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Abstract Aboveground carbon dynamics were estimated in two strips (30 m × 150 m) clear-cut in the Peruvian Amazon following 15 years after clearing. While strip 1 recovered 50 % (64 Mg C ha^{-1}) of its pre-clearing carbon stocks, the clear-cut portion of strip 2 recovered 39 % (53 Mg C ha^{-1}). During this period, there were high carbon sequestration rates (> 6 Mg C ha^{-1} year^{-1}) from tree growth and low carbon emission rates from tree mortality (up to 3 Mg C ha^{-1} year^{-1}) in both strip 1 and the clear-cut portion of strip 2. Recruits make up most of the tree composition in both strips, and therefore play an important role in carbon sequestration. However, there were high carbon emission rates from timber harvesting and processing (> 20 Mg C ha^{-1} year^{-1}). A thinning treatment, applied 7 years after clearing, did not affect carbon dynamics. In the 15-year period total carbon emissions (+) were greater than sequestration (−). Net carbon fluxes for strip 1 and the clear-cut portion of strip 2 were +26 and +68 Mg C ha^{-1}. Comparable carbon dynamics were found in the deferment-cut treatment, applied to the south half of strip 2. To conclude, the 15 year-old clear-cut strips are not carbon sinks yet, and will require > 140 years to reach old-growth carbon stocks.

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Introduction

Tropical logged forests in addition to having high carbon sequestration rates are also large reservoirs of biomass and biodiversity. Carbon sequestration rates in Amazonian logged forests vary from 1 to > 4 Mg C ha^{-1} year^{-1} (Blanc et al. 2009; Mazzei et al. 2010), and overall they are much greater than unlogged forest stands (< 1 Mg C ha^{-1} year^{-1}) (Phillips et al. 2008).

Several factors, however, influence aboveground carbon dynamics in a logged forest such as tree recruitment, growth and mortality (Vieira et al. 2004; Blanc et al. 2009). After the initial harvest (15–100 m^3 ha^{-1}), logged forests exhibit higher tree mortality rates (> 2 % year^{-1}) than unlogged forests (> 1 % year^{-1}), which can last more than 8 years after logging (Silva et al. 1995; de Graaf et al. 1999; Blanc et al. 2009). After this period, biomass in a logged forest accumulates due to growth of residual trees and tree recruitment. The contribution of tree recruitment to aboveground carbon stocks is usually small, ranging from < 1 to about 2 Mg C ha^{-1} year^{-1} (Blanc et al. 2009; Mazzei et al. 2010).

Due to felled trees and collateral damage, high carbon emissions are produced during and after timber harvesting. That is, untrained loggers can severely damage up to 10–20 other trees for every harvested tree (Putz et al. 2008). Thus, logging operations leave behind large amounts of logging slash and woody debris, which eventually decompose releasing carbon emissions (Keller et al. 2004). Furthermore, due to high mortality rates after timber harvesting, carbon emissions increase and remain high (> 10 Mg C ha^{-1} year^{-1}) for several years after logging (Blanc et al. 2009). Reduced impact logging (RIL) techniques can reduce carbon emissions when compared to conventional logging (Pinard and Putz

1996; Pinard and Cropper 2000), but these techniques are not commonly used in the Amazon. Another source of carbon emissions is processing of timber products, but these emissions vary depending on sawmill transformation efficiency (Lentini et al. 2005; Pereira et al. 2010).

In this study, we examined aboveground carbon dynamics after the application of strip clear-cutting in the Peruvian Amazon. Although strip clear-cutting has been used extensively in North America and Europe (Smith 1986; Matthews 1989), it is not commonly used in the Amazon. Strip clear-cutting of tropical forest was proposed by Tosi (1982) and Hartshorn (1989a) in the early 1980s for the management of 44,000 ha of old-growth forest in the Palcazu Valley, Peru (Hartshorn 1989b). Thus, this system is also known as the Palcazu forest management model.

In strip clear-cutting, native gap-dependent timber species are harvested simulating gap dynamics (Hartshorn 1989a). High intensity harvesting (250 m^3 ha^{-1}) is concentrated in long, narrow clear-cut strips (30–40 m wide) with rotation cycles of 30–40 years (Hartshorn 1989a). In the clear-cut strips, all timber (≥5 cm dbh), regardless of species, is harvested and used locally or sold to attain maximum value from the strips (Hartshorn 1989a). Animal traction is used to reduce soil compaction and natural regeneration of seeds and stump sprouts is permitted (Hartshorn 1989a; Gorchov et al. 1993).

Yet, several studies have raised concerns about the sustainability of strip clear-cutting. A high species richness was found two and a half years after clearing two experimental strips in the Palcazu, indicating a high regeneration capacity for this system (Hartshorn 1989a). However, the composition at the early stages of regeneration was dominated by pioneer species of low commercial value (Gorchov et al. 1993). Fifteen years into the regeneration of two clear-cut strips, commercial species still had low densities and comprised only 8–15 % of the original basal area (Rondon et al. 2009a). Furthermore, commercial stems had low diameter growth rates (<0.3 cm $year^{-1}$) even after the application of silvicultural thinning (Dolanc et al. 2003). Growth projections predicted a low timber production (3–8 m^3 ha^{-1}) at the time of a second harvest (Rondon et al. 2009b), which questions the long-term sustainability of the system (Cornejo and Gorchov 1993; Rondon et al. 2010).

Before strip clear-cutting is implemented at a large scale, it is necessary to examine the potential use of clear-cut strips as carbon sinks. In 1989 two experimental strips (30 × 150 m) were clear-cut in the Peruvian Amazon to study seed dispersal and tree regeneration (Gorchov et al. 1993). A deferment-cut treatment was applied to the south half of one of the strips (Cornejo and Gorchov 1993). Due to low growth rates in commercial trees, in 1996 a thinning treatment was applied to both strips (Dolanc et al. 2003).

Objectives

In this study we report aboveground carbon dynamics 15 years after strip clear-cutting in the Peruvian Amazon, focusing on carbon sequestration from tree growth and tree recruitment, and carbon emissions from tree mortality, harvested trees and timber processed. A second objective was to test the effect of thinning on carbon dynamics. In addition we compared carbon dynamics in the clear-cut strips and the deferment-cut treatment.

Methods

Study site

This study took place at the *Centro de Investigaciones Jenaro Herrera* (CIJH), located near the Ucayali River, 200 km south of Iquitos, Loreto, Peru. The mean annual temperature is 26.5 °C and the mean annual precipitation is 2521 mm (Spichiger et al. 1989). A relatively dry period occurs from June to August, but rainfall varies highly each month of the year (Ascorra et al. 1993; Rondon et al. 2009b). Soils are sandy-loam and the vegetation is considered lowland tropical rainforest on high terrace (Spichiger et al. 1989). The plant families with the highest densities are Sapotaceae, Fabaceae, Lecythidaceae, Chrysobalanaceae, Lauraceae, and Myristicaceae (Spichiger et al. 1996).

History of the strips

In 1989, two (30 × 150 m) strips, 150 m apart, were clear-cut in primary high terrace forest at CIJH (Fig. 1). This area had been selectively logged 15–20 years previously, but the forest maintained an intact canopy. Strip 1 was clear cut in April–May, 1989 and strip 2 in October–November, 1989. Lianas and shrubs were cut before tree felling. Most trees >5 cm in diameter at breast height (dbh) were felled in each strip using directional felling to ensure the cut trees landed in the strips (Gorchov et al. 1993). A few large trees leaning out of the strips (>28 cm dbh, N = 5 in strip 1 and N = 13 in strip 2) were not cut to avoid damage to the surrounding forest (Cornejo and Gorchov 1993), and were removed from the data analysis. An experimental deferment-cut treatment was implemented in the south half of strip 2, where only commercial trees ≥30 cm dbh and "other" species (trees of no commercial value >5 cm dbh) were harvested in 1989, and smaller commercial trees were left uncut (n = 56, 5–28 cm dbh) to grow for the next harvest (Cornejo and Gorchov 1993).

In both strips all timber harvested was locally used or carried off site. A survey of trees (≥5 cm dbh) was conducted in both strips prior to the 1989 felling (Cornejo and Gorchov 1993). After clearing, each strip was divided into 20 subplots (15 × 15 m) (Fig. 1). While strip 1

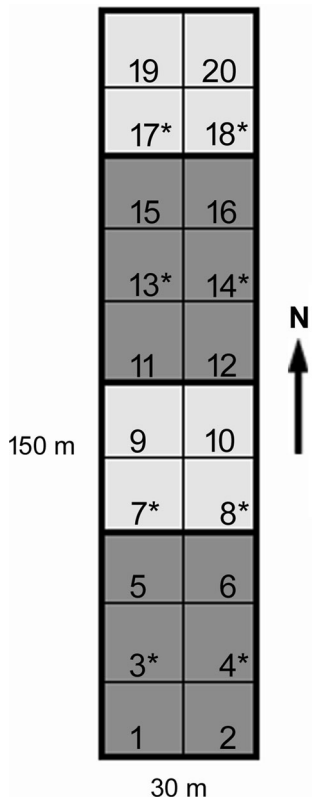

Fig. 1 Diagram of each of the two strips (30 × 150 m) clear-cut in 1989 at *Centro de Investigaciones Jenaro Herrera* (CIJH), Loreto, Peru, with 20 subplots (15 × 15 m) marked in each strip. In the south half portion of strip 2 (subplots 1–10), 56 trees (5–28 cm dbh) were left uncut as part of a deferment-cut treatment. Subplots with an *asterisk* (*) were censused for all saplings (>2 m tall), and the advanced regeneration and stump sprouts were censused throughout each strip. Thinned subplots in 1996 are shaded in *dark gray*. This is a modified figure from Dolanc et al. (2003)

consists of 20 subplots, each portion of strip 2, the clear-cut and deferment-cut, consists of 10 subplots (Fig. 1). Stump sprouts and the advanced regeneration (trees <5 cm dbh that survived the 1989 clearing, but >2 m tall in 1989) were identified and tagged in all subplots of strips 1 and 2. Recruits (new saplings reaching 2 m tall) were identified and censused in 8 out of 20 subplots of each strip (Fig. 1). Hereinafter referred to as demography subplots, for they have a complete inventory of recruits, advanced regeneration and tree stumps. In strip 2, each felling portion, the clear-cut and deferment-cut, has 4 demography subplots for a total of 8 subplots.

Censuses in the clear-cut strips took place once a year: 1990–1994, 1996 and 2000. An experimental silvicultural thinning treatment took place in 1996, where pioneer trees (all *Cecropia* spp. and trees <10 m tall of the genus *Alchornea* and the family Melastomataceae) were girdled by machete in 12 subplots of each strip (Dolanc et al. 2003; Fig. 1). The last censuses in strip 1 and 2 took place in 2004 and 2005, when all trees ≥5 cm dbh were measured in all 20 subplots of each strip (Rondon et al. 2009a).

Tree identification

Tree identification was conducted using Gentry (1993) and Spichiger et al. (1989, 1990). Voucher specimens were deposited at the CIJH herbarium, at the Universidad Nacional de la Amazonía Peruana (UNAP) herbarium (Herbario Amazonense, AMAZ), and at Miami University herbarium (Willard Sherman Turrell herbarium, MU). Voucher specimens of difficult taxa were brought for identification to the Missouri Botanical Garden herbarium (MOBOT). In the pre-clearing inventory several taxa were not identified to the species-level, but identification was done either to the genus or family level. Only 1 % of these trees had no identification to the family level. In the following censuses, great efforts were taken to identify tree taxa (Dolanc et al. 2003; Rondon et al. 2009a). In the 2004/2005 censuses, all tree taxa ≥5 cm dbh were identified to either species or genus level, and only <1 % of the trees had no identification to the species level.

Aboveground carbon dynamics

Although censuses in the strips took place in 1989, from 1990 to 1994, in 1996, 2000, 2004 (strip 1) and 2005 (strip 2), we analysed census data of the years 1989, 1992, and 1996 onward in order to obtain sufficient tree growth between time intervals to evaluate carbon dynamics. Carbon dynamics per subplot were modelled in strip 1, the clear-cut and deferment-cut portions of strip 2 using all trees ≥5 cm dbh. In 1996 a complete census took place only in strip 2 (stump sprouts, advanced regeneration and recruits), but not in strip 1, where measurements of stump sprouts and advanced regeneration were not taken (Dolanc et al. 2003). Thus, we obtained the annual diameter increment for the period 1992–1994 to project tree size for 1996.

Thus, carbon stocks (Mg C ha^{-1}) were estimated for six time periods: in the pre-clearing (1989) considered old-growth forest [calculated with all trees >7.5 cm dbh in strip 1, only data available for the subplots for the year 1989 (Rondon et al. 2009a)], after clearing or year zero (1989), three years after clearing (1992), seven years after clearing (1996), 11 years after clearing (2000), and about 15 years after clearing (2004/2005). Since we have complete tree inventories for year zero (stumps, advanced regeneration, and trees left uncut in deferment-cut of strip 2) and year 15, we used all 20 subplots in strip 1 and all 10 subplots in the clear-cut and deferment-cut portions of strip 2 to estimate wood specific gravity and carbon stocks, but for the remaining years we used data from the demography subplots which include recruits.

Carbon sequestration from tree growth and recruitment, and carbon emissions from tree mortality were estimated using demography subplots. These subplots were also used to estimate annual diameter increment (cm $year^{-1}$), mortality (% $year^{-1}$) and recruitment (%

year^{-1}) of new trees following Sheil et al. (1995) and Sheil et al. (2000) [Table S1 of Supplementary Material (SM)]. Carbon emission rates from timber harvested and timber processed were estimated using pre-clearing data. We also estimated the net carbon flux by adding total carbon sequestration and total carbon emissions.

Wood specific gravity (WSG)

Since wood specific gravity (WSG) shows regional variation across the Amazon, it is critical to include in the allometric Eqs. (1, 2) values belonging to the particular region studied (Baker et al. 2004). Therefore, to estimate WSG for trees in the strips, we compiled a dataset of WSG values for a total of 144 tree taxa from the Peruvian Amazon, published in Baker et al. (2004) and in Woodcock (2000). As in other studies, we allocated WSG to tree taxa in the strips based on taxonomy (Baker et al. 2004; Baraloto et al. 2011). That is, we obtained WSG means at the species-level, at the generic-level and at the family-level. WSG at the species-level was allocated to the matching tree species in the strips. In absence of matching species we allocated WSG at the generic level. In the absence of generic-level matching was done at the family-level.

For those tree species lacking of a match with the compiled dataset, we assigned WSG values based on successional status. Thus, we classified the compiled dataset as well as the non-matched tree species in the strips as old-growth, successional or unknown status (Appendix 1 and Table S2 of SM). We obtained the mean successional WSG (0.489 g cm^{-3}), and allocated this value to all successional, non-matched tree species in the strips. To avoid overestimating WSG in the strips, which affects biomass estimates (Baker et al. 2004), we allocated the overall mean WSG (0.553 g cm^{-3}) to old-growth, non-matched tree species in the strips, instead of using the old growth WSG value (>0.6 g cm^{-3}). We also allocated the overall mean WSG to trees in the strips with unknown successional status or unknown taxonomic identification, as it was done in other studies (Baker et al. 2004; Baraloto et al. 2011).

Carbon stocks and carbon sequestration rates

We estimated aboveground biomass (AGB) (Mg ha^{-1}) per subplot using allometric equations. AGB for trees between 5 and 10 cm dbh was estimated using allometric Eq. (1) for moist tropical forest of southern Mexico (Chave et al. 2004), which includes WSG (g cm^{-3}), dbh (cm), and mean WSG per subplot:

$$AGB_1 = \frac{WSG * e^{(-1.9703+2.1166*\ln(dbh))}}{mean(WSG)} \quad (1)$$

AGB for trees >10 cm dbh was estimated using allometric Eq. (2) for general moist forest (Chave et al. 2005). AGB (kg) values were divided by 1000 to obtain values in megagrams (Mg), equivalent to 1 ton. To standardize height measurements taken in the post-clearing censuses, we developed an allometric equation for total height (h) (3) using diameter and height data of harvested trees in JH (p value ≤ 0.05, $r^2 = 0.83$) (Rondon 2008; Rondon et al. 2009b).

$$AGB_2 = 0.0509 * WSG * dbh^2 * h \quad (2)$$

$$h = 9.871 * \ln(dbh) - 8.0063 \quad (3)$$

Carbon accumulation during forest regeneration was determined by estimating carbon stocks (Mg C ha^{-1}) for trees ≥5 cm dbh, per subplot for each census. After estimating AGB per subplot, AGB was converted into carbon assuming a conversion factor of 0.5 (Penman et al. 2003). These trees comprised advanced regeneration, sprouting stumps, and recruits. In this system, recruits make up most of the tree composition, whereas the advanced regeneration and sprouting stumps are less numerous (Gorchov et al. 1993; Rondon et al. 2009a). In the strips there are two types of recruits: the new recruits or cohort of trees that appeared in each census, and the previously established recruits that had survived into the next census.

To assess carbon balance of the AGB pool in the strips, we differentiated between carbon inputs or gains by recruitment of new trees and by tree growth (Vieira et al. 2004; Blanc et al. 2009; Mazzei et al. 2010). Carbon stocks (Mg C ha^{-1}) added by recruitment of new trees in each census was estimated per subplot, using biomass allometric equations and Penman et al. (2003)'s carbon conversion factor. Annual carbon sequestration from recruitment of new trees was calculated by dividing recruitment carbon stocks by time interval between censuses. Carbon stocks and sequestration rates by tree growth per subplot were estimated using the advanced regeneration, sprouting stumps and established recruits ("recruit growth"). After obtaining carbon stocks from tree growth for each census, we estimated annual carbon sequestration by obtaining the difference in growth carbon stocks between two censuses, and dividing it by time interval between censuses.

Carbon emissions from tree mortality

Carbon losses or emissions from tree mortality were estimated per subplot for each census, using an exponential decay model (Olson 1963; Chambers et al. 2000). Before using the decay model, decomposing biomass of dead trees was estimated with Eqs. 1 and 2, and was classified as coarse woody biomass (CWB) or fine woody biomass (FWB). CWB is defined as large pieces of standing dead trees (≥10 cm dbh) with a slow decomposition, whereas FWB is defined as smaller pieces of dead trees such as twigs or tree branches (<10 cm in diameter) with a rapid decomposition (Chambers et al. 2004; Keller et al. 2004).

For dead trees ≥10 cm dbh, decomposing AGB consisted of CWB and FWB (Chambers et al. 2004) (Eq. 4):

$$AGB = CWB + FWB \tag{4}$$

CWB was estimated indirectly using a CWB fraction equation developed by Chambers et al. (2004) (Eq. 5):

$$f_{CWB} = 0.774 + 0.0018 * dbh \tag{5}$$

Once CWB was estimated, FWB was estimated with Eq. 4, as the difference between AGB and CWB. Decomposing AGB of dead trees < 10 cm dbh was considered to consist entirely of FWB (CWB = 0).

Decay model

We used the exponential decay model (Eq. 6) to estimate change in decomposing CWB and FWB (Mg ha^{-1}) individually (Olson 1963; Chambers et al. 2000):

$$M_t = M_o * e^{-K*t}, \tag{6}$$

where M_o is the initial mass, M_t is the final mass at year t and K is the fractional mass lost per year or decomposition rate. When using the decay model, decomposition rate (K) becomes a crucial parameter. Two models for estimating K in the Amazon have been developed: For French Guyana Hérault et al. (2010) parameterized a model for trees with WSG ranging from 0.230 to 1.241 g cm^{-3}, and for central Amazonia Chambers et al. (2000) parameterized a model for trees with lower WSG ranging from ~0.30 to ~0.90 g cm^{-3}. Thus far, a K model for western Amazonia has not been developed yet. Due to the comparability between WSG values from western Amazonia and those from central Amazonia (Baker et al. 2004; Baraloto et al. 2011), in this study we used the Chambers et al. (2000) model to estimate K for CWB (Eq. 7, Figure S1 of SM). K for old-growth species (≥10 cm dbh) was lower than successional species (Figure S1 of SM). Following Blanc et al. (2009) and Keller et al. (2004), K for FWB was set at 0.2 year^{-1}.

$$K = (1.104 - 0.670 * WSG - 0.163 * \log(dbh))^2 \tag{7}$$

Carbon emissions from timber harvesting

We estimated carbon emissions from decomposing biomass after the 1989 timber harvest. Decomposing biomass consisted of stumps from harvested trees, classified either as CWB (harvested trees were ≥10 cm dbh), or FWB (harvested trees were < 10 cm dbh). We also estimated FWB from harvested trees ≥10 cm dbh (e.g., branches).

Stump biomass of harvested trees ≥10 cm dbh was estimated by multiplying WSG by stump volume (V), which in turn was estimated using the Smalian's formula (Eq. 8) (Baker et al. 2007):

$$V = L * \left[\frac{\pi * \left(\frac{d_1}{2}\right)^2 + \pi * \left(\frac{d_2}{2}\right)^2}{2}\right], \tag{8}$$

where L is length (m), d_1 and d_2 are diameters (m). Stump height measurements were not taken; thus, we estimated the heights of stumps from harvested trees ≥30 cm dbh and from trees < 30 cm dbh to measure approximately 0.5 and 0.15 m, respectively. The second or upper diameter (cm) in Eq. 8 was estimated with a taper function (Eq. 9) based on dbh (cm) and height (m) (Chambers et al. 2000):

$$d_2 = 1.59 * dbh * h^{-0.091} \tag{9}$$

In order to use the exponential decay model (Eq. 6), we used Eq. 7 to estimate K for stumps of harvested trees ≥10 cm dbh classified as CWB.

We assumed that there were two types of FWB associated with timber extraction in the strips: decomposing biomass (e.g., branches and twigs) due to harvested trees ≥10 cm dbh, and stumps from harvested trees < 10 cm dbh. We used Eqs. 1, 4, and 5 to estimate decomposing FWB due to harvested trees ≥10 cm dbh. Biomass of decomposing stumps of harvested trees < 10 cm dbh was estimated with the volume equation for a cylinder multiplied by WSG (Keller et al. 2004). To estimate change in either decomposing FWB or decomposing stumps of harvested trees < 10 cm dbh, we used the exponential decay model by setting K at 0.2 year^{-1} (Keller et al. 2004; Blanc et al. 2009).

Carbon emissions from timber processing

We also estimated carbon emissions from timber processing based on the assumption that all trees (≥5 cm dbh) were harvested and taken to a local sawmill. Biomass of harvested logs belonging to trees ≥10 cm dbh was estimated by subtracting stump biomass from total CWB estimated using Eq. 5. Biomass of harvested logs belonging to trees < 10 cm dbh was estimated by subtracting stump biomass from the total AGB (Eq. 1).

Transformation efficiency for sawmill operations was estimated by dividing volume of timber products (m^3) by volume of roundwood (m^3) produced in a particular year, following methods of Lentini et al. (2005) and Pereira et al. (2010). Since transformation efficiency varies in the Amazon, we used the 2010 timber production data for Peru and standard conversion factors of timber products to roundwood (Ramirez-Arroyo et al. 2011). Using this data, we obtained a transformation efficiency of 38 %, meaning that 62 % of log harvested biomass was transformation waste. This value for transformation efficiency is within the range of other reported values in the Amazon, varying from 33 to 41 % (Blanc et al. 2009; Pereira et al. 2010). Thus, emissions from timber processing were estimated assuming that transformation efficiency for larger harvested logs (≥10 cm dbh) and smaller harvested logs (5 to < 10 cm

dbh) was 38 %. The exponential decay model was used to estimate change in decomposing biomass, where transformation waste was assumed to be burned or decomposed in the first year ($K = 1$ year^{-1}) (Blanc et al. 2009).

Net carbon flux (NCF)

The NCF (Mg C ha^{-1}) is the cumulative sum of carbon gains or removals, described as negative fluxes (−), and carbon losses or emissions, described as positive fluxes (+) (Penman et al. 2003). Total carbon emissions consisted of emissions from tree mortality, timber harvested and timber processed. NCFs were estimated for each subplot during the 15 year-period. Since we did not have data on tree mortality for all 20 subplots in the strips, but only for the demography subplots, total carbon emissions for the remaining subplots were estimated indirectly. Using the demography subplots, we obtained a conversion factor by adding carbon emissions from timber harvested and emissions from timber processed, and dividing the sum by total carbon emissions. In strip 1, the conversion factors for thinned and unthinned subplots were 0.88 and 0.9, respectively. In strip 2, the conversion factors for thinned and unthinned subplots in the clear-cut portion were 0.77 and 0.93, and for thinned and unthinned subplots in the deferment-cut portion were 0.78 and 0.98, respectively (Fig. 1).

Data analysis

In this study, we analysed carbon dynamics on each strip separately, using subplots as replicates. In strip 1, a one-way ANOVA was used to test the effect of thinning on total carbon sequestration, total carbon emissions and NCF; thinning was a fixed factor and subplots were replicates. In strip 2, we used a two-way ANOVA with two fixed factors, thinning and felling treatment (clear-cut versus deferment-cut), and their interaction.

Total carbon emissions of each strip were transformed to the reciprocal to meet the normality assumption of ANOVA. All statistical analyses were performed using the statistical program R version 3.1.1 (R Core Team 2014) and RStudio (2014) version 0.98.1062, with $a = 0.05$. An important caveat in this study is that subplots within each strip are not independent.

We also estimated the time required for a clear-cut stand to reach old-growth AGB (trees > 10 cm dbh). We pooled together biomass data from strip 1 and the clear-cut portion of strip 2 from the post-clearing censuses starting from year 0 to year 15. Since the CIJH site has a history of selective logging, we obtained biomass data for old-growth stands (mean = 282 Mg ha^{-1}) from the Iquitos region (Baker et al. 2004), assumed to be about 300 years-old (Vieira et al. 2005). The Chapman-Richards function, a non-linear regression, was fit to the biomass data (Eq. 10):

$$A(t) = A_{max} * (1 - e^{-b_1 * t})^{b_2}, \quad (10)$$

where A is biomass at year t (= 0, 3, 7, 11 and 15), A_{max} is the maximum biomass, b_1 and b_2 are coefficients.

Results

Carbon stocks in the pre-clearing period ranged between 128 and 140 Mg C ha^{-1} (Fig. 2). In the pre-clearing period, carbon stocks in strip 1 were slightly lower than in strip 2 (Fig. 2), because they were estimated using a greater threshold for tree size, > 7.5 cm dbh, instead of ≥5 cm dbh. After clearing, carbon stocks in strip 1 and the clear-cut portion of strip 2 declined to 0 and 1.5 Mg C ha^{-1}. In the clear-cut portion of strip 2, two trees ≥19 cm dbh were left uncut after clearing, but they either fell or broke a year later. Carbon stocks in the deferment-cut portion of strip 2 declined to 18 Mg C ha^{-1}, as part of the felling treatment (Fig. 2). Fifteen years after clearing, strip 1 recovered about 50 % (64 Mg C ha^{-1}) of its pre-clearing carbon stocks, and the clear-cut portion of strip 2 recovered 39 % (53 Mg C ha^{-1}). The deferment-cut portion of strip 2 recovered 45 % (63 Mg C ha^{-1}) of the pre-clearing carbon stocks.

Carbon dynamics after strip clear-cutting

During most of the 15-year period carbon sequestration rates from tree growth, consisting of established recruits, advanced regeneration and sprouting stumps, were

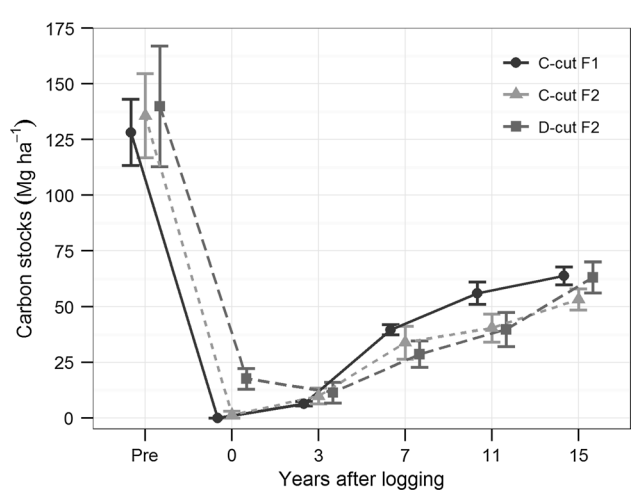

Fig. 2 Carbon stocks (±SE) before clearing (Pre), after clearing (year = 0) and over a 15-year period after clearing in strip 1 (C-cut F1), the clear-cut (C-cut F2) and deferment-cut portions of strip 2 (D-cut F2)

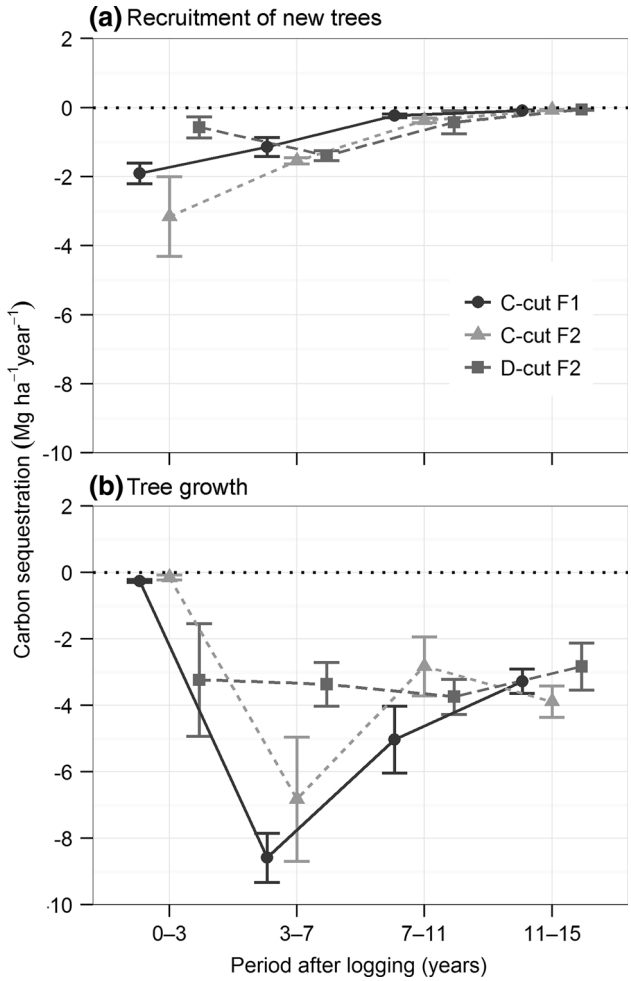

Fig. 3 Carbon sequestration (±SE) (**a**) from tree recruitment and (**b**) from tree growth, over a 15-year period after clearing in strip 1 (C-cut F1), the clear-cut (C-cut F2), and deferment-cut portions (D-Cut F2) of strip 2

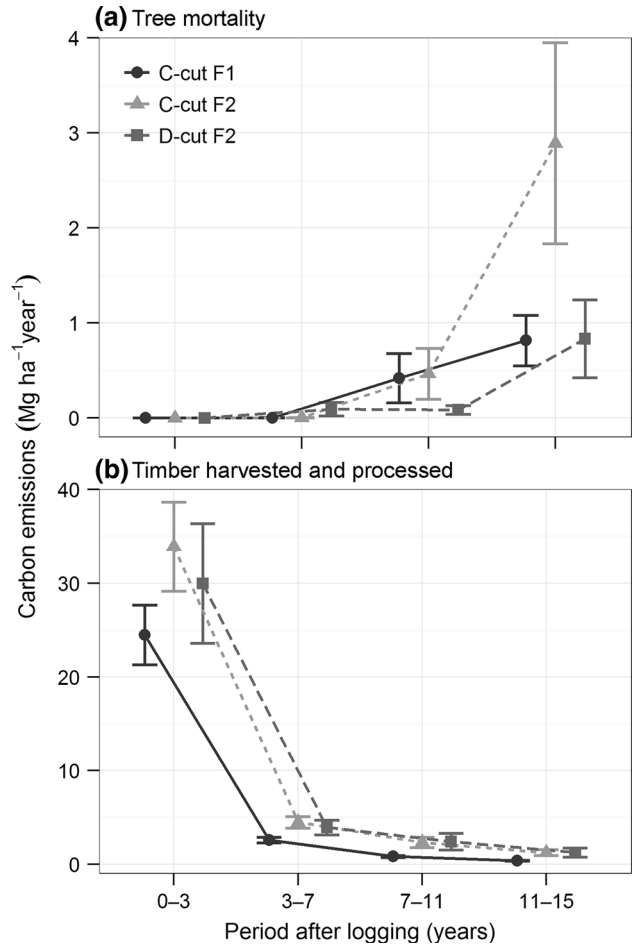

Fig. 4 Carbon emissions (±SE) (**a**) from tree mortality (**b**) and from timber harvested and processed, over a 15-year period after clearing in strip 1 (C-cut F1), the clear-cut (C-cut F2), and deferment-cut (D-cut F2) portions of strip 2

higher than sequestration rates from recruitment of new trees, reaching >[−] 6 Mg C ha^{-1} year^{-1} (Fig. 3). Nevertheless, three years after clearing sequestration rates from recruitment of new trees were higher than those from tree growth in both strip 1 (about [−] 2 vs. [−] 0.3 Mg C ha^{-1} year^{-1}) and clear-cut portion of strip 2 (about [−] 3 vs.[−] 0.1 Mg C ha^{-1} year^{-1}). Sequestration rates from recruitment of new trees declined to [−] 0.1 Mg C ha^{-1} year^{-1} in both strip 1 and the clear-cut portion of strip 2, 15 years after clearing (Fig. 3). In the deferment-cut portion of strip 2, carbon sequestration from tree growth remained at about [−] 3 Mg C ha^{-1} year^{-1} and was higher than recruitment of new trees (Fig. 3).

Although tree mortality produced low carbon emissions, harvesting and processing of timber produced high carbon emissions. Carbon emissions from timber harvested (stumps and FWB) and timber processed ranged between [+] 24 to 34 Mg C ha^{-1} year^{-1} in strip 1 and in the clear-cut portion of strip 2, whereas carbon emissions from tree mortality ranged [+] 1 to 3 Mg C ha^{-1} year $^{-1}$, peaking 15 years after clearing (Fig. 4). In the deferment-cut portion of strip 2, timber harvested and timber processed produced carbon emissions >[+] 30 Mg C ha^{-1} year^{-1}, but tree mortality produced carbon emissions of about [+] 1 Mg C ha^{-1} year^{-1} (Fig. 4).

The effect of thinning and felling on carbon dynamics

Total carbon sequestration was not affected by thinning in strips 1 ($F = 1.217$, $P = 0.286$) and 2 ($F = 0.269$, $P = 0.611$), nor by felling in strip 2 ($F = 0.781$, $P = 0.390$), but it was affected by the interaction of thinning and felling in strip 2 ($F = 5.30$, $P = 0.035$) (Fig. 5). While thinning increased total carbon sequestration in the clear-cut portion of strip 2, thinning reduced carbon sequestration in the deferment-cut portion (Fig. 5).

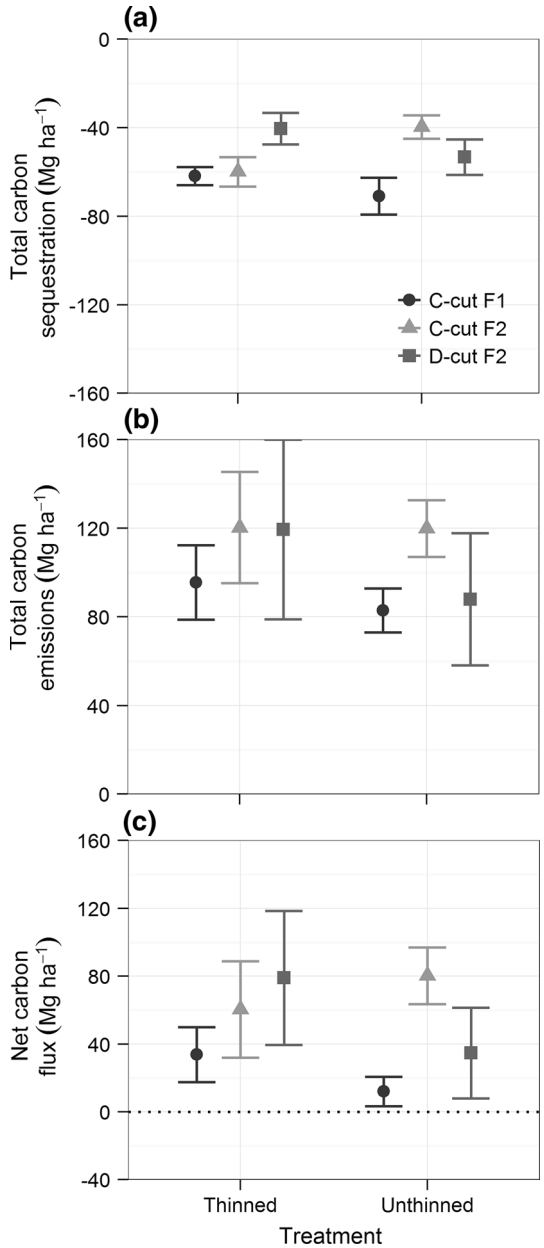

Fig. 5 Total carbon (±SE) (**a**) sequestration, (**b**) emissions and (**c**) net carbon flux over a 15-year period in thinned and unthinned subplots in strip 1 (C-cut F1), the clear-cut (C-cut F2), and deferment-cut (D-cut F2) portions of strip 2

Total carbon emissions in strip 1 and in the clear-cut portion of strip 2 were [+] 91 and [+] 120 Mg C ha^{-1}, whereas in deferment-cut portion of strip 2 were [+] 100 Mg C ha^{-1}. The reciprocal of total carbon emissions was not affected by thinning in strips 1 (F = 0.0155, P = 0.902) and 2 (F = 0.0752, P = 0.787), nor by felling in strip 2 (F = 2.388, P = 0.1418), nor by the interaction of thinning and felling in strip 2 (F = 0.008, P = 0.928) (Fig. 5).

Overall, total carbon emissions (+) were greater than total carbon sequestration (−) in both strips. Net carbon fluxes (NCFs) in strip 1 and in the clear-cut portion of strip 2 were +26 and +68 Mg C ha^{-1}. NCF in the deferment-cut portion of strip 2 was +61 Mg C ha^{-1}. NCFs were not affected by thinning in strips 1 (F = 0.934, P = 0.348) and 2 (F = 0.138, P = 0.715), nor by felling in strip 2 (F = 0.048, P = 0.839), nor by the interaction of thinning and felling in strip 2 (F = 0.95, P = 0.344) (Fig. 5).

The time for a clear-cut stand to reach old-growth biomass (282 Mg ha^{-1}) or carbon stocks of 141 Mg C ha^{-1} was estimated to be about 148 years. The parameters for the Chapman-Richards function fitted to the biomass data were as follows:

$$A(t) = 282 * (1 - e^{-0.045*t})^{1.264},$$

with SE of 0.01 for b_1 = 0.045, SE of 0.27 for b_2 = 1.264, and adj. r^2 = 0.89 (Figure S2 of SM).

Discussion

Aboveground biomass (AGB) and carbon sequestration rates found in the strips were within the range of reported values for the region (Table 1). In the pre-clearing period, AGB for trees >10 cm dbh in the strips was comparable to values reported for sites in Loreto, Peru (Baker et al. 2004; Baraloto et al. 2011), but higher than values of Bajo Calima in western Colombia (Table 1).

After 11 years of regeneration, the AGB of the strips ranged between 71 and 101 Mg ha^{-1} and were comparable to the AGB of 12-year old stands (67–93 Mg ha^{-1}) from Bajo Calima (Table 1). The successional plots from Bajo Calima have a land-use history of clear-cutting for pulp (Faber-Langendoen 1992). Moreover, during the 15-year period carbon sequestration rates in the strips (Fig. 3) were also comparable to those of secondary forests in early succession ([−] 2–10 Mg C ha^{-1} year^{-1}) (Hughes et al. 1999; Fehse et al. 2002), but were higher than selectively logged forest ([−] 2 to >4 Mg C ha^{-1} year^{-1}) (Blanc et al. 2009).

In this system recruitment plays an important role in carbon sequestration. After 15 years of regeneration, recruits (established and new) comprised 81 and 76 % of the total tree regeneration in strips 1 and 2, whereas advanced regeneration comprised 16 and 18 %, and stump sprouts comprised 3 and 6 % (Rondon et al. 2009a). Due to high recruitment rates of new trees three years after clearing (Table S1 of SM), recruitment of new trees made a greater contribution to aboveground carbon sequestration than tree growth from advanced regeneration and sprouting stumps (Fig. 3). However, seven years after clearing, recruitment rates of new trees began to decline and it has continued for more than seven years (Table S1 of SM), as in other logged forests (Silva et al. 1995; de Oliveira 2005; Blanc et al. 2009). When recruitment of new trees declined, tree growth made a greater contribution to carbon sequestration (Fig. 3). Nevertheless, the growth of established recruits, mostly pioneer species of low commercial value and high

Table 1 Aboveground biomass (AGB) for trees >10 cm dbh, from two clear-cut strips (C-cut F1, C-cut F2) and a deferment cut treatment (D-cut F2) in Jenaro Herrera, Loreto, Peru, before logging, and 15 years after logging; from unlogged forests in northwestern Amazonian sites in Loreto, Peru; and from unlogged forests (Front) and logged forests (Succ. Front) of different ages in Bajo Calima, western Colombia

Jenaro Herrera, Loreto, Peru (this study)				Three NW Amazonian sites, Loreto, Peru*	Bajo Calima, western Colombia**		
AGB (Mg ha^{-1})	C-cut F1	C-cut F2	D-cut F2		AGB (Mg ha^{-1})	Front 1 plots	Front 2 plots
Unlogged Pre-logging	256.4	263.9	275.2	241.0–312.8	Unlogged	235.9	188.7

Years after clearing	C-cut F1	C-cut F2	D-cut F2	Age of forest (years)	Succ. Front 1 plots	Succ. Front 2 plots
0	0.0	2.9	34.2	0.4	7.8	6.6
3	3.4	9.5	17.9	4	7.5	7.9
7	67.6	49.2	43.4	8	29.0	61.3
11	100.7	70.9	70.7	12	66.7	93.1
15	118.0	97.6	119.6	18	63.7	No data

Baker et al. (2004)*: Allpahuayo, Sucusari and Yanamono
Faber-Langendoen (1992)**

basal areas (e.g. *Cecropia* spp. *Alchornea* sp. and trees from the Melastomataceae family), remained dominant 15 years after clearing (Rondon et al. 2009a).

Tree mortality in strip 1 and in the clear-cut portion of strip 2 released low carbon emissions, up to [+] 3 Mg C ha^{-1} year^{-1}. Yet Blanc et al. (2009) found higher emission rates from tree mortality ([+] 1–10 Mg C ha^{-1} year^{-1}) under selective logging in French Guiana. These differences occur because in the strips most of the dead trees were small in size (≤ 26 cm dbh) and their average WSG was 0.546 g cm^{-3} (Figure S1 of SM), within the lower range for north western Amazonia WSG values but lower than those for central and eastern Amazonia (>0.65 g cm^{-3}) (Baker et al. 2004), which results in trees having low biomass, low carbon content, and high decomposition rates (Figure S1 of SM). Despite low emission rates from tree mortality in the strips, emissions tended to increase over the 15-year period (Fig. 4) due to higher mortality rates 7–15 years (3–16 % year^{-1}) after the clearing than to previous years (2–4 % year^{-1}) (Table S1 of SM). Although a thinning treatment, applied seven years after clearing, did not significantly affect either tree recruitment or survival (Dolanc et al. 2003), it increased mortality rates (Table S1 of SM).

Harvesting (stumps and FWB) and processing of timber produced high carbon emission rates (> [+] 20 Mg C ha^{-1} year^{-1}) in both strip 1 and the clear-cut portion of strip 2. Our estimates of emission rates from timber harvested and processed were higher than those from selective logging, > [+] 10 Mg C ha^{-1} year^{-1} (Blanc et al. 2009), presumably due to the higher harvest intensity of strip clear-cutting. Compared to selective logging, the amount of timber removed in the clear-cut strips was 102–109 trees ha^{-1}, which totaled 166–184 m^3 ha^{-1} of roundwood (Cornejo and Gorchov 1993), a much higher volume than the volume typically removed from selective logging, which is about 19–40 m^3 ha^{-1} (Nepstad et al. 1999).

In the 15-year period, carbon emissions exceeded carbon sequestration in both strip 1 and the clear-cut portion of strip 2, resulting in positive net carbon fluxes (NCFs) (Fig. 5). NCFs in strip 1 and the clear-cut portion of strip 2 ranged between +26 and +68 Mg C ha^{-1}, and were greater than the NCFs of selective logged plots (+14 Mg C ha^{-1}) and selective logged plots with timber stand improvement (+57 Mg C ha^{-1}) over a 20-year period (Blanc et al. 2009). Blanc et al. (2009) found that application of a timber stand improvement treatment, where non-commercial species were girdled, increased carbon sequestration, carbon emissions, and NCFs. In the strips, however, silvicultural thinning did not have an effect on carbon dynamics (Fig. 5), but significantly increased tree growth rates of commercial species (Dolanc et al. 2003), and significantly reduced the relative abundance of pioneer species (Rondon et al. 2009a).

Carbon dynamics in the deferment-cut portion of strip 2 were comparable to strip 1 and the clear-cut portion of strip 2 (Figs. 2, 3, 4, 5), but felling treatments differed in tree species composition. Deferment-cut (where commercial trees <30 cm dbh were left uncut) significantly reduced the relative abundance of pioneer species and significantly increased the relative abundance of commercial species relative to the clear-cut portion of strip 2 (Rondon et al. 2009a). Although thinning in the deferment-cut tended to reduce relative abundance of pioneers (Rondon et al. 2009a), thinning tended to increase carbon emissions and NCFs (Fig. 5), but in neither case was this significant.

Since in this study we only examined part of the necromass, it is very likely that carbon emissions in this system might be even higher. In this study we only quantified stump biomass and fine woody biomass produced after timber harvesting, as well as biomass from tree mortality produced during regeneration. We did not quantify fallen woody debris during regeneration, which is typically estimated with line-intercept

sampling (Keller et al. 2004). Furthermore, we did not quantify the pre-clearing necromass to make comparisons with the post-clearing. It is well known that logged forests have greater amounts of CWD than unlogged forest (Gerwing 2002; Keller et al. 2004); and as logging damage increases, biomass in the necromass also increases (Pinard and Putz 1996), which eventually decays resulting in carbon emissions.

Sustainability of strip clear-cutting

Our results indicate that applying strip clear-cutting for timber extraction has the potential to impact the carbon balance of the surrounding forest landscape, due to an increase in carbon emissions. Old-growth forests in the Amazon are considered carbon sinks, because growth on average exceeds mortality (Phillips et al. 2008), but this is not the case for the 15 year-old clear-cut strips, where carbon sequestration has not offset yet carbon emissions, mainly from harvesting and processing of timber. As in other secondary forests (Brown and Lugo 1990), aboveground biomass accumulated rapidly in the clear-cut strips during the first 15 years (Figure S2 of SM); nevertheless, tree mortality and growth have not reached an equilibrium yet (Figs. 3, 4). Pioneer species that established early in succession have dominated composition of the strips for years and will likely die off within <25 years (Denslow and Guzman 2000), releasing more carbon emissions. Our results also indicate that the strips will require >140 years to reach old-growth carbon stocks, and thus offset carbon emissions (Figure S2 of SM), similarly to other secondary forests in the Amazon (100–190 years) (Saldarriaga et al. 1988; Fearnside and Guimaraes 1996), but many more years than selectively logged forests (45–100 years) (Blanc et al. 2009; Huang and Asner 2010; Mazzei et al. 2010).

Nevertheless, this system could qualify for payments for ecosystem services (PES) that allow for timber harvesting. For instance, under the REDD+ (Reducing Emissions from Deforestation and forest Degradation and enhancing forest carbon stocks) framework forest owners and users are paid to reduce emissions and increase carbon sequestration (Angelsen et al. 2009). REDD+ incentivizes forest users and owners to conserve and manage forests sustainably to deter them from investing in other land-uses such as cattle ranching and agriculture (Angelsen et al. 2009). A PES framework such as REDD+ may benefit strip clear-cutting, whose financial viability is a major concern (Rondon et al. 2010).

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