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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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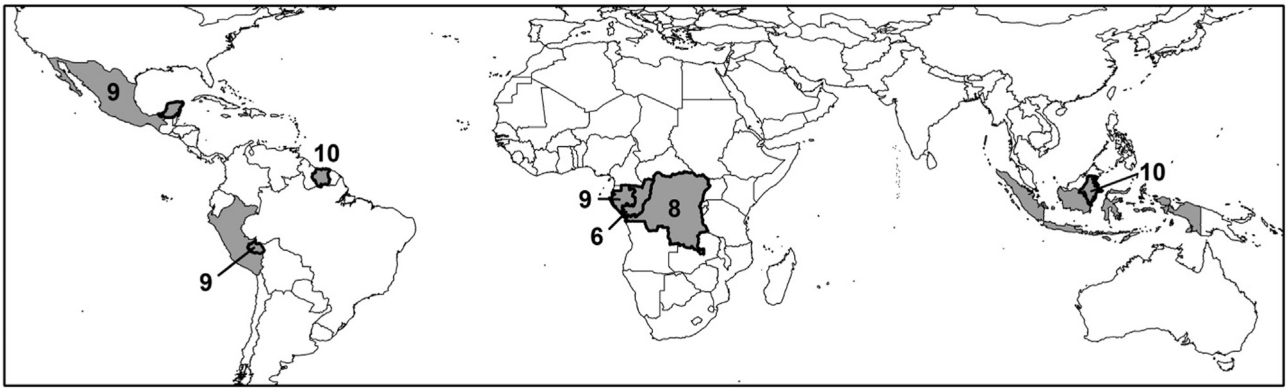

Fig. 1. Location of and number of sample blocks included in this study. Sample regions are outlined in bold, from west to east: Yucatan Peninsula, Mexico (YucP), Madre de Dios, Peru (MdD); Suriname; Gabon; Republic of Congo (RoC); Democratic Republic of Congo (DRC); and East and North Kalimantan, Indonesia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

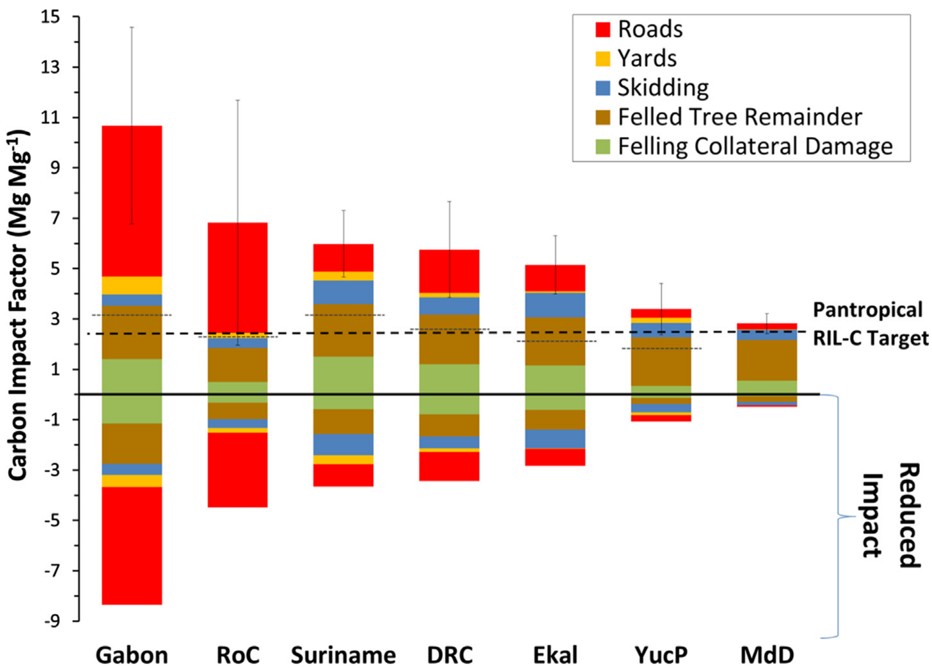

Fig. 2. Measured baseline carbon emissions (above x-axis) and carbon reduction potential (below x-axis) for the seven regions surveyed in this study. Emissions values are expressed as the carbon impact factor, the ratio of carbon lost from logging damage to carbon extracted as timber (Mg Mg^{-1}). Error 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.

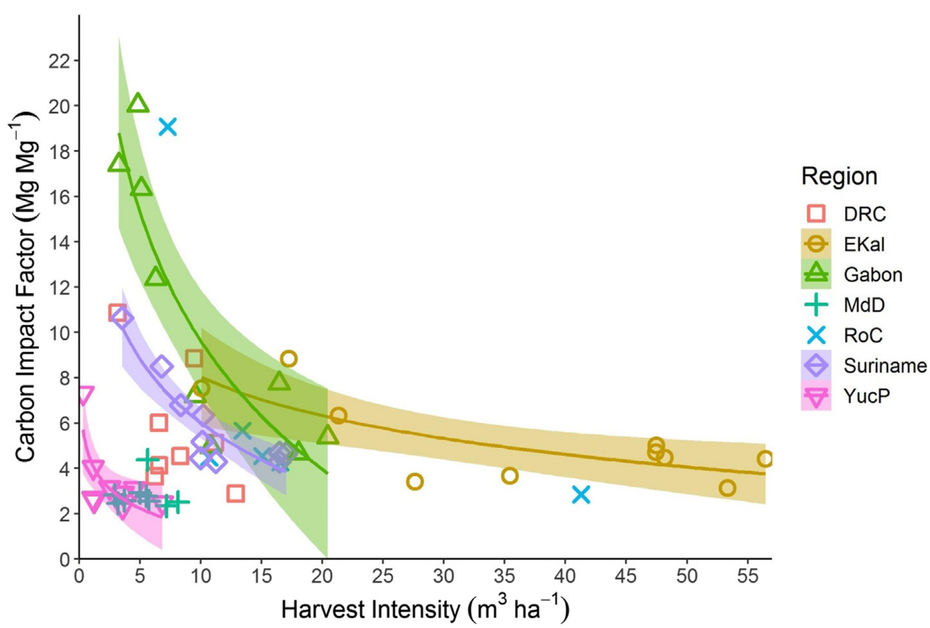

Fig. 3. Effect of harvest intensity on logging emissions by region. Emission values are expressed as carbon impact factors [carbon lost from logging damage per carbon extracted as timber (Mg Mg^{-1})]. We only show fitted lines for regions where the effect of harvest intensity is significant.

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

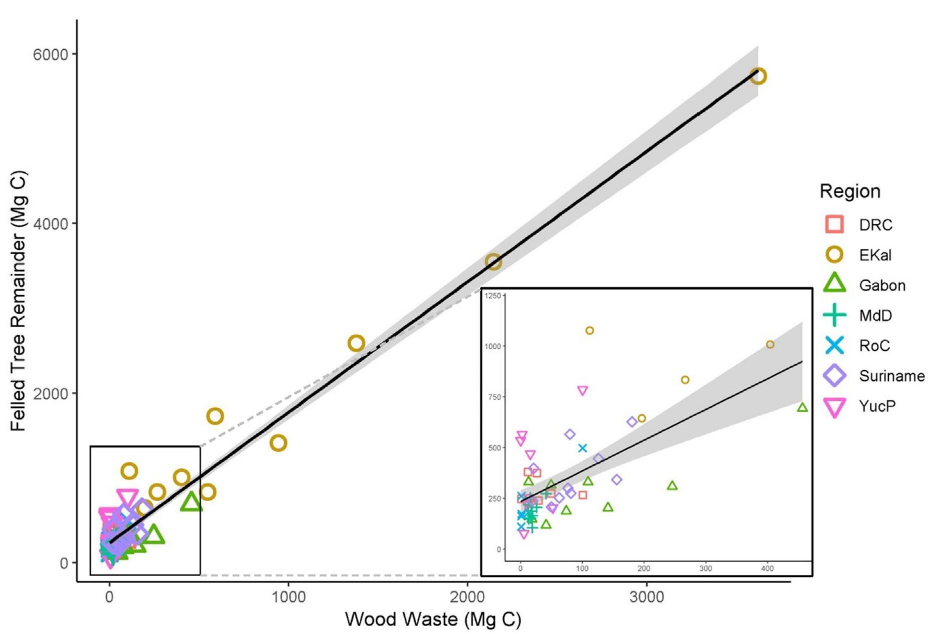

Fig. 4. Relationship between wood waste and felled tree remainder for all sample blocks by region, with inset displaying relationship in sample blocks with little wood waste.

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.

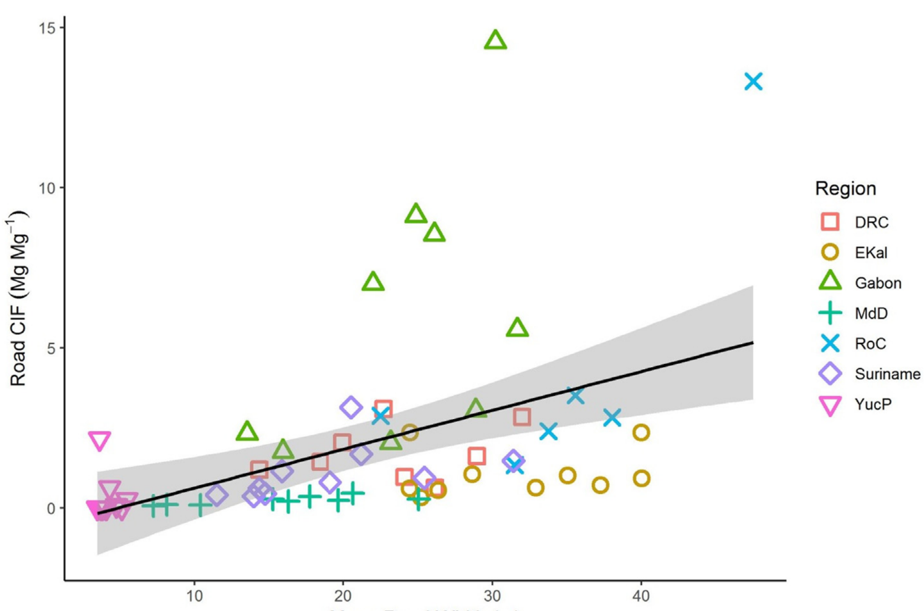

Fig. 5. Correlation between haul road width and road emissions. Road emissions expressed as carbon impact factor: carbon lost from road construction per carbon extracted as timber (Mg Mg^{-1}).

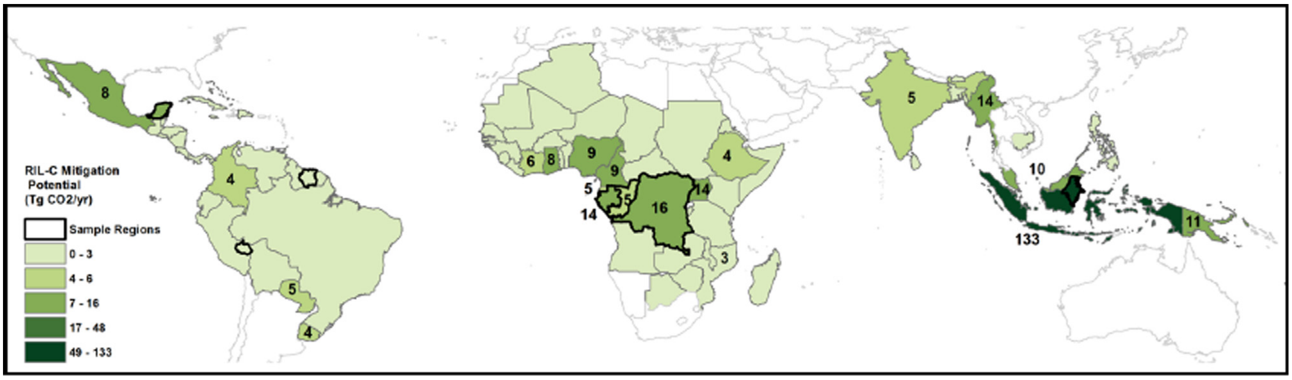

Fig. 6. Potential RIL-C emissions reductions by country. Sample regions are outlined in bold.

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 $\mathrm{Mg\,Mg^{-1}}$, 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 $\mathrm{Mg\,Mg^{-1}}$, 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 $\mathrm{Mg\,Mg^{-1}}$ (Table 3). The remaining 0.10 $\mathrm{Mg\,Mg^{-1}}$ 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 $\mathrm{Mg\,ha^{-1}}$ is also close to a *meta*-analysis estimate 19.9 $\mathrm{Mg\,ha^{-1}}$ 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.

References

Aalde, H., Gonzalez, P., Gytarsky, M., Krug, T., Kurz, W.A., Lasco, R.D., Martino, D.L., McConkey, B.G., Ogle, S., Paustian, K., Verchot, L., 2006. Generic methodologies applicable to multiple land-Use categories. In: Eggleston, H.S., Buendia, L., Miwa, K.,

Ngara, T., Tanabe, K. (Eds.), IPCC Guidelines for National Greenhouse Gas Inventories. IPCC, Hayama, Japan. pp. 1–59.
Alli, B.O., 2008. Fundamental principles of occupational health and safety. Int. Labour Off. Geneva 221. https://doi.org/10.1017/CBO9781107415324.004.
Andrade, R.B., Balch, J.K., Parsons, A.L., Armenteras, D., Roman-Cuesta, R.M., Bulkan, J., 2017. Scenarios in tropical forest degradation: Carbon stock trajectories for REDD+. Carbon Balance Manag. 12, 1–7. https://doi.org/10.1186/s13021-017-0074-0.
Appanah, S., Putz, F.E., 1984. Climber abundance in virgin dipterocarp forest and the effect of pre-felling climber cutting on logging damage [Peninsular Malaysia]. Malaysian For. 47, 335–342.
Asner, G.P., Knapp, D.E., Broadbent, E.N., Oliveira, P.J.C., Keller, M., Silva, J.N., 2005. Selective logging in the Brazilian Amazon. Science 310, 480–482. https://doi.org/10.1126/science.1118051.
Baccini, A., Walker, W., Carvalho, L., Farina, M., Sulla-Menashe, D., Houghton, R.A., 2017. Tropical forests are a net carbon source based on aboveground measurements of gain and loss. Science (80-) 358, 230–234. https://doi.org/10.1126/science.aam5962.
Bartoń, K., 2018. Package “MuMIn” Title Multi-Model Inference. Cran-R 74. doi: Available at http://www.cran.r-project.org/web/packages.
Baruch-Mordo, S., Kiesecker, J., Kennedy, C.M., Oakleaf, J.R., Opperman, J.J., 2019. From Paris to practice: sustainable implementation of renewable energy goals. Environ. Res. Lett. 14. https://doi.org/10.1088/1748-9326/aaf6e0.
Bicknell, J.E., Struebig, M.J., Edwards, D.P., Davies, Z.G., 2014. Improved timber harvest techniques maintain biodiversity in tropical forests. Curr. Biol. 24, R1119–R1120. https://doi.org/10.1016/j.cub.2014.10.067.
Blaser, J., Sarre, A., Poore, D., Johnson, S., 2011. Status of tropical forest management 2011. ITTO Technical Series, No. 38. International Tropical Timber Organization, Yokohama, Japan.
Bonner, M.T.L., Schmidt, S., Shoo, L.P., 2013. A meta-analytical global comparison of aboveground biomass accumulation between tropical secondary forests and monoculture plantations. For. Ecol. Manage. 291, 73–86. https://doi.org/10.1016/j.foreco.2012.11.024.
Chave, J., Andalo, C., Brown, S., 2005. Tree allometry and improved estimation of carbon stocks and balance in tropical forests. Oecologia 145, 87–99. https://doi.org/10.1007/s00442-005-0100-x.
Chave, J., Coomes, D., Jansen, S., Lewis, S.L., Swenson, N.G., Zanne, A.E., 2009. Towards a worldwide wood economics spectrum. Ecol. Lett. 12, 351–366. https://doi.org/10.1111/j.1461-0248.2009.01285.x.
CONAFOR, 2012. National Forest and Soil Inventory, report 2004-2009.
Conway, S., 1976. Logging Practices: Principles of Timber Harvesting Systems, first ed. Miller Freeman Publications, San Francisco, California, USA.
de Avila, A.L., van der Sande, M.T., Dormann, C.F., Peña-Claros, M., Poorter, L., Mazzei, L., Ruschel, A.R., Silva, J.N.M., de Carvalho, J.O.P., Bauhus, J., 2018. Disturbance intensity is a stronger driver of biomass recovery than remaining tree-community attributes in a managed Amazonian forest. J. Appl. Ecol. https://doi.org/10.1111/1365-2664.13134.
De Pellegrin Llorente, I., Hoganson, H.M., Carson, M.T., Windmuller-Campione, M., 2017. Recognizing spatial considerations in forest management planning. Curr. For. Reports 3, 308–316. https://doi.org/10.1007/s40725-017-0068-x.
Dykstra, D., Heinrich, R., 1996. FAO model code of forest harvesting practice. Food and Agriculture Organization (FAO), Rome.
Ellis, E., Armenta Montero, S., Hernández Gómez, I.U., Romero Montero, J.A., Ellis, P.W., Rodriguez-Ward, D., Blanco-Reyes, P., Putz, F.E., 2019. Reduced-impact logging practices reduce forest disturbance and carbon emissions in community managed forests on the Yucatán Peninsula, Mexico. For. Ecol. Manage 437, 396–410. https://www.sciencedirect.com/science/article/pii/S0378112718322138.
Ellis, P.W., Griscom, B.W., Walker, W., Gonçalves, F., Cormier, T., 2016. Mapping selective logging impacts in Borneo with GPS and airborne lidar. For. Ecol. Manage. 365, 184–196. https://doi.org/10.1016/j.foreco.2016.01.020.
Erb, K.-H., Kastner, T., Plutzar, C., Bais, A.L.S., Carvalhais, N., Fetzel, T., Gingrich, S., Haberl, H., Lauk, C., Niedertscheider, M., Pongratz, J., Thurner, M., Luyssaert, S., 2017. Unexpectedly large impact of forest management and grazing on global vegetation biomass. Nature. https://doi.org/10.1038/nature25138.
FAO, 2016. Global Forest Resources Assessment 2015. FAO, Rome, Italy.
FAO, 2010. Global Forest Resources Assessment 2010. Agriculture, Rome.
Fayolle, A., Ngomanda, A., Mbasi, M., Barbier, N., Bocko, Y., Boyemba, F., Couteron, P., Fonton, N., Kamdem, N., Katembo, J., Kondaoule, H.J., Loumeto, J., Maïdou, H.M., Mankou, G., Mengui, T., Mofack, G.I., Moundounga, C., Moundounga, Q., Nguimbous, L., Nsue Nchama, N., Obiang, D., Ondo Meye Asue, F., Picard, N., Rossi, V., Senguela, Y.P., Sonké, B., Viard, L., Yongo, O.D., Zapfack, L., Medjibe, V.P., 2018. A regional allometry for the Congo basin forests based on the largest ever destructive sampling. For. Ecol. Manage. 430, 228–240. https://doi.org/10.1016/j.foreco.2018.07.030.
Feldpausch, T.R., Jirka, S., Passos, C.A.M., Jasper, F., Riha, S.J., 2005. When big trees fall: damage and carbon export by reduced impact logging in southern Amazonia. For. Ecol. Manage. 219, 199–215. https://doi.org/10.1016/j.foreco.2005.09.003.
Fick, S., Hijmans, R.J., 2017. Worldclim 2: New 1-km spatial resolution climate surfaces for global land areas. Int. J. Climatol. 37, 4302–4315.
Forest Stewardship Council, 2018. Ecosystem Services Procedure: Impact Demonstration and Market Tools [WWW Document]. FSC-PRO-30-006 V1-0. URL https://ic.fsc.org/en/document-center/id/328 (accessed 10.22.18).
Frolking, S., Palace, M.W., Clark, D.B., Chambers, J.Q., Shugart, H.H., Hurtt, G.C., 2009. Forest disturbance and recovery: A general review in the context of spaceborne remote sensing of impacts on aboveground biomass and canopy structure. J. Geophys. Res. 114, G00E02. https://doi.org/10.1029/2008JG000911.
FSC, 2002. FSC International Standard: FSC Principles and Criteria for Forest Stewardship, FSC-STD-01-001. Bonn, Germany.
Goodman, R., Aramburu, M.H., Gopalakrishan, T., Putz, F., Gutierrez, N., Alvarez, J.L.M., Aguilar-Amuchastegui, N., Ellis, P., 2019. Carbon emissions and potential emissions reductions from low-intensity selective logging in southwestern Amazonia. For. Ecol. Manage. https://doi.org/10.1016/j.foreco.2019.02.0372.
Goodman, R.C., Baker, T.R., Phillips, O.L., 2012. Part 1: Analysis of data and methodologies of the biomass plots installed in Madre de Dios. Consultancy to Establish the Technical Guidelines to Complete a Carbon Baseline and Design a Measurement and Monitoring System for Carbon Stocks in the Region of Madre de Dios, Peru. University of Leeds, Leeds, UK.
Goodman, R.C., Phillips, O.L., Baker, T.R., 2014. The importance of crown dimensions to improve tropical tree biomass estimates. Ecol. Appl. 24, 680–698. https://doi.org/10.1890/13-0070.1.
Government of Suriname, 2018. Forest Reference Emission Level for Suriname's REDD+ Programme. Paramaribo, Suriname.
Griscom, B.W., Adams, J., Ellis, P.W., Houghton, R.A., Lomax, G., Miteva, D.A., Schlesinger, W.H., Shoch, D., Siikamäki, J.V., Smith, P., Woodbury, P., Zganjar, C., Blackman, A., Campari, J., Conant, R.T., Delgado, C., Elias, P., Gopalakrishna, T., Hamsik, M.R., Herrero, M., Kiesecker, J., Landis, E., Laestadius, L., Leavitt, S.M., Minnemeyer, S., Polasky, S., Potapov, P., Putz, F.E., Sanderman, J., Silvius, M., Wollenberg, E., Fargione, J., 2017. Natural climate solutions. Proc. Natl. Acad. Sci. 114, 11645–11650. https://doi.org/10.1073/pnas.1710465114.
Griscom, B.W., Ellis, P.W., Burivalova, Z., Halperin, J., Marthinus, D., Runting, R.K., Ruslandi, R., Putz, F.E., 2019. Reduced-impact logging in Borneo to minimize carbon emissions and impacts on sensitive habitats while maintaining timber yields. For. Ecol. Manage.
Griscom, B.W., Ellis, P.W., Putz, F.E., 2014. Carbon emissions performance of commercial logging in East Kalimantan, Indonesia. Glob. Chang. Biol. 20, 923–937. https://doi.org/10.1111/gcb.12386.
Holmes, T.P., Blate, G.M., Zweede, J.C., Pereira, R., Barreto, P., Boltz, F., Bauch, R., 2002. Financial and ecological indicators of reduced impact logging performance in the eastern Amazon. For. Ecol. Manage. 163, 93–110. https://doi.org/10.1016/S0378-1127(01)00530-8.
Indrajaya, Y., van der Werf, E., Weikard, H.P., Mohren, F., van Ierland, E.C., 2016. The potential of REDD+ for carbon sequestration in tropical forests: supply curves for carbon storage for Kalimantan, Indonesia. For. Policy Econ. 71, 1–10. https://doi.org/10.1016/j.forpol.2016.06.032.
Jarvis, A., Reuter, H.I., Nelson, A., Guevara, E., 2008. Hole-filled SRTM for the globe Version 4, available from the CGIAR-CSI SRTM 90m [WWW Document]. Database. Greenbelt, MD, EUA CGIAR Consort. Spat. Inf. URL http://www.cgiar-csi.org/2010/03/108/uot;http:/srtm.csi.cgiar.org (accessed 1.3.19).
Jürgensen, C., Kollert, W., Lebedys, A., 2014. Assessment of industrial roundwood production from planted forests. FAO Plant. For. Trees Work. Pap. FP/48/E, 40.
Kamarudin, N., Ismail, M.H., Jusoff, K., Mohd Sood, A., 2011. Evolution and development of forest harvesting in Peninsular Malaysia. Malaysian For. 74, 79–102.
Keller, M., Palace, M., Asner, G.P., Pereira, R., Silva, J.N.M., 2004. Coarse woody debris in undisturbed and logged forests in the eastern Brazilian Amazon. Glob. Chang. Biol. 10, 784–795. https://doi.org/10.1111/j.1529-8817.2003.00770.x.
Keller, M., Varner, R., Silva, H., Asner, G.P., 2005. Soil-atmosphere exchange of nitrous oxide, nitric oxide, methane, and carbon dioxide in logged and undisturbed forest in the Tapajos National Forest, Brazil. Earth Interact. 9, 1–28. https://doi.org/10.1175/EI125.1.
Kuznetsova, A., Brockhoff, P.B., Christensen, R.H.B., 2017. **lmerTest** Package: tests in linear mixed effects models. J. Stat. Softw. 82. https://doi.org/10.18637/jss.v082.i13.
MacDicken, K.G., 2015. Global forest resources assessment 2015: what, why and how? For. Ecol. Manage. 352, 3–8. https://doi.org/10.1016/j.foreco.2015.02.006.
Marshall, A.R., Coates, M.A., Archer, J., Kivambe, E., Mnendendo, H., Mtoka, S., Mwakisoma, R., de Figueiredo, R.J.R.L., Njilima, F.M., 2017. Liana cutting for restoring tropical forests: a rare palaeotropical trial. Afr. J. Ecol. 55, 282–297. https://doi.org/10.1111/aje.12349.
Marshall, D.D., Iles, K., Bell, J.F., 2004. Using a large-angle gauge to select trees for measurement in variable plot sampling. Can. J. For. Res. 845, 840–845. https://doi.org/10.1139/X03-240.
McGroddy, M., Daufresne, T., Hedin, L., 2004. Scaling of C: N: P stoichiometry in forests worldwide: implications of terrestrial Redfield-type ratios. Ecology 85, 2390–2401.
Medjibe, V.P., Putz, F.E., 2012. Cost comparisons of reduced-impact and conventional logging in the tropics. J. For. Econ. 18, 242–256. https://doi.org/10.1016/j.jfe.2012.05.001.
Medjibe, V.P., Putz, F.E., Starkey, M.P., Ndouna, A.A., Memiaghe, H.R., 2011. Impacts of selective logging on above-ground forest biomass in the Monts de Cristal in Gabon. For. Ecol. Manage. 262, 1799–1806. https://doi.org/10.1016/j.foreco.2011.07.014.
Mitchard, E.T.A., 2018. The tropical forest carbon cycle and climate change. Nature 559, 527–534. https://doi.org/10.1038/s41586-018-0300-2.
Mokany, K., Raison, R.J., Prokushkin, A.S., 2006. Critical analysis of root : shoot ratios in terrestrial biomes. Glob. Chang. Biol. 12, 84–96. https://doi.org/10.1111/j.1365-2486.2005.001043.x.
Navarro-Martínez, A., Palmas, S., Ellis, E., Blanco-Reyes, P., Vargas-Godínez, C., Iuit-Jiménez, A., Hernández-Gómez, I., Ellis, P., Álvarez-Ugalde, A., Carrera-Quirino, Y., Armenta-Montero, S., Putz, F., 2017. Remnant trees in enrichment planted gaps in Quintana Roo, Mexico: reasons for retention and effects on seedlings. Forests 8, 272. https://doi.org/10.3390/f8080272.
Newell, J.P., Vos, R.O., 2012. Accounting for forest carbon pool dynamics in product carbon footprints: challenges and opportunities. Environ. Impact Assess. Rev. 37, 23–36. https://doi.org/10.1016/j.eiar.2012.03.005.
Pearson, T.R.H., Brown, S., Casarim, F.M., 2014. Carbon emissions from tropical forest

degradation caused by logging. Environ. Res. Lett. 9, 034017. https://doi.org/10.1088/1748-9326/9/3/034017.
Pearson, T.R.H., Brown, S., Murray, L., Sidman, G., 2017. Greenhouse gas emissions from tropical forest degradation: an underestimated source. Carbon Balance Manag. 12, 11. https://doi.org/10.1186/s13021-017-0072-2.
Pinard, M.A., Putz, F.E., 1996. Retaining forest biomass by reducing logging damage. Biotropica 28, 278–295.
Pinard, M.A., Putz, F.E., Tay, J., Sullivan, T.E., 1995. Creating timber harvest guidelines for a reduced-impact logging project in Malaysia. J. For. 41–45.
Pingoud, K., Skog, K.E., Martino, D.L., Tonosaki, M., Xiaoquan, Z., 2006. Harvested Wood Products. In: IPCC Guidelines for National Greenhouse Gas Inventories, pp. 1–33.
Piponiot, C., Sist, P., Mazzei, L., Peña-Claros, M., Putz, F.E., Rutishauser, E., Shenkin, A., Ascarrunz, N., de Azevedo, C.P., Baraloto, C., França, M., Guedes, M., Coronado, E.N.H., d'Oliveira, M.V.N., Ruschel, A.R., da Silva, K.E., Sotta, E.D., de Souza, C.R., Vidal, E., West, T.A.P., Hérault, B., 2016. Carbon recovery dynamics following disturbance by selective logging in amazonian forests. Elife 5. https://doi.org/10.7554/eLife.21394.
Putz, F.E., Pinard, M.A., 1993. Reduced-impact logging as a carbon-offset method. Conserv. Biol. 7, 755–757. https://doi.org/10.1046/j.1523-1739.1993.7407551.x.
Putz, F.E., Sist, P., Fredericksen, T., Dykstra, D., 2008a. Reduced-impact logging: challenges and opportunities. For. Ecol. Manage. 256, 1427–1433. https://doi.org/10.1016/j.foreco.2008.03.036.
Putz, F.E., Zuidema, P., Pinard, M., Boot, R.G., Sayer, J., Sheil, D., Sist, P., Vanclay, J.K., 2008b. Improved tropical forest management for carbon retention. PLoS Biol. 6, e166. https://doi.org/10.1371/journal.pbio.0060166.
Putz, F.E., Zuidema, P.A., Synnott, T., Peña-Claros, M., Pinard, M.A., Sheil, D., Vanclay, Jerome K., Sist, P., Gourlet-Fleury, S., Griscom, B.W., Palmer, J., Zagt, R., 2012. Sustaining conservation values in selectively logged tropical forests: the attained and the attainable. Conserv. Lett. 5, 296–303. https://doi.org/10.1111/j.1755-263X.2012.00242.x.
R Development Core Team, 2014. R: A Language and Environment for Statistical Computing.
Read, J.M., 2003. Spatial analyses of logging impacts in Amazonia using remotely sensed data. Photogramm. Eng. Remote Sensing 69, 275–282.
Réjou-Méchain, M., Tymen, B., Blanc, L., Fauset, S., Feldpausch, T.R., Monteagudo, A., Phillips, O.L., Richard, H., Chave, J., 2015. Using repeated small-footprint LiDAR acquisitions to infer spatial and temporal variations of a high-biomass Neotropical forest. Remote Sens. Environ. 169, 93–101.
Romero, C., Guariguata, M., Sills, E.O., Cerutti, P.O., Lescuyer, G., Putz, F.E., 2017. Evaluation of the impacts of Forest Stewardship Council (FSC) certification of natural forest management in the tropics. Int. For. Rev. 19, 1–14. https://doi.org/10.17528/cifor/004347.
Roopsind, A., Caughlin, T.T., van der Hout, P., Arets, E., Putz, F.E., 2018. Trade-offs between carbon stocks and timber recovery in tropical forests are mediated by logging intensity. Glob. Chang. Biol. https://doi.org/10.1111/gcb.14155.
Runting, R.K., Ruslandi, Griscom, B., Strueb, M.J., Satar, M., Meijaard, E., Burivalova, Z., Cheyne, S.M., Deere, N.J., Game, E.T., Putz, F.E., Wells, J.A., Wilting, A., Ancrenaz, M., Ellis, P., Khan, F.A.A., Leavitt, S.M., Marshall, A.J., Possingham, H.P., Watson, J.E.M., Venter, O., 2019. Larger gains delivered by improved management over sparing-sharing for tropical forests. Nat. Sustain. 2, 53–61.
Ruslandi, Cropper, W.P., Putz, F.E., 2017. Effects of silvicultural intensification on timber yields, carbon dynamics, and tree species composition in a dipterocarp forest in Kalimantan, Indonesia: an individual-tree-based model simulation. For. Ecol. Manage. 390, 104–118. https://doi.org/10.1016/j.foreco.2017.01.019.
Sasaki, N., Asner, G.P., Pan, Y., Knorr, W., Durst, P.B., Ma, H.O., Lowe, A.J., Koh, L.P., Putz, F.E., 2016. Sustainable management of tropical forests can reduce carbon emissions and stabilize timber production. Front. Environ. Sci. 4, 1–13. https://doi.org/10.3389/fenvs.2016.00050.
The Nature Conservancy, TerraCarbon LLC, 2016a. Methodology for improved forest management through reducted impact logging (RIL-C) [WWW Document]. VM0035. URL https://verra.org/methodology/vm0035-methodology-for-improved-forest-management-through-reduced-impact-logging-v1-0/ (accessed 10.22.18).
The Nature Conservancy, TerraCarbon LLC, 2016b. Performance Method for Reduced Impact Logging in East and North Kalimantan [WWW Document]. VM0047. URL https://verra.org/methodology/vmd0047-performance-method-for-reduced-impact-logging-in-east-and-north-kalimantan-v1-0/ (accessed 10.22.18).
Tropical Forest Foundation Indonesia, 2015. Generic standard for RIL [WWW Document]. URL http://www.tff-indonesia.org/index.php/en/r-i-l/generic-standard-for-ril (accessed 5.1.15).
Umunay, P., Gregoire, T., Gopalakrishna, T., Ellis, P., Putz, F., 2019. Selective logging emissions and potential emission reductions from reduced-impact logging in the Congo Basin. For. Ecol. Manage. 437, 360–371. https://www.sciencedirect.com/science/article/pii/S0378112718321960.
United Nations Framework Convention on Climate Change, 2015. COP 21 Climate Agreement. United Nations Framework Convention on Climate Change, Paris.
van der Heijden, G.M.F., Powers, J.S., Schnitzer, S.A., 2015. Lianas reduce carbon accumulation and storage in tropical forests. Proc. Natl. Acad. Sci. 112, 13267–13271. https://doi.org/10.1073/pnas.1504869112.
Weishampel, J.F., Hightower, J.N., Chase, A.F., Chase, D.Z., 2012. Use of airborne LiDAR to delineate canopy degradation and encroachment along the Guatemala-Belize border. Trop. Conserv. Sci. 5, 12–24.
Wenger, A.S., Atkinson, S., Santini, T., Falinski, K., Hutley, N., Albert, S., Honring, N., Watons, J.E.M., Mumby, P.J., Jupiter, S.D., 2018. Predicting the impact of logging activities on soil erosion and water quality in steep, forested tropical islands. Environ. Res. Lett. 13, 44035.
World Resources Institute, 2017. CAIT Climate Data Explorer [WWW Document]. URL https://www.climatewatchdata.org/ (accessed 12.21.17).
Zalman, J., Ellis, P., Crabbe, S., Roopsind, A., 2019. Opportunities for carbon emissions reduction from selective logging in Suriname. For. Ecol. Manage.
Zanne, A.E., Lopez-Gonzalez, G., Coomes, D.A., Ilic, J., Jansen, S., Lewis, S.L., Miller, R.B., Swenson, N.G., Wiemann, M.C., Chave, J., 2009. Global wood density database.