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Carbon emissions and potential emissions reductions from low-intensity selective logging in southwestern Amazonia

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ABSTRACT

Forests in southwestern Amazonia are increasingly being converted for agriculture, mining, and infrastructure development; subjected to low-intensity selective logging of high value timber species; and designated as conservation areas and indigenous reserves. To understand the impacts of forestry in this region, we evaluated carbon emissions from felling, skidding, and hauling in five FSC-certified concessions where workers were trained in reduced-impact logging (RIL) and in four non-certified concessions where workers were not trained in RIL in Madre de Dios, Peru. Emissions estimates did not differ by certification status, so we established a single baseline for selective logging emissions. Total carbon emissions from selective logging were low per hectare (4.9–11.6 Mg ha^{-1}) due to low logging intensities (2.9–8.1 m^3 ha^{-1}). Despite the unique architecture of trees in the southwestern Amazon (short stems and large crowns), emissions per volume and per ton carbon in the extracted timber were also relatively low (1.55 Mg m^{-3} and 4.04 Mg Mg^{-1}, respectively). Only emissions per area scaled with logging intensity. Emissions were dominated by the felled tree itself (in extracted logs and residuals), whereas hauling infrastructure (roads and log landings) contributed comparatively little. Unintended emissions could be reduced by 46% if concessions were able to achieve the best demonstrated outcomes in each source category and by 54% with additional improvements. Less than 5% of timber was lost due to hollow sections. We determined that it would be overly cautious to avoid cutting all trees with any hollow sections, and it would actually increase emissions per unit timber extracted if no other trees were cut in place of the hollow trees. At the tree level, certified concessions had higher log recovery and damaged fewer commercial species during felling, which should increase their current and future timber yields. It is important to both understand and improve carbon dynamics in managed forests in this emerging hotspot for greenhouse gas emissions from deforestation and forest degradation.

1. Introduction

Globally, approximately 11% of annual net greenhouse gas emissions and 14% of carbon emissions are from forestry and other land uses, mostly in developing tropical and subtropical countries (Goodman and Herold, 2014). Reducing tropical deforestation and degradation have long been considered important to reduce global carbon emissions, but the contribution of forest degradation has only recently been quantified over large scales (Berenguer et al., 2014; Baccini et al., 2017; Erb et al., 2017). Forest degradation reportedly accounts for one quarter (Pearson et al., 2017) to over two thirds (Baccini et al., 2017) of all forest emissions in tropical countries. In Central and South America, half (Hosonuma et al., 2012) to two thirds (Pearson et al., 2017) of degradation emissions are from logging. Both reducing carbon emissions from forestry operations and ensuring sustainability of forest management are important to address given that over half of all remaining tropical forests are dedicated to wood production (Blaser et al., 2011).

Working with forest management enterprises (e.g., concessions and community-based forest management) has the potential to improve conservation outcomes by reducing degradation through improved harvest practices and reducing deforestation by creating profitable business models based on retaining forests as forests (Griscom and Goodman, 2015). Natural forest management and avoided forest conversion are both high-potential and low-cost “natural climate solutions” to mitigate

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climate change (Griscom et al., 2017). At the same time, unnecessarily destructive logging is a major form of degradation that generally precedes and can even promote deforestation (Asner et al., 2006).

Due to high diversity and the marketability of timber from only a few tree species, logging in the tropics is typically selective but can nevertheless result in substantial carbon emissions. Carbon losses from selective logging come from clearing for haul roads and log yards, collateral damage around felled trees and on skid trails, crop tree residuals (i.e., branches, stumps, and sections of the stem left in the forest, etc), and the life cycle of the extracted wood itself. By employing what have become known as reduced-impact logging (RIL) practices (Putz and Pinard, 1993), these emissions can be substantially reduced (Johns et al., 1996; Pinard and Putz, 1996; West et al., 2014). Recommended RIL practices include planning harvest operations (e.g., mapping and marking commercial trees; planning roads, log yards, and skid trails; and using directional felling techniques to avoid damage to future crop trees and streams), cutting lianas on trees to be harvested at least 6 months before felling, and following guidelines on tree felling and skidding (Pinard et al., 1995; Dykstra and Heinrich, 1996). RIL guidelines typically also include practices intended to reduce the biodiversity and hydrological impacts of logging and to improve worker safety, but we here focus on only those practices likely to reduce carbon emissions (RIL-C; Griscom et al., 2014). In this study we also disregard the substantial post-logging carbon benefits of RIL such as increased rates of carbon stock recovery (Lincoln, 2008; Vidal et al., 2016).

Despite the large amount of research conducted on RIL across the tropics (FAO, 2004), we are aware of no studies on RIL in Peru. Given that logging contributes substantially to Peru's carbon emissions, which the country has committed to reduce (MINAM, 2015), we conducted this study to establish a baseline from which improvements can be measured. Furthermore, while Peru is globally known for its deforestation (Asner et al., 2010; Hansen et al., 2013; Robiglio et al., 2014) and problems with illegal logging (Gutierrez-Velez and MacDicken, 2008; Finer et al., 2014), the country also hosts efforts to improve forest management practices that deserve attention.

Among the interventions intended to promote responsible forest management in general and RIL in particular, Forest Stewardship Council (FSC) certification looms large. Since its founding in 1993, numerous studies reported on FSC's impacts but few were designed to avoid positive selection biases, ignored contextual changes of likely importance, and suffered from other deficiencies such as small sample sizes (Romero et al., 2017; Komives et al., 2018). In a meta-analysis of this literature, Blackman et al. (2017) found no conclusive evidence that forest certification has positive environmental outcomes. In another meta-analysis that disregarded the quality of the included studies, Burivalova et al. (2017) reported that the environmental outcomes from certified and RIL forest management (including C emissions) were better than conventional management in 76% of case studies (worse in 6% and no difference in 18%). However, this and other studies also concluded that the benefits of RIL decline when logging intensity is taken into account because certified concessions and those that claim to employ RIL practices tend to harvest at lower intensities than their uncertified and conventionally logged counterparts (Medjibe et al., 2013; Griscom et al., 2014; Martin et al., 2015; Burivalova et al., 2017).

The Peruvian Amazon is an important focal geography where many global issues are currently at play. Peru has fourth highest area of forest cover in tropics, and nearly half of the country's 68 million ha are classified as permanent production forest (Blaser et al., 2011). The southwestern Amazon and specifically the MAP region (Madre de Dios, Peru–Acre, Brazil–Pando, Bolivia) is undergoing rapid deforestation and forest degradation following completion of the Interoceanic Highway (Baraloto et al., 2015; Alarcón et al., 2016), and deforestation and degradation of natural forests represents almost half of Peru's greenhouse gas emissions (MINAM, 2013). Legal and illegal logging has already degraded much of the natural forest in the Peruvian Amazon (Asner et al., 2010), and the legal timber industry is expanding (Cossio et al., 2014).

We employed the methods of Griscom et al. (2014) to estimate carbon emissions from forestry concessions in Madre de Dios, Peru. Specifically, our objectives were to: (i) establish a baseline for carbon emissions from selective logging in the region; (ii) assess whether RIL training associated with FSC certification reduced carbon emissions; and (iii) estimate potential emissions reductions through RIL-C. Because of the low logging intensity (Cossio et al., 2014) and distinctive architecture of trees in southern Peruvian forests (relatively short stems and large crowns; Goodman et al., 2014), we hypothesized baseline emissions per ha to be lower and emissions per unit timber extracted to be higher than the tropical average. As field staff in all certified concessions were trained in RIL while those employed by non-certified concessions were not, we expected emissions from certified concessions to be lower while fully recognizing that any differences cannot be attributed to certification due to other differences among the concessions and lack of a counterfactual design.

2. Methods

2.1. Site and socio-economic description

Our study was conducted within Tahuamanu Province of Madre de Dios, Peru. Forests here are broadly classified as lowland, moist, *terra firme* forest (Whitmore, 1998; Achard et al., 2002), and "bamboo-dominated" forests are common in this region (Carvalho et al., 2013). Mean annual temperature is 24.5 °C; mean annual precipitation is 1811 mm with a 3–4 month dry season (Hijmans et al., 2005). The area is relatively flat with medium gradient hills and elevation ca. 250–375 masl (FAO et al., 1998).

We evaluated nine annual cutting blocks or forest management units (FMUs) from six different concessions: 5 FMUs from three FSC certified concessions and 4 FMUs from three non-certified concessions. All concessions have management plans and operated under governmental oversight. Workers in all FSC certified concessions received RIL training through World Wildlife Fund (WWF)–Peru in 2008 on pre-harvest inventory and skid trail mapping, pre-harvest liana cutting, directional felling, improved bucking, and plunge cuts to test for hollowness. All FSC-certified concession managers were trained to conduct inventories of all crop trees, road building best management practices, and GIS-based haul road planning.

All certified concessions were highly vertically integrated (level 3 in Bray et al., 2006), owned sawmills, and marketed sawn timber primarily to international markets. The non-certified concessions sold either standing trees or roundwood with little or no added value processing (levels 1 and 2 in Bray et al., 2006) to domestic markets only. Logging was subcontracted in one certified concession (to family members of the former concession owners) and in one non-certified concession. All concessions paid workers monthly salaries with the exception of special jobs with daily wages. Worker retention, especially of chainsaw and skidder operators, was problematic because of poor living conditions, low wages, and more profitable opportunities in the region (personal communication). Workers regularly left after one or two seasons to work in the gold mining industry. Within the forestry sector, concession managers tried to hire workers who were already trained elsewhere. In one certified concession, some workers were retained during the wet season to work in the sawmill. Forestry engineers, who plan and direct the harvests, often stayed longer, especially in certified concessions.

Timber harvest and extraction took place during the dry season of each year, *ca*. May–October. All concessions were in their first cutting cycle, with the exception that some mahogany (*Swietenia macrophylla*) trees were selectively harvested before 2000.

2.2. Field methods

We conducted field work in 2014 based on methods in Griscom et al. (2014) and Pearson et al. (2014). We tracked the length of all

Table 1
Sample size (n) and mortality rates (1 year after logging event), assumed aboveground (AG) emissions in trees that survive (as proportion of original AG carbon), and AG and belowground (BG) emissions scenarios.*

Damage class	n	Mortality rate	AG emissions in survivors	Emissions scenarios	
				AG	BG
G: uprooted, laying on ground	666	0.854	1.00	1.000	0.854
S: trunk Snapped below first branch	265	0.509	1.00	1.000	0.509
L: Leaning ≥ 10° from vertical	191	0.136	0.00	0.136	0.136
C: ≥50% of crown lost	914	0.115	0.33	0.407	0.115
B: ≥100 cm^2 of bark lost	122	0.074	0.00	0.074	0.074

* For trees uprooted and those with trunks snapped below the first branch (G and S), we assumed 100% aboveground losses. For trees with bark damage and those that were leaning, we assumed no aboveground carbon loss from trees that survived and 100% loss for trees that die; thus, proportion of aboveground emissions = mortality rate. For trees with ≥ 50% crown damage, we estimated that 44% of aboveground biomass is in the tree crowns (Goodman et al., 2013, 2014) and 75% of crowns were lost. Thus, we expect 33% of aboveground emissions from trees that survive (88.5%) and 100% carbon loss from the 11.5% of trees that die (0.33 × 0.885 + 1.00 × 0.115 = 0.407). For all damage categories, we assume that all roots will be lost when the tree dies and no roots will be lost if the tree survives; thus, we used mortality rates for belowground emissions scenarios.

roads in each cutting block with handheld GPS units. At 15 points separated by 200 m, we measured the width of both the active surface (area compacted by vehicles) and total road corridor (i.e., perpendicular distance between trees > 10 cm dbh; stem diameter at 1.3 m or above buttresses). While tracking roads, we counted all log landings in each cutting block and measured up to 10 log landings per concession (some had < 10). We recorded log landing dimensions and shapes outside of the active road surface.

Skid trail networks were sampled by randomly selecting a distance from where the primary access road enters the FMU. We started the skid trail evaluation at the closest skid trail to this point and followed all branches. All subsequent skid trails evaluated were on the same side of the road in a randomly selected direction. We tracked 2–3 km of skid trails in each cutting block with a GPS and marked the stumps of every tree felled by harvest crews whether or not any timber was removed.

We assessed skidding impacts in 15 10-m long plots located every 100 m along the mapped skid trails. For all damaged vegetation with dbh ≥ 10 cm, we recorded dbh and damage classes (as defined by Griscom et al., 2014): toppled below 1.3 m; cut above 1.3 m but below the crown; > 50% crown loss; > 100 cm^2 of bark removed; and unnatural lean > 10°. In concessions harvested in 2013 (i.e., one year prior to our sample), we also recorded whether each damaged tree had survived or died (Table 1). We assumed that all trees < 10 cm dbh were completely destroyed. Since there was no evidence of soil surface disturbance by skidder blades, we assumed that skid trails were 3 m wide—the most common width of skidder blades in this area.

To estimate timber volumes harvested (and associated extracted log emissions) and the residual biomass of trees felled for harvest (crop tree residuals), we evaluated 15–19 felled trees per FMU, taken as every other tree in the skid trail network evaluated. We measured stump heights, dbh of the felled tree[1], length of the extracted log, diameter at the base and top of the extracted log (top of stump and base of crown, respectively), length and width of crown, and dimensions of any abandoned logs. In the case of hollow sections, we measured length and diameter of the cavity at the base and top of the hollow section.

We evaluated felling collateral damage around each of the measured felled trees described above. We measured dbh and recorded life form, damage category, and survival (as per skid trail damage assessments) on all affected woody vegetation with dbh ≥ 5 cm.

2.3. Emissions estimates

Our committed emissions estimates include carbon in above- and belowground biomass but not in the soil; belowground biomass was estimated as 0.235 × AGB (Mokany et al., 2006) and carbon content as 47% of dry mass (IPCC 2006). Emissions from roads and log landings were estimated for entire FMUs as the product of area cleared (total road corridors) and mean carbon density in this forest type. All concession representatives interviewed maintained that their tractor operators avoid trees ≥ 40 cm dbh, so we estimated C emissions from carbon density of trees < 40 cm dbh as 62.32 Mg ha^{-1} (Goodman et al., 2012) and assumed that no timber was extracted during road construction.

Collateral damage from skidding and felling were estimated in two steps: (1) For each damaged tree, we estimated its original above- and belowground C; and (2) we estimated the proportion of C lost in each damage category in both above- and belowground tree components. First, we estimated aboveground biomass (AGB) of all damaged trees using Goodman et al. (2014) model II.1 with dbh and wood density (0.563 g cm^{-3}; mean wood density in plots within Madre de Dios, Peru (Baker et al., 2004)). Second, we estimated above- and below-ground carbon losses as the product of live tree carbon stocks and mortality rate (Table 1). Because damaged trees may die in later years (Putz and Brokaw, 1989; Shenkin et al., 2015), we consider our 1-year mortality estimates to be conservative. We calculated skid trail emissions as the sum of emissions from damaged trees ≥ 10 cm dbh per m of skid trail and C density of smaller vegetation (8.45 Mg ha^{-1}; Goodman et al., 2012).

Crop tree AGB was estimated using Goodman et al. (2014) model I.1CR with dbh, height, wood density, and crown radius. Commercial log mass was estimated using Smalian's formula. Crop tree residuals were all above- and belowground biomass carbon minus the carbon in the commercial log (if extracted).

We calculated C emissions per ha, per m^3 of timber extracted, and per Mg C in extracted timber. Carbon impact factor (CIF) is the latter (emissions in Mg Mg^{-1}) excluding emissions from extracted timber itself (which is, by definition, 1 Mg Mg^{-1}). Thus, CIF can be considered the unintended carbon emissions. Hauling emissions (roads and log yards) were assessed for entire cutting blocks whereas all other activities were assessed only within the sampled skid trail network. Since the majority of emissions came from non-hauling sources, we focused our metrics on the area sampled in the skid trail network and scaled hauling emissions down to this area. Because there are extensive areas where trees are not harvested for no apparent reason (Ellis et al., 2016), our scaling-down factor is the ratio of extracted timber in our sampled skid trail network to the total volume of timber reported by for each FMU (Table 2).

2.4. Statistical analysis

We compared emissions from each of the six sources (i.e., extracted

[1] Since logs were usually removed, we measured dbh at the top of the butt log (typically above buttresses). If only stumps < 1.3 m height remained, we used dbh reported in the pre-harvest inventories.

Table 2
Concession and forest management unit (FMU) characteristics: Timber harvests reported in the entire FMU and measured in the skid trail network sample area; scaling factor to convert emissions from road and log landings of the whole FMU to sampled area; areas of the entire concession, FMU, and sample area; harvest intensity as trees and volume of timber extracted per ha; and volume (vol.) extracted as a percent of volume authorized for the corresponding FMU.

		Timber harvested (m^3)			Area (ha)			Harvest intensity		Vol. extracted/
FMU	Year logged	FMU	Sample area	Scaling factor*	Concession	FMU	Sample area**	Trees ha^{-1}	m^3 ha^{-1}	Vol. authorized (%)
Certified										
1	2014	9036	428	0.047	46,505	2,435	115.3	0.33	3.71	53
2a	2013	6028	304	0.050	45,974	1,883	95.0	0.22	3.20	–
2b	2014	2660	327	0.123		532	65.4	0.37	5.00	62
3a	2013	9519	286	0.030	49,370	3,247	97.5	0.31	2.93	15
3b	2014	37,101	477	0.013		6,725	86.4	0.57	5.52	26
Non-certified										
4	2013	5659	273	0.048	14,621	789	38.0	0.71	7.17	51
5	2013	1680	318	0.189	5,905	295	55.8	0.38	5.70	52
6a	2013	1671	124	0.074	19,267	299	22.3	0.81	5.58	45
6b	2014	2795	332	0.119		344	40.9	0.83	8.13	45
Means (SE)										
Certified					47,283 (1055)	2964 (1039)	91.9 (8.1)	0.36 (0.06)	4.07 (0.51)	39 (11)
Non-certified					13,264 (3917)	432 (129)	39.2 (6.9)	0.68 (0.11)	6.64 (0.61)	48 (2)
Overall					30,274 (7820)	1839 (707)	68.5 (10.6)	0.50 (0.08)	5.22 (0.58)	44 (5)

* Scaling factor = (Timber in sample area)/(Timber reported for FMU).
** Sample area = (Area of FMU) × (Scaling factor).

log, crop tree residuals, felling collateral damage, skidding, roads, and log landings) in certified/RIL trained/vertically integrated and non-certified/not RIL trained/not vertically integrated concessions (hereafter "certified" and "non-certified") using t-tests and analysis of covariance (ANCOVA) with harvest intensity as a covariate ($n = 9$). We treated all FMUs as replicates (i.e., independent), even when they occurred within the same concession. We justify the assumption of independence on the basis of the FMUs being logged in different years and —since there is very high turnover of personnel—by different crews. We also evaluated whether certification status affected emissions from felled trees (extracted log and residuals) and from felling collateral damage using individual trees as replicates (84 in certified and 67 in non-certified). We used t-tests and linear regression (lm in R) with certification as a dummy variable to test for differences. Likewise, we related collateral damage from felling, crop tree residuals, and carbon in the extracted log to aboveground biomass of the felled tree using linear regression with certification status as a dummy variable. Baseline committed emissions per ha, per volume of timber extracted, and per Mg C in the extracted timber were determined using linear regression with harvest intensity (m^3 ha^{-1}) as the independent variable (Griscom et al., 2014) and certification status as a dummy variable. Non-significant terms were removed until a minimum adequate model was reached. Thus, if certification was not significant, we combined all data to establish baseline emissions. We verified the assumptions of linear regression using normal Q-Q plots and the Anderson-Darling test for normality; homogeneous variance and linearity were evaluated by plotting residuals against fitted values; and data were transformed when necessary. All analyses were performed in R, version 3.5.0.

2.5. Potential emissions reductions

To estimate the potential gains from application of RIL-C practices, we assessed "RIL-C" emissions reductions potential at two levels of implementation. Level 1 avoidable emissions are the best demonstrated outcomes from each source category and RIL-C practice among the nine cutting blocks analyzed. Level 2 avoidable emissions are feasible higher performance levels based on professional judgement and outcome analysis (see Table 3).

As part of our Level 2 analysis, we examined the contributions of unharvested boles that were felled and abandoned due to heartrots and hollows. We first estimated the wood volume left in the forest due to hollowness as all the stem wood (butt and commercial logs) cut and abandoned with hollowness: volume lost due to hollowness = volume of abandoned wood × length of hollow section/total log length. We then looked for relationships between the cavity size at the point where a plunge cut would be performed and losses of timber due to hollowness. Since there were no detectable relationships between cavity diameter, diameter of the solid portion of the log, or proportion of hollowness compared to total or relative amount of wood lost (Figs. A1–A3), we could not formulate any guidelines for avoiding the felling of partially hollow trees. In light of these results, we carried out a theoretical analysis in which no hollow trees were felled. In this analysis there were no emissions from the hollow trees that were not felled, no felling collateral damage, and no skid trails to those trees; likewise, no timber was extracted from these trees.

3. Results

3.1. Logging practices

All FMUs were logged at low intensities (2.9–8.1 m^3 ha^{-1}) and harvested only 15–62% of government-authorized volumes (Table 2). All five certified FMUs harvested at lower intensities than the four non-certified FMUs in terms of trees ($p = 0.045$) and volumes ($p = 0.017$) per ha. Felled trees were on average 101.8 (standard error 2.3) cm dbh and 38.6 (0.6) m tall. Mean extracted logs were 15.6 (0.3) m long, 11.3 (0.6) m^3, and 4.3 (0.3) Mg C. Certified concessions were much larger than non-certified concessions ($p = 0.009$). FMUs also ranged in size (295–6725 ha) and tended to be larger in certified concessions (not significant). Certified FMUs also had wider haul roads than non-certified FMUs (total corridor means = 17.9 (1.0) and 12.7 (4.2) m, respectively), but these differences were not significant. All concessions principally harvested *Dipteryx micrantha* (locally known as "shihuahuaco"), a dense-wooded Fabaceae. This species comprised 55% of all trees harvested, 72% of timber volumes, and 76% of C exported. In one FMU, over 98% of C extracted was in *D. micrantha.*

3.2. Committed emissions

Total carbon emissions per hectare were greater in non-certified

Table 3
Explanation of Levels 1 and 2 RIL-C emission reduction scenarios.

Logging activity category	RIL practices	Emissions category	Level 1 implementation	Level 2 implementation
Felling and log recovery	Pre-harvest inventory, plunge cut, planned skidding	Emissions from trees felled and abandoned.	CIF of abandoned felled trees in best recorded FMU (0 in all but but two FMUs)	No further reduction
	Directional felling to reduce collateral damage	Emissions from collateral damage of surrounding forest during felling	CIF of collateral damage in best recorded FMU (0.40 in P2a)	Assuming 5% reduction in collateral damage from liana cutting during pre-harvest inventory
	Improved bucking & log recovery	Emissions from trees felled with some volume extracted (not incl abandoned)	CIF of felled tree (log + remainders) in best recorded FMU (2.04 in P3b)	Assuming 10% increase of log recovery
Skidding:	Skid trail planning, long-line winching	Emissions from mortality resulting from skidding damage	CIF of skid trails in best recorded FMU (0.16 in P6b)	Reducing skid trail length to each tree by 30 m (mean skid trail length is 138 m $tree^{-1}$, so 22% reduction)
Hauling:	Narrower haul roads	Emissions from mortality resulting from clearing road corridors	CIF of best recorded FMU (0.05 in P4; mean active road 2.5 m wide & total road corridor 7.2 m wide)	No further reduction

FMUs —10.30 (0.65) vs. 6.23 (0.91) Mg ha^{-1}; $p = 0.008$. Using conservative estimates of forest carbon stocks,[2] only 3–9% of forest carbon stocks were lost from the ecosystem (mean 6.3%). Emissions per unit timber (volume and C) were slightly lower in certified FMUs in each source category (i.e. crop tree residuals, collateral damage, etc) except roads (Fig. 1), but differences between certified and non-certified FMUs were never significant. Thus, total emissions per unit timber extracted were slightly lower in certified FMUs but not significantly different from non-certified FMUs—1.53 (0.09) vs 1.59 (0.17) Mg m^{-3} and 3.93 (0.20) vs. 4.18 (0.54) Mg Mg^{-1} in certified and non-certified, respectively. Only four logs were abandoned of which 3 were hollow throughout the bole. Hollowness only marginally lowered harvest volumes, and primarily manifested as cutting longer butt logs so that all commercial logs were solid. On average, only 0.21 (0.08) m^{-3} ha^{-1} or 4.8 (1.7) % of timber of merchantable size was left in the forest due to hollowness, primarily in butt logs.

The majority of emissions (per Mg C extracted) for al FMUs came from crop trees themselves (extracted log + crop tree residuals) whereas logging infrastructure (skid trails + log yards + roads) contributed little (Fig. 1). On average, nearly two thirds of emissions (in Mg Mg^{-1}) come from the crop trees: 40% as residuals left in forest and 25% removed in logs. Of the total emissions, felling collateral damage accounted for only 14%, skid trails 11%, roads 8%, and log yards 0.5%. Excluding emissions from harvested timber, then relative contribution of crop tree residuals increases to 54% of CIF, felling collateral damage to 20%, skid trails to 15%, roads to 11%, and log yards to 0.6%. In the language of Pearson et al. (2014), mean (standard error) extracted log emissions (ELE) was 0.39 (0.01) Mg m^{-3}, logging damage factor (LDF; felling collateral damage + crop tree residuals) was 0.84 (0.06) Mg m^{-3}, and logging infrastructure factor (LIF; roads + log yards + skid trails) was 0.32 (0.06) Mg m^{-3}.

Training in directional felling and improved bucking shows some evidence of reducing damage and increasing log recovery. At the FMU level, certified FMUs did not have significantly lower collateral damage in terms of C emissions, but they damaged less than half as many residual commercial species as non-certified FMUs (0.60 vs. 1.33 per felled tree; $p = 0.043$). When felled trees were examined individually, CIF of felling collateral damage averaged lower in certified FMUs (0.54 vs. 0.78; $p = 0.028$). Compared to AGB of the crop tree, mass of extracted logs was greater in certified FMUs than in non-certified FMUs (Fig. 2).

Baseline or total predicted committed emissions (emissions vs. logging intensity) did not differ between certified and non-certified FMUs. Only emissions expressed per hectare varied with harvest intensity, and these baseline emissions are estimated as a function of harvest intensity (Fig. 3A). Baseline carbon emissions per unit timber volume and carbon were independent of harvest intensity and estimated as mean values for all concessions combined: 1.55 (0.09) Mg m^{-3} and 4.04 (0.25) Mg Mg^{-1} (Fig. 3B and C).

3.3. Potential emissions reductions

There was fairly wide variation in emissions from each source among the 9 FMUs evaluated (Fig. 1). No one FMU consistently performed best (i.e., had lowest CIF in all source categories; Table 3), but one FMU did have the highest (worst) CIF values in most source categories. Level 1 potential emissions reductions were determined by combining the lowest six CIF values (one from each source category) from five different FMUs, and Level 2 implementation scenarios are explained in Table 3. Potential emissions reductions are larger from the

[2] Carbon density of trees and palms with dbh ≥ 10 cm for forests with bamboo in the southwestern Amazon (Table 4.2 in Goodman, R.C., 2013. Tropical Tree and Palm Allometry and Implications for Forest Carbon Dynamics in Southwestern Amazonia. In, School of Geography. University of Leeds, Leeds, UK, p. 213.)

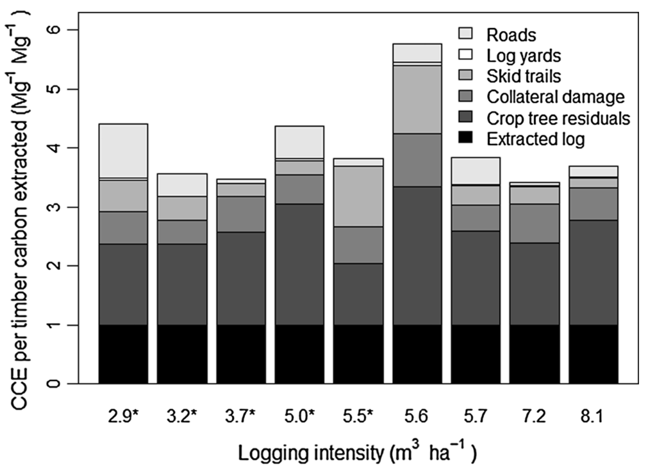

Fig. 1. Total committed carbon emissions (CCE) per carbon in extracted timber (Mg Mg^{-1}) from each source in each forest management unit. By definition, Mg Mg^{-1} of the extracted log = 1.

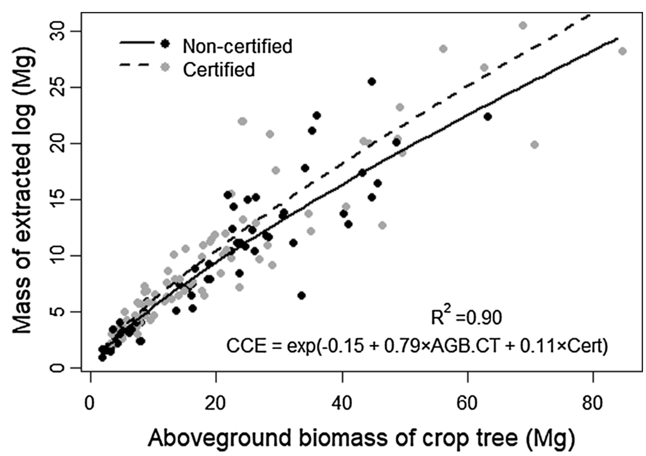

Fig. 2. Dry mass of the extracted log vs. aboveground biomass of the crop tree (AGB.CT) and certification status (Cert; 0 if non-certified and 1 if certified). Certified concessions tended to extract higher proportions of the felled tree mass.

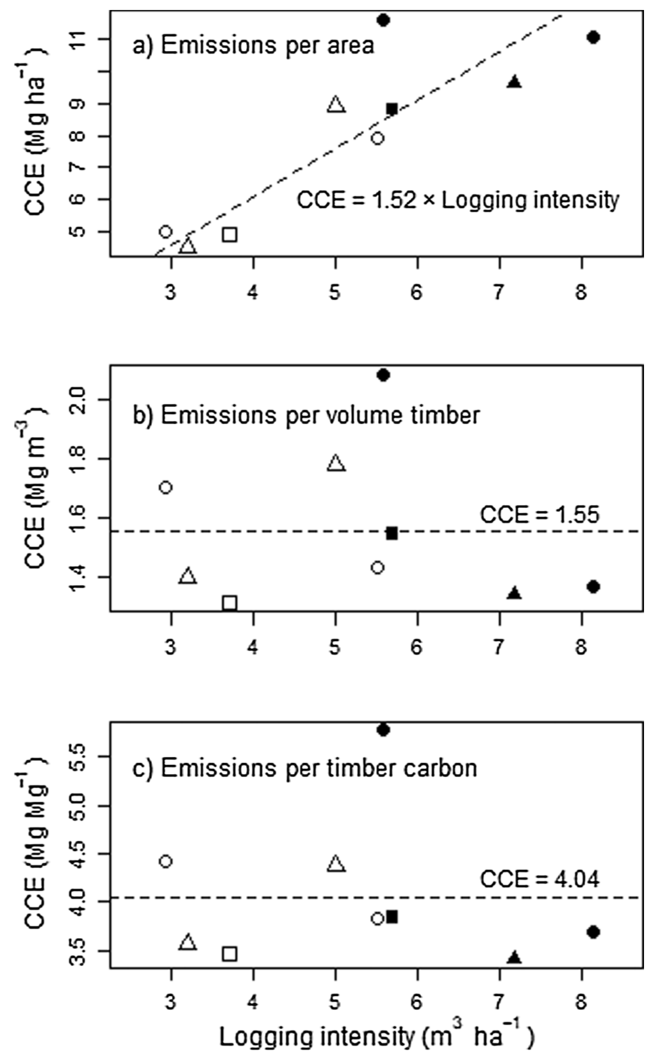

Fig. 3. Baseline committed carbon emissions (CCE) for nine FMUs in Tahuamanu Province, Madre de Dios, Peru reported per ha (a), per volume of timber extracted (b), and per carbon mass extracted in timber (c). Only CCE per ha (a) varies with logging intensity. Hollow symbols are certified forest management units (FMUs) and solid symbols are non-certified FMUs. Same symbols represent FMUs within the same concession. Dotted lines show predicted total emissions.

baseline to Level 1 than from Level 1 to the theoretical improvements of Level 2 (Fig. 4).

Mean CIF (the "unintended" emissions) could potentially be reduced by more than half, from 3.04 to 1.65 or 1.41 with Level 1 and 2 RIL-C implementation, respectively. The greatest potential for reduction is improved log recovery (which simultaneously increases the denominator and decreases crop tree residuals), followed by skid trail planning and cable-winching, directional felling to reduce collateral damage, and narrowing haul roads. There are potential emissions reductions from decreasing the number of abandoned trees (to zero) and making smaller and fewer log landings, but so few emissions come from these sources that overall emissions reductions are nearly negligible.

In our theoretical scenario —in which no trees with hollowness at the point where chainsaw operators would perform a plunge cut were felled— showed that, if no replacement trees were cut, logging intensity would be reduced from 2.9 to 8.1 (mean 5.2 (0.6)) m^3 ha^{-1} to 0.8–7.9 (mean 4.4 (0.7)) m^3 ha^{-1} (Fig. A5). Under this scenario, mean emissions per hectare would be reduced from 8.04 (0.90) to 6.69 (0.94) Mg ha^{-1}, but mean total CIF would increase from 3.04 to 3.38 (0.47) due to reductions in timber extracted from several FMUs (Fig. 4). This result was driven primarily by a dramatic increase in CIF from roads in one FMU with a hypothetical harvest intensity < 1 m^3 ha^{-1} and by the fact that CIF of crop tree residuals decreased only marginally. Since the observed FMU with the lowest CIF of crop tree residuals felled zero hollow trees, potential emissions reductions from never cutting hollow trees are already included in our Level 1 RIL-C implementation.

4. Discussion

4.1. Logging practices and emissions baselines

Harvest intensities for all concessions in our study were much lower than reported for other tropical forests, with the exception of other parts of the southwestern Amazon (Rutishauser et al., 2015) and Gabon (Medjibe et al., 2013). As found elsewhere (e.g., Blackman et al., 2018), certified concessions were much larger than non-certified concessions, which is no surprise given that the costs of certification (Ruslandi et al., 2014) would be prohibitive for small concessions.

Like prior studies, concessions or FMUs practicing RIL harvested at lower intensities and consequently had lower emissions per ha (Pinard and Putz, 1996; Medjibe et al., 2013; Martin et al., 2015; Vidal et al.,

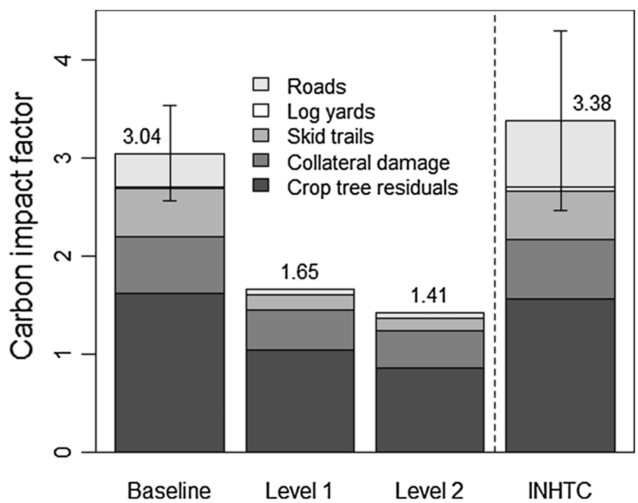

Fig. 4. Four scenarios for carbon impact factor (CIF; Mg C emitted per Mg C in timber extracted; Mg Mg^{-1}) of selective logging in Amazonian Peru: Baseline (mean observed), Level 1 (minimum observed CIF of each source), Level 2 (Level 1 + theoretical reductions), and baseline (mean) if no hollow trees were cut (INHTC). See Table 3 for explanations of Levels 1 and 2. Bars show 95% confidence limits of total CIF.

2016). As observed in Indonesia (Griscom et al., 2014), there was no difference in carbon emissions between certified and non-certified concessions once logging intensity was taken into account (Fig. 3). In contrast to Indonesia (Griscom et al., 2014), our data showed no evidence that efficiency of logging increased with harvest intensity (in terms of Mg m^{-3} or CIF; Fig. 3B and C). This pattern may change with more samples that cover a wider range of harvest intensities, as indicated in the theoretical scenario in which no hollow trees were harvested (Fig. A4).

Our hypothesis that the distinct tree architecture and low logging intensity in our study area would cause logging operations to have higher than average carbon emissions was not supported. Compared to emissions reported in the first pantropical review of the topic (in Mg m-3; Pearson et al., 2014), extracted log emissions were high in Peru (mean 0.39 vs. 0.25–0.38 Mg m^{-3}), indicating that timber species in Peru have denser wood; logging infrastructure emissions were at the low end of reported values (0.32 vs. 0.24–0.98 Mg m^{-3}); and logging damage factor was average and similar to values for other South American countries (0.84 vs. 0.71–1.23 Mg m^{-3} in Bolivia, Brazil, and Guyana). Volumes and masses of extracted logs in Peru were much smaller than in Republic of Congo and Indonesia; larger than in Belize, Bolivia, and Guyana; and nearly identical to Brazil (Pearson et al., 2014). Compared to trees in Pearson et al. (2014)'s Brazilian study, mean dbh of felled trees in Peru was larger, mean log length was shorter, and percent of felled tree extracted was exactly the same (43%). In the most recent pantropical analysis (Ellis et al., 2019), Peru has the absolute lowest CIF. CIF was over three times greater in Gabon than in Peru. In strong contrast to Peru, roads were the largest source of emissions in the countries with the highest CIFs from logging (Gabon and Republic of the Congo; Ellis et al., 2019).

The vast majority of emissions from the selectively logged forests we studied in Amazonian Peru were from the crop trees themselves (commercial timber plus residuals), which cannot be changed by improved practices. That said, metrics that describe committed emissions relative to the amount of timber extracted are highly sensitive to the partitioning of the crop tree and hence benefit from improved felling and bucking practices. Furthermore, we considered all the carbon in extracted logs to be committed emissions due to lack of site-specific data and because the proportion of wood that end up in long-lived wood products is extremely low (Lauk et al., 2012), especially in the tropics (Earles et al., 2012). We did not consider milling efficiency or product use but fully advocate improving timber recovery from extracted logs and the use of long-lived forest products.

4.2. Potential emissions reductions

A CIF of 3.04 means that for every ton of carbon extracted in commercial timber, over three times that amount is lost from the forest in the process (in addition to the 1 Mg in the timber itself). Reducing CIF from 3.04 to 1.65 (Level 1) would mean a reduction in "unintended" C losses by almost half compared to current practices. It could be argued that choosing the lowest CIF in each category produces unrealistic potential emissions reductions, as those values may have resulted from unique circumstances. However, the best CIF values from each source were never far from the second or third lowest values, so we believe that Level 1 emissions are achievable with careful training, planning, practice, and supervision. Level 2 potential emissions reductions are admittedly more aspirational but have been implemented in other locations.

RIL detractors often referred to it as "reduced-income logging", but our results show that the best way to reduce CIF is to increase yields from the trees cut for that purpose. Specifically, increasing the recovery of timber extracted both increases the denominator and decreases crop tree residuals, even though there is no reduction in overall carbon emissions. Timber recovery could be increased by trimming buttresses to utilize more of the stem and extracting very large branches as timber. This change in practices might sound simple, but there are logistical, technological, and regulatory constraints. For example, the irregularity of branches and stems with buttresses make transportion more difficult and reduce milling efficiency. More research is needed on the wood properties of branches and buttresses, as they likely differ from stem wood of the same species. In terms of regulations in Peru, forest transport permits are issued for a given species, location, and estimated timber volumes from inventory data on dbh, stem height, and stem form. If greater volumes were authorized (e.g., for the extraction of branches), then it would open the system to further corruption, since the documents could be sold to loggers felling trees illegally elsewhere (Finer et al., 2014). It is thus unclear how changing these regulations would affect the already prevalent contribution of legal logging to illegal logging (Finer et al., 2014).

For Level 2 potential emissions reductions, we propose introducing cable winching for the last 30 m of skid trails, though the emissions reductions from that change of practice are small. There is theoretically potential to reduce haul road emissions by 87% (from baseline to Level 1) by reducing road lengths and widths, but very narrow roads might not be favorable for practical reasons such as drying. Log yards account for < 1% of total emissions, so they provide few options for emissions reductions. There are also modest potential emission reductions from directional felling but in Peru, as elsewhere, the first priority is worker safety and the second is avoidance of damage to future crop trees, neither of which may translate into much carbon retention. CIF of felling collateral damage is on average 16% less in certified concessions where workers were trained in directional felling, but this difference was not significant and when compared at the FMU level or when individual crop tree biomass was related to collateral damage. Given the mortality rates observed in this study and prolonged mortality of all trees damaged from logging (Sist et al., 2014; Shenkin et al., 2015), we suggest that to reduce emissions and maintain stand structure loggers avoid damaging large trees in any way, even scraping bark. In the interest of sustaining timber yields, avoiding future crop trees should probably be prioritized. Liana cutting in advance of felling should decrease collateral damage, improve worker safety, and increase post-logging rates of stand recovery (Putz, 1991).

4.3. Forest management, degradation, and deforestation

There is a fine line between forest management and forest degradation through selective logging. One side of that line represents the argument that forest management that is economically viable and ecologically sound helps protect forests from conversion; the other side argues that logging degrades forests and facilitates deforestation. Thus, extensive low-intensity selective logging can lead to among the best or worst outcomes for forest carbon stocks and biodiversity conservation (Griscom et al., 2018).

One definition of degradation is the "reduction in the overall capacity of a forest to supply goods and services including carbon storage, climate regulation, and biodiversity conservation" (Berenguer et al., 2014). Here we measured only the maintenance of forest carbon stocks, but this may be an indicator of carbon stock recovery rates as well. In studies that spanned the Amazon Basin, the proportion of forest C loss during logging operations was found to be the best predictor of forest C recovery time, and a 10% C loss is expected to recover in < 20 years (Rutishauser et al., 2015). In our study, < 10% of forest carbon stocks were lost from selective logging, even with a conservative estimate of forest carbon stocks (Goodman, 2013). Thus, we would expect C stocks to recover in all FMUs before the next allowed harvest in 20 years. However, the recovery of carbon stocks does not equate to the recovery of commercial timber (West et al., 2014).

Contrary to broad calls to reduce logging intensities in tropical forests (Sist et al., 2003; Burivalova et al., 2017; Romero and Putz, 2018), logging intensity in Madre de Dios seems to be low enough. Harvest intensities in all concessions were well below what has been recommended as sustainable: ≤ 8 trees ha^{-1} in Guyana (Roopsind et al., 2018) and 3–4 trees or 10–14 m^3 ha^{-1} with 40 year rotations in eastern Brazil (Sist and Ferreira, 2007). However, sustainability depends on the structure of the residual stand (Sist and Ferreira, 2007), and sustainable forest management in "bamboo-dominated" forests of the southwestern Amazon is particularly difficult due to the scarcity of future crop trees (Rockwell et al., 2014). Furthermore, the strong dependence of forest industries in the region on large *Dipertyx* trees is probably not sustainable. Forestry concessions will likely have to shift species in subsequent rotations, as found across the tropics (Putz et al., 2012) and recommended for this forest type in particular (Rockwell et al., 2014). In a model simulation of eastern Amazonian forests, timber yields could be sustained for 2 and 3 cutting cycles but the composition shifted from high-value, shade-tolerant species, like *Dipteryx*, towards lower-value but faster growing species (Macpherson et al., 2012). We would expect the same future in Peru.

From a carbon perspective, the greatest contribution of forest certification is not the reduction in emissions from forestry operations through RIL but the reduction in deforestation (Griscom et al., 2018). Certification has been found to reduce deforestation in some cases (Miteva et al., 2015) but not always (Blackman et al., 2018). At least in the early days of the Interoceanic Highway, forestry concessions were effective at resisting deforestation in Madre de Dios (Chávez Michaelsen et al., 2013), and securing land tenure dramatically decreased deforestation rates across Peru (Hajek and Che Piu, 2016). However, the fate of forest concessions after their 40-year contracts end is unknown. It is important to manage these forests sustainably because illegal or unplanned logging often degrades forests and catalyzes deforestation (Asner et al., 2006; Pinheiro et al., 2016).

In Peru, the "S" in FSC is often assumed to stand for as "Sustainable" rather than "Stewardship", and concession managers themselves may be mistaken about the nature of their operations (e.g., certification indicates that they are sustainable and have zero net carbon emissions over the length of the rotation). In our experience in Madre de Dios, FSC certification benefits worker safety and treatment, but the issue of sustainability is not fully addressed. This is not a problem unique to Peru. As found in the eastern Amazon, regulations nor RIL ensure sustainability (Macpherson et al., 2012). FSC does address/require sustainable timber yields (criterion 5.6), but sustainability is difficult to enforce or even predict. This phenomenon is troublesome since depleted and abandoned forests are subject to conversion (Romero and Putz, 2018). In our study, certified concessions damaged fewer commercially important species during felling, which is a metric that should be emphasized during certification.

4.4. Limitations and future research

Our emissions estimates from skidding and felling collateral damage may be underestimates because trees that initially re-sprout often die in subsequent years (Putz and Brokaw, 1989) and damaged trees continue to die years after logging (Sist et al., 2014; Shenkin et al., 2015). On the other hand, cumulative mortality rates reported for resprouts and trees with "other major damage" after 8 years (Shenkin et al., 2015) were lower than our 1-year mortality rates for highly damaged trees (snapped and uprooted). In any case, we have advanced methods to account for carbon emissions from collateral damage by accounting for both immediate C losses from crown damage and initial mortality of trees with "minor" damage (e.g., bark damage or slight leaning). We suggest more in-depth studies on long-term mortality rates from logging damage. Our data suggest that there may be different mortality rates within each damage class depending on whether the damage occurred during skidding or felling, but we lacked sufficient data to test this idea or to develop separate mortality rates.

In this study, we tracked only biomass carbon from selective logging and did not consider soil carbon losses, fuel use during extraction or transport, or any other carbon emission sources. There are few data on the effect of selective logging on soil carbon, but a recent assessment reported no effect (Berenguer et al., 2014). This finding aligns with our observations that there was very little soil disturbance from logging operations, except on roads and log landings, which occupy a very small proportion of the landscape. Future studies might consider soil carbon losses due to erosion from logging roads, which can be can be several meters deep.

While roads contribute relatively little to forest biomass C losses from logging (this study) and affect only a small portion of the land area (median 1.7% across tropics; Kleinschroth and Healey, 2017), their secondary effects are far-reaching. In particular, roads fragment forests and increase access to previously remote areas, thereby potentially increasing the occurrence of hunting, invasive species, in-migration, conversion to agriculture, and fires (Kleinschroth and Healey, 2017). These deleterious effects of logging and roads on biodiversity can be reduced by post-logging road closure, along with other recommended road-related RIL practices (Bicknell et al., 2014) and controls on logging intensity (Burivalova et al., 2014).

Complete assessments of the effects of selective logging carbon emissions should also take into account how logging increases the likelihood and severity of forest fires. Selective logging increases fire frequency and intensity by increasing fuel loads (collateral damage and crop tree residuals) and altering the microclimate (e.g., elevated temperature and desiccation in forest canopy gaps; Holdsworth and Uhl, 1997; Cochrane and Laurance, 2008). Forest fires release large quantities of carbon (Withey et al., 2018) and may reduce forest biomass and timber stocks for decades due to high tree mortality (Silva Camila et al., 2018). Fire is thus troublesome from both climate and timber yield sustainability perspectives. Wildfires and the link between fires and logging are expected to intensify with climate change and the associated droughts (Cochrane and Laurance, 2008; Withey et al., 2018). Thus, logging practices that reduce the risks and intensities of fire (Holdsworth and Uhl, 1997) deserve attention and may even have synergies with reducing CIF in the short term.

The results of our hollow tree analysis were unexpected and not fully conclusive. First, we found no clear way to predict the vertical extent of heart rots and hollows in standing trees or the amount of timber lost due to hollowness. Several trees with large hollow sections near the ground still yielded substantial quantities of commercial timber from upper parts of their boles. It is possible to improve predictions of the severity of hollowness before felling (Kennard et al., 1996), and it is likely that the chainsaw operators in our study areas have already done so. Indeed, of the 29 felled trees we measured with some hollowness, only three were entirely unusable. However, we did not collect data on the number of trees tested for hollowness by chainsaw operators with plunge cuts or otherwise. This is an interesting topic, as learning to not cut trees with little or no useable timber not only reduces carbon emissions, it also reduces time, fuel waste, and wear-and-tear on equipment while retaining trees that are important for stand structure and wildlife. Most importantly, felling hollow trees is especially dangerous for workers (Conway, 1976). On the other hand, deciding not to fell trees with any sign of hollowness would be overly cautious and actually increase CIF in many FMUs primarily because it reduces timber yields while logging infrastructure remains the same—if no replacement trees are felled in lieu of those skipped due to hollowness. The possibility of replacement is plausible in Peru, where concessions harvest much less than they are authorized and that is available, but this may not be the case in other forests. This issue of cutting hollow trees should be explored in more detail and is likely to become more important in future rotations, as the proportion of hollow trees tends to increase with each harvest while timber stocks decrease (FEP, pers. obs.).

5. Conclusions and recommendations

From a carbon perspective, Madre de Dios in the southwestern Amazon has among the best carbon outcomes from selective logging of all tropical countries studied. Nonetheless, three times more carbon is emitted from logging operations than is extracted, and this amount could be reduced by half through RIL-C harvesting practices. FSC certification was not created specifically to reduce carbon emissions, and we find little evidence that it does so in Peru. We call for a clearer link between certification, RIL, RIL-C, and sustainability of timber yields. We also suggest that timber companies in the region increase the number species they harvest, assist natural regeneration of desired timber species, and protect and release future crop trees. Finally, we note that RIL training is somewhat futile unless the trained workers remain in the forestry industry and suggest that companies improve financial incentives, living conditions, and employee treatment to increase retention. Managers and forestry engineers trained and dedicated to RIL-C can also emphasize the importance of these practices daily and create a culture of RIL-C within each logging camp.

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.foreco.2019.02.037.

References

Achard, F., Stibig, H.J., Eva, H., Mayaux, P., 2002. Tropical forest cover monitoring in the humid tropics: TREES project. Trop. Ecol. 43, 9–20.

Alarcón, G., Díaz, J., Vela, M., García, M., Gutiérrez, J., 2016. Deforestación en el sureste de la amazonia del Perú entre los años 1999–2013; caso Regional de Madre de Dios (Puerto Maldonado – Inambari). Revista de Investigaciones Altoandinas 18, 12.

Asner, G.P., Broadbent, E.N., Oliveira, P.J.C., Keller, M., Knapp, D.E., Silva, J.N.M., 2006. Condition and fate of logged forests in the Brazilian Amazon. PNAS 103, 12947–12950.

Asner, G.P., Powell, G.V.N., Mascaro, J., Knapp, D.E., Clark, J.K., Jacobson, J., Kennedy-Bowdoin, T., Balaji, A., Paez-Acosta, G., Victoria, E., Secada, L., Valqui, M., Hughes, R.F., 2010. High-resolution forest carbon stocks and emissions in the Amazon. PNAS 107, 16738–16742.

Baccini, A., Walker, W., Carvalho, L., Farina, M., Sulla-Menashe, D., Houghton, R.A., 2017. Tropical forests are a net carbon source based on aboveground measurements of gain and loss. Science 358, 230–234.

Baker, T.R., Phillips, O.L., Malhi, Y., Almeida, S., Arroyo, L., Di Fiore, A., Erwin, T., Killeen, T.J., Laurance, S.G., Laurance, W.F., Lewis, S.L., Lloyd, J., Monteagudo, A., Neill, D.A., Patino, S., Pitman, N.C.A., Silva, J.N.M., Martinez, R.V., 2004. Variation in wood density determines spatial patterns in Amazonian forest biomass. Global Change Biol. 10, 545–562.

Baraloto, C., Alverga, P., Quispe, S.B., Barnes, G., Chura, N.B., da Silva, I.B., Castro, W., da Souza, H., de Souza Moll, I.E., Del Alcazar Chilo, J., Linares, H.D., Quispe, J.G., Kenji, D., Marsik, M., Medeiros, H., Murphy, S., Rockwell, C., Selaya, G., Shenkin, A., Silveira, M., Southworth, J., Vasquez Colomo, G.H., Perz, S., 2015. Effects of road infrastructure on forest value across a tri-national Amazonian frontier. Biol. Conserv. 191, 674–681.

Berenguer, E., Ferreira, J., Gardner, T.A., Aragão, L.E.O.C., De Camargo, P.B., Cerri, C.E., Durigan, M., Oliveira, R.C.D., Vieira, I.C.G., Barlow, J., 2014. A large-scale field assessment of carbon stocks in human-modified tropical forests. Glob. Change Biol. 20, 3713–3726.

Bicknell, J.E., Struebig, M.J., Edwards, D.P., Davies, Z.G., 2014. Improved timber harvest techniques maintain biodiversity in tropical forests. Curr. Biol. 24, R1119–R1120.

Blackman, A., Goff, L., Rivera Planter, M., 2018. Does eco-certification stem tropical deforestation? Forest Stewardship Council certification in Mexico. J. Environ. Econ. Manage. 89, 306–333.

Blackman, A., Raimondi, A., Cubbage, F., 2017. Does forest certification in developing countries have environmental benefits? Insights from Mexican corrective action requests. Int. Forest. Rev. 19, 247–264.

Blaser, J., Sarre, A., Poore, D., Johnson, S., 2011. Status of Tropical Forest Management 2011. In: ITTO Technical Series 38. International Tropical Timber Organization, Yokohama, Japan.

Bray, D.B., Antinori, C., Torres-Rojo, J.M., 2006. The Mexican model of community forest management: the role of agrarian policy, forest policy and entrepreneurial organization. For. Pol. Econ. 8, 470–484.

Burivalova, Z., Hua, F., Koh Lian, P., Garcia, C., Putz, F., 2017. A critical comparison of conventional, certified, and community management of tropical forests for timber in terms of environmental, economic, and social variables. Conserv. Lett. 10, 4–14.

Burivalova, Z., Sekercioglu, C.H., Koh, L.P., 2014. Thresholds of logging intensity to maintain tropical forest biodiversity. Curr. Biol. 24, 1893–1898.

Carvalho, A.L.D., Nelson, B.W., Bianchini, M.C., Plagnol, D., Kuplich, T.M., Daly, D.C., 2013. Bamboo-dominated forests of the southwest Amazon: detection, spatial extent, life cycle length and flowering waves. PLoS ONE 8, e54852.

Chávez Michaelsen, A., Huamani Briceño, L., Fernandez Menis, R., Bejar Chura, N., Valera Tito, F., Perz, S., Brown, I., Domínguez Del Aguila, S., Pinedo Mora, R., Alarcón Aguirre, G., 2013. Regional deforestation trends within local realities: land-cover change in southeastern Peru 1996–2011. Land 2, 131.

Cochrane, M.A., Laurance, W.F., 2008. Synergisms among fire, land use, and climate change in the Amazon. SPIE.

Conway, S., 1976. Logging Practices: Principles of Timber Harvesting Systems. Miller Freeman Publications, San Francisco.

Cossio, R., Menton, M., Cronkleton, P., Larson, A., 2014. Community forest management in the Peruvian Amazon: A literature review. In: Working Paper 136. CIFOR, Bogor, Indonesia.

Dykstra, D., Heinrich, R., 1996. FAO Model Code of Forest Harvesting Practice. Food and Agriculture Organization of the United Nations, Rome, Italy.

Earles, J.M., Yeh, S., Skog, K.E., 2012. Timing of carbon emissions from global forest clearance. Nat Clim Change 2, 682–685.

Ellis, P.W., Griscom, B., Walker, W., Gonçalves, F., Cormier, T., 2016. Mapping selective logging impacts in Borneo with GPS and airborne lidar. For. Ecol. Manage. 365, 184–196.

Ellis, P.W., Gopalakrishna, T., Goodman, R.C., Roopsind, A., Griscom, B.W., Umunay, P. M., Zalman, J., Ellis, E., Mo, K., Gregoire, T., Putz, F.E., 2019. Climate-effective reduced-impact logging (RIL-C) can halve selective logging carbon emissions in tropical forests. For. Ecol. Manage.

Erb, K.-H., Kastner, T., Plutzar, C., Bais, A.L.S., Carvalhais, N., Fetzel, T., Gingrich, S., Haberl, H., Lauk, C., Niedertscheider, M., Pongratz, J., Thurner, M., Luyssaert, S., 2017. Unexpectedly large impact of forest management and grazing on global vegetation biomass. Nature 553, 73.

FAO, 2004. Reduced impact logging in tropical forests: Literature synthesis, analysis and prototype statistical framework. In: Food and Agriculture Organization of the United Nations. Forest Harvesting and Engineering Programme. Working Paper No. 1, Rome, Italy.

FAO, ISRIC, UNEP, CIP, 1998. Soil and terrain database for Latin America and the Caribbean (v2.0), 1:5M. scale (CD-ROM). In: Land and Water Digital Media Series 5. FAO, Rome.

Finer, M., Jenkins, C.N., Sky, M.A.B., Pine, J., 2014. Logging concessions enable illegal logging crisis in the Peruvian Amazon. Sci. Rep. 4, 4719.

Goodman, R.C., 2013. Tropical Tree and Palm Allometry and Implications for Forest Carbon Dynamics in Southwestern Amazonia. In: School of Geography. University of Leeds, Leeds, UK, p. 213.

Goodman, R.C., Baker, T.R., Phillips, O.L., 2012. Part 1: Analysis of data and methodologies of the biomass plots installed in Madre de Dios. Consultancy to establish the technical guidelines to complete a carbon baseline and design a measurement and monitoring system for carbon stocks in the region of Madre de Dios, Peru. University of Leeds, Leeds, UK.

Goodman, R.C., Herold, M., 2014. Why maintaining tropical forests is essential and urgent for a stable climate. CGD Working Paper 385. Washington, DC: Center for Global Development.

Goodman, R.C., Phillips, O.L., Baker, T.R., 2013. Data from: The importance of crown dimensions to improve tropical tree biomass estimates. In: Dryad Data Repository.
Goodman, R.C., Phillips, O.L., Baker, T.R., 2014. The importance of crown dimensions to improve tropical tree biomass estimates. Ecol. Appl. 24, 680–698.
Griscom, B., Ellis, P., Putz, F.E., 2014. Carbon emissions performance of commercial logging in East Kalimantan, Indonesia. Glob. Change Biol. 20, 923–937.
Griscom, B.W., Adams, J., Ellis, P.W., Houghton, R.A., Lomax, G., Miteva, D.A., Schlesinger, W.H., Shoch, D., Siikamäki, J.V., Smith, P., Woodbury, P., Zganjar, C., Blackman, A., Campari, J., Conant, R.T., Delgado, C., Elias, P., Gopalakrishna, T., Hamsik, M.R., Herrero, M., Kiesecker, J., Landis, E., Laestadius, L., Leavitt, S.M., Minnemeyer, S., Polasky, S., Potapov, P., Putz, F.E., Sanderman, J., Silvius, M., Wollenberg, E., Fargione, J., 2017. Natural climate solutions. Proc. Natl. Acad. Sci. 114, 11645–11650.
Griscom, B.W., Goodman, R.C., 2015. Reframing the sharing vs sparing debate for tropical forestry landscapes. J. Trop. For. Sci. 27, 145–147.
Griscom, B.W., Goodman, R.C., Burivalova, Z., Putz, F.E., 2018. Carbon and biodiversity impacts of intensive versus extensive tropical forestry. Conserv. Lett. 11, e12362.
Gutierrez-Velez, V.H., MacDicken, K., 2008. Quantifying the direct social and governmental costs of illegal logging in the Bolivian, Brazilian, and Peruvian Amazon. For. Pol. Econ. 10, 248–256.
Hajek, F., Che Piu, H., 2016. Analysis of the status of administrative acts related to the reduction of forest cover (authorizations for forest land use change and titling processes). Adapted from MINAM, 2016. National Forest Strategy and Climate Change and Climate Investment Funds, 2013. FIP Investment Plan for Peru.
Hansen, M.C., Potapov, P.V., Moore, R., Hancher, M., Turubanova, S.A., Tyukavina, A., Thau, D., Stehman, S.V., Goetz, S.J., Loveland, T.R., Kommareddy, A., Egorov, A., Chini, L., Justice, C.O., Townshend, J.R.G., 2013. High-resolution global maps of 21st-century forest cover change. Science 342, 850–853.
Hijmans, R.J., Cameron, S.E., Parra, J.L., Jones, P.G., Jarvis, A., 2005. Very high resolution interpolated climate surfaces for global land areas. Int. J. Climatol. 25, 1965–1978.
Holdsworth, A.R., Uhl, C., 1997. Fire in Amazonian selectively logged rain forest and the potential for fire reduction. Ecol. Appl. 7, 713–725.
Hosonuma, N., Herold, M., De Sy, V., De Fries, R.S., Brockhaus, M., Verchot, L., Angelsen, A., Romijn, E., 2012. An assessment of deforestation and forest degradation drivers in developing countries. Environ. Res. Lett. 7.
Johns, J.S., Barreto, P., Uhl, C., 1996. Logging damage during planned and unplanned logging operations in the eastern Amazon. For. Ecol. Manage. 89, 59–77.
Kennard, D.K., Putz, F.E., Niederhofer, M., 1996. The predictability of tree decay based on visual assessments. J. Arboric. 22, 249–254.
Kleinschroth, F., Healey, J.R., 2017. Impacts of logging roads on tropical forests. Biotropica 49, 620–635.
Komives, K., Arton, A., Baker, E., Kennedy, E., Longo, C., Newsom, D., Pfaff, A., Romero, C., 2018. Conservation impacts of voluntary sustainability standards. How has our understanding of the conservation impacts of voluntary sustainability standards changed since the 2012 publication of “Toward sustainability: The roles and Limitations of certification?”. In. Meridian Institute, Washington, DC.
Lauk, C., Haberl, H., Erb, K.-H., Gingrich, S., Krausmann, F., 2012. Global socioeconomic carbon stocks in long-lived products 1900–2008. Environ. Res. Lett. 7 034023.
Lincoln, P.R., 2008. Stalled gaps or rapid recovery: the influence of damage on post-logging forest dynamics and carbon balance. University of Aberdeen.
Macpherson, A.J., Carter, D.R., Schulze, M.D., Vidal, E., Lentini, M.W., 2012. The sustainability of timber production from Eastern Amazonian forests. Land Use Policy 29, 339–350.
Martin, P.A., Newton, A.C., Pfeifer, M., Khoo, M., Bullock, J.M., 2015. Impacts of tropical selective logging on carbon storage and tree species richness: a meta-analysis. For. Ecol. Manage.
Medjibe, V.P., Putz, F.E., Romero, C., 2013. Certified and uncertified logging concessions compared in gabon: changes in stand structure, tree species, and biomass. Environ. Manage. 51, 524–540.
MINAM, 2013. Iniciativas para reducir las emisiones de carbono. In: Peru Ministerio del Ambiente, Lima.
MINAM, 2015. Estrategia Nacional ante el Cambio climático, Ministerio del Ambiente. Lima, Perú.
Miteva, D.A., Loucks, C.J., Pattanayak, S.K., 2015. Social and environmental impacts of forest management certification in indonesia. PLoS ONE 10 e0129675.
Mokany, K., Raison, R.J., Prokushkin, A.S., 2006. Critical analysis of root: shoot ratios in terrestrial biomes. Glob. Change Biol. 12, 84–96.
Pearson, T.R.H., Brown, S., Casarim, F.M., 2014. Carbon emissions from tropical forest degradation caused by logging. Environ. Res. Lett. 9.
Pearson, T.R.H., Brown, S., Murray, L., Sidman, G., 2017. Greenhouse gas emissions from tropical forest degradation: an underestimated source. Carbon Balance Manage. 12, 3.
Pinard, M.A., Putz, F.E., 1996. Retaining forest biomass by reducing logging damage. Biotropica 28, 278–295.
Pinard, M.A., Putz, F.E., Tay, J., Sullivan, T.E., 1995. Creating timber harvest guidelines for a reduced-impact logging project in Malaysia. J. Forest. 93, 41–45.
Pinheiro, T.F., Escada, M.I.S., Valeriano, D.M., Hostert, P., Gollnow, F., Müller, H., 2016. Forest degradation associated with logging frontier expansion in the Amazon: The BR-163 region in southwestern Pará, Brazil. Earth Interact 20, 1–26.
Putz, F.E., 1991. Silvicultural effects of lianas. In: Putz, F.E., Mooney, H.A. (Eds.), The Biology of Vines. Cambridge University Press, Cambridge, UK, pp. 493–501.
Putz, F.E., Brokaw, N.V.L., 1989. Sprouting of broken trees on Barro Colorado Island, Panama. Ecology 70, 508–512.
Putz, F.E., Pinard, M.A., 1993. Reduced-impact logging as a carbon-offset method. Conserv. Biol. 7, 755–757.
Putz, F.E., Zuidema, P.A., Synnott, T., Peña-Claros, M., Pinard, M.A., Sheil, D., Vanclay, J.K., Sist, P., Gourlet-Fleury, S., Griscom, B., Palmer, J., Zagt, R., 2012. Sustaining conservation values in selectively logged tropical forests: the attained and the attainable. Conserv. Lett. 5, 296–303.
Robiglio, V., Armas, A.D., Silva Aguad, C., White, D., 2014. Beyond REDD+ readiness: land-use governance to reduce deforestation in Peru. Clim. Pol. 14, 734–747.
Rockwell, C.A., Kainer, K.A., d'Oliveira, M.V.N., Staudhammer, C.L., Baraloto, C., 2014. Logging in bamboo-dominated forests in southwestern Amazonia: caveats and opportunities for smallholder forest management. For. Ecol. Manage. 315, 202–210.
Romero, C., Putz, F., 2018. Theory-of-change development for the evaluation of Forest Stewardship Council Certification of sustained timber yields from natural forests in Indonesia. Forests 9, 547.
Romero, C., Sills, E.O., Guariguata, M.R., Cerutti, P.O., Lescuyer, G., Putz, F.E., 2017. Evaluation of the impacts of Forest Stewardship Council (FSC) certification of natural forest management in the tropics: a rigorous approach to assessment of a complex conservation intervention. Int. Forest. Rev. 19, 36–49.
Roopsind, A., Caughlin, T.T., Hout, P., Arets, E., Putz, F.E., 2018. Trade-offs between carbon stocks and timber recovery in tropical forests are mediated by logging intensity. Glob. Change Biol. 24, 2862–2874.
Ruslandi, Klassen, A., Romero, C., Putz, F.E., 2014. 15 Forest Stewardship Council Certification of natural forest management in Indonesia: Required improvements, costs, incentives, and barriers. In: Katila, P., Galloway, G., de Jong, W., Pacheco, P., Gerardo, M. (Eds.), Forests Under Pressure: Local Responses to Global Issues. IUFRO, Vienna.
Rutishauser, E., Hérault, B., Baraloto, C., Blanc, L., Descroix, L., Sotta, E.D., Ferreira, J., Kanashiro, M., Mazzei, L., d'Oliveira, M.V.N., de Oliveira, L.C., Peña-Claros, M., Putz, F.E., Ruschel, A.R., Rodney, K., Roopsind, A., Shenkin, A., da Silva, K.E., de Souza, C.R., Toledo, M., Vidal, E., West, T.A.P., Wortel, V., Sist, P., 2015. Rapid tree carbon stock recovery in managed Amazonian forests. Curr. Biol. 25, R787–R788.
Shenkin, A., Bolker, B., Peña-Claros, M., Licona, J.C., Putz, F.E., 2015. Fates of trees damaged by logging in Amazonian Bolivia. For. Ecol. Manage. 357, 50–59.
Silva Camila, V.J., Aragão Luiz, E.O.C., Barlow, J., Espirito-Santo, F., Young Paul, J., Anderson Liana, O., Berenguer, E., Brasil, I., Foster Brown, I., Castro, B., Farias, R., Ferreira, J., França, F., Graça Paulo, M.L.A., Kirsten, L., Lopes Aline, P., Salimon, C., Scaranello Marcos, A., Seixas, M., Souza Fernanda, C., Xaud Haron, A.M., 2018. Drought-induced Amazonian wildfires instigate a decadal-scale disruption of forest carbon dynamics. Philos. Trans. Roy. Soc. B: Biol. Sci. 373, 20180043.
Sist, P., Ferreira, F.N., 2007. Sustainability of reduced-impact logging in the Eastern Amazon. For. Ecol. Manage. 243, 199–209.
Sist, P., Fimbel, R., Sheil, D., Nasi, R., Chevallier, M.H., 2003. Towards sustainable management of mixed dipterocarp forests of South-east Asia: moving beyond minimum diameter cutting limits. Environ. Conserv. 30, 364–374.
Sist, P., Mazzei, L., Blanc, L., Rutishauser, E., 2014. Large trees as key elements of carbon storage and dynamics after selective logging in the Eastern Amazon. For. Ecol. Manage. 318, 103–109.
Vidal, E., West, T.A.P., Putz, F.E., 2016. Recovery of biomass and merchantable timber volumes twenty years after conventional and reduced-impact logging in Amazonian Brazil. For. Ecol. Manage. 376, 1–8.
West, T.A.P., Vidal, E., Putz, F.E., 2014. Forest biomass recovery after conventional and reduced-impact logging in Amazonian Brazil. For. Ecol. Manage. 314, 59–63.
Whitmore, T., 1998. An Introduction to Tropical Rain Forests. Oxford University Press, Oxford.
Withey, K., Berenguer, E., Palmeira Alessandro, F., Espírito-Santo Fernando, D.B., Lennox Gareth, D., Silva Camila, V.J., Aragão Luiz, E.O.C., Ferreira, J., França, F., Malhi, Y., Rossi Liana, C., Barlow, J., 2018. Quantifying immediate carbon emissions from El Niño-mediated wildfires in humid tropical forests. Philos. Trans. Roy. Soc. B: Biol. Sci. 373, 20170312.