, & Lo, 2005). We address whether, under the circumstances found in our case study, a disturbance to the natural carbon cycle in tropical forests due to human interventions lead to accumulation or loss of carbon, or whether the system can be considered neutral in terms of carbon cycling. Such questions are especially useful in tropical countries, as answers to these bear relevance about the potential of forest management in national REDD+ (Reduced Emissions from Deforestation and Degradation) or other climate mitigation strategies.

2 | METHODS

2.1 | Approach and system boundaries

This study focuses on the exploitation of natural wet and moist forests in the Northern Caribbean region of Costa Rica. This region is responsible for 83% of timber harvest from natural forests in the country (MINAE, 2011, 2012, 2013). Changes in biogenic carbon pools are quantified along all stages within the product system, that is, harvesting operations, sawmilling, transformation into end products, product and co-product use, and end of life management. Products are defined as the intended output of the milling process (e.g., sawnwood), while co-products are by-products (i.e., slabs, bark, edges, off-cuts, sawdust, and shavings) with a market value (e.g., as fuelwood or pellets).

The temporal boundary used here is one rotation period which in tropical forests can vary from 15 to 60 years (Rutishauser et al., 2015; Sasaki, Chheng, & Ty, 2012). In Costa Rica, the rotation period is determined based on information on forest recovery through a pre-harvest inventory. There is variation among production forests but it has not been quantified. For this reason, we fixed this period to the minimum allowable length of

15 years according to national legislation (MINAE, 2002, 2009). As a result our estimate of recovery of forest carbon stocks are probably conservative, given that in practice cutting cycles are likely considerably longer.

Based on these boundaries, we approximate carbon emissions due to biomass decomposition and combustion when they occur, together with forest regrowth. This is done for an average hectare of natural tropical forest in Costa Rica, where timber is extracted for wood products and co-products. Therefore, the sum of all carbon gains and losses are allocated to one hectare of natural tropical forest.

Soil carbon was excluded based on probable limited changes from this stock due to small impacted area and a short duration of the impact (Pearson et al., 2014), together with large uncertainties around its estimation (Baccini et al., 2012; de Rosa et al., 2017; Mohren, Hasenauer, Köhl, & Nabuurs, 2012). In terms of processes, recycling was excluded given lack of data, with only 1–2% reported by the furniture industry (Solera, 2014). In both cases, evidence that a continuous cover system with a proportionally low impacted area (Pearson et al., 2014) and without residue collection can lead to soil carbon increases (Helin et al., 2013), and that the effect of recycling results in the prolongation of the life of products, seem not to challenge the conservativeness of this BioC-LC. Finally, because harvesting does not cause deforestation in Costa Rica (Arroyo-Mora, Svob, Kalacska, & Chazdon, 2014), land use change was also excluded and the “no use” scenario becomes the reference (Helin et al., 2013).

Foreground data was collected from the revision of all management plans within the study region during 2010–2016 and field questionnaires for all stages except end of life management. For EoL, background data were taken from national reports (Table S1 in the Supporting Information).

2.2 | Carbon stocks in forest biomass

Plots from the National Forest Inventory within our study region (Programa REDD/CCAD-GIZ -SINAC 2015) were used together with a site-specific allometric equation (Fonseca, Alice, Rojas, Villalobos, & Porras, 2016) (Table S1 in the Supporting Information) to determine average carbon per hectare. This equation estimates all ecosystem biomass, that is, above and belowground tree biomass, herbaceous vegetation, and necromass.

2.3 | Wood harvest and logging damage

To account for wood harvest and carbon emissions from logging operations, we reviewed the management plans submitted to the regional offices of the Ministry of Environment and Energy (MINAE) in Costa Rica during the period 2010–2016. A total of 107 forest management plans and their corresponding audit reports were reviewed.

Reported extracted volumes over bark from felled standing trees (> 60 cm minimum harvestable diameter) and deadwood, were converted to biomass using wood densities (g cm^{-3}) (Chave et al., 2009). Species were grouped according to their traditional classification as hardwoods, semi-hardwoods, and softwoods, with some remaining as unclassified (Zúñiga-Méndez, 2016). This was done to accommodate species for which wood densities were not available. Average wood density was determined per group and weighted by the group's contribution to total volume. Further conversion into harvested carbon (H) was calculated using a site-specific carbon fraction for tree stems (Fonseca et al., 2016).

Logging damage was estimated based on the area impacted as reported in the reviewed management plans (Table S1 in the Supporting Information) and the carbon stock in the forest biomass. We assumed that residual large trees (> 40 cm DBH) are not damaged during the construction of infrastructure or during felling operations. The ecosystem carbon excluding these trees was $55.83 \text{ Mg C ha}^{-1}$ or 55% of total ecosystem carbon and within the range of 28–56.2% reported in the literature (Sasaki et al., 2012). In the case of gaps from felling, we also included the additional carbon from the tree compartments from extracted logs which remain in the forest as slash (i.e., leaves, branches, and roots). This was done using the biomass expansion factors and root to shoot ratios (Fonseca et al., 2016). In case of harvested deadwood, no carbon emissions due to gap formation were calculated since this extraction does not involve felling.

Decomposition was included assuming exponential decay with a 0.1 year^{-1} (Houghton et al., 2000) decay constant. We report carbon emissions (LD_{15}) and stocks in the system after the 15-year period.

$$LD_{15} = (Gp + LgDck + PrmRd + ScRd + SkTr) \times (1 - e^{-k_1 t}) \quad (1)$$

where

LD_{15} = Carbon in logging damage decomposed by year 15 (Mg C ha^{-1})

Gp = Initial amount of carbon from felling gaps (Mg C ha^{-1})

$LgDck$ = Initial amount of carbon from logging decks (Mg C ha^{-1})

$PrmRd$ = Initial amount of carbon from primary roads (Mg C ha^{-1})

$ScRd$ = Initial amount of carbon from secondary roads (Mg C ha^{-1})

$SkTr$ = Initial amount of carbon from skid trails (Mg C ha^{-1})

k_1 = Decay rate of deadwood; 0.1 year^{-1}

t = years; 15 years

2.4 | Forest regrowth

As we lacked observations on forest regrowth in our study region, we used results from a meta-analysis of 10 logged Neotropical forests (Rutishauser et al., 2015) to estimate the time it takes for forest carbon to recover to pre-logging carbon stock (RT). RT is a function of carbon lost, that is, the sum of logging damage (LD_0) and extracted wood (H). RT was used to determine the growth rate until the initial biomass was reached.

$$RT = \left(\frac{(H + LD_0) \times 100}{CSFB} \right)^\theta \quad (2)$$

where

RT = Recovery time (years)

H = Harvest; sum of carbon extracted, both standing trees and deadwood ($H_{St} + H_{Dw}$) (Mg C ha^{-1})

LD_0 = Carbon from logging damage ($Gp + LgDck + PrmRd + ScRd + SkTr$) (Mg C ha^{-1})

$CSFB$ = Carbon stock in forest biomass (Mg C ha^{-1})

$\theta = 1.106 \pm 0.022$

$$FR_{15} = \min \left(H + LD_0; t \times \frac{(H + LD_0)}{RT} \right) \quad (3)$$

where

FR_{15} = Carbon from forest regrowth by year 15 (Piponiot et al., 2016)

t = rotation period

2.5 | Sawmill biomass and carbon flow

Based on the available forest management plans we identified a total of 42 sawmills and selected those processing timber from natural forests. After an initial contact, we selected 20 sawmills to include in our survey. Most selected sawmills were located within the study region; four were located at ~ 100 km distance.

We developed a questionnaire for sawmilling and gathered data on all biomass inputs and outputs. The reported types and amounts of wood products, co-products and residues were used to develop the carbon flow within the sawmill.

Products were grouped according to their end use and classified as short, mid-, or long-term to assign half-lives. For example, all sawnwood used as formwork was considered a "short-term" product, while the remaining sawnwood that does require further transformation at the milling stage (i.e., planing and molding of wood flooring, boxboard, moldings, scantlings, beams, etc.), were grouped into one single category, that is, "construction" and classified as "long-term" products.

All forms of by-products (e.g., slabs, edges, sawdust, etc.) were traced independently but grouped into co-products depending on their end use. For example, slabs, sawdust, and shavings are all used for pellets, while edges and off-cuts are used in the furniture industry. Although being a co-product, this last end use was further classified as a "mid-term" product due to its half-life.

2.6 | Transformation into end use products

Long-term and mid-term products (i.e., wood used in construction and furniture) require an additional transformation outside the mill before becoming products in use. For these, we used a questionnaire for the secondary transformation industry to determine the fraction of wood that becomes residues and that is sent to EoL (i.e., TL_f in Equations (8) and (9)). Other categories have no further transformation (e.g., formwork), or it makes no difference given that complete carbon loss is assumed to occur on the year of harvest (e.g., pellets).

2.7 | End use phase of the life cycle

No carbon emissions from the main wood products take place at this stage but there is a flow of carbon from products in use to EoL. To quantify this flow, we used first order decay and half-lives that varied according to products. As described in section 2.6, products were classified as short, mid-, and long-term, and assigned half-lives of 2, 25, and 35 years (i.e., k_4 , k_5 and k_6 in Equations (8) and (9)) (IPCC, 2014).

For co-products, carbon can be lost during use. In the case of fuelwood and pellets used for bioenergy, we assumed emissions take place immediately on year 0. For the decomposition of sawdust and shavings used as flooring for stables or as compost in nurseries, we assumed the same rate as forest biomass (i.e., k_1) given the conditions under which these will decompose. This may result in a slight underestimation of carbon emissions due to the smaller particle size of this co-product.

$$Plt_0 = H \times (SwdPlt_f + ShvPlt_f + SBPlt_f) \quad (4)$$

where

Plt_0 = initial amount of carbon in wood used for pellets (Mg C ha⁻¹)

$SwdPlt_f$ = fraction from harvest that becomes sawdust and is used as pellets

$ShvPlt_f$ = fraction from harvest that becomes shavings and is used as pellets

$SBPlt_f$ = fraction from harvest that becomes slabs and bark and is used as pellets

$$Fw_0 = H \times (SwdFw_f + ShvFw_f + SBFw_f) \quad (5)$$

where

Fw_0 = initial amount of carbon from wood used as fuelwood (Mg C ha⁻¹)

$SwdFw_f$ = fraction from harvest that becomes sawdust and is used as fuelwood

$ShvFw_f$ = fraction from harvest that becomes shavings and is used as fuelwood

$SBFw_f$ = fraction from harvest that becomes slabs and bark and is used as fuelwood

$$SSN_{15} = H \times ((SwdSSN_f + ShvSSN_f) \times (1 - e^{-k_1 t})) \quad (6)$$

where

SSN_{15} = carbon in wood used for stables, stalls, and nurseries decomposing by year 15 (Mg C ha⁻¹)

$SwdSSN_f$ = fraction from harvest that becomes sawdust and is used in stables, stalls, or nurseries

$ShvSSN_f$ = fraction from harvest that becomes shavings and is used in stables, stalls, or nurseries

2.8 | End of life

We approximated the EoL of products and residues by determining the amount that decompose in solid waste disposal sites (SWDS) and those that are open burned. This distribution was taken from the National GHG Inventory (Chacón, Jiménez, Montenegro, Sassa, & Blanco, 2012). Once the fraction of products reached SWDS, we used the default value of 0.5 as the decomposable degradable organic carbon fraction (DOC_f) and half-lives differentiated by wood type; that is, 20 years for wood products, slabs, and bark, and 10 years for sawdust and shavings at the mill dump (Pipatti et al., 2006).

Carbon emissions from the mill dump [Equation (7)] were estimated separately for groups of wood residues due to differing half-lives. In Equation (8), we first estimate the flow of each wood product category (STP , MTP , and LTP) into SWDS using half-lives described in the previous section (i.e., wood products retired from service). Then, once in a SWDS, we determine the outflow/emissions using exponential decay and a single half-life for all products (k_3). Transformation losses were subtracted from products and accounted separately because these flow directly to SWDS on year 0. Finally, emissions were estimated only for the fraction that is effectively lost (i.e., DOC_f).

For the fraction that is open burned [Equation (9)], we applied the same logic were transformation losses are subtracted from products, and the outflow from products in use is estimated based on each products' half-life. The main difference is that all carbon was assumed to be lost as soon as residues were disposed of or products were retired from service.

$$SmR_{15} = (H \times (SwdR_f + ShvR_f) \times (1 - e^{-k_2 t})) + (H \times (SBR_f \times (1 - e^{-k_3 t}))) \times DOC_f \quad (7)$$

where

SmR_{15} = carbon in sawmill residues decomposing by year 15 (Mg C ha⁻¹)

$SwdR_f$ = fraction from harvest that becomes sawdust residues during milling

$ShvR_f$ = fraction from harvest that becomes shavings residues during milling

SBR_f = fraction from harvest that becomes slabs and bark residues during milling

DOC_f = fraction of degradable organic carbon that can decompose; 0.5

$k_2 = \ln(2)/10$

$k_3 = \ln(2)/20$

$SWDS_{15} = SWDS_f$

$$\begin{aligned} & \times \left(\sum_{i=0}^{14} \left((H \times ((STP_f \times e^{-k_4 i}) \times (1 - e^{-k_4})) + ((MTP_f \times (1 - TL_f) \times e^{-k_5 i}) \times (1 - e^{-k_5})) \right. \right. \\ & \left. \left. + ((LTP_f \times (1 - TL_f) \times e^{-k_6 i}) \times (1 - e^{-k_6})) \right) \times (1 - e^{-k_3(t-i)}) \right) + (H \times ((MTP_f \times TL_f) + (LTP_f \times TL_f)) \times (1 - e^{-k_3 t})) \\ & \times DOC_f \end{aligned} \quad (8)$$

where

$SWDS_{15}$ = carbon in wood decomposing at SWDS during the 15-year period ($Mg\ C\ ha^{-1}$)

$SWDS_f$ = fraction of wood decomposing at SWDS

STP_f = fraction of wood harvest that becomes short-term products

MTP_f = fraction of wood harvest that becomes mid-term products

LTP_f = fraction of wood harvest that becomes long-term products

TL_f = fraction of wood that becomes residues and is sent to EoL during the final transformation of mid and long-term products

$k_4 = \ln(2)/2$ (STP)

$k_5 = \ln(2)/25$ (MTP)

$k_6 = \ln(2)/35$ (LTP)

$i = \text{years } 0-14$

$$\begin{aligned}
 OB_{15} = & (1 - SWDS_f) \\
 & \times \left(\left((H \times STP_f) \times (1 - e^{-k_4 t}) \right) + \left(\left((H \times MTP_f) \times (1 - TL_f) \right) \times (1 - e^{-k_5 t}) \right) + (H \times MTP_f \times TL_f) \right) \\
 & + \left(\left((H \times LTP_f) \times (1 - TL_f) \right) \times (1 - e^{-k_6 t}) \right) + (H \times LTP_f \times TL_f) \quad (9)
 \end{aligned}$$

where

OB_{15} = carbon in wood products open burned during the 15-year period ($Mg\ C\ ha^{-1}$)

2.9 | System balance, uncertainty, and sensitivity analyses

All previous equations were combined to obtain the system's net carbon balance [summarized in Equation (10) for illustration purposes only]. Although we did not account for carbon storage in products or SWDS directly, the difference between the inflow and outflow from these reservoirs indicates storage.

$$\text{System Balance} = (LD_{15} + SmR_{15} + Plt_0 + Fw_0 + SSN_{15} + EoL_{15}) - FR_{15} \quad (10)$$

To evaluate the effect of parameter variability and the uncertainty of the carbon balance, we determined the probability density functions for all 32 parameters used in the analysis (Table S2 in the Supporting Information). We randomly sampled from their distributions, calculated the carbon balance, and repeated this procedure through Monte Carlo simulations 10,000 times. We then calculated the mean and confidence interval of the carbon balance.

To assess the sensitivity of carbon balance to variation in parameters, we also performed a sensitivity analysis. This differs from a Monte Carlo analysis in that it applies equal changes (+/− 10%) to each parameter separately and then evaluates the effect on the carbon balance.

3 | RESULTS

3.1 | Harvest, logging damage, and forest regrowth

A total of 7,756 ha were harvested in the region during 2010–2016, with an average area per forest management plan of 80 ha ($n = 97$; $\sigma = 67$), ranging from 3.5 to 325 ha. Average standing tree harvest was $11.08\ m^3\ ha^{-1}$ ($n = 65$; $\sigma = 6.23$) and deadwood was $1.95\ m^3\ ha^{-1}$ ($n = 54$; $\sigma = 7.02$), resulting in a total harvest of $13.03\ m^3\ ha^{-1}$ (over bark).

Average wood density of hardwoods, semi-hardwoods, softwoods, and “unclassified” species were 0.66, 0.45, 0.33, and $0.52\ g\ cm^{-3}$, respectively. The weighted average wood density ($0.4965\ g\ cm^{-3}$) was used together with a carbon content of 0.447 (Fonseca et al., 2016) to determine carbon in the harvest. Total harvested carbon was $2.89\ Mg\ C\ ha^{-1}$, distributed in $2.46\ Mg\ C\ ha^{-1}$ ($n = 65$; $\sigma = 1.38$) from felled trees and $0.43\ Mg\ C\ ha^{-1}$ ($n = 54$; $\sigma = 1.56$) for deadwood. Carbon stock at the ecosystem level was estimated using NFI average basal area ($25.8\ m^2\ ha^{-1}$; $n = 9$; $\sigma = 8.79$) and resulted in $101.72\ Mg\ C\ ha^{-1}$ ($n = 9$; $\sigma = 32.9$). Carbon in the diametric classes damaged during harvesting (i.e., DBH < 40 cm) represented $55.83\ Mg\ C\ ha^{-1}$ ($n = 9$; $\sigma = 15.81$).

According to the 31 forest management plans that reported area impacted by logging, gaps from tree felling represented the largest amount with 3.62 ha or 5.3% of the total forest area. Secondary roads represent 1.01 ha (1.8%) and skid trails 0.8 ha (1.4%). Primary roads and logging decks inside the forest caused only a marginal impact with 0.25 (0.2%) and 0.11 ha (0.2%), respectively. Total carbon impacted during logging,

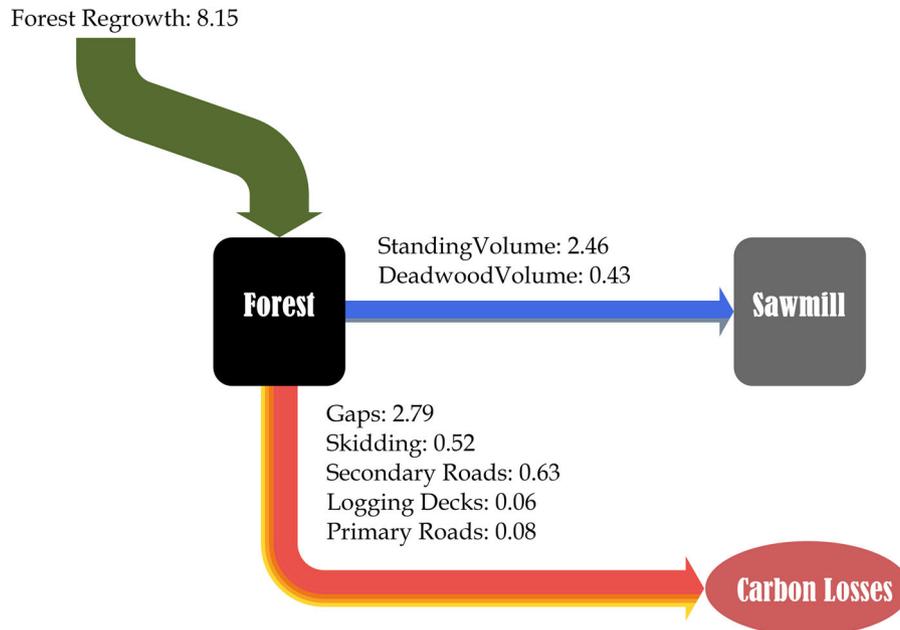


FIGURE 1 Forest carbon flows during one logging cycle of 15 years for an average hectare of exploited tropical forest in Costa Rica ($\text{Mg C ha}^{-1} 15 \text{ year}^{-1}$; box sizes are not indicative of the size of the stock but black boxes represent carbon storage)
Note. Tabular data can be found in Table S3 in the Supporting Information.

excluding harvest, was $5.26 \text{ Mg C ha}^{-1}$. Of this amount, $4.09 \text{ Mg C ha}^{-1}$ was lost due to decomposition during the 15-year period, while $1.17 \text{ Mg C ha}^{-1}$ (22.3%) remains in the forest as necromass (Figure 1; Table S3 in the Supporting Information).

We estimated 9.99 years for the full recovery of carbon stocks in logged forests. During this period, carbon stocks increased at a rate of $0.82 \text{ Mg C ha}^{-1} \text{ year}^{-1}$. This estimate was obtained using total harvested volume plus all carbon from logging damage, that is, 8% of total initial ecosystem carbon or $8.15 \text{ Mg C ha}^{-1}$. In order not to overestimate recovery time and because the original model (Rutishauser et al., 2015) assumes committed emissions, we used logging damage on year 0 instead of our 15-year estimate considering decomposition.

3.2 | Sawmill and wood transformation

The main products from milling are boards (27%) and laths (20%) used as formwork, which is a low quality and short-term product used to mold concrete in construction. The category “construction” or long-term products (i.e., laths for framing, beams, scantlings, moldings, floors, boardbox, etc.) represent an additional 17%, and together constitute the 64% milling efficiency (Table S4 in the Supporting Information; Figure 2).

The remaining 36% is distributed among slabs and bark (13%), edges and off-cuts (13%), sawdust (8%), and shavings (2%; Table S4 in the Supporting Information), of which some became co-products. For example, sawmills reported selling 100% of edges and off-cuts to the furniture industry, 6.2% of slabs and bark is used to produce pellets for bioenergy, 4.6% for fuelwood, and only 2.6% of slabs and bark end at the mill dump (Table S5 in the Supporting Information). In the case of sawdust and shavings, the most important use is flooring for stables, stalls, or as organic matter used in nurseries (7% and 2%, respectively). Small amounts of sawdust and shavings were also used for pellets, fuelwood, or will be discarded (Table S5 in the Supporting Information).

Products and co-products flow to a next transformation stage or directly to the use phase of the life cycle (Figure 2). From the 2.89 Mg C from harvest, only 0.09 Mg C end at the mill dump. Given that slabs and bark represent over 80% of wood going to this dump and have a half-life of 20 years, only 0.02 Mg C (0.7% of harvest) are lost during the 15-year period, while 0.07 Mg C (3% of harvest) remain stored at the dump.

The transformation of products into end uses was limited to those used in construction or edges and off-cuts for the furniture industry. Out of the 0.49 Mg C of long-term and 0.37 Mg C of mid-term products that result from the milling process, 9.8% (SD = 7.29) or 0.05 and 0.04 Mg C , respectively, are sent to EoL management (Figure 2).

3.3 | Product use and end of life

During the 15-year period and given the 2-year half-life, 99.4% of all formwork has been transferred to EoL (Figure 3). However, 67% of construction wood and 60% of edges and off-cuts used in the furniture industry remain stored in the product pool. Carbon emissions during this phase

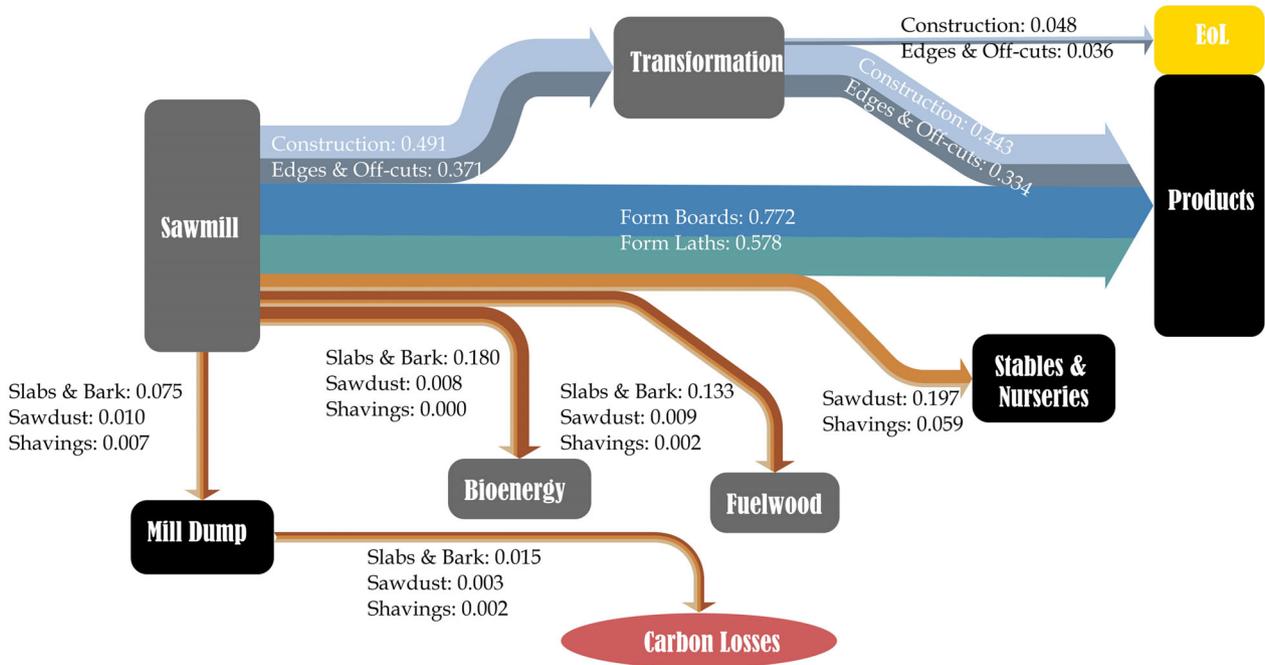


FIGURE 2 Carbon flows during sawmill processing and final product transformation of wood sourced from one hectare of natural forest in Costa Rica, during one logging cycle of 15 years ($\text{Mg C ha}^{-1} 15 \text{ year}^{-1}$; box sizes are not indicative of the size of the stock but black boxes represent carbon storage)

Note. Tabular data can be found in Table S4 in the Supporting Information.

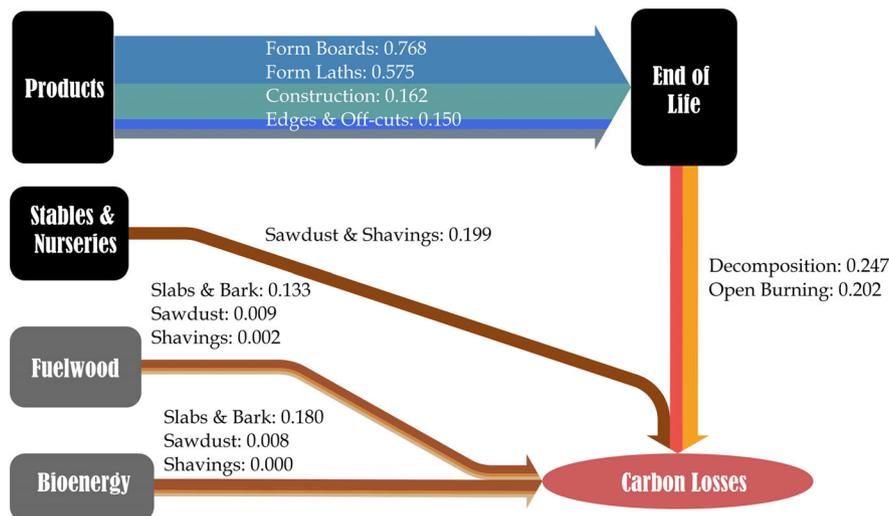


FIGURE 3 Carbon flows incurred by retirement (to EoL) and loss of wood products obtained from one hectare of exploited natural forest in Costa Rica, during one logging cycle of 15 years ($\text{Mg C ha}^{-1} 15 \text{ year}^{-1}$; box sizes are not indicative of the size of the stock but black boxes represent carbon storage)

corresponded to decomposition of sawdust and shavings used in stables and nurseries or the combustion of firewood and pellets, both assumed to have occurred during the year of harvest.

At SWDS, carbon emissions were determined by the fraction of degradable organic carbon that is effectively lost due to biomass decomposition (DOC_t) and the 20-year half-life assumed for all types of wood. As a result, from the 1.57 Mg C transferred to SWDS from products in use and the 0.09 Mg C from the final transformation shown previously, only 0.45 Mg C were lost during the 15-year period while 1.22 Mg C remained stored (Figure 3).

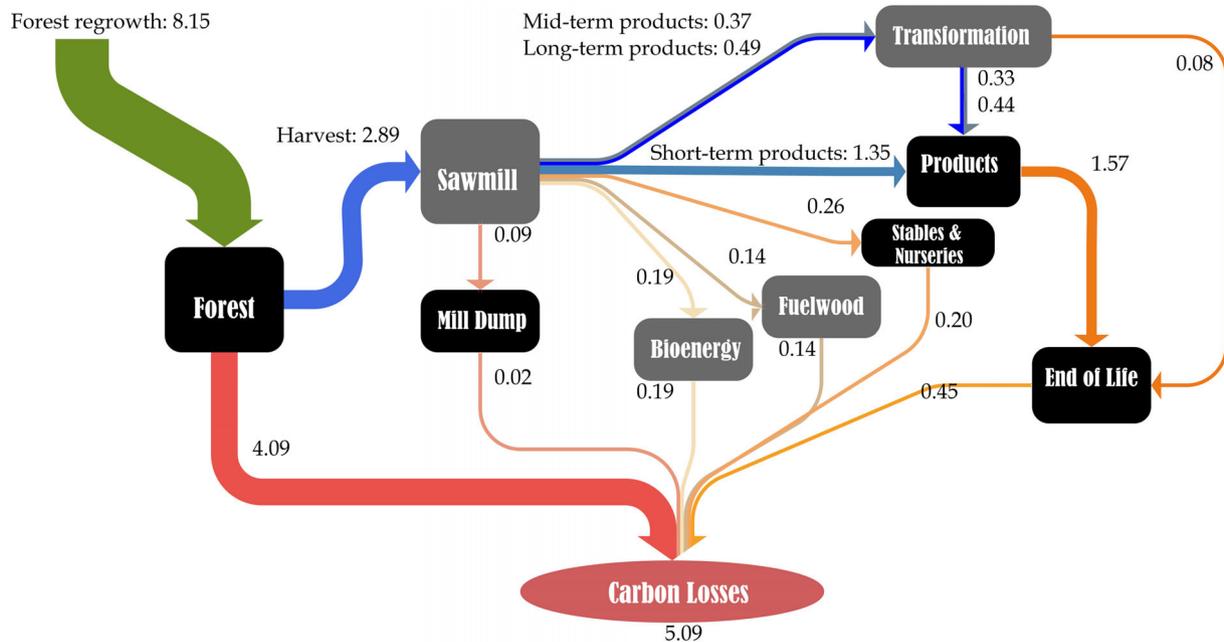


FIGURE 4 Carbon flow of wood products from one hectare of exploited natural forest in Costa Rica during a 15-year rotation period ($\text{Mg C ha}^{-1} 15 \text{ year}^{-1}$; box sizes are not indicative of the size of the stock but black boxes represent carbon storage)

3.4 | System balance

Life Cycle carbon emissions from the management of natural tropical forests for wood production in Costa Rica were $5.09 \text{ Mg C ha}^{-1} 15 \text{ year}^{-1}$ 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., $3.08 \text{ Mg C ha}^{-1}$) 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 $-3.06 \text{ Mg C ha}^{-1} 15 \text{ year}^{-1}$.

3.5 | Uncertainty and sensitivity analyses

The Monte Carlo simulations shifted the average carbon balance from -3.06 to $-2.19 \text{ Mg C ha}^{-1} 15 \text{ year}^{-1}$. This shift is due to the asymmetry of the distributions of some parameters (e.g., standing and deadwood harvest). The 95% confidence intervals of the carbon balance when taking parameter uncertainty and variation into account ranged from -5.26 to $1.86 \text{ Mg C ha}^{-1} 15 \text{ year}^{-1}$. This confidence interval includes the value of 0, implying that parameter variation can lead to carbon emissions from natural forest management. Yet, the probability of finding negative values is considerably larger, $\sim 80\%$.

Sensitivity analyses showed that carbon balance was most sensitive to rotation length (Figure S1 in the Supporting Information). All other things remaining equal, a longer rotation length resulted in higher carbon emissions due to its effect on retirement and decomposition rates. The decay rate of biomass at the forest (i.e., k_1) was the second most important parameter. This effect also shows the important role of carbon in necromass in the forest for the carbon balance after one logging cycle. The third most important parameter influencing the balance is the fraction of wood ending in SWDS (SWDS_f).

We found clear differences in the output of the sensitivity analysis when conducted for emissions and regrowth separately. For example, parameters such as harvest (H) and logging damage (gaps in particular) only affected regrowth despite H interacting with most parameters or logging damage being associated to carbon emissions. In both cases, an increase resulted in lower carbon emissions due to larger forest and anthropogenic reservoirs. In the case of the rotation period, it mainly affected emissions and had a marginal effect on regrowth under these circumstances.

None of the parameter changes in the sensitivity analysis resulted in a positive carbon balance. Yet, the results of the MC analysis showed that if multiple parameters are varied simultaneously, positive balance values can be obtained. Combined, this suggests that in scenarios with long logging cycles, high harvest intensity, high damage, high shares of short-term products, and/or low retirement to landfill, carbon balance will likely be positive. The cases with a positive carbon balance in our Monte Carlo simulations represent such (combinations) of variables.

4 | DISCUSSION

4.1 | The ecosphere meets the technosphere

This study presents a complete life cycle carbon balance for wood harvesting in the tropics following recommendations from the LCA framework. We find that indeed there are large probabilities for a carbon neutral outcome and confirm that it is at the forest where the largest exchanges of carbon occur (Butarbutar et al., 2016; Newell & Vos, 2012). Therefore, ignoring this phase from the life cycle of wood and biogenic carbon in general under the carbon neutral assumption is not justified (Geng et al., 2017; Klein et al., 2015; Knauf, 2015; Lippke et al., 2011). Especially considering the probabilities for the system to become a source of carbon emissions (Keith, Lindenmayer, Macintosh, & Mackey, 2015; Pioniot et al., 2016).

On the other hand, assuming committed emissions (i.e., the immediate release of carbon when harvesting takes place) is known to overestimate losses (Jordan, Hu, Arvesen, Kauppi, & Cherubini, 2018). In our study, this methodological difference largely explains why results for carbon emissions ($4.06 \text{ Mg C ha}^{-1}$ or 0.31 Mg C m^{-3} ; Table S3 in the Supporting Information) are well below those reported (Pearson et al., 2014) (i.e., $6.8\text{--}50.7 \text{ Mg C ha}^{-1}$ or $0.99\text{--}2.33 \text{ Mg C m}^{-3}$), although site-specific circumstances also play a role. For a better interpretation of these differences, we discuss three possible local practices that help to explain our results.

First, harvest intensity is known to determine forest carbon emissions (Martin, Newton, Pfeifer, Khoo, & Bullock, 2015; Putz et al., 2008). In our study, harvest intensity (13.03 m^3 or 2.89 t C ha^{-1} per logging cycle) is in the lower end of ranges reported in the literature (i.e., 10 to above $30 \text{ m}^3 \text{ ha}^{-1}$ and $1.5\text{--}8.5 \text{ t C ha}^{-1}$ (Pearson et al., 2014; Rutishauser et al., 2015; Sasaki et al., 2016), while logging damage is comparable with that from even lower reported harvest intensities (i.e., $9 \text{ m}^3 \text{ ha}^{-1}$ and 6.7 Mg C ha^{-1}) (Pearson et al., 2014). This is partly explained because in Costa Rica ~20% of the harvest is collected deadwood which does not require felling, and felling is the largest source of carbon emissions in a continuous cover harvesting system (i.e., 38–51% (Pearson et al., 2014); and 80% from this study). As a result, during the logging cycle (even if this is a relatively short one), carbon emissions are fully recovered in our study system.

Second, because of the small size of forest patches in Costa Rica (3.5–325 ha), logging hardly requires infrastructure such as primary roads and logging decks inside the forest. Furthermore, national standards for forest management (MINAE, 2002) require measures to reduce road impact consistent with those recommended in the literature (Laurance, Goosem, & Laurance, 2009). Most importantly, roads are closed once harvesting activities have taken place and are left for the forest to recover. This reduces the damage while increasing the contribution of gaps from felling on the overall damage.

Finally, different from the 50 cm threshold used in similar studies (Numazawa et al., 2017; Pearson et al., 2014; Pioniot et al., 2016), we assumed instead that trees $> 40 \text{ cm DBH}$ did not experience logging damage. This is based on the outcome of questionnaires to harvesting operators and foresters, who reported even lower tree sizes depending on the type of damage (i.e., between 10 and 30 cm). Consistent with our findings from these questionnaires, skidding together with cable winch as done in Costa Rica has been reported not to cause damage to trees $> 10 \text{ cm}$ (Griscom, Ellis, & Putz, 2014).

Clarifying these differences or assumptions is also important given that we estimate regrowth based on total biomass lost (Rutishauser et al., 2015). Despite the relatively low logging damage, the sum of harvest and damage represents 9% of the initial ecosystem carbon and is within the expected range (3–15%) (Pearson et al., 2014). Furthermore, as a measure of the conservativeness of the recovery rate used, the resulting mean annual increment of forest carbon stocks (i.e., $0.82 \text{ Mg C ha}^{-1} \text{ year}^{-1}$) is close to the lower end of those found in the Amazon, Borneo, and Nicaragua ($0.66\text{--}1.5 \text{ Mg C ha}^{-1} \text{ year}^{-1}$) (Rutishauser et al., 2015; Sasaki et al., 2016).

Finally, the amount of biomass damaged due to harvesting and left at the forest to decompose also represents an important temporal carbon stock at the end of the analyzed period. Regardless of efforts to correctly estimate damaged biomass, this stock depends on decay rates that are associated to large uncertainties (Pearson et al., 2014; Pioniot et al., 2016). In tropical forests, half-lives for biomass decay can vary from 1 to 69 years (Pearson et al., 2014) depending on the type of necromass. For this reason, other studies differentiate decay rates based on this (e.g., fine and large necromass) (de Rosa et al., 2017; Lun, Li, & Liu, 2012; Pioniot et al., 2016), but the largest uncertainties are mostly related to large necromass. To avoid overestimating this stock we used a conservative decay rate with a half-life of ~7 years, which is close to the average from the 0.6–14 year reported range (Hérault et al., 2010).

4.2 | The life cycle of biogenic carbon in the technosphere

A mass flow analysis such as the one used to trace carbon along the life cycle of wood has been recommended in the literature to consider the multifunctionality of wood and avoid misrepresenting its contribution in the overall balance (Geng et al., 2017; Jasinevičius et al., 2018). Essential for this analysis was the use of foreground data to determine all milling outputs (products, co-products, and residues) and estimate milling efficiency. By doing so, we were able to categorize wood products far beyond commonly used classifications (e.g., sawnwood, panels, pulp, and paper) and had more flexibility to assign specific half-lives until the EoL.

The relatively high milling efficiency (i.e., 63.67%) is the result of the main products, that is, boards and laths (46% of the total harvest), being used as formwork. This is a very low quality end product that allows the maximization of sawnwood use. Most reported milling efficiencies tend to

be around 50% (Butarbutar et al., 2016; Ofoegbu, Ogonnaya, & Babalola, 2014; Ramasamy, Ratnasingam, Bakar, Halis, & Muttiah, 2015; Sasaki et al., 2016), but our result is still within a range of 40–70% reported in the literature for tropical countries (Ofoegbu et al., 2014; Sasaki et al., 2016).

Despite the effect that formwork has on the milling efficiency, it was also because of the short half-life from this wood product that the stock of carbon from HWPs was relatively small, in accordance with the strong effect of retirement rates of wood products on carbon storage (Miner, 2010). In this work the carbon stock in products was largely determined by products with half-lives larger than a rotation period, and according to our sensitivity analysis there is no effect from prolonging this half-life. Since this is contrary to what has been repeatedly found in the literature (Brunet-Navarro et al., 2017), it is important to clarify that it is not small changes in half-life what affects the balance (e.g., $\pm 10\%$ used in the sensitivity) but a radical change in wood use (e.g., from formwork to mid- or long-term products).

The most important anthropogenic reservoirs delaying carbon emissions were solid waste disposal sites (SWDS). Few studies include this reservoir given the limited data (Clavreul et al., 2012) which was also the case in our study. To partly compensate for a potential overestimation of carbon allocated to landfills, we chose the 0.5 default value for all types of residues (Pipatti et al., 2006) as the fraction of carbon that will be lost (i.e., DOC_f) through anaerobic biomass decomposition. This fraction can vary from 0 to 0.65 (Barlaz, 2006; De la Cruz et al., 2013; Micales & Skog, 1997; Ximenes et al., 2015), and under tropical conditions, an average value of 0.18 has been reported (Ximenes et al., 2015). Therefore, it is very likely that our choice overestimates carbon emissions.

4.3 | Uncertainties and variation due to the choice of system boundaries

In life cycle studies the choice of system boundaries can be highly subjective and have a large effect on results (Geng et al., 2017; Klein et al., 2015; Knauf, 2015; Lippke et al., 2011; Newell & Vos, 2012). We made an attempt to avoid these decisions by including all major processes and by collecting foreground or local data as far as possible. By doing so, we reduced some of the model's uncertainty but increased parameter variability. Variability being the most common measure of uncertainty in LCA (Heijungs & Huijbregts, 2004). However, decisions regarding the reference unit to which the impact is attributed, that is, the functional unit, could not be avoided. In our study, this functional unit could be described as one hectare of natural tropical moist or wet forests in Costa Rica from which wood for various uses is harvested in 15-year rotation cycles.

Using a hectare as part of the functional unit is possible because we trace biogenic carbon exclusively and it is usually measured using this unit, especially at a regional level. It basically only allows the comparison with other forests (perhaps other land uses), and is therefore not frequently used in LCA where the common metric for wood products is cubic meter (Lippke et al., 2011). Its use can be further justified based on the goal of the analysis (de Rosa et al., 2017; Lippke et al., 2011; Perez-Garcia, Lippke, Cornick, & Manriquez, 2005), and has the additional benefit of avoiding part of the multifunctionality problem, that is, the allocation of impacts to products and co-products based on some allocation rule, mass being the most common (Sandin, Peters, & Svanström, 2016).

The most critical aspect of this unit is the choice of the analyzed period. Long-term processes associated to forestry have always conflicted life cycle studies (Ter-Mikaelian, Colombo, & Chen, 2015) because these rely on the assumption of a static system (Clavreul et al., 2012). We followed the “whole rotation approach” (Klein et al., 2015) given that it provides a time frame that allows the inclusion of regrowth, decomposition, and carbon storage until a next cycle and because it is the most common in the LCA of forestry (Ter-Mikaelian et al., 2015). However, this approach is not entirely free from criticisms.

Given that it assumes that the forest has never been logged before, it is subject to what has been termed the “start-up effect” (Miner, 2006). Wood will continue to be retired from the system, combusted, or will decompose in the years following this rotation, and these emissions are not accounted for, thus leading to an overestimation of carbon storage during the first rotation. To correct for this effect, methods used in HWP inventories estimate “inherited emissions” (i.e., carbon emissions from previous harvests) (Pingoud, Skog, Martino, Tonosaki, & Xiaoquan, 2006) while other proposed methods recommend estimating the existing stocks after 100 years (Miner, 2006). The main advantage from this approach is that reliable data on previous harvests can be difficult to obtain.

To present the most conservative estimate for the carbon balance, we recalculated the balance and its uncertainty for a 100-year period. This resulted in a lower average $-1.36 \text{ Mg C ha}^{-1} 100 \text{ year}^{-1}$ where the 95% CI $[-4.43, -0.14]$ is always negative. Due to a longer rotation period, logging damage is almost fully decomposed but there is still some carbon stored in products and mainly in SWDS. Most importantly, this longer period reduces chances of forests not recovering the initial carbon.

The main contribution from this work has been to show the importance of biogenic carbon and the effects of expanding the system boundaries to include all major processes in the life cycle of tropical timber, something that was lacking in the literature (Murphy, 2004; Numazawa et al., 2017; Piponiot et al., 2016). Our results provide evidence for the hypothesis that managed forests could potentially contribute more to climate change mitigation than unmanaged forests (Lundmark, Bergh, Nordin, Fahlvik, & Poudel, 2016), although this remains highly controversial and opposing evidence is also available. Converting managed forests to protected areas has been shown to lead to higher carbon accumulation (Keith et al., 2015), especially when the reference scenario involves high harvest intensities and logging damage. In any case, under circumstances where forests are being harvested, it is a combination of short logging cycles, low harvesting intensities and high mass allocation into long-term products what has the greatest probabilities of avoiding some carbon emissions (Liu et al., 2017).

5 | CONCLUSIONS

According to our analysis, selective logging in a 15-year cycle with subsequent timber use, may delay biogenic carbon emissions due to the storage of carbon in forests and anthropogenic reservoirs; allowing the forest to recover before the next cycle of use. However, forest management may also act as a disturbance leading to an acceleration of carbon emissions, for example, through higher harvesting intensities with high logging damage, leading to insufficient recovery time until the next logging event, or by allocation of wood to short-term uses. When considering carbon storage, low impact logging and long-term product use are crucial.

Our results imply that forest management and subsequent use of wood products may indeed contribute to total carbon storage, also when considering harvest and wood processing losses. Decisive factors in that case are low-intensity and low-impact selective logging, efficient wood processing, and allocation of wood to product categories that have substantially long life-spans. In addition, end-of-life is important, and final allocation of used wood products to landfill may comprise an important storage component, although the final allocation to landfill may be unwanted for other reasons and should be reconsidered for re-use or use for bioenergy in which fossil fuels are replaced. This allows sustainable forest management combined with efficient product use to contribute to carbon storage, while a continued resource use adds to valuation of forested land, and thereby supports conservation of the forest resource.

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CONFLICT OF INTEREST

The authors have no conflict to declare.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

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