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Changes in ecosystem carbon stocks from the conversion of disturbed forest to oil palm plantation in Ucayali, Peru

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ABSTRACT

While Peru has pledged to reduce greenhouse gas emissions, oil palm plantations are currently under expansion in the Peruvian Amazon at the risk of forest conversion. This study aimed to characterize the structure and composition of remnant disturbed forests adjacent to oil palm plantations in the Peruvian Amazon region of Ucayali, to further determine the carbon stock loss/gain from such transition. The thesis was based on data collected by CIFOR in a 2015 field campaign, which included four forest plots and six oil palm stands. The latter ones using a space-for-time substitution approach to assess the carbon stock change over a rotation period. All carbon pools were analyzed and compared over the land use transition. In the case of oil palm plantations, per pool time-averaged carbon stocks were developed from estimated growth models in a 30 year-old rotation.

Forest composition and structure analysis at the study site evidenced past anthropogenic disturbance, probably due to previous logging activities. Species characterization did not reveal deterministic differences between plots and all presented a certain level of species homogenization. Previous forest practices showed a negative effect in composition and structural parameters, the mean basal area was found to be 22 (SE 1.4) m^2 ha^{-1}. Overall, the conversion from disturbed forests to oil palm plantations resulted in a carbon debt scenario, as forests presented a total carbon stock of 140.7 (SE 5.8) Mg C ha^{-1} and the time-averaged carbon stock of oil palm plantation 74.3 (SE 2.2) Mg C ha^{-1}. Above ground carbon was the main contributing pool; followed by soil organic carbon and necromass.

Remnant disturbed forests at the study site are at systematic risk of conversion. This research contributes to the current land planning discussions on where to settle new areas for oil palm production. In terms of carbon footprint, logged forest and secondary forests should be excluded from the scope. Instead, conversion should be directed to highly degraded lands such as pastures and shrubs, where further studies are still needed.

Key words

Carbon stock, land use change, forest structure, forest composition, disturbed forest, oil palm plantation

ACRONYMS

AIC	Akaike Information Criterion
AGB	Aboveground biomass
AGC	Aboveground carbon
Ab	Abundance
BA	Basal Area
CIFOR	Center for International Forestry Research
CWD	Coarse woody debris
DBH	Diameter at breast height
DW	Dead wood
dm	Dry matter
Fr	Frequency
IPCC	Intergovernmental Panel on Climate Change
IVI	Species Importance Value Index
LULUCF	Land Use, Land Use Change and Forestry
MRV	Monitoring, Reporting, Verification
MINAM	Peruvian Ministry of Environment (in its Spanish acronym)
MINAGRI	Peruvian Ministry of Agriculture and Irrigation (in its Spanish acronym)
NSP	Nuevo San Pedro hamlet
SCS	Soil carbon stock
SE	Standard error
TAL	Tupac Amaru Limon hamlet
UFCCC	United Nations Framework Convention on Climate Change

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I. Introduction

1.1 Background and problem statement

In 2015 Peru submitted its Indented Nationally Determined Contributions (INDC) to the UNFCCC pledging to reduce 30% of greenhouse emissions based on a business as usual scenario by 2030 (PERU, 2015). Peru had already been working towards climate change mitigations and adaptation actions through international schemes such as Reducing Emissions from Deforestations and forest Degradation (REDD) and Nationally Appropriate Mitigation Actions (NAMAs). According to Peru´s first Biennial Update Report, in 2010 deforestation and degradation represented around 35% of national greenhouse gas emissions (MINAM, 2014). Even though the forest sector is the biggest contributor to national greenhouse gas emissions, it also offers a giant opportunity for climate change mitigation and adaptation action through carbon removals from forest restoration, conservation of natural primary forest and sustainable forest management. Therefore, in its recent climate change law and national strategy on forest and climate change, Peru acknowledges the potential of the land use, land use change and forestry (LULUCF) sector to achieve its mitigation and adaptation climate goals (MINAM, 2016).

In order to assure compliance with such commitments, the monitoring, reporting and verification (MRV) of emissions and removals coming from LULUC still need to be improved. Currently, national reports still use Intergovernmental Panel on Climate Change (IPCC) Tier 1 default values for all land use types except for primary forests (MINAM, 2014). Moreover, despite constant anthropogenic pressures on Peruvian forests, limited research has been done on structure and composition of disturbed forests after logging activities; neither has a full carbon stock assessment on permanent crops been done. This research aims to support national carbon accountability by providing sound and credible estimates of emissions associated with the transition of disturbed forests to oil palm plantations and its associated emissions to guide decision makers for its sustainable development. Disturbed forest here refers to the mosaic of anthropogenic intervened forests as found in the region of Ucayali, which include residual forests resulting from past logging activities and secondary forests re-growing after long fallows (Smith, 1999).

This study is of great relevance for the region of Ucayali and areas surrounding the city of Pucallpa, as both the national and regional government are encouraging oil palm expansion (GRE Ucayali, 2016a; MINAGRI, 2016). Nevertheless, little research has been conducted on oil palm plantations in Peru, despite the need to understand to what extent this cash crop can be developed sustainably at landscape level. For instance, in order to address the environmental impacts of oil palm plantations related to climate change it is important to investigate if they are a source or a sink of carbon, compared to other vegetation types. Most research on oil palm sustainability aspects has been carried out in Southeast Asia, where the crop has mainly expanded (Hansen et al., 2015). However, as international demand continues to grow, the South American continent needs to address the sustainable development of this crop to prevent risky destructive experiences as happened in Indonesia or Malaysia. As a matter of fact, Peru's stakeholders have ruled for developing the crop on degraded lands, avoiding primary forests, although such areas have not been defined yet (GRE Ucayali, 2016a; MINAGRI, 2016. This study aims to provide information that supports its delimitation.

The present research is part of Module 3 of the Global Comparative Study on REDD+ led by the Center for International Forestry Research (CIFOR). CIFOR is working on creating better LULUCF information to improve accuracy on mitigation information and strategies to climate change. As part of this project, CIFOR established inventory plots in oil palm plantations and disturbed forests in June 2015. This master thesis has focused on the analysis of these inventory data and complemented it with additional site visits.

1.2 Research questions and objectives

The main research question of this study is how much carbon is lost from forest disturbance and ultimately conversion to oil palm plantations in the Peruvian Amazon. To address this question, two main objectives have been identified with additional sub research questions (SRQ) and respective hypothesis (H).

- Asses the composition, structure and carbon stocks of disturbed forests in the Peruvian Amazon region of Ucayali

  SRQ1: How does degradation affect forest composition and structure?

  H1: The sampled forest in the study site shows evidence of disturbance according to composition and structural characteristics

- Determine the carbon stock changes from the transition of disturbed forests to oil palm plantation

SRQ2: How do carbon stocks in oil palm plantations evolve in time and by pool over one rotation period?

SRQ3: Are there significant difference in carbon stocks between disturbed forest and oil palm plantation?

H2: Disturbed forest have a higher carbon stock than adjacent oil palm plantation

II. State of the art on land use, land use change and carbon stocks

2.1 Land use history of Ucayali and the degradation effects on forest structure and composition

Peru stands among the ten countries with largest forest cover in the world and the second largest in South America (FAO, 2016). Peru´s Amazon forest represents around 56% of its entire territory (MINAM, 2015). It is a mega diverse country and most ecosystem services and biodiversity are associated with the Amazon tropical forest, which represents around 94% of total forest land (ibid). The forests in Peru can be classified according to climatic characteristics (rainforest, sub-humid forests) or geomorphological conditions (terrace, hills, floodplains, mountains, etc.) (ibid). In the region of Ucayali, where the study site is located, forest types include terrace forests (low, medium, high), hill forests (low, medium, high); mountain forests; flooded forests (locally named Aguajal – a *Mauritia flexuosa palm*- dominanted forest); riverside forests and secondary forests (ibid). Along a natural forest altitudinal gradient starting in the floodplains of Ucayali and reaching humid highlands in the region of Pasco, Gamarra et al. (2015) described a decrease in forest species richness and diversity with altitude increase.

During 2001-2014, Peru lost 1,653,129 ha of Amazon forests at a mean rate of 118,080 ha $year^{-1}$ (MINAM, 2016). During this period, 77% of deforestation occurred in areas ≤ 5 ha (MINAM, 2016). Therefore, small-scale migratory agriculture, mostly related with slash and burn practices, is probably the main source of forest loss in Peru, especially since 45% of deforestation occurs in areas with no assigned forest rights associated with unsolved land tenure and forest governance issues (MINAM, 2015). Fires in the Amazon are mostly anthropogenic, as non-degraded moist and wet forest don't easily propagate fire (Baldoceda et al., 2001). Since 2007, industrial crops were identified as additional drivers of deforestation, mainly oil palm and cacao in Loreto and Ucayali Amazon regions; as well as mining happening mainly in Madre de Dios (MINAM, 2016).

Anthropogenic disturbance in Ucayali started in 1943, when the Lima-Pucallpa highway was finished; which triggered the development of secondary roads and the establishment of new settlements in the direction of Tournavista, Nueva Requena and Curimana (Galván et al., 2000). Around 1980, there was an expansion of agricultural and cattle raising activities; first as subsistence farming and later in the 1990s, as cacao and perennial crop production such as fruits, oil palm and coca fields at further distance from the open roads (ibid; Fujisaka and White, 1998). By 1996, Pucallpa settlers had already cleared more than half of their lands leaving a heterogeneous landscape mainly composed of fallows and pastures; with permanent and annual crops in lower proportion (Fujisaka and White, 1998). At the same time, forest logging turned into an intense activity in the area.

Between 2000 and 2016, while natural forest cover decreased from 96,763 ha to 95,601 ha in Ucayali; secondary vegetation and grasslands grew in 15% and 53% respectively (Geobosques, 2018). Most recently, forest cover has mostly been converted to croplands and grasslands, whereas secondary vegetation was the result of agricultural and grassland fallows (Geobosques, 2018). In slash and burn agriculture, the main reason behind regeneration is soil recuperation. According to a study conducted in Ucayali, fallow periods are usually short but can be extended when agricultural productivity declines or when income is not sufficient to prepare the upcoming agricultural rotation (Smith, 1999). In fact, the study found that only 13% of the secondary forest were meant for full recovery and forest production, while the rest was part of the agricultural rotation system (ibid). Even though commercial use of secondary forest is undervalued; there are still uses the local population can draw upon forest functions such as wood for local construction, low value sawn wood and food provision (Galván et al., 2000). If ecosystem services were additionally accounted for, remnant and secondary forests could be a more competitive land use. The pressure over remnants or residual forests is reduced in improved short fallows sites with available secondary forest (Smith, 1999). However, too short fallows could only sustain annual cropping which drives further forest degradation (Lojka et al., 2011).

Although past forest management plans where inaccessible during this research to improve understanding over past land use intensity among forest concessions; forest statistical yearbooks revealed that Ucayali's wood quotas have constantly been among the highest in the country. Back in 1998, there were 184 forest enterprises and 50 additional permits for forest production in Ucayali (MINAGRI, 1999). Among Amazon regions, Ucayali had the highest round wood (293,757.60 m^3) and sawn timber production (148,256.46 m^3) in the country. The most harvested species in the region were *Cedrelinga catenaeformis, Hura crepitans, Cedrela odorata, Chorisia sp., Virola sp., Iryanthera sp., Copaifera sp.* (ibid).

Many studies focus on the effects of different types of disturbances on the forest's structure and composition. A study in African semi-deciduous forests found, through a comparison between unlogged stands and 18-year-old post-harvested stands, that although selective logging did not have a significant effect on tree species composition, it altered forest structure in terms of recruitment by reducing sapling and tree stem densities (Hall et al., 2003). On the other hand, a study in the tropical forests of Central Kalimantan found that even a 55-year-old succession was not enough for a secondary forest to reproduce primary forest species composition and diversity; though forest structural characteristics (tree basal area, height and biomass) might bear a resemblance (Brearley et al., 2004). Another study on the long-term effects of logging in the mountain rainforest of Madagascar stated that after logging and subsistence agriculture there was a proliferation of non-native or invasive species that affected the local diversity and forest structure (Brown and Gurevitch, 2004).

There is no common agreement on how forest structure and species composition changes as a consequence of logging in tropical forests; therefore, it is important to have a case-by-case examination. Jakovac et al. (2016) ran an analysis on Brazilian Amazonian secondary forests to test which indicators were more deterministic on succession species composition and defined that land use history was crucial. Land use intensity and management practices played a stronger role in secondary forest composition than landscape characteristics, which include, for instance, the presence of primary forest patches left as seed bank sources (ibid). The more intense the management of the forest, the more likely that the diversity decreases and early successional communities to be more similar to each other; regardless of their initial composition state (Jakovac et al., 2016).

Ucayali's lowland forests have an overall rapid regeneration rate after slash and burn activities in comparison with poorer soils site's in the Amazon, even though floristic composition is still distinctive to primary forest (Tournon and Riva, 2001). Similar to Jakovac et al. (2016), Alva Vásquez & Lombardi (2000) reported that among the most important factors that influence forest succession in terms of composition and productivity in Ucayali were soil fertility, previous land use intensity, productive activities in the surroundings and approachability to primary forest patches. For instance, secondary forests along the Federico Basadre Highway near to the city of Pucallpa have lower tree density in small DBH classes in comparison to areas located further from the highway (Kommeter, 1987). This is due to the fact that areas closest to Pucallpa city have been longer and more intensively disturbed, have poorer soils and less primary forests in the landscape matrix as seed banks (ibid).

2.2 Oil palm expansion in the Peruvian Amazon

Palm oil represents around 35% of worldwide vegetable oil, followed by soy oil (28.7%) (MINAGRI, 2016). Oil palm plantations are mainly concentrated in Southeast Asia, with Indonesia and Malaysia accounting for 86% of overall palm oil production (Potter, 2015). In South America, Colombia was the pioneer, which together with Ecuador and Brazil complete 3% of the world's crop cover (MINAGRI, 2016). Oil palm agro ecological demands narrow its niche to tropical countries with high precipitations regimes and average temperatures of around 24-28°C (Pirker et al., 2016). While lands in Southeast Asia become scarce, governments in Africa and South America are strongly promoting palm oil production as a way to reduce poverty, produce biofuel and supply food (ibid). In Peru, by 2014 palm oil production represented 19% of the national value; still national supply was not enough to fulfill the demand and 96% of imports came from Indonesia (Borasino, 2016). By 2015, palm oil covered 77,537 ha of the territory, with San Martin and Ucayali among the most important regions. Considering the growth in Ucayali alone, while in 2008 the expansion of oil palm was of 5,000 ha, by 2016 it covered already 35,000 ha; which means a 15,000 ha increase every 4 years (GRE Ucayali, 2016a). The regional plan is to reach 60,000 ha by 2026 (ibid).

Oil palm is a largely controversial crop. Its accelerated expansion is supported by its high yield performance per ha and low price in comparison to other oil seeds (e.g. soy, rapeseed); if well planned, it could reduce shifting cultivation over forested areas (MINAGRI, 2016; Pirker et al., 2016). However, its negative reputation is inherited mainly from experiences in Southeast Asia where its imminent development has been at expense of forests on mineral and peat soils while declining biodiversity, emitting highly significant greenhouse gasses and polluting soils and streams (Potter, 2015). In South America, oil palm plantations in large monocultures have also shown to drastically change mammal species composition in Colombian savannah ecosystems (Pardo et al., 2018). Additionally, oil palm expansion has been associated to social and gender issues, including land grabbing in unsolved land tenure rights or aggravating inequality (Potter, 2015). In Ucayali, oil palm has been reported to increase farmers' livelihoods to obtain a net monthly income of about 360-900 USD in 5ha production areas, but at the expense of soil pollution from indiscriminate use of chemical fertilizers (ibid).

In Peru, oil palm has long been considered a crop of national interest; therefore, taxation incentives were provided to farmers with the purpose to tackle the economic and ecological effects of shifting cultivations and eradicating illicit crops (Dammert, 2016). The 2000-2010 national plan for the promotion of oil palm stated that the crop should be grown in deforested or degraded lands, however, there was no tool or legal framework that specified where these potential areas were located (Barrantes et al., 2016). On the other hand, a biofuel law also triggered Peru's palm oil production. The intentions of this law was to regulate the consumption of ethanol and biofuel in gasoline and petrol as an effort to reduce greenhouse emissions from the car park while giving an opportunity to agro industrial crops, promote employment and replace illicit crops (Dammert, 2016). Nevertheless, it does not protect biofuel local production and has ended up being a perverse incentive through which biofuel has been imported from other regional countries with lower subsidies prices, mainly from Argentina (Dammert, 2016). Nowadays, almost 100% of palm oil produced in Peru is engaged in food supplies; the biodiesel mill owned by one of the most important industrial stakeholders is currently inoperative (Borasino, 2016).

Palm oil production was historically driven by private small-scale investors who organized themselves into associations (e.g. COCEPU in Ucayali) and were initially financially helped by international programs fighting illicit farming (Barrantes et al., 2016). Compared to other crops, it seems that oil palm farmers have greater economic conditions (in terms of income and assets) and, by being organized in associations, have better access to services (Borasino, 2016). A national financial group (Romero Group) is also an important industrial stakeholder; they now own 40% of total productive area and run two processing plants that account for 68% of total palm oil production in the country (Barrantes et al., 2016). However, in Ucayali, small-medium holders still dominate the market, COCEPU has 657 members and 5,632 planted hectares in the region (Potter, 2015).

Oil palm yield depends on planting materials, age of the palm, fertilizer application, harvesting and maintenance regimes and climatic factors (Potter, 2015). Management techniques differ between industrial and small-holder plantations, which result not only in different yields, but also in different greenhouse gas footprints (Chase and Henson, 2010; Khasanah et al., 2015a; Henson, 2017). In the district of Campo Verde, in Ucayali, for instance, 53% of smallholders do not use inputs such as fertilizers and pesticide (Potter, 2015). Peruvian smallholders are usually challenged with less access to technology, loans and technical assistance; which translates to lower yields and unsustainable practices (MINAGRI, 2016). In Malaysia, better climatic conditions combined with breeding techniques and improved fertilization have led to higher yields; which could be a way to decrease pressure on forested areas (Potter, 2015). Nevertheless, frontier expansion in Indonesia forests has been associated, not to smallholders (<3ha), but to absent farmers that engage in oil palm for investment purposes (Schoneveld et al., 2017).

While Indonesia's oil palm production was on the world spotlight for environmental catastrophe, it triggered a movement towards sustainability through the creation of standards and certifications (e.g. Roundtable on Sustainable Palm Oil, International Sustainability and Carbon Certification, Indonesian Sustainable Palm Oil) as well as no deforestation platforms (e.g. High Carbon Stock) (Pirard et al., 2017). There were also corporate sustainability commitments that pledged for zero deforestation and international agreements that fully engaged to abolish deforestation (New York Declaration on forests) (Pirard et al., 2017). After the creation of such initiatives, the challenge to reach sustainability on the ground in Indonesia is now related to weak governance, no law enforcement (e.g. issues like tenure), lack of technical and management capacity building to smallholders to increase productivity and the fragile product traceability system (Pirard et al., 2017; Sharma et al., 2017).

If oil palm production is to continue to grow, the road to sustainability does not necessarily depend on high yield production alone; but also on the adequate delimitation of new suitable lands that respect international and national sustainability commitments. Pirker et al. (2016) defined worldwide suitable lands for oil palm based on biophysical characteristics (climate, soil and topography), land availability, biodiversity and carbon richness. In Peru, similar academic efforts have been made to delimitate potential areas for oil palm expansion (Glave and Vergara, 2016; Barrantes et al., 2016); which have been acknowledged by the national government and partially incorporated into the new sustainable oil palm production plan (2016-2025) (MINAGRI, 2016). The plan, although finalized, has not been officially approved yet. Under its scope, areas for oil palm production include i) land with agricultural purpose; ii) areas without forest cover (primary forest, secondary forest or land under restoration) or degraded; and iii) areas suitable for oil palm production (MINAGRI, 2016). However, much of the spatial studies that would support the planning have not been equally finalized for the entire country. In Ucayali, a competitiveness plan for oil palm production (2016-2026) has been developed by oil palm stakeholders. This plan states that the agricultural frontier will increase without compromising primary forests or impacting the environment (GRE Ucayali, 2016a). Hence, even though there is the political will to avoid deforestation and commit with a sustainable oil palm production; a clearer and more tangible delimitation of areas for expansion still needs to be defined to ensure transparency, facilitate monitoring, and comply with the country's legal framework. Such actions are necessary if the country aims for certification or sustainable production to later access international markets that call for no deforestation and High Carbon Stock (HCS) oil palm production.

2.3 Carbon stocks of Amazon forests and oil palm plantations

Several efforts have been done to determine the biomass distribution in Amazon forests, from ground data extrapolations to remote sensing metric correlations (Asner, 2014; Malhi et al., 2006; Saatchi et al., 2007). Saatchi et al. (2007) found that the northeastern Amazonian region had the highest stocks with Above Ground Biomass (AGB) values between 300-400 Mg ha^{-1} due to low human disturbances, forest structure and a hot and wet climatic regime. Malhi et al. (2006) also found, from an extrapolation exercise from forest plots, that the Central Amazonian and northeast Amazonian presented the highest AGB. Western regions of the Amazon, including the lowlands of Peru, register biomass ranging from 200 to 300 Mg ha^{-1} and data suggests that forests are more dynamic, have higher productivity, and a larger number of medium and smaller size trees than the central and eastern Amazon (Saatchi et al., 2007). AGB of degraded forests and secondary vegetation from different ages in the Amazon fell under the interval of 25-70 Mg ha^{-1} (ibid). According to Saatchi et al. (2007), the natural spatial variation of biomass in the Amazonian region is associated with different vegetation types and climatic variables, such as number of dry months (rainfall < 100mm) during the year or the amount of rainfall in the driest quarter; though temperature shows a poor correlation with biomass. Poorter et al. (2015) also found a close dependence between aboveground biomass and rainfall, structural forest attributes (i.e. tree density and size) and species richness.

In Ucayali, and other lowland regions of western Amazonia, the incision from rivers decreases by 30-50% the vegetation carbon stock in comparison to forests located away from the river basin (Asner, 2014). Forests in large floodplains of Ucayali and Loreto contain 50-80% less AGB carbon stocks than neighboring terra firme forests (ibid). Similarly, over an altitudinal gradient from lowland Ucayali' forests to upper mountain forests in other regions of the country, it was noted that the lowland forest plot contained less aboveground carbon stock (112 Mg C ha^{-1}) than the transitional to pre mountain and pre mountain forests (132 and 122 Mg C ha^{-1} respectively); mainly due to lower wood density and diameters at breast height (DBH) in lowland forests (Gamarra et al., 2015).

Natural forests in the Amazon are net carbon sinks with an overall higher productivity than mortality (Brienen et al., 2015). Carbon stocks are influenced by forest dynamics (recruitments vs. mortality) (Gamarra et al., 2015). Furthermore, based on ground monitoring being done since the 1990's, there is a trend of decreased carbon accumulation associated with increased tree mortality that could be attributed, among other sources, to climate variability (Brienen et al., 2015). When the anthropogenic variable is included, forested lands result in greenhouse gas pollutants. A pantropical analysis based on 12 years of remote sensing data and above ground biomass inventories suggests that tropical forests are a net carbon source; due to the ongoing deforestation and forest degradation, which isn't compensated by carbon sequestration resulting from forest growth (Baccini et al., 2017).

The IPCC guidelines for national inventories propose two generic methods for estimating CO_2 emissions or removals from changes in carbon pools; the stock change method and gain-loss method (GFOI, 2013). The stock change method, which is the one applied in the present study, is based on surveyed data and estimates emissions or removals as the differences in carbon stocks between estimates made in two different periods in time (ibid). A developed forest system could be considered as a fairly constant carbon stock; which in a land use change scenario is interrupted with forest clearance, slash and burn and crop establishment. After an initial large carbon loss, a gradual accumulation follows, which depends on the rate of carbon accumulation of the new land system (Palm et al., 2000). The carbon sequestration potential of a crop would be overestimated if the maximum carbon stored in the system were used, therefore, it is better to compare the time-averaged carbon stocks of land use systems (ibid).

Neither secondary forests nor logged forest are at equilibrium. Along a forest succession, there is an initial rapid carbon accumulation stage which slows downs and tends to stabilize around 40 years after regrowth (Orihuela-Belmonte et al., 2013). Even though age plays an important role on tree carbon accumulation rates; previous land use intensity and site quality are also equally relevant (ibid). Such dynamics in disturbed forests were not part of the scope of this study nor were resources available to establish chronosequence plots on disturbed forests with the same sampling intensity as in oil palm plantations. Therefore, it is assumed that the sampled forest plots at the study site are representative of the remnant disturbed forest mosaic found next to the on growing oil palm plantations.

Furthermore, Khasanah et al. (2015a) followed the concept of assessing carbon neutral or carbon debt free oil palm transitions in Indonesia by comparing carbon stocks preceding conversion to the oil palm time-averaged carbon stocks. The paper studied different carbon pools over 25-year rotation stands under different management systems. On mineral soils, the time-averaged aboveground carbon (AGC) of an oil palm plantation was 42.07 Mg C ha^{-1} in commercial plantations; whereas 37.76 Mg C ha^{-1} among smallholders (ibid). Understory biomass and dead organic matter pools presented much smaller carbon stocks (ibid).

III. Material and Methods

3.1 Study area and site characteristics

The study was conducted in the Campo Verde district, in the Peruvian Amazon region of Ucayali, within the Nuevo San Pedro (NSP) and Tupac Amaru Limon (TAL) hamlets located respectively around 20 and 40 km from Pucallpa city (Figure 1). Very high relative humidity, continuous rain and permanent warm temperatures are characteristics of the study site (GRE Ucayali, 2016b). Annual temperatures remain relativly constant between 24 °C and 26 °C, with an annual mean of 25 °C. Average annual precipitation fluctuates between 1600-2300 mm and there are two to four dry months a year (precipitation < 100 mm month^{-1}) between May and August, according to meteorological stations in Campo Verde (ibid). The study area is located on a terrace elevated 160 and 192 m a.s.l. at TAL and NSP, respectively (CIFOR, unpublished). According to FAO's classification system, both sites are located in acidic Luvisol or Cambisol soil types (pH ranges between 4.5-4.7 or 5.6-7.8), well drained and clay-loam content (GRE Ucayali, 2016c).

A space-for-time substitution approach was employed to assess changes in carbon stocks in oil palm plantations. This extensively used method infers past trajectories from contemporary patterns and assumes that the observed differences can be attributed to land-use change and are not inherent site differences (Blois et al., 2013). Four disturbed forest and six oil palm plantation sites among TAL and NSP were selected. It was beyond the study's scope nor were resources available to perform a similar chronosequence study to capture the disturbed forest succession dynamics. Forest plots were assumed to be representative of the forest dynamics found next to oil palm plantations boundaries on the study site, with similar land use history. Annex 1 and 2 evidence the land use transition at the study site through different year's satellite imagery, from which it was inferred that sampled plots were representative from disturbed forest patches left at the site. Three of the forest sites were located at the side of young oil palm plantations aged one, four and seven-year-old, nearby TAL where oil palm establishment was most recent (Figure 1). The fourth forest site was in the vicinity of the oldest plantations aged fifteen, twenty-three and twenty-eight-years-old, close to NSP. Satellite imagery observations show older forest disturbance and land clearing history for the NSP oil palm plantations plots, whereas TAL cleared plots have been more recently intervened (Annex 1 and 2).

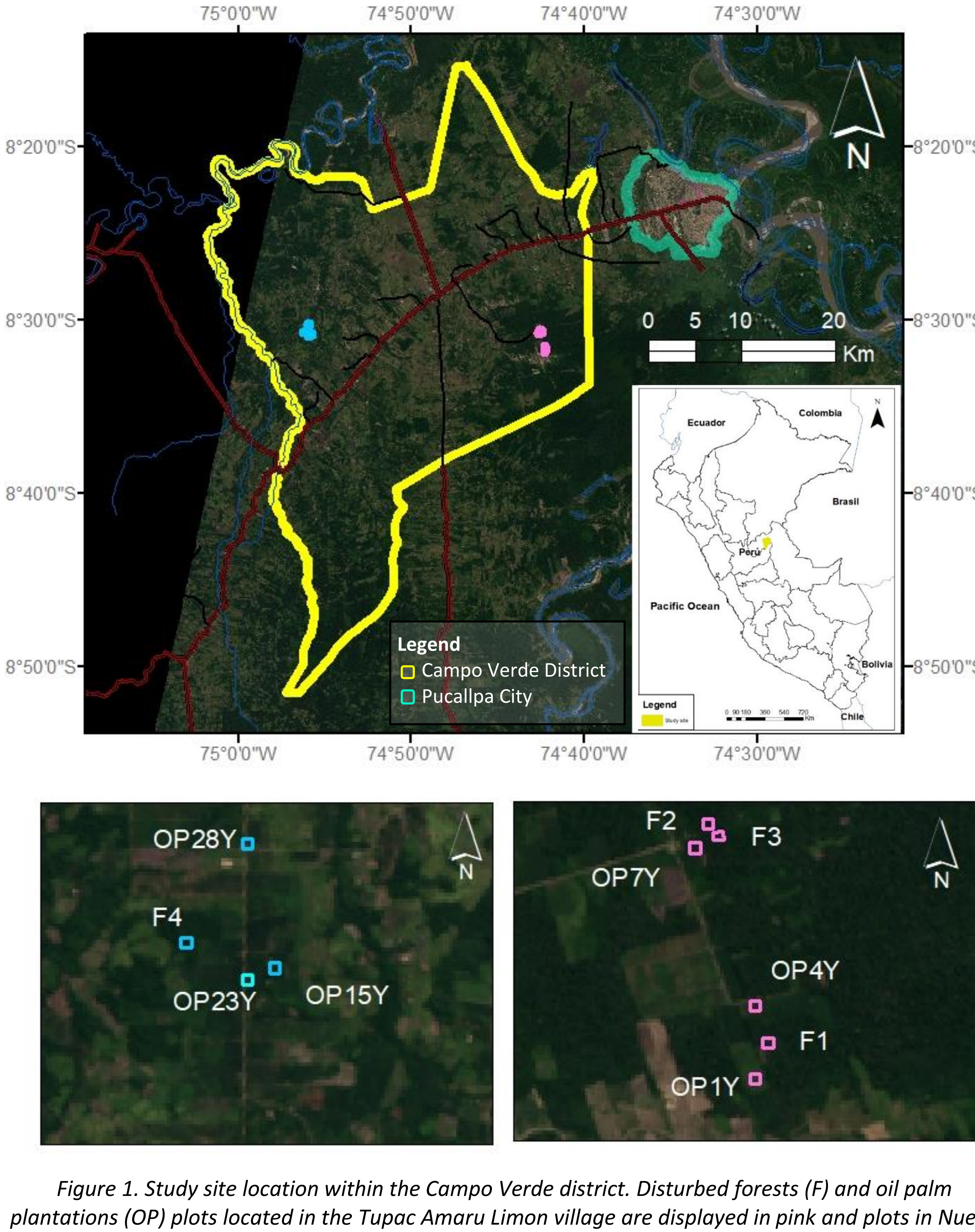

Figure 1. Study site location within the Campo Verde district. Disturbed forests (F) and oil palm plantations (OP) plots located in the Tupac Amaru Limon village are displayed in pink and plots in Nuevo San Pedro in blue. Landsat 8 OLI 2015 satellite image in RGB (432).

The oil palm (*Elaeis guineensis*) plantations were all situated in logged forest and had been either converted directly to plantation or left as fallow, used as coca field or cultivated with a ground cover for some years before being planted. All of them were in their first rotation and most of them underwent at least one land-clearing fire prior to planting (CIFOR, unpublished). Pictures of the sampling sites can be found in Annex 3. Plantations included oil palm ASD (Costa Rica) and CIRAD high yield varieties. Old oil palm plantation stands were exposed to a more intense fertilization regime than younger stands. Although irregularly applied (depending on smallholders practices), the dose was around 1.5-3.5 (kg palm^{-1} year^{-1}) brought in two applications of rock phosphate (24% $P_2 0_3$ + 33% C_3O), urea (46% Nitrogen), potassium chloride (60% K_2O) and compounds of MgO + B + S + K_2O + P_2O in different concentrations. Younger stands (<5 years) received less than 2 kg of each per palm and year.

Table 1. Location, characteristics and land use (LU) history of forest (F) and oil palm plantation (OP) sites sampled in the hamlets of Tupac Amaru Limon (TAL) and Nuevo San Pedro (NSP), nearby Pucallpa. (CIFOR, unpublished).

Plot	Village	Coordinate S°	Coordinate W°	Patch size (ha)	Distance to OP/F edge (m)	Interim LU before OP	Duration interim LU (years)	Land-clearing fires (n)
F1	TAL	08°31'39.6"	74°42'14.4"	4	10	n.a.	n.a.	n.a.
F2	TAL	08°30'40.0"	74°42'30.3"	17	n.a.	n.a.	n.a.	n.a.
F3	TAL	08°30'35.6"	74°42'32.0"	12	n.a.	n.a.	n.a.	n.a.
F4	NSP	08°30'42.0''	74°56'14.0''	2	200	n.a.	n.a.	n.a.
OP1Y	TAL	08°31'51.2"	74°42'18.6"	2	50	Fallow	3	1
OP4Y	TAL	08°31'30.6"	74°42'19.3"	2	20	n.a.	n.a.	1
OP7Y	TAL	08°30'42.9"	74°42'37.1"	6	200	Kudzú[1]	1	1
OP15Y	NSP	08°30'51.4"	74°55'47.5"	2.5	250	Kudzú[1]	n.a.	1
OP23Y	NSP	08°30'55.2"	74°55'54.0"	4	400	Coca	Unknown	Unknown
OP28Y	NSP	08°30'13.5"	74°55'55.0"	25	400	n.a.	n.a.	1

[1] N-fixing leguminous species (*Pueraria* sp.) planted as ground cover. All oil palm plantations were in their first rotation

3.2 Sampling design

The approach used for carbon stock sampling was specifically designed to capture the spatial variability of logged forests, accommodate the regular planting scheme of oil palm plantations while maintaining harmonized sampling methods across land uses. This thesis analyzed the data from forest and oil palm inventories carried out by CIFOR between June and August 2015. CIFOR's sampling team was composed of 4 people and took around 2-3 days to complete the fieldwork of each plot, depending on the land use type and age of the plantation (CIFOR, unpublished).

Additional visits to Pucallpa were done by the author to gather information on disturbance history at the site. The information was collected through interviews with members of regional forestry agencies, environmental agencies, research institutions, small-holders and review of grey literature available in local libraries. A compound of Ucayali's forest studies since the beginning of the 1980's was made by the author, including literature that exists only in printed editions and are somewhat forgotten. Moreover, during the author's visit to Pucallpa in March 2018, new measurements of oil palm heights were carried out to improve above ground carbon estimations. The following section describes the inventory protocol CIFOR followed during CIFOR's sampling campaign of 2015, as well as the re-measurements carried out in oil palm plantations as part of this thesis fieldwork. Pictures of both field campaigns supporting sampling methods can be found in Annex 4.

3.2.1 Disturbed forests

All the following described activities were developed by CIFOR in the 2015 sampling campaign (CIFOR, unpublished). A nested plot with different sampling intensities was established. Above-ground biomass (AGB) in standing alive and dead trees with a diameter at breast height (DBH) > 30 cm was measured in a 100 x 100 m^2 plot. The plot was positioned at a minimum distance of 20 m from the edge of the forest. AGB of standing alive and dead trees with a DBH between 5 and 30 cm was measured in three 16 x 36 m^2 transects positioned randomly within the 1-ha plot See Figure 2 for sampling disposition. The DBH of trees was measured using a metric tape. Tree measurements were done according to RAINFOR field manual (Phillips et al., 2016), at 1.3 m height (DBH) when possible. In case of deformities, climbers or buttress, the point of measurement (POM) was adjusted and recorded[1]. All tree measurement where he height for measuring diameter in typical trees was Dead trees were classified into three decay status (D): Trees without leaves that retained most branches (D1), trees without leaves and small branches that retained some large branches (D2), and trees without leaves and branches that retained mainly the stem (D3) (Kauffman and Donato, 2012).

[1] For instance, POM should be moved 50 cm above the top of the buttress, 2 cm below deformities or 50 cm above the highest stilt root (Phillips et al., 2016).

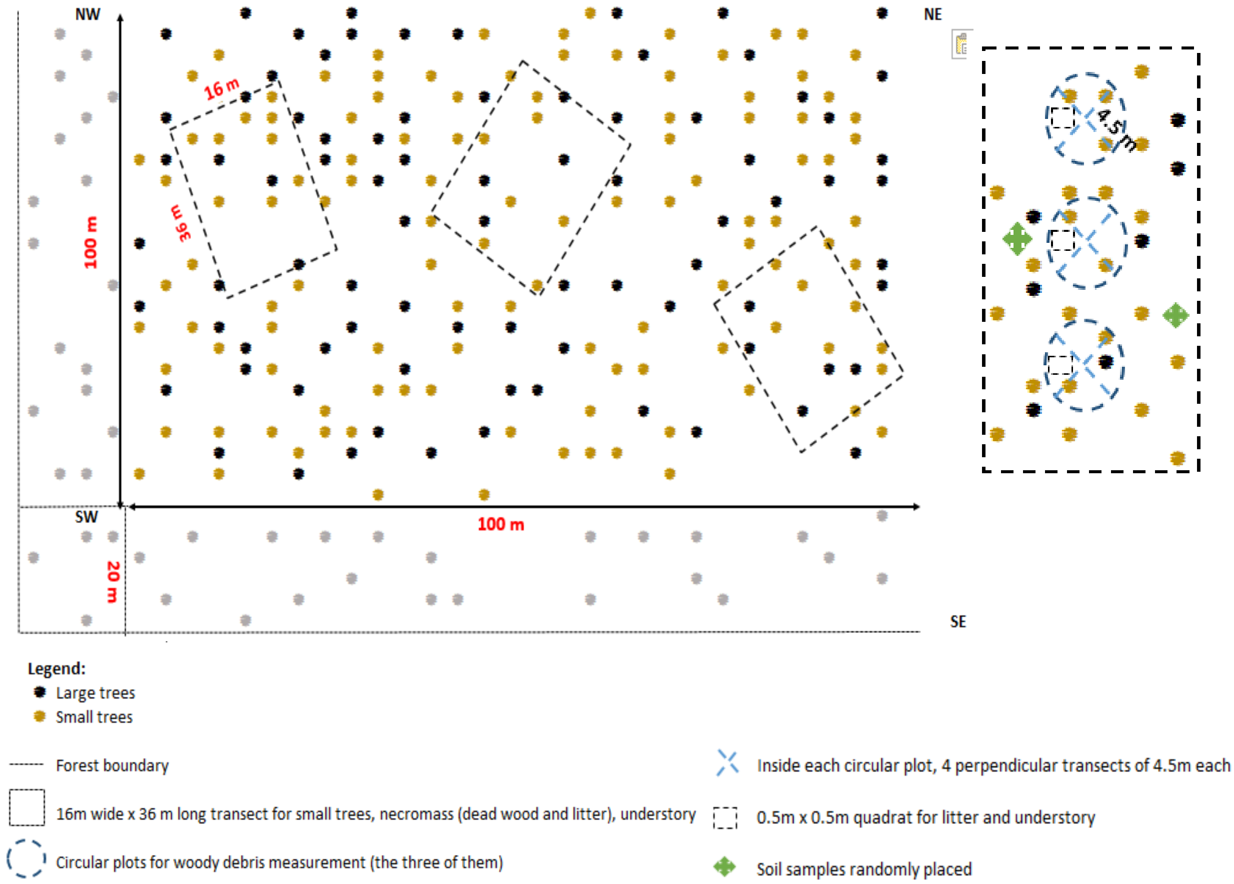

Figure 2. Plot design in oil palm plantations. Pools inventoried included AGB, understory, woody debris, litter and soil.

Tree species were identified at species, genus, and family level by a botanist. Whenever identification could not be performed in situ, leaf samples were collected in the field, air-dried and identified at the herbarium of the "Instituto de Investigación de la Amazonia Peruana" (IIAP). Species-specific wood gravity values were obtained from the global wood density database (Zanne et al., 2009) and other references (Chave et al., 2006; Goodman et al., 2014). Whenever species-specific values were not available, a family- or genus-level average was used, and in the absence of family-level values the average wood density of the plot was used (Baker et al., 2004).

Woody debris was measured non-destructively using the linear intersect technique e.g. by counting the woody pieces intersecting a vertical sampling plane (Kauffman and Donato, 2012). In each 16 x 36 m^2 transects, three 4.5 m radius circular plots were established. In all circular plots, four 4.5 m woody debris transects were laid out perpendicular to each other. Diameter and decay status (sound or rotten) of all large woody debris (diameter ≥ 8 cm) were recorded. In the case of medium woody debris (diameter 2 – 8 cm), all pieces were counted but some additionally were measured. In each transect, the diameter of a maximum of ten randomly chosen medium pieces were recorded. [2]

Woody specific gravity was determined for each wood category (medium, large-sound, large-rotten). Maximum ten pieces were randomly collected (inside the circular plot but outside of transects) for each size class. Their volume was estimated by measuring their size (length, diameter) and their dry mass by weighing them after oven-drying at 65˚C until constant mass. The plot average woody debris specific gravity in each size class was calculated from the ten pieces.

Understory vegetation and litter were sampled in a 0.5 m x 0.5 m quadrat established in each circular plot. All the vegetation growing on the surface (e.g. ferns) was destructively sampled and all the litter collected. The wet mass of understory and litter was weighed in the field and one sub-sample per transect was oven-dried at 65˚C until constant mass and weighed.

Soil pits positions were chosen at random within the 16 x 36 m^2 transects. Two 30 cm deep soil pits were excavated per transect. Soil cores from the 0-10, 10-20 and 20-30 cm depths were sampled using a metallic ring (5cm in diameter, 5 cm in height) and weighed. In the laboratory they were further oven-dried at 105˚C for 24 hours and weighted to determine their bulk density.

3.2.2 Oil palm plantation

Above ground biomass (AGB) in standing alive and dead palms was measured by CIFOR in 100 x 100 m^2 plots (about 13 rows and 12 lines for a 9-m planting density in a triangular design) established at a minimum distance of 18 m from the plantation boundary (CIFOR, unpublished). The height of the palms was measured from the soil surface until the highest leaf using a clinometer or a metric tape for small palms. Within the 1-ha plot three 16 m wide x 36 m long transects were established about 16 m apart (2 rows apart) (Figure 3). A 16 m transect width was chosen in order to include two inter-rows, one with and

[2] The diameter of laying woody debris was recorded at the point where the transect line crosses the piece. If the transect encountered only a corner of the pieces, it was not considered. Pieces were recorded more than once if they crossed twice (or more) the transect (Kauffman and Donato, 2012).

one without decomposing fronds. The 36 m transect length was chosen in order to include three oil palms within the sampling row. All dead fronds decomposing on the ground were counted along the three transects. Three fronds per transect were randomly chosen, cut into small pieces and weighed. Subsamples were fresh-weighed, oven-dried in the laboratory at 65˚C until constant mass and weighed (CIFOR, unpublished).

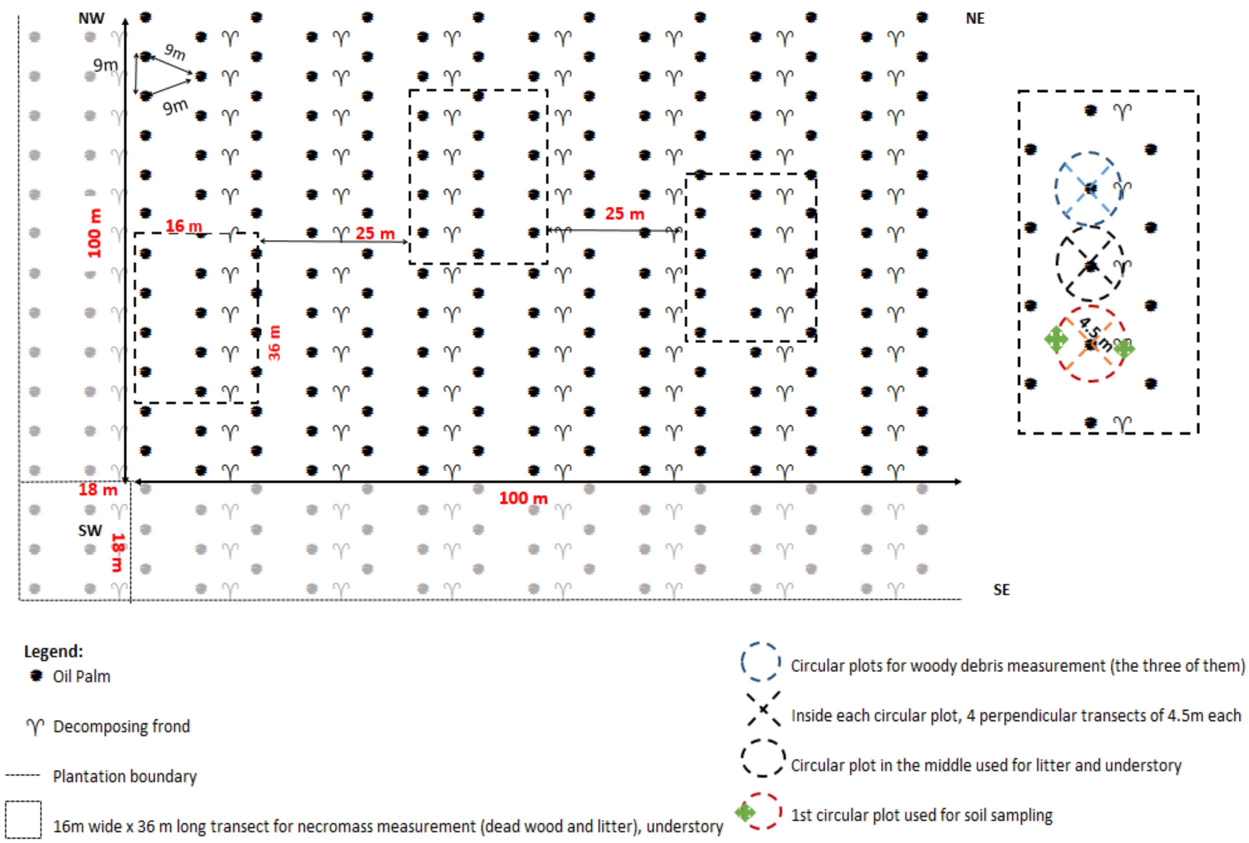

Figure 3. Plot design in oil palm plantations. Pools inventoried included AGB, understory, woody debris, litter and soil.

Measurements of woody debris, understorey, litter and soil followed the same protocol as forests with one exception. Understorey and litter were sampled in the circular plot located in the middle of each 16 x 36 m^2 transect; and soil pits were not randomly placed. Two samples were taken in one of the other two circular plots of each transect; one pit was located 4.5 m from the palm in the inter-row where fronds were left to decompose and the other one was diametrically opposed (CIFOR, unpublished).

The selected allometric equation for oil palm above ground biomass estimation uses palm trunk height as independent variable. Therefore, the author preformed supplementary field measurements during a field visit in March 2018 to assess palm trunk heights in relation to total oil palm height to later adjust former measurements. A previously sampled site in Nuevo San Pedro was revisited where seven oil palm aged 1, 4, 6, 9, 12, 17 and 25 stands were selected. Both total and trunk height of 30 to 40 individuals per stand were measured by the author with the help of a field assistant using a clinometer or a metric tape in the case of younger palms. The youngest palms (1 year old) had no visible trunk, so no trunk height was recorded. Total height was measured as described before and trunk height was estimated by measuring from the ground to the most recent emerging frond (Khasanah et al., 2015a).

3.3 Data analysis

3.3.1 Composition and structure of disturbed forest

Trees were identified to species level when possible, or else to genus or family level. Species richness refers to the number of species recorded for a group of organism in a specific area (Kindt and Coe, 2005). Therefore, species, genus and family abundance was calculated to characterize species richness, which was expressed as the sum of individuals of the same group found in each plot. Big trees (BT, DBH ≥ 30 cm) and small trees (ST, 10 cm ≤ DBH < 30 cm) were extrapolated by their own sampling intensity to compute abundance per ha (N/ha). Non identified individuals were not displayed on the results.

A detrended correspondence analysis (DCA) was performed with the use of the vegan Package in R to explore the differences or similarities in species composition between plots (Honorio-Coronado et al., 2015). The community data used was a matrix sorted by species and the corresponding abundance fraction of each species per plot. The DCA shows the 25 most abundant species from all plots (N/ha). The DCA also highlighted additional abundant big trees (n > 5 individual ha^{-1}), species from woody families (Myristicaceae, Myrtaceae, Meliaceae, Lauraceae, Chrysobalanaceae) and species typical of highly degraded forest.

Species Importance Value Index (IVI) was computed to identify the most ecological important species per plot, following the equation by Honorio-Coronado et al. (2015). As frequency counts are sensitive to sampling size (Smith et al., 1986) the IVI analysis was preformed separately for big and small trees.

$$IVI = 100 * \left[\frac{Ab_i}{Ab_t} * \frac{Do_i}{Do_t} * \frac{Fr_i}{Fr_t}\right] \qquad (1)$$

Where: *i* refers to individual

t refers to total

Ab is abundance, the number of individuals of a species (i) in the total plot (t);

Do expresses dominancy in basal area per ha (m^2 ha^{-1});

Fr is frequency, the number of species occurrence within a plot (BT) or a subplot (ST).

Other parameters used to characterize forest structure included basal area (m^2 ha^{-1}), average DBH and height, abundance and volume (m^3 ha^{-1}) disaggregated by plot and DBH class (in the case of abundance and basal area). DBH classes thresholds were defined as shown in Table 2 (Mueller-Dombois and Ellenberg, 1974). The analysis of composition and structure of forest plots was performed for trees with a DBH ≥ 10cm, in accordance with the scientific found. Tree volume calculation considered a 0.5 form factor as a conservative approach for Amazon disturbed forest (Galván et al., 2000).

Table 2. DBH classes categories.

Class	DBH (cm)	Height (m)
I	DBH<10	H<4m
II	DBH<10	H>4m
III	10<=DBH<30	-
IV	30<=DBH<50	-
V	DBH>=50	-

The alpha fisher index was calculated using the vegan package from the R program to test species diversity. As the tests is sensitive to counts, a species matrix was built in R with abundance (N ha^{-1}) as the dependent variable to calculate the biodiversity index.

3.3.2 Carbon stocks of disturbed forest

Aboveground biomass (AGB) (kg dm $tree^{-1}$) of standing alive and dead trees (status D1 and D2) was calculated following Chave et al. (2014).[3] Just a few allometric models haven been developed for Peruvian

[3] Chave et al. (2014) model precision is defined as σ = 0:357; AIC = 3130; df = 4002.

Amazon tree species. The Chave et al. (2014) pantropical model is considered to suit the climatic and ecological characteristics of the study site; because, firstly, it includes Peruvian lowland forest' data for its elaboration and secondly, has been used by other peer reviewed carbon accounting scientific work on lowland Amazon forests (Duque et al., 2017; Honorio-Coronado et al., 2015).

$$AGB_{tree} = 0.0673 \times [\rho \times DBH^2 \times H]^{0.976} \qquad (2)$$

Where: ρ is the wood specific gravity (g m^{-3}),

DBH is the diameter at breast height (cm),

H is the tree total height (m).

Heights were not measured in the field, but instead computed as (Chave et al., 2014):

$$H_{tree} = e^{[0.893-E+0.760\times LN(DBH)-0.0340\times[LN(DBH)]^2]} \qquad (3)$$

Where E is defined as the factor of environmental stress on the diameter-height tree allometry. It was estimated as the mean E value of the four forest plots (-0.075) using the raster tools provided by Chave et al. (2014). AGB by plot was later computed as the sum of AGB of small trees and big trees, each extrapolated to one ha by their correspondent sampling sizes of 1ha and 0.1728ha. For dead trees of status D1 and D2, 2.5% and 15% of the AGB estimate by was subtracted to account for the absence of biomass in leaves and leaves and small branches (Kauffman and Donato, 2012).

AGB of standing dead trees of status D3 was calculated as the volume of a cylinder multiplied by the wood specific gravity (0.5, default value). The height of D3 trees was measured from the soil surface until the highest point using a clinometer. Each plot's specific dead large sound wood carbon content (defined in laboratory) was used to estimate carbon stocks (Mg C ha-1) for both alive and standing dead pools.

Fallen dead wood calculations were based on Kauffman and Donato (2012) protocol. The volume of medium woody debris (2-8cm diameter) per circular plot and transect was calculated as:

$$Volume\ (m^3 ha^{-1}) = \pi \times \left[\frac{N_i \times QMD_i^2}{8 \times L}\right] \qquad (4)$$

Where: N_i is the count of intersecting woody debris pieces;

QMD$_i$ the quadratic mean diameter per transect: QMD = $\sqrt{(\Sigma d_i{}^2)/n}$;

and L the transect length (m) 18m for each transect.

Large fallen wood (sound and rotten) volumes per circular plot and transect were calculated from individual diameter measurements:

$$Volume\ (m^3 ha^{-1}) = \pi \times \left[\frac{d_1^2 + d_2^2 + \cdots + d_m^2}{8 \times L}\right] \qquad (5)$$

Where: d_1, d_2, etc. are the diameters of intersecting pieces of large dead wood (cm);

and L the transect length (m), 18m for each transect.

Woody debris mass was calculated as the volume multiplied by its mean specific gravity (medium, large-sound and large-rotten). Specific gravity (Mg dm m^{-3}) and carbon contents were determined for each wood category upon the 10 pieces per class collected in each plot. Necromass (Mg dm ha^{-1}) and carbon stocks (Mg C ha^{-1}) of woody debris were calculated for each circular plot and transect as the sum of the three classes. Three replicates for each category (medium, large-sound, large-rotten) made up of three mixed sub-subsamples randomly chosen were analysed for carbon & nitrogen content.

Biomass and carbon stocks calculations for litter and understory followed a similar procedure as oil palm fronds (see next section). Subsamples per transect were used to calculate a wet/dry mass ratio, from which a plot ratio was defined. The plot average dry wet mass ratio obtained was used to calculate the transect's dry mass of understory and litter collected. Carbon stocks (Mg C ha^{-1}) per transect were defined by extrapolating the dry mass per correspondent litter and understory sampling size, as well as carbon content. Three replicates of understorey and litter each made up of three mixed sub-subsamples from the three transects were analysed for carbon & nitrogen content.

Soil carbon stock per depth layer (Mg C ha^{-1} $layer^{-1}$) was calculated as:

$$SCS_l = C \times BD \times d \times 100 \qquad (6)$$

Where: C is the carbon fraction of the total carbon content;

BD is the soil bulk density (g dry mass cm^{-3});

and d is the soil layer depth (cm)

The total soil carbon content per sampling site and transect was the result of the sum of the three samplings depths. There were 2 sampling sites per transects making a total of six SCS total values per plot. Each soil sample was analysed for carbon and nitrogen contents.

The root to shoot ratio used was 0.235, as recommended for tropical/subtropical moist forest with less than 125 Mg dm ha^{-1} (Mokany et al., 2006). Carbon content from living AGBs was considered for root carbon estimations.

3.3.3 Carbon stock in oil palm plantation

Data analysis in oil palm plantations was mainly focused on carbon stocks estimates. No allometric equations were found for oil palm above ground biomass (AGB) estimations in the Amazon, they have been mostly developed in Southeast Asia. Khasanah et al. (2015a) used semi-destructive sampling to assess models per soil type (mineral or peat) to compare linear regressions vs. power models. Their best fitted regression in mineral soils was chosen for this study.

$$AGB_{OP}\ (Mg\ palm^{-1}) = 0.0923 \times H + 0.13333 \quad (R^2 = 0.8544) \qquad (7)$$

Where H is the height of the palm trunk (m).

Before going over AGB plot calculations, a previous data preparation step was required in order to compute palm trunk height from palm total height. Based on the seven plots measured in the Nuevo San Pedro village in 2018, a regression between the stand age and the trunk height (H_{trunk}) /palm height (H_{palm}) ratio was established (Figure 4). A direct trunk height / palm height relationship was not possible to build due to the effect of planting density over trunk height vs. age (Henson, 2006), which is especially relevant among small holders that do not follow a homogenous planting density.

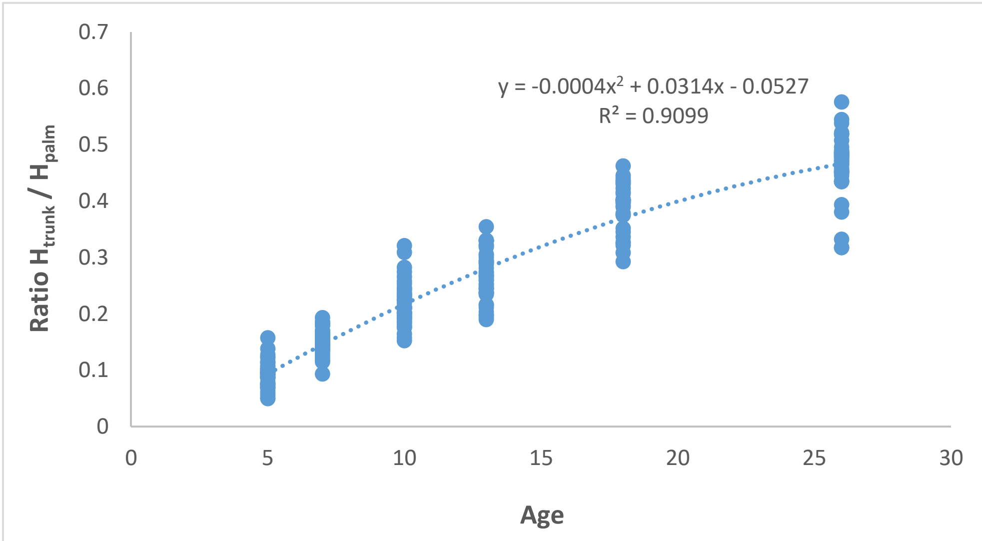

Figure 4. Relationship between stand age and H_{trunk}/H_{palm} ratio using measurements from the 2018 field campaign.

Based on the model in Figure 4, a H_{trunk}-to-H_{palm} ratio was calculated for each stand age of the fully-inventoried plots (Table 3), which was used to estimate a trunk height value for each individual palm from its palm height.

Table 3. Trunk height/ Palm height ratio for previous inventoried plots.

Oil palm stand age	H_{trunk}/ H_{palm} ratio
1	-0.0217[4]
4	0.0665
7	0.1475
15	0.3283
23	0.4579
28	0.5129

Individual palm oil biomass (Mg dm palm^{-1}) was later added up to a hectare level (Mg dm ha^{-1}). The specific dead large sound wood carbon content from each plot (defined in laboratory) was used to estimate carbon stocks (Mg C ha^{-1}) for both alive and standing dead pools.

[4] The young stand had no visible trunk height because it was planted under the surface. Therefore a trunk height of 0 was considered.

In the case of fronds, the subsamples fresh and dry weights, in addition to transects fronds counts, were used to estimate the transect frond biomass and carbon stocks of each plot. The plot average dry/wet mass ratio obtained from each subsample was used to calculate the dry mass of the nine fronds. The dry mass of fronds per ha (Mg dm ha^{-1}) was computed from the frond plot average dry mass and the number of counted fronds per transect times the transect size. Furthermore, three replicates each made up of three mixed sub-subsamples from each transect were analysed for carbon and nitrogen content.

Fallen dead wood, understory, litter and soils total carbon (Mg C ha^{-1}) followed the same procedure as described in the previous data analysis section. In the case of soils, site refers to the position of the sample, whether it coincides with the decomposed fronds position or not. Root biomass was estimated from shoot biomass using the root-to-shoot ratios for oil palm plantations in mineral soils. In young stands (1-6 year old plots) a 0.61 root/shoot ratio was applied (Syahrinudin, 2005), 0.29 for medium age (7-19 year old plots) (ibid) and 0.19 for older stands (> 19 year old plots) (Khalid et al., 1999).

3.4 Statistics

For disturbed forest plots, incase replicates were available (all pools except AGB and standing dead wood), they were used to compute means and standard errors per pool and plot. Later, they were used to estimate representative mean carbon stocks (Mg C ha^{-1}) of disturbed forests. Standard error propagation techniques followed Lo (2005).

Oil palm carbon stocks from different age stands were used to infer growth models and generate rotation time average carbon stocks for a 30 year period. Regression models (exponential, logarithmic, polynomial, among others) were tested for carbon stocks (Mg C ha^{-1}) along the chronosequence and the best fitted model per pool was chosen based on Akaike Information Criterion (AIC). When available, replicates were used for regressions. If not, the model was the result of the six different age stands average. In the case of root biomass estimation over the oil palm rotation, roots to shoots ratios were applied to yearly aboveground carbon (AGB) values in accordance to their stand age. Estimations over the modeled rotation years were considered independent, no correlation in time is assumed.

Comparisons of two means (e.g. secondary forest vs. oil palm plantation stocks) were performed applying the T test, under the assumption of measurements being independently drawn from a normal distribution

N (μ,σ^2) with unknown μ and $\sigma2$. When dealing with more than two groups, ANOVA was tested followed by a Tukey pair-wise comparison. When residuals were not normally distributed, the non-parametric Kruskal Wallis test with pairwise comparison was performed.

All tests were conducted with a 5% probability level. Statistical programs used were open source Infostat (version 2018) (Di Rienzo et al., 2018) and R (version 3.3.2), including packages such as vegan and ggplot2.

IV. Results

4.1 Degradation effects on forest composition and structure

4.1.1 Forest composition

A total of 573 living trees with a DBH ≥ 10 cm were recorded across the four forest plots. 161 taxa were identified, from which 99 were differentiated at species level, 48 at genus level and 13 at family level. Main abundant taxa are displayed in Figure 5. It reveals species composition coincidences among plots, as well as a dominancy of the Aracaceae family (palm trees) in disturbed forests (See Annex 5 for the list of species). These forests harbored mostly medium to fast growing species (*Oenocarpus bataua, Socratea exhorriza, Cecropia sp.1, Cecropia sciadophylla, Euterpe precatoria, etc*.) as well as species representative from gaps or disturbed forest *(Pourouma sp, Zygia latifolia, etc.).* Wood-valuable species from the families Myristicaceae (*Iryanthera juruensis, Virola sp.1*), Myrtaceae (*Calyptranthes speciosa*) and Lauraceae (*Nectandra sp.1*) also stand out among the most abundant ones (Gentry, 1996; Pennington et al., 2004).

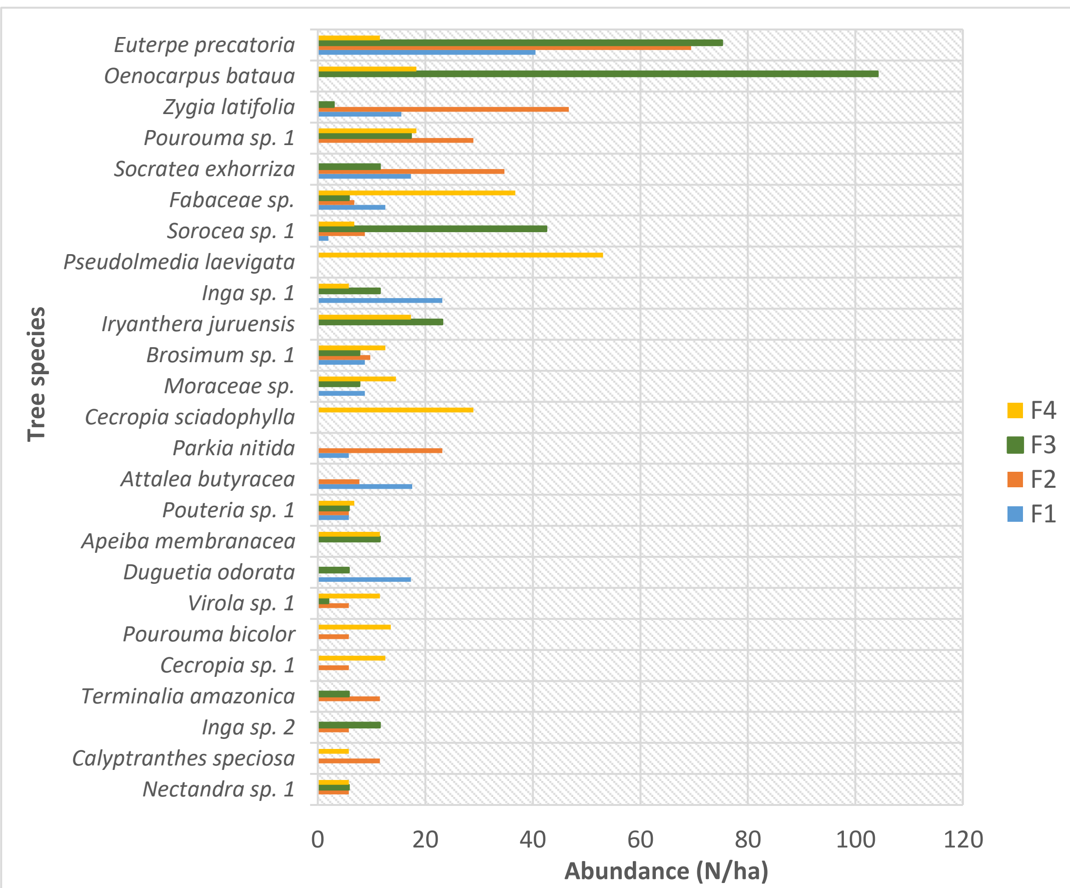

Figure 5. 25 most abundant taxa characterized by plot.

The detrended correspondence analysis (DCA) (Figure 6) explores differences and similarities in floristic composition between plots. Forest plot number three exhibited more dominance of individuals from the Aracaceae family than plot number four. Pioneer species such as *Cecropia sciadophylla, Cecropia sp.1* and *Pseudolmedia laevigata* were mainly found in plot number four. All sampled sites presented species characteristic of highly intervened forests, such as species from the Miconia genus and *Vismia baccifera*. Additional species coming from valuable woody families (Myristicaceae, Myrtaceae, Meliaceae, Lauraceae, Chrysobalanaceae) also stood out mostly in plot number four and one, but were less evident in plot number three.

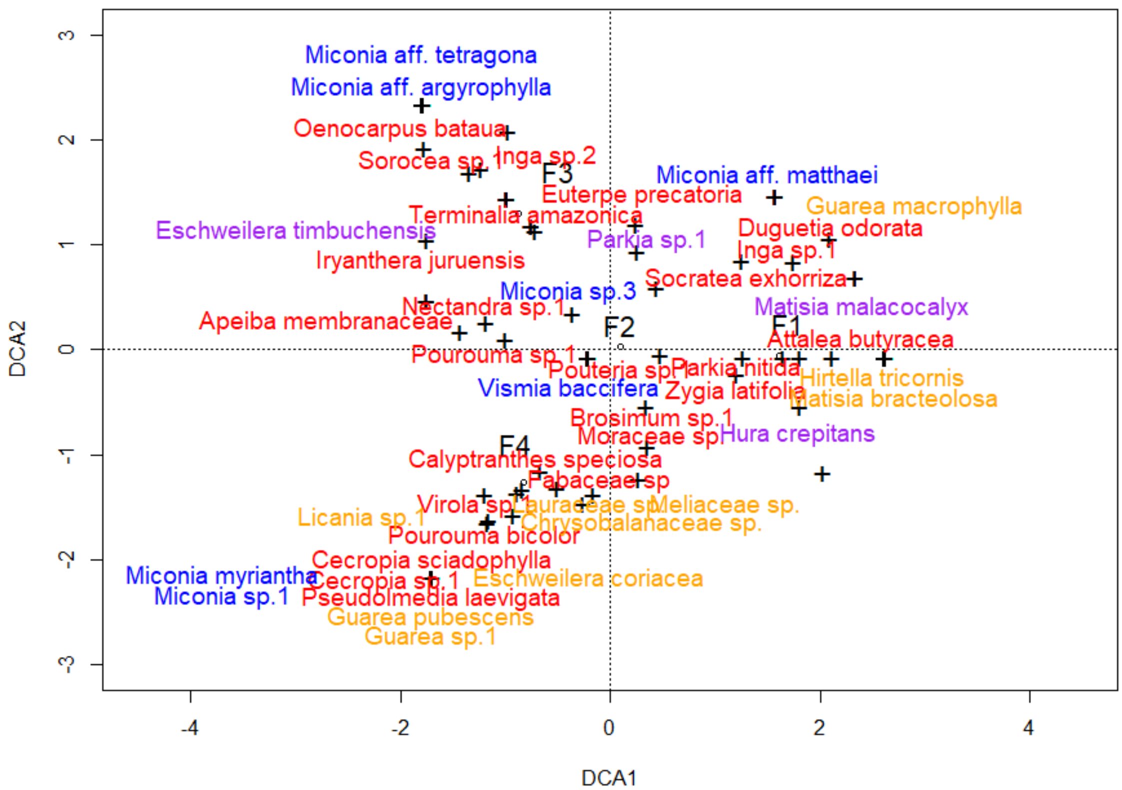

Figure 6. Detrended correspondence analysis (DCA) out of the (+) 161 taxa abundance values found in the forest plots. In red, the 25 most abundant species); in orange, examples of additional species from woody families; in purple, additional abundant big trees (>5 individuals ha^{-1}) and in blue, species typical of highly degraded forest.

The first five species with higher importance value index (IVI) per plot are shown in Table 4. In all four plots, they represented between 21 to 63% of the taxa of the plot, which reveals more or less homogeneous stands. Most importantly, the highest IVI species were not necessarily the most abundant ones because forest structure parameters (basal area) give additional importance to the taxa. That is the case of *Hura Crepitans, Eschweilera timbuchensis, Parkia sp. 1, Pouteria trilocularis, Guatteria sp. 1,* among others, that have their higher contribution from basal area. Full IVIs data is presented in Annex 5, including abundance data, dominance and frequency information per species.

Table 4. Species Importance Value Index (IVI) for the main five species per plot and sampling density.[5]

		Species	Absolute			Relative			IVI
			Ab	**BA**	**Fr**	**Ab**	**BA**	**Fr**	
F1	**Big Trees >30cm**								
	1	*Hura crepitans*	5	15.0	1	8	16	4	27
	2	*Matisia malacocalyx*	6	4.8	1	9	5	4	18
	3	*Attalea butyracea*	6	3.7	1	9	4	4	17
	4	*Zygia latifolia*	4	6.0	1	6	6	4	16
	5	*Myrcia sp. 1*	4	5.9	1	6	6	4	16
		Others	41	58.2	23	62	62	82	206
	Small Trees <30cm								
	1	*Euterpe precatoria*	7	3.7	3	9	5	5	19
	2	*Inga sp. 1*	4	3.4	2	5	5	4	13
	3	*Brosimum alicastrum*	3	3.0	2	4	4	4	12
	4	*Attalea butyracea*	2	3.9	1	2	6	2	10
	5	*Duguetia odorata*	3	1.4	2	4	2	4	9
		Others	**62**	**52.6**	**47**	**77**	**77**	**82**	**236**
F2	**Big Trees >30cm**								
	1	*Zygia latifolia*	12	21.2	1	33	37	7	78
	2	*Brosimum sp. 1*	4	5.5	1	11	10	7	28
	3	*Sorocea sp. 1*	2	6.1	1	6	11	7	23
	4	*Pouteria trilocularis*	3	3.9	1	8	7	7	22
	5	*Guatteria sp. 1*	2	5.0	1	6	9	7	22
		Others	13	14.9	9	36	26	64	127
	Small Trees <30cm								
	1	*Zygia latifolia*	6	18.5	3	7	25	6	38
	2	*Euterpe precatoria*	12	5.8	3	14	8	6	28
	3	*Parkia nitida*	4	3.4	3	5	5	6	15
	4	*Socratea exhorriza*	6	1.3	3	7	2	6	15
	5	*Pourouma sp. 1*	5	1.5	2	6	2	4	12
		Others	**50**	**42**	**38**	**60**	**58**	**73**	**191**

[5] Ab refers to Abundance (N ha^{-1}), BA to basal area (m2 ha^{-1}) and Fr to frequency. Total IVI per plot and sampling intensity is 300%.

		Species	Absolute			Relative			IVI
			Ab	BA	Fr	Ab	BA	Fr	
F3	**Big Trees >30cm**								
	1	*Eschweilera timbuchensis*	5	12.6	1	13	20	5	37
	2	*Pouteria trilocularis*	4	7.3	1	10	12	5	26
	3	*Zygia latifolia*	3	3.9	1	8	6	5	18
	4	*Parkia sp. 1*	3	3.6	1	8	6	5	18
	5	*Moraceae sp.*	2	4.7	1	5	7	5	17
		Others	23	30.8	17	58	49	77	184
	Small Trees <30cm								
	1	*Oenocarpus bataua*	18	20.4	3	18	25	5	48
	2	*Euterpe precatoria*	13	4.8	3	13	6	5	24
	3	*Sorocea sp. 1*	7	4.6	3	7	6	5	18
	4	*Iryanthera juruensis*	4	5.2	2	4	6	3	14
	5	*Pourouma sp. 1*	3	1.7	3	3	2	5	10
		Others	**54**	**44**	**48**	**55**	**55**	**77**	**187**
		Overall total	**139**	**143.8**		**100**	**100**	**100**	**300**
F4	**Big Trees >30cm**								
	1	*Hieronima sp. 1*	2	4.4	1	4	7	4	14
	2	*Moraceae sp.*	3	2.7	1	6	4	4	14
	3	*Parkia sp. 1*	2	3.3	1	4	5	4	13
	4	*Apeiba membranaceae*	2	3.1	1	4	5	4	12
	5	*Hura crepitans*	2	3.0	1	4	5	4	12
		Others	40	46.7	23	78	74	82	234
	Small Trees <30cm								
	1	*Pseudolmedia laevigata*	9	6.2	3	8	6	5	18
	2	*Indeterminado_Fabaceae*	6	3.5	2	5	3	3	11
	3	*Oenocarpus bataua*	3	4.4	2	3	4	3	10
	4	*Cecropia sciadophylla*	5	2.4	2	4	2	3	10
	5	*Guatteria glauca*	3	4.1	1	3	4	2	8
		Others	**91**	**88**	**54**	**78**	**81**	**84**	**243**
		Overall total	**168**	**171.5**		**100**	**100**	**100**	**300**

In terms of floristic richness, the alpha fisher biodiversity index ranged between 11.5 and 19.7 (Table 5). Plot number one presented the highest value and number two the lowest. The latest was also the plot with lowest tree abundance. Overall, the four plots have a 16.6 (SE 1.8) diversity index.

Table 5. Biodiversity Index per plot.

	F1	F2	F3	F4	Mean	SE
Alpha.Fisher	19.7	11.5	18.3	16.8	16.6	1.8
Abundance (N ha^{-1})	534	511	612	728	597	48.8

4.1.2 Forest structure

The mean tree DBH and height from all four disturbed forest stands amounted to 24.3 cm (SE 2.3 cm) and 20.1 ± 2 m, respectively, with the first plot reaching the maximum (Table 6). Although plot one had the highest mean DBH, plot four had the top basal area due to greater tree abundance. The overall distribution of both structure variables was a classic inverted "J" (Figure 7), showing a higher frequency of small trees (in DBH and height) with a slight bounce in the height distribution around medium/tall size trees.

Table 6. Forest plots overall structure information.

Plot	Mean plot tree DBH (cm)	Mean plot tree height (m)	Stand BA (m^2 ha^{-1})	Stand Volume (m^3 ha^{-1})
F1	27.2	21.3	22.8	232.8
F2	23.4	19.7	18.6	167.7
F3	23.3	19.6	21.4	192.3
F4	23.4	19.8	25.3	230.6
Mean	**24.3**	**20.1**	**22.0**	**205.9**
SE	**2.3**	**1.0**	**1.4**	**15.8**

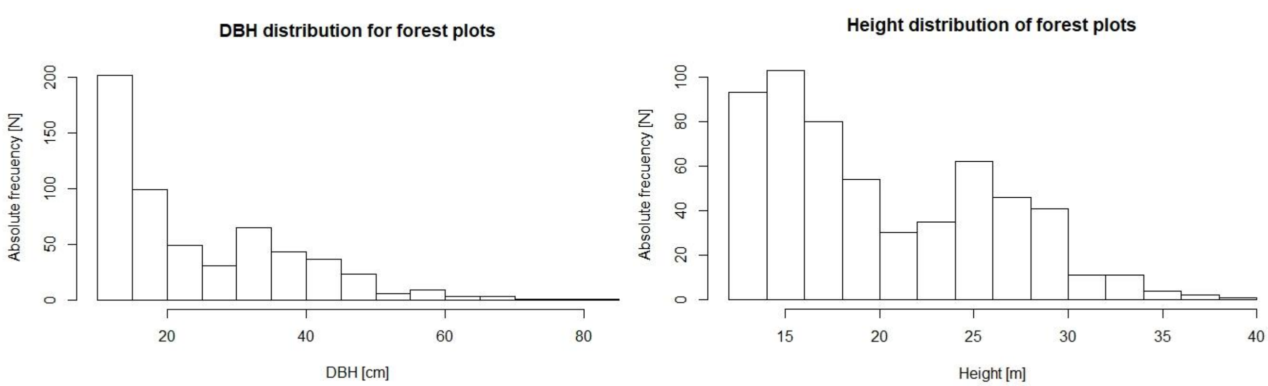

Figure 7. Overall distribution of (a) DBH (cm) and (b) height (m) for all forest plots.

Furthermore, Figure 8 segregates abundance and basal area distribution by plot per DBH class. The largest tree abundance in plot 4 is mainly pending from the smallest DBH class trees. Plot number one, on the other hand, has less trees per ha but a slightly higher contribution from large trees in comparison to the other plots. Abundance also explains basal area. Therefore, the smallest DBH class contributes more to F4 basal area; whereas forest plot one presents the highest basal area in the largest class.

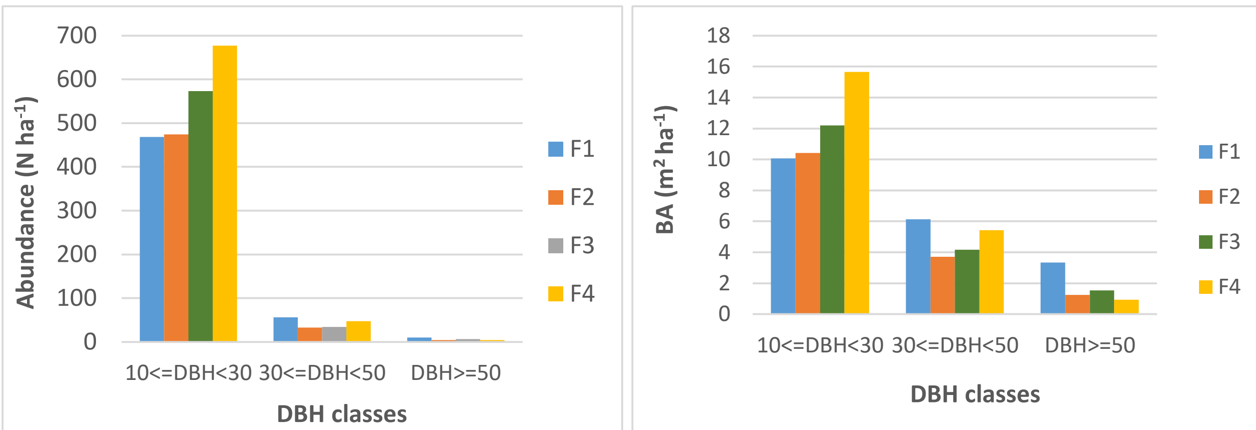

Figure 8. DBH classes per plot for (a) abundance and (b) basal area (BA).

4.2 Carbon stock losses over the transition

4.2.1 Carbon stocks in disturbed forests

Mean carbon stocks and standard errors (Mg C ha^{-1}) by pool are shown in Table 7, full plot information can be found in Annex 6. Living aboveground biomass stocks varied between 61.8 and 79.2 Mg C ha^{-1}, with an overall mean of 71.3 Mg C ha^{-1} (SE 4.2 Mg C ha^{-1}). Similarly to volume results (Table 6), the highest Aboveground Carbon (AGC) and standing dead wood was found in F1 (Table 7). Significant differences between plots were tested using ANOVA for soils understory and litter and Kruskal-Wallis for fallen dead wood. Calculations did not show significant difference between plot means ($\rho > 0.05$), except for fallen dead wood ($\rho = 0.0185$). The total carbon stock of disturbed forest plots was 140.7 Mg C ha^{-1} (SE 5.8 Mg C ha^{-1}); with the highest carbon contribution coming from ABC, followed by soils.

Table 7. Mean carbon stocks (Mg C ha^{-1}) and standard errors of disturbed forest plots in the study site.

Carbon pool	F1		F2		F3		F4		Mean	SE
	x̄	SE	x̄	SE	x̄	SE	x̄	SE		
Tree living AGC*	79.2	-	61.8	-	66.6	-	77.6	-	71.3	4.2
Standing DW*	6.1	-	2.8	-	2.1	-	3.1	-	3.5	0.9
Fallen DW **	9.2^{a}	1.7	25.7^{b}	5.6	8.4^{a}	2.0	19.8^{a}	7.1	15.8	2.4
Litter	2.0	0.4	2.0	0.7	1.3	0.1	2.5	0.0	1.9	0.2
Understory	0.9	0.3	1.2	0.5	0.3	0.1	0.6	0.1	0.7	0.2
Roots*	18.6	n.a.	14.5	n.a.	27.7	n.a.	18.2	n.a.	19.8	2.8
Soils	32.1	3.4	23.7	3.9	26.0	2.0	28.9	1.5	27.7	1.4
Total Carbon Stock									**140.7**	**5.8**

*No replicates were recorded for these pools within the pools
** Letters a and b indicate a significant difference between plots within a pool (ρ= 0.0185).

4.2.2 Carbon stocks evolution in time by pool in oil palm plantations

Mean heights showed a logarithmic growth along the oil palm chronosequence (Figure 9). Small and medium size old palm farmers do not always follow standard management techniques. Therefore, planting density was not homogenous across stands, influencing both mean stand heights (Figure 9) and AGC. The 15-year-old oil palm stand had the highest planting density (211 palms ha^{-1}) as well as the highest average total palm height (12.7 m).

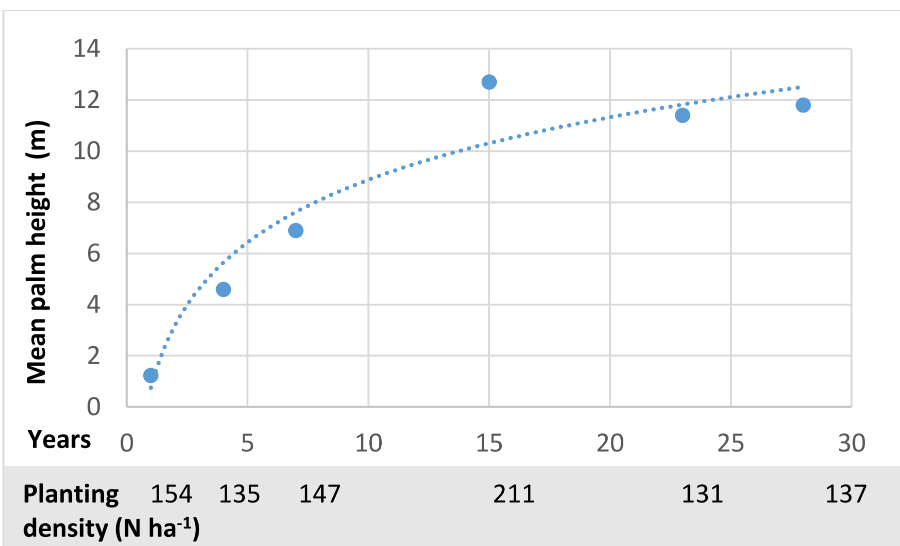

Figure 9. Mean total palm height per stand age. Planting density per plot is displayed at the bottom of the figure.

The carbon stocks of all plots are fully presented in Annex 6. Best fitted carbon growth models are shown in Table 8. All models were logarithmic. AGC and frond pools increased over the years; whereas understory, dead wood and soils presented an inverted trend (Figure 10). There was no clear trend or significant differences in carbon stocks between plots for litter (ANOVA, ρ>0.9999), therefore, no growth model was developed for this pool. The average for litter across plots was 0.42 Mg C ha^{-1} (SE 0.07 Mg C ha^{-1}) (Figure 10d). Although soil samples were taken in sites with fronds and without them expecting a difference in total soil carbon content (0-30cm depths), the null hypothesis of homogenous carbon stocks in both sites could not be rejected (T test, ρ=0.1819). Hence, all soil samples were taken equally as replicates to build the growth model. Yearly oil palm carbon stocks (Mg C ha^{-1}) estimated by each pool's model are presented in Annex 7.

Table 8. Best fitted carbon growth model per pool.

Carbon stock	n	AIC	Parameter	Estimate	SE	T	p - value	Growth stock models in (Mg C ha^{-1})
AGC living	6	47.7	b1	13.3	1.7	7.6	0.0006	$AGC_i = b1*LN(Age_i) + Ɛ_i$
Fronds	18	90.9	b1	1.8	0.3	6.6	<0.0001	$Fronds_i = b1*LN(Gge_i) + Ɛ_i$
Fallen dead wood	54	418.3	b1	-3.2	1.3	-2.4	0.0187	$DW_i = b1*LN(Age_i)+b0 + Ɛ_i$
			b0	11.9	3.2	3.7	0.0005	
Understory	18	26.5	b1	-0.2	0.1	-2.5	0.0241	$Understory_i = b1*LN(Age_i)+b0+ Ɛ_i$
			b0	1.0	0.2	4.4	0.0005	
Soil carbon (0-30cm)	36	269.2	b1	-3.2	1.4	-2.3	0.0303	$SCS_i =b1*LN(Age_i)+b0+ Ɛ_i$
			b0	40.0	3.3	12	<0.0001	

4.2.3 Carbon stocks differences over the land use system transition

Mean carbon stocks found in disturbed forest adjacent to oil palm plantations were, in most cases, larger than the highest carbon stocks the crop could reach over its rotation period (ca. 30 years) (Figure 10). AGC, dead wood, and litter have a systematic higher carbon stock in disturbed forests than oil palm plantations. Initially, understory had a similar stock to forest, though later decreased. Litter and understory were the least significant carbon pools in both land use systems. Total soil carbon stock (0-30cm depth), on the other hand, revealed a different outcome. Along the 30-year oil palm rotation soil carbon stock declined over time until reaching a value similar as in forests stands (Figure 10f).

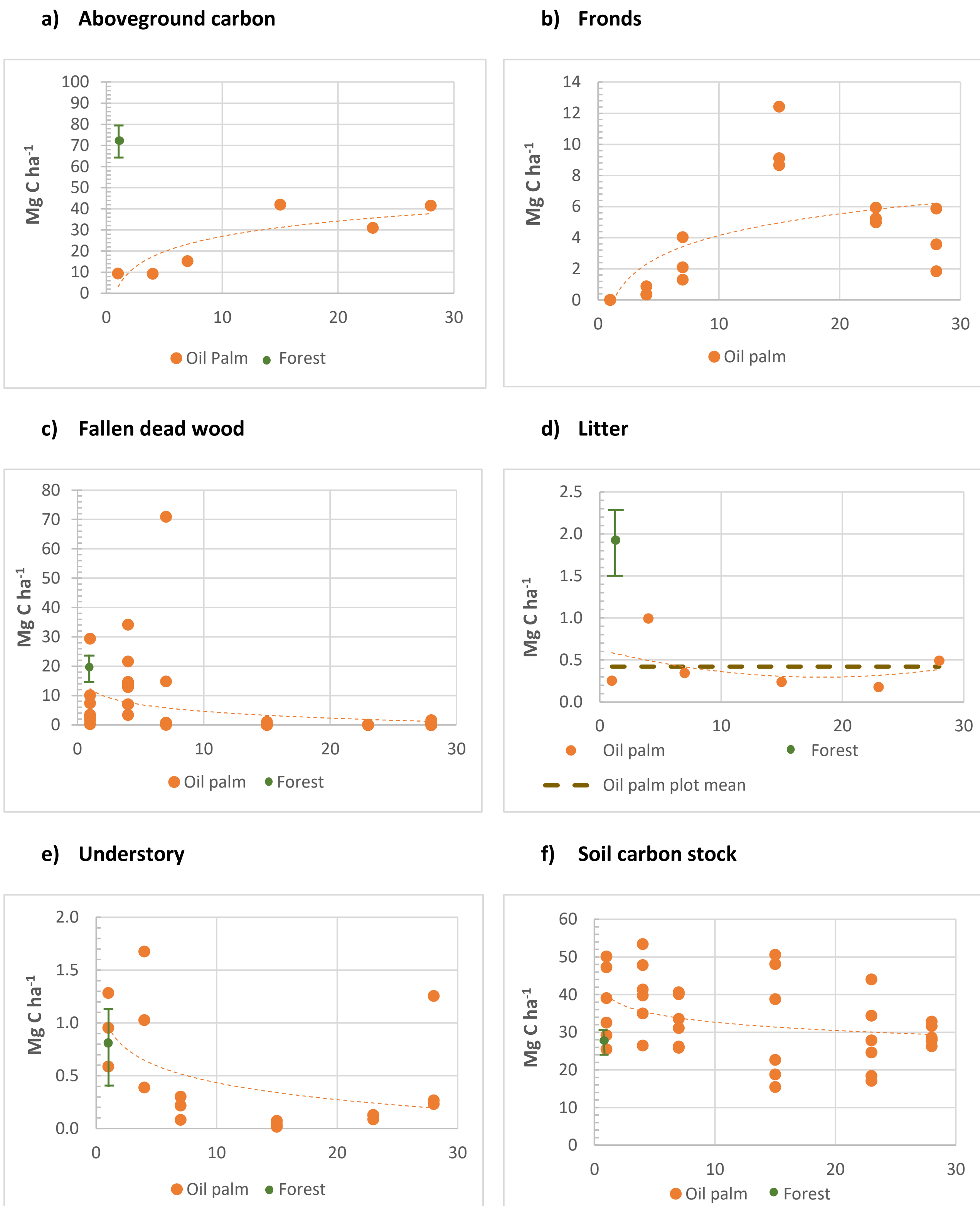

Figure 10. Carbon stock increase or decrease as a function of the age of oil palm plantation. Average carbon stocks and confidence intervals (95% of confidence) in disturbed forests are displayed for comparison purposes.

Disturbed forests in the study site presented an overall carbon stock of 140.7 (SE 5.8) Mg C ha^{-1}; which is almost twice the average stock of a 30-year oil palm rotation 74.3 (SE 2.2) Mg C ha^{-1} (Table 9). As a matter of fact, when addressing the land use transition scenario of the remnant disturbed forest in Campo Verde to oil palm plantation, at any stage of the 30-year rotation, it represented a net carbon source of greenhouse emissions. This is mainly due to the AG carbon pool contribution; whereas the mean in disturbed forest was 71.3 (SE 4.23) Mg C ha^{-1}, the average stock over an oil palm rotation period was significantly inferior (33.02, SE 2.06 Mg C ha^{-1}) (Table 9). Figure 11 visually displays significant differences between carbon stocks of both land use systems. In general, carbon stocks of oil palm plantations were significantly different from disturbed forests in the study site, except for understory (Kruskal Wallis H test ρ= 0.0754). Total soil carbon stock (0-30 cm) was the only pool that had a significantly higher stock in oil palm plantations compared to disturbed forests (T test, ρ= 0.0227).

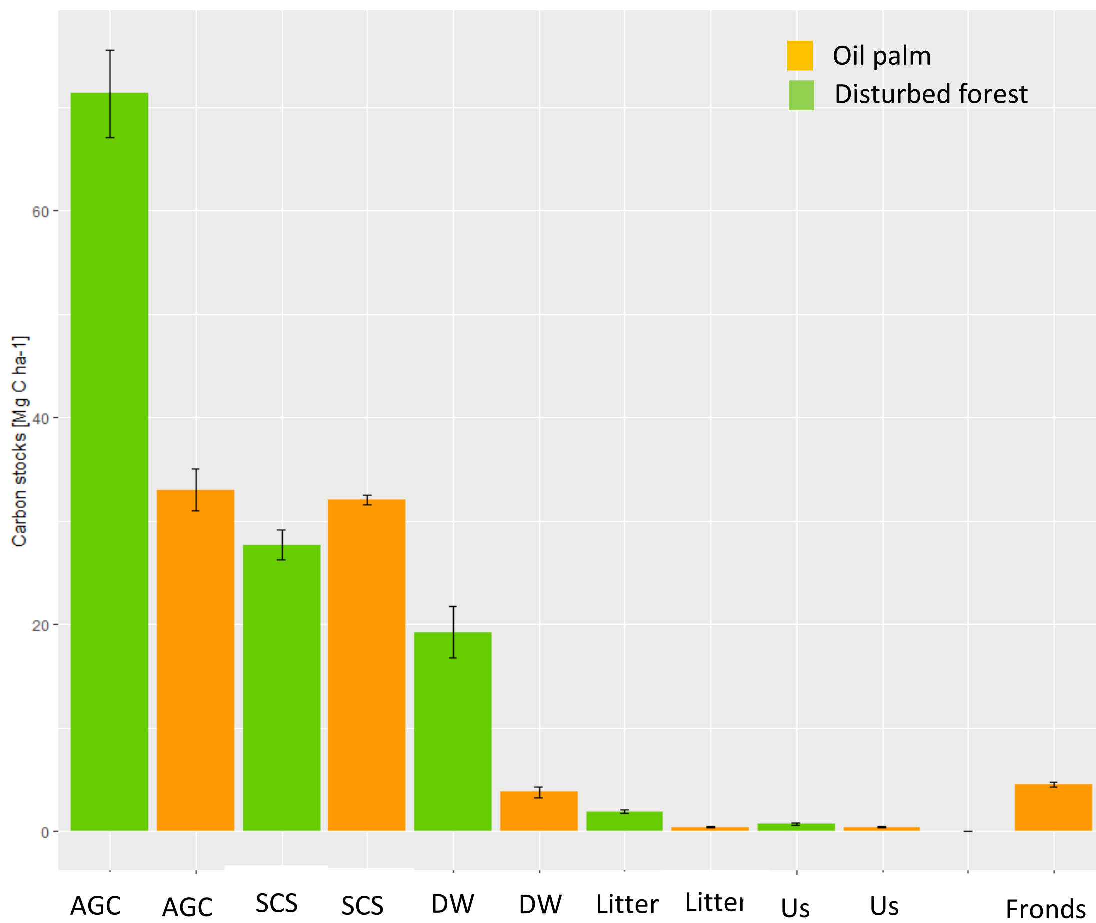

Figure 11. Carbon stocks (Mg C ha^{-1}) and standard errors per pool in disturbed forests and oil palm plantations.

Table 9. Mean disturbed carbon stocks and time-average carbon stock for a 30-year oil palm rotation in Campo Verde district.

	Oil palm plantations			Disturbed forests		
Carbon pool	**n**	**Carbon stock** ($Mg\ C\ ha^{-1}$)	**SE**	**n**	**Carbon stock** ($Mg\ C\ ha^{-1}$)	**SE**
Fronds	18	4.6	0.3	4	n.a.	n.a.
Litter	18	0.4[a]	0.1	4	1.9[b]	0.2
Understory	18	0.4	0.04	4	0.7	0.2
Soil carbon stock	36	32.1[a]	0.5	4	27.7[b]	1.4
Dead wood	54	3.8[a]	0.5	4	19.2[b]	2.5
AGC	6	33.0[a]	2.1	4	71.3[b]	4.2
Roots	6	9.00[a]	0.2	4	19.8[b]	2.8
Overall stock		**74.3**[a]	**2.2**		**140.7**[b]	**5.8**

Letters a and b indicate a significant difference between land use systems per pool ($\rho < 0.05$)

Deeper analysis into the soil revealed not only that there were significant differences between land use systems at total soil depth (0-30 cm), but also at different layers (Kruskal Wallis H test $\rho = 0.0444$). As expected, carbon stocks decreased significantly in the soil profile in both; disturbed forest (Kruskal Wallis H test $\rho = <0.0001$) and oil palm plantations (Kruskal Wallis H test $\rho = <0.0001$) (Figure 12). Bulk density between depths in land classes was also analyzed to discard soil compaction over land use change. No significant differences in bulk density was found for either depth (0-10cm Kruskal Wallis H test $\rho = 0.8385$; 10-20cm Kruskal Wallis H test $\rho = 0.9879$; 20-30cm Kruskal Wallis H test $\rho = 0.4455$) so no adjustments in soil carbon stocks between land uses was needed.

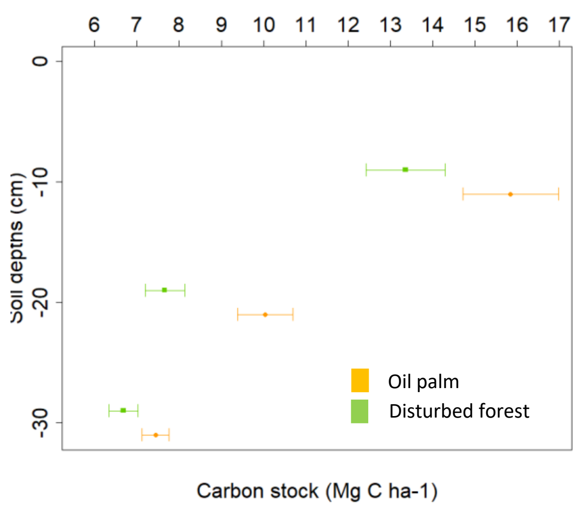

Figure 12. Soil carbon stocks ($Mg\ C\ ha^{-1}$) and standard errors per soil depth for disturbed forest and average oil palm rotation.

Nitrogen content did not present significant difference between land use total soil profiles (Kruskal Wallis H test ρ= 0.3431), nor between land uses within depths (0-10 cm Kruskal Wallis H test ρ= 0.3844; 10-20 cm ANOVA ρ= 0.0944; 20-30 cm Kruskal Wallis H test ρ=0.6204). Neither did nitrogen contents among sampling sites (with or witout fronds) in oil palm plantations, there was no contribution from fronds in nitrogen contents in the soil (Kruskal Wallis H test ρ= 0.317).

V. Discussion

5.1 Forest composition and structure in Amazonian primary and disturbed forests

5.1.1 Forest composition

Among the forest plots evaluated, the Aracaceae family is the most abundant from them all, followed by the Fabaceae, Moraceae and Urticaceae families. A floristic study in the Amazon reveals some taxa hyperdominance; as for only 277 species represent 50% of all individuals in the region (ter Steege et al., 2013). Among them, the Aracaceae family was reported as most dominant; from which species such as *Euterpe precatoria, Oenocarpus bataua* and *Socratea exhorriza* stood out among the 20 most abundant species in the Amazon (ibid). Such species were also frequently found in the sampled forest plots. However, the Aracaceae family was not equally reported in all plots, it was mostly frequent in forest plots number three, two and one respectively. Palm species are apparently more abundant on nutrient-rich soils (Nebel et al., 2001).

Furthermore, in the forest stands sampled, both frequently primary forest species and pioneers (*Cecropia sp., Inga sp.*) or species growing in gaps (*Pourouma sp., Pourouma bicolor*) were recorded (Estrada Tuesta, 1993; Gentry, 1996; Pennington et al., 2004). This mixed composition evidences forest disturbance probably due to previous logging activities in the area. Past vegetation inventories in undisturbed lowland Ucayali forests also reported some of the dominant species found in this study, such as *Oenocarpus bataua, Socratea exhorrhiza, Apeiba membranacea, Pseudolmedia laevigata, Iryanthera juruensis* and *Cecropia sciadophylla* (Gamarra et al., 2015). On the other hand, Ucayali secondary forests studies ranked among most common species *Inga sp., Cecropia sp., Vismia sp.* and *Nectandra sp.* (Rios Trigoso, 1990). During a secondary forest succession from two to 20 year old stands in the study area; species such as *Cecropia sp.* and *Inga sp.* were dominant in young stages of the succession; whereas *Apeiba membranaceae, Nectandra sp.* and *Hirtella tricornis* showed up later on in the succession (Kommeter, 1987). In terms of regeneration, most abundant species also were, among others, *Cecropia sp., Vismia sp., Miconia sp., Inga sp.* (ibid).

Moreover, looking at the detrended correspondence analysis to explore differences in floristic composition between plots (Figure 6), the “Y” axis component could be attributed to topographic features or flooding regimes. Whereas plots one to three are located in medium terrace forests, plot number four is based in high terrace forests with lower floodings (GRE Ucayali, 2016d). Nevertheless, species characterization did not reveal deterministic differences among plots. Species of flooded or periodically flooded lands (*Euterpe precatoria, Zygia latifolia, Parkia nitida*) were mainly found in plot number two but were actually present in all four plots; whereas species of non-flooded lands (*Pourouma sp.1, Socratea exhorriza*, etc) were also frequent in plot two but still present in all four (Gentry, 1996; Pennington et al., 2004). According to Nebel et al. (2001), Sapotaceae and Chrysobalanaceae families are most commonly found under waterlogged conditions, contrary to Lauraceae, Bombacaceae, and Meliaceae that exist in land less exposed to flooding. Such trend could not be confirmed from this study data.

In the Amazon, alpha diversity increases from east to west, as well as from south to equatorial western side of the region (Wittmann et al., 2006). Diversity is also linked with stand age, which means the later in the succession the forest is, the more chances to find higher diversity (ibid). Furthermore, biodiversity in forests is also correlated with inundation regimes; undisturbed forests in flood plains have fewer species than adjacent non flooded terra firme forests (Gamarra et al., 2015; Honorio-Coronado et al., 2015; Nebel et al., 2001; Wittmann et al., 2006). High diversity is also associated with soil factors (such as nutrients, texture, pH or drainage) (Gamarra et al., 2015; Phillips et al., 2003). Fisher alpha index expresses the species diversity in a specific sampling area; in an undisturbed Ucayali floodplain forest alpha fisher diversity was found to be 105.8 (Gamarra et al., 2015). Phillips et al. (2003) found 224 ± 39.6 and 78 ± 23.4 alpha fisher diversity values for 1 ha undisturbed forests plots in the Peruvian Amazon regions of Loreto and Madre de Dios respectively. In comparison, the disturbed sampled forest plots in this study have a significantly lower alpha fisher index.

Indeed, the first five species with highest IVI of each sampled plot and subplot accounted for more than 21% to 63% of each site's taxa, which reveals a certain level of forest homogenization (Galván et al., 2000). Coincidently, an Ucayali primary forest inventory in 1983 reported among the highest IVI species in forest terraces individuals such as *Inga sp., Hura crepitans, Guatteria sp., Nectandra sp.* (ONERN, 1983). It is interesting to point out that other commercial species with high IVI found in those primary forests were *Dipteryx odorata* (Shihuahuaco), *Hevea brasiliensis* (Shiringa)*, Coussapoa grandeseps* (Uvilla) (ibid), which were not found among the sampled plots probably as a consequence of past logging activities. On the other hand, a secondary forest research near Pucallpa (above 10 year old stands) determined *Guazuma crinite* and *Cecropia sp*. as species with the highest IVI values; though *Inga sp., Apeiba membranaceae,* *Parkia sp.* were also recorded (Galván et al., 2000). Although *Gauzuma crinite* is a commercially relevant species of secondary forests in Ucayali; remarkably, not one individual with a DBH>10cm was found in this study.

Like primary forest, disturbed forests are also suppliers of ecosystem services. In terms of provisioning; species have commercial uses or local appliances, such as rural constructions, sawed boards, low value sawn wood, etc. *Apeiba membranaceae, Parkia sp*., *Eschweilera timbuchensis, Brosimum sp., Pseudolmedia laevigata* are commercialized as sawn wood, parquet, for local construction and firewood (Galván et al., 2000; Rios Trigoso, 1990; Pennington et al., 2004). *Inga sp.* has appliances in shelter wood in agroforestry, soil regeneration, fruits provision and firewood (Rios Trigoso, 1990). *Duguetia odorata* is used in rural construction (Galván et al., 2000); Pourouma sp. for fruits and firewood (Galván et al., 2000; Rios Trigoso, 1990; Pennington et al., 2004); *Cecropia sp.* for wood pulp (Rios Trigoso, 1990); *Iryanthera juruensis* for medicine, oils, and fruits (Silva et al., 2005); etc. In general, palm species are valuable for food, oils and the trunk as construction (Pennington et al., 2004). Under proper management, and considering the rapid recuperation rate of Ucayali lowland forest (Tournon and Riva, 2001), disturbed forests could reach higher productivity and be of more interest to farmers in terms of forest use.

Moreover, secondary forests mitigate the agricultural frontier expansion under slash and burn practices into remnant forest or undisturbed primary forest; therefore, allow conservation of ecosystem services. The main purpose of young secondary vegetation in the forest landscape is mainly soil regeneration; actually, only a small proportion of them reach a later succession stage suitable for forest production or restoration (Smith, 1999). If old secondary vegetation or residual forests would be maintained, gains in biodiversity and greenhouse gas mitigation, if translated into payment for ecosystem services schemes, could become a much more competitive land use and fit under join biodiversity and climate change mitigation schemes. According to Cavanaugh et al. (2014), tropical forests carbon stocks increase congruently with taxonomic diversity. More recent studies in the Brazilian Amazon revealed that such a positive correlation between biodiversity and above ground carbon stocks is evident in secondary and disturbed primary forest; whereas in forest ecosystems above 100 Mg C ha^{-1} the relationship is disrupted (Ferreira et al., 2018).

5.1.2 Forest structure

The "J" inverted DBH distribution of the sampled forest plots is characteristic of natural lowland forests, from floodplains to pre-montane forests (Gamarra et al., 2015; Honorio-Coronado et al., 2015). Most individuals were confined among the smallest DBH classes with a slight recovery around the 30-40cm DBH class (Figure 7). Tree heights also follow a "J" inverted distribution, though in two stages. The first group corresponds to the understory and canopy layer (below 25 m trees); while the rebound reflects the emergent tree layer (>25 m) (Gamarra et al., 2015) which could probably be attributed to remnants of unlogged trees.

Basal area distribution also shows a decline in higher diameter classes, which may be the result of the ongoing forest succession and/or previous logging activities (Nebel et al., 2001). The mean basal area found in the four forest stands, 22 m^2 ha^{-1} (SE 1.4 m^2 ha^{-1}), is less than what has been found in Amazon undisturbed forests; 28.95 m^2 ha^{-1} (Baker et al., 2004) in the Peruvian South east Amazon; 29.7 m^2 ha^{-1} in Ucayali (Tournon and Riva, 2001) or 22.6-28.8 m^2 ha^{-1} in the Amazon northwest floodplains (Nebel et al., 2001). Still, it is higher than a 20 year old secondary forest in Ucayali with a basal area value of 18.9 m^2 ha^{-1} and trees less than 15 m height (Tournon and Riva, 2001). Actually, along a secondary forest succession in Ucayali, young vegetation presented an overall 6.38 (SE 1.72) m^2 ha^{-1} basal area; 8-10 year old regrowth 10.6 (SE 2.04) m^2 $ha^{-1}$and above 10 year old stands reported 11.5 (SE 2.69) m^2 ha^{-1} (Galván et al., 2000). Like the present study, all cited studies also considered individuals with DBH > 10 cm. Primary undisturbed forest volume results were hard to compare because previous inventories considered individuals with DBH > 25 cm for volume assessments (wood production volumes). However, results were higher than around 10 year old secondary stands in Pucallpa (82.4 m^3 ha^{-1}) (Galván et al., 2000) and similar to 12 year old *Gauzuma crinite* dominant stands (230.59 m^3 ha^{-1}) (Guerra Arevalo, 2008). Certainly, previous land use practices have had a negative influence on forest structural parameters such as basal area and volume (Guerra Arevalo, 2008).

5.2 Carbon stock assessment

5.2.1 From primary forests to oil palm plantations in mineral soils

Forest structural parameters are correlated with carbon stocks, as they constitute to biomass estimations (Sullivan et al., 2017). Within the four forest plots in the study site, number one had the highest tree carbon stock and tree volume; whereas forest plot number four had greater basal area and tree density. These differences might be due to the fact that, as mentioned before, forest plot number one is influenced by higher DBH classes in comparison to other plots; which consequently contributes to volume and stock estimations. Even though no replicates were available to test significant differences among means, above ground carbon stocks results fit within a comparable logged primary forest distribution in the Brazilian state of Para ca. 80-240 Mg C ha^{-1} (Ferreira et al., 2018). They were also congruent with other studies on old tropical secondary forests in Mexico 89 Mg C ha^{-1} (Orihuela-Belmonte et al., 2013) and logged forests in Borneo 83.2 ± 6.8 Mg C ha^{-1} (Berry et al., 2010); but lower than reported by Barbaran (2002) in Ucayali logged forests 121.5 Mg C ha^{-1}.[6]

When comparing the study results against Peruvian Amazon undisturbed primary forest described in the literature, significant differences were found. Pallqui et al. (2014) reported for Southeast Peruvian lowland forests 137.3 ± 10.4 Mg C ha^{-1}; Honorio-Coronado et al. (2015) estimated 90 ± 25 Mg C ha^{-1} for Northeast floodplain forests and Gamarra et al. (2015) found 112.08 Mg C ha^{-1}in Ucayali's forest floodplains. As expected, results show that logging activities on the Peruvian Amazon have an effect on aboveground carbon stocks and, at the same time, old secondary forests have a similar AG carbon stock as logged primary forests. This confirms the assumption that, in terms of carbon stocks, both long fallows and logged forest fit under the concept of disturbed forest. Even though the author could not find information on land use intensity in the plots based on management plans of past logging companies, the assumption of such forests being previously logged could not be rejected based on the study's composition, structure and carbon stocks results.

From the 140.7 (SE 5.8) Mg C ha^{-1} carbon stocks found in the disturbed forests at the study site, above ground carbon was the main contributing pool (50%); followed by soil organic carbon and necromass.

[6] When values were reported by authors in above ground biomass the default IPCC carbon content value of 0.47 IPCC was used (Vol. 4, Chp.4, IPCC, 2006).

Necromass, the sum of fallen, standing woody debris and litter, presented an overall mean of 21.19 (SE 2.52) Mg C ha^{-1}. Just accounting woody debris (19.27, SE 2.51 Mg C ha^{-1}), results turn out to be lower than encountered in forest concessions in Ucayali terra firme forests 40.22 (SE 1.6) Mg C ha^{-1} (Meza Doza, 2016) but higher than undisturbed forest floodplains in Southwest Peruvian Amazon 8.3 ± 1.1 Mg C ha^{-1} (Baker et al., 2007) and old tropical secondary forests in Mexico 6.5 Mg C ha^{-1} (Orihuela-Belmonte et al., 2013). This pool is tough to compare because tree morality not only depends on local natural circumstances such as competition, pests and diseases but also varies with logging intensity and management systems, which have not been fully unveiled. Additional conflicts in the comparison arise with methodological limitations. While some studies define coarse woody debris starting 2 cm wood diameter (Palace et al., 2012) others extend the threshold to 10 cm (Baker et al., 2007). Soil carbon stock, on the other hand, is still an understudied carbon pool in Peruvian forests. Frazão et al. (2013) reported for native forests in the state of Para, Brazil, 30.2 ± 3.4 Mg C ha^{-1} in a 30 cm depth profile; which does not differ much from the results obtained in this study (27.70, SE 1.44 Mg C ha^{-1}). Meza Doza (2016) found lower soil stocks (16.21 ± 3.2 Mg C ha^{-1}) in Ucayali terra firme forest concessions. No significant differences would be expected in soil carbon in a natural forest baseline if disturbances came mainly from logging activities; even the Intergovernmental panel on Climate Change (IPCC) considers no stock difference in such pool within forest remaining forest lands.

Regardless of their climate change contribution potential, remnant forests at the study site are at imminent risk of land conversion. As a matter of fact, according to Forest Global Watch – a global monitoring platform-, two of the 2015 sampled forests plots have already been deforested by the time this document was elaborated. Moreover, as mentioned, Ucayali's current political plan is to double the oil palm farming area in the region in the next 10 years (GRE Ucayali, 2016a). Consequently, the question as to whether the transition from disturbed forests to oil palm plantations is carbon neutral or contributes to greenhouse gas emissions needs to be addressed, if the country's ongoing climate change mitigation efforts and national and international commitments are to be respected. Previous experiments have been done mainly in Asia (Khasanah et al., 2015b; Agus et al., 2013; Germer and Sauerborn, 2008) and less extensively in Latin America (Henson, 2012). Khasanah et al. (2015a) reported a time average aboveground carbon stock for independent farmers in a mineral soil of 37.76 ± 0.33 Mg C ha^{-1}. Agus et al. (2013) and Germer & Sauerborn (2008) reported, based on literature review, 36 Mg C ha^{-1} ± 10.4 Mg C ha^{-1} and around 30 Mg C ha^{-1} respectively. Here 42.21 ± 2.1 Mg C ha^{-1} was calculated. The difference may be due to a longer rotation period considered. When including soil carbon stocks and estimated roots, the full time average carbon stock in oil palm plantations became 74.3 ± 2.2 Mg C ha^{-1}.

Going deeper into the soil dynamics over the transition, results showed significant differences in soil carbon stocks between both land use systems in a 30 cm soil profile. Whereas Guillaume et al. (2016) found higher soil carbon stocks and carbon contents in rubber vs. oil palm plantations in Indonesia; our results resemble Frazão et al. (2013), in the sense that soil carbon was significantly higher in oil palm plantations than adjacent forests (15% higher). Frazão et al. (2013) also found significant differences between plantations' ages and a decline over time (from 33.2 Mg C ha^{-1} at 4 year old to 22.7 Mg C ha^{-1} at 25). Similarly, the results of this study show higher values at the first years of the rotation (39.9, SE 3.32 Mg C ha^{-1}) which decrease at the end (29.2, SE 8 Mg C ha^{-1}) to similar values found at the forest sites. The increase in soil carbon stocks between land uses could slightly neutralized the carbon budget over the transition; nevertheless, soil contribution was considerably offset by above ground carbon debt. Inputs from above ground and below ground organic components may have contributed to the initial soil carbon peak, probably coming from the dead organic pool which was also interestingly higher at the beginning of the rotation. Deeper analysis on the decomposition contribution from other pools into the soil needs to be further addressed to better understand soil carbon dynamics over the transition. Additionally, only the first rotation has been addressed here, when probably the common slash and burn activities for land use change have a significant contribution. It would also be interesting to see if soil carbon contents tend to stabilize in future oil palm rotations. Unlike (Khasanah et al., 2015b) & Frazão et al. (2013), this study's results did not find any difference in carbon or nitrogen content between sampling sites (inter-row with and without fronds). No contribution from the stacking of fronds was evidenced in the study site.

Overall, data showed that after disturbed forest clearing and oil palm plantation installment the outcome was a carbon debt scenario. Here, only land conversion emissions were addressed, not a full life cycle assessment of oil palm production. Emissions from fertilizer production and application (N_2O emissions), methane and carbon dioxide gasses during the production of crude oil palm have been excluded. For that, a comprehensive life cycle assessment should be carried out, including the know-how on fertilization practices of small/medium oil palm farmers in Peru. CIFOR is planning to pursue such study in the next years to complete a full life cycle assessment. Still, land use change emissions are, in a logged forest predecessor system, the main net source of emissions in an oil palm life cycle assessment (Bessou et al., 2014; Chase and Henson, 2010). Hence, Chase and Henson (2010) recommend that the most effective way to reduce greenhouse gas emissions from oil palm production would be to prioritize expansion over low carbon stocked areas, followed by capturing the methane emitted during transformation processes.

Comparing among different land use systems, it is clear that the later in the succession the forest system is, the larger the carbon debt over oil palm conversion is. Germer & Sauerborn (2008) reported a 657 ± 361 Mg CO_2e ha^{-1} greenhouse gas balance on mineral soils over the transition of natural tropical forests to oil palm plantations. Emissions became much more significant if the conversion was taken over tropical peatlands (1314 ± 679 Mg CO_2e ha^{-1}) due to peat oxidation dynamics. On the other hand, if degraded grasslands were to be the previous land system, the conversion to oil palm plantations would turn out to be debt free and even a stock gain of -136 ± 37 Mg CO_2e ha^{-1} (ibid). Other studies, that include also rubber plantations in the budget, support the idea that grasslands (3.4 Mg C ha^{-1}) or shrubs (34 Mg C ha^{-1}) conversion would be the only carbon debt free transition (Khasanah et al., 2015a; Agus et al., 2013; Chase and Henson, 2010). Soy bean, a complementary crop for worldwide oil production, has a lower average aboveground carbon stock than oil palm (2.13 ± 0.67 Mg C ha^{-1}) (Bonini et al., 2018); which means that among oil producing crops competitors, oil palm has not only the higher yield per hectare but has also the less carbon debt.

Moreover, higher yield oil palm production does not necessarily reduce the pressure over natural forests. A remote sensing exercise over the period of 2000 to 2010 in the Peruvian Amazon revealed that even though high yield oil palm production has occupied less land than smallholders production systems, 72% of its expansion has been over forested areas, mainly to avoid land tenure conflicts (Gutiérrez-Vélez et al., 2011). Land intensification alone is not necessarily associated with less deforestation (Porro et al., 2014). Hence, high yield oil palm production may contribute to land sparing, but only if accompanied with incentives that promote transition over highly degraded lands and reinforce no intervention over natural forests (Gutiérrez-Vélez et al., 2011).

5.2.2 Oil palm models in the literature

Aside from the litter pool, results showed that carbon stocks along an oil palm rotation tend to logarithmically increase or decrease depending on the vegetative component. In terms of aboveground biomass, Henson (2017) confirmed a linear increment with age up to at least 27 year-old plantations. When mature, either the stock is maintained until clearance or decreased due to palm mortality (Henson, 2012). This study's results followed a similar dynamic, though with some additional variability. The chosen allometric model (Khasanah et al., 2015a) also depends on palm height, which is influenced by planting density. Hence, the heterogeneous planting density among smallholders had an effect over height vs. age models (Henson, 2006). Furthermore, Henson (2017) also observed a difference in oil palm aboveground biomass dynamic depending on the type of management followed. Likewise, Khasanah et al. (2015a) reported a significant difference over growth models between plantation managements, with independent farmers resulting in less oil palm carbon stocks. Small/medium holders may not always have access to technology nor training; resulting in heterogeneous planting density, over/under fertilization, lack of irrigation, among others. Such practices do not only affect the carbon cycle, but the nitrogen cycle as well; over fertilization rate and drainage are the main contributors to N losses in oil palm mature state plantations (Pardon et al., 2017).

Similar to our results, in terms of other pools, Henson (2012) also reported a decrease of understory over time. As expected, kudzu or other vegetation that are dominant at younger stages start to disappear as the canopy closes. On the contrary, even if an increase in litter has been reported in the literature (ibid), no trend was evident among the cronosequence study. Khasanah et al. (2015a) also revealed a negative trend over time in necromass attributed to previous forest decomposition.

Most studies on oil palm growth models have been mainly developed in the Southeast Asian continent. Oil palm yield is affected by age, type of management and agro ecological site characteristics (mainly associated with temperature, soils and precipitation) (MINAGRI, 2016). All of them vary around oil palm production sectors in Peru, with Tocache (San Martin) registering the highest yields (ibid). It would be interesting to see if these oil palm models could be replicated in other oil palm production sites in Latin America or Peru, in regions such as San Martin, Loreto and Huánuco, considering different management systems and site characteristics.

VI. Conclusions and recommendations

6.1 Conclusions

Overall, the study revealed that a disturbed forest had a significantly higher total carbon stock than what an oil palm plantation could reach over a 30 year rotation period; turning this land use transition into a carbon debt scenario. Such research is of especial relevance to Peru because it supports, with ground evidence, that logged or old growth secondary forests have a climate change mitigation potential if not converted to oil palm plantations. The timing of such results is crucial, especially because both national and regional stakeholders have expressed the political will to sustainably expand the oil palm farming frontier over degraded lands; which according to this study's results, should exclude such degraded forests.

First of all, the composition and structural characterization of the studied forests support the hypothesis of previous disturbance at the site, probably due to past logging activities. In terms of composition, both remnant species from primary forest as well as pioneers or growing in gap species were found. Even though there were wood valuable species at the site (Myristicaceae, Myrtaceae, Lauraceae), the majority was medium-fast growing species such as *Oenocarpus bataua, Socratea exhorriza, Cecropia sp.1 & Cecropia sciadophylla, Euterpe precatoria.* Moreover, IVI results exposed some level of forest homogenization, which evidently coincide with lower biodiversity index vs. referential lowland primary forests. Most importantly, comparisons with previous primary forest inventories in Ucayali's lowlands point out the clear absence of some commercial species (i.e. *Dipteryx odorata, Hevea brasiliensis, Coussapoa grandeseps, Cedrelinga catenaeformis, Cedrela odorata, Chorisia sp.*) and maintenance of just few (*Hura crepitans, Virola sp., Iryanthera sp.),* backing up the historical logging premise.

Logging activities have had also an effect on the forest structural component. Basal area, for instance, was found to be lower than at undisturbed Amazon forests, though still higher then regrowth vegetation stands. In general, DBH and height distributions followed the natural J inverted distribution with a small rebound on higher diameter classes, which could be attributed to previous unlogged trees. Although some differences have might been noticed between forest plots in terms of species composition or structural parameters (probably due to land use intensities), no deterministic discrepancies were ultimately found, which reflects that they all were representative replicates from the current forest landscape in Campo Verde, Ucayali.

Furthermore, forest carbon stocks after logging were proven to be significantly reduced in comparison to reference primary forests baselines in the Peruvian Amazon. The literature refers to over 90 Mg C ha^{-1}AGC stocks in Peruvian lowland forest, this study results determined 71.3 (SE 4.23) Mg C ha^{-1} in disturbed forests, similar to those found in old secondary forest stands. Indeed, what is clear is that degraded forest still have a significantly higher total carbon stock (140.7, SE 5.8 Mg C ha^{-1}) than an oil palm plantation time average stock (74.3 ± 2.2 Mg C ha^{-1}). As a matter of fact, along the whole 30 year rotation; most pools have a higher stock in forests than in adjacent oil palm plantations, with soil organic carbon being the only exception. Nevertheless, as AG carbon stock is the main carbon contributor, the difference in soil organic carbon between land uses does not offset the other stocks. Litter and understory were the least significant pools in both land uses. Whereas in oil palm plantations soils, no contribution in carbon or nitrogen content was shown from stacking fronds, as no significant differences were proven between sampling sites.

Oil palm stock growth models per pool presented similar trends as found in literature review. Whereas AGB and fronds stocks increased over the rotation period; dead wood, understory and soil carbon stocks decreased over time. The litter component, on the other hand, did not show any significant tendency. Such trends are also influenced by farmer's management techniques, like planting density affecting palm height and therefore, biomass. These results are representative of smallholders in Campo Verde, complementary studies in oil palm industrial plantations are pending.

It is apparent that Peru has set as an objective to increase oil palm production to satisfy not only the national demand but also embrace the international markets. Most local stakeholders have pledged for sustainability and no deforestation of primary forest, and for promoting the expansion of the crop over degraded lands. However, the definition of such degraded lands remains unclear and the results of this study support the hypothesis that, in terms of greenhouse gasses, disturbed forests should not be converted into oil palm plantations. Conversion should be directed to highly degraded lands such as pastures and shrubs, where further studies should be assessed. Remnant forests and old secondary forests are not only relevant for the mitigation of greenhouse gas emissions but also for forest adaption synergic strategies. They are positively correlated with biodiversity and unlike monoculture, they are important for the connectivity of ecosystems functions. For instance, they would be of great advantage as natural pest controls to the threats already affecting oil palm producers in the area. Furthermore, they could be subject of forest management, as some of the present species could be of interest of productive use of local populations. Therefore, even though increasing yield in existing oil palm plantations could help control further deforestation, if expansion is inevitable, it should stick to low carbon stock lands where the transition would result in carbon gains. Additionally, such practices would be a step on the way of compliance with international sustainable standards and certifications that rule for no deforestation and discourage oil palm production in high carbon stock lands.

6.2 Recommendations

Even though carbon loss from forest conversion has been documented to be the main source of emissions over an oil palm lifecycle assessment, it would be necessary to study emissions resulting from other stages of the crop production or transformation in Peru. For instance, CIFOR plans to assess N_2O emissions resulting from N fertilizer application. In order to do so, a deeper study on fertilization regimes needs to be completed among smallholders, as such practices highly depend on management systems. Comparisons between industrial and smallholder producers would be interesting to fulfill as well, as other authors in Asia have found differences in greenhouse gas emissions between management systems.

Further analysis on soil decomposition and other contributing pools, such as dead organic matter, could be performed to better understand the soil carbon peak at the beginning of the oil palm rotation. A question that also arises is if such trend is only particular in land use change (and probably slash and burn practices) or whether soil stock would stabilize or decline in following rotations. On the other hand, oil palm research has been vastly developed in the Asian continent. As the crop continues to expand in Africa and South America, research on the matter should also continue to grow in Latin America. Other sites in Peru with different yield performances could be taken as example.

Lastly, in terms of lessons learned among methodological aspects that could improve the performance of future studies one could consider georeferencing trees when establishing nested plots, doing direct measurements of tree heights, evaluating palm trunks instead of total heights. Such field protocol recommendations could improve data analysis and interpretation but, perhaps, at the expense of time. Furthermore, replicating the chronosequence study in disturbed forests to represent growth and decay would allow to compare both land uses dynamics. Additionally, including plots on undisturbed primary forests would be of high value to represent the entire transition with primary information, from degradation to land use change. For the purpose of forest composition analysis, for instance, it would strongly improve results as most old inventories in Ucayali primary forests had a strong limitation in species identification with common names; having to roughly later attribute a scientific name.

VII. References

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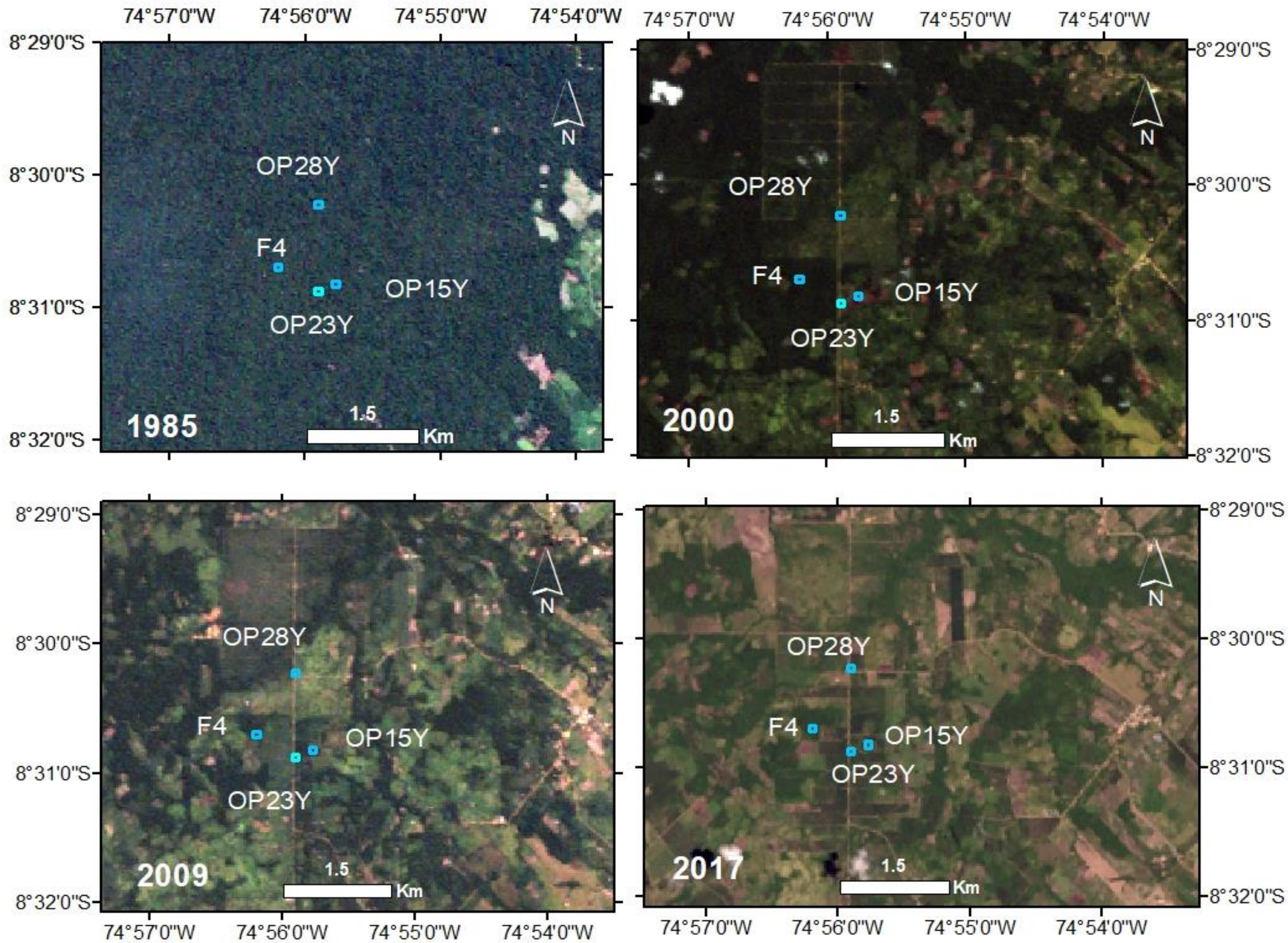
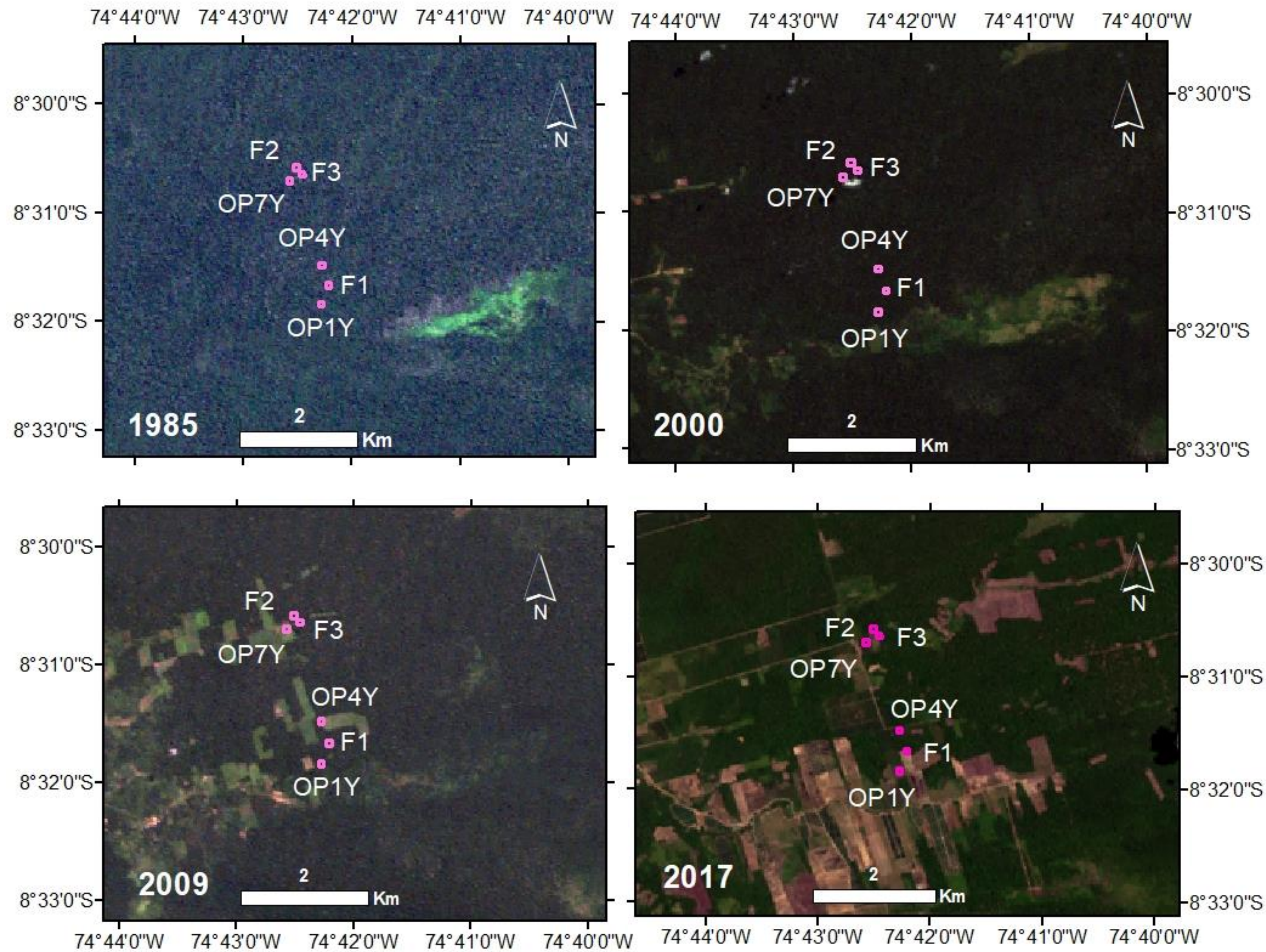
VIII. ANNEX

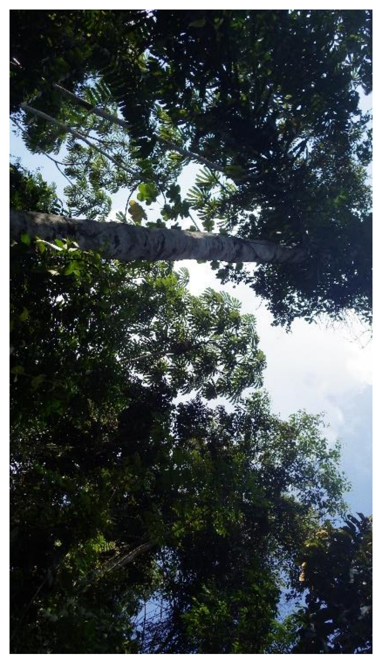
Annex 1. Forest disturbance transition over a 20 year old period at the study sites of Nuevo San Pedro using Landsat 5TM and Landsat 8 OLI satellite imagery in RGB.



Annex 2. Forest disturbance transition over a 20 year old period at the study sites of Tupac Amaru Limon using Landsat 5TM and Landsat 8 OLI satellite imagery in RGB.



Annex 3. Sampling sites

		
1-year old palm	4-year old palm	7-year old palm
		
15-year old palm	28-year old palm	Disturbed forest © 2015 Kristell Hergoualc´h

Annex 4. Sampling methods

Aboveground biomass – oil palm heights with clinometer

Aboveground biomass –distance to estimate oil palm height

Fronds

Soil pits in inter-rows with fronds

Soil samples at three depths

Fallen dead wood – counts and diameter
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Fallen dead wood – specific gravity samples
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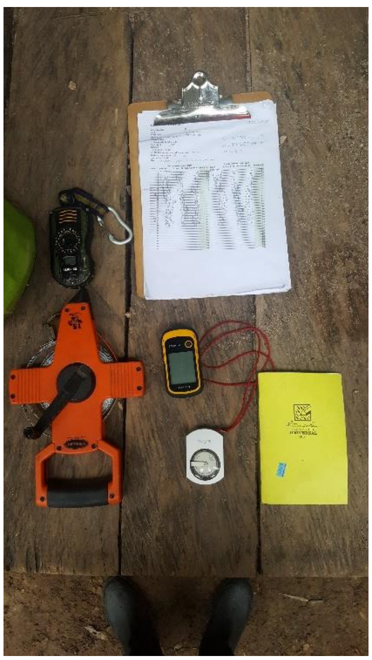

Field instruments

Understory
© 2015 Kristell Hergoualc´h

Annex 5. List of species and IVI per plot for big trees (DBH ≥ 30 cm)

Species	Family	IVI			
		F1	F2	F3	F4
Apeiba membranaceae	Malvaceae	0	0	12	12
Apeiba sp. 1	Malvaceae	7	11	0	0
Annonaceae sp.	Annonaceae	6	0	0	0
Attalea butyracea	Arecaceae	17	14	0	0
Brosimum sp. 1	Moraceae	13	28	16	9
Byrsonima sp. 1	Malpighiaceae	0	0	0	7
Calycophyllum sp. 1	Rubiaceae	0	0	0	7
Castilla ulei	Moraceae	8	0	0	0
Cecropia latiloba	Urticaceae	0	12	8	0
Cecropia sp. 1	Urticaceae	0	0	0	7
Ceiba sp. 1	Malvaceae	0	0	9	0
Chrysobalanaceae sp.	Chrysobalanaceae	6	0	9	0
Couratari guianensis	Lecythidaceae	7	0	0	0
Coussapoa cf. cupularis	Urticaceae	0	0	0	6
Dypterix micrantha	Fabaceae	6	0	0	0
Euphorbiaceae sp.	Euphorbiaceae	0	0	0	7
Eschweilera sp. 1	Lecythidaceae	0	0	0	8
Eschweilera timbuchensis	Lecythidaceae	0	0	37	0
Fabaceae sp.	Fabaceae	8	15	0	11
Garcinia macrophylla	Clusiaceae	6	0	0	0
Garcinia sp. 1	Clusiaceae	0	0	11	0
Genipa americana	Rubiaceae	7	0	0	0
Guarea sp. 1	Meliaceae	0	0	0	11
Guatteria sp. 1	Annonaceae	0	22	0	8
Hieronima sp. 1	Phyllantaceae	0	0	0	14
Hura crepitans	Euphorbiaceae	27	12	10	12
Hyeronima alchorneoides	Phyllantaceae	0	18	9	0
Inga sp.	Fabaceae	10	0	0	9
Iryanthera sp. 1	Myristicaceae	0	0	9	0
Lauraceae sp.	Lauraceae	13	0	9	0
Lecythidaceae sp.	Lecythidaceae	6	0	0	0
Matayba inelegans	Sapindaceae	9	0	0	0
Matisia malacocalyx	Malvaceae	18	0	10	0
Meliaceae sp.	Meliaceae	6	0	0	0

Species	Family	IVI			
		F1	F2	F3	F4
Micrandra sp. 1	Euphorbiaceae	7	0	0	0
Micropholis venulosa	Sapotaceae	0	0	0	7
Moraceae sp.	Moraceae	12	0	17	14
Myristicaceae sp.	Myristicaceae	0	0	0	8
Myrcia sp. 1	Myrtaceae	16	0	0	0
NN	NN	23	11	22	54
Neea sp. 1	Nyctaginaceae	0	11	0	8
Oenocarpus bataua	Arecaceae	0	0	0	8
Pachira sp. 1	Malvaceae	0	21	0	0
Parkia sp. 1	Fabaceae	8	0	18	13
Pourouma bicolor	Urticaceae	0	0	0	9
Pourouma guianensis	Urticaceae	0	0	8	0
Pourouma sp. 1	Urticaceae	0	0	0	7
Pouteria cf. bilocularis	Sapotaceae	0	0	0	8
Pouteria sp. 1	Sapotaceae	0	0	0	7
Pouteria trilocularis	Sapotaceae	0	22	26	0
Protium sp. 1	Burseraceae	0	0	8	0
Pseudolmedia laevigata	Moraceae	0	0	0	9
Pterocarpus rohrii	Fabaceae	8	0	0	0
Ruizodendron ovale	Annonaceae	11	0	0	0
Simarouba amara	Simaroubaceae	6	0	8	0
Sorocea sp. 1	Moraceae	12	23	12	8
Tachigali guianensis	Fabaceae	0	0	0	10
Virola sp. 1	Myristicaceae	0	0	13	0
Zygia latifolia	Fabaceae	16	78	18	0
TOTAL		**300**	**300**	**300**	**300**

List of species and IVI per plot for small trees (10≤ DBH <30cm)

Species	Family	IVI			
		F1	F2	F3	F4
Abuta grandifolia	Menispermaceae	0	0	3	0
Apeiba membranaceae	Malvaceae	0	3	4	6
Astrocaryum murumuru	Arecaceae	4	0	0	0
Attalea butyracea	Arecaceae	10	5	0	0
Batocarpus sp. 1	Fabaceae	0	0	6	0
Brosimum alicastrum	Moraceae	12	0	0	0
Brosimum sp. 1	Moraceae	4	4	3	6
Byrsonima crispa	Malpighiaceae	0	0	0	3
Byrsonima sp. 1	Malpighiaceae	0	0	0	5
Calycophyllum spruceanum	Rubiaceae	0	0	3	0
Calyptranthes speciosa	Myrtaceae	0	7	0	3
Casearia sp. 1	Salicaceae	3	0	0	0
Castilla ulei	Moraceae	5	0	0	0
Cecropia sciadophylla	Urticaceae	0	0	0	10
Cecropia sp. 1	Urticaceae	0	4	0	5
Cecropia sp. 2	Urticaceae	3	0	0	0
Celtis schippii	Ulmaceae	4	0	0	0
Chrysobalanaceae sp.	Chrysobalanaceae	0	0	0	3
Chrysophyllum venezuelanense	Sapotaceae	0	0	3	0
Clarisia racemosa	Moraceae	0	0	3	0
Cordia sericicalyx	Boraginaceae	3	0	0	0
Couepia racemosa	Chrysobalanaceae	0	4	0	0
Couepia sp. 1	Chrysobalanaceae	0	5	0	0
Dacryodes peruviana	Burseraceae	4	0	0	0
Dendropanax arboreus	Anacardiaceae	0	4	0	0
Duguetia odorata	Annonaceae	9	0	3	0
Ecclinusa guianensis	Sapotaceae	0	0	7	0
Endlicheria aff. verticillata	Lauraceae	0	7	0	0
Eschweilera coriacea	Lecythidaceae	0	0	0	6
Eugenia sp. 2	Myrtaceae	4	0	0	0
Euterpe precatoria	Arecaceae	19	28	24	4
Euphorbiaceae sp.	Euphorbiaceae	0	0	0	3
Fabaceae sp.	Fabaceae	7	4	4	11
Garcinia aff. macrophylla	Clusiaceae	0	0	4	0
Garcinia macrophylla	Clusiaceae	0	0	8	0
Glycydendron amazonicum	Euphorbiaceae	0	0	0	5
Guapira sp. 1	Nyctaginaceae	3	0	0	0
Guarea macrophylla	Meliaceae	9	0	3	0

Species	Family	IVI			
		F1	F2	F3	F4
Guarea pubescens	Meliaceae	0	0	0	3
Guatteria glauca	Annonaceae	0	0	0	8
Guatteria pilosula	Annonaceae	0	3	4	0
Guatteria sp. 1	Annonaceae	0	4	0	3
Hebepetalum humirifolia	Linaceae	0	0	3	0
Hebepetalum humiriifolium	Linaceae	0	0	0	5
Heisteria nitida	Olacaceae	4	0	0	0
Hirtella tricornis	Chrysobalanaceae	6	0	0	0
Inga acrocephala	Fabaceae	4	0	0	0
Inga sp. 1	Fabaceae	13	0	4	3
Inga sp. 2	Fabaceae	0	3	6	0
inga sp. 3	Fabaceae	0	0	4	0
Inga sp. 4	Fabaceae	0	0	3	0
Iryanthera juruensis	Myristicaceae	0	0	14	6
Iryanthera sp. 1	Myristicaceae	4	0	0	0
Laetia procera	Salicaceae	0	0	0	3
Lauraceae sp.	Lauraceae	0	0	0	3
Leonia glycycarpa	Violaceae	0	4	0	0
Leonia sp. 1	Violaceae	4	0	0	0
Licania sp. 1	Chrysobalanaceae	0	0	0	3
Mabea speciosa	Euphorbiaceae	0	0	3	0
Matayba inelegans	Sapindaceae	9	0	0	0
Matisia bracteolosa	Malvaceae	4	0	0	0
Maytenus macrocarpa	Chrysobalanaceae	4	0	0	0
Meliaceae sp.	Meliaceae	0	4	0	3
Miconia aff. argyrophylla	Melastomataceae	0	0	3	0
Miconia aff. matthaei	Melastomataceae	4	0	3	0
Miconia aff. tetragona	Melastomataceae	0	0	3	0
Miconia myriantha	Melastomataceae	0	0	0	3
Miconia sp. 1	Melastomataceae	0	0	0	3
Miconia sp. 3	Melastomataceae	3	0	0	0
Micropholis trunciflora	Sapotaceae	0	0	0	3
Moraceae sp.	Moraceae	4	0	4	7
Mouriri vernicosa	Memecylaceae	0	0	0	3
Myrcia sp. 1	Myrtaceae	0	7	0	0
Naucleopsis sp. 1	Moraceae	5	0	0	0
Nealchornea yapurensis	Euphorbiaceae	4	0	4	0
Nectandra sp. 1	Lauraceae	0	4	4	4
Neea divaricata	Nyctaginaceae	0	3	3	0
NN	NN	46	41	22	65

Species	Family	IVI			
		F1	F2	F3	F4
Ocotea aff. cernua	Lauraceae	0	3	0	0
Oenocarpus bataua	Arecaceae	0	0	48	10
Ormosia sp. 1	Fabaceae	0	0	4	4
Parkia nitida	Fabaceae	4	15	0	0
Perebea sp. 1	Moraceae	0	0	3	0
Pourouma bicolor	Urticaceae	0	3	0	5
Pourouma guianensis	Urticaceae	0	7	0	0
Pourouma minor	Urticaceae	0	0	0	3
Pourouma sp. 1	Urticaceae	0	12	10	5
Pouteria bangii	Sapotaceae	0	0	5	0
Pouteria caimito	Sapotaceae	4	0	0	0
Pouteria sp. 1	Sapotaceae	4	4	4	3
Pouteria sp. 5	Sapotaceae	4	0	0	0
Pouteria trilocularis	Sapotaceae	0	8	0	0
Protium nodulosum	Burseraceae	0	0	0	6
Protium sagotianum	Burseraceae	0	0	0	6
Protium tenuifolium	Burseraceae	5	0	0	0
Pseudolmedia laevigata	Moraceae	0	0	0	18
Psychotria japurensis	Rubiaceae	3	0	0	0
Rinorea flavescens	Violaceae	0	0	3	0
Roucheria punctata	Linaceae	0	0	0	3
Ruizodendron ovale	Annonaceae	4	0	0	0
Sapium marmieri	Euphorbiaceae	0	0	4	0
Sapindaceae sp.	Sapindaceae	0	0	0	3
Sapotaceae sp.	Sapotaceae	0	0	0	3
Simarouba amara	Simaroubaceae	0	3	0	0
Siparuna cuspidata	Siparunaceae	0	0	3	0
Socratea exhorriza	Arecaceae	7	15	4	0
Solanum sessiliflorum	Solanaceae	0	3	3	0
Sorocea hirtella	Moraceae	0	6	0	0
Sorocea muriculata	Moraceae	4	0	0	0
Sorocea sp. 1	Moraceae	0	8	18	3
Sterculia sp. 1	Malvaceae	3	0	0	0
Tachigali macbridei	Fabaceae	0	0	0	3
Tachigali paniculata	Fabaceae	0	0	0	3
Talisia cerasina	Sapindaceae	5	0	0	0
Tapirira guianensis	Anacardiaceae	0	0	4	3

Species	Family	IVI			
		F1	F2	F3	F4
Tapura guianensis	Dichapetalaceae	0	0	3	0
Terminalia amazonica	Combretaceae	0	8	3	0
Theobroma glaucum	Malvaceae	9	0	0	0
Theobroma sp. 1	Malvaceae	0	0	0	3
Theobroma speciosum	Malvaceae	0	4	0	0
Tovomita sp. 1	Clusiaceae	0	0	0	6
Trichilia sp. 1	Meliaceae	0	0	0	3
Ulmaceae sp.	Ulmaceae	4	0	0	0
Vine sp.	Vine	7	0	0	0
Virola calophylla	Myristicaceae	6	0	0	0
Virola pavonis	Myristicaceae	0	5	0	0
Virola sp. 1	Myristicaceae	0	4	0	6
Virola sp. 2	Myristicaceae	0	0	6	0
Vismia baccifera	Hypericaceae	0	4	0	0
Vismia macrophylla	Hypericaceae	0	0	4	0
Zygia latifolia	Fabaceae	8	38	0	5
Zygia sp. 1	Fabaceae	0	0	3	0
TOTAL		**300**	**300**	**300**	**300**

Annex 6. Carbon stocks (Mg C ha^{-1}) for all replicates and plots

Carbon pool	transect	circular plot	Forest				Oil palm					
			F1	F2	F3	F4	OP1Y	OP4Y	OP7Y	OP15Y	OP23Y	OP28Y
Aboveground carbon			79.24	61.84	66.55	77.58	9.31	9.21	15.20	41.94	30.98	41.47
Fronds	1		-	-	-	-	0.00	0.88	4.02	8.66	4.98	5.87
Fronds	2		-	-	-	-	0.00	0.36	2.10	9.11	5.94	1.85
Fronds	3		-	-	-	-	0.00	0.35	1.30	12.42	5.23	3.57
Dead wood standing			6.09	2.77	2.10	3.09	-	-	-	-	-	-
Dead wood fallen	1	1	14.58	54.53	1.66	12.62	29.45	34.23	70.93	0.09	0.02	0.71
Dead wood fallen	1	2	8.74	31.42	18.79	16.02	3.51	7.01	0.71	0.03	0.04	0.82
Dead wood fallen	1	3	14.65	48.88	17.50	17.96	1.74	12.90	0.60	0.04	0.04	0.39
Dead wood fallen	2	1	6.31	26.50	6.67	74.40	0.34	14.66	0.65	0.31	0.07	0.42
Dead wood fallen	2	2	11.87	18.33	6.24	23.73	3.15	3.42	0.46	1.08	0.10	0.00
Dead wood fallen	2	3	12.55	17.91	5.21	11.47	2.40	13.90	0.83	0.16	0.04	0.23
Dead wood fallen	3	1	0.99	10.95	6.13	6.99	7.48	7.01	0.14	0.01	0.00	0.32
Dead wood fallen	3	2	11.38	16.80	3.93	5.74	10.15	21.66	0.09	0.02	0.00	1.64
Dead wood fallen	3	3	1.85	4.70	9.61	9.50	7.02	14.89	0.17	0.01	0.00	0.22
Litter	1		2.56	1.10	1.19	2.53	0.16	1.42	0.32	0.33	0.23	0.13
Litter	2		1.23	1.60	1.17	2.39	0.53	0.90	0.39	0.30	0.16	0.21
Litter	3		2.14	3.30	1.40	2.47	0.07	0.66	0.33	0.09	0.14	1.14
Understory	1		1.41	0.93	0.37	0.47	0.95	1.67	0.08	0.07	0.13	0.23
Understory	2		0.25	2.25	0.15	0.61	0.59	1.03	0.22	0.03	0.13	0.27
Understory	3		0.92	0.44	0.39	0.67	1.28	0.39	0.30	0.02	0.09	1.26
Roots			18.62	14.53	27.66	18.23	5.68	5.62	4.41	12.16	5.89	7.88
Soil	1	F	48.20	14.01	23.73	33.22	39.05	47.84	26.20	50.58	17.06	28.56
Soil	1	NF	28.73	11.80	22.95	25.24	47.22	26.42	25.82	22.61	18.35	28.08
Soil	2	F	30.58	27.12	29.39	28.22	25.45	53.38	33.52	38.75	27.80	32.79
Soil	2	NF	28.36	36.85	33.06	26.66	50.09	41.34	40.09	15.41	34.35	26.25
Soil	3	F	32.12	30.17	27.29	34.02	29.21	39.75	31.12	48.04	44.03	27.81
Soil	3	NF	24.81	22.28	19.71	26.26	32.50	34.92	40.56	18.79	24.57	31.63

Annex 7. Yearly oil palm carbon stocks (Mg C ha^{-1}) per pool estimated by growth models

Aboveground carbon -AGC-

AGC (Mg C ha^{-1}) = 13.27*LN(Stand age)

Age (year)	AGC (Mg C ha^{-1})	Standard error
1	0	0
2	9.20	1
3	14.58	2
4	18.40	2
5	21.36	3
6	23.78	3
7	25.82	3
8	27.59	4
9	29.16	4
10	30.56	4
11	31.82	4
12	32.97	4
13	34.04	4
14	35.02	5
15	35.94	5
16	36.79	5
17	37.60	5
18	38.36	5
19	39.07	5
20	39.75	5
21	40.40	5
22	41.02	5
23	41.61	5
24	42.17	6
25	42.71	6
26	43.23	6
27	43.74	6
28	44.22	6
29	44.68	6
30	45.13	6

Fronds

Fronds (Mg C ha^{-1}) = 1.83*LN(Stand age)

Age (year)	Fronds (Mg C ha^{-1})	Standard error
1	0	0
2	1.27	0
3	2.01	0
4	2.54	0
5	2.95	0
6	3.28	1
7	3.56	1
8	3.81	1
9	4.02	1
10	4.21	1
11	4.39	1
12	4.55	1
13	4.69	1
14	4.83	1
15	4.96	1
16	5.07	1
17	5.18	1
18	5.29	1
19	5.39	1
20	5.48	1
21	5.57	1
22	5.66	1
23	5.74	1
24	5.82	1
25	5.89	1
26	5.96	1
27	6.03	1
28	6.10	1
29	6.16	1
30	6.22	1

Fallen dead wood -DW- DW (Mg C ha^{-1}) = -3.23*LN(Stand age)+11.85		
Age (year)	DW (Mg C ha^{-1})	Standard error
1	11.85	3
2	9.61	4
3	8.30	5
4	7.37	5
5	6.65	5
6	6.06	6
7	5.56	6
8	5.13	6
9	4.75	6
10	4.41	6
11	4.10	6
12	3.82	6
13	3.57	7
14	3.33	7
15	3.10	7
16	2.89	7
17	2.70	7
18	2.51	7
19	2.34	7
20	2.17	7
21	2.02	7
22	1.87	7
23	1.72	7
24	1.58	7
25	1.45	7
26	1.33	7
27	1.20	8
28	1.09	8
29	0.97	8
30	0.86	8

Understory –U- U (Mg C ha^{-1}) = -0.23*LN(Stand age)+0.97		
Age (year)	Understory (Mg C ha^{-1})	Standard error
1	0.97	0
2	0.81	0
3	0.72	0
4	0.65	0
5	0.60	0
6	0.56	0
7	0.52	0
8	0.49	0
9	0.46	0
10	0.44	0
11	0.42	0
12	0.40	0
13	0.38	0
14	0.36	0
15	0.35	0
16	0.33	0
17	0.32	0
18	0.31	0
19	0.29	0
20	0.28	0
21	0.27	0
22	0.26	0
23	0.25	1
24	0.24	1
25	0.23	1
26	0.22	1
27	0.21	1
28	0.20	1
29	0.20	1
30	0.19	1

Soil **Soil (Mg C ha^{-1}) = -3.15*LN(Stand age)+39.90**		
Age (year)	**Soils Mg C ha^{-1})**	**Standard error**
1	39.9	3
2	37.72	4
3	36.44	5
4	35.53	5
5	34.83	6
6	34.26	6
7	33.77	6
8	33.35	6
9	32.98	6
10	32.65	7
11	32.35	7
12	32.07	7
13	31.82	7
14	31.59	7
15	31.37	7
16	31.17	7
17	30.98	7
18	30.80	7
19	30.63	7
20	30.46	7
21	30.31	8
22	30.16	8
23	30.02	8
24	29.89	8
25	29.76	8
26	29.64	8
27	29.52	8
28	29.40	8
29	29.29	8
30	29.19	8

Root (Mg C ha^{-1}) = 0.61*AGC **Root (Mg C ha^{-1}) = 0.29*AGC** **Root (Mg C ha^{-1}) = 0.19*AGC**		**Age < 7** **7≤ Age < 20** **Age ≥ 20**
Age (year)	**Root (Mg C ha^{-1})**	**Standard error**
1	0	0
2	5.61	1
3	8.89	1
4	11.22	1
5	13.03	2
6	14.50	2
7	7.49	1
8	8.00	1
9	8.46	1
10	8.86	1
11	9.23	1
12	9.56	1
13	9.87	1
14	10.16	1
15	10.42	1
16	10.67	1
17	10.90	1
18	11.12	1
19	11.33	1
20	7.55	1
21	7.68	1
22	7.79	1
23	7.91	1
24	8.01	1
25	8.12	1
26	8.21	1
27	8.31	1
28	8.40	1
29	8.49	1
30	8.58	1