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Editorial

Introduction to the special issue: Reduced-impact logging for climate change mitigation (RIL-C)

"Reduced-impact logging (RIL)," a phrase used in over 5000 articles since it was coined 25 years ago (Putz and Pinard, 1993), is often heralded as a way to balance environmental protection with timber production in selectively logged forests in the tropics. Despite this enthusiasm, RIL adoption remains limited for several familiar reasons, such as high cost, poor transparency; and lack of incentive mechanisms, training, and pressure from environmental groups (Putz et al., 2000). In this special issue, we address a related, but hitherto poorly recognized barrier to RIL: the previous lack of a widely accepted, outcome-based standard for evaluating RIL performance. In the absence of this standard, there was a cacophony of RIL claims with no ability to justify, measure, or compare them.

In this special issue we build upon previous work (Griscom et al., 2014) to develop a set of improved practices and related measures referred to collectively as Reduced-Impact Logging for Climate Change Mitigation (RIL-C), with emphasis on the "C" to denote the additional carbon stored in managed forests. With RIL-C, the authors' shared goal is to promote more environmentally sound tropical forestry that is also economically viable over the long term by focusing on climate-effective RIL outcomes that can be measured inexpensively and consistently.

In 2016, The Nature Conservancy and partners published a methodology for measuring and validating RIL-C outcomes (The Nature Conservancy and TerraCarbon LLC, 2016); in 2018, this methodology was recommended for making carbon claims as part of the Forest Stewardship Council's Ecosystem Services Procedure (FSC-ES, Forest Stewardship Council, 2018). The methods used by the authors in this issue are consistent with these standards, thereby providing means for credible justification of RIL-C claims across multiple geographies. Data were collected in 56 forest management enterprises in selectively logged forests of 7 tropical countries using a consistent field-based measurement system. This system is designed to comprehensively measure logging disturbances and set a baseline against which impact reductions can be evaluated.

Our emphasis on carbon is intended to motivate uptake of the broader suite of RIL interventions; we have no intention to undermine the important non-carbon benefits that RIL delivers. More work is needed to develop outcome-based systems for RIL benefits to worker safety, financial profitability, water quality, biodiversity, and soil health. In the meantime, carbon is presently the only ecosystem service with a transparent and consistent system for measuring, monitoring, reporting, verifying, and validating RIL performance (RIL-C MRV). To measure and compare selective logging emissions across the tropics, we employ a Carbon Impact Factor (CIF), which expresses collateral damage emissions relative to the carbon in extracted timber (Feldpausch et al., 2005). By including mass on both sides of the ratio, CIF adjusts for regional variation in wood density and harvest intensity so as to inform efforts to improve forest management in ways that maintain timber supplies.

A RIL-C MRV system can overcome some barriers to RIL uptake, but it does not directly address the primary problem of lack of effective incentives. Nevertheless, we believe that the science necessary to operationalize RIL-C MRV systems can also be used to quantify climate mitigation opportunities, and that knowledge of these opportunities can motivate stakeholders to incentivize action to satisfy regional climate commitments. Specifically, we believe that if decision-makers responsible for jurisdictional initiatives are made aware of the cost-effective carbon savings from RIL-C best practices they will activate instruments to operationalize them.

It is too early to tell if climate-based incentives will result in sustained improvements in tropical forest management, but promising signals are emerging. In East Kalimantan, Indonesia for example, one logging concession recently demonstrated emissions 50% below the regional baseline and certified their claim using the RIL-C methodology (Ichwan, 2018). Other countries (e.g. Mexico, the Republic of Congo, and Democratic Republic of Congo) are considering jurisdictional RIL-C initiatives. Suriname recently included RIL-C in its Forest Reference Emissions Level submitted to the United Nations (Government of Suriname, 2018). Other incentive mechanisms such as FSC-ES, the World Bank Forest Carbon Partnership Facility (FCPF), the Green Climate Fund, private sector supply chain commitments, the voluntary carbon market, and various hybrid approaches provide additional opportunities to capture the carbon benefits of RIL-C practices. And finally, in countries such as Gabon, RIL-C is being considered as a mechanism to demonstrate performance as a large portion of its Nationally Determined Contribution to the Paris Agreement (NDC). Indeed, the research presented in this special issue indicates that at least 9 tropical countries could deliver more than 50% of their NDC commitments through deployment of RIL-C alone.

Given this landscape of opportunities, there is utility in providing accurate baseline estimates of carbon emissions from selective logging, establishing feasible RIL-C targets, and incentivizing RIL-C practices that allow forest managers to reach production targets while reducing carbon emissions. The articles in this special issue represent a collective effort to move towards these applied science goals at both regional and pan-tropical scales. In the first five articles, we present selective logging emissions data from forest management enterprises in East Kalimantan Province in Indonesia, the Congo Basin (Democratic Republic of Congo, Gabon, and Republic of the Congo), Madre de Dios Department in Peru, Suriname, and three Mexican states on the Yucatan Peninsula. Each analysis considers regionalized emission reduction benefits available from adoption of RIL-C practices. In the final paper, we synthesize these data to draw conclusions, estimate pan-tropical logging emissions, set a pan-tropical RIL-C target, and highlight the most promising RIL-C practices for achieving that target.

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While adoption of RIL in general and RIL-C in particular will increase the sustainability of timber yields by reducing stand damage and increasing post-logging rates of timber stock recovery (Roopsind et al., 2018), we acknowledge that additional silvicultural treatments are often necessary. Similar to RIL research, much has been said on this important subject, but very little seems to have improved the situation on the ground (Putz, 2017). Neither RIL nor RIL-C will solve the sustainability problem, but we hope that increased scrutiny provided by outcome-based systems like RIL-C will encourage efforts to ensure that harvest intensities and intervals are supported by science. Although we do not directly address the relationship between RIL and sustainability in this issue, we are happy to see RIL-C performance metrics integrated into independent standards to ensure sustainability, such as FSC-ES.

Our generation faces enormous challenges if we are to meet increasing demands for wood and carbon storage. If we continue to harvest timber from tropical forests unsustainably and destructively, we will spend down our natural capital and reduce opportunities to meet the challenges of the future. We hope that RIL-C contributes to an alternative vision where humans, forestry, and forest ecosystems can thrive in synergy.

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Reduced-impact logging in Borneo to minimize carbon emissions and impacts on sensitive habitats while maintaining timber yields

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ABSTRACT

We define two implementation levels for reduced-impact logging for climate mitigation (RIL-C) practices for felling, skidding, and hauling in dipterocarp forest concessions of East and North Kalimantan. Each implementation level reduces logging emissions by a consistent proportion below the business-as-usual emissions baseline, which varies with harvest intensity. Level 1 reflects the best recorded emissions performance for each type of practice. Level 2 is more ambitious but feasible based on workshop feedback from concession managers and forestry experts, and confirmed by a recent demonstration. At Level 1 emissions can be reduced by 33%, avoiding emissions of 64.9 ± 22.2 $MgCO_2$ per ha harvested, on average. At Level 2 emissions can be reduced by 46%, avoiding 88.6 ± 22.7 $MgCO_2$ ha^{-1}. The greatest emissions reductions derive from (i) not felling trees that will be left in the forest due to commercial defects, and (ii) use of long-line cable winching to avoid bulldozer impacts.

We also quantify the potential to avoid logging steep slopes and riparian habitats, while holding to our RIL-C accounting assumption that timber yields are maintained to avoid problems of leakage and product substitution. Logging damage to riparian areas < 50 m from perennial streams could be avoided by re-locating harvests to less sensitive areas that currently are not accessed due to lack of spatial planning. In all but the steepest concessions, all slopes > 40% could similarly be avoided. The combined areas of these sensitive habitats (steep slopes and riparian buffers) represented 16% of each cutting block on average.

Implementation of RIL-C practices would deliver 8% (Level 1) and 11% (Level 2) of Indonesia's pledged reductions to their forest reference emissions level as a nationally determined contribution to the Paris Climate Agreement. In concert with RIL-C practices, 30% of logging concession areas could be permanently protected from logging and conversion to minimize impacts on biodiversity, soils, and water quality, thereby expanding Indonesia's protected areas by one third and achieving 93% of Indonesia's Aichi Target 11 (the effective conservation of at least 17% of lands). Both these Paris Climate Agreement and Aichi outcomes could be delivered with no reductions in timber yields and substantial improvements in worker safety and sustainability of the natural forest timber sector.

1. Introduction

Tropical and subtropical countries emit an estimated 0.85 $GtCO_2$ yr^{-1} from selective harvests of timber from natural forests (Ellis et al. this issue), equivalent to direct emissions from commercial buildings globally (Lucon et al., 2014), and ¼ of average annual emissions from tropical forest cover loss since 2001 (Gibbs et al., 2018). At least 39 tropical countries report plans to reduce these emissions in their

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Nationally Determined Contribution (NDC) to the United Nations Framework Convention on Climate Change Paris Agreement (Ellis et al. this issue). One of these countries, Indonesia, is considering a national regulation that requires reduced impact logging for climate mitigation, or RIL-C, as part of fulfilling their NDC to the Paris Climate Agreement. Ideally, RIL-C can be implemented while also delivering explicit biodiversity and other conservation outcomes to which Indonesia committed to under the Convention on Biological Diversity.

Studies on tropical forestry over the past decades reported that reduced-impact logging (RIL) in experimental cutting blocks reduced emissions from logging by 30–50% (Johns et al., 1996; Mazzei et al., 2010; Medjibe et al., 2013; Pinard and Putz, 1996; Vidal et al., 2016). Given the experimental nature of these studies, it remains unclear whether these levels of emission reductions can be realized and verified at operational scales, and what individual RIL interventions should be prioritized to achieve these results. More fundamentally, there is evidence that emissions benefits of some attempts at RIL disappear when logging intensities are taken into account (Sist et al., 2003; Martin et al., 2015). It is also not clear whether the potential climate mitigation benefits of RIL are associated with forest certification (Griscom et al., 2014; Martin et al., 2015; Miller et al., 2011), which may reflect that certification systems, such as the Forest Stewardship Council (FSC), while requiring that many RIL practices are employed, were not designed to achieve or verify emissions reductions (Romero and Putz, 2018).

Confusion about the scalable and verifiable climate benefits of RIL motivated development of RIL-C, a set of improved selective logging practices focused on verifiable emission reductions (Griscom et al., 2014). RIL-C accounting methodologies, for example, differ from prior RIL standards in emphasizing the avoidance of felling trees that are subsequently not harvested due to the presence of hollows in their stem. RIL addresses a broader set of issues beyond carbon emissions that are not considered by RIL-C, such as minimizing erosion, set asides for biodiversity conservation, and worker safety. Neither RIL-C or RIL address other dimensions of improved natural forest management (NFM) such as sustainable silvicultural practices, social equity, milling practices, and other aspects of biodiversity conservation (Fig. 1).

Here we propose a set of RIL-C practices, and aligned RIL set asides, for East and North Kalimantan. We estimate the potential of individual RIL-C practices to reduce carbon emissions, and we extrapolated carbon benefits across natural timber production forests in Indonesia. We consider the potential to achieve additional non-carbon conservation outcomes, in particular biodiversity and hydrology, that could accompany implementation of RIL-C practices by improved spatial planning of areas set aside from logging.

We focus on RIL-C outcomes that can be verified by or are consistent with a new methodology designed to account for climate benefits (The Nature Conservancy and TerraCarbon LLC, 2016). This methodology uses the same field data employed here to establish a regional emissions baseline for East and North Kalimantan, from which emissions reductions can be achieved through the four RIL-C practices to be described here. The methodology establishes "impact parameters" for each RIL-C practice that allow efficient and robust field verification of logging impacts. Crediting of emissions reductions are restricted to legally authorized concessions on previously logged forests.

We limit our analysis to discrete RIL-C practices that have measurable carbon emissions reductions associated with explicit and feasible improvements in practices, and that do not change timber yields. These constraints allow for commercially viable scaling of RIL-C practices as contributions to achievement of national emissions reductions goals without displacing demand for timber elsewhere (i.e., leakage) and/or promoting the substitution of wood products by high-carbon emission materials such as concrete, steel, or aluminum (Oliver et al., 2014).

The effects of selective logging in tropical forests on carbon, biodiversity, and other ecosystem services vary with harvest intensities, practices employed, and spatial distributions of impacts (Burivalova et al., 2014; Griscom et al., 2018). A study in East and North Kalimantan reported that on average over one third of harvest block areas are not actually logged under conventional practices but that the unlogged areas did not occur in low stand volume areas, nor were they concentrated in ecologically sensitive areas (Ellis et al., 2016). This finding presents the opportunity to reduce the deleterious impacts of logging on biodiversity and hydrology by setting aside sensitive riparian and steep slope areas without reducing timber yields. Set asides could be established along riparian buffer zones, given their importance for wildlife as well as for the integrity of freshwater ecosystems, downstream water quality, and attenuation of peak flows. Set asides might also be designated in areas with steep slopes, which are prone to excessive soil disturbances associated with skidding, with attendant

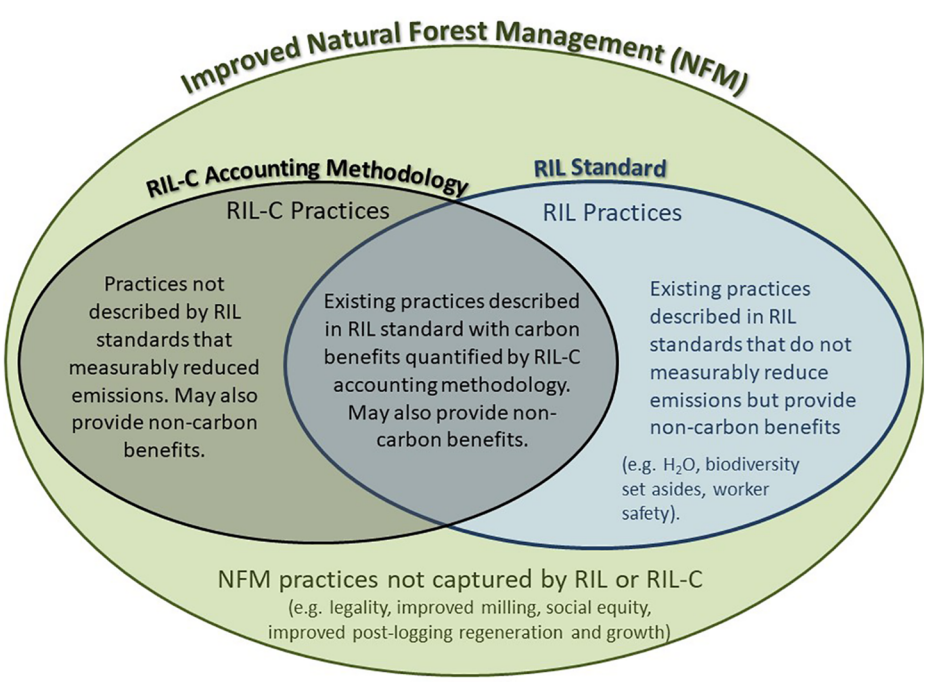

Fig. 1. We use the term "improved natural forest management" (NFM) to refer to the broadest range of improved interventions in forests managed for timber production. We use the more specific term "reduced-impact logging" (RIL) to refer to the direct impacts of harvesting on forest ecosystems and people, as specified by existing RIL standards (Dykstra and Heinrich, 1996). We use the term "reduced-impact logging for climate mitigation" (RIL-C) to refer to the set of RIL practices, and practices not included in RIL standards, that are defined by measurable thresholds (Griscom et al., 2014) and that can be verified by a carbon accounting methodology.

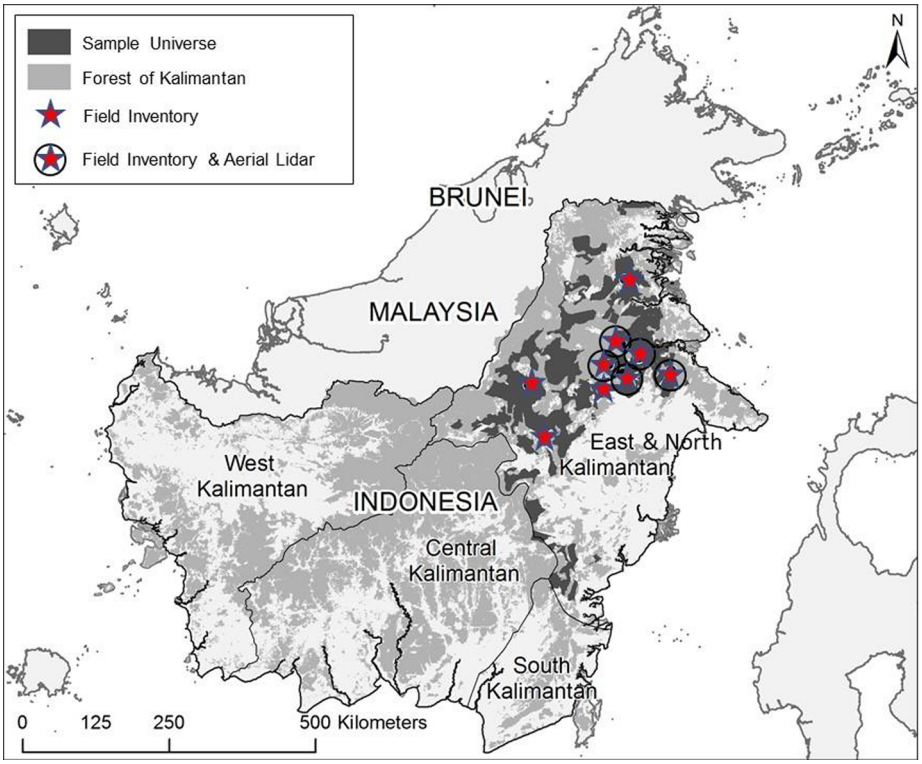

Fig. 2. Location of the nine logging concessions where field inventory (N = 9) and aerial lidar (N = 5) data were collected in native dipterocarp forests in East and North Kalimantan, Indonesia.

increases in erosion and reductions in site productivity (Putz et al., 2018; Sidle et al., 2004). We include here an analysis of improved set aside planning that can happen in tandem with RIL-C efficiency practices that do not adjust harvest intensity and timber volume extracted.

2. Methods

2.1. Study sites and field measurements

We carried out field surveys in nine logging concessions selected in a stratified random manner in dipterocarp forests of East and North Kalimantan, Indonesia (Fig. 2). To stratify the samples, we conducted a cluster analysis of all the major timber concessions in the region using Wards Linkage Euclidian distance. We identified three clusters (dark grey areas in Fig. 1) based on slope, elevation, percent cover by each of three major soil types (Inceptisols, Oxisols, Ultisols), distance from the nearest city, and percent cover of major vegetation types (wetlands, primary forest, logged forest, and non-forest vegetation). From each cluster two non-FSC certified concessions were randomly selected, as well as the one concession that was FSC-certified prior to 2013. See Griscom et al. (2014) for further sampling details.

Commercial logging in our study area involved the construction of haul roads mostly along ridge lines and the use of bulldozers to skid logs ≥60 cm diameter at breast height (dbh) to log yards adjacent to haul roads. Two of the FSC-certified concessions we sampled were testing an alternative 'monocable winch' system in which logs were skidded using a ca. 100 m winch cable drawn by an 18 horsepower engine attached to a narrow metal sled anchored to a tree.

Within each selected concession (Fig. 2), we randomly selected a cutting block (ca. 100 ha) harvested within the previous 12 months. In each block we employed 10 different targeted sampling methods to quantify tree damage and death that resulted from felling, skidding logs from stumps to log yards, and forest clearing for haul roads and log yards, as summarized in Table 1. For more details on the study areas and sampling, see Griscom et al. (2014).

2.2. Calculations of carbon emissions

For each logging concession we estimate gross committed above and below-ground emissions (i.e., conversion of live above and below-ground tree biomass to necromass) due to logging. As consistent with Griscom et al. (2014) and Ellis et al. (this issue), we converted measurements of tree diameter into biomass using the 'model II moist forest stands' equation from Chave et al. (2005) and below-ground biomass was estimated from shoot biomass using Equation 1 from Mokany et al. (2006). It was beyond the scope of this study to identify tree species with high confidence, given the high species diversity of these forests, so we assumed average wood density for lowland forests in tropical Asia (0.57) (Reyes et al., 1992). For biomass-to-carbon conversions, we used the standard carbon fraction of 0.5 (Lugo and Brown, 1992). We used a committed emissions approach to necromass decay to be consistent with most of the literature on RIL (e.g., Medjibe et al., 2011; Putz et al., 2008) and other articles in this issue.

Our accounting methods differ in one notable way from other articles in this special issue: we calculate emissions per ha of areas accessed for logging, rather than per ha of annual harvest blocks. This allows our numbers to be consistent with the RIL-C verification methodology discussed above, which excludes areas not accessed. Also, given the unusually large and variable portions of cutting blocks not accessed in East and North Kalimantan (Ellis et al., 2016), this focuses our analysis of factors influencing emissions levels on the areas where emissions are occurring. We also report overall emissions per ha of harvest blocks for comparison. We extrapolate avoidable emissions to the country of Indonesia based on our estimated emissions per m^3 of roundwood extracted, and national roundwood volume statistics (BPS - Statistics Indonesia, 2018), both of which are independent of harvest areas.

2.3. Estimation of baseline and potential emission reductions

We identified four RIL-C practices that have the potential to

Table 1
Summary of field sample methods used in concessions to quantify impacts from logging.

Measurement description	N per concession (mean)	N total	Concessions sampled	Comments
Felling				
Felled trees tallied	119	1075	9	total area sampled: 226 ha
Felled trees measured	16	148	9	plot size varied by gap size
Felled tree gap plots	10	91	9	plot size varied by gap size
Skidding				
Skid trail lengths measured (km)	3.7	33.5	9	total area sampled: 226 ha
Skid trail networks mapped	4.4	40	9	total area sampled: 226 ha
Skid trail plots	19	167	9	plots: 10 m × skid trail width
Skid trail side plots	5	48	7	plots: 10 m × 5 m
Hauling				
Road width measurements	24	217	9	width of road corridor
Log yards measured	5	47	7	length and width
Biomass prism plots	16	80	5	20 & 40 BAF prism used

contribute to reductions in carbon emissions: (1) "FELL1" - avoided felling of trees from which no wood is extracted (includes felled tree and collateral damage emissions); (2) "FELL2" - improved bucking to maximize timber extraction per tree felled, allowing reduced total number of trees harvested to deliver the same roundwood volume; (3) "SKID" - improved skidding to reduce mortality of non-commercial trees; and, (4) "HAUL" - reduced haul road corridor widths and sizes of log landings. We conducted field inventories in each concession to estimate the level of performance with respect to each of these four practices. By averaging the performance of sampled concessions across each practice, we created a separate baseline for each practice.

For each practice we identified the "leader" concession as the one with the highest performance (lowest emissions normalized by harvest intensity). We estimated the emissions that would be avoided if all concessions implemented each of the four practices at the level demonstrated by the "leader" for each practice. We refer to this as "Level 1" implementation of RIL-C: achieving the upper end of existing practices (see Table 2).

As an aspirational estimate of emission reductions, we also estimated a "Level 2" of RIL-C implementation, based on a hypothetical level of improvement that could be achieved for each of the four practices. We identified these levels through a workshop with 19 concession managers from the study region and 16 other forestry practitioners held in Tanjug Redeb, East Kalimantan, on May 28, 2013. We distributed a questionnaire one week before the meeting, to allow participants sufficient time to consider quantitative performance levels that are ambitious but feasible for each practice (see Appendix A). The specific definitions of Level 1 and Level 2 practices that we modeled for each of the four practice categories (FELL1, FELL2, SKID, HAUL), are presented in Table 2.

In the process of characterizing RIL-C practices, we identified two potential additional improved practices that could generate emissions reductions; however, we were unable to determine their climate benefits. These potential yet unquantified RIL-C practices were: (i) directional felling to avoid collateral damage; and, (ii) liana cutting to reduce collateral damage due to inter-connected canopies. We also did not quantify the potential emissions reductions from reducing haul road length given practitioner feedback that adjusting haul road length is expensive, and we were unable to determine the extent to which road length could be feasibly reduced.

2.4. Analysis of slope and riparian set asides

We used publicly available 30 m digital elevation model data from the Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER) with ArcGIS v10.1 software to estimate set aside areas needed for two types of sensitive habitat: stream buffer zones and steep slopes. Our mapping of set asides was restricted to observed areas of logging activity, as determined from aerial lidar data which we acquired for five of the nine concessions (Ellis et al., 2016), and by field delineation using hand held Garmin 60 CSX GPS units in the remaining four concessions. Methods used for aerial lidar data acquisition and analysis are reported by Ellis et al. (2016). Details of our methods for analysis of the ASTER dataset to derive slope are reported by Putz et al. (2018), where it was determined that ASTER provides very conservative estimates of the extent of steep slope areas. We defined steep slopes as > 40%, and riparian zones as 50 m on either side of perennial streams. Streams were delineated using the flow accumulation tool within ArcGIS v10.4, using 30 m ASTER data.

2.5. Statistical analyses

We used analysis of covariance (ANCOVA, equal slopes, Tukey method for comparisons) to compare baseline emissions from the sampled logging concessions (N = 9) with the two modelled RIL-C emissions levels. While it was our intention to arrive at different emissions levels, we did not know a priori if each implementation level would result in significantly different emissions levels, since our modeling only fixed the performance variables defined in Table 2 and we did not control for all other concession-specific variables that influence emissions levels (e.g. slope, forest structure). To determine the sensitivity of the two proposed RIL-C implementation levels to logging intensity, we conducted a simple linear regression. All variables were approximately normally distributed. All statistical analyses were performed in Minitab © v. 16.2.2.

3. Results and Discussion

3.1. Definition of Level 1 and Level 2 RIL-C implementation

We found no clear relationship between concession cluster membership and logging emissions. The cluster of more remote high elevation concessions (A, D, F) had a wide range of logging emissions levels, overlapping with emissions levels for lower elevation least remote concessions (E, G, H) and intermediate concessions (B, C, I). The nine sampled concessions differed substantially in existing emissions associated with each of the four logging practices, but no one concession was the "leader" for all practices (Fig. 3). Concession H showed the lowest emissions, per m^3 of timber harvested, for each of the two types of felling practices. Skidding emissions were lowest in concession F. We selected concession D as the "leader" for haul road emissions performance because it demonstrated the lowest mean road width. Two other concessions (I and H) had wider haul road corridors but slightly lower total haul road emissions due to lower haul road length. However, concessions I and H were not selected as the "leader" because road length was not considered in ranking emissions performance, based on

Table 2
Reductions in biomass carbon emissions, expressed as MgC ha^{-1} and MgC m^{-3}, due to demonstrated (Level 1) and potential (Level 2) target implementation levels for four RIL-C practices. Emissions reduction estimates represent the differences between baseline emissions and emissions under a given RIL-C level of implementation. Uncertainty is reported as ± 95% confidence intervals from the mean.

Logging Activity Category	RIL-C Practice Name and Description	Mean Emissions Baseline MgC ha^{-1} (MgC m^{-3})	Level 1 Emissions Reductions MgC ha^{-1} (MgC m^{-3})	Level 2 Emissions Reductions MgC ha^{-1} (MgC m^{-3})	Level 1 Implementation	Level 2 Implementation
Felling and log recovery: Includes pre-harvest inventory, felling, bucking and extraction practices.	FELL1: Emissions from trees felled and abandoned.	8.39 ± 4.50 (0.24 ± 0.11)	6.06 ± 4.08 (0.17 ± 0.10)	7.48 ± 4.34 (0.21 ± 0.10)	Best recorded (concession H): 7.7% of felled trees are abandoned	3% of trees felled for harvest are left in the forest with no roundwood removed.
	FELL2: Emissions from trees felled with some volume extracted.	21.86 ± 2.41 (0.62 ± 0.11)	1.86 ± 1.16 (0.06 ± 0.04)	2.72 ± 0.94 (0.08 ± 0.03)	Best recorded (concession H): 10.6% of commercial log length left in forest	5% of commercial log length left in forest
Skidding: Includes pre-harvest inventory and dozer skid trail planning, long-line winching, directional felling.	SKID: Emissions from mortality resulting from skidding damage	12.51 ± 3.81 (0.35 ± 0.12)	6.73 ± 3.81 (0.18 ± 0.11)	9.62 ± 3.81 (0.27 ± 0.11)	Best recorded (concession F): 0.5 MgC per 100 m of skid trail, and 115.4 m of skid trail per ha of cutting block. Excludes areas using monocable winch system.	50% of Level 1, due to use of long-line cable winching across 50% of areas accessed.
Hauling: Narrower haul roads and smaller log landings.	HAUL: Emissions from mortality resulting from clearing road corridors and log landings	9.58 ± 1.84 (0.28 ± 0.08)	3.05 ± 1.52 (0.09 ± 0.05)	4.35 ± 1.55 (0.12 ± 0.05)	Narrowest recorded mean haul road corridors, and absence of separate log yards (concession D): 25 m mean road & yard corridor width	20 m road & yard corridor width.

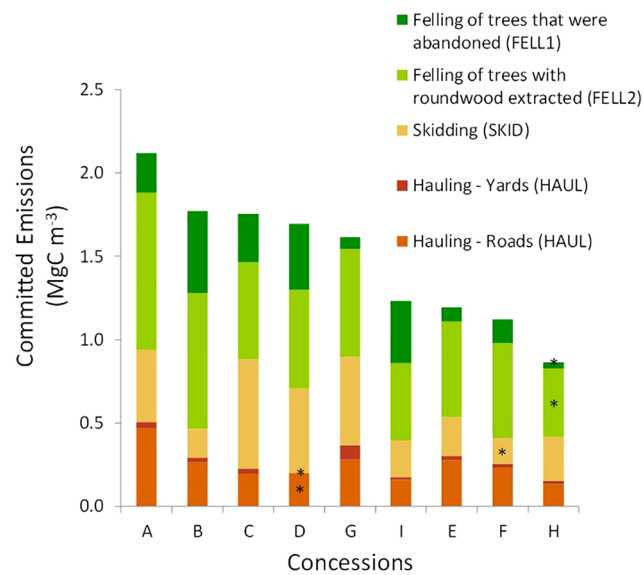

Fig. 3. Existing above- and below-ground carbon biomass emissions per m^3 of timber extracted associated with the four RIL-C practices across nine concessions in East and North Kalimantan. Two different components of hauling impacts (roads and yards) were measured, and then lumped into one improved practice category (HAUL). The best performers for each of the four logging practices are identified with asterisk (*). Note that for haul roads, the best performer was determined by mean haul road width (lowest in D), rather than haul road emissions (lowest in H), because emissions are also determined by road length, which was not considered adjustable at reasonable cost.

expert workshop participants' conclusion that road length is controlled by topography and not adjustable at reasonable cost. Concession D also demonstrated that logs can be yarded along the margins of haul roads, as recommended by Pinard et al. (1995) such that additional clearings are not needed for log yards. The quantitative definitions of Level 1 RIL-C (based on these leaders) and Level 2 RIL-C (based on additional improvements informed from our RIL-C workshop) are presented in Table 2.

3.2. Effectiveness of RIL-C across logging intensities

There is a positive linear relationship between timber volume extracted and carbon emissions (MgC ha^{-1}) in the sampled concessions ($R^2 = 0.35$, $F_{1,8} = 3.72$, $p = 0.095$). This relationship would hold if all concessions reached the performance of each "leader" in all four of the practices, (Level 1: $R^2 = 0.59$, $F_{1,8} = 10.01$, $p = 0.016$), and if all concessions reached the performance across all four practices deemed feasible by concession managers (Level 2: $R^2 = 0.59$, $F_{1,8} = 10.23$, $p = 0.015$; Fig. 4). As such, the approximate proportion of avoidable logging emissions with RIL-C practices (red and blue lines in Figs. 4 and 5) are, by design, retained at a wide range of harvest intensities. The greater slope of the measured (baseline) emissions as compared with Level 1 and Level 2 emissions indicates that concessions with higher stocking of commercial timber volume have incrementally higher potential to reduce emissions with improved practices. The positive relationship between harvest intensity and committed biomass emissions is driven by felling emissions, as compared to skidding and hauling emissions that are not sensitive to harvest intensities, as detailed by Griscom et al. (2014). Thus, the absolute reference level for measuring reduced felling emissions (Fell 1 and Fell 2 RIL-C practices in Table 2) within a given concession varies as a function of harvest intensity. To accommodate this harvest intensity dependence, we defined the implementation levels determining Levels 1 and 2 for FELL1 and FELL2 as proportions, rather than absolute levels (i.e. percent of trees felled and abandoned, percent of commercial log length left in forest). As such,

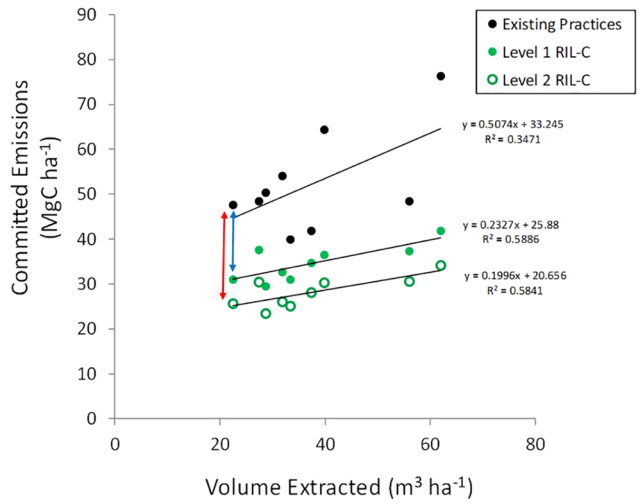

Fig. 4. We modeled the potential to reduce emissions in areas accessed for harvests, due to logging impacts on above- and below-ground carbon biomass if RIL-C practices were implemented. Black dots indicate measured carbon emissions from logging in each of nine logging concessions sampled. Green dots indicate modeled carbon emissions from logging if all four RIL-C practices were implemented at levels described in the Table 2. The blue (Level 1) and red (Level 2) arrows indicate the avoided emissions due to full implementation of RIL-C practices in the concession with the lowest harvest intensity.

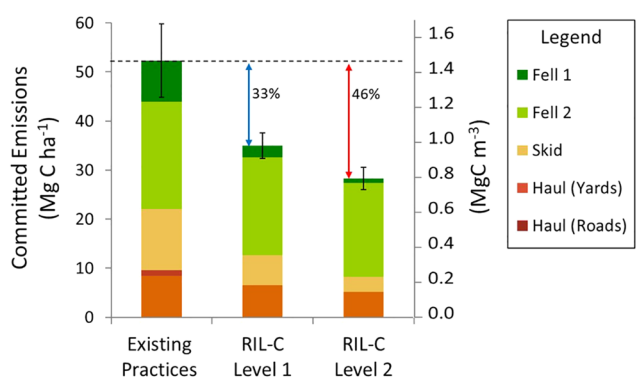

Fig. 5. Measured committed emissions of above- and below-ground carbon biomass under existing practices, and modeled emissions with implementation of four RIL-C practices at two different levels (see Table 2). Emissions (MgC) are reported per ha accessed for logging (left axis) and per cubic meter of timber extracted (right axis). Error bars are 95% confidence intervals based on variability among concessions in measured and modelled emissions per ha. Level 1 RIL-C avoids 33% of logging emissions (blue arrow) while Level 2 avoids 46% of emissions (red arrow) without changing timber yields.

RIL-C is designed (and explicitly required in the Verra RIL-C methodology) to avoid driving changes in harvest intensity as discussed above.

3.3. Avoidable emissions

The baseline was determined as the average committed emissions of above and below-ground carbon from tree biomass across all nine concessions: 52.3 ± 7.4 MgC ha^{-1}. If implemented at Level 1 (all concessions perform at the level of "leaders" on all four practices), emissions could be reduced from this baseline by 33% (by 17.7 MgC ha^{-1}) to 34.6 MgC ha^{-1} (95% CI ± 2.6 MgC ha^{-1}) ($T(2, 26) = -5.69$, $p < 0.0001$). If implemented at Level 2 (all concessions achieve the aspirational levels across all four practices), emissions could be reduced from the baseline by 46% (by 24.2 MgC ha^{-1}) to 28.2 ± 2.3 MgC ha^{-1} ($T(2, 26) = -8.43$, $p < 0.0001$), as shown in Fig. 5. Level 2 ambition, which we modeled but did not measure in the field, has been recently demonstrated. The Narkata Rimba logging concession in East Kalimantan achieved 50% verified RIL-C emissions reductions following our accounting methods (SCS Global Services, 2018). Avoidable emissions per ha of harvest blocks (including areas not harvested) are 12.07 MgC ha^{-1} (Level 1) and 16.77 MgC ha^{-1} (Level 2), representing the same proportional reductions from baseline. These are conservative estimates of avoidable emissions because the legal minimum basal diameter limit for roundwood has been reduced from 60 cm when we conducted our field work to 40 cm now. Further, research is needed to explore additional improved felling practices we identified but were unable to quantify (directional felling and liana cutting) that could yield additional emissions reductions.

Reducing the proportion of trees that are unnecessarily felled, because they are left in the forest (FELL1), is the individual practice offering the largest avoidable emissions at Level 1 (Table 2). This improved practice is also easily achieved. The primary reason for leaving felled boles in the forest is small hollow zones at tree centers – averaging 5% of trunk volume as void - which can be detected with a "plunge cut" in which the chainsaw blade is used to probe for rotten and missing wood. If the blade enters the stem in a vertical orientation, it results in minimal structural and vascular damage to the tree. Concession managers were particularly receptive to this practice, given that it should save money and improve worker safety (reduced working hours and risks to chainsaw operators, reduced fuel and wear on chainsaws). This practice influences the commercial value of future harvest cycles in positive and negative ways. On the one hand, hollow trees of commercial size (≥60 cm dbh) are left standing to occupy growing space while generating low or no commercial value. On the other hand, each hollow tree not felled avoids mortal damage to three smaller trees (10–60 cm dbh) of commercial species, on average (Griscom et al. 2014).

Use of long-line winching is the practice offering the largest avoidable emissions at Level 2 (Table 2). Concession managers we convened were skeptical that "monocable winch" machines being field tested are feasible for more widespread adoption, due to the lower log extraction rates and unreliable machinery. Other long-line winching technologies used extensively in Malaysian Borneo should be explored that have similarly low skidding impacts yet extraction rates as high or higher than bulldozer skidding (Norizah et al., 2012). However, widespread adoption of these higher powered long-line cable systems could allow access to yet steeper slopes and may increase the need to enforce harvest restrictions.

If the potential emissions reductions we estimate here for concessions in East and North Kalimantan were achieved in natural forest logging concessions across Indonesia (the majority of which occur in structurally similar dipterocarp forests), Indonesia would avoid carbon emissions of 12.1 ± 3.7 $TgCO_2$ yr^{-1} with Level 1 RIL-C, or 16.7 ± 4.2 $TgCO_2$ yr^{-1} with Level 2 RIL-C. This assumes emissions reductions of 0.49 ± 0.15 MgC m^{-3} (Level 1) and 0.68 ± 0.17 MgC m^{-3} (Level 2) we describe here were achieved for the 6.7 million m^3 yr^{-1} tropical hardwood extraction reported in Indonesia (average 2006–2015) (BPS - Statistics Indonesia, 2018).

3.4. Avoidable impacts to sensitive habitats

The combined areas of sensitive habitats (i.e., steep slopes and riparian buffers) represented on average 16% of each cutting block, less than half the average area of annual cutting blocks not accessed (37%) as reported by Ellis et al. (2016) for the same concessions we sampled (Table 3). This presents an opportunity to re-distribute areas not accessed for timber to avoid sensitive habitats, while remaining consistent with RIL-C assumption of no change from business-as-usual timber yields.

However, the proportion of cutting blocks that fell into these sensitive habitats was highly variable, ranging from 1.6% to 50.7%. All

Table 3
Area of sensitive habitats in nine logging concessions in East and North Kalimantan.

Concession Code	Total Area (ha)	Area > 40% slope (as % of total area)	Area within 100 m perennial stream buffer (as % of total area)	Area of intersection of steep slopes and riparian buffer (ha)	Area of steep slope and riparian set asides (as % of total area)
A	344	1.2%	0.4%	0	1.6%
B	1775	10.7%	3.0%	0	13.7%
C	504	10.3%	2.9%	0	13.1%
D	378	50.7%	0.0%	0	50.7%
E	73	0.0%	5.3%	0	5.3%
F	55	11.6%	11.2%	0	22.8%
G	1841	4.8%	4.4%	0	9.2%
H	1259	17.3%	4.9%	2	22.0%
I	107	38.5%	0.0%	0	38.5%
All	6335	12.5%	3.5%	2	15.9%

concessions could have accommodated set asides for riparian buffers (50 m on either side of perennial streams), which ranged from 0 to 11% of harvest block area. Two of the nine concessions – those with extremely steep terrain - would need to set aside more than 30% of their cutting blocks to fully protect areas with greater than 40% slopes. This finding is conservative, given that the ASTER dataset provides a conservative estimate of the extent of steep slope areas, as compared with higher resolution Lidar data (Putz et al. 2018). The opportunity to redistribute harvested areas also assumes that stand biomass, which Ellis et al. (2016) found to be no different inside harvest block areas not accessed than in sensitive areas that were harvested, is a good proxy for commercial volume in these dipterocarp forests – an assumption that should be tested with additional sampling.

The potential for all concession managers to avoid riparian impacts without negatively impacting their bottom line is particularly important. Riparian buffer zones are widely considered to be crucial to the maintenance of biodiversity. In Australian timber production forests, riparian buffer zones maintained bat activity as high as in mature forest (Lloyd et al., 2006). The importance of riparian buffer zones on mammal, bird, and freshwater biodiversity has even been shown in tropical landscapes that have undergone a more dramatic conversion such as to oil palm (Giam et al., 2015), cattle pastures (Lees and Peres, 2008), and slash-and-burn agriculture (Iwata et al., 2003). We also argue that permanent set asides would benefit biodiversity given that some taxa do not fully recover their population sizes even after 35 years of post-logging recovery (Burivalova et al., 2015). We also caution that protection of only two types of sensitive habitat as defined here will not capture the range of habitat or spatial scales needed for comprehensive biodiversity conservation in this complex tropical forest landscape (Groves and Game, 2016).

4. Conclusions

One third of emissions from logging can be avoided without reducing timber yields if logging concessionaires implement RIL-C practices at levels already demonstrated by the best performing commercial operators (Level 1). Although we believe that this level of emissions reductions could be achieved at low cost or cost savings, evidence in support of this assumption is mixed (Medjibe and Putz, 2012). Level 1 RIL-C practices require no changes in machinery, but do require training in improved felling, bucking, skid trail planning, and haul road and yard construction. Cost savings are possible because each practice requires less use of machinery (chainsaws, bulldozers, and excavators). The greatest emissions reductions are achieved by one of the simplest improved practices that is likely to reduce cost: not felling trees that will be left in the forest due to commercial defects.

Nearly half (46%) of logging emissions can be avoided without reducing timber yields if logging concessionaires consistently implement RIL-C logging practices that regional concession managers and forestry practitioners indicate are ambitious but feasible. The largest of these additional emissions reductions would require adoption of long-line cable winching – a practice that has the added benefit of reducing the construction of bulldozer skid trails, a major cause of soil erosion in the rugged landscapes where logging is currently carried out in Indonesia (Putz et al., 2018). It is unclear whether carbon incentives are sufficient to catalyze this shift in yarding equipment

We also find that, in most logging concessions, improved spatial planning of harvesting activities could avoid impacts in riparian and steep slope habitats without changing business as usual harvest volumes, or changing RIL-C emissions reductions opportunities described above. Improved skid trail planning and skidder operator training and supervision could have the dual benefit of achieving Level 1 RIL-C skidding emissions while avoiding logging on steep slopes and in riparian zones.

RIL-C practices will drive other positive forest management outcomes. RIL-C will improve long term timber yields by reducing damage to future crop trees. Implementation of RIL-C requires worker training, which should deliver a social benefit of reduced work-related injuries and deaths. Similarly, the RIL-C aligned set-asides described here, and the planning and careful construction of skid trails and roads required to reduce carbon emissions will reduce biodiversity impacts and hydrological disruptions typically associated with logging (Chappell and Thang, 2007). However, RIL-C does not include the silvicultural practices necessary to sustain timber yields (e.g., liberation thinning and enrichment planting) nor does it address the fundamental soundness of rules governing harvest intensities and frequencies (Ruslandi et al., 2017). RIL-C is intended as one component of improved natural forest management systems that should also be explicitly designed to sustain timber yields, maintain biodiversity, minimize hydrological impacts, and maximize social equity, worker safety, and other aspects of ecological integrity and human well-being. For example, to deliver both carbon and many of these broader outcomes, RIL-C can be implemented along with existing RIL standards that include explicit worker safety procedures (e.g. Elias et al., 2001), and with Forest Stewardship Council certification (https://us.fsc.org/en-us/certification), which includes consideration of social, economic, and ecological outcomes.

If RIL-C practices were fully implemented in all native forest logging concessions in Indonesia, emission reductions achieved would contribute from 8.2% (Level 1) to 11.3% (Level 2) of forest emissions reductions pledged as part of Indonesia's Nationally Determined Contribution to the Paris Climate Agreement. This assumes 26% reductions to Indonesia's Forest Reference Emissions Level of 0.57 $GtCO_{2e}$ yr^{-1} (Republic of Indonesia, 2016). These potential Level 1 emissions reductions are equivalent to the CO_2 emissions from 13 of the coal fired power plants in Kalimantan, Sulawesi, and Papua, plus four of Sumatra's coal power plants. Level 2 RIL-C emissions reductions are equivalent to the same coal power plant emissions plus emissions from the remaining 12 of Sumatra's coal power plants which average higher emissions than those in eastern Indonesia (https://www.carbonbrief.org/mapped-worlds-coal-power-plants).

Furthermore, if 30% of logging concession area is formally and permanently set aside to target the protection of sensitive habitats,

Indonesia could deliver 7.4 million ha of high diversity forest protection, complementing existing protected areas (22.6 million ha) to reach 93% of Indonesia's Aichi target 11 (the effective conservation of at least 17% of lands). While this improved protection within concessions could deliver the majority of Indonesia's remaining protected area targets, a complementary protected areas network would be needed to fully deliver the "ecologically representative" and "well connected" requirements of the Aichi targets. Both these Paris Climate Agreement and Aichi outcomes could be delivered with no reductions in timber yields and substantial improvements in worker safety and sustainability of the natural forest timber sector.

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Appendix. A. Information sent to participants in advance of expert workshop on Level 2 RIL-C

A workshop was held May 28th, 2013 in Tanjung Redeb, Berau, Indonesia with 35 concession managers and regional and pantropical forestry experts. The workshop was designed for expert and practitioner inputs to inform the definition of Level 2 RIL-C implementation levels. The following information (Table A.1) was provided to workshop participants prior to the workshop to structure discussions.

Table A1
Information sent to 35 concession managers and forestry experts, in advance of RIL-C workshop held May 28th, 2013 in Tanjung Redeb, Berau, Indonesia.

Type of change in practice and potential savings in committed emissions (average)	Proposed RIL-C quantitative target for discussion	Issues to be discussed
Felling: Avoid felling defective trees and trees from which logs are unlikely to be recovered due to steep slopes, etc.Bucking: Improved bucking to recover all commercial roundwood.	Trees felled with no log extracted due to defect ≤ 5% of total felled. Trees felled with no log extracted due to inaccessibility ≤ 5% of total felled. No sound log sections ≥ 1 m left in the forest.	Why is no wood harvested from so many felled trees (~30% in our study)? Before felling, how effectively can survey crews and/or fellers identify defective trees with little or no commercial value? Is the chainsaw bar "plunge test" method used? What is the shortest length and smallest diameter log that can be commercially extracted? Why isn't the "butt log" extracted? Can stump heights be lowered to increase volume of commercial "butt logs"?
Felling: Better felling practices in order to reduce wood waste, reduce collateral damage, and reduce skidding distances.	Cut vines on trees to be harvested (and future crop trees) ≥ 6 months before felling. Directionally fell trees to reduce skidding distance, reduce collateral damage, and avoid damage to roundwood.	How many months in advance of felling do survey crews work? Can vine cutting be included as a survey crew responsibility? What are the limitations in availability of training in directional felling? What are the priority goals for directional felling (skid distance vs. log damage vs. collateral damage). To what extent are there trade-offs among these goals?
Skidding: Bulldozer Skidding: Plan skid trails	Dozer skid trail planning to current FSC/TFF standards.	We assume this is feasible in all concessions, since it is currently implemented in FSC concessions, but is this assumption correct?
Skidding; Cable winch skidding: Skid logs with mono-cable winches (MCW) or comparable long-line cable system (e.g., LogFisher)	≥70% of area harvested with long-line cable system.	What is an ambitious yet feasible proportion of felling block areas where MCWs or other long-line cable system can replace bulldozers? Is there another type of winching technology (e.g. LogFisher, truck-mounted winches) that we should be exploring? Is there a risk that use of MCW use will increase the areas accessed within felling blocks (e.g., very steep areas)?
Hauling: Narrower haul roads	Mean width of primary haul road corridors ≤ 24 m; secondary hauls roads should be < 20 m wide.	What is an ambitious yet feasible target for minimal haul road corridor width (includes roadside clearings)? Why is there so much variability in haul road corridor width?

(*continued on next page*)

Table A1 (*continued*)

Type of change in practice and potential savings in committed emissions (average)	Proposed RIL-C quantitative target for discussion	Issues to be discussed
		Are there big opportunities to reduce the total length of haul roads? (we have assumed the answer is no) Does the minimum feasible haul road corridor width depend upon soil type, geology, and/or topography? Is proximity to a gravel pit critical? What is the difference between primary and secondary haul roads?
Hauling: Small roadside log landings	Log landings < 30 m^2 per ha of logged block (< 30% of in CL).	Given that this goal is reached in FSC-certified concessions, we assume that it is feasible, but is this assumption correct?
Set Asides: No logging near permanent streams. No logging on steep slopes. No logging in high conservation value forests (HCVF).	No felling of trees within demarcated buffer zones along perennial streams. No trees felled or bulldozer skidding on slopes > 40% (averaged over 0.25 ha) No trees felled or bulldozer skidding on 10% of cutting block set aside as HCVF.	Can we assume that "conventional logging" ignores all set asides? If not, what are "conventional logging" set aside levels? How are perennial (permanent) streams defined in the field? Are streamside buffer zones demarcated in the field? Over what areas are slopes averaged? What percent of forests are set aside as HCVF for FSC? Are HCVF designated as "no logging" or rather "lower impact logging"? How much of the average cutting block will be out-of-bounds for logging for each set aside rule?

Appendix B. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.foreco.2019.02.025.

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Selective logging emissions and potential emission reductions from reduced-impact logging in the Congo Basin

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ABSTRACT

To estimate carbon emissions from selective logging in Central Africa, we employed the reduced-impact logging for carbon emissions reductions (RIL-C) protocol to quantify baseline carbon emissions from legal timber harvests by source (i.e., hauling, skidding, and felling). We modeled the relationships between emissions and biophysical conditions, logging practices, and forest policies and then used these models to estimate potential emission reductions from full implementation of RIL-C practices. We applied the method in 8 forest management enterprises (FMEs; i.e., concessions) in the Democratic Republic of Congo (DRC), 9 in Gabon, and 6 in the Republic of Congo (RoC). Committed logging emissions expressed per cubic meter of timber harvested (to control for differences in logging intensities) ranged from 0.63 Mg C m^{-3} in a FME in RoC to 4.8 Mg C m^{-3} in a FME in Gabon, with an overall average of 2.1 Mg C m^{-3}. Logging emissions were dominated by damage caused by road and log landing construction (i.e., hauling; 50%) and felling (43%; includes carbon in extracted logs). Total emissions represented only about 9% of unlogged forest biomass carbon stocks. Average emissions were highest in Gabon (2.65 Mg C m^{-3}) followed by DRC (1.84 Mg C m^{-3}) and RoC (1.54 Mg C m^{-3}). Emissions from concessions certified by the Forest Stewardship Council (FSC, N = 6) and those that were not certified (N = 17) did not differ. Nearly half (51%) of logging emissions could be avoided without reducing timber yields if all best examples of RIL-C logging practices observed were applied in the same FME. At the country level, if all FMEs were to utilize these practices, emissions reductions would be 34% in RoC, 45% in DRC, and 62% in Gabon. When combined with country-level logging statistics, emissions from selective logging as currently practiced in the six countries of the Congo Basin are equivalent to 40% of the region's total emissions from deforestation.

1. Introduction

Avoidance of tropical deforestation and forest degradation is recognized as a key climate change mitigation strategy that was formally recognized in the Paris Agreement (Bustamante et al., 2016; Carodenuto et al., 2015). Compared to forest degradation, carbon emissions from deforestation are relatively easy to measure with remote sensing (e.g., Avitabile et al., 2012; Zarin et al., 2016). In contrast, quantifying emissions from degradation (i.e., loss of carbon from forests that remain forests) requires detailed site-specific information that is hard to derive from passive remote sensing imagery (Herold et al., 2011; Ryan et al., 2014; Zhuravleva et al., 2013). Given that field measurements require suitable sampling protocols and financial support, it is no surprise that global emissions from forest degradation are poorly quantified (Agyeman et al., 1999; Baccini et al., 2017; Hosonuma et al., 2012; Morales-Barquero et al., 2015, 2014; Thompson et al., 2013).

Several studies have demonstrated the important contribution of forest degradation to global carbon emissions (Ernst et al., 2013; Hosonuma et al., 2012; Ryan et al., 2012; Zhuravleva et al., 2013). Globally, an estimated 850 million hectares of tropical forest were degraded between 1990 and 2010, emitting 10–40% of total net emissions globally (Houghton et al., 2012). A more recent study suggests that forest degradation is responsible for the majority (69%) of carbon emissions from tropical ecosystems (Baccini et al., 2017). Similarly, regional-scale analyses estimate that 22–57% of total forest emissions are from degradation (Asner et al., 2012; Hosonuma et al., 2012). In tropical Africa, a recent study reported that degradation

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accounted for 81% of emissions (Baccini et al., 2017) of which ~32% was from timber harvests (Hosonuma et al., 2012; Thompson et al., 2013). These studies provide motivation to focus on forest degradation from selective logging but, given that they were based on remote sensing with few field measurements, they do not allow detailed insights into the forest management practices and biophysical factors (e.g., topography, climate, and soils) that influence emissions.

Selective logging occurs in at least 20% of the world's tropical forests (Blaser et al., 2011; Pearson et al., 2017). While most studies on this topic focused largely on the extent of damage to residual stands and carbon emissions (e.g., Medjibe et al., 2011), emissions sources (e.g., from haul roads, skid trails, and collateral damage from felling) are less well quantified (but see Griscom et al., 2014; Pearson et al., 2014), especially in Africa. This data scarcity causes uncertainty in estimates of greenhouse gas emissions from logging. It is known that excessive emissions result from uncontrolled selective logging by untrained crews operating in the absence of detailed harvest plans and without incentives to minimize avoidable stand and soil damage (e.g., Pinard and Putz, 1996; Asner et al., 2005; Pearson et al., 2017). There is a clear need for fine-scale direct measurements of emission-causing activities that allow decoupling of overall emissions estimates into causative activities. This information can be used to design effective ways to reduce these emissions through improved practices.

To minimize the deleterious environmental impacts of selective logging, foresters have pushed for improved harvesting practices for nearly a century (e.g., Bryant, 1923; Dykstra et al., 1996; Ewel et al., 1980; Hendrison and Wageningen., 1990). In 1993 these well-established practices became known as "reduced-impact logging" (RIL) (Putz and Pinard, 1993) with a focus on carbon emission reductions. Overall, RIL practices are intended to minimize the disruption of tropical forest carbon and water cycles via pre-harvest tree selection, cutting lianas before logging, directional felling, and planning of skid trails (e.g., Pinard et al., 1997; Miller et al., 2011). Positive effects of the adoption of RIL techniques on rates of post-logging biomass and timber volume recoveries are substantial and well- recognized in the literature (e.g., Miller et al., 2011; Vidal et al., 2016). Knowing this, efforts should now focus on developing consistent guidelines and practices, standardizing worker training, reducing worker turnover and injuries, and developing suitable methods to measure and monitor carbon emissions (Griscom et al., 2014; Pearson et al., 2014). Such an approach should allow disaggregation of the recommended RIL practices so as to measure the emissions reductions associated with each.

Tested in Indonesia, the RIL-C protocol focuses on carbon emission reductions (Ellis et al., 2016; Griscom et al., 2014) to reflect increased concerns about climate change as well as increased opportunities to benefit financially from those reductions (e.g., REDD+, voluntary carbon markets, Nationally Determined Contributions to the UN Paris Agreement, and corporate commitments) (VCS, 2016). The carbon emission reduction benefits of RIL derive from increased efficiency and from respect for rules related to riparian buffer zones, slope restrictions, and sometimes by protecting big trees. Although following RIL guidelines does not assure long-term timber yield sustainability (e.g., Ruslandi et al., 2017), their adoption can reduce adverse environmental impacts (e.g., soil compaction and erosion, and collateral stand damage) and enhance worker safety. The RIL-C monitoring protocol allows disaggregation and measurement of the emissions-causing practices (felling, yarding, and hauling) and allows estimation of potential emission reductions from their improvement. Nonetheless, there remains a need to set performance levels by which the implementation of RIL practices can be assessed and compared with the baseline emissions to estimate RIL additionality and effectiveness in emissions reductions.

Here we apply the RIL-C monitoring protocol developed for Indonesia (Griscom et al. 2014) to Congo Basin forests to quantify logging emissions and potential emission reductions. The objectives of this study are: (1) to quantify emissions from hauling, skidding, and felling in 23 concessions that span three Congo Basin countries; (2) to analyze the relationships between total emissions and a range of explanatory variables including biophysical and spatial variables, logging practices, and policies; and, (3) to estimate potential emissions reductions with full RIL implementation. We place our modeled estimates of potential RIL reductions in emissions into a broader context by estimating the magnitude of potential reductions across three sampled countries (Democratic Republic of Congo, Gabon, and Republic of Congo) as a contribution to each countries' pledge, as signatories of the Paris Climate Agreement, to 20–50% emissions reductions, as specified in their Nationally Determined Commitments (NDC).

2. Methods

2.1. Reduced-impact logging for carbon emission reductions (RIL-C) methodology

The RIL-C protocol was developed to measure emissions from selective logging in Indonesia by its main source (either felling, skidding, or hauling), and to estimate the possible emissions reductions from adoption of improved logging practices (Ellis et al., 2016; Griscom et al., 2014). This protocol was approved for use in East and North Kalimantan by Verified Carbon Standard (VCS). It includes predetermined additionality benchmarks and crediting baselines that serve to reduce operational costs while mitigating emissions through simplified monitoring, reporting, and verification. The method divides logging-induced carbon emissions into those from felling, skidding (i.e., timber yarding), and hauling (i.e., haul roads and log yards); committed emissions (Mg C) are expressed per cubic meter of timber harvested and per hectare. We estimated changes in biomass pools and related emissions directly by measuring losses in live biomass and incidental damage to other trees. In this sense, the method follows the gain-loss approach as opposed to estimating the difference in carbon emissions and removals from pre- and post-logged forest (Plugge and Köhl, 2012).

2.2. Site descriptions

Democratic Republic of Congo (DRC): The DRC's 155 million hectares of forest constitutes one third of all forests in the Congo Basin; annual deforestation rates reached 4% between 2000 and 2014 in forest with > 50% tree crown cover (Abernethy et al., 2016). In 2016, 81 concessions were operational of which 57 operated with timber licenses in an area of ~ 10.7 million ha, while 21 concessions were timber licenses to communities (~4 million ha), and three timber licenses (394,359 ha) on hold by the government (WRI–Domaine Forestier de la RDC 2016). Of all these industrial logging concessions, at the time of this study (2017) only seven had validated management plans and none were Forest Stewardship Council (FSC) or PEFC certified (Blaser et al., 2011; de Wasseige et al., 2015). Large-scale industrial timber harvesting is not fully developed in DRC due to lack of infrastructure and political instability. Most of the logs are exported towards Europe and Asia, with little pre-export processing. It should be noted that artisanal and illegal logging greatly increased during the 1996–2002 conflict and was often followed by deforestation (DRC – RPP 2010). DRC is the only country in the region that allocates logging concessions to communities, but their operations are rudimentary and yield little timber.

Republic of Congo (RoC): The RoC's 24 million ha forest covers 71% of the country with an annual deforestation rate of 1.6% between 2000 and 2014 (Abernethy et al., 2016). To promote sustainability, in 2000 the government of RoC required logging concessions to operate according to government-approved forest management plans. In mid-2009, there were 52 large-scale concessions covering nearly 12 million hectares, 8 million in the northern region and about 4 million in the south and central regions of the country. Following governmental policies, these concessions were often divided into management units of about 50,000 ha (Blaser et al., 2011). In 2016, 51 logging concessions covered 12.6 million ha of which about 4.6 million ha were under

government-approved management plans and a total of 3 million ha (or 12 FME) were FSC certified (de Wasseige et al., 2015; Karsenty and Ferron, 2017). Almost all logging concessions are owned by foreign companies, with little development of community-owned concessions.

Gabon: The timber industry plays an important role in the economy of Gabon in terms of its contributions to GDP, foreign exchange, and employment. With 24 million ha of forest, Gabon is the most forested country in the region (88%) and, in 2014, suffered the lowest deforestation rate of 1.1% (Abernethy et al., 2016). In 2015, 150 companies operated with timber licenses that covered 14.3 million ha (Karsenty and Ferron, 2017) or > 50% of total forested area (de Wasseige et al., 2015). Gabon's timber sector, which is dominated by foreign companies, exported about 4 million cubic meters of industrial logs in 2000 out of which 70% was in the form of raw round logs. In 2009, prior to its 2010 log export ban, Gabon produced an estimated 3.4 million m^3 of industrial logs, out of which 1.87 million m^3 of logs and 157,000 m^3 (roundwood equivalent) of sawnwood were exported, which made Gabon the world's second largest exporter of tropical hardwoods in that year (Blaser et al., 2011; Rana and Sills, 2017). As of 2015, 2.4 million ha of forest in 25 FMEs were FSC certified (17%), 50% of forest was included in management plans registered with the government, and the remainder was being harvested under temporary logging permits (de Wasseige et al., 2015). Efforts to promote sustainable forest management, certification, and log export bans are endorsed by the government as ways to promote economic development and reduce deforestation.

2.3. Field sampling

Data were collected in 23 commercial forest concessions in the Congo Basin (nine concessions in Gabon, eight in DRC, and six in RoC) that were selected to cover a wide range of management categories, logging practices, and pre-logging biomass carbon stocks (Table 1). Sample blocks were selected with a stratified random procedure to ensure a representative sample of FMEs based on factors such as their size, soil type, elevation, carbon density, certification status, and worker training in RIL practices. If a randomly selected sample block was inactive or in accessible, it was replaced by a new randomly selected sample block from the same stratum. Concessions were categorized as FSC-certified (FSC), with registered management plans (MP), or with only temporary logging permits (TP). For consistency, we combined MP and TP as non-certified concessions to compare with FSC-certified concessions throughout. In annual cutting blocks in each FME we measured the widths of active roads and road corridors at 20 locations. In addition, and where possible we measured the areas of 10 log yards in each cutting block. Skidding and felling were measured in a randomly selected 50 ha sub-block in each concession. For each sampled sub-block, we measured skid trail lengths, mapped skid trail networks, and measured trees ≥ 10 cm DBH that were damaged by skidding in 15 skid trail plots. To account for felling damage, we first selected 30% (309 of the 1039) of the felled trees for measurement of dendrometric variables (stump, tree and log diameter, height, and diameter of any hollows or heartrots), then measured any trees ≥ 10 cm DBH that were killed (e.g. uprooted or snapped) or that suffered bark or crown damage during felling. Finally, the biomass carbon stocks of unlogged forest were estimated with data from 15 prism plots (345 in total) measured in an adjacent, unlogged block within each sampled concession. Detailed information on field measurements can be found in Griscom et al. (2014) and conversion of field measurements into estimates needed for application of the carbon accounting equations are presented in the supplemental material (S1 – Logging Equations in Ellis et al. 2019, in this issue).

2.4. Carbon emissions accounting

Total emissions from timber harvests were estimated as the sum of three sources: (1) hauling (H), which includes log landings, haul roads, and road corridors; (2) skidding (S), calculated as emissions from skid trail plots times the length of the skid trail network; and (3) felling (F) that combines emissions from harvested trees (H) and those that suffered collateral damage. Committed emissions from above- and below-ground biomass are expressed both as Mg C m^{-3} and as total Mg of carbon emitted per Mg of timber harvested (referred to as the Carbon Impact factor or CIF), to account for differences in wood densities and logging intensities (Feldpausch et al., 2005). To estimate logging emissions per hectare (Mg C ha^{-1}), total emissions (E) were divided by the areas of sampled blocks. We used equation (12) from Fayolle et al. (2018) and wood density data from the Global Wood Density database to calculate aboveground biomass, while we estimated below-ground biomass from shoot biomass using equation (1) from Mokany et al. (2006). We converted biomass into carbon using a standardized carbon fraction of 0.47 (Brown and Lugo, 1992; Chave et al., 2014; Fayolle et al. 2018). Details about the methods used to estimate committed emissions from field measurements are provided in the supplementary information (S1).

Hauling emissions *(H)*. Hauling emissions included emissions from destruction of trees ≥ 10 cm for creation of logging roads (R) and log landings (L). To calculate the area of forest cleared for haul roads, we measured the width of haul road surfaces and adjacent strips of felled and bulldozed trees (hereafter 'haul road corridors') at 100 m intervals. While in the field we distinguished between new haul roads (clearing of forest) and old haul roads (re-clearing of previously cleared road corridors). Emissions from new haul roads (R_N in Mg C) were estimated as:

$$R_N = \frac{\bar{w_R} * l_{NR}}{10000} * BD$$

where $\bar{w_R}$ is the mean haul road corridor width (m), l_{NR} is the length of newly constructed haul roads allocated to a sample block (m), and BD is the biomass carbon density of adjacent unlogged forest (Mg C ha^{-1}). Emissions from re-clearing old roads (R_O in Mg C) were estimated as:

$$R_O = \frac{(\bar{w_R} - \bar{w_{AR}}) * l_{OR}}{10000} * LR * SS$$

where $\bar{w_{AR}}$ is the mean measured haul road corridor width (m), $l_{OR}$is the

Table 1
Key characteristics of the 50 ha areas sampled in 23 selected concessions grouped by country and management status: FSC = certified by FSC; MP = concession with a management plan validated by the government; and, TP = concession operating with a temporary permit but working toward completion of a management plan.

Country	Status	N	Mean slope (%)	Mean elevation (m)	Trees harvested (# ha^{-1})	Volume harvested (m^3 ha^{-1})	Mean unlogged carbon density (Mg C ha^{-1})
DRC	MP	3	18	483	1.2	21	129
	TP	5	16	467	2.4	51	207
Gabon	FSC	3	9	353	3.2	46	225
	MP	3	8	360	1.4	56	126
	TP	3	7	377	1.1	22	250
RoC	FSC	3	25	425	3.3	40	157
	MP	2	24	408	3.1	18	161
	TP	1	26	420	2.4	38	179

Table 2
Mean estimates (± SE) of field variables measured to estimate emissions from hauling, skidding, and felling in 23 logging concessions (DRC = 8, Gabon = 9, and RoC = 6) of which 6 were FSC certified. The waste index estimates the percentage of wood left in forest due to poor utilization of merchantable portions of logs. Treatments with the same superscripts did not differ (ANOVA Tukey's HSD, P > 0.05).

Country	Mean road width (m)	Mean corridor width (m)	Skid trail width (m)	Mean DBH (cm)	Mean extracted log length (m)	Waste (%)
DR Congo,	5.5(0.4)[a]	23.3(2.03)[a]	3.8(0.1)[a]	118(6.2)	18(1.4)[a]	8.1(0.4)[a]
Gabon	5.9(0.4)[b]	24.0 (2.1)[b]	5.8(0.3)[a,b]	107(4.7)	21(1.1)[a]	30.1(0.8)[b]
R of Congo	8.5(0.2)[a,b]	34.8(3.3)[a,b]	3.7(0.3)[b]	117(11)	19(1.3)[a]	3.5(0.2)[a]
Status						
FSC	7.5(0.6)[a]	28.5(4.7)[a]	4.5(0.5)[a]	106(3.9)	20(1.5)[a]	17(1.1)[a]
Non-FSC	6.1(0.4)[a]	26.0(1.6)[a]	4.5(0.3) [a]	116(5.2)	19(0.9)[a]	15(0.4)[a]
Mean (N = 23)	6.4(0.3)	27(1.7)	4.5(0.3)	113(4.1)	19.5(0.7)	16(0.4)

length of re-cleared old haul road, LR is the average logging rotation with 30 years as the default value, and SS is the secondary forest carbon sequestration rate (Mg C ha^{-1} yr^{-1}) with a default value of 2.726 (Bonner et al., 2013). Finally, emissions from log landings (L, Mg C) were estimated from the average log landing area in each concession (ha), the number of log yards per length of haul road surveyed (m^{-1}), and length of haul roads allocated to a sample block (m).

Skidding (S). Skidding emissions were computed as the product of the emissions from the average skidding collateral damage (CD_S) per area (Mg C ha^{-1}) and the area of skid trails (A_S). GPS tracks of skid trail centerlines were used to calculate meters of skid trail per ha, adjusted to cutting block area based on the areas of gaps and overlaps for adjacent skid trail access areas. Skidding emissions were therefore calculated as:

$$S = C\bar{D}_S * A_S$$

where the average skidding collateral damage per area of skidding in sample block ($C\bar{D}_S$, Mg C ha^{-1}) was derived from the skidding collateral damage for all skidding damaged trees measured (Mg C) divided by area of all skid plots measured (ha).

Felling (F). Felling emissions represent those that occurred when a tree or several trees were felled and created a canopy gap. The resulting emissions are from the above-ground and below-ground biomass of stumps and portions of felled trees left as dead wood in the forest, and adjacent trees ≥ 10 cm killed or severely damaged. Felling emissions were calculated as:

$$F = CD_F + HT$$

where CD_F is the collateral damage emissions from felling in the sampled block (Mg C), calculated from the mean collateral damage emissions per measured tree (Mg C tree^{-1}) multiplied by the number of felled trees (stumps) measured. HT is the emissions from harvested trees (Mg C), derived from the average harvest tree emissions ($\hat{HT}$) per measured tree in the sampled block (Mg C tree^{-1}) multiplied by the number of felled trees (stump count). Additional details about calculation of gross committed emissions are summarized in S1 (logging equations file).

2.5. Statistical analyses

We fitted linear models on untransformed data to predict felling emissions due to collateral damage, and extracted and unextracted log emissions at the felling gap level (N = 309). We also fitted regression models for total emissions in Mg C m^{-3} at the concession level (N = 23). We considered log diameter, mean log length, volume of wood extracted, tree density, logging intensity, and slope as potential predictor variables. We specified country or certification status as qualitative variables in the model because of the varied harvest treatments in use over time which combined to affect C stocks. We fitted our models using the gls () function in R version 3.3.1 (R Core Team, 2016). Our final model for total emissions as a function of logging intensity is:

$$Y = \beta_0 + \beta_1 X_1 + \beta_2 X_2 + \beta_3 X_3 + \epsilon$$

where

Y = In(TotalEm3)

X_1 = In(Logging Intensity)

X_2 = indicator variable for DRC ($X_2 = 1$ if country ≡ DRC, 0 otherwise)

X_3 = indicator variable for Gabon ($X_3 = 1$ if country ≡ Gabon, 0 otherwise)

According to this model, three parallel lines on the logarithmic scale are produced because when

- country ≡ DRC, $Y = \beta_0 + \beta_1 X_1 + \beta_2 + \epsilon$
- country ≡ Gabon, $Y = \beta_0 + \beta_1 X_1 + \beta_3 + \epsilon$
- country ≡ RoC, $Y = \beta_0 + \beta_1 X_1 + \epsilon$

Parallel lines are produced bacause β_2 and β_3 are just offsets to the RoC intercepet term, β_0 on the logarithemic scale.

2.6. Emissions reduction scenarios with RIL-C implementation

We considered the following two scenarios about RIL-C implementation based on our field data: (1) a reference scenario that shows the expected emissions if current relationships and trends continue; (2) a 'RIL-C Level 1 scenario' that models the impacts on emissions if the FMEs achieved the best observed performance levels for each of the four RIL-C practices.

3. Results

3.1. Field measurements

The sampled concessions vary substantially in physical features and timber extraction methods (Table 2). Across all sites, logging road corridor width averaged 27 ± 1.7 m (mean ± 1 SE throughout), with a mean active surface width of 6.4 ± 0.3 m. Road were widest in RoC (34.8 ± 3.3 m and 8.5 ± 0.2 m, respectively; Table 2). Skid trails were substantially wider in the nine concessions in Gabon (5.8 ± 0.3 m), than in the eight concessions in DRC (3.8 ± 0.1 m), or the six concessions in RoC (3.7 ± 0.3 m). Maximum slopes measured 15 m up and down-hill from each stump were 10% in Gabon, 28% in RoC, and 30% in DRC, with corresponding means of 8%, 25%, and 17% respectively.

Of the 309 felled trees measured across the 23 concessions, the largest were around 200 cm DBH (mean = 113 ± 4.1) and the longest logs extracted were 35 m (mean = 19.5 m ± 0.7 l; Table 2). The proportions of trees felled from which no logs were extracted ranged from 3.5% in RoC to 30.1% in Gabon (Table 2). The 6 FSC concessions felled and abandoned trees without extracting any wood just as often as the 17 non-certified concessions (17% and 15% of felled trees, respectively; P = 0.12).

Table 3
Committed emission from hauling, skidding, and felling by country and averaged by concession status (mean ± SE). The carbon impact factor (CIF; see SI) is expressed in units of Mg C emitted per Mg C in the harvested wood. Treatments with the same superscripts did not differ (ANOVA Tukey's HSD, P > 0.05 and 95% CI).

Country	Hauling Emissions Mg C m^{-3}	Skidding Emissions Mg C m^{-3}	Felling Emissions Mg C m^{-3}	Total Emissions Mg C m^{-3}	Carbon Impact Factor Mg C Mg C^{-1}
DR Congo (N = 8)	0.52 ± 0.1^{a}	0.2 ± 0.05^{a}	1.1 ± 0.15^{a}	1.84 ± 0.3^{a}	5.8 ± 0.97^{a}
Gabon (N = 9)	1.60 ± 0.4^{b}	0.11 ± 0.04^{a}	0.97 ± 0.22^{a}	2.65 ± 0.5^{a}	10.7 ± 1.9^{a}
R of Congo (N = 6)	0.93 ± 0.3$^{a, b}$	0.07 ± 0.04^{a}	0.54 ± 0.06^{a}	1.54 ± 0.3^{a}	6.8 ± 2.5^{a}
Status					
FSC (N = 6)	1.20 ± 0.3^{a}	0.10 ± 0.04^{a}	0.84 ± 0.3^{a}	2.14 ± 0.6^{a}	9.7 ± 3.1^{a}
Non-FSC (N = 17)	0.98 ± 0.2^{a}	0.14 ± 0.03^{a}	0.94 ± 0.11^{a}	2.05 ± 0.3^{a}	7.3 ± 1.1^{a}
Mean (N = 23)	1.04 ± 0.2	0.13 ± 0.03	0.91 ± 0.11	2.1 ± 0.25	8.0 ± 1.1

3.2. Committed emissions from selective logging

Selective logging in the 23 concessions generated a mean of 2.1 ± 0.25 Mg C m^{-3} (18.4 Mg C ha^{-1}) of committed emissions, which represents a transfer of 9% of live above- and below-ground tree carbon biomass into necromass based on a mean pre-harvest forest biomass of 202 Mg C ha^{-1} (Table 3). Emissions per cubic meter of timber extracted in Gabon were 30% higher than in DRC (2.65 ± 0.5 Mg C m^{-3} in Gabon vs 1.84 ± 0.3 Mg C m^{-3} in DRC) and 42% higher than in RoC (1.54 ± 0.3; Table 3). Carbon emissions per cubic meter of timber harvested did not differ between FSC-certified and non-certified concessions (Table 3).

Felling emissions, the sum of emissions from unextracted logs and portions thereof, collateral damage, and extracted wood, contributed 43% of the total emissions (Table 3). Of the felling emissions, the felled tree remainder, or portion of felled trees left on-site, represented 22%, collateral damage caused by felling of trees selected for harvest represented 13%, and 9% of emissions were from the extracted wood. Hauling emissions that combine emissions from roads and log yards accounted for 50% of emissions, of which 45% of were from logging roads (only 5% from log yards), while skidding damage contributed only 6%.

Committed emissions from hauling (1.04 ± 0.2 Mg C m^{-3}; Table 3) varied almost three-fold between the lowest value in DRC (0.52 Mg C m^{-3}) to the highest in Gabon (1.6 Mg C m^{-3}). In all cases, log landings contributed relatively little to hauling emissions (11% in Gabon, 10% in DRC, and 5% in RoC). Road emissions represented as much as 60% of total emissions in Gabon and RoC compared to < 28% of total emissions in DRC (Table S1). FSC concessions showed somewhat lower hauling emissions than non-certified concessions, but the difference was not significant (Table 3). Committed emissions from skid trails also differed by country from 4% and 5% of total emissions in Gabon and RoC, respectively, to 11% in DRC.

Committed emissions from logging gaps varied by less than a factor of two between the lowest felling damage in RoC and to the highest in DRC (Table 4). Emissions from tree remainders (i.e., the stumps, tops, and logs left in the forest) accounted for 50% (DRC) and 52% (Gabon and RoC) of the total emissions per tree harvested. Collateral damage emissions varied from 34% (Gabon) and 30% (DRC) to just 20% (RoC), while emissions from extracted wood contributed 15% in Gabon, 23% in DRC, and 26% in RoC. FSC concessions showed 9% lower emissions from felling than non-certified concessions. Volumes extracted per tree ranged from 10.5 m^3 (Gabon) to 13.3 m^3 (RoC) with extracted masses of 6.7, 8.6 and 9.6 Mg C for RoC, Gabon, and DRC, respectively.

We observed large differences among concessions in carbon emissions per unit volume of timber extracted, which ranged from 4.8 Mg C m^{-3} in concession GAB9 to 0.63 Mg C m^{-3} in RoC6 (Fig. 1). Emissions from roads and the remainders of cut trees represented the major sources of emissions, followed by collateral damage and extracted timber, while log yards and skidding emissions remained the lowest. For some concessions in Gabon (GAB9 and GAB6), road emissions were three to four-times higher than from the concessions with the lowest emissions from this source.

The proportion of above and below ground biomass carbon of unlogged forest emitted from all logging sources averaged 9% across all concessions with 10% for the highest intensity logging (RoC and Gabon), and as little as 7% in DRC (Table 5).

3.3. Landscape characteristics and logging emissions

We explored for the factors responsible for differences in committed emissions by concession, management status, and country. In the 23 concessions, committed emissions per m^3 of timber extracted decreased with timber volumes harvested per ha ($P = 0.04$; Fig. 2a). In contrast, committed emissions per ha were not related to harvested volumes ($P = 0.84$; Fig. 2b). Emissions did not vary with certification status ($P = 0.62$). However, when fitting separate models, committed emissions per m^3 harvested decreased when volume of wood extracted increased, representing 81% the variation among the 6 FSC concessions ($P = 0.02$) and only 7% of the variation among the 17 non-certified concessions ($P = 0.29$). The decrease in committed emissions per ha did not differ as a function of certification status ($P = 0.35$). For FSC concessions, volumes extracted explained 30% of the variation in committed emissions per ha ($P = 0.26$), while it explained only 2% of the variation in non-certified concessions ($P = 0.60$).

Emissions did not increase strongly with size of felled trees

Table 4
Timber volumes extracted and emissions (mean ± 1 SE) per felled tree and the resulting emissions from collateral damage aggregated by country and certification status.

Country	Volume extracted per tree (m^3)	Carbon extracted tree (Mg C)	Collateral damage emissions (Mg C m^{-3})	Extracted wood emissions (Mg C m^{-3})	Tree remainder emissions (Mg C m$^{-3)}$	Felling emissions (Mg C m^{-3})
DR Congo	11.1 (1.3)	9.6	0.33 (0.07)	0.25 (0.02)	0.55 (0.09)	1.1 (0.15)
Gabon	10.5 (2.1)	6.7	0.33 (0.09)	0.15 (0.01)	0.5 0.13)	0.97 (0.22)
RO Congo	13.3 (3.4)	8.6	0.11 (0.03)	0.15(0.02)	0.28 (0.04)	0.54 (0.06)
Status						
FSC	10.1 (2.3)	7.5	0.23 (0.11)	0.16 (0.02)	0.45 (0.18)	0.84 (0.3)
N-FSC	11.8 1.5)	8.5	0.28 (0.05)	0.19 (0.02)	0.46 (0.06)	0.94 (0.11)
Mean (N = 23)	11.4 (1.3)	8.2	0.27 (0.05)	0.18 (0.01)	0.46 (0.06)	0.91 (0.11)

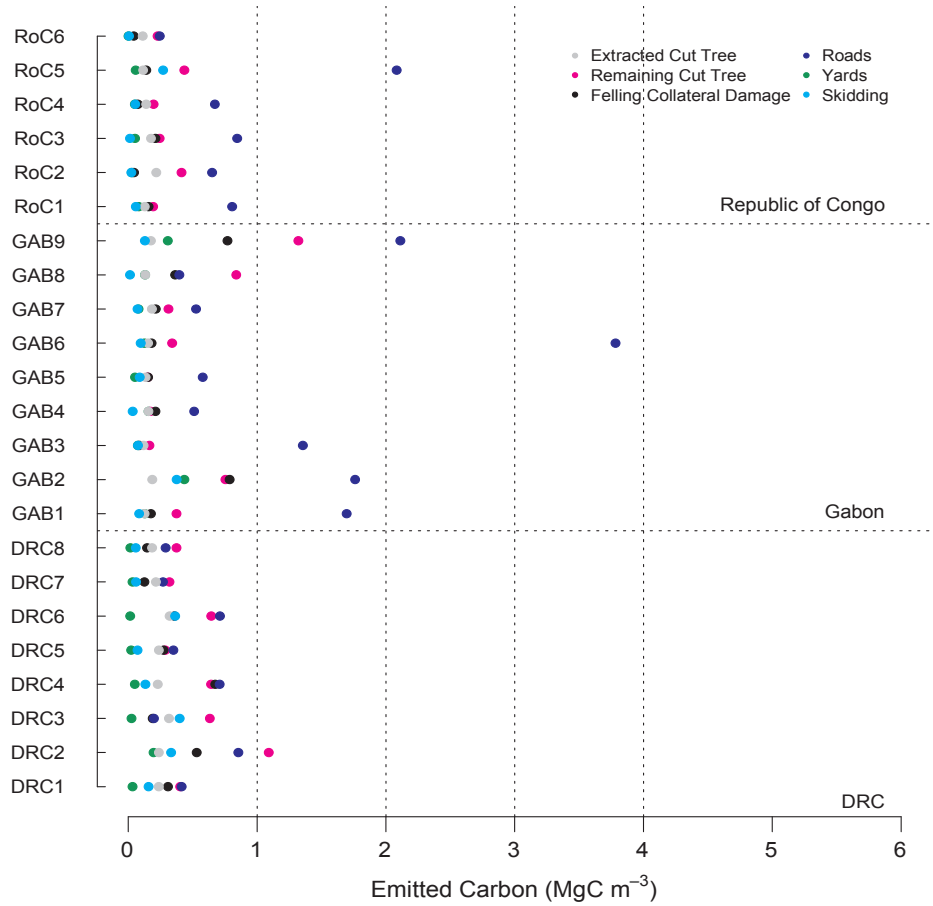

Fig. 1. Committed emissions (biomass carbon transformed into necromass) by six categories of logging impacts in the twenty-three concessions sampled in the three Congo Basin countries.

(P = 0.11); the linear relationship only explained 12% of the variation. Likewise, there is only a weak relationship between collateral damage and the number of felled trees in felling gaps ($P = 0.36$). Emissions varied with neither certification status nor country.

Fifty-two percent of the variability among concessions in felling emissions was explained by the densities of trees felled for harvest ($R^2 = 0.54$, $P < 0.001$; Fig. 3). When concession status (i.e., certified or not) and country were added as variables, the model did not improve.

Although felling accounted for almost half of the overall emissions from logging, there was no relationships between felling emissions and either diameter of trees harvested or slope (8–30%) near the trees (Fig. 4). However, collateral damage emissions at the tree level increased with slope in DRC concessions ($R^2 = 0.44$; $P < 0.001$), but not in Gabon ($R^2 = 0.02$; $P = 0.14$) or RoC ($R^2 = 0.08$; $P < 0.001$). Felling emissions per m^3 extracted increased with slope at the concession level ($R^2 = 0.24$; $P = 0.02$). Harvested trees in Gabon and DRC concessions were typically larger than those in the RoC (Fig. 4), but when country was included as a variable in the model, no differences in tree size were detected.

3.4. Potential for emissions reductions with RIL-C

No single concession had the lowest emissions for each RIL-C activity, but based on the best-performing concessions for each RIL-C activity (e.g., the concession with the lowest felling emissions), the overall Level 1 emissions reduction would be 51% (Table 6). Most of those reductions would be from not felling trees from which no timber is extracted and from reduction of road corridor widths to 22 m. Additional emissions reductions could be achieved through better planned and shorter skid trails, especially in DRC where skidding contributed 11% of total emissions.

Table 5
Committed emissions per ha from selective logging expressed relative to the pre-harvest carbon density of 202 Mg C ha^{-1}. Treatments with the same superscripts did not differ (ANOVA Tukey's HSD, P > 0.05 and 95% CI).

Country	Emissions (Mg C m^{-3})	Emissions (Mg C ha^{-1})	Carbon Impact Factor, Mg C emitted per Mg C extracted	Biomass carbon stocks (Mg C ha^{-1})	Proportion of biomass carbon transferred to necromass
DR Congo	1.84(0.3)	13.6(1.9) [a]	5.8(0.97)	202	0.07
Gabon	2.65(0.5)	20.8(1.9) [b]	10.7(1.9)	202	0.10
R of Congo	1.54(0.3)	21(1.8) [b]	6.8(2.5)	202	0.10
Status					
FSC	2.14(0.6)	19.7(2.1) [a]	9.7(3.1)	202	0.10
Non-FSC	2.05(0.3)	17.9(1.6) [a]	7.3(1.1)	202	0.09
Mean (N = 23)	2.1(0.25)	18.4(1.3)	8(1.1)	202	0.09

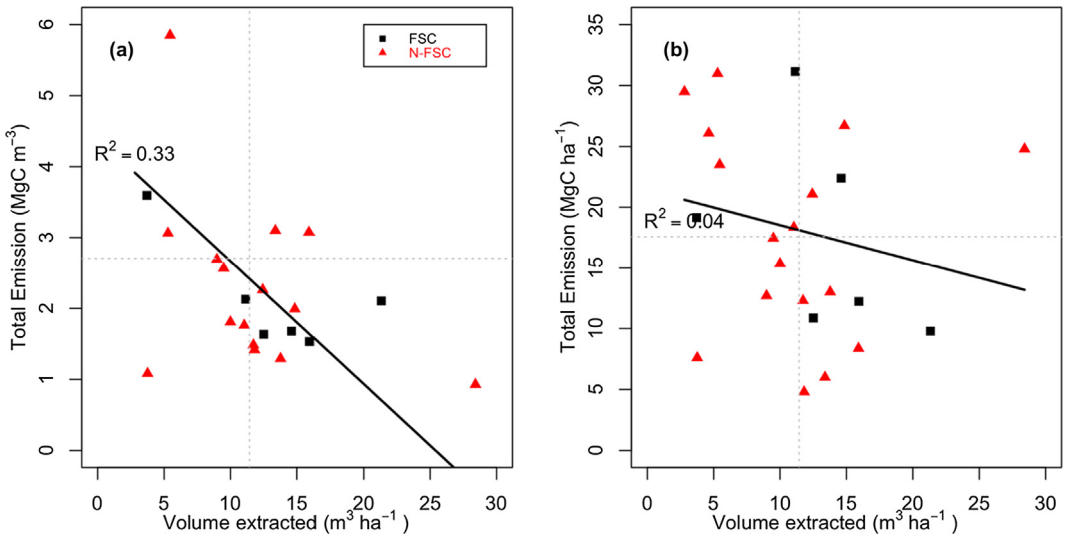

Fig. 2. Total emission per cubic meter of timber extracted (a) and per hectare (b) as a function of harvest intensity expressed as volume per hectare in FSC certified (squares) and un-certified concessions (triangles). With increases in logging intensity, emissions decreased per cubic meter ($P < 0.01$, $R^2 = 0.33$) but did not decrease per hectare ($P = 0.35$, $R^2 = 0.04$). Expressed in either way, there was no apparent relationship with certification status ($P > 0.89$).

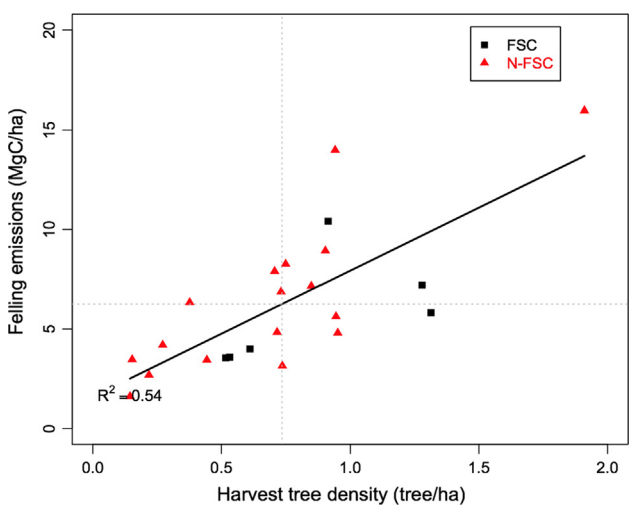

Fig. 3. Simple linear regression model between felling emissions and the number of harvested trees per ha in sampled block for FSC-certified and un-certified (N-FSC) concessions. Harvest tree density explained 54% of the variability among concessions in per ha felling emissions ($P < 0.001$).

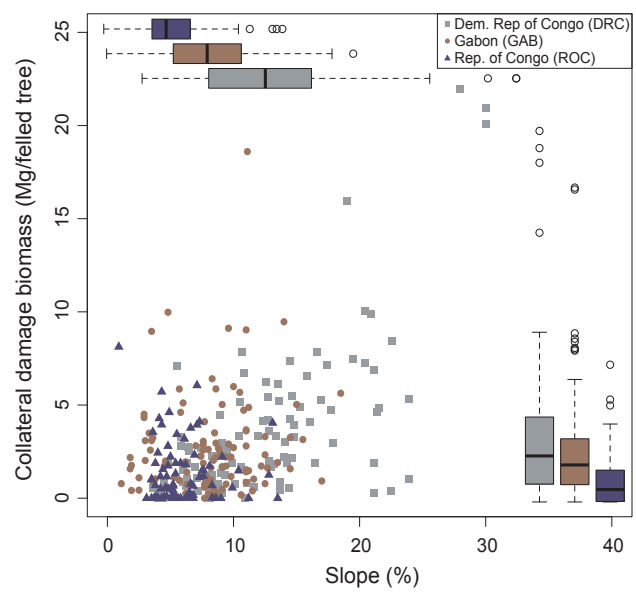

Fig. 4. Slope explains only 27% of the variation across the three countries in collateral damage emissions per felled tree ($R^2 = 0.27$; $P < 0.001$), but both collateral damage and slope differed among countries (Tables 1 and 4). There is no correlation between collateral damage emissions and slope in Gabon and RoC ($R^2 < 0.01$; $P \geq 0.2$), where slopes were fairly gentle, but a positive correlation in DRC ($R^2 = 0.44$; $P < 0.001$) where slopes were often steep. Grey squares = concessions in DRC (n = 102 trees), orange circles = Gabon (n = 135 trees), and purple triangles = RoC (n = 75 trees). Boxplots show variation in slope and collateral damage emissions per country; outliers represented by open circles.

Emission reductions potentials were highest in Gabon (62% with Level 1 implementation), 45% in DRC, and 34% in RoC (Fig. 5). With 53% and 58% of total emissions from roads in Gabon and RoC, respectively, concessions in those countries could reduce their road-related emissions by more than half if their roads were the width of the four best concessions in DRC, one certified concession in Gabon, and one uncertified concession in RoC. All three countries have the potential to reduce emissions from collateral damage emissions by 52%, 43% by not felling trees that yield no timber, and 29% by better bucking to maximize wood extraction.

4. Discussion

4.1. Selective logging intensity and stand damage

We estimate that committed emissions from selective logging in the Congo Basin average 18.4 Mg C ha^{-1}, 2.1 Mg C m^{-3}, or 8 Mg C Mg C^{-1} (destroyed biomass per m^3 of timber harvested). This impact represents a transfer of 9% of above- and below-ground tree biomass to necromass. These estimates are similar to those reported by Pearson et al. (2014) for RoC (8.9 Mg C ha^{-1} and 0.99 Mg C m^{-3}), Indonesia (1.49 Mg C m^{-3}), and Guyana (2.33 Mg C m^{-3}). In terms of per hectare reductions in biomass due to selective logging in Gabon, Medjibe et al. (2013) reported a loss of 2.9% in an FSC-certified concession and 6.3% in a nearby uncertified concession for an average of 4.1% average loss of pre-logging biomass (i.e., 17.2 Mg C ha^{-1} or ~ 10 Mg C ha^{-1} based on our methods). Our estimated average committed emission is about 51% higher than the results from these two previous studies but about 64% to 80% lower than those reported by Griscom et al. (2014) for East

Table 6
Emissions reductions from demonstrated (Level 1) target implementation levels for four RIL-C practices. We selected the mean of the best recorded concessions with the lowest values for each operation.

Logging Activity Category	Emissions Category	Mean Emissions Baseline (Mg C m^{-3})	RIL-C Level 1 Emissions Reductions (Mg C m^{-3})	Level 1 Implementation description
Felling and log recovery: Includes pre-harvest inventory, felling, bucking and extraction practices	Emissions resulting from collateral damage	0.27	0.14	Best recorded used directional felling, liana cutting (RoC concessions), and worker training
	Emissions from trees felled and abandoned	0.46	0.23	Avoided felling hollow trees
	Emissions from trees felled with some volume extracted	0.18	0.13	Maximize extraction of wood so that only 5–10% of merchantable wood is left in forest)
Hauling: well-planned road network, narrower haul roads and smaller log landings.	Emissions from clearing road corridors and log landings. Includes emissions from poorly planned road networks and road edge tree deaths	1.04	0.44	Mean road corridor width < 22 m and find the optimal ratio between lengths of roads and skid trails to shorten the former (data not shown)
Skidding: well-planned skid trail networks	Emissions from skidding damage	0.13	0.106	Grand mean of best recorded (concessions in Gabon and RoC)

Kalimantan, Indonesia (51.1 Mg C ha^{-1}), Mazzei et al., 2010 for Brazil (94.5 Mg ha^{-1}), and Pinard and Putz (1996) for Malaysia (104 Mg ha^{-1}).

In the Pearson et al. (2014) study, felling emissions, which include logging damage and extracted log emissions expressed per cubic meter of timber harvested dominated the total emissions in Indonesia (55%), Guyana (58%) and RoC (76%). Griscom et al. (2014) estimated that 59% of committed emissions from selective logging in Indonesia were from felling (38% was from the remainder of felled trees and 21% from collateral damage), 24% from skidding, and 16% from hauling. We estimated 43% of emissions in the three Congo Basin countries we studied were from felling, 50% from hauling, and only 6% from skidding, although these factors varied by country. Elsewhere in Gabon, Medjibe et al. (2013) reported 13.1–24.2 Mg ha^{-1} from felling, 28.8–54.0 Mg ha^{-1} from the extracted logs, and 5.9 – 12.1 Mg ha^{-1} from skidding (road emissions were not reported). These three factors from Medjibe et al. (2013) are similar to those we obtained for Gabon, despite the high and variable extraction rates among the sampled concessions.

Unlike the finding of Medjibe et al. (2013) in Gabon, we found that average emissions in FSC-certified and non-certified concessions did not differ, but FSC concessions showed somewhat higher roads emission and lower emissions from felling and skidding. One FSC concession in Gabon (GAB9) showed the highest emissions of all 23 concessions studied. To assure that RIL practices are employed and to track emissions reductions, we recommend that FSC auditors employ a version of the RIL-C sampling protocol.

Emissions from felling varied substantially among the 23 concessions distributed across the three countries (Fig. 1). This variation was observed despite similar densities of harvested trees (Fig. 3), and the lack of a relationship between felled tree sizes or the number of felled trees per gap and logging emissions. We observed variation in collateral damage emissions among countries with differences slope ranges but overall, slope did not explain much variability in emissions, (Fig. 4) as reported elsewhere (Griscom et al. 2014; Putz et al. 2018). However, in DRC with wide range of slope angles (7–30%), collateral damage emissions were higher on steep areas (Fig. 4), as expected since slopes may affect many physical processes due to gravitational acceleration and geometry (Putz et al. 2018). To our surprise, emissions were not higher on the steepest slopes in RoC (20–28%) or in Gabon (7–10%).

4.2. RIL-C impact performance methodology

The substantial carbon benefits from improved tropical forest management demonstrated in this study justify payments for emissions reductions by REDD + and other climate change mitigation programs. One advantage of this "natural climate solutions pathway" (Griscom et al., 2017) is that because no reductions in timber yields are required, it entails no risk of leakage due to displacement of logging. In our RIL-C study, the relationship between carbon emissions per m^3 of wood extracted and harvest intensity followed similar negative curvilinear trends in all three countries, while there was no indication that FSC certification was associated with reduced emissions (Fig. 6; see S2). RIL-C impact performance promotes increased production rates and profits to support economic development through fulfillment of social obligations while achieving low carbon emissions by not felling trees that yield no timber and by increased wood recovery from the trees that do. These RIL-C practices allow higher timber yields with fewer trees felled, less skidding induced mortality, and reduced emissions from forest clearing of for haul roads and log yards (Feldpausch et al., 2005; Pearson et al., 2014).

In Gabon, more than half non-certified concessions showed high total carbon emissions (Mg C m^{-3}) despite low harvesting intensities, while in RoC and DRC, non-certified concessions emitted relatively little. While emissions (Mg C m^{-3}) varied greatly among concessions, there was no difference between those that were FSC certified and those

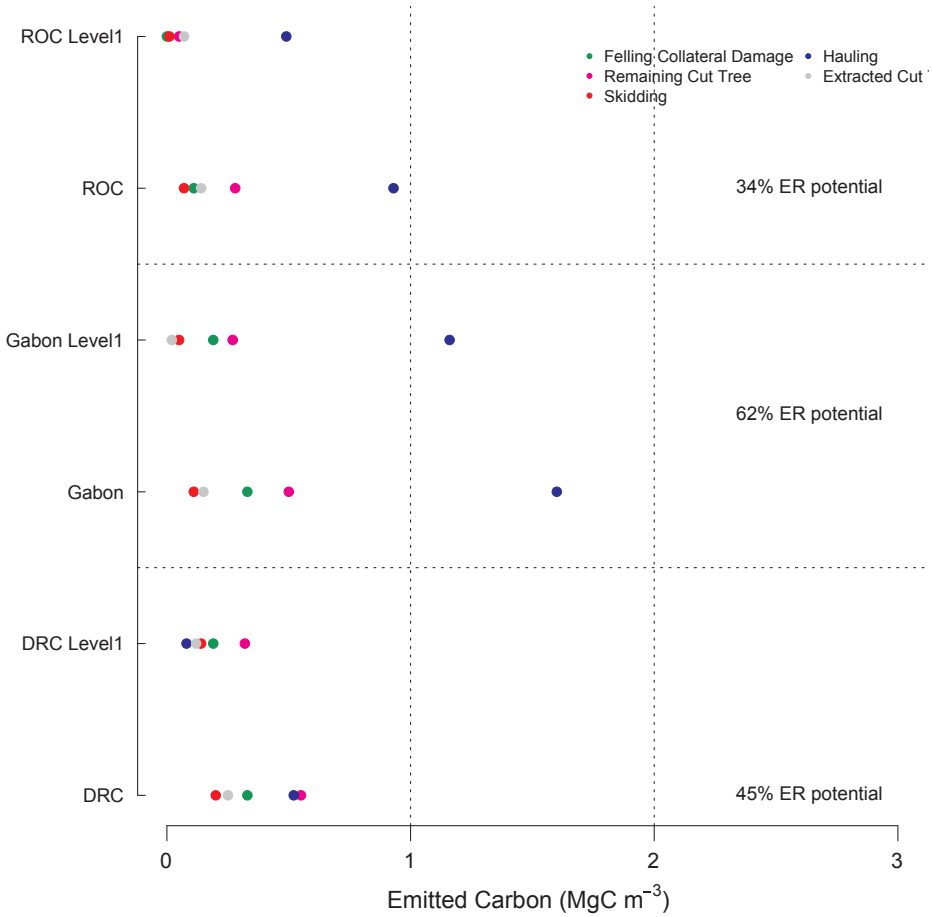

Fig. 5. Committed emissions per cubic meter of timber extracted from existing practices and with implementation of the four RIL-C practices described in Table 6. Mean emissions from existing practices can be reduced by 51% through implementation of RIL-C practices Level 1. ER = emission reductions.

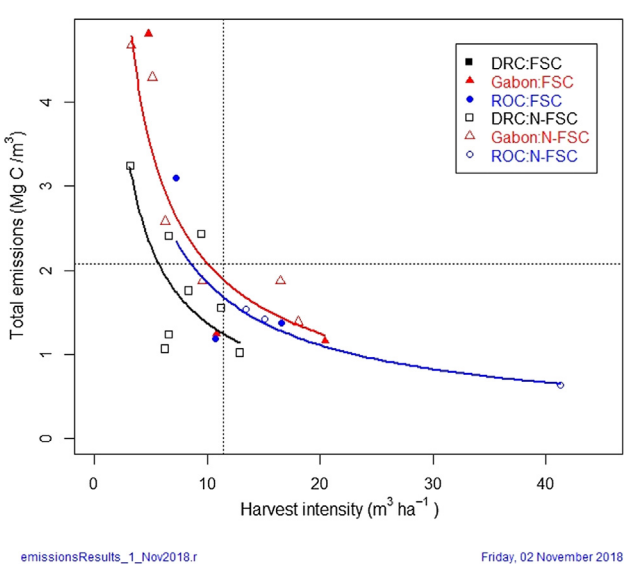

Fig. 6. Carbon emissions (Mg C m^{-3}) versus harvest intensity in DRC, Gabon, and RoC for FSC-certified and non-certified concessions (N-FSC; for model structure and ANOVA see Table S3; for the statistical method see Gregoire (2015).

that were not at least partially due to high variation coupled with small sample sizes and potential positive sampling bias (i.e., only good performing concessions were sampled). This variation can be explained by differences in logging practices driven by management status or other factors that remain to be investigated. Due to the large variation in logging emissions from FSC-certified concessions, we suggest that FSC criteria and indicators (FSC and Council, 2002) be coupled with a monitoring approach such as the RIL-C to clarify what practices deserve attention and to increase assurance of responsible management.

Results in this study also suggest that by reducing road corridor widths and by maximizing timber extraction, substantial emissions reductions are possible. Road corridors are wide to promote surface drying, but better road drainage and surfacing with gravel are suitable substitutes, especially if coupled with better layouts and overall engineering. Thus, opportunities for reduced emissions depend in part on willingness to invest in high quality roads. While logging roads are easily monitored with remote sensing or by FSC auditors in the field, it will be harder to track felling emissions from abandoned logs and poor bucking (Pearson et al. 2014), both of which are tracked with the RIL-C methodology. We recommend that some form of this methodology be incorporated into national standards to quantify logging emissions if national emission reduction targets are to be met in these high forest cover countries.

4.3. Committed emissions from selective logging vs. deforestation

Selective logging of merchantable timber, which takes place over the whole Central African region, emits little carbon per hectare because the harvest intensities are low, but the total emissions are substantial. To estimate these emissions at national scales we used reported industrial roundwood production data from FAO FRA 2015 and applied the relevant emission factors obtained in this study (Table 7). To contextualize these values, we compared the gross emissions with those

Table 7
Gross committed emissions from selective logging of natural forests based on concession averages from each country, emissions from deforestation, and the maximum mitigation potential from implementing RIL-C practices generated from this study.

Country	Roundwood production* (10^3 m^3 yr^{-1})	Total emissions from logging+ (Tg C yr^{-1})	Total emissions from deforestation# (Tg C yr^{-1})	Ratio of logging to deforestation	Maximum mitigation potential with RIL-C (Tg C yr^{-1})
DRC	4,611	5.99	23	0.27	2.70
Gabon	1,987	2.15	3.97	0.54	1.33
RoC	1,779	1.1	3.29	0.33	0.37
CAR	623	0.65	4	0.16	0.33
Cameroon	3,264	3.4	7	0.48	1.73
Equatorial Guinea	750	0.8	1.2	0.6	0.40
Region	2,169	2.34	7.0	0.4	1.14

* Round wood harvest data from FAO's Forest Resources Assessment reported in 2015.
\# Total emissions from deforestation are from Harris et al.2012.
\+ Total emissions excluding logging roads.

from deforestation using data from Harris et al. (2012). Such estimates would be more reliable if based on country-specific emissions factors and of course if based on more accurate estimates of harvested volumes from natural forests (Pearson et al. 2014). For this analysis, we excluded hauling emissions because Harris et al. (2012) may have already included these areas as deforestation (Pearson et al. 2014). We then applied the potential emissions reductions from RIL-C implementation in each country. We applied regional average for countries with no field sampled concessions from our study.

An average of 40% of total emissions from deforestation and forest degradation are from harvesting timber in the six countries Congo Basin countries (Table 7). The DRC, with the highest emissions from deforestation, logging emissions still represent 27% of total land-use change emissions. In countries with low deforestation emissions, such as Equatorial Guinea, RoC, Gabon, and Cameroon, logging emissions contribute higher proportions to the totals, from 33% in RoC to 60% in Equatorial Guinea. To meet their Nationally Determined Commitments (NDCs), these countries might focus on opportunities to reduce logging emissions through implementation of RIL-C practices. We estimate 1.14 (Tg C yr^{-1}), an equivalent of 51% of maximum mitigation potential to reduce emissions from degradation.

4.4. Limitations of the study

The opportunistic nature of sampling concessions may insinuate a bias into our results, especially if the concessions that granted access to us maintained management practices that were above average. We recognize this possibility but lack any means to validate whether or not it affected in our results. We regard the trends revealed by the 23 concessions that were studied are of great value, notwithstanding the possibility of sampling bias. We also note that the bias is likely positive, which means that our estimates of potential emissions reductions from use of RIL-C practices are conservative.

5. Management implications

Future timber yields from selectively logged tropical forests, which are critical for the long-term economic well-being of countries in Central Africa, will vary with harvest intensities and the manners in which timber is harvested. Forest industries contribute up to 7% to the economies of Congo Basin countries, and, in Gabon, they are the second largest employer after the government (de Wasseige et al., 2009). If logging is wasteful, timber stocks will decline rapidly, thereby compromising the ability of the forest to support future extractive economic activities (Umunay et al., 2017). The design of possible actions for reducing logging damage and associated emissions by improving logging practices depend on detailed data about the practices and their consequences. As such, direct measurements, like those employed in this study, allow quantification of damage and associated emissions from each source. Hauling emissions (50%) and emissions from felling damage (43%) are the largest sources of emissions in most of the Congo Basin concessions studied. We suggest that efforts to reduce emissions from these sources include extracting more timber per felled tree and reducing waste. Felling damage would also be reduced by improved directional felling and minimization of incidental damage to surrounding trees through pre-felling liana cutting. Emissions from infrastructure could be reduced by better road planning, shorter roads, and narrower road corridors; efforts should be made to find the optimal ratio between lengths of roads and skid trails. Finally, especially in areas that are steep or where soils are particularly erosion-prone, cable yarding of timber could replace the opening of skid trails up to the stump of each felled tree. We estimated that by applying these RIL-C Level 1 improved practices, emissions could be reduced by 51%. We also believe that the RIL-C monitoring protocol employed in this study represents a robust and cost-effective accounting system that can be used in the design and implementation of performance-based emissions reductions mechanisms. Other improved forest management practices that do not necessarily reduce carbon emissions (e.g., safety gear for forest workers and use of post-logging silvicultural treatments) also deserve attention if countries in the region are to move towards sustainable forest management.

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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.2019.01.049.

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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.)

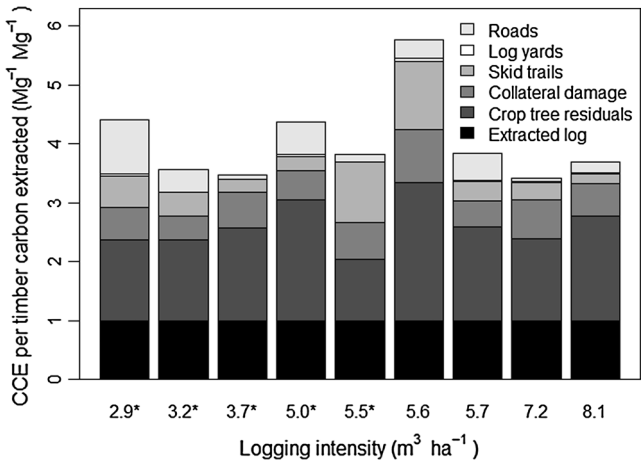

Fig. 1. Total committed carbon emissions (CCE) per carbon in extracted timber (Mg Mg^{-1}) from each source in each forest management unit. By definition, Mg Mg^{-1} of the extracted log = 1.

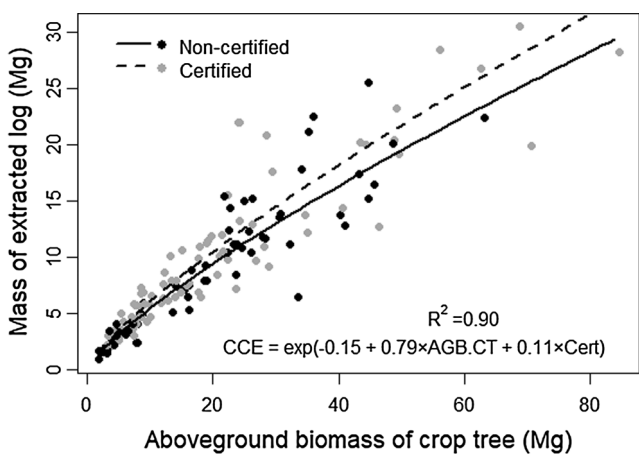

Fig. 2. Dry mass of the extracted log vs. aboveground biomass of the crop tree (AGB.CT) and certification status (Cert; 0 if non-certified and 1 if certified). Certified concessions tended to extract higher proportions of the felled tree mass.

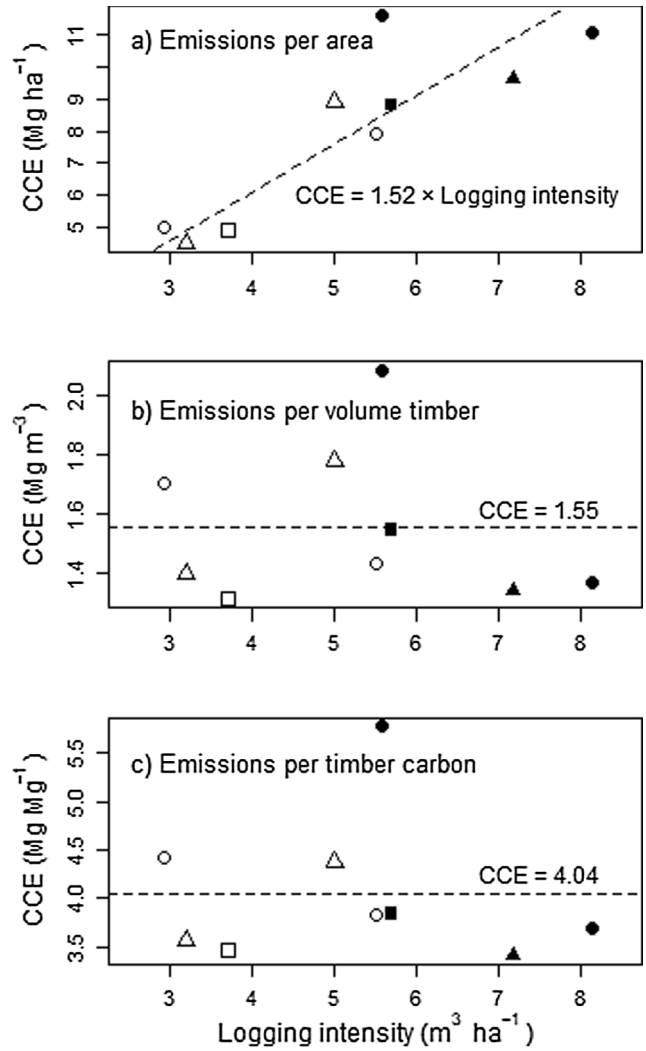

Fig. 3. Baseline committed carbon emissions (CCE) for nine FMUs in Tahuamanu Province, Madre de Dios, Peru reported per ha (a), per volume of timber extracted (b), and per carbon mass extracted in timber (c). Only CCE per ha (a) varies with logging intensity. Hollow symbols are certified forest management 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.,

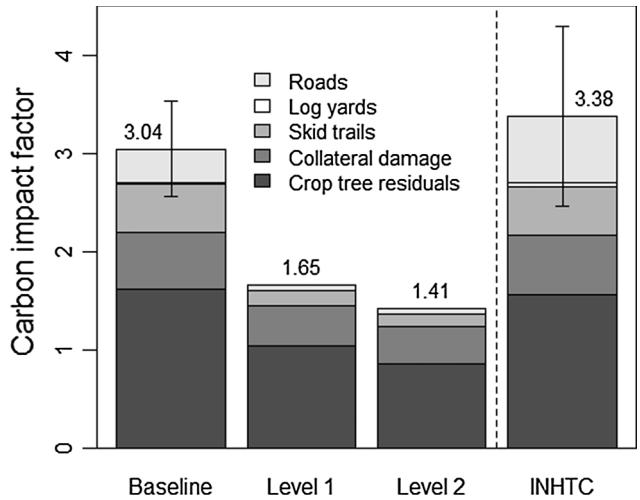

Fig. 4. Four scenarios for carbon impact factor (CIF; Mg C emitted per Mg C in timber extracted; Mg Mg^{-1}) of selective logging in Amazonian Peru: Baseline (mean observed), Level 1 (minimum observed CIF of each source), Level 2 (Level 1 + theoretical reductions), and baseline (mean) if no hollow trees 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.

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Opportunities for carbon emissions reduction from selective logging in Suriname

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1. Introduction

Improved forest management practices in tropical forests managed for timber can make a large contribution to climate change mitigation efforts (Houghton and Nassikas, 2018; Putz and Pinard, 1993). Well-managed tropical forests retain more carbon in vegetation and soils (e.g., Johns et al., 1996; Pinard & Putz, 1996) and more rapidly sequester carbon after logging (Roopsind et al., 2018; Vidal et al., 2016). Under the Paris Climate Accord, tropical countries can apply these reduced carbon emissions from improvements in forest management to meet their nationally determined commitments (NDCs; UNFCCC, 2013). Where selective logging is the principal forestry practice, these emission reductions could come from switching from conventional timber harvests to logging that includes a suite of reduced-impact logging (RIL) practices with trained forestry personnel (Sasaki et al., 2016). Emissions reductions achieved from improved forest management would be eligible for compensation under existing climate financing schemes, such as voluntary carbon markets (VCS, 2016) and the UN-REDD+ program (FCPF, 2018).

To claim emissions reductions payments from improved forest management, accurate and consistent methodologies are required to measure performance relative to an established forest reference emission level (FREL). A FREL sets the baseline against which emissions reduction targets are established and for subsequent monitoring under performance-based carbon payment programs (Angelsen, 2012; UNFCCC, 2013). In this study, we apply an emission assessment protocol (Ellis et al., in press) developed for selective logging to establish Suriname's FREL (Government of Suriname, 2018). This emission assessment protocol was piloted in Kalimantan Indonesia (Griscom et al., 2014) and subsequently approved for use by the Voluntary Carbon Standard (VCS, 2016). We complement the Griscom et al., (2014) protocol with elements from the FREL methodology developed by Winrock International for logging emissions in Guyana, Brazil, Belize, and Gabon (Brown et al., 2014; Pearson et al., 2014).

1.1. Forest management in Suriname

Suriname is a high forest cover low deforestation country, with the highest forest cover in the world at 93% (15.2 million hectares) and an annual deforestation rate assessed between 2000 and 2012 at 0.04% (5676 ha yr^{-1}; SBB, 2017). Logging in Surname is a major economic activity and is characterized by the selective removal of high-value tree species. Logging results primarily in forest degradation and is the second largest source of carbon emissions after deforestation from gold mining in Suriname (Government of Suriname, 2018). In order to improve forest management and reduce forestry related emissions, Suriname has proposed RIL guidelines embedded in a draft national logging code of practice to be applied across all forest management enterprises (van der Hout, 2011).

These forest management enterprises, whether community forests or industrial concessions leased from the government, are divided into forest harvesting units (FHUs, or "kapvaks" in Dutch) which are approximately 100 ha from which timber can be harvested for 2 years. Average timber production is estimated at 8.80 m^3 ha^{-1} with a maximum allowable harvest of 25 m^3 ha^{-1} at logging rotations of 25 years (SBB, 2017; Werger, 2011). Harvesting practices in FHUs are categorized into three management systems: (1) conventional logging, where there is no forest management planning, including no pre-harvest forest inventory; (2) controlled logging, where forest management plans are prepared prior to timber harvests; and (3) controlled logging certified by the Forest Stewardship Council (FSC), hereafter referred to as FSC-logging that includes application of a broader suite of sustainable forest management practices (Table 1). Rules regarding protected species and minimum felling diameter are applicable to all three logging types, but other recommended RIL practices such as directional felling and winching are not mandated across all FHUs.

Conventional logging is usually permitted in areas where there is a possibility that overlapping land-use claims (e.g., sub-surface alluvial gold mining) could preclude sustainable forest management or, in the case of very small-scale community operations, where license holders lack the capacity and/or capital for intensive pre-harvest planning.

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Table 1
Forest management systems applied in Suriname (conventional logging, controlled logging, and FSC-logging) and their associated forest management guidelines and logging practices (NIMOS et al., 2017).

Management Type	Pre-harvest timber inventory	Skid trail and road planning	Forest management plans/annual harvest plans	Maximum allowable harvest ($25\,m^3$/ha)	Minimum harvest cycle (25 years)	Third-party audits	Additional RIL practices (directional felling, winching, liana cutting)
Conventional logging (C)	No	No	No	√	√	No	No
Controlled logging (R)	√	√	√	√	√	No	Not strictly applied
FSC logging (FSC)	√	√	√	√	√	√	√

Controlled logging is done according to the national legal RIL requirements (e.g., pre-harvest inventory and preparation of harvest plans; (van der Hout, 2011) and the FSC certified concessions are required to apply a higher level of RIL practices that include trained forestry personnel. These three logging systems are coded as C, R and FSC for conventional, controlled and FSC-logging respectively (Table 1). In addition to variation among management systems applied at the FHU level, there is also variation in logging machinery (e.g., skidders, bulldozers, excavators) utilized, as well as in the technical skills of forestry workers.

In 2016 there was approximately 2 million ha of forests issued to forest management enterprises with annual active production areas of 32,328 ha (283 FHUs) classified under conventional logging and 18,134 ha (185 FHUs) under controlled timber harvests, and an estimated 6200 ha (62 FHUS) under FSC-logging (SBB, 2017). This study is an input to Suriname's determination of its FREL for development of its National REDD + program. We hypothesized that logging systems that apply more RIL practices would have lower emissions per cubic meter of wood harvested, with emissions highest in FHUs under conventional logging, intermediate in FHUs under controlled logging, and lowest in the FSC-logged FHUs.

2. Methods

2.1. Study sites and sampling design

The study was conducted in the 50–200 km-wide lowland tropical forest belt of Suriname that stretches East-West just above the 4° N parallel (Fig. 1). In 2017 we assessed carbon emissions in 10 logged FHUs across the different forest management systems; conventional logging (N = 4), controlled logging (N = 4), and FSC-logging (N = 2). As there were only 2 FSC certified forest management enterprises at the time of the field surveys, we decided to sample both and spread the other 8 sampling locations over the other 2 management types in several different forest management enterprises. We randomly selected FHUs that met the following criteria: (1) the FHU was logged less than 6 months prior to emissions assessment (to ensure logging impacts were still visible); (2) all harvest operations were completed and legal access would not occur until the next harvest; and, (3) there was no evidence of other land uses, especially gold mining. The exclusion of mined areas prevented emissions inflation from disturbance not due to logging.

We sampled a randomly selected half (50 ha) of each sampled FHU, except the first FHU, which we sampled in its entirety (100 ha) during the development and testing of the sampling protocol. We categorized carbon emissions from logging into the following sources: (1) extracted log emissions (ELE) - carbon removed from the forest in the extracted section of the felled tree; (2) logging damage factor (LDF) - carbon from the unextracted sections of the felled trees (i.e., branches, roots) and trees damaged or killed during felling (i.e., collateral felling damage); and, (3) Logging infrastructure factor (LIF) – carbon lost from skid trails, log decks (i.e., areas where logs are temporarily stored before being trucked from the forest) and haul roads. The total emissions factor (TEF; $Mg\,C\,m^{-3}$) for each FHU is the combined emissions from ELE, LDF, and LIF.

2.2. Carbon accounting method

2.2.1. Unlogged forest biomass

Each logged FHU sampled for carbon emissions was paired with an adjacent unlogged FHU of similar forest type and terrain that was proposed to be logged by the forest management enterprise. We used a variable plot sampling method with a 40 BAF prism ($40\,ft^2\,acre^{-1}$; $9.18\,m^2\,ha^{-1}$), with a total of 15 unlogged biomass plots that were paired with the logged FHUs that was sampled for carbon emissions. The unlogged biomass plots were established along transects at 100 m intervals. Once a tree was determined to be 'in' the biomass plot with

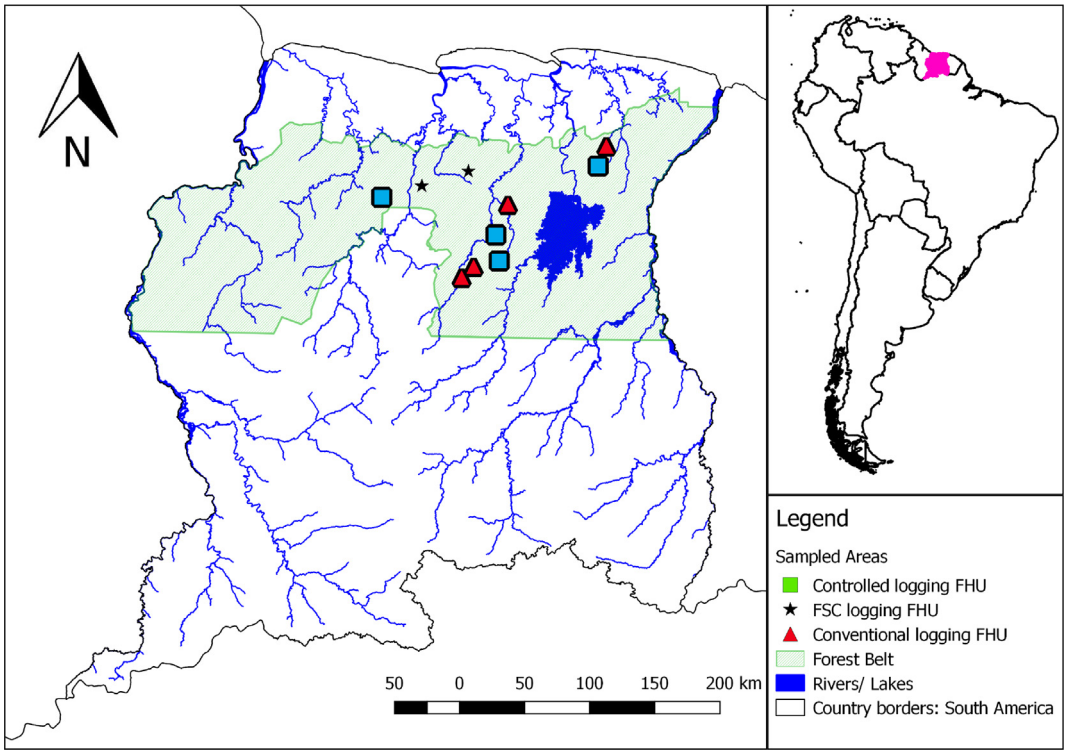

Fig. 1. Locations of forest harvest units (FHUs) sampled for carbon emissions by forest management classification (controlled logging, conventional logging, and FSC-logging) in Suriname. Inset map on the upper right is of South America with Suriname highlighted in pink.

the 40 BAF prism, its diameter at 1.3 m aboveground (DBH) was measured and its species recorded. We excluded areas such as creeks and swamps, as the primary goal of the unlogged biomass plots was to infer carbon lost from logging roads and log decks, which are generally constructed on ridges and in other areas with well-drained soils in the FHUs. We estimated the aboveground biomass of each tree with the 'model II moist forest stands' allometry proposed by Chave et al., (2005), and belowground biomass utilizing Eq. (1) from Mokany et al. (2006). We converted total tree biomass to carbon using the conversion factor of 0.47 (IPCC, 2006). We calculated carbon stocks per unit area as the basal area per unit area $*$ average BBAR; where BBAR is equivalent to $\frac{\text{tree carbon}}{\text{tree basal area}}$, also referred to as a mean-of-ratios-estimator (Bitterlich, 1984; Marshall et al., 2004).

2.2.2. Extracted timber volumes

Within the 50 ha sample area of each logged FHUs, all felled tree stumps were counted, and locations recorded with a Garmin GPS 62 s handheld unit. In a minimum of 25 felling gaps in each logged FHU, we recorded the height of stumps, length, and diameter of all log sections, and tree height. The status of each log section, whether present or absent (i.e., removed from the felling gap), was recorded. The length of the extracted section (if any) was determined as the distance from the stump or butt log to the top-cut below the tree crown or upper log section present. We assumed the absent log section is equivalent to the extracted timber and calculated its volume with the Smalian scaling formula (cm^3). The total extracted timber volume at the FHU level was then estimated based on the total number of tree stumps counted and the average volume extracted per stump based on the felled trees measured. As our volume estimate of the harvested tree was done at the stump, it does not account for subsequent bucking and rejection of logs or log sections within the forest that would reduce the overall timber volume recovered.

2.2.3. Extracted log emissions (ELE)

To simplify the carbon accounting process and to follow IPCC Tier 1 methods (IPCC, 2007) and the literature (Griscom et al., 2014; Pearson et al., 2014), we assumed that all carbon in the harvested portion of the tree (i.e., extracted timber) is emitted at the time of felling due to lack of data on wood processing recovery and decay rates of associated wood products. We converted the extracted timber volumes (cm^3) to extracted carbon as: ELE (Mg C m^{-3}) = vol (cm^3) $*WD_s * 10^{-6} * 0.47$, where WD_s is species-specific wood density in g cm^{-3} (Chave et al., 2009; Zanne et al., 2009), 0.47 is the biomass-to-carbon ratio and 10^{-6} is the conversion factor to Mg. We also accounted for the missing mass in hollow portions of log sections in our biomass estimates of the felled trees based on the diameter and length of the decayed sections. If the hollow was only detected at one end of the log section, we assumed the hollow was half the length of the log, applying the bottom diameter of the hollow as the top diameter. The missing biomass estimated based on the dimensions of the hollow was then subtracted from the total tree biomass to account.

2.2.4. Logging damage factor (LDF)

The logging damage factor reflects the pool of dead carbon created in felling gaps where harvest volumes and ELE was measured. The LDF includes branches and roots of the harvested tree (unextracted biomass) and trees killed or severely damaged during harvesting (felling damage).

$$\text{LDF (Mg C m}^{-3}) = \frac{\sum_{\text{gap}_i}^{n}\left(\frac{([(\text{biomass of tree}_i - \text{ELE}_i) + \text{CD}_{\text{gap}_i}])}{\text{Gap}_i \text{ volume (m}^3)}\right)}{\text{Number of gaps}}$$

Here, biomass of $tree_i$ refers to the total biomass of each felled tree in felling gap_i, ELE_i is the extracted biomass from $tree_i$ and is subtracted from the total tree biomass to estimate the unextracted biomass for $tree_i$, $CDgap_i$ is the carbon emissions from trees killed during felling in a specific gap and Gap_i volume (m^3) is the timber extracted from the felling gap, as a single felling gap may contain several harvested trees. The diameter used to estimate the biomass of $tree_i$ applying the Chave et al. (2005) allometry when the log was extracted and there was no

rejected log section, was the stump diameter as stem taper is minimal if there are no stem deformities. In instances where there was a rejected lower log section, due to buttresses or swollen stem, the top diameter of the rejected log section above any deformities was used to estimate the tree biomass. Only trees classified as snapped (stem broken at > 1.3 m) or grounded (stem broken at < 1.3 m or uprooted) were included in the LDF as felling damage, while trees with other damage types (e.g., bark damage and partial crown loss) were assumed to survive post-logging, as we lack data on the post-logging morality rates of the different damage classes (Fig. S1). Trees felled and not extracted were included as part of the LDF.

2.2.5. Skid trail emissions factor (SF)

All skid trails in the sampled FHUs were mapped with a Garmin 62 s GPS handheld unit to estimate the area covered by skid trails. At 200 m intervals along the skid trails in each sampled FHU, 10 m-long plots were established (N = 15 per FHU) to assess damage and death of trees ≥10 cm and to measure skid trail widths. The mapped length and average width of the skid trails were used to estimate the skid trail area (ha) in a FHU. Carbon emissions for the area occupied by skid trails was estimated based on the tree mortality recorded in the skid plots (Mg C ha^{-1} of skid trail) and the total area occupied by skid trails in a FHU. As the skid trail emissions was scaled up to the FHU level, the timber production estimated at the FHU could be used as the denominator to estimate the carbon emission factor (Mg C m^{-3}).

$$\mathrm{SF}\ (\mathrm{Mg\ C\ m^{-3}}) = \left(\frac{\mathrm{Mg\ C\ ha^{-1}\ of\ skid\ trail} * \mathrm{skid\ trail\ area\ (ha)\ in\ FHU}}{\mathrm{timber\ production\ (m^3)\ in\ FHU}}\right)$$

2.2.6. Haul road emissions factor (HF)

To estimate the carbon emissions from haul roads we utilized the baseline carbon stocks measured in unlogged biomass plots (Mg C ha^{-1}). We estimated the area deforested by haul roads in each FHU based on their respective length and width. Haul road widths were defined as the perpendicular distance between undamaged trees ≥10 cm DBH on either side of haul roads at the forest edge, with 10 measurements recorded for each FHU. We determined road intensity (length haul road per m^3 extracted) as follows: (1) We combined GPS mapped haul roads in and near our focal FHU with remotely detected haul roads from Sentinel 2A satellite imagery (Copernicus, 2017) and Suriname forestry agency's national GIS database (SBB, 2016) on logging road infrastructure. (2) We treated haul roads similar to a catchment area of a river drainage system, as many haul roads are used to extract logs from several FHUs. We used our estimates of extracted harvest intensities (m^3 ha^{-1}) from the sampled FHUs to scale up our timber production estimates across the entire catchment area served by the haul road in our focal FHU. (3) With the estimated timber production from the multiple FHUs served by the haul roads and the calculated area occupied by the haul roads, we estimate the carbon emitted from haul roads per cubic meter of wood harvested.

2.2.7. Log deck emissions factor (DF)

Similar to haul roads, we treated log decks as completely deforested areas. We measured the lengths and widths of 10 log decks in and around each sampled FHU and estimated their areas based on their respective shapes. We then counted the number of log decks within each sampled FHU in the field and calculated the total area occupied by log decks based on the average log deck size. Carbon emissions from log deck construction were then estimated based on the area deforested using the baseline carbon stocks for the FHU. We combined emissions from skid trails, haul roads, and log decks to estimate the logging infrastructure factor (LIF; Mg C m^{-3}). We acknowledge the reuse of logging infrastructure such as haul roads and log landings in subsequent harvest rotations would reduce the overall emissions estimated relative to the timber extracted. We, however, lack data on the re-use of logging infrastructure as the majority of logging concessions in Suriname have not begun a second rotation.

2.3. Statistical analysis

We fitted linear mixed effect models for carbon emissions from collateral felling damage estimated at the gap level (N = 239) and emissions from trees killed during the construction and use of skid trails from our skid trail plots (N = 152). We included the timber volume extracted (m^3) at the gap level (harvest intensity), management type, and slope (%) as predictor variables in the model on emissions from collateral felling damage. We did not have information on the intensity of use of skid trails based on the number of passes a machine would have made across the skid plots, and thus limits our ability to infer how the intensity of use influences skid trail emissions. We fitted the emissions model for collateral felling damage and skid trail emissions with FHUs as random effects to account for baseline differences among FHUs. For exploratory purposes, we fit a regression model with logging intensity (log transformed) as a predictor of the total emissions factor (TEF; Mg C m^{-3}) at the FHU level. We are cautious about our statistical inferences at the FHUs with respect to the relationship between TEF and logging intensity due to our small sample size of N = 10 FHUs (4 FHUs in conventional logging, 4 FHUs in controlled logging, and 2 FHUs in FSC-logging). The small sample size limits our confidence in making strong statistical inference, especially quantifying uncertainty and including additional predictor variables such as the logging system, terrain (slope) or baseline biophysical characteristics of the FHUs by means of random effects as we did for the models on collateral felling damage emissions and skid plot emissions (Greenland et al., 2000; Ogundimu et al., 2016). We built and implemented our models with the rstanarm package (Stan Development Team, 2017) that employs a Bayesian estimation routine for regression models in R (R Core Team, 2014). We incrementally added predictor variables, including interactions and the random effects, comparing each model with leave-one-out-cross-validation (LOO) and model averaging using the loo package to ensure model complexity did not reduce model performance, and report the best fit model in our results (Vehtari et al., 2018).

3. Results

Average harvest intensity was 11.73 m^3 ha^{-1} (SE: ± 1.56), which resulted in carbon emissions of 2.44 Mg (SE: ± 0.36) for every cubic meter of timber extracted (Table 2). Unextracted biomass of harvested trees (0.70 Mg C m^{-3}; SE: ± 0.08; 29%) and collateral felling damage (0.57 Mg C m^{-3}; SE: ± 0.07; 23%) were the main sources of carbon emissions (Table 2 & Fig. 2).

Logging infrastructure associated with haul roads, skid trails, and log decks accounted for 21% (0.51 Mg C m^{-3}; SE: ± 0.16), 13% (0.31 Mg C m^{-3}; SE: ± 0.08) and 2% (0.06 Mg C m^{-3}; SE: ± 0.01) of logging related emissions, respectively (Table 2 & Fig. 2).

Carbon emissions were highest under conventional logging (3.23 Mg C m^{-3}; SE: ± 0.74), followed by controlled logging (1.96 Mg C m^{-3}; SE: ± 0.25), and FSC logging (1.82 Mg C m^{-3}; SE: ± 0.09; Table 2 & Fig. 3). Extracted timber volumes were lowest in conventionally logged FHUs (8.38 m^3 ha^{-1}; SE: ± 1.87; 1.5 trees ha^{-1}) compared to controlled logging (13.41 m^3 ha^{-1}; SE: ± 7.44; 2.9 trees ha^{-1}) and FSC logging (15.06 m^3 ha^{-1}; SE: ± 3.80; 2.5 trees ha^{-1}). Logging intensity (m^3 ha^{-1}; log transformed) explained 60% (R^2 95% CI: 0.3–0.8) of the variation in the total emissions factor (TEF – Mg C m^{-3}) across the 10 FHUs (Fig. 4).

3.1. Logging infrastructure

Skid trails averaged 5.73 m (SE: ± 0.30) wide and covered 703.06 (SE: ± 47.15) m^2 ha^{-1} of logged forest across all FHUs. There was

Table 2
Timber volume extracted ($m^3 ha^{-1}$) and logging related carbon emissions ($Mg C m^{-3}$) by extracted and unextracted felled tree biomass, collateral felling damage, skid trails, haul roads and log decks across the 10 sampled forest harvest units (FHUs) classified by logging system (C = conventional logging; R = controlled logging; FSC = Forest Stewardship Council certified logging).

	Harvested Volume ($m^3 ha^{-1}$)	Extracted Wood Emissions ($Mg C m^{-3}$)	Unextracted Wood Emissions ($Mg C m^{-3}$)	Collateral Damage Emissions ($Mg C m^{-3}$)	Skid Trail Emissions ($Mg C m^{-3}$)	Haul Road Emissions ($Mg C m^{-3}$)	Log Decks Emissions ($Mg C m^{-3}$)	Total Emission Factor (TEF) ($Mg C m^{-3}$)
C1	11.74	0.29	0.43	0.32	0.14	0.36	0.04	1.57
C2	10.91	0.29	0.57	0.81	0.37	0.62	0.01	2.67
C3	7.33	0.33	0.80	0.90	0.78	0.80	0.03	3.65
C4	3.55	0.34	1.28	0.78	0.73	1.81	0.10	5.05
R1	16.20	0.26	0.50	0.24	0.11	0.12	0.05	1.29
R2	16.70	0.31	0.65	0.66	0.10	0.19	0.10	2.02
R3	8.60	0.31	0.83	0.40	0.46	0.35	0.10	2.46
R4	12.13	0.30	0.55	0.75	0.13	0.31	0.05	2.09
FSC1	9.80	0.30	0.75	0.44	0.09	0.25	0.08	1.90
FSC2	20.32	0.29	0.62	0.37	0.16	0.25	0.04	1.73
Mean (SE)	**11.73 (1.56)**	**0.30 (0.01)**	**0.70 (0.08)**	**0.57 (0.07)**	**0.31 (0.08)**	**0.51 (0.16)**	**0.06 (0.01)**	**2.44 (0.36)**
C - Mean (SE)	8.38 (1.87)	0.31 (0.01)	0.77 (0.19)	0.70 (0.13)	0.50 (0.15)	0.90 (0.32)	0.05 (0.02)	3.23 (0.74)
R - Mean (SE)	13.41 (7.44)	0.30 (0.01)	0.63 (0.07)	0.52 (0.12)	0.20 (0.09)	0.24 (0.05)	0.08 (0.02)	1.96 (0.25)
FSC - Mean (SE)	15.06 (3.80)	0.30 (0.01)	0.69 (0.06)	0.41 (0.03)	0.12 (0.03)	0.25 (0.00)	0.06 (0.02)	1.82 (0.09)

71.60 m^2 (SE: ± 11.84) of skid trail for every cubic meter of wood harvested (301 m^2 per tree felled; SE: ± 47.15), and 122 m of skid trail per hectare of forest across all the sampled FHUs (SE: ± 6.99; Fig. S2).

FSC-logged FHUs had the highest skid trail densities (mean = 759.92 $m^2 ha^{-1}$; SE: ± 226.56), with skid trail density per cubic meter of extracted timber lowest in FSC-logged FHUs (51.50 $m^2 m^{-3}$; SE: ± 2.95; Table 3).

Skid trail emissions per cubic meter of wood extracted in FSC-logged FHUs was 0.12 $Mg C m^{-3}$ (SE: ± 0.03), which was 40% and 76% lower compared to controlled (0.20 $Mg C m^{-3}$; SE: ± 0.09) and conventionally logged (0.50 $Mg C m^{-3}$; SE: ± 0.015) FHUs, respectively (Table 2). Excavators were found to be used for skidding in 70% of forest management enterprises, sometimes in combination with wheel skidders.

Carbon emissions from the skid trail plots (N = 156) indicated a reduction of 0.16 Mg C (95% CI, −0.34 to 0.004) and 0.19 Mg C (95% CI, −0.42 to 0.04) for every 10 m of skid trail constructed under controlled logging and FSC logging, respectively compared to conventionally logged FHUs (0.30 Mg C, 95% CI 0.15–0.46). Slope had a

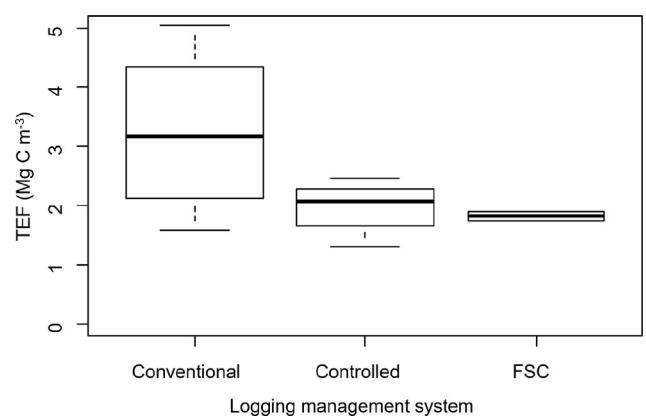

Fig. 3. Total logging-related carbon emissions from FHUs under conventional logging (N = 4), controlled logging (N = 4), and FSC logging (N = 2) expressed per cubic meter of wood removed from the felling gap.

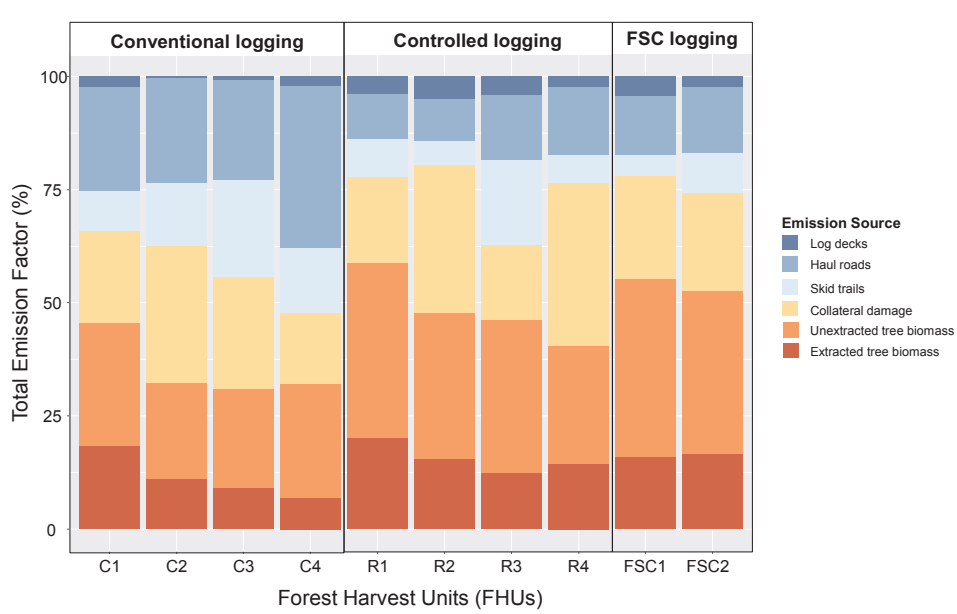

Fig. 2. Total emission factor (TEF) broken down by emission source (extracted tree biomass, unextracted felled tree biomass, collateral felling damage, skid trails, haul roads, and log decks) based on forest management type (conventional logging - C, controlled logging - R and FSC logging - FSC).

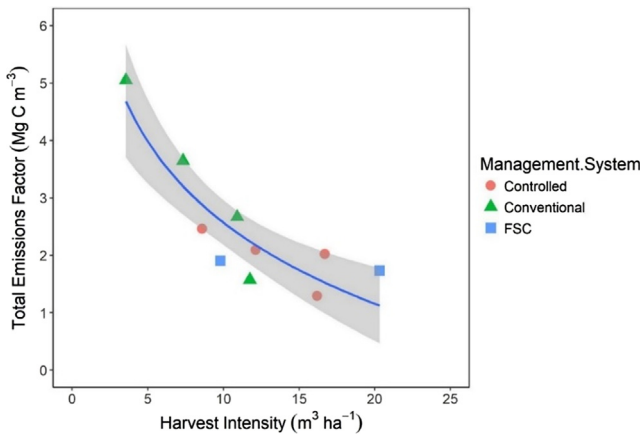

Fig. 4. Relationship between harvest intensity (m^3 ha^{-1}) and total carbon emissions. We fit a log curve (blue line) with the 95% CI captured by the grey band. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

small positive effect on overall skid trail emissions but was not significant (0.01 Mg C, 95% CI −0.01 to 0.02; Fig. 5).

Haul road emissions varied by a factor of 15 among FHUs, with haul road emissions from conventional logging 70% higher (0.90 Mg C m^{-3} SE: ± 0.32) than controlled logging (0.24 Mg C m^{-3}; SE: ± 0.05) and FSC logging (0.25 Mg C m^{-3}; SE: ± 0.00; Table 2). Haul road widths were also 40% wider in conventional logging (24.66 m; SE: ± 2.50) with density of haul roads per cubic meter of wood extracted three times (31.93 m^2 m^{-3}; SE: ± 8.18) that of controlled logging (9.17 m^2 m^{-3}; SE: ± 2.14) and FSC logging (10.89 m^2 m^{-3}; SE: ± 0.17; Table 3).

Log decks accounted for the smallest source of logging emissions, with

0.08 Mg C m^{-3} (SE: ± 0.02) in controlled logging, 0.05 Mg C m^{-3} (SE: ± 0.02) in conventional logging and 0.06 Mg C m^{-3} (SE: ± 0.02) in FSC logging (Table 2). The average size of log decks ranged from 0.07 ha (SE: ± 0.01) in conventional logging to 0.11 ha (SE: ± 0.01) in controlled logging (Table 3).

3.2. Felled tree emissions

Extracted C emissions were similar among the three logging systems (0.30 Mg C m^{-3}) with C emissions associated with the unextracted sections of felled trees highest in conventional logging (0.77 Mg C m^{-3}; SE: ± 0.19), followed by FSC logging (0.69 Mg C m^{-3}; SE: ± 0.06) and controlled logging (0.63 Mg C m^{-3}; SE: ± 0.07; Table 2 & Fig. S3). Felled trees that had no timber extracted constituted 10.26% (SE: ± 5.84) and 8.16% (SE: ± 9.45) of all stumps recorded in conventional logging and controlled logging respectively, and 3.64% (SE: ± 0.09) in FSC-logged FHUs (Table 4).

In our sample of 255 harvested trees in 239 felling gaps, 1277 other trees ≥10 cm DBH were uprooted or snapped (Figs. S1 and S4). An additional 470 trees lost > 50% of their crown, and 115 were leaning > 10 degrees (assumed to be from logging impact). The number of trees killed per felled tree was highest in conventional logging (6 trees killed per tree felled; Table 4).

Collateral felling damage emissions measured at the gap level, averaged 0.57 Mg C m^{-3} (SE: ± 0.07) across all FHUs, and was higher in conventional logging (0.70 Mg C m^{-3}; SE: ± 0.13) compared to controlled logging (0.52 Mg C m^{-3}; SE: ± 0.12) and FSC logging (0.41 Mg C m^{-3}; SE: ± 0.03; Table 2). Logging intensity had a significant effect on collateral felling damage emissions that resulted in an increase of 0.12 Mg C (95%; CI: 0.03–0.22) for every additional cubic meter of timber harvested (Figs. S5 and S6). Application of controlled logging and FSC logging reduced collateral felling damage emissions by 1.0 Mg C (95% CI, −3.15 to 1.15) and 1.3 Mg C (95% CI, −3.84 to 1.22), respectively, relative to conventional logging (3.3 Mg C, 95% CI, 1.6–5.0). Slope did not have a significant effect on felling damage emissions (−0.04, 95% CI, −0.14 to 0.06; Fig. 6).

4. Discussion

Overall carbon emissions from the forestry sector in Suriname (2.44 Mg C m^{-3}) were similar to those reported in Guyana (2.33 Mg C m^{-3}), a neighboring country that shares biophysical characteristics associated with Guiana Shield forests (Hammond, 2005; Pearson et al., 2014). Logging emissions measured in this study and the Pearson et al. (2014) study for Guyana, were dominated by the unextracted portions of the harvested trees and trees killed during felling (collateral felling damage), with logging intensity driving overall emissions compared to other countries (Ellis et al., in press). The high logging emissions related to the extracted and unextracted portions of the felled tree in Suriname and Guyana, is in part driven by the high wood densities across Guiana Shield forests (mean wood

Table 3
Logging machinery and infrastructure characteristics for log decks, skid trails, and haul roads in sampled FHUs by logging system (C = conventional logging; R = controlled logging; FSC = Forest Stewardship Council certified logging).

Forest Harvest Unit (FHU)	Machinery used	Average log deck area (ha)	Average skid trail width (m)	Skid trail density (m^2 ha^{-1})	Skid trail area per timber volume (m^2 m^{-3})	Skid trail length per tree harvested (m)	Average haul road width (m)	Haul road density (m^2 ha^{-1})	Haul road area per timber volume (m^2 m^{-3})
C1	Bulldozer	0.11	5.76	674.82	57.50	70	25.45	8.23	17.85
C2	Excavator	0.05	5.63	553.26	50.72	60	31.43	9.40	27.07
C3	Excavator	0.08	6.37	870.74	118.86	78	21.22	9.40	27.22
C4	Excavator	0.06	6.03	539.11	151.78	101	20.54	9.61	55.59
R1	Excavator & wheel skidder	0.09	5.02	690.53	42.62	38	14.73	5.96	5.42
R2	Excavator & wheel skidder	0.13	4.29	697.18	41.74	47	11.51	8.32	5.73
R3	Excavator	0.10	7.06	811.11	94.26	54	15.86	6.25	11.53
R4	Excavator	0.14	4.93	673.97	55.55	59	19.12	8.89	14.01
FSC1	Wheel skidder	0.07	5.02	533.37	54.45	53	14.35	7.55	11.05
FSC2	Wheel skidder	0.09	7.15	986.48	48.55	45	13.97	15.59	10.72
Mean (SE)		**0.09 (0.01)**	**5.73 (0.30)**	**703.06 (47.15)**	**71.60 (11.84)**	**60.54 (5.85)**	**18.82 (1.93)**	**8.92 (0.84)**	**18.62 (4.77)**
Mean C (SE)		0.07 (0.01)	5.95 (0.16)	659.48 (76.72)	94.72 (24.43)	77.21 (8.86)	24.66 (2.50)	9.16 (0.31)	31.93 (8.18)
Mean R (SE)		0.11 (0.01)	5.32 (0.60)	718.20 (31.35)	58.54 (12.32)	49.58 (4.56)	15.31 (1.57)	7.36 (0.73)	9.17 (2.14)
Mean FSC (SE)		0.08 (0.01)	6.08 (1.06)	759.92 (226.56)	51.50 (2.95)	49.11 (4.30)	14.16 (0.19)	11.57 (4.02)	10.89 (0.17)

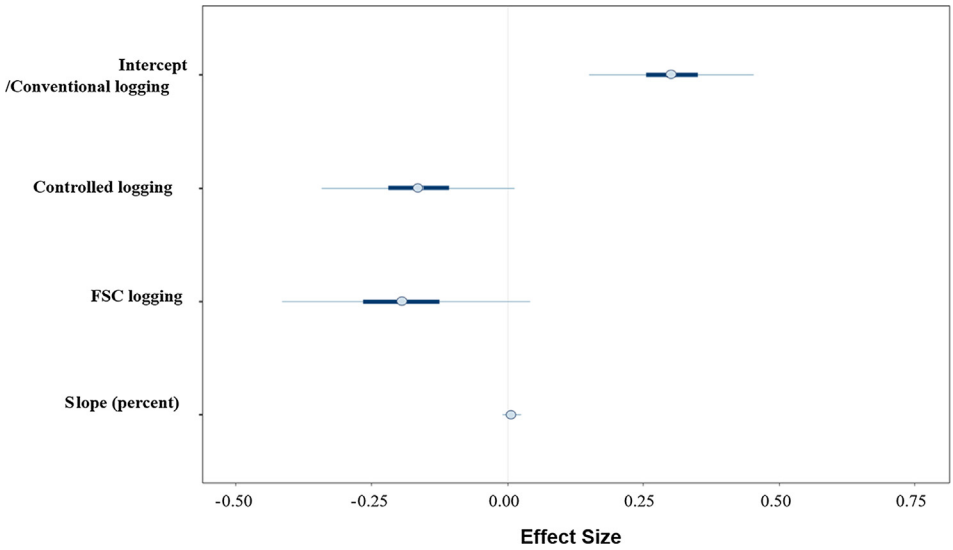

Fig. 5. Model coefficients for carbon emissions from 10 m long skid trail plots (N = 156). The intercept term on the y-axis is equivalent to C emissions from conventional logging. The coefficient parameters for controlled logging, FSC logging, and slope represent the effect of these parameters relative to conventional logging (Intercept). Circles represent mean effect size, thick horizontal lines are the 50% credible intervals, and thin horizontal lines are the 95% credible intervals (R^2 = 0.11).

Table 4
Average diameters and lengths of logs extracted from sampled FHUs (numbers of trees noted parenthetically) and percentage of felled trees that were cut from which no timber was extracted by logging system (C = conventional logging; R = controlled logging; FSC = Forest Stewardship Council certified logging).

Forest Harvest Unit (FHU)	Average extracted log length (m)	Average diameter of bottom section of extracted log (cm)	# Felling gaps sampled	Carbon stock in adjacent unlogged FHU (Mg C ha^{-1})	Felled trees not extracted (%)	# Trees felled per hectare	# trees killed per tree felled	Average mortally damaged collateral tree DBH
C1	18.00	73.37	26.00	200.25	11.54	1.31	4.65	24.26
C2	16.62	74.98	25.00	227.70	8.00	2.17	7.24	26.75
C3	13.72	64.68	26.00	295.61	3.85	3.19	5.08	26.08
C4	13.69	72.50	17.00	325.68	17.65	2.29	6.41	23.77
R1	15.85	65.50	26.00	230.52	0.00	2.29	3.62	19.96
R2	16.64	68.66	29.00	323.29	17.24	3.48	4.79	25.49
R3	13.41	72.07	26.00	305.38	15.38	2.55	4.58	22.31
R4	15.19	66.06	25.00	222.08	0.00	2.24	5.52	26.02
FSC1	14.21	71.57	28.00	224.04	3.57	1.96	4.36	23.66
FSC2	15.90	80.96	27.00	231.83	3.70	2.63	4.52	25.70
Mean (SE)	**15.32 (1.54)**	**71.04 (1.58)**	**25.50 (3.24)**	258.64 (15.14)	**8.09 (6.90)**	**2.41 (0.19)**	**5.08 (0.34)**	**24.40 (0.66)**
Mean C (SE)	15.51 (2.16)	71.38 (2.29)	23.50 (4.26)	262.31 (29.12)	10.26 (5.84)	2.24 (0.38)	5.85 (0.60)	25.21 (0.71)
Mean R (SE)	15.27 (1.38)	68.07 (1.50)	26.50 (1.73)	270.32 (25.73)	8.16 (9.45)	2.64 (0.29)	4.63 (0.39)	23.44 (1.42)
Mean FSC (SE)	15.06 (1.19)	76.27 (4.69)	27.50 (0.71)	227.93 (3.90)	3.64 (0.09)	2.29 (0.33)	4.44 (0.08)	24.68 (1.02)

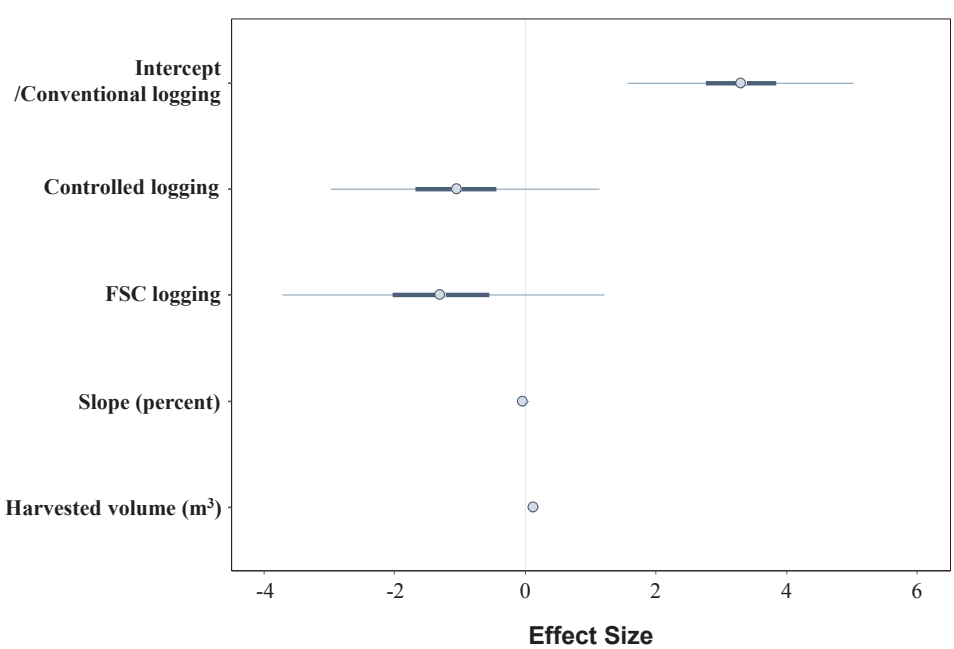

Fig. 6. Model coefficients for collateral felling emissions assessed at the gap-level (N = 239). The coefficient parameters for controlled logging, FSC logging, slope, and harvested volume assessed at the gap represent the effect of these parameters relative to conventional logging (intercept). Circles represent mean effect size, thick horizontal lines are the 50% credible intervals, and thin horizontal lines are the 95% credible intervals (R^2 = 0.14).

density = 0.63 g cm^{-3}; Johnson et al., 2016). For example, concessions C3 and C4, that applied conventional logging had the lowest harvest volumes, but the highest extracted log emissions as they favored species with high wood densities (0.73 and 0.75 g cm^{-3}, respectively).

Carbon emissions was about 40% lower under controlled logging (1.96 Mg C m^{-3}) and FSC logging (1.82 Mg C m^{-3}) compared to conventional logging (3.23 Mg C m^{-3}). The higher C emissions per cubic meter of timber extracted in conventionally logged FHUs is explained by low timber production in these FHUs, which was on average 50% less in conventionally logged FHUs compared to FHUs logged under controlled and FSC systems. We attribute the lower production volumes in conventionally logged FHUs to the lack of pre-harvest inventories, which can result in identification of 20% more harvestable stems (Marn and Jonkers, 1981; Putz et al., 2008) and can lead to an increase of 15% in timber volume harvested (Barreto et al., 1998). Additionally, as conventionally logged FHUs construct the same amount of infrastructure (Table 3) as the other management systems, but extract less timber, they have higher logging infrastructure emissions (Table 2). Though higher logging intensities reduce the total carbon emissions factor (Fig. 4), an appropriate balance needs to be struck, as higher logging intensities both deplete forest carbon stocks and lead to slower recovery time of forest carbon (Roopsind et al., 2018, 2017; Rutishauser et al., 2015). Additionally, higher logging intensities also negatively impact other ecosystem services such as biodiversity (Bicknell et al., 2014; Burivalova et al., 2014; Putz et al., 2012; Roopsind et al., 2017).

Carbon emissions from conventional logging also resulted in the highest collateral felling damage emissions (Table 2). The lower felling damage in the controlled and FSC logged FHUs compared to conventional logging (Table 4) could be a direct result of the application of directional felling practices with trained forestry workers. Observations in the field and communication with tree fellers though, gave the impression that the suite of practices intended to reduce felling damage and improve safety of forestry workers such as liana cutting, and directional felling are not fully applied or understood. A previous study conducted in Suriname has shown that the correct application of directional felling practices in under controlled logging reduces felling damage and protects future crop trees, in addition to aiding extraction (Hendrison, 1990). We cannot however definitively attribute the lower level of felling damage to the use of directional felling, as though FSC logged FHUs are required to apply directional felling and are audited on a regular basis for compliance, controlled FHUs that share the same level of collateral felling damage emissions in our study, are not mandated to apply directional felling.

In terms of logging infrastructure, haul roads, which result in clear patterns of forest loss and classified as deforestation, was 38% wider in conventionally logged forests than haul roads in controlled and FSC logged forests (Table 3). The average haul road width (24.7 m) observed in conventionally logged FHUs is however within Suriname's draft logging code that recommends a maximum width of 25 m (van der Hout, 2011). As we found the width of hauls in the control and FSC logged forests to be on average 10 m less than the recommended maximum width in the national logging code, we would recommend a downward revision of the maximum haul road width to reduce forest loss and associated carbon emissions. FSC and controlled FHUs also had shorter skid trails per extracted tree, an indication of effective skid trail planning. We were unable though to correlate our haul road emissions to use of specific machinery, and skill levels of machine operators, that explains forest road and other infrastructure quality (Hendrison, 1990; Majnounian et al., 2009; van der Hout, 1999).

The unextracted wood from felled trees was the largest source of emissions across all FHUs and logging systems. We suspect that the total carbon emissions from the unextracted portions of harvested trees are even higher than reported in our study due to repeated trimming and culling of logs before they are milled. To reduce these emissions, much of which can be considered wood waste (i.e., potential exists for utilization), there needs to be better access to appropriate timber milling machinery and training in crosscutting for improved wood utilization. Another source of carbon emission associated with the unextracted carbon pool that could be reduced with improved logging practices are trees felled and then rejected because of poor form, breakage, heart rots, or hollows or simply forgotten in the forest. We found the number of rejected trees were similar in the conventionally logged and controlled logged FHUs, but 50% less in FSC logged FHUs. This difference potentially reflects the benefits of training fellers to use practices such as plunge cuts, which FSC forestry workers indicated they were trained to apply.

In 2016, conventional, controlled, and FSC logged FHUs produced 309,569 m^3, 141,948 m^3, and 51,443 m^3 of timber, respectively in Suriname. Based on the carbon emission estimates per cubic meter of wood from our study, the forestry sector was responsible for approximately 1.37 million Mg C emissions, approximately 67% of the 2.06 million Mg C emissions from deforestation estimated for 2014–2015 (Government of Suriname, 2018).

Pre-harvest inventories, used to inform road and skid trail planning, represent a large opportunity for emissions reductions. For example, based on the 2016 timber production data and the emission factors per management type determined in this study, emissions could have been reduced by 393,263 Mg C if pre-harvest inventories and road planning were conducted in conventionally logged FHUs (effectively making them controlled logging FHUs). Additional emissions reductions across all management systems, albeit at a smaller scale relative to the phasing out of conventional logging, can be achieved by improved bucking practices, as the unextracted portions of harvested trees was the single largest source of carbon emissions. Improved bucking would also have the added benefit of higher timber volume production without felling additional trees or constructing additional roads and skid trails, as observed under the higher timber recovery in FSC-logged FHUs. Increase recovery of timber would, however, require investments in training and milling machinery that accepts shorter and less uniform logs.

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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.2019.02.026.

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Reduced-impact logging practices reduce forest disturbance and carbon emissions in community managed forests on the Yucatán Peninsula, Mexico

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ABSTRACT

On the Yucatan Peninsula in Mexico, communities (*ejidos*) that selectively log their forests help reduce deforestation and are an important source of timber for national and international markets. If carried out without proper planning and reduced-impact logging (RIL) practices, forest disturbances and carbon emissions from these harvests can be substantial. To assess variation in logging-induced emissions and to estimate potential reductions in those emissions, we estimated carbon impacts from damage to trees > 5 cm DBH in the annual cutting areas of ten forest-managing ejidos. Baselines were developed for emissions from felling, skidding and transport of timber and then ejidos were compared with respect to whether they were Forest Stewardship Council (FSC) certified, size of annual cutting area, logging intensity, and implementation of RIL practices, particularly directional felling, skid trail planning, and the use of small modified agricultural tractors instead of large forestry skidders. The carbon impacts of enrichment planting in multiple-tree felling gaps (400–1800 m^2) were also evaluated. Carbon emissions from selective logging averaged 1.52 $Mg\,m^{-3}$ but ranged 1.19–2.55 $Mg\,m^{-3}$ among the 10 ejidos. Most emissions were from the remnants of trees felled for their timber (73%), followed by skidding (11%), transport infrastructure (i.e. logging roads and landings; 8%), and collateral damage from felling (7%). Our analyses indicate that FSC certification was not associated with any difference in carbon emissions from selective logging but that employment of RIL practices resulted in fewer damaged trees and lower carbon emissions even in ejidos with high logging intensities. Use of modified agricultural tractors for log yarding (i.e., skidding) reduced C emissions by 0.15 $Mg\,m^{-3}$ or 5 $Mg\,km^{-1}$ of skid trail. Greater collateral damage was found in multiple felling gaps but the increased emissions were offset by reductions in the remnants of harvest trees. Adoption of RIL-C practices by all community forestry ejidos in the region would contribute substantially to the Mexican forest sector's efforts to mitigate climate change.

1. Introduction

Carbon emissions from tropical forest degradation now exceed those resulting from deforestation (Baccini et al., 2017), and a major cause of this degradation is selective logging (Griscom et al., 2009; Simula and Mansur, 2011; Morales-Barquero et al., 2014). Emissions from selective logging of tropical forests (0.5 Gt $year^{-1}$ of C; Putz et al., 2008b) can be reduced, and post-logging rates of forest recovery can be increased through implementation of reduced-impact logging (RIL) practices (Asner et al., 2010; Putz et al., 2008a). Selective logging should therefore not be considered degradation when harvesting and other silvicultural practices are applied by trained and supervised workers in ways that minimize biomass impacts, promote recovery, and sustain production of timber and other environmental services; well managed logged forests may even sequester more carbon than un-logged or unmanaged forest (Bray et al., 2011; Putz and Romero, 2015; Griscom

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et al., 2014). The carbon benefits of timber stand management are further enhanced if the harvested wood is utilized in place of carbon costly materials such as aluminum, steel, and concrete (Putz and Romero, 2015).

Despite the global potential for sustainable forest management (SFM) to supply timber and mitigate climate change, in reality it is not widespread in the tropics where only an estimated 10% of permanent forest areas are managed sustainably (Poudyal et al., 2018). Accordingly, at least since the Paris Agreement, the United Nations' REDD + program (Reducing Emissions from Deforestation and Degradation) has endorsed SFM as an important instrument to enhance carbon stocks and to reduce emissions in tropical developing countries. Furthermore, given that close to a quarter of the worlds forests are controlled by indigenous peoples or rural communities (White and Martin, 2002; Garnett et al., 2018), inclusion of community forest management (CFM) is critical to achievement of improved forest outcomes (Herold and Skutsch, 2011). Community forests provide important livelihood contributions to more than half a billion rural people world-wide, so integrating CFM into REDD + activities on the ground can be a low risk high success strategy to meet REDD + goals (Agrawal and Angelsen, 2009).

In Mexico, CFM helps maintain forest cover while it economically benefits rural forest communities (Bray et al., 2003; Bray et al., 2004; Antinori and Bray, 2005; Ellis and Porter-Bolland, 2008; Ellis et al., 2017). Over 60% of Mexican forests are owned by communities (Madrid et al., 2009) and CFM has contributed substantially to the countrýs forest sector for 40 years, providing an important source of timber as well as carbon reserves for climate change mitigation. Both nationally and internationally, CFM is promoted by government institutions and NGOs as a "climate smart" land use that also helps conserve biodiversity (Agrawal and Angelsen, 2009; CONAFOR, 2010; Bray et al., 2011). Recent carbon dynamics modeling indicates that Mexican forests under CFM, when combined with increased timber production and substitution of wood for carbon-intensive materials, are effective carbon sinks (Olguin et al., 2018). Moreover, recent field studies and household surveys strengthen the argument that conservation goals are better achieved when secure tenure, communal land management, and stronger community governance are in place (Rodriguez and Fleischman, 2018). It is in fact these qualities that have facilitated implementation of REDD+ in Mexico (Herold and Skutsch, 2011), bringing more attention to CFM as a promising strategy to combat climate change in Mexico and beyond.

On the Yucatan Peninsula, CFM takes place over an extensive area of tropical forests (about 7 million ha) and is conducted by *ejidos* (communities holding communal land tenure), mostly in the states of Campeche and Quintana Roo. Both states are important producers of tropical timber for national and international markets: annual volumes harvested range from 70,000 to 150,000 m^3 with approximate market values of $6-$12 million (SEMARNAT, 2010; 2016). Forest ecosystems on the Yucatan Peninsula also have high conservation values, forming part of the Selva Maya, the largest contiguous tropical forest region in Central America and Mexico (Rodstrom et al., 1998), including large protected areas such as Sian Kaan and Calakmul Biosphere Reserves. These forests, particularly in Quintana Roo, have also played a major role in the development of sustainable CFM in the Mexican tropics; ejidos such as Noh Bec, Caobas, Petcacab and Tres Garantías were pioneers in the development of community-based forestry enterprises. These developments began with strong support and subsidies by the state and federal government in collaboration with the German government (GTZ), under the Plan Piloto Forestal (1984–1998; Ellis et al., 2014a).

Forestry on the Yucatan Peninsula involves the selective removal of the commercially valuable timber that is often present at low densities (1–10 trees ha^{-1}; Ellis et al., 2014b). Ejidos engaged in CFM are legally required to have an authorized forest management plan (FMP) for timber harvests that need to follow a polycyclic silvicultural system with a 75-year rotation and 25-year cutting cycles (Ellis et al., 2014a). The 25-year cutting cycle takes into consideration a range of growth rates (0.4–0.8 cm $year^{-1}$) of the commercial species, although most FMPs focus on mahogany (*Swietenia macrophyla*), the most valuable timber species. Based on mandatory forest inventories, FMPs consider three size classes of timber: repopulation (10–25 cm DBH); reserve (25–35 cm DBH); and, harvestable (> 35 cm DBH). In the case of precious timber, mahogany and Spanish cedar (*Cedrela odorata*), and of chicle (*Manilkara zapota*), minimum cutting diameters (MCD) are larger (55 cm DBH) (Navarro-Martínez et al., 2018). Ultimately, how many and which trees are harvested varies with the volumes of different species demanded by buyers, and average only 30% of permitted volumes (Ellis et al., 2015; Rodriguez-Ward et al., 2016). Although the silvicultural system applied by CFM in the Mexican Selva Maya has sustained harvests for over 40 years, there are still concerns about regeneration and stocks of valuable timber species, particularly mahogany (Snook 2005a; Ellis et al., 2014a). Moreover, as noted above, there is recent interest in Mexico about the role of CFM in biodiversity conservation and climate change mitigation (AGECC, 2010; Cronkleton et al., 2011). Nevertheless, realizing the potential of CFM on the Yucatan Peninsula is impeded by silvicultural research gaps coupled with often perverse public policies and illegal logging (Ellis et al., 2014a; Ellis et al., 2015).

The potential of RIL to reduce carbon emissions from selective logging is well established (see Putz et al., 2008a; Asner et al., 2010; Griscom et al., 2014). RIL minimizes forest damage by applying practices such as preharvest inventories, planned logging road networks, directional felling, and winching (Bicknell et al., 2014). Across the tropics, RIL practices demonstrably reduce collateral damage and consequent carbon emissions: in Brazil, RIL showed a loss of 17% above-ground biomass compared to 26% with conventional logging (CL) (West et al., 2014); in Sabah, Malaysia, CL damaged 41% of residual trees < 60 cm DBH compared to 15% when implementing RIL practices (Pinard and Putz, 1996); and, in East Kalimantan, Indonesia, tree injury and death was lower with RIL (30%) compared to CL (48%) (Bertault and Sist, 1997). Research on carbon impacts from selective logging in the tropics that distinguished emissions from felled trees and from impacts on biomass due to felling, skidding and construction of logging infrastructure (i.e., log landings and logging roads) have also been conducted (Pearson et al., 2005; Pearson et al., 2006; Brown et al., 2011). Most recently, in six tropical countries Pearson et al. (2014) found large differences in gross carbon emissions from selective logging that ranged from 6.8 Mg ha^{-1} in Brazil to 50.7 Mg ha^{-1} in Indonesia. Given that carbon emissions per hectare predictably increase with logging intensity (i.e., volume of timber extracted per hectare), logging practices were best compared using the indicator of emissions per cubic meter harvested (i.e., Mg m^{-3} of C). Pearson et al. (2014) reported these carbon emissions as 0.66 Mg m^{-3} in Republic of Congo, 1.49 Mg m^{-3} in Indonesia, and 2.33 Mg m^{-3} in Guyana. Based on similar methods, Griscom et al. (2014) reported that carbon emissions from selective logging in Kalimantan, Indonesia averaged 51.1 Mg ha^{-1} and 1.5 Mg m^{-3}, with no overall differences between Forest Stewardship Council (FSC) certified and non-certified concessions other than lower skidding emissions from certified concessions. Despite the known contributions of selective logging to greenhouse gases and the recognized potential of RIL to reduce these emissions, there is still much to learn about improving practices that cause these emissions.

Tropical forest management can be improved in many different ways, such as through silvicultural treatments to increase the growth and regeneration of timber species, setting aside high-value conservation areas, implementing worker safety measures, and RIL, which is the focus of this study (Burivalova et al., 2017). Forest certification by the FSC, which is based on more than just timber harvesting practices, aims to promote SFM by meeting the high environmental and social standards associated with markets for certified timber. However RIL figures prominently in FSC principles and criteria (Gullison, 2003;

Rametsteiner and Simula, 2003; Ebeling and Yasué, 2009; but see Romero and Putz, 2018). In our research, we specifically focus on RIL practices that directly relate to carbon emissions from harvesting activities (felling, skidding, and transport of logs) and that can be measured in the field as described by Griscom et al. (2014) and Pearson et al. (2014). Known as RIL-C, these improved logging practices include directional felling, improved log extraction methods (i.e., skidding or yarding), skid trail and road planning, improved bucking, and long-line winching (Broadbent et al., 2006; Wit and Van Dam, 2010; Griscom et al., 2014). To allow comparisons among forests selectively logged at very different intensities, we express emissions both per hectare and per cubic meter of wood harvested. The only similar study in Mexico was conducted by Pearson et al. (2005) for logging in the temperate forests in the state of Chihuahua.

Estimating forest carbon emissions from selective logging and defining baselines are essential to the improvement of community-based forest management and emission reduction strategies on the Yucatan Peninsula. The outcomes of the research presented will aid in setting up results-based actions by the REDD + MRV component and strengthen the integration of community-based forestry as an important "natural climate solution", defined by Griscom et al. (2017) as conservation, restoration and improved management practices in natural terrestrial biomes to mitigate climate change.

To that end we adopted the RIL-C protocol (Griscom et al., 2014; Pearson et al., 2014) to quantify carbon emissions from selective logging in ten CFM ejidos on the Yucatan Peninsula. Biomass impacts from harvested trees and collateral damage from felling and skidding logs, in addition to those from the construction of log landings and logging roads, were measured in the field to assess carbon emissions and to establish baselines for harvesting practices in the logging landscape sampled. Furthermore, we evaluated how tree damage and carbon emissions relate to forest management certification by the FSC and the implementation of specific RIL-C practices (directional felling, skid trail planning and the use of modified agricultural tractors for skidding). Ejidos are assessed in terms of their performance in committed emissions (as per the IPCC Tier 1 accounting approach; Davis et al., 2014), and potential reductions in carbon impacts from implementing RIL-C practices are estimated.

2. Methods

2.1. Logging landscape

We quantified carbon impacts from logging in the forested landscape of the Yucatan Peninsula. Located in southeast Mexico, the peninsula lies atop a karstic plateau that emerged from the ocean during the Tertiary and Quaternary (Lugo and García, 1999; Bautista-Zúñiga et al., 2005). We focused on the logging landscape in the southeastern quadrant of the Peninsula (Fig. 1) where the topography is mostly flat with elevations ranging 0–400 m above sea level, with some hilly terrain characteristic of the central and southern portions (Lugo and García, 1999; Orellana et al., 1999). The climate of the study region is warm and humid, with mean annual temperatures of 24–26 °C and annual precipitation of 800–1200 mm, with a pronounced November-April dry season with < 60 mm of rain per month (Gutiérrez-Granados et al., 2011; Koleff et al., 2012). In upland areas of the logging landscape, soils are mostly rendzinas (leptosols and phaeozems) that are shallow to moderately deep, rocky, poor in organic matter, and well drained. In depressions the dominant soils are gleysols and vertisols which are moderately deep to deep, rich in organic matter, and poorly drained (Vester and Martínez, 2007; Bautista-Zúñiga et al., 2011).

Vegetation of the logging landscape varies with geomorphology, soil, and climate (Miranda, 1978; Durán and Olmsted, 1999; Flores-Guido et al., 2010) as well as from natural and anthropogenic disturbances that resulted in a landscape mosaic of forest at different successional stages (Ellis and Porter-Bolland, 2008). Hurricanes and fires are common, as are human impacts that date from the Maya civilizations thousands of years ago; the forests adapted to these frequent disturbances and are considered very resilient (Turner, 1981; Whigham et al., 1991; Snook, 1996; Snook and López, 2003; Navarro-Martínez et al., 2012). Forests are of low (10–15 m), medium (15 to 30 m), and high-stature (30–35 m) and vary from semi-evergreen (25–50% dry season leaf loss) to semi-deciduous (50–75% dry season leaf loss) (Miranda and Hernández-X., 1963; Flores and Espejel, 1994; Pérez-Salicrup, 2004). Above-ground forest biomass in the logging landscape reportedly averages 77 Mg ha^{-1} of C (Morfín-Ríos et al., 2015; CONAFOR, 2017), but ranges 50–90 Mg ha^{-1} (Douterlungne et al., 2013; Santos et al., 2015).

Logging in the study region is mostly in medium to high-stature forests that are semi-evergreen (Snook et al., 2005; Hernández-Stefanoni et al., 2006). These forests typically support around 100 tree species per hectare with common upland species including *Brosimum alicastrum*, *Manilkara zapota*, *Talisia olivaeformis*, *Bursera simaruba*, *Lonchocarpus longistylus*, *Nectandra salicifolia*, *Psidium sartorium*, *Guettarda combsii*, *Vitex gaumeri*, *Caesalpinia gaumeri* and *Lysiloma bahamense* (Hernández-Stefanoni et al., 2006; Gutiérrez-Granados et al., 2011). In flooded areas, *Hemotoxylon campechianum* and *Metopium brownei* are common, but many of the same species still occur (Flores and Espejel, 1994; Pérez-Salicrup, 2004). Over half of the tree species have commercial timber value (Vester and Martínez, 2007; Toledo-Aceves et al., 2009). As noted above, forests in the area typically recover quickly from disturbances (Whigham et al., 1991; Negreros-Castillo and Mize, 1993; Bonilla-Moheno, 2010; McGroddy et al., 2013,). Mahogany, a key timber species, occurs in higher densities on the Yucatan than in other regions in Latin America, and requires large canopy openings (at least 5000 m^2) for successful regeneration (Snook, 2005b). The ecological resilience and abundance of high-value timber favored the historical importance of forest management on the Yucatan Peninsula (Snook, 1998).

Ejidos engaged in forest management must delimit their management areas and specify the extent of each annual cutting area (ACA) during their authorized logging period. Mandatory forest inventories are conducted to determine existing volumes and potential harvest volumes within ACAs. Harvests are limited to 15–20 trees ha^{-1}, with authorized volumes of 7–20 m^3 ha^{-1}, depending on the ejido, but harvests are typically only 20–40% of what is authorized (Ellis et al., 2015). Timber harvests are based on a selection of mature, dead, and sick trees. Typically, trees to be felled are selected and marked in advance by forest technicians or logging crew chiefs, and are subsequently felled, bucked, and extracted, generally using a skidder (articulated forestry tractor; Fig. 2) that drags the timber to log landings of 400–1200 m^2 (Ellis et al., 2014b) locally called "bacadillas." Subsequently, logs are trucked to local sawmills or to points of sale outside the forest. Even ejidos that own sawmills sell some roundwood to local or national buyers for processing elsewhere. Timber harvests are typically conducted during the January to May dry season.

Some forest communities recently opted to use modified agricultural tractors for skidding rather than the much larger skidders that remain operating in the region (Fig. 2). These 85–100 HP farm tractors are fitted with: (1) a caged cabin; (2) extra protection for radiators, tire valves, lights, motor and axles; (3) thicker tires; (4) a front-mounted blade; (5) a rear-mounted winch; and, (6) spark protector for the exhaust. The cost of these outfitted tractors ranges $20,000–50,000 USD and are easily operated by community members. They are also cheaper to maintain by local mechanics than the forestry skidders, which are not readily available in Mexico and cost upwards of $20,000 for a very used 1970s model. Most skidders on the Yucatan Peninsula are legacies of the Plan Piloto Forestal Project (PPF), which promoted community forestry in the 1980s and 1990s (Ellis et al., 2015). These skidders are harder to operate than farm tractors, they sometimes need to be rented, and often require contracted operators. Ejidos that own their own skidders suffer shortages of replacement parts and qualified mechanics,

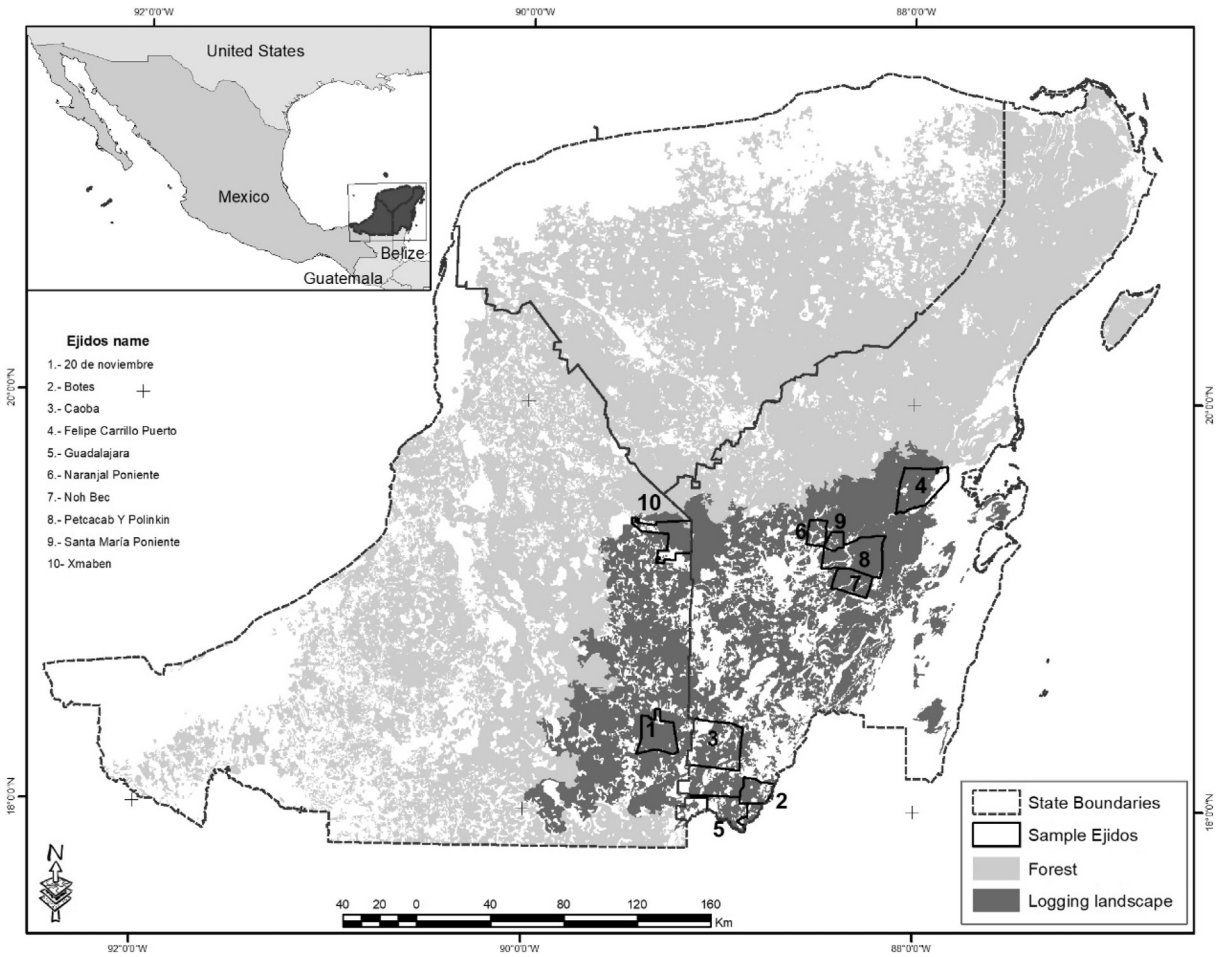

Fig. 1. The study logging landscape on the Yucatan Peninsula with the sampled ejidos indicated by number (see Table 1 for a key).

both of which are expensive. For these reasons, the opportunity to use modified agricultural tractors is especially beneficial for small ejidos that formerly relied on contracted skidders, which were costly and perceived to cause excessive disturbance.

2.2. Sampled CFM ejidos

We sampled ten CFM ejidos of which seven were selected at random. For the remaining three, we chose the only two ejidos in the region that were FSC certified at the time (Noh Bec and Caobas), and one ejido (20 de Noviembre) that was previously selected for a pilot study by The Nature Conservancy (TNC; Fig. 1). The latter was sampled by its forest technician, Caobas was sampled by a doctoral student as part of her dissertation project (S. Armenta-Montero, in preparation), and the others were sampled by our team with assistance from local community forest technicians. Random selection of the seven ejidos was from a population of 33 ejidos with active management plans and on-going logging that were stratified into those with large (> 500 ha) and small (< 500 ha) ACAs (Table 1). Given the high likelihood of positive selection bias in regard to the FSC certified communities, the results of

Fig. 2. A forestry skidder (left) and a modified agricultural tractor (right) used for log yarding on the Yucatan Peninsula.

Table 1
Characteristics of sampled ejidos. Forest biomass includes both above- and below-ground masses of trees > 7 cm DBH. DF = directional felling; STP = skid trail planning; MT = modified agricultural tractor.

Ejido	Forest biomass (Mg C ha^{-1})	Annual cutting area (ACA; ha)	Forest Mgmt. Area (ha)	Reported harvested volume from ACA (m^3)	Logging intensity (m^3 ha^{-1})	RIL-C practices	FSC
Caobas	76.6	Large (1059)	32,265	1605	Low (1.1)	DF, MT, STP	Yes
Noh Bec	64.2	Large (1008)	18,000	7000	High (6.9)	DF, STP	Yes
Petcacab*	76.8	Large (1180)	41,776	9000	High (7.6)	DF, STP	No*
Botes**	73.0	Small (400)	7358	500	Low (1.2)	MT,	No**
Guadalajara**	53.9	Small (240)	12,334	950	Medium (3.5)	MT	No**
Felipe Carrillo Puerto	95	Large (1843)	24,780	1600	Low (0.9)	None	No
Naranjal Poniente*	83	Small (300)	7500	1000	Medium (2.3)	None	No*
Santa María Poniente	78.6	Small (200)	4800	800	Medium (2.5)	None	No
X-Maben (Campeche)	88.7	Small (350)	2644	400	Low (1.1)	None	No
20 de Noviembre**	51.9	Large (1000)	22,725	700	Low (0.7)	None	No**

* Previously certified but lost FSC certification.
** Underwent certification auditing process but did not attain it.

our comparisons of FSC and non-FSC operations should be interpreted with caution.

Ejidos in this study vary in forest biomass as well as in overall extents, areas dedicated to forest management, and ACAs (Table 1). The ejidos also differ in other aspects of their forest environments as well as in characteristics of their management. For example, moisture regimes vary from the driest in Xmaben towards the northwest limit of the study region to the wettest in Guadalajara and Botes at the southern extreme.

Ejido forests also vary in their history of use and disturbance, as reflected in their biomass, tree species composition, and stocking of timber species. Ejidos also vary culturally, socioeconomically and institutionally as well as in their experience with forest management, organization, and governance. For example, some ejidos harvest timber through working groups (internal groups of community members) allocated different portions of the ACA, while others communally manage their forest with assigned members for specific logging and transportation tasks. Some ejidos in the study hired non-community members to run the heavy equipment. Ejidos also differ in their experience with natural forest management. For example, Noh Bec, Petcacab, Caoba, and Guadalajara have participated in forest management since the PPF period (1983) and were associated with that prominent tropical forestry project. Guadalajara, Caobas and Botes are members of the *Sociedad de Productores Forestales de Quintana Roo S.C.* whereas Naranjal Poniente and Santa Maria Poniente belong to the *Organizacion Ejidal de Productores Forestales de la Zona Maya S.C.*, both very influential local forestry organizations in operation since the early 1980s (Snook 2005a). Noh Bec and Petcacab formerly belonged to these forestry associations but Noh Bec left when they created their own community forestry office and Petcacab switched to an independent forestry technician. Of all the ejidos studied, Xmaben has the least experience in forest management and the least well-developed community forest organization and enterprise. As PPF was established in the state of Quintana Roo, the two ejidos from Campeche, 20 de Noviembre and Xmaben, did not experience as much influence from PPF nor the above-mentioned technical organizations.

ACAs are sub-divided into smaller management sub-blocks that are typically 100 ha. RIL-C practices implemented by at least some ejidos were directional felling (DF) and skid trail planning (STP) in FSC-certified ejidos (Noh Bec and Caobas) as well as in Petcacab, which is in the process of re-obtaining certification. It should be noted that all three ejidos lost their FSC certification due to ecological damage and economic problems caused by Hurricane Dean in 2007. Noh Bec and Caobas recovered certification in 2012 and 2013, respectively. Modified agricultural tractors (MTs) for skidding, which we consider a RIL-C practice, were employed in Botes and Guadalajara, which are members of a prominent forestry association and had undergone FSC certification audits but never attained certification, and in most (80%) of the FSC-certified ejido, Caobas. The three RIL-C practices considered for this study were also identified among the four most important improved management practices to reduce carbon impacts in our logging landscape, in addition to more efficient utilization of large branches and residues from felled trees (Villaseñor and Gonzalez, 2016). As noted above, Noh Bec, Caobas, and Petcacab were among the first established and most advanced community forestry enterprises, owning substantial sawmills that are also used by neighboring ejidos. In addition, Naranjal Poniente, Botes and 20 de Noviembre own small sawmills. Roundwood is also sold in all ejidos and processed in sawmills in nearby Felipe Carrillo Puerto, Chetumal, and Cancun.

2.3. Logging disturbance sampling

Forest disturbance from logging operations in the 2013 ACAs of Caobas and 20 de Noviembre were sampled January-April 2014; for the other eight ejidos we sampled the 2014 ACAs during March-November 2015. Two randomly selected 100 ha harvest blocks were sampled in ejidos with large ACAs, only one in ejidos with small ACAs. Delimited ACAs and sub-blocks were georeferenced from maps in forest management plans or from shapefiles provided by ejido forest technicians. Selected harvest blocks were then mapped in the field using Garmin GPS Map 60csx. In each sub-block all felled trees (stumps), skid trails, log landings, and logging roads were georeferenced. For field measurements of logging impacts on forest biomass, we adopted for Yucatan conditions the methodology provided by Griscom et al. (2014) and Pearson et al. (2014).

Felling biomass impacts (F) were calculated from samples of 5–15% of all trees felled for harvest in the harvest blocks (hereafter, harvest trees or HT). Stumps were selected by randomly choosing skid trails and then sampling every other stump or felling gap encountered. For each sampled gap, we recorded the location of each HT stump, determined its species, and measured the stump's diameter and height, removed log length (stump to canopy base), and diameter of the canopy base. The log harvested and removed from the forest is referred to as RW, and the stump, roots and canopy left in the forest make up the harvest tree remnants (HTR), which added to RW equals HT. In addition, collateral damage (CD) from felling was recorded for all trees > 5 cm DBH by species, DBH, and type of damage categorized as: TL (totally fallen), TS (trunk snapped), DC (> 50% damaged crown), BS (bark stripped), and LN (leaning ≥ 10°). Finally, damaged trees were categorized by species as commercial (COT), non-commercial (NCOT), or palm (P). When encountered, we skipped gaps caused by the felling of multiple trees (*bosquetes*) since in most cases they were already cleared for enrichment planting (Navarro-Martínez et al., 2017). Bosquetes were present in Noh Bec, Petcacab, Guadalajara and Caobas, and were established as part of a national reforestation program. To evaluate the carbon impacts of these larger multiple-tree felling gaps, we measured felling carbon impacts (HT and CD) in 10 bosquetes in Noh Bec, applying the

same methodology described for single-tree felling gaps.

Skidding impacts (S) were based on measures of damage to trees > 5 cm DBH in 15–20 skid trail plots in each sampled block that were 10 m long and the width of the trail. The plots were placed on randomly selected skid trails (see above) at random distances from the road or log landing. As in felling CD, species, DBH, and type of damage were recorded for damaged trees.

Estimates of emissions from logging roads (R) were based on measurements of their lengths and widths every 200 m within sampled harvest blocks. Similarly, estimates of forest disturbance from all log landings (L) in the blocks were based on measures of their areas. Emissions were estimated only for new roads; most roads in this logging landscape were initially constructed in the 1950s and were only rehabilitated for this round of logging (Ellis et al., 2015), which caused only minor emissions.

Reference levels of un-logged forest biomass in the sampled blocks were obtained by measuring basal area with a 2 BAF metric prism at 15–20 randomly selected points in unlogged stands past the ends of randomly selected sampled skid trails. In Caobas and 20 de Noviembre, un-logged forest biomass estimates were obtained from forest inventories of the sampled sub-blocks conducted prior to logging. This inventory consisted of 500 m^2 circular plots in which the DBH and height of all trees > 10 cm DBH were measured prior to logging.

Sampling intensities varied among ejidos but was at least 100 ha in small ACA ejidos, and 200 ha in large ones (Table 2). Larger areas were included in the ejidos sampled first (2013; Caobas and 20 de Noviembre) with the goal of including entire ACAs (847 and 1000 ha, respectively). The number of harvested trees tallied and georeferenced in the sampled harvest blocks of each ACA (from 190 to 704) varied with logging intensities and the sampled area. The number of felled trees sampled for felling impacts in the ACAs also varied (from 19 to 348), but in all cases except for 20 de Noviembre, 5–17% of harvest tree gaps in the sub-blocks were sampled. The length of sampled skid trails per ejido ranged 6–21 km whereas logging road lengths ranged 0.4–2.9 km. The number of log landings sampled in harvest blocks ranged 1–10, and the total area they occupied was relatively small (from 0.1 to 0.9 ha). Basal area measurements in un-logged forest of sampled sub-blocks (16–25 m^2 ha^{-1}) mostly reflect the differences in forest conditions prior to logging (Table 2).

2.4. Calculation of carbon emissions

The total emissions from logging, E, is equal to F + S + R + L. For F and S; DBHs of felled and damaged trees were used to estimate their biomass and carbon using allometric equations. Chave et al. (2014) was used for above-ground biomass (AGB) using model 4 (AGB = 0.0673(WD*DBH2*H)0.976), where WD = wood density and H = tree height. H was calculated with the equation (H = exp (0.893 − ENV + 0.760*In(DBH) − 0.0340*In(DBH)2)), where ENV is an indicator of environmental stress related to temperature, rainfall, and geographical location (Chave et al., 2014). Wood density data for biomass calculations were obtained from the Global Wood Density Database (Chave et al., 2009). Below-ground biomass of felled and damaged trees was estimated using Mokany et al. (2006), which allows estimation for individual trees with a root: shoot ratio of 0.205 for moist tropical forest < 125 Mg AGB. A carbon factor of 0.47 was used to convert biomass to carbon (C). Carbon in the removed log (RW) was calculated using field measurements of log length, upper diameter, and lower diameter. The biomass of the tree crown and stump (HTR) was estimated by subtracting the biomass of RW from total harvest tree biomass. We considered emissions according to the type of damage to trees from felling or skidding, where TL and TS were 100%, and DC 20% emissions. Finally, as indicated above, R and L carbon impacts were estimated by calculating the biomass removed during infrastructure construction. Carbon impacts (Mg) from selective logging in CFM ejidos were extrapolated to Mg m^{-3} based on RW volumes and Mg ha^{-1}, based on the total area sampled in the ACA. Soil carbon emissions were not considered but seemed minimal.

2.5. Baselines and statistical assessments

Based on the carbon emissions from selective logging in the ten sampled ejidos, we calculated emission baselines for the different emission sources resulting from harvest operations. Specifically, baselines consist of the means of carbon impacts from felling, skidding, and logging infrastructure construction (CD, RW, HTR, S, L, R and E), calculated as Mg ha^{-1} and Mg m^{-3}. Linear regression was used to examine relationships between carbon impacts (Mg ha^{-1} and Mg m^{-3}) and roundwood volume harvested (m^3) and logging intensity (m^3 ha^{-1}) in sampled harvest blocks. Total emissions (E) and emissions from the different logging activities in the sampled ejidos are then compared with the derived regional baselines.

We use ANOVAs on our sample of ten ejidos (N = 10) to assess the correlations of carbon emissions (Mg m^{-3}) from selective logging with FSC certification, implementation of RIL-C practices, and management characteristics, such as size of the ACA and LI (m^3 ha^{-1}). Mixed models were used to test the effects of implementing specific RIL-C practices (DF, STP, and MT), FSC certification, and ejido forest management characteristics (ACA and LI) on emissions and on the number of damaged trees from felling (CD) and skidding (S). These complex models are based on the same principle as general linear models and make it possible to use repeated measures and include random factors. The explanatory variables can be quantitative or qualitative, and referred to as fixed and random factors (XLStat, 2017). For the case of felling CD, we evaluated RIL-C practice of DF, and for skidding carbon impacts (S), we evaluated the RIL-C practices of STP and MT. Other explanatory variables or fixed effects used in the mixed model included FSC, ACA, and LI. Since most ejidos were selected at random and considering the large variability among them, as described above (e.g. forest environments, socioeconomic and cultural), we used ejido as a random effect in

Table 2
Mapping and sampling effort in ACAs of CFM study ejidos.

Ejido	ACA sampled (ha)	# Mapped stumps	# Sampled stumps	Mapped skid trails (km)	# Skid trail plots	Mapped logging roads (km)	# Logging road meas.	# and Area of log landings (ha)	# Biomass sample sites & basal area (m^2 ha^{-1})
Caobas	847	467	70	6.9	21	3.1	10	10 (0.40)	n/a
Noh-Bec	182	704	116	20.8	20	1.5	10	2 (0.75)	15 (18.7)
Petcacab	170	495	77	19.0	15	2.7	20	9 (0.87)	12 (21.8)
Botes	308	194	20	10.3	14	0.5	10	1 (0.12)	10 (20.9)
Guadalajara	270	370	19	16.9	18	1.0	10	2 (0.44)	10 (16.0)
F. Carrillo Puerto	240	255	38	16.4	15	2.9	20	4 (0.33)	15 (26.3)
Naranjal Poniente	116	190	69	9.1	17	1.2	8	3 (0.18)	15 (23.3)
Sta. Ma. Poniente	188	197	30	10.6	15	0.4	9	6 (0.56)	15 (22.3)
Xmaben	409	102	21	6.2	15	2.2	10	3 (0.29)	16 (24.7)
20 de Noviembre	1000	348	320	12.9	95	2.1	4	5 (0.34)	n/a

Table 3
Total carbon emissions and logging intensities from selective logging in sampled ejidos on the Yucatan Peninsula.

Ejido	Mg ha^{-1} of C	LI in sampled sub-blocks (m^3 ha^{-1})	Mg m^{-3} of C
Caobas	1.5	1.3	1.2
Noh Bec	9.0	6.7	1.3
Petcacab	5.6	4.6	1.2
Botes	1.7	1.2	1.4
Guadalajara	4.5	3.6	1.3
Felipe Carrillo Puerto	2.2	1.2	1.9
Naranjal	3.5	2.5	1.4
Sta. Ma. Pte	3.0	2.4	1.2
Xmaben	0.74	0.3	2.5
20 de Noviembre	1.1	0.7	1.6
Mean	*3.3*	*2.4*	*1.5*

the mixed model. Skid trail plots and felling gaps (HTs) sampled in each ejido are then used as sample units and treated as replicates in the mixed model. In the case of 20 de Noviembre where all HT and skid trails were sampled, we randomly selected 30 skid trail plots and 35 HT to have a comparable random sample. Tukey (HSD) pairwise comparison tests were also applied to compare carbon emissions means from felling and skidding impacts between categories of FSC (yes or no), ACA (large or small), LI (low medium and high), DF (yes or no), STP (yes or no), and MT (yes or no).

3. Results

3.1. Overall carbon emissions from selective logging

Selective logging induced carbon emissions from the 10 ejidos sampled on the Yucatan Peninsula averaged 3.3 Mg ha^{-1} but ranged from 0.8 to 9.0 Mg ha^{-1} (Table 3). Emissions per hectare were closely correlated with LI (m^3 ha^{-1}) in the sampled sub-blocks ($R^2 = 0.99$, F (1,8) = 617.4, $p < 0.0001$, Fig. 3). On the other hand, carbon emissions per cubic meter of timber harvested (Mg m^{-3}) was weakly correlated with LI ($R^2 = 0.28$, F (1,8) = 3.1, $p < 0.11$, Fig. 3). Since C emissions per area (Mg ha^{-1}) were correlated with logging intensity, we did not use this indicator for further comparison of emissions performance by ejidos. Instead, we use the indicator Mg m^{-3}, which was not correlated to LI, to present and compare carbon emissions performance.

The average total carbon emissions per volume of timber harvested for all 10 sampled ejidos was 1.5 Mg m^{-3} but varied from 1.2 to 2.5 Mg m^{-3}. Relative to this baseline, Xmaben, Felipe Carrillo Puerto, and 20 de Noviembre, which implemented none of the tracked RIL-C practices, all had carbon impacts > 1.6 Mg m^{-3}. Four of the five ejidos with the best performance (< 1.5 Mg m^{-3}) implemented RIL-C practices and two were FSC certified. Caobas, in first place, is FSC-certified and implements all three RIL-C practices; Petcacab in second, implements DF and STP; Guadalajara in fourth implements MT; and Noh Bec in fifth, is FSC certified and implements DF and STP. Santa Maria Poniente and Naranjal Poniente, although they do not use any of the monitored RIL-C practices, performed well, ranking third and sixth place respectively. Finally, in seventh, also below our baseline, was Botes which implements MT.

3.2. Carbon emissions from selective logging disaggregated by operations

Carbon emissions per cubic meter of timber harvested (Mg m^{-3}) from specific logging operations varied substantially among ejidos (Fig. 4 and Table 4). Harvest tree remnants (HTR) left in the forest after felling consistently constituted the major portion of carbon emissions, with a mean of 0.76 Mg m^{-3} (0.62–0.95 Mg m^{-3}), which represents 28–61% of total ejido emissions. After HTR, the round-wood extracted of felled trees (RW) represented average carbon emissions of 0.30 Mg m^{-3} (21%), quite consistently across ejidos (0.20–0.36 Mg m^{-3}; 11–26% of total emissions). Skidding (S) was the third largest cause of carbon emissions with an average of 0.18 Mg m^{-3} (11% of total emissions) but varying widely (0.03–0.46 Mg m^{-3} or 2–18% of total emissions). Ejidos with the lowest skidding impacts, Botes and Guadalajara (0.03 and 0.06 Mg m^{-3} and 2 and 5% of total emissions, respectively), used MT exclusively. Noh Bec and Petcacab, which implemented STP, were also among the ejidos with low skidding carbon emissions (both 0.11 Mg m^{-3} and 8% of total emissions). Following skidding, collateral damage from felling (CD) caused carbon emissions that averaged 0.12 Mg m^{-3} or 7% of total emissions. CD varied widely (0.04–0.29 Mg m^{-3} and 3–11% of total emissions), and ejidos that implemented DF (Noh Bec, Caobas, and Petcacab) had the lowest emissions. As noted above for overall carbon impacts, Xmaben, Felipe Carrillo Puerto and 20 de Noviembre performed above the baselines for felling and skidding damage. Carbon emissions from logging infrastructure construction were very low relative to other disturbances, averaging only 0.07 Mg m^{-3} (5% total emissions) but varying 0.02–0.21 Mg m^{-3} (2–8% of total emissions). There were only road emissions (R) from the five ejidos with new logging roads; they averaged 0.08 Mg m^{-3} (4% of total emissions) for all 10 ejidos. Xmaben, where all the roads were new, was the only ejido with substantial carbon emissions from road development (0.61 Mg m^{-3}), which constituted 24% of its total emissions.

For most baseline indicators of selective logging emissions by operation, Xmaben stands out as the worst performer, with the highest skidding impacts and a substantial portion of their carbon emissions from building new logging roads to access the ACA. Skidding impacts were also high in 20 Noviembre and Felipe Carrillo Puerto whereas felling collateral damage emissions were high in, Xmaben and 20 de Noviembre. The best overall performers (Noh Bec, Guadalajara, Botes and Caobas) had much lower carbon impacts from felling CD, S, L and R. These also include both FSC certified ejidos (Noh Bec and Caobas), plus all three ejidos that used MT for skidding (Botes, Caobas and Guadalajara).

The ANOVA model applied to assess the relationships between total carbon emissions and FSC certification, application of one or more RIL practices, and other management characteristics explained 76% of the variability in total carbon impacts (Mg m^{-3}) from selective logging for the ten sampled ejidos (N = 10) and was significant at the 10%, but not 5% significance level [F(4, 5) = 4.05, p = 0.079]. Significant explanatory variables at the 10% level of significance included RIL (p = 0.049), LI (p = 0.068), and ACA (p = 0.096). Model parameters show ejidos not applying RIL-C practices had higher emissions (p = 0.049), but no difference was found between FSC and non FSC certified ejidos. Furthermore, model parameters indicated that ejidos with large ACAs had lower carbon emissions (p = 0.096) and ejidos with low logging intensities (LI) had high emissions (p = 0.027). However, as mentioned above, ejidos with small ACAs and low to medium harvest intensities also performed well, with the implementation of RIL (Botes and Guadalajara) or without any visible RIL practices in their 2014 ACA (Santa Maria and Naranjal Poniente). The worst performers (Felipe Carrillo Puerto, Xmaben and 20 de Noviembre) did not implement the monitored RIL-C practices, had low harvest intensities, and, except for X-Maben, also had large ACAs.

3.3. Felling emissions

Descriptive statistics for the number of damaged trees and carbon emissions from felling (CD) in the ten ejidos (Table 5) show that the three ejidos with the lowest number of damaged trees (Caboas, 3.0; Petcacab, 4.5 and Noh Bec, 5.1) all implemented DF and all had the lowest carbon emissions per felled tree (Noh Bec, 0.18; Petcacab, 0.12 and Caobas, 0.13 Mg). Interestingly, Caobas with lowest number of damaged trees per tree felled, was the only ejido sampled that was

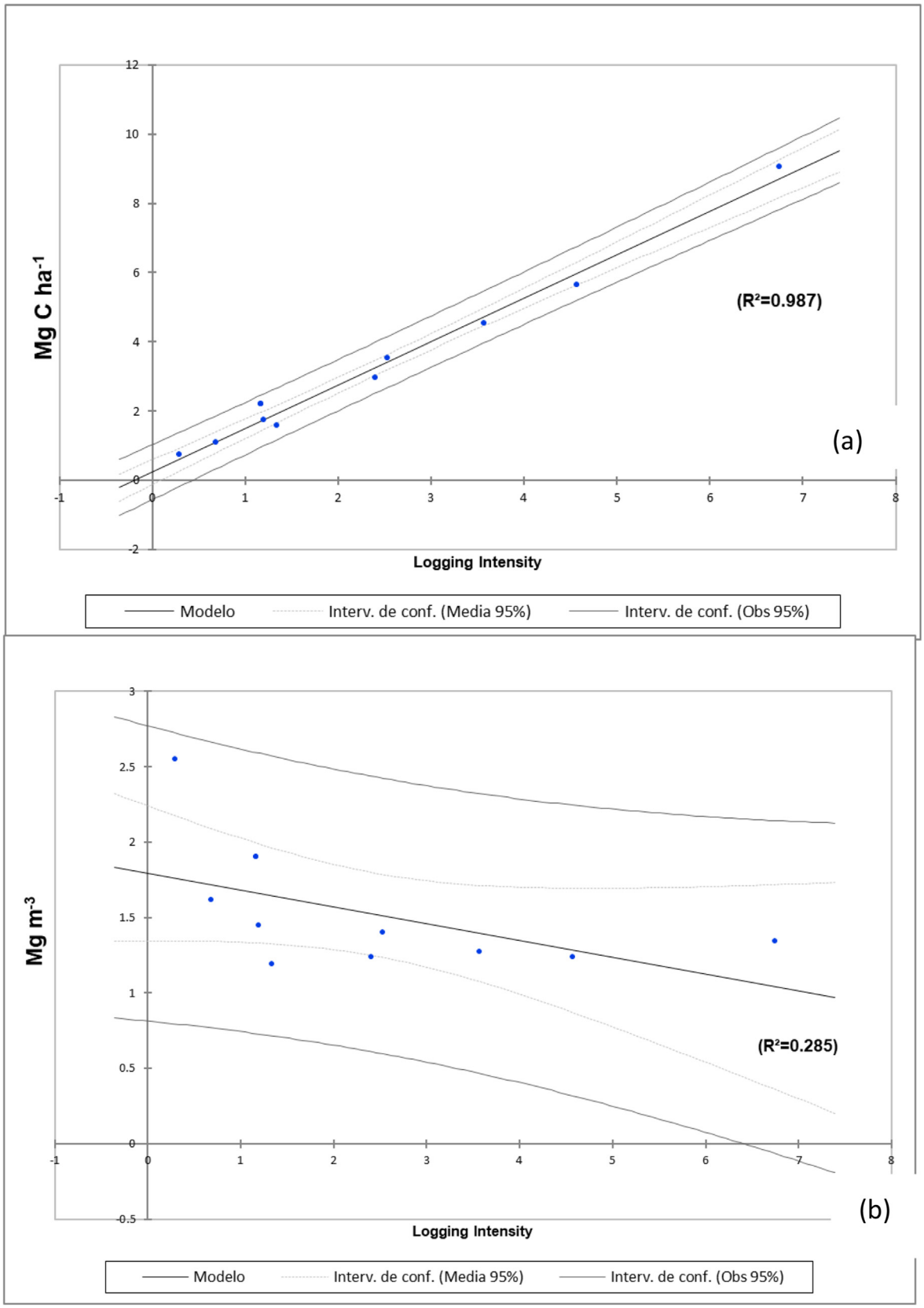

Fig. 3. Carbon emissions per area of ACA (Mg ha^{-1}) were strongly related to logging intensity (LI) expressed as cubic meters of timber harvested per hectare (a), but carbon emissions per volume of harvested timber (Mg m^{3}) were not strongly related to LI (b).

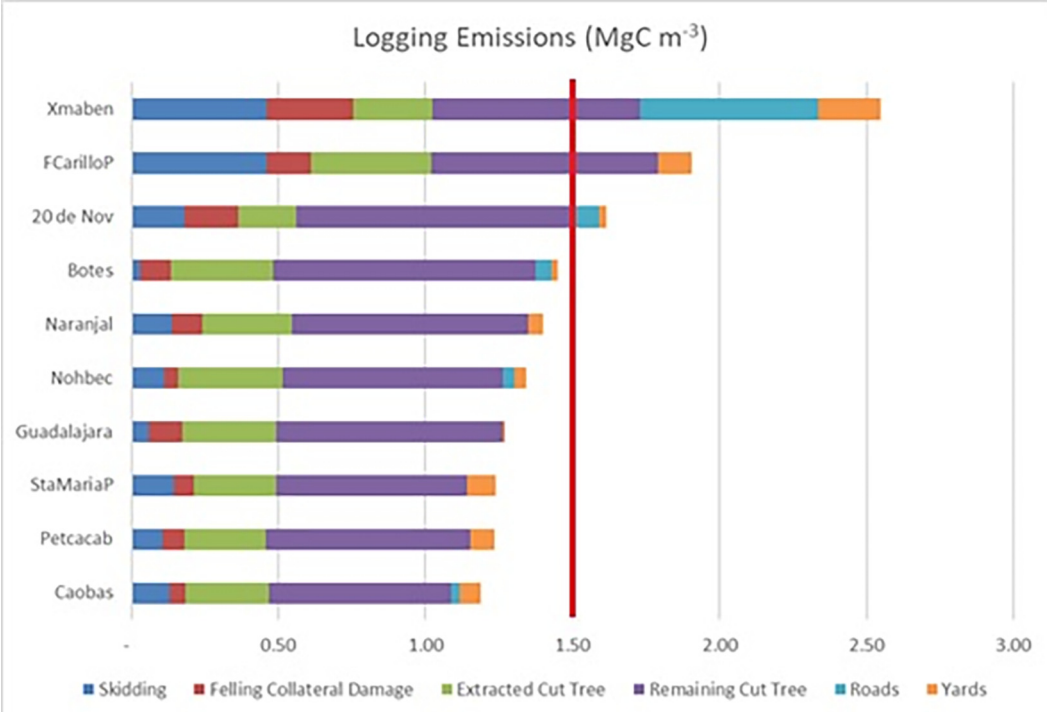

Fig. 4. Carbon emissions (Mg m^{-3}) from selective logging operations in community forestry ejidos on the Yucatan Peninsula. The red line indicates the total impact baseline calculated as the mean total emissions (Mg m^{-3}) of all ten ejidos.

cutting lianas prior to harvesting (at least 6 months). Again, as in the overall results, X-Maben and 20 de Noviembre had high carbon emissions per tree felled.

The mixed model type II results for the number of damaged trees per tree felled show that all fixed effects considered contributed to the model: FSC [F(1, 4) = 31.18, p = 0.005], DF [F(1, 4) = 18.02, p = 0.013], ACA [F(1, 4) = 8.75, p = 0.042] and LI [F(2, 4) = 17.39, p = 0.011]. In contrast, model parameters (Table 6) show that the most influential and significant variable was DF, indicating that ejidos that applied DF during felling damaged significantly fewer trees [t = 7.78, p < 0.001]. The model also shows that ejidos with high LI also damaged less trees per tree felled [t = 5.78, p = 0.004]. Tukey (HSD) comparisons show that there were no differences between ejidos with and without FSC certification (p = 0.19) or ejidos with large or small ACAs (p = 0.14). However, as reflected in the mixed model results, Tukey (HSD) comparisons show ejidos that implemented DF damaged fewer trees (p = 0.001), and ejidos with high intensities also damaged fewer trees than ejidos with medium (p = 0.013) and low (p = 0.017) intensities. Ejidos applying DF on average damaged four fewer trees per tree felled, and ejidos with high LI damaged on average two more trees per tree felled. With respect to carbon emissions per tree felled the mixed model type II results indicated that the fixed effects did not contribute to the model: FSC [F(1, 4) = 2.56, p = 0.185], DF [F(1, 4) = 1.94, p = 0.236], ACA [F(1, 4) = 0.79, p = 0.792] and LI [F(2, 4) = 0.56, p = 0.61] and mixed model parameters also showed no significance of the explanatory variables (Table 7). Tukey (HSD) also failed to show significant differences among FSC, DF, ACA and LI categories.

3.4. Skidding emissions

The number of damaged trees per 10 m long skid trail plot in ejidos ranged from 1.3 to 4.1, and the three ejidos that used a modified agricultural tractor for log yarding (MT) damaged the lowest numbers (Table 8). These results indicate that use of modified agricultural tractors for skidding can reduce forest impacts by more than 100 trees

Table 4
Carbon emissions per unit volume of timber extracted (Mg m^{-3}) from felling, skidding and logging infrastructure (log landings and roads) in sampled ejidos ordered from lowest to highest. F-RW = felled roundwood harvested, F-HTR = felled tree remnants in forest, CD = collateral damage from felling, S = skidding, L = log landings, R = haul roads, E = RW + HTR + CD + S + L + R.

Ejido	F-RW (% E)	F-HTR (% E)	F-CD (% E)	S (% E)	L (% E)	R (% E)	E
Caobas	0.28 (23.7)	0.62 (52.1)	0.05 (4.6)	0.13 (11.0)	0.07 (6.1)	0.03 (2.5)	1.19
Petcacab	0.27 (22.2)	0.70 (56.2)	0.07 (6.0)	0.11 (8.7)	0.09 (6.9)	0.00	1.24
Sta. Ma. Poniente	0.28 (22.5)	0.65 (52.5)	0.07 (5.4)	0.15 (11.8)	0.10 (7.9)	0.00	1.24
Guadalajara	0.32 (25.1)	0.77 (60.5)	0.12 (9.1)	0.06 (4.6)	0.01 (0.7)	0.00	1.27
Nohbec	0.36 (26.5)	0.75 (60.0)	0.04 (3.3)	0.11 (8.5)	0.04 (3.0)	0.04 (2.8)	1.34
Naranjal	0.30 (21.6)	0.80 (57.4)	0.11 (7.6)	0.14 (9.7)	0.05 (3.7)	0.00	1.40
Botes	0.35 (23.9)	0.89 (61.5)	0.10 (7.2)	0.03 (2.1)	0.02 (1.6)	0.05 (3.7)	1.45
20 de Noviembre	0.20 (12.3)	0.95 (58.6)	0.18 (11.3)	0.18 (11.1)	0.03 (1.6)	0.08 (5.1)	1.62
Felipe Carrillo Pto.	0.41 (21.5)	0.77 (40.6)	0.15 (7.9)	0.46 (24.1)	0.11 (5.9)	0.00	1.90
Xmaben	0.27 (10.6)	0.71 (27.7)	0.29 (11.5)	0.46 (18.1)	0.21 (8.4)	0.61 (23.7)	2.55
Baseline *Mean*	*0.30 (21.0)*	*0.76 (52.3)*	*0.12 (7.4)*	*0.18 (11.0)*	*0.07 (4.6)*	*0.08 (3.8)*	*1.52*

Table 5
Numbers of damaged trees and carbon emissions from felling collateral damage (CD) in community forestry ejidos ordered from lowest to highest emissions (Mg of C) and indicating forest management characteristics of ejidos. ACA = annual cutting area, LI = logging intensity, SD = standard deviation, HT = harvest tree.

Ejido	ACA	LI	RIL-C felling practice	Mean # damaged trees per tree felled (SD)	Total # damaged trees (# ha^{-1})	Mean DBH of damaged trees (SD)	Mean Mg of C from CD per tree felled (SD)
Noh Bec	Large	High	DF	5.1 (2.5)	3740 (21)	9.2 (4.5)	0.08 (0.09)
Petcacab	Large	High	DF	4.5 (2.0)	2299 (8.5)	11.0 (6.1)	0.12 (0.18)
Caobas	Large	Low	DF	3.0 (1.8)	1404 (1.6)	15.2 (7.1)	0.13 (0.17)
Santa María Poniente	Small	Medium	None	5.2 (2.3)	962 (5.1)	12.1 (6.2)	0.16 (0.16)
Felipe Carrillo Puerto	Large	Low	None	6.1 (2.7)	2678 (11.1)	10.9 (5.1)	0.16 (0.14)
Naranjal Poniente	Small	Medium	None	5.6 (2.8)	2358 (20.3)	11.2 (6.3)	0.21 (0.05)
Botes	Small	Low	None	5.1 (2.0)	984 (3.2)	11.7 (6.9)	0.26 (0.02)
Guadalajara	Small	Medium	None	5.8 (3.6)	1908 (7.1)	14.0 (10.0)	0.32 (0.62)
X-Maben (Campeche)	Small	Low	None	6.4 (2.2)	922 (2.2)	12.2 (7.8)	0.34 (0.53)
20 de Noviembre	Large	Low	None	7.2 (2.1)	2436 (2.4)	16.3 (7.2)	0.34 (0.24)

Table 6
Mixed model parameters for number of damaged trees per felled tree with FSC, DF, ACA and LI as fixed factors. ACA = annual cutting area, LI = logging intensity, SD = standard deviation, HT = harvest tree.

Source	Coefficient	Standard error	GL	t	Pr > |t|
Intercept	5.756	0.374	480	15.397	< 0.0001
FSC-NO	0.000				
FSC-YES	0.558	0.352	4	1.585	0.188
DF-NO	0.000				
DF-YES	−4.181	0.537	4	−7.781	0.001
ACA-SMALL	0.000				
ACA-LARGE	0.867	0.472	4	1.837	0.140
LI-LOW	0.000				
LI-MED	−0.209	0.434	4	−0.481	0.655
LI-HIGH	2.129	0.362	4	5.877	0.004

Table 7
Mixed model parameters for carbon emissions per felled tree with FSC, DF, ACA and LI as fixed factors. ACA = annual cutting area, LI = logging intensity, SD = standard deviation, HT = harvest tree.

Source	Coefficient	Standard error	GL	t	Pr > |t|
Intercept	0.281	0.063	480	4.476	< 0.0001
FSC-NO	0.000				
FSC-YES	−0.035	0.108	4	−0.323	0.763
DF-NO	0.000				
DF-YES	−0.077	0.145	4	−0.531	0.624
ACA-SMALL	0.000				
ACA-LARGE	−0.035	0.086	4	−0.411	0.702
LI-LOW	0.000				
LI-MED	−0.074	0.079	4	−0.938	0.401
LI-HIGH	−0.053	0.109	4	−0.492	0.648

km^{-1}. Skid trail carbon emissions in measured plots ranged 0.01–0.09 Mg per 10 m. Four of the five lowest carbon impacts (< 5Mg C km^{-1} of skid trail) were measured in ejidos with MT and/or STP.

Table 9
Mixed model parameters for number of damaged trees from skidding with FSC, DF, ACA and LI as fixed factors. ACA = annual cutting area, LI = logging intensity, SD = standard deviation, HT = harvest tree.

Source	Coefficient	Standard error	GL	t	Pr > |t|
Intercept	1.421	0.437	157	3.248	0.001
FSC-NO	0.000				
FSC-YES	0.200	0.713	4	0.281	0.793
MT-YES	0.000				
MT-NO	1.917	0.429	157	4.466	< 0.0001
STP-NO	0.000				
STP-YES	−0.897	0.988	4	−0.907	0.416
ACA-SMALL	0.000				
ACA-LARGE	0.910	0.559	4	1.629	0.179
LI-LOW	0.000				
LI-MED	−0.017	0.485	4	−0.035	0.974
LI-HIGH	0.248	0.756	4	0.328	0.759

Again, carbon emissions were lowest in Botes and Guadalajara where MTs were used exclusively.

Mixed model Type II results show that MT was the sole contributor to the model: FSC [$F_{(1,\ 3)} = 0.44$, $p = 0.555$], MT [$F_{(1,\ 3)} = 36.43$, $p = 0.009$], STP [$F_{(1,\ 3)} = 0.023$, $p = 0.888$], ACA [$F_{(1,\ 3)} = 4.2$, $p = 0.131$] and LI [$F_{(2,\ 3)} = 0.013$, $p = 0.987$] and model parameters (Table 9) also show that MT was the most significant variable ($p = 0.027$), showing that ejidos that did not use MT damaged more trees by skidding. All other variables (FSC, STP, ACA and LI) were not significant in the model. Tukey (HSD) tests also confirm that the only differences in skidding emissions were between ejidos that used MT and those that did not ($p = 0.027$), with no effects of the other fixed variables detected (FSC, STP, ACA and LI). Mixed model results for carbon impacts from skidding were similar. Type II test results showed that MT was the most influential fixed effect and all other were not significant: FSC [$F_{(1,\ 3)} = 0.004$, $p = 0.953$], MT [$F_{(1,\ 3)} = 6.37$, $p = 0.086$], STP [$F_{(1,\ 3)} = 1.178$, $p = 0.357$], ACA [$F_{(1,\ 3)} = 0.094$, $p = 0.78$] and LI

Table 8
Damaged trees and carbon emissions (S) from 10 m long skid trails plot ordered from lowest to highest with relevant forest management characteristics of the ejidos.

Ejido	ACA	LI	RIL-C practices	Mean # damaged trees/plot (SD)	# Damaged trees km^{-1}	Mean DBH (SD)	Mean Mg of C/plot (SD)	Mg C km^{-1}
Botes	Small	Low	MT	1.3 (1.1)	129	7.8 (5.4)	0.01 (0.02)	1
Guadalajara	Small	Medium	MT	1.5 (1.1)	150	8.9 (4.5)	0.03 (0.04)	3
Naranjal Poniente	Small	Medium	None	2.4 (2.0)	240	7.7 (4.3)	0.04 (0.05)	4
Petcacab	Large	High	STP	3.6 (1.5)	360	8.8 (3.5)	0.04 (0.03)	4
Caobas	Large	Low	MT STP	2.0 (1.4)	200	16.5 (12.3)	0.05 (0.04)	5
Noh Bec	Large	High	STP	3.8 (1.8)	380	9.3 (3.7)	0.07 (0.06)	7
Felipe Carrillo Puerto	Large	Low	None	3.3 (1.7)	330	9.1 (4.3)	0.08 (0.10)	8
Santa María Poniente	Small	Medium	None	4.1 (2.5)	410	8.8 (4.0)	0.08 (0.07)	8
X-Maben (Campeche)	Small	Low	None	3.5 (1.9)	350	10.2 (4.7)	(0.09) (0.07)	9
20 de Noviembre	Large	Low	None	5.5 (1.9)	550	8.9 (4.5)	0.09 (0.6)	9

Table 10
Mixed model parameters for carbon emissions from skidding with FSC, DF, ACA and LI as fixed factors. ACA = annual cutting area, LI = logging intensity, SD = standard deviation, HT = harvest tree.

Source	Coefficient	Standard error	GL	t	Pr > |t|
Intercept	1.421	0.437	157	3.248	0.001
FSC-NO	0.000				
FSC-YES	0.200	0.713	4	0.281	0.793
MT-MT	0.000				
MT-NONE	1.917	0.429	157	4.466	< 0.0001
STP-NONE	0.000				
STP-STP	−0.897	0.988	4	−0.907	0.416
ACA-SMALL	0.000				
ACA-LARGE	0.910	0.559	4	1.629	0.179
LI-LOW	0.000				
LI-MED	−0.017	0.485	4	−0.035	0.974
LI-HIGH	0.248	0.756	4	0.328	0.759

[$F_{(2, 3)} = 0.319$, $p = 0.749$. Mixed model parameters of carbon impacts from skidding (Table 10) also show that ejidos not using MT had somewhat higher emissions although not significant at the 5% level ($p = 0.092$). Tukey (HSD) tests failed to detect differences between all fixed effect categories.

3.5. Bosquete impacts

In the additional 10 multiple-tree felling sites sampled in Noh Bec (bosquetes) carbon emission from CD was higher (0.1 Mg m^{-3}) than in the single-tree gaps of the same ejido (0.04 Mg m^{-3}), but this was offset by lower HTR carbon emissions (0.67 Mg m^{-3}) compared to the single tree gaps (0.75 Mg m^{-3} of C). Skidding and hauling emissions were identical, since the same infrastructure was used to access bosquetes as single-tree gaps.

4. Discussion

4.1. Carbon emissions from selective logging on the Yucatan Peninsula

We evaluated forest disturbance and carbon emissions from selective logging in ten ejidos on the Yucatan Peninsula to establish baselines from which to assess potential emissions reductions from implementing RIL-C practices. Results from this study should facilitate monitoring and implementing carbon credit programs for community forestry enterprises in the region. The per hectare total carbon emission baseline we measured (3.3 Mg ha^{-1}) was much lower than the lowest values reported by Pearson et al. (2014) for Brazil (7 Mg ha^{-1}), and our highest carbon emissions (9.0 Mg ha^{-1}) was threefold lower than the highest values reported for Indonesia (51 Mg ha^{-1}; Griscom et al., 2014; Pearson et al., 2014) and Guyana (30 Mg ha^{-1}; Pearson et al., 2014). We found a very strong positive correlation ($R^2 = 0.98$) between emissions per hectare and logging intensity (m^{-3} ha^{-1}), as also noted by Griscom et al. (2014) and Pearson et al. (2014) and reported previously for Indonesia (Bertault and Sist, 1997) and Guyana (Blanc et al., 2009). The low observed per hectare emissions from selective logging on the Yucatan Peninsula need to be interpreted in terms of the very low logging intensities; the overall mean intensity for the 10 ejidos was only 1.4 trees ha^{-1} and 2.4 m^3 ha^{-1} with maxima of 4 trees ha^{-1} and 8 m^3 ha^{-1}. These intensities are far lower than the recommended maxima suggested by researchers concerned about the effectiveness of RIL (8 tree ha^{-1}; Sist et al., 2003; Roopsind et al., 2018) and maintenance of biodiversity (10 m^3 ha^{-1}; Burivalova et al., 2014).

On the Yucatan Peninsula, the mean carbon emissions per volume of harvested timber (1.5 Mg m^{-3}) was midway among the 0.99 to 2.33 Mg m^{-3} values reported by Pearson et al. (2014) for forests across the tropics that varied in biomass, standing stocks of timber, logging intensity, and the sizes and wood densities of harvested trees, all factors that can influence carbon impacts from selective logging. However, as in the Griscom et al. (2014) study in Indonesia, we also observed large regional variation in total emissions from selective logging (1.2–2.5 Mg m^{-3}). In our logging landscape, ejidos varied in forest types, biomass and management characteristics such as logging intensity, size of ACA and application of improved practices. In this study, all the sampled ejidos were in their second 25-year logging cycle and may have been logged prior to the 1980s by the parastatal concession MIQRO (Ellis et al., 2014a). A long history of forest use and management, natural and anthropogenic disturbances, and environmental differences among the sampled ejidos explain the large variations in forest biomass, structure and composition, which also influence harvested species and volumes extracted (Ellis et al., 2015). This landscape-level diversity needs to be considered when assessing carbon emissions as well as when implementing landscape-scale conservation and development strategies.

4.2. Variation in carbon emissions and RIL-C practices among ejidos

Differences in forest environments and current and historical forest management interventions complicate the assessment of carbon emissions with respect to implementation of RIL-C practices. On the Yucatan Peninsula, ejidos often manage their forests for a diversity of products, not exclusively timber; some ejidos harvest primarily large diameter trees of high-value timber with medium to high wood densities, such as mahogany and chicozapote, while others harvest common species of smaller size classes and lower wood densities for charcoal and polewood (Sierra-Huelsz et al., 2017). Further complicating RIL-C performance assessments is the fact that RIL has never been researched and piloted in Mexico as it has in other tropical countries such as Malaysia, Guyana, Gabon and Brazil (e.g., Putz et al., 2008a; Blanc et al., 2009; Medjibe et al., 2013; Vidal et al., 2016). Although improved forest management, sustained yields, and biodiversity conservation are pursued in the region, RIL is still an acronym that is pretty much absent from the vocabulary of community foresters and forestry institutions in Mexico. And while dozens of forest management plans developed for ejidos in our logging landscape mention the application of improved practices such as DF and STP, the reality on the ground is that very few implement them. Surprisingly, even though the Yucatan Peninsula has been globally recognized since the 1980s for its cases of successful and sustainable community-based forest management, it still lacks genuine RIL extension efforts.

RIL-C practices, such as DF and STP, are implemented by some ejidos, but mostly because of their pursuit of FSC certification coupled with training programs, outside of a larger RIL extension endeavor. As mentioned above, the particular ejidos associated with FSC and involved with PPF (Noh Bec, Caobas, Petcacab) have also been central in the development of community forest management in the region. Key PPF impacts on forestry in the region pertain to the creation of permanent forest areas, devolved forest management authority to the ejidos, and the establishment of six community forestry enterprises including the purchase of sawmills and extraction machinery (Wilshusen, 2005). These three ejidos mentioned above, along with Tres Garantias, Chacchoben, and Nuevo Guadalajara, were the six main beneficiaries of the PPF project during the 1980s and 1990s. However, our results show that other ejidos with a history of involvement with major local silvicultural organizations (*Sociedad de Productores Forestales de Quintana Roo S.C.* and *Organizacion Ejidal de Productores Forestales de la Zona Maya S.C.*), also performed relatively well and were not FSC certified (e.g. Santa Maria Poniente), but may have been certified or pursued certification in the past (e.g. Naranjal, Botes and Guadalajara). Improved forest management and RIL practices (although not labeled as such), which were initiated by the PPF and followed through by these forestry institutions a decade before the first FSC certified ejidos, may have been central in reducing forest impacts from selective logging in the region. With respect to the recent practice of using MT,

which this study shows substantially reduces skidding emissions, the ejidos that opted for this technology did so because it reduces costs and eliminates the need to out-source skidding operations or to sell standing timber directly to buyers who then organize all harvest operations. For ejidos with small ACAs, MTs are an appropriate skidding technology. Our results indicate that rather than FSC certification, it is RIL practices that reduce carbon emissions, benefits that may be maintained in most ejidos due to their experience and association with community forestry associations since the PPF project. Even though some of these ejidos have medium to high logging intensities, the implementation of RIL-C practices such as DF, STP and MT reduced their carbon emissions from timber harvest operations

Despite variation among ejidos in logging-induced carbon emissions, this research demonstrated similar patterns in carbon impacts as in other tropical regions where selective logging is used. Our results also affirm that the majority of carbon emissions (73%) originate from the harvested trees (Griscom et al., 2014; Pearson et al., 2014), as the timber removed (21%) and the remnants of crowns and branches left in the forest (52%). Thus, to reduce carbon emissions from selective logging effectively and efficiently, attention is warranted to felling and bucking practices that result in greater timber recovery from trees felled for that purpose (e.g., lower stumps). These practices are hardly considered by most logging operations on the Yucatan Peninsula, although recently markets have developed for smaller and irregular bole parts and large branches for handicrafts.

Among the ten ejidos for which we measured selective logging-induced carbon emissions, three performed particularly badly (Felipe Carrillo Puerto, Xmaben and 20 de Noviembre). These poor performers did not implement any RIL-C practices and logged at the lowest intensities. In the worst-case ejido in terms of carbon emissions (2.5 Mg m^{-3}), we learned during field work that the ejido was going to suspend its forestry operations due to internal governance and management problems. Instead of carrying out the logging themselves, they had arranged for the buyer to enter the forest and harvest the agreed species and volumes, which consisted of a small volume of high-value species and was done in a visibly reckless manner. In contrast, the best performers had mostly adopted at least one of the three RIL-C practices we tracked, although in the case of Santa Maria Poniente, the ejido was not documented as implementing RIL-C during the 2014 harvest. Generally, most ejidos performed well and differences in harvest intensities were not associated with carbon impacts below the baselines. For example, Petcacab, which is in process of returning to FSC certification, had the second highest logging intensity but also the second lowest carbon emissions. On the other hand, Noh Bec with the highest logging intensity (6.7 m^3 ha^{-1}) and a large ACA (> 500 ha), performed midway despite having the most experience with FSC certification, implementation of improved practices and well-organized and planned harvest operations, strategically establishing log landings, roads, and main skid trail networks in their ACAs. Caobas, the other FSC certified ejido, also has a large ACA but harvested at low intensity (1.1 m^3 ha^{-1}) and had the lowest carbon emissions per cubic meter of timber extracted. Loggers in Caobas very efficiently reduced harvest waste and minimized collateral damage by applying DF and STP RIL-C practices, but also by using a MT during harvest operations. Small CFM ejidos that belonged to a prominent forestry organization, such as Guadalajara (previously FSC certified) and Botes (which had undergone FSC certification audit but was never certified), were also good performers, and controlled their emissions from skidding through use of a modified agricultural tractor (MT), the only RIL-C practice they implemented. Several other ejidos with small ACAs (< 500 ha) and medium logging intensities (from 2 to 4 m^3 ha^{-1}) showed low emissions but without the obvious implementation of RIL-C practices, which argues for the importance of the tradition and experience of proper forest management by some forest communities and their technicians. Overall, it is important to keep in mind that given the lack of support for climate change mitigation, minimizing carbon emissions is not among the many goals of forest managers in the region. When these emissions are minimized, it is for other reasons.

As mentioned above, not only are harvest intensities very low in our logging landscape compared to other tropical countries, harvest volumes are often less than half of what is permitted. Explanations for these conditions are complex and involve environmental, institutional, cultural and economic barriers to increased production. Understanding and overcoming these barriers could contribute substantially to the economic benefits from small-scale community forestry enterprises in the region while maintaining carbon stocks. Our results are assuring, in the sense that community forestry ejidos on the Yucatan have the potential to increase their timber volumes and logging intensities and at the same time reduce carbon emissions by applying RIL-C practices.

4.3. Potential reduction of carbon emissions with RIL-C practices

Given that impacts from felling and skidding still constitute substantial portions of carbon emissions from selective logging in our study region (7.4% and 11.0%, respectively), any improvements in these activities are important. Comparisons of felling and skidding impacts in sampled ejidos may answer the question of why emissions from some ejidos with medium-to-high logging intensities are lower per cubic meter of timber harvested than in the three ejidos that log at very low intensities. With regards to felling, all three ejidos that implemented DF (Noh Bec, Petcacab and Caobas) damaged the least number of trees (3–5) and had the lowest carbon impacts (0.08–0.13 Mg) per tree felled.

The greatest differences in damaged trees and emissions per felled tree were found between ejidos that implemented DF and those that did not. Caobas, the only ejido that cut lianas on trees to be felled prior to harvesting, had the least damage and emissions from felling, demonstrating the potential of including this practice in a package of RIL-C practices. Differences between large and small ACAs are small, but low logging intensity ejidos damaged more trees and showed higher carbon impacts from felling. These results suggest that the practice of DF rather than reduced logging intensity is responsible for reductions in collateral damage from felling. As noted above, other ejidos (e.g., Botes and Santa María Poniente) did not implement DF but nevertheless performed relatively well, which shows the difficulty in determining when and to what extents RIL-C practices (e.g., DF) are actually implemented.

The skidding emissions data demonstrate that all ejidos that used MT, implemented STP, or both, in the case of Caobas, performed well. Botes and Guadalajara, the only two ejidos that used MT exclusively, were the two best performers, with skid trail carbon emissions half or less than the average (1 and 3 Mg km^{-1}, respectively versus 6 Mg km^{-1}). The potential to reduce skidding carbon emissions by 5 Mg km^{-1} is worthy of attention. The carbon emission reduction potential of STP was less (3 Mg km^{-1}) but still large. Again, this indicates that on the Yucatan Peninsula it is not the intensity of logging that results in lower carbon emissions, but instead how the harvesting is done (Bicknell et al., 2014). The potential of RIL-C practices to reduce carbon emissions from selective logging by ejidos on the Yucatan Peninsula is well demonstrated by this research, but the other benefits of these practices deserve attention. Directional felling, for example, can reduce damage to future crop trees (Galante et al., 2012) and improved bucking techniques can increase the volume of wood obtained from felled trees and thereby increase economic gains. As mentioned earlier, the use of MT reduces costs of skidding and dependence on external contractors. Proper planning of skid trails, log landings and haul roads also reduce the costs of timber harvesting and any subsequent silvicultural treatments. There are also many biodiversity benefits of RIL that deserve attention (Burivalova et al., 2014; Bicknell et al., 2014).

Despite the rewards that adoption of RIL practices can bring ejidos, Mexico, and the world, there are still barriers that could explain why few ejidos on the Yucatan Peninsula do so. The presumed market benefits of FSC certification could motivate adoption of RIL practices insofar as those practices are required by FSC auditors. Despite the

major focus in the past five years by CONAFOR and international institutions (e.g. PNUD, Rainforest Alliance) on promotion of forest certification in the region, up to this date (late 2018), only Petcacab, which was enroute to certification when we sampled it in 2015 and one more ejido we did not sample, became certified, totaling four in our study area. The costs of certification are clearly an obstacle that only larger ejidos with larger management operations and harvest volumes can overcome. More generally, for the majority of smaller and less productive ejidos there is still insufficient institutional and financial support to promote RIL and other improved forest management practices. This condition is particularly unfortunate given that extension services and training for some practices such as DF and STP are inexpensive. Credit or subsidies to purchase smaller forestry or modified agricultural tractors (MT) could also be a cost-effective mechanism to promote RIL and to increase timber production and profits for smaller ejidos while reducing deleterious carbon and biodiversity impacts. CONAFOR and other institutions do provide subsidies for improved forest management, but they benefit mostly the larger and more productive ejidos.

Efforts to reduce stand damage caused by selective logging on the Yucatan Peninsula face a paradox that emerges because most of the high-value timber species are light demanding (Fredericksen and Putz, 2003). Of these species that require large canopy openings to regenerate, mahogany, is the best known (e.g., Snook, 2005a, 2005b) but there are others, including Spanish cedar and tzalam (*Lysiloma bahamensis)*. To promote regeneration of these species, large canopy gaps created by felling multiple trees (*bosquetes*) are considered ideal (Navarro-Martínez et al., 2017), but the creation of such openings releases substantial carbon. These bosquetes are further cleaned and reforested with high-value and light-demanding timber species such as mahogany and ciricote (*Cordia dodecandraa*); if the planted trees grow quickly, the carbon debt is at least partially repaid, but only over several decades. Where RIL-C practices are not employed, the resulting hurricane-mimicking damaged areas, such as the wide skid trails, roads, and log landings, are also suitable sites for both natural regeneration and enrichment planting. Our research demonstrated that in bosquetes, the increased carbon impacts from collateral damage could be offset by the improved bucking of HT remnants. The integration of these multiple-tree felling gaps alongside conventional selective logging applying RIL-C practices can still result in substantial carbon emission reductions.

A new development in silvicultural practices is underway on the Yucatan that involves a severe tradeoff between timber production and carbon sequestration. Recently piloted in two ejidos with small trees and a scarcity of high-value timber, the silvicultural system calls for clearcutting patches of up to 3 ha and commercializing the cut trees as polewood, charcoal, and saw timber (Negreros-Castillo et al., 2018). The clearcuts are then to be planted with seedlings of commercial timber species, but the future yields, carbon dynamics, and biodiversity impacts of this new system are not known. A small clear-cut silvicultural system in the tropics is bound to bring some challenges for implementation of RIL. In contrast, this study demonstrates that use of RIL-C practices in more traditional selective logging for shade tolerant species combined with multiple-felling gaps to regenerate shade-intolerant species while maintaining logging intensities $< 10\,m^3\ ha^{-1}$ could reduce carbon emissions, increase production, and conserve biodiversity in large portion of the Yucatan Peninsula's forests.

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Reduced-impact logging for climate change mitigation (RIL-C) can halve selective logging emissions from tropical forests

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ABSTRACT

Selective logging causes at least half of the emissions from tropical forest degradation. Reduced-impact logging for climate (RIL-C) is proposed as a way to maintain timber production while minimizing forest damage. Here we synthesize data from 61 coordinated field-based surveys of logging impacts in seven countries across the tropics. We estimate that tropical selective logging emitted 834 Tg CO_2 in 2015, 6% of total tropical greenhouse gas emissions. Felling, hauling, and skidding caused 59%, 31%, and 10% of these emissions, respectively. We suggest that RIL-C incentive programs consider a feasible target carbon impact factor of 2.3 Mg emitted per Mg of timber extracted. Operational modifications are needed to achieve this target, such as reduced wood waste, narrower haul roads, and lower impact skidding equipment. Full implementation would reduce logging emissions by 44% (366 Tg CO_2 year^{-1}) and deliver 4% of the nationally determined contributions to the Paris Climate Agreement from tropical countries, while maintaining timber supplies.

1. Introduction

Tropical forest degradation (carbon losses from forests that remain forests) is responsible for much of contemporary (69%) and historic (27%) carbon emissions from tropical ecosystems (Baccini et al., 2017; Erb et al., 2017). Selective logging, which occurs in at least 20% of the world's tropical forests, is estimated to account for at least half of these anthropogenic forest degradation emissions (Blaser et al., 2011; Pearson et al., 2017). The need to reduce the deleterious environmental impacts of logging is widely recognized, but uncontrolled selective logging by untrained crews remains the major cause of tropical forest degradation and associated carbon emissions (Asner et al., 2005; Pearson et al., 2017).

Improved natural forest management represents a potentially large natural climate solution to global climate change, but this mitigation opportunity is highly uncertain (Griscom et al., 2017). Reduced-impact logging (RIL)—a set of improved timber harvesting guidelines for selectively logged natural forests—is of particular interest because of its relative low costs and numerous co-benefits. The carbon benefits of RIL have been studied at numerous sites across the tropics (e.g., Feldpausch et al., 2005; Medjibe et al., 2011; Pearson et al., 2014) as are the benefits to biodiversity (Bicknell et al., 2014). However, to our knowledge only one study (Putz et al., 2008b) estimated the pan-tropical climate mitigation potential of RIL, but it was based on field data from only two sites (Keller et al., 2004; Pinard and Putz, 1996).

The term RIL, which refers to sets of well-established timber harvesting practices (e.g., Conway, 1976), was first applied to an improved forest management project in Malaysia (Putz and Pinard, 1993). Since then various versions of RIL were codified internationally (Dykstra and Heinrich, 1996) and in various countries around the tropics (e.g., Pinard et al., 1995; Tropical Forest Foundation Indonesia, 2015). Here we use RIL-*C* to refer to a subset of recommended RIL practices that are explicitly promoted to reduce carbon emissions, an emphasis that reflects concerns about climate change and forest degradation as well as opportunities to benefit from reductions in carbon emissions, e.g., REDD+, voluntary carbon markets, Nationally Determined Contributions to the UN Paris Climate Agreement (NDCs, United Nations Framework Convention on Climate Change, 2015), and corporate

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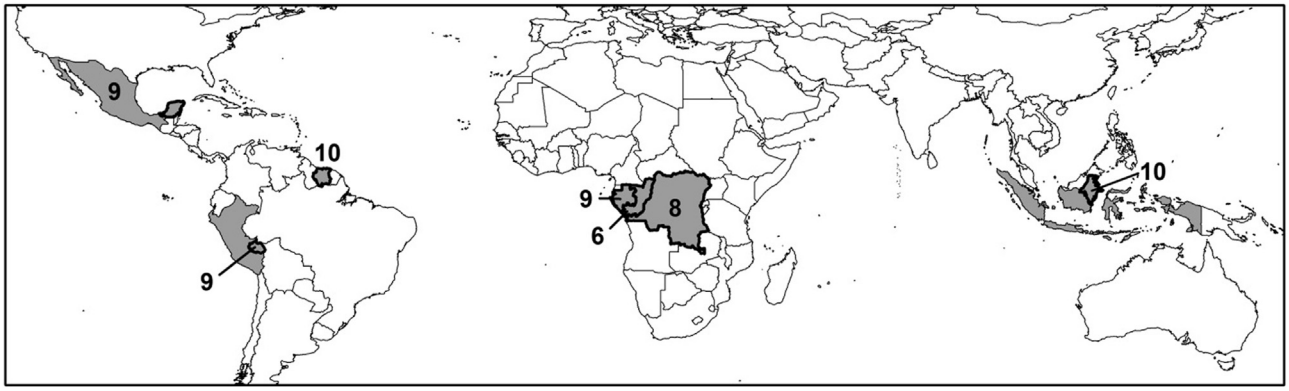

Fig. 1. Location of and number of sample blocks included in this study. Sample regions are outlined in bold, from west to east: Yucatan Peninsula, Mexico (YucP), Madre de Dios, Peru (MdD); Suriname; Gabon; Republic of Congo (RoC); Democratic Republic of Congo (DRC); and East and North Kalimantan, Indonesia (EKal).

commitments [see Fig. 1 in Griscom et al. (2019), for a detailed review of related terminology].

RIL-C practices are defined by their capacities to deliver measurable climate change mitigation outcomes without reductions in timber yields. Examples of RIL-C practices include improved felling and bucking for greater wood utilization (thus reducing waste), directional felling to avoid collateral damage, skid trail planning, long-line winching, and narrower haul road construction. Many of these practices can be implemented at low cost without dramatic changes to existing operational systems (Holmes et al., 2002; Indrajaya et al., 2016). Despite these opportunities, adoption of RIL and RIL-C practices remains low, partly because they lack robust, efficient emission reduction verification systems and appropriate rewards (Putz et al., 2012). Unlike deforestation, selective logging is notoriously difficult to monitor with available satellite imagery (Frolking et al., 2009; Read, 2003; Réjou-Méchain et al., 2015; Weishampel et al., 2012), so one challenge is to provide cost-effective, consistent, and reliable field-based protocols to measure those emissions. To provide such a protocol, the Nature Conservancy worked with partners to develop and validate a third-party Verra (formerly VCS) methodology for verification of RIL-C practices in the tropics (The Nature Conservancy and TerraCarbon LLC, 2016a), with a customized module for measuring RIL-C in East and North Kalimantan, Indonesia (The Nature Conservancy and TerraCarbon LLC, 2016b). The RIL-C methodology provides an outcome-based approach to measure logging emissions and thereby validate claims about the employment of RIL practices, using carbon as the performance metric. It facilitates implementation by applying easily measured field-based carbon metrics ("impact parameters") that allow independent monitoring entities to audit performance.

In this issue of Forest Ecology and Management, we compile field data from 61 sample blocks in 56 tropical forest management enterprises (FMEs) in seven countries on three continents to set logging emission baselines that comply with the RIL-C Verra methodology. We then estimate the opportunity for RIL-C practices to reduce emissions below these baselines. This paper synthesizes results at the pantropical scale, while the other papers in this issue focus on results from each FME and region with analyses designed to inform regional climate-effective logging policies. Here we (1) calculate a historical logging emissions baseline for the tropics, (2) evaluate patterns across regions, (3) propose a new RIL-C pantropical best performance target, (4) estimate the pantropical maximum potential climate mitigation of RIL-C implementation, and (5) provide global insights into which RIL-C practices are likely to generate the largest emission reductions.

Carbon benefits of RIL derive from both increased logging efficiency and respect for rules related to riparian buffer zones, slope restrictions, and other set-asides within harvest blocks. Here we focus on RIL practices that maintain timber yields and thereby avoid risks of leakage (i.e., displacement of logging to outside the area of study. RIL-C efficiency is expressed as an emissions factor, either in Mg C per ha harvest block, per m^3 extracted timber, or per Mg of extracted timber. The latter, which we refer to as the carbon impact factor (CIF) when excluding emissions for the extracted timber itself, corrects for differences in wood density and carbon content among harvested timbers. When combined with activity data on the spatial extent of logging and timber volumes harvested, these emissions factors can be used to estimate logging emissions and emission-reduction benefits at scales from harvest blocks to FMEs, regions, countries, and the tropics.

2. Methods

2.1. Study sites

This paper compiles data from coordinated field campaigns in seven national or sub-national regions described in detail in other papers in this special issue: Democratic Republic of Congo (DRC), Gabon, and Republic of Congo (RoC; Umunay et al., 2019); East and North Kalimantan Provinces, Indonesia (EKal; Griscom et al., 2019); Madre de Dios Department, Peru (MdD; Goodman et al., 2019)); Suriname (Zalman et al., 2019); and three Mexican states on the Yucatan Peninsula (YucP; Ellis et al., 2019). In each region (Fig. 1), field data were collected in 6–10 (mean = 8.7) harvest blocks within active, legally permitted FMEs. In five cases (once in Ekal, once in Suriname, and three times in MdD) two blocks were sampled in the same FME but were harvested at different times by different crews. Sample blocks, which represent spatially distinct areas of active harvesting, ranged 22–1060 ha and were often coincident with planning units used by forest managers (i.e., forest management units, "petaks," "kapvaks," or "sub-blocks"). Relevant regional harvesting statistics are presented in Table 1.

Sample blocks were selected with a stratified random procedure to ensure a representative sample of FMEs based on factors such as their size, soil type, elevation, carbon density, certification status, and worker training in RIL practices. If a randomly selected sample block was inactive or inaccessible, it was replaced by a new randomly selected sample block from the same stratum. At least two Forest Stewardship Council (FSC)-certified FMEs were selected in each region, except in DRC, where there were none. Certified and uncertified FMEs share many characteristics, but they were not fully matched, so we did not account for likely positive selection bias, and our comparisons should therefore be considered naïve (Romero et al., 2017).

2.2. Field data collection

We adapted field methods from two previous studies (Griscom et al., 2014; Pearson et al., 2014). We mapped all skid trails in each harvest block using wide-area augmentation system-enabled Garmin GPS

Table 1
Key characteristics of samples from harvest blocks. All carbon (C) values represent above- and below-ground biomass.

Region	No. sample blocks	Mean area sampled (ha)	Mapped skid trail length (km)	Mapped haul road length (km)	No. felled trees measured	No. felled trees counted	Mean log length (m)	Mean felled tree DBH (cm)	Mean felled tree C (Mg)	Mean pre-harvest forest C density (Mg ha^{-1})	Mean harvest intensity (m^3 ha^{-1})
DRC	8	77.9	29.3	5.1	102	317	18.5	117.9	33.8	202.1	8.0
Gabon	9	100.9	28.9	18.4	135	498	20.9	107.2	27.7	202.1	10.5
EKal	10	117.1	35.0	73.6	132	1173	25.3	87.9	50.6	233.3	36.5
YucP	9	320.0	118.4	17.3	460	2969	7.8	62.6	14.2	76.6	2.8
MdD	9	68.5	35.7	67.4	151	262	17.3	101.8	18.6	NA	5.2
RoC	6	51.7	18.1	4.9	75	236	18.8	117.0	28.0	202.1	17.4
Suriname	10	57.8	68.0	125.4	255	1167	17.9	74.1	21.2	236.9	11.0
All	61	116.2	333.5	312.1	1310	6622	18.2	93.6	27.9	173.8	13.3

receivers. We also counted all felled trees extracted from these skid trail networks except in EKal and MdD, where skid trails were subsampled. In EKal, we scaled our subsamples based on LiDAR-mapped skid trail densities over 5620 ha in six of the nine sample blocks (Ellis et al., 2016); for the remaining three blocks, we used the LiDAR-based average. In MdD, we scaled our subsamples using the ratio between field-measured extracted timber volumes and reported volumes for the entire FME.

To estimate emissions from tree felling, including those from the portions of felled tree left in the forest (hereafter, felled tree remainder), we visited an average subsample of 21 recently felled trees in each sample block. At each felled tree, we recorded the location ± 5 m, tree species, stump height, diameter at breast height (DBH) when possible, total tree height (except for YucP and EKal), and diameter and length of all present and absent log sections up to the first major branch of the felled tree, noting any hollows. We inferred diameters and lengths of extracted logs from the distances between and diameters of remaining sections. To ensure accuracy of inferred log extraction lengths, any felled trees remainders that displayed evidence of sliding down hill after felling (or being moved during yarding) were dropped from the sample. To avoid bias toward sampling tree gaps with multiple felled trees, we selected felled trees from a systematic subsample of felling gaps, regardless of whether they were caused by the felling of single or multiple trees. Measurements were taken for all trees within selected felling gaps. In each felling gap, we recorded DBH and damage class of all trees ≥10 cm DBH that were damaged as a result of tree felling (see Table S1 for damage classes).

To assess damage from skidding (transport of timber from felling site to the roadside), we established an average of 16 plots 10 m long, with width defined by width of the skidding damage, distributed evenly throughout the mapped skid trail networks. As for felling damage, in each skid trail plot, we recorded DBH and damage class of all trees ≥10 cm DBH. In EKal, where bulldozers (i.e., crawler tractors) were used for skidding and the soil surface is often bladed off, trees < 20 cm DBH were often buried by debris. To account for this process, in EKal we measured the density of all trees 10–20 cm DBH in 5 × 10 m plots located 5 m from the edge of each skid trail plot, as described in Griscom et al. (2014). Given that skidding emissions from trees 10–20 cm DBH was < 1% of skidding emissions in Ekal sample blocks, only trees > 20 cm DBH were measured in other geographies where bulldozers were used (Gabon, MdD, and Suriname).

To estimate the area of forest clearing from newly constructed haul roads, we mapped an average of 5 km of haul roads in and adjacent to the sampled blocks using a Garmin® GPS. We measured widths of the active road surface and of the total haul road corridor between the nearest standing tree boles at an average of 18 points along these roads. Along these mapped roads, we also measured the area of an average sample of 7 log yards using field-based measurements of length, width, and shape or from the GPS-based area calculated from tracing the yard perimeter.

To estimate the carbon density of forests cleared during road and log yard construction, we established an average of 15 biomass plots in pre-harvest blocks adjacent to the sampled blocks. We used a nested variable-radius sampling "Big BAF" system (Griscom et al., 2014; Marshall et al., 2004), except in MdD and YucP. Following this methodology, for trees > 10 cm DBH selected by a larger basal area factor angle wedge gauge, we recorded DBH, species, (and total tree height in Gabon, DRC and RoC). These trees were used to calculate a biomass-to-basal area factor. We then tallied all trees selected with a smaller BAF angle gauge to calculate basal area. Small and large BAFs were calibrated to conditions in each region, as described by Marshall et al. (2004). In YucP, biomass-to-basal area factors were calculated from available inventory data (CONAFOR, 2012). In MdD, a regional average 62.3 Mg C ha^{-1} was used for trees with DBH ≤ 40 cm from Goodman et al. (2012) because trees > 40 cm DBH were avoided during road construction, as reported by forest managers and observed in the field. Soil carbon emissions were not assessed in this study.

Our field methods differ from those used by Pearson et al. (2014) in how we mapped felling gaps, skid trails, and haul roads. Instead of using remote-sensing imagery and pre-harvest maps, we relied solely on field-based GPS maps of skid trails and haul roads (and LiDAR in EKal), as described above. Teams of 2–4 people typically completed a sample block in 3–4 days.

2.3. Data processing

We calculated baseline emissions using a consistent set of equations and variables (see Supplementary Equations). Similar to previous studies (Griscom et al., 2014; Pearson et al., 2014), we use the "gain-loss method" equation 2.4 from Intergovernmental Panel on Climate Change (IPCC) National Guidelines (Aalde et al., 2006). This equation is recommended by IPCC in place of the "stock-difference method" when carbon fluxes are a small proportion of stocks, as is the case for selective logging emissions from tropical forests. We analyzed, aggregated, and summarized the data at the scales of plots, sample blocks, FMEs, regions, and all tropical countries where there is commercial selective logging. We categorized data into six emissions sources: (1) roundwood *timber* extracted from felled trees (RW_F); (2) the roots, crowns and branches of felled trees that remain on site (*felled tree remainder*); (3) *felling collateral damage* from trees killed by felling operations; (4) collateral damage from log transport (i.e., yarding) from felling sites to log yards (*skidding*); (5) forest cleared during haul *road* construction; and, (6) forest cleared for *log yard* construction. Sources 1–3 were associated with *felling*, source 4 with *skidding*, and sources 5 and 6 with *hauling*. Note that these categories differ slightly from those used by Pearson et al., (2014), but can be easily cross-walked: Extracted Log Emissions (ELE = timber), Logging Damage Factor (LDF = felling collateral damage + felled tree remainder), and Logging Infrastructure Factor (LIF = skidding + hauling). All ranges reported in this paper are expressed as ± 95% confidence limits.

We take a committed emissions approach to accounting for emissions from all pools, including extracted timber, following the IPCC Tier 1 accounting assumption "… that all carbon biomass harvested is oxidized in the removal [harvest] year" (Pingoud et al., 2006). In this way, potential mitigation from improved milling efficiency, increased carbon storage in durable wood products, permanent wood storage in landfills, energy generation from wood waste, and the substitution of wood for concrete, steel, or aluminum are not included in our calculations of the mitigation potential of RIL-C.

To calculate tree biomass from DBH, wood density, and height (and crown diameter when available), we applied the best available allometric equation for each region. We used Chave et al. (2005) model II.3 Moist for EKal and Suriname and II.3 Dry for YucP. For MdD we used the Goodman et al. (2014) model I.1CR for felled trees and model II.1 for all others. In Gabon, RoC, and DRC we used Fayolle et al. (2018) regional model 12. We estimated below-ground root biomass with root-to-shoot ratios for each region's forest type (Mokany et al., 2006). To calculate volume of extracted timber from felling, we used a Smalian frustum formula from field measurements of the distance between remaining log sections and diameters of remaining sections at either end. We converted volume to carbon using wood density (Chave et al., 2009; Zanne et al., 2009) and 0.47 carbon fraction (McGroddy et al., 2004). Hollow volumes were subtracted from log section volumes also using Smalian's frustum formula. When hollows were observed at only one end of a section, we assumed a hollow volume equal to a cone with height equal to half the log's length. Overall, hollows represented 0.5% of total felled tree biomass.

We calculated collateral damage emissions from skidding and felling using mortality rates from damage scenarios described in Table S1, adapted from Goodman et al. (2019) using pantropical average for proportion of AGB in the tree crown.

We estimated the timber extracted during haul road construction that was not captured in our calculations from felling sites by applying harvest intensities from our field data to the total haul road area. For this purpose, we first mapped the "area accessed" (Griscom et al., 2014) based on the 95th percentile of GPS recorded skid trail-to-stump minimum Euclidian distances for each region (5571 stumps, 333 km skid trails total). We then delineated skid trail "buffers" in GIS using these distances. We applied the field-measured harvest volumes per area accessed as a proxy for available timber in the area of newly constructed haul roads. This additional hauling timber (RW_H) was added to the felling timber (RW_F) to calculate the total timber harvested used as the denominator of Mg C m^{-3} and CIF (Mg Mg^{-1}) emissions estimates presented below.

To estimate the area of newly cleared haul roads, we assigned the mean haul road density (m ha^{-1} in sample blocks) for the entire region to all sample blocks in that region before multiplying by the mean sample block-specific haul road widths. We chose more generalized road length densities in place of sample block specific ones because sample road lengths for any given FME were rarely large enough to capture the variability in road density confidently. Furthermore, practitioner feedback indicated that reducing haul road length is expensive and often infeasible (Griscom et al., 2019), so it would not represent a viable RIL-C practice.

To estimate the area of previously constructed and re-used haul roads, we assumed trees would regrow in the haul road clearing corridor but not on the active road surface where soil conditions and continued road use greatly inhibit regrowth. To calculate the carbon density of the vegetation in these roadside strips of regeneration, we multiplied the area times the average harvest cycle by the tropical secondary forest carbon sequestration rate estimated as 2.73 Mg ha^{-1} yr^{-1} (Bonner et al., 2013).

Carbon emissions from soil and litter, fossil fuels used during logging, and activities outside the sample blocks such as base camp construction and operation were beyond the scope of this study and were not considered.

We express C emissions in three ways: (1) emissions per area (Mg ha^{-1}) by dividing all sample block source emissions by the total area of the sample block, acknowledging that the area of the sample block accessed for extraction is often much smaller than the permitted harvest block (Ellis et al., 2016); (2) emissions per volume of timber extracted (Mg m^{-3}) by dividing all sample block emissions by the extracted volume; and, (3) the carbon impact factor (CIF) which divides emissions sources 2–6 by the extracted volume (source 1) expressed in Mg C, referred to as "mean carbon export ratio" by Feldpausch et al. (2005). CIF adjusts for variation in wood density among study areas and provides an intuitive sense of the efficiency of logging operations. As the ratio of biomass C damaged to biomass C utilized for timber, lower CIF values correspond to more efficient operations. In this paper, we use CIF when comparing regions but use Mg m^{-3} when scaling emissions factors to country and pantropical scales because harvest volume data are more available. Emissions reported in Mg ha^{-1} vary with harvest intensities, so we use this metric only for comparisons with other published data.

We used mixed effects models to evaluate effects of various factors on logging emissions, specifying regions as the random intercepts. We selected best fit models to explore how CIF (total and by source) is effected by FSC certification, and the following environmental variables at sample block and regional scales: average terrain slope (percent, Jarvis et al., 2008), average pre-harvest carbon density (Mg ha^{-1}), average wood density of felled trees (g cm^{-3}), average height of felled trees (m), average heights of measured trees in the pre- harvest blocks (m), and average annual precipitation (mm, Fick and Hijmans, 2017). We also used mixed effects models to evaluate the relationship between harvest intensity and logging emissions (total CIF), again controlling for variation among regions.

To test for differences in emissions due to the use of different skidding equipment, we fitted a one-way ANOVA model followed by Tukey HSD post-hoc tests. We tested for correlations (Pearson's) between haul road width and road emissions. All statistical analyses were done using the R packages (R Development Core Team, 2014), lmerTest (Kuznetsova et al., 2017), and MuMIn (Bartoń, 2018).

2.4. Emissions reductions and RIL-C best practices

None of the 61 sample blocks implemented the complete suite of RIL-C practices. Therefore, to estimate potential RIL-C emissions reductions for each region, we selected the best performance (in terms of CIF) from all sample blocks for each emissions source. In this way, theoretical best-case scenarios were compiled from emission source data for each region; this is consistent with "level 1" RIL-C implementation as described by Griscom et al. (2019). We estimated a RIL-C pantropical best performance as the average of "level 1" best performance compilations from each region.

To investigate the factors that might influence emissions reductions, we used linear models to analyze CIF correlations with wood waste from felled trees and inefficient bucking, skidding equipment, skid trail density, haul road width, and worker training. Wood waste was defined as any non-hollow, undamaged wood from the felled tree left at the felling site that was between 50 cm above the ground and the first large branch, separated into felled tree wood waste from felled trees with no timber extracted ($RW_n = 0$), and bucking wood waste from felled trees with some timber extracted ($RW_n > 0$). Winching distances were defined by the distance from each stump to the nearest skid trail, based on 5571 stumps and 333 km of tracked skid trails.

We did not explicitly investigate the effects of set-asides (e.g. riparian buffers, steep slopes), pre-felling liana cutting, haul road planning, marking future crop trees, or road and skid trail construction best management practices (e.g. water bars and culverts). Emissions reductions from these activities would be additional to those identified here, but we expect them to be minimal and/or challenging to monitor because of leakage concerns, data scarcity, circumstances out of

manager control, low variability in our dataset, or with effects on carbon pools not covered in this study, respectively.

2.5. Pantropical logging emissions

We used emissions factors calculated in this study together with harvest volume data from the Food and Agriculture Organization of the United Nations Forest Resources Assessment (FRA; FAO, 2016), to estimate pantropical baseline logging emissions and potential emissions reductions. First, we calculated the average C emission factors (Mg m^{-3}) for the seven sampled regions and assumed our samples were representative of the logging conditions for that entire country. Then, we estimated the volumes of commercial timber harvested from natural forest in each country using the extracted timber volumes data from FRA for 2015. The FRA collects industrial timber production statistics from national governments, but these data do not distinguish between timber from natural and plantation forests. To remove plantation-sourced timber from our statistics, we relied on a unique dataset that estimates global plantation production volume (Jürgensen et al., 2014) to estimate the plantation productivity (m^3 ha^{-1}) for each tropical country. We divided 2010 plantation areas (FAO, 2010) by the output volume (m^3 for year 2012; Jürgensen et al., 2014) and used regional averages for countries with missing data. We then multiplied the estimated plantation productivity by the reported 2015 plantation area (FAO, 2016) to derive the total 2015 timber production volume from plantations for each country. Lastly, we subtracted this number from total 2015 timber production volume to obtain country-level natural forest timber production in 2015. These natural forest timber volumes served as the activity data that, when multiplied by our country emissions factors, provide country-wide baseline selective logging estimates for the seven countries sampled in this study.

To extrapolate this sample to the other 77 tropical countries with FRA-reported rates of timber extracted from natural forests by selective logging, we conducted an expert consultation process to cluster countries with similar logging conditions, extraction intensities, and harvesting equipment, with each country cluster assigned the parameters from one of the seven sampled countries. We then assigned all countries in a cluster the emissions factor from their representative sample country, multiplied by natural forest harvest volumes, and thereby obtained country-level estimates of baseline logging emissions. Summing these baseline estimates across the 84 timber-producing tropical countries provides an estimate of pantropical carbon emissions from selective logging of natural forests. To estimate maximum potential emissions reductions from RIL-C best practices, we subtracted the pantropical best RIL-C performance from each country's baseline logging emissions and summed the differences. To determine RIL-C's contribution to NDCs, we compared national RIL-C potential emissions reductions against the NDCs reported by Baruch-Mordo et al. (2019).

3. Results

3.1. Baseline emissions

Mean CIF baseline carbon emissions for all 61 sample blocks was 5.7 ± 1.0 Mg Mg^{-1} (1.8 ± 0.2 Mg m^{-3} and 20.8 ± 4.6 Mg ha^{-1}). Variation in CIF was high among (± 1.9 Mg Mg^{-1}) and within geographies (± 0.4 to ± 4.9 Mg Mg^{-1}), and across emissions sources (Fig. S3). Gabon's CIF (10.7) was almost four times higher than MdD's (2.8). Congo Basin countries displayed the highest emissions from hauling and the greatest variation in total emissions. Emissions in MdD and YucP were the lowest and varied the least, with both dominated by felled tree remainder emissions (Fig. 2). On average, haul roads and felled tree remainders are the largest emission sources (35% and 33%), followed by felling collateral damage and skidding (17% and 10%). Log yards are the smallest source of emissions (5%). The percentage of pre-logging carbon stocks emitted as a result of logging was fairly consistent, on average 11.2 ± 1.7%, with EKal highest (21.7%), YucP the lowest (4.8%). The CIF of the best performing FME (Guadalajara, in YucP) was 2.3 Mg Mg^{-1}.

Considering regions as random effects in mixed effects models, we found different results for different fixed effects. When accounting for within-region variation, FSC-certified FMEs did not differ from uncertified FMEs ($F_{(1,56)} = 0.07$, $p = 0.78$, Fig. S4). Considering environmental variables out of manager control (terrain slope, carbon density, wood density, tree height, and precipitation) only slope had a significant effect on CIF emissions overall or by source ($p = 0.005$), and together explained 52% of CIF variation. Across all sample blocks, the log of harvest intensity significantly decreased with CIF and explained 28% of the variation in CIF emissions when controlling for the random effect of regions ($F_{1,58.5} = 31.6$, $p < 0.0001$). Evaluation of the effect of harvest intensity on CIF by region revealed significant effects only in Gabon, Ekal, YucP, and Suriname. In these regions, log function asymptotes ranged from 1.7 in YucP to 3.5 CIF in Suriname (mean = 2.6 ± 0.7, Fig. 3). Harvest intensities were by far the lowest in the Yucatan Peninsula (mean 2.8 compared to 13.3 m^3 ha^{-1} mean across all regions).

3.2. Emissions reductions and RIL-C best practices

The average RIL-C pantropical best performance (2.3 Mg Mg^{-1}) was 60% lower than the mean pantropical baseline, with lower intra-region variation than for baseline emissions (± 0.4 Mg Mg^{-1}). Subtracting this global mean best performance from baseline values for each region provides estimated potential emissions reductions by region, which ranged from 8.4 Mg Mg^{-1} in Gabon (79%) to 0.5 Mg Mg^{-1} (18%) in MdD (Fig. 2).

Most of RIL-C's emission reduction benefits are derived from RIL-C practices that minimize the hauling footprint, reduce wood waste, and improve skidding (Table 2). Increases in wood waste explain 96% of the linear model's variation in felled tree remainder emissions (Fig. 4). Most wood waste in our sample (79%) is a result of poor log recovery (felling hollow trees, failing to extract all felled trees); the remainder (21%) is from poor bucking practices (e.g., high stumps, too much crown wood). Skidding emissions were significantly different for concessions using different equipment ($F_{4,56} = 2.81$, $p < 0.05$, Fig. S5). FMEs using heavy equipment such as articulated skidders, bulldozers and excavators emitted eight times more carbon per km of skid trail than those that used small footprint skidding equipment such as modified farm tractors and forestry skidders (known in Mexico as "tree farmers") (Table 3). Skidding emissions showed a weak but significant ($F_{1,59} = 4.28$, $p = 0.04$, adjusted $R^2 = 0.05$) relationship with skid trail density (m [ha sample block]$^{-1}$; Fig. S6).

Haul road width explains 19% of the linear model's variation in road emissions CIF ($F_{1,5} = 15.0$, $p = 0.0003$), Fig. 5). We found no correlation between the lengths of skid trails and haul roads.

We found no evidence that training in directional felling reduced felling collateral damage ($F_{1,47} = 0.40$, $p = 0.53$). At the sample block scale, neither mean tree biomass (Mg $tree^{-1}$) nor mean DBH was correlated with felling collateral damage emissions. There was also no relationship between winching distances (7.0 ± 1.6 m) and skidding emissions.

3.3. Pantropical logging emissions and emissions reductions

Using baseline emission factors from Fig. 2 together with reported harvest volumes sourced from the FRA data, we estimate that the baseline logging emissions for 83 timber-producing tropical countries is 834 Tg CO_2 $year^{-1}$, which exceeds Mexico's total annual greenhouse gas emissions (World Resources Institute, 2017). Using estimated potential emissions reductions from Fig. 2 (shown as negative CIF), the total expected emissions reductions sum to 366 Tg CO_2 $year^{-1}$, 44% of baseline emissions (see Fig. 6). These potential emissions reductions are

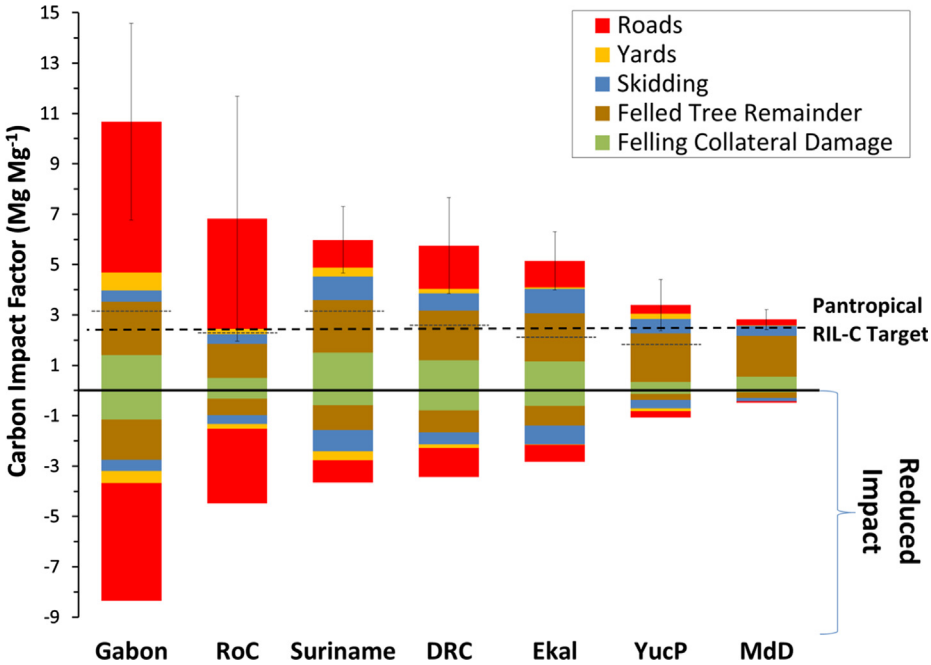

Fig. 2. Measured baseline carbon emissions (above *x*-axis) and carbon reduction potential (below *x*-axis) for the seven regions surveyed in this study. Emissions values are expressed as the carbon impact factor, the ratio of carbon lost from logging damage to carbon extracted as timber (Mg Mg^{-1}). Error bars show the 95% confidence intervals for total baseline emissions. Potential reduced-impact emissions reductions were calculated as the difference between the regional baseline and the average RIL-C pantropical best performance target (thick dashed line, 2.3 Mg Mg^{-1}). Thin dotted lines correspond to regional best case scenarios, calculated as the best performance from all sample blocks for each emissions source. The pantropical target is the average of all regional best case scenarios.

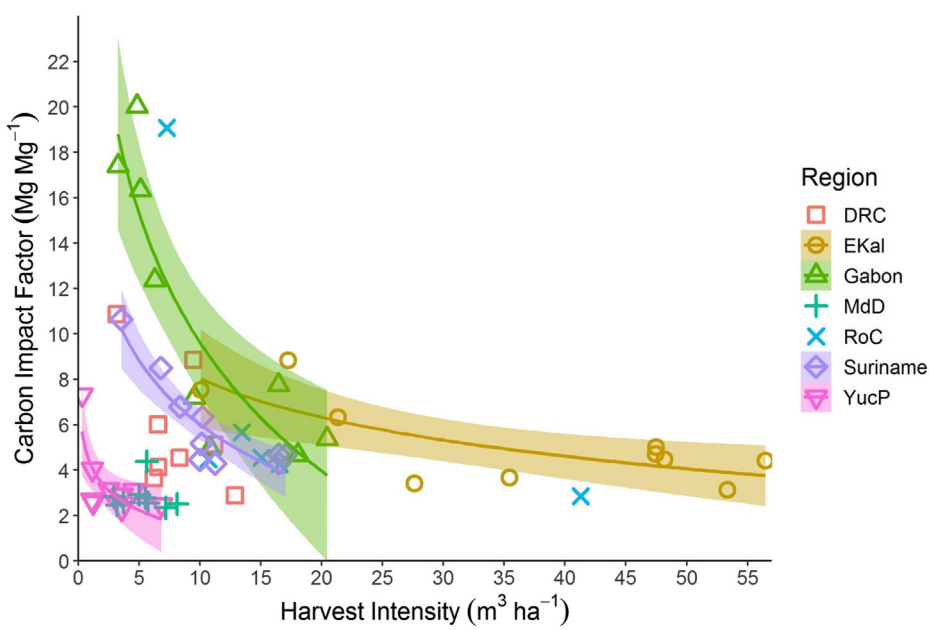

Fig. 3. Effect of harvest intensity on logging emissions by region. Emission values are expressed as carbon impact factors [carbon lost from logging damage per carbon extracted as timber (Mg Mg^{-1})]. We only show fitted lines for regions where the effect of harvest intensity is significant.

more than Mexico's entire unconditional 2010 NDC. Logging emissions represent 6% of these 83 countries' total greenhouse gas emissions (World Resources Institute, 2017). For the 58 tropical timber-producing countries who have pledged quantifiable NDCs to the Paris Agreement, RIL-C can contribute 4% to their aggregate emissions reductions targets. Nine of these 58 countries show potential RIL-C mitigation contributing to more than half of their NDC commitment: Uganda, Gabon, Côte d'Ivoire, Solomon Islands, Equatorial Guinea, Republic of Congo, Guinea, Central African Republic, and Liberia.

4. Discussion

4.1. Baseline emissions

High inter- and intra-region variation in baseline CIF indicates large opportunities for operational improvements to reduce emissions from selective logging in the tropics (Fig. S3, Fig. 2). The lack of evidence for association between logging emissions and FSC certification may reflect that FSC standards were designed to ensure sustainability and promote environmental responsibility, not to reduce carbon emissions. Furthermore, FSC's principles, criteria, and indicators are not specific enough to affect operational changes that generate measurable ecosystem service outcomes. Fully aware of this challenge, FSC recently released an Ecosystem Services Procedure for FME audits that recommends the

Table 2
Estimated emissions reductions from RIL-C practices. Note that directional felling emissions reductions (*) are theoretical, as we have no evidence that directional felling training led to lower felling collateral damage.

RIL-C Practice	RIL-C savings (CIF, Mg Mg^{-1})	% of total savings	Emissions source
Minimize hauling footprint			Roads and yards
Build narrower haul roads	1.38	40%	
Clear smaller log yards	0.17	5%	
Reduce wood waste			Felled tree remainder
Recover all merchantable wood	0.47	14%	
Do not fell hollow trees	0.20	6%	
Buck felled logs efficiently	0.18	5%	
Improve felling			Felling collateral damage
Use directional felling	0.59*	17%	
Improve skidding			Skidding
Use low-impact skidding equipment	0.33	10%	
Plan out skidding routes	0.10	3%	
Total	3.42		

RIL-C methodology to demonstrate carbon impact (Forest Stewardship Council, 2018). This standard is available to existing FSC-certificate holders who want to document their carbon-related performance.

The paucity of detected relationships between logging emissions and environmental variables is surprising, but corresponds with the results of previous research (e.g., Griscom et al., 2014). More studies of these relationships between logging emissions and slope, carbon density, timber stocking, tree height, other biophysical variables are needed, but the paucity of evidence to date suggests that operational decisions exert an outsized influence on logging impacts. For example, we were surprised to find that mean annual precipitation appeared unrelated to road width, given the reported need to increase road corridors to facilitate "daylighting" in wetter climates.

The negative effect of harvest intensity on CIF is strong, but heterogeneous, and warrants further study. It is unclear why some regions show a strong effect of harvest intensity (Ekal, Gabon, Suriname, and YucP), but others do not (DRC, MdD, RoC). It also appears that the negative effect of harvest intensity saturates at high levels (above ~25 m^3 ha^{-1}), and converges at ~2.6 CIF, perhaps because forest managers can more efficiently utilize logging infrastructure and extract more wood from targeted portions of sample blocks with high stocking, but after they exhaust these areas of timber, must build new infrastructure into more marginal territory (Fig. 3). Only in the Yucatan Peninsula does it appear that significant increases in harvest intensity could drive emission levels below the target CIF of 2.3 Mg Mg^{-1} (YucP asymptote = 1.7 Mg Mg^{-1}).

4.2. Emissions reductions and RIL-C best practices

Our average RIL-C pantropical best performance CIF is 2.3 Mg Mg^{-1} (0.63 Mg m^{-3} of C). This target is similar to the only other two published estimates of field-measured RIL emissions we could find in the literature: 2.4 Mg Mg^{-1} from Para, Brazil (Feldpausch et al., 2005; Keller et al., 2004) and 0.62 Mg C m^{-3} from Sabah, Malaysia (Pinard and Putz, 1996). More research is needed to evaluate RIL performance in other regions, but given this alignment across 9 different counties (±14% uncertainty when including Brazil and Malaysia), we suggest that RIL incentive programs consider a pantropical target CIF of 2.3 Mg Mg^{-1}. This target balances practicality with ambition and provides a measurable benchmark to evaluate progress.

Given that carbon emissions from selective logging decreases with harvest intensity, as discussed above, target CIF values might also vary with intensity. YucP is the only region where intensification appears able to drive CIF values below our theoretical best performance, but even here, values below 2.3 are not directly observed. Therefore, for simplicity, practicality, and to motivate and guide RIL-C implementation, we feel confident that a CIF of 2.3 Mg Mg^{-1} serves as reasonable and achievable target. However, when RIL-C monitoring systems are designed, it is important to control for the potentially perverse incentive of logging intensification. Therefore, we recommend RIL-C performance methodologies include safeguards that limit increases in timber extraction and tie performance to improved practices known to limit impacts, as specified in the RIL-C Verra Methodology (The Nature Conservancy and TerraCarbon LLC, 2016a).

To realize RIL-C benefits, forest managers need to know how to

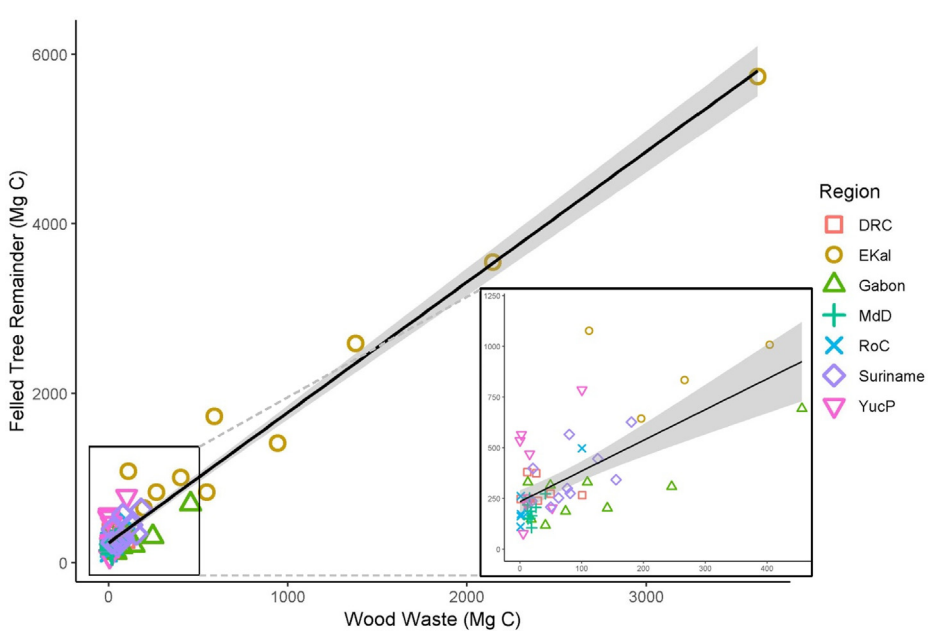

Fig. 4. Relationship between wood waste and felled tree remainder for all sample blocks by region, with inset displaying relationship in sample blocks with little wood waste.

Table 3
Skidding emissions differences based on skidding equipment (mean ± 95% confidence limits). Asterisks mark significant pairwise differences based on Tukey HSD tests. Skidding emissions are expressed as the carbon impact factor: the ratio of carbon lost from skidding to carbon extracted as timber (Mg Mg^{-1}).

	Skid trail width (m)	Skidding intensity (m m^{-3})	Skidding C emissions intensity (Mg km^{-1})	Skidding emissions CIF (Mg Mg^{-1})	Max tree biomass (Mg C)	Sample blocks (n)
Modified farm tractor	3.1 ± 0.4	51.9 ± 27.5	4.1 ± 3.4	0.16 ± 0.06	46.7	3
Forestry skidder	3.9 ± 0.2	106.6 ± 38.7	7.2 ± 1.3	0.79 ± 0.39	12.2	6
Articulated skidder	2.3 ± 0.8	26.8 ± 4.5	17.9 ± 5.2	0.51 ± 0.17*	53.1	25
Excavator	3.0 ± 2.6	47.8 ± 7.1	24.4 ± 8.2	1.21 ± 0.56*	25.5	6
tracked bulldozer	6.3 ± 1.6	15.9 ± 13.0	38.8 ± 13.6	0.69 ± 0.27	71.6	21
All	3.9 ± 0.8	34.2 ± 8.2	24.0 ± 5.9	0.65 ± 0.14	71.6	61

adapt their harvesting operations to optimize emissions reductions with minimal costs. More research is needed to provide causal links between best practices and emissions reductions, but this paper, together with the regional studies in this special issue, provide a starting point for identifying RIL-C best practices with high likelihood of reducing carbon emissions.

Improvements in road construction constitute the largest source of potential emissions reductions (1.38 Mg Mg^{-1}, 40%) but are also likely to be the most costly. Following our best-case scenario logic, subtracting minimum road widths from average road width per geography, road widths could be feasibly reduced by 1–12 m (18–54%). To compensate for the reduced direct sunlight on the often-wet road surfaces, loggers would need to improve their road engineering to increase drainage, and use more gravel on road surfaces to increase trafficability. The latter would entail substantial costs where hard rock is scarce, but improved roads might reduce hauling costs and increase the length of the time the roads are passable. Minimizing log yard area could reduce emissions an additional 0.17 Mg Mg^{-1} (5%); temporary storage of logs on roadsides is an option but might require better scheduling of overall harvest operations.

Reduced wood waste is a smaller but more cost-effective RIL-C best practice that contributes at least 25% (0.84 Mg Mg^{-1}) of the total potential emissions reductions need to reach the 2.3 CIF target. Activating this mitigation opportunity involves three potential interventions, all of which improve utilization through reducing wood waste from felled trees. First, simple planning and communication between fellers and skidder operators can ensure that all merchantable logs are recovered, potentially reducing the CIF by 0.47 Mg Mg^{-1}. Seventy percent of the unextracted felled tree waste in our sample had no hollows or evident damage. Improving log extraction will not only reduce emissions, but improve operational efficiency, since it increases volume extracted per unit machine time and labor.

Second, training and motivating tree fellers to avoid felling hollow trees could reduce much of the remaining 30% of felled tree wood waste (0.20 Mg Mg^{-1}). Many strategies exist for pre-felling evaluation of hollowness. For example, fellers may utilize a chainsaw plunge-cut to test for hollows at the tree base before initiating the felling process. If the detectably hollow trees are subsequently not felled, wood waste and carbon emissions will both be reduced, valuable wildlife habitat and forest structure will be maintained, and the personal risks to the tree fellers will be reduced (Conway, 1976). This is an obvious benefit for one of the most dangerous professions in the world, where every 10th logger in the tropics is likely to die from a work-related accident (Alli, 2008).

Third, improved bucking of felled trees can generate additional operational efficiency and emissions reductions. Twenty-one percent of wood waste in our field sample was generated from felled trees with some extracted timber ($RW_F > 0$). Of this, 77% showed no signs of heart-rot, hollows, or other defects, indicating that 0.18 Mg emissions could be avoided per Mg of timber harvested if fellers bucked non-hollow log sections up to the first large branch and down to 0.5 m from the ground. For trees with large buttresses, trimming buttresses to the bole before felling not only avoids wood waste, but improves accuracy and safety of felling, again providing additional operational benefits.

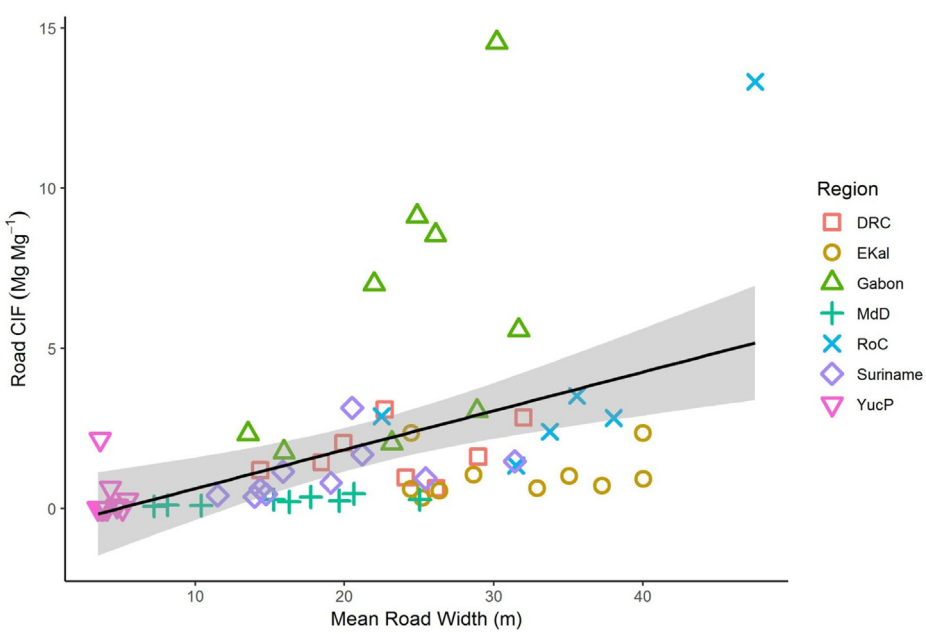

Fig. 5. Correlation between haul road width and road emissions. Road emissions expressed as carbon impact factor: carbon lost from road construction per carbon extracted as timber (Mg Mg^{-1}).

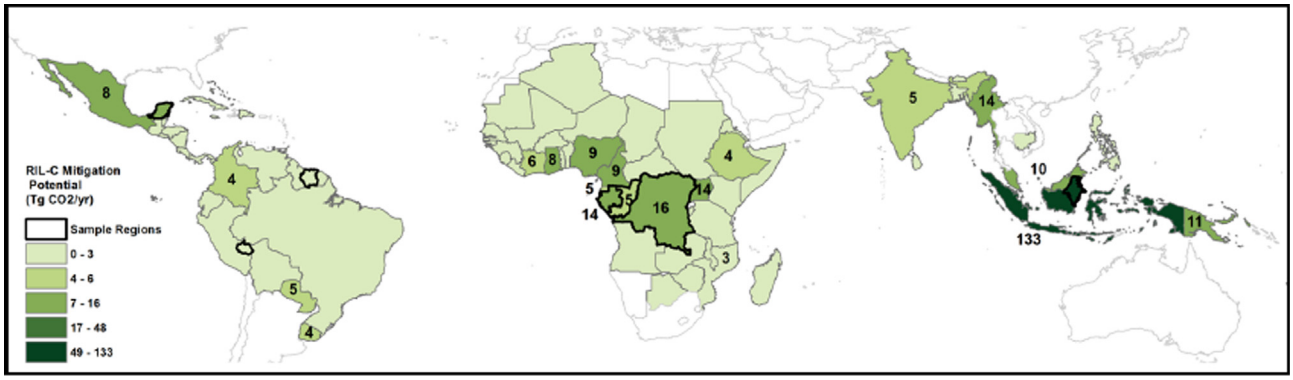

Fig. 6. Potential RIL-C emissions reductions by country. Sample regions are outlined in bold.

All three of these strategies for improved wood utilization could provide additional emission reductions to skidding and hauling, because they increase timber yields for similar infrastructure footprints, thereby reducing the CIF for skidding and hauling.

Felling collateral damage emissions are the third largest source of emissions reductions (0.59 Mg Mg^{-1}, 17%), but the pathway to implementation is less clear. We found no relationship between feller training and reduced felling collateral damage. Either feller training was insufficient, the trainees were not motivated to utilize their training to reduce emissions, or they were not the ones doing the felling (attrition of skilled loggers is common). It is also important to note that only one of 56 FMEs cut lianas at least 6 months prior to felling (the ejido Caobas in YucP; Ellis et al., 2019), as per RIL recommendations (Appanah and Putz, 1984; Pinard et al., 1995). While more research is needed to examine relationships between liana cutting and felling collateral damage, it is noteworthy that felling emissions in Caobas were very low. In addition to cutting lianas, felling emissions might be reduced by: (1) providing incentives to fellers to avoid damaging large trees; (2) marking future crop trees, as stipulated in many RIL standards; and (3) instituting programs to retain and reward tree fellers that show skill and experience in directional felling—these master fellers could earn higher wages by training less skilled staff and passing on their knowledge.

Improved skidding is the smallest potential source of emissions reductions (0.43 Mg Mg^{-1}, 13%). The resistance of loggers to changes in skidding machinery notwithstanding, if rubber-tired skidders replaced bulldozers and farm tractors replaced forestry skidders, skidding emissions could be reduced by 50% and generate an average emission reduction of 0.33 Mg Mg^{-1} (Table 3). The remaining 0.10 Mg Mg^{-1} could be achieved through improved skid trail planning and longer line winching, although the weak relationship between skid trail density and skidding emissions (Fig. S6) suggests that the impact would be relatively small. While we detected no relationship between skidding emissions and skid trail-to-stump distances, we believe this is because none of the FMEs in this study used long line winching technology (skid to stump distances were only 7.0 ± 1.6 m) such as the modified excavators that are widely used in Malaysia (Kamarudin et al., 2011). Findings in this special issue (Griscom et al., 2019) indicate that these technologies could generate additional emission reductions not observed in our sample by limiting the length of skid trails needed to reach felled trees.

Increased post-logging regrowth and avoided soil impacts are additional sources of carbon mitigation provided by RIL-C, but not addressed in this paper. We do not attempt to measure the carbon storage from post-RIL increased growth, but this source of climate mitigation is likely additional to estimates reported in this study (de Avila et al., 2018; Piponiot et al., 2016; Roopsind et al., 2018). Similarly, reduced soil disturbance from RIL road and skid trail construction has been shown to decrease greenhouse gas emissions (Keller et al., 2005), but a paucity of the post-logging soil-atmosphere flux data limits inclusion of soil respiration in RIL-C carbon budgets.

Many improved forest practices such as liana cutting, extending rotations, set-asides, fuel efficiency, and increased wood product storage are not included here, but would provide additional climate change mitigation. We did not have sufficient data to evaluate the potential for pre-harvest liana-cutting to limit felling impacts and reduce emissions, but preliminary research suggests this additional RIL-C practice could provide additional leakage-free emissions reductions (Marshall et al., 2017; van der Heijden et al., 2015). Extending the time between harvests would also augment carbon storage by increasing the time-averaged landscape-level carbon stocks, but this would necessitate at least temporary reductions in timber production (Griscom et al., 2017; Sasaki et al., 2016). Setting aside special areas such as high conservation value forests, riparian zones, or other sensitive areas could also invoke leakage concerns by excluding portions of the permitted logging area from harvest operations. Resulting leakage be mitigated by more thoughtful planning of existing no-impact zones (Ellis et al., 2016), which occupied 57% of the total sample block area surveyed in this study. As a demand-side intervention, we also did not address the increased mitigation from use of wood products, especially those with long residence times and those that replace concrete, steel, or aluminum. By accounting for 100% of wood product emissions as per IPCC Tier 1 recommendations (Pingoud et al., 2006) we allow for future research to estimate additional mitigation from wood product storage when life cycle inventories demonstrate that wood product inputs exceed outputs, or wood outputs show longer landfill residence times (Newell and Vos, 2012).

4.3. Pantropical logging emissions and emissions reductions

To the best of our knowledge, only three other studies estimate total carbon emission from selective logging in the tropics: 1870 Tg CO_2 (Putz et al., 2008b) based on six sample blocks in two countries, 1090 Tg CO_2 (Pearson et al., 2017) based on 13 sample blocks in six countries, and a model-based estimate of 1923 Tg CO_2 using an average of all logging entries reported in Table 3 of Sasaki et al., (2016). Our estimate (834 Tg CO_2) is lower than these, but it is based on a larger field sample (61 sample blocks in seven countries). Comparing our study to that of Pearson et al. (2014), emissions factors for Indonesia and the Guiana Shield (where Pearson et al sampled 4–5 sample blocks) align well (< 10% difference), but our results differ for RoC and Central America, where Pearson et al.'s sampling densities in these countries were very low (Table 4). Expressed as emissions per hectare, our pantropical average baseline C emissions estimate of 20.8 Mg ha^{-1} is also close to a *meta*-analysis estimate 19.9 Mg ha^{-1} that draws on all the aforementioned studies (Andrade et al., 2017).

Our estimated pantropical RIL-C mitigation potential is conservative compared to other studies. It is 50% lower than the model-based

Table 4
Comparison of baseline C emissions factors (Mg m^{-3}) from this study to (Pearson et al., 2017). Parenthetical numbers are sample sizes. *Central America values (for Belize and the Yucatan Peninsula, Mexico, respectively) are for felling only, as Pearson excludes skidding and hauling in their estimate.

	Indonesia	Guiana Shield	RoC	Central America*
(Pearson et al., 2014)	1.49 (5)	2.33 (4)	0.99 (1)	1.54 (1)
This study	1.61 (10)	2.15 (10)	1.54 (6)	1.04 (9)

estimate from Sasaki et al. (2016) and 38% lower than (Putz et al., 2008b), at least partially because we sampled regions with lower potential emissions reductions (MdD and YucP) and apply those lower emissions factors conservatively to over half of the tropical timber-producing countries assessed. Our mitigation potential is 47% of the maximum natural forest management mitigation potential reported by Griscom et al., (2017), but only 22% lower than Griscom et al.'s \$100 Mg^{-1} CO_2 2-degree pathway estimate (468 Tg CO_2 $year^{-1}$). More consistent reporting of country-level harvest volumes to replace the self-reported FRA data would likely improve the accuracy of all estimates (MacDicken, 2015).

4.4. Barriers and trade-offs

It is important to emphasize that RIL does not ensure sustainability (Putz et al., 2008a). To be effective as a conservation intervention, timber yields should be sustained and other safeguards should be in place lest managed forests become susceptible to more damaging land uses that yield greater short-term financial profits. Therefore, when evaluating logging performance, it is important to pair RIL-C as a performance metric with standards for sustained yield, worker safety issues, and the various non-carbon ecosystem services. The FSC Ecosystem Services procedure provides a potential vehicle for this pairing by combining the RIL-C methodology with other FSC standards. In particular, criteria 5.6 that stipulates that "the rate of harvest of forest products shall not exceed levels which can be permanently sustained" (FSC, 2002).

Cost is often cited as a barrier to RIL implementation, but evidence for this is inconclusive. Sasaki et al. (2016) as well as Medjibe and Putz (2012) both reviewed evidence for the cost effectiveness of RIL but both failed to find consistent results due at least in part to methodological differences among the few published studies on this topic. By disaggregating RIL performance into different logging emissions sources and best practices, the RIL-C approach allows forest managers to make financially sound decisions about where to invest their efforts in improvement of selective logging practices.

There are often trade-offs between RIL objectives and silvicultural ones. For example, to compensate for the production losses of overlogged forests, silvicultural intensification may be required (e.g., Ruslandi et al., 2017). Light-demanding species may also require clearing larger gaps to promote regeneration (Navarro-Martínez et al., 2017). To be effective, RIL should be part of a landscape approach to forest management where higher intensity silviculture, set-asides, and RIL are balanced to achieve multiple objectives (De Pellegrin Llorente et al., 2017; Runting et al., 2019).

4.5. Recommendations for management

Carbon is a useful but incomplete metric for RIL. On one hand, RIL standards have struggled for consistency across the tropics (Medjibe and Putz, 2012). On the other hand, FSC standards lack the specificity needed to drive and document measurable improvements. RIL-C attempts to fill this gap by providing a universal measurable indicator of performance. Admittedly, RIL-C does not capture important RIL benefits to biodiversity (Bicknell et al., 2014), soil erosion (Wenger et al., 2018), and other ecosystem services referenced above; nor does it address the very real concerns about worker safety. More research is needed to understand synergies and trade-offs between carbon and other ecosystem services in natural forest management. However, carbon measurement systems are currently the most robust and are ready to measure results now.

Given the challenges countries face in reporting degradation baselines, higher-tier accounting systems are needed to evaluate opportunities and demonstrate performance against climate goals (Andrade et al., 2017). Furthermore, high uncertainty from IPCC Tier 1 default values are being propagated into forest dynamic process models that inform our own actions in the face of climate change, which impedes our ability to innovate and develop appropriate policy actions (Mitchard, 2018).

RIL-C meets this challenge by applying the following actions: (1) motivate regional investment in RIL by providing a rough country-level RIL-C mitigation estimate from Table S2 (see Fig. 6); (2) set a regional logging emissions baseline with a field campaign that follows methods outlined above; (3) identify best practices to reduce emissions; and, (4) provide a rapid field based auditing protocol to quantify and verify implementation of these practices, with the capacity to correct for the influence of any covariates that effect source emissions. Countries with high logging emissions could implement this approach to deliver large portions of their NDCs at relatively low costs. As reported above, for nine countries, particularly the less developed countries of Central and West Africa, the potential RIL-C mitigation reported here represents more than half of their stated NDCs. Many countries are unaware of this potential, but others are spearheading the process now, and are including components of RIL-C in their NDC revisions (e.g., Gabon), Forest Reference Emission levels (Government of Suriname, 2018), and national forestry regulations (e.g., Indonesia). We hope that the results of this study motivate more explicit inclusion of RIL-C in national climate mitigation efforts as countries prepare to finalize their NDCs in 2020, and begin measuring performance in biennial transparency reports.

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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.2019.02.004.

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