Heterogeneity of tree diversity and carbon stocks in Amazonian oil palm landscapes

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Abstract

Background: Quantitative effects of large-scale oil palm expansion in the Neotropics on biodiversity and carbon stocks are still poorly documented.

Aims: We evaluated differences in tree species composition and richness, and above-ground carbon stocks among dominant land cover types in Pará state, Brazil.

Methods: We quantified tree species composition and richness and above-ground carbon stock in stands in remnant primary rain forest, young secondary forest, oil palm plantation and pastures.

Results: We sampled 5,696 trees with a DBH ≥ 2 cm, of 413 species in 68 families, of which 381 species were recorded in primary forest fragments. We found significant differences in species richness and carbon stock among the four land cover classes. Carbon stocks in remnant primary forest were typically over 190 Mg ha^{-1}, while those in other land cover types were typically less than 60 Mg ha^{-1}.

Conclusion: Oil palm plantations have a species-poor tree community given active management; old plantations have a standing carbon stock which is comparable to that of secondary forest and far greater than that of pastures. Private forest reserves within oil palm company holdings play an important role in preserving primary forest tree diversity in human-modified landscapes in Amazonia.

Keywords: land cover; plantations; biofuel; community composition; arboreal biomass

Introduction

Driven by increasing demand for food and biofuel, the expansion in the acreage of oil palm (Elaeis guineensis Jacq.) has become a major driver of loss of primary forest habitats globally (Fitzherbert et al. 2008). Oil palm is the most important vegetable oil in terms of worldwide quantity of production and now accounts for about 10% of the world's permaculture area (Pirker et al. 2016). Forest conversion to oil palm plantation has most acutely impacted Indonesia and Malaysia. For example, an analysis of FAO data has found that, during the period from 1990 to 2005, more than half of oil palm expansion in Malaysia occurred at the expense of forests (Koh and Wilcove 2008). Recent decades have seen expansion of oil palm plantations in Africa and South America (Butler and Laurence 2009). In particular, the Amazon basin has been earmarked as the next major frontier for expansion (Reis and Guzmán 2015). The state of Pará is currently responsible for 95% of the palm oil production in Brazil.

The regional agro-ecological zoning exercise for oil palm (ZAE-Dendê) has identified existing deforested and degraded areas as targets for oil palm planting (Brasil 2010). The official Brazilian program for oil palm expansion (PPSOP) has been justified, in addition to its catalysing regional development, by evoking that perennial crops such as oil palm may help restore soil quality, protect against soil erosion and contribute to carbon sequestration (Homma et al. 2000). However, an increase in the oil palm acreage has generated concerns that expansion may come at the expense of land for existing agricultural production, leading to opening up more areas for agriculture and thus indirectly resulting in deforestation (Butler and Laurence 2009). Ninety per cent of this expansion in oil palm production occurred on former pasture land rather than forest and direct conversion of primary forest to oil palm declined from 4% from 2006–2010, to less than 1% between 2010–2014 across 50,000 km2 of eastern Pará (Benami et al. 2018).

There is a historical landscape degradation associated with oil palm expansion in the eastern Amazon region (Almeida et al. 2020). The impacts of this expanding oil palm acreage on Amazonian biodiversity have been little studied. However, it is known that plantations offer poor habitats for birds (Lees et al. 2015) and mammals (Mendes-Oliveira et al. 2017) and reduce the integrity of aquatic ecosystems (Cunha et al. 2015). Before large-scale planting of oil palm is undertaken solely on the basis of statements that it is a relatively green source of energy with low environmental impacts, a thorough assessment of environmental impacts is required (Yusoff 2006; Basiron 2007; Tan et al. 2009; Moni and Sulaiman 2013). In Brazil, there has been pressure since 2005 by diverse stakeholders (including politicians and producers) to legally declare oil palm plantations a ‘low impact’ form of land use. If such a definition were accepted, it could greenlight oil palm planting instead of restoring illegally deforested land parcels with native vegetation which land-owners are currently required to do (Lees and Vieira 2013). The legally mandated maintenance of native forests on private landholdings (so-called ‘legal reserves’ amounting to 80% of a property in most parts of Amazonia) is essential for maintaining a multitude of ecosystem services (incl. carbon sequestration) and conserving biodiversity. Some of the largest native forest remnants in eastern Pará are found on private properties, but the value of these forests is undermined by their being subject to disturbance from fire and logging (Berenguer et al. 2014; Moura et al. 2014).

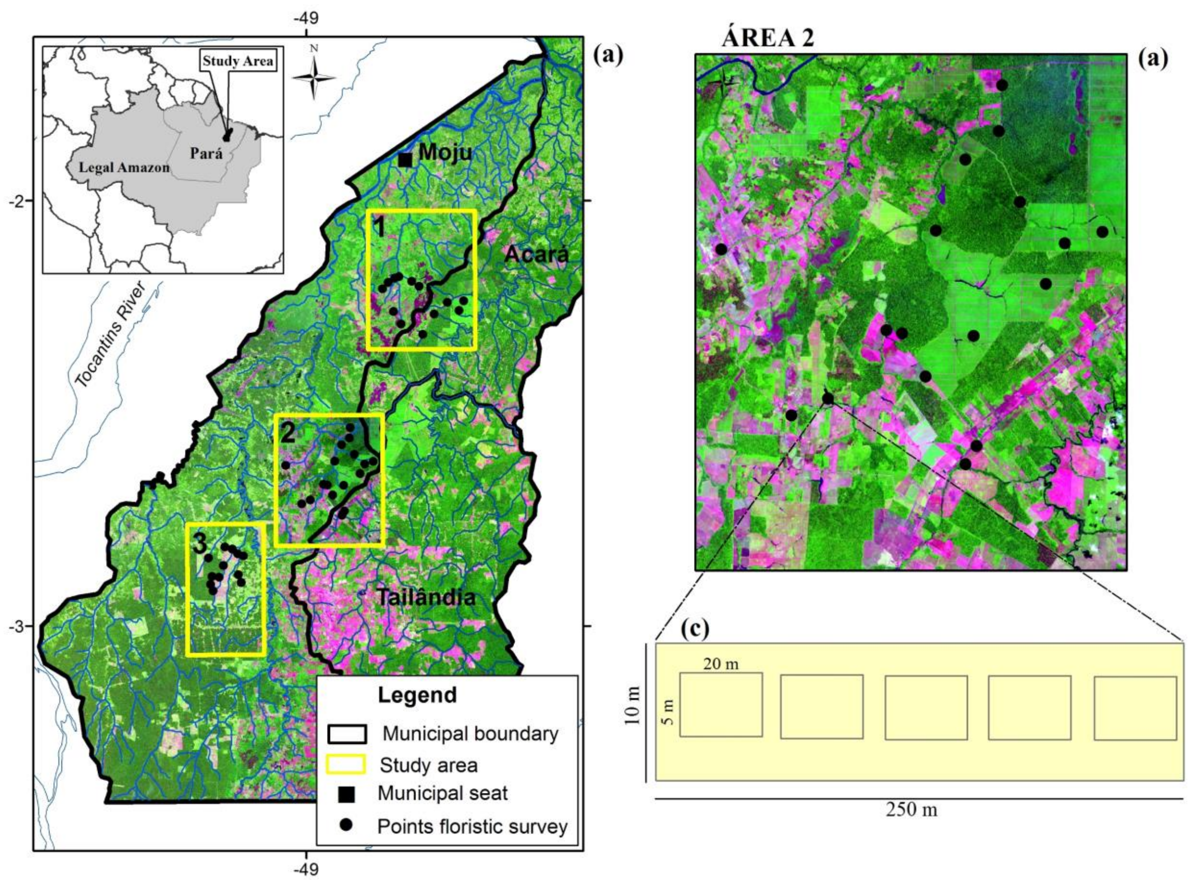
The replacement of forests by crop-pasture matrices has predictably severe consequences for biodiversity (Peres et al. 2010) and farm management decisions and choices of land use influence the conservation of adjacent forest remnants. The quantification of the structure and diversity of tree species in different land cover types is important since they may provide resources and habitat for several forest-associated species and are thus been a vital instrument in assessing forest conservation status and a measure of the sustainability. The conversion of primary forest to oil palm has been reported to result in a reduction of carbon stock by over 50%, with the above-ground biomass loss contributing to 89% of the overall carbon loss (Morel et al. 2011). However, planting oil palm on abandoned pastures increases carbon sequestration and storage relative to those in pasture. Our objective was to provide an evidence-base for conservation/legal designation for restoration priorities on the basis of the relative values/impacts of land cover types. We quantified tree species richness and composition and above-ground carbon stock in four dominant land cover types: remnant primary forest, young secondary forest, oil palm plantation and pasture land. We assessed biodiversity and standing carbon stock values, relative to primary forest, of production areas (Peres et al. 2010), secondary forests (Almeida and Vieira 2001; Ferreira et al. 2018), and primary forest remnants, most of them degraded by fire and logging (Barlow et al. 2016). Understanding how much native woody vegetation persists in production areas, even in the face of active attempts at suppression, is important as it is likely to facilitate persistence of wildlife in these habitats (Gilroy et al. 2015).

Materials and methods

Study area

The study was conducted in three areas (Figure 1) that encompass parts of the municipalities of Moju, Acará and Tailândia, over 2,588 km2 in north-east Pará state (hereafter Moju) which includes the landholdings of three different oil palm producing companies. The three areas differ somewhat in their history of use, land-tenure strategies and time since the start of production. All these landholdings retain varyingly sized patches of primary forest in a complex landscape with high connectivity to the surrounding skeletal forest matrix.

Figure 1. Primary forest, secondary forest, oil palm plantation and pasture were sampled in three sampling areas (1, Moju; 2, Tailândia; 3, Acará) in Pará state, Brazil (a). Background image (Landsat 8) shows land cover in 2013. The close-up of Area 2 showing survey localities (b); and sampling design (c).



There are four dominant land cover types in the area. Primary forest (PF) comprises dense tropical rain forest and more open rain forest of 30–40 m tall canopy, with occasional emergent (Putz and Redford 2010). Some of these forest remnants have been subject to recent, often intense human exploitation with selective logging and accompanying logging roads and log landings resulting in forest degradation and at least occasional wildfire events. Secondary forests (SF) are successional forests that follow land abandonment (Putz and Redford 2010), which in Amazonia includes typically slash-and-burn systems or degraded cattle pastures. Oil palm (OP) plantations have mean canopy height in mature plantations of 8 m and a density of stems 136–160 ha−1. Cattle pastures (Past) are a dominant production land cover type in southern Amazonia as well as in our study area.

Soils are dystrophic yellow latosols (EMBRAPA 2009) and forests are naturally dense lowland evergreen tropical forests growing on low plateau tablelands that are not inundated periodically, as well as on fluvial terraces and floodplains. The regional climate is hot and humid with an annual mean temperature of 26.9ºC. Precipitation varies between 2,000 and 3,000 mm year−1, with most rainfall occurring from January to June. The local population consists mainly of smallholders, rural workers, riverine and quilombola (traditional communities of Afro-Brazilian descent) communities. In recent years, the region has undergone a strong expansion of productive activities, focusing on the service sector and agriculture (IBGE 2010) and is currently the centre of a large-scale oil palm expansion in Brazil.

Floristic and structural inventories

Study plots of 2,500 m2 in size were established in PF, SF, OP and Past to collect floristic and structural data. We employed a stratified random sampling design, where plots were distributed across the study region in proportion to the per cent cover of land cover types to increase the likelihood of capturing important internal heterogeneities within land cover types. Vegetation data were gathered from 31 plots, comprising: 17 in PF, four in SF of varying ages (5–20-year-old), seven in variable-aged OP (12–25-year-old) and three in cattle pastures (5–22-year-old). All live trees and palms with a diameter at breast height (DBH) >10 cm were measured in 10 m × 250 m plots and smaller individuals DBH > 2 < 10 cm were sampled in five subplots of 5 m × 20 m within each plot, following the methodology of Gardner et al. (2013). Tree heights were measured from the base of the tree to the highest point from the ground (parallel to the main trunk) using a 5-m ranging pole for shorter trees and visual estimation for taller ones. All plots were separated by at least 1,500 m from each other to minimise spatial dependence and the plots were located at least 100 m from the nearest border with another habitat type to avoid edge effects.

Trees were identified to species level in the field by expert botanists; herbarium samples were collected for identification when field identification was deemed impossible. Such samples were deposited at the Herbário João Murça Pires, Museu Paraense Emílio Goeldi, Belém. Species were assigned to families based on the APG IV system (APG IV 2016).

Data analysis

To compare species accumulation curves among different land cover types, we constructed sample-based rarefaction curves, using the Mau Tao estimate (Colwell et al. 2012) with 95% confidence intervals using the R package ‘vegan’ v. 2.5–4 (Oksanen et al. 2019).

The data recorded were quantitatively analysed for frequency, density and dominance and the relative values of these parameters were calculated and summed in an Importance Value Index (IVI) (Cottam and Curtis (1956) using Mata Nativa 2 (CIENTEC 2006) as follows: IVIi = RFi + RDi + RBAi where, IVIi is the Importance Value Index of species i; RFi is the relative frequency of species i; RDi is the relative density of species i and RBAi is the relative basal area of species i. Above-ground biomass (Mg ha-1) was estimated for each land cover type using allometric equations for individuals with DBH ≥ 2 cm (Table 1). For species recorded in PF, SF and Past, biomass was calculated using an equation adjusted for tropical forests (Chave et al. 2005). For the genus Cecropia, we used an equation developed by Nelson et al. (1999), using DBH (cm) and stem height (m) as variables. The biomass of E. guineensis was estimated using an equation from Corley and Tinker (2003) that considers the total height (m) and DBH ≥ 1 cm. Values of wood density were extracted from Chave et al. (2009) and Zanne et al. (2009). The carbon stock was estimated for each land-use using the above-ground biomass estimates, assuming that total stem carbon represented 50% of above-ground biomass, following Berenguer et al. (2014).

Table 1. Equations used to estimate the biomass for all species with a DBH ≥ 2 cm, for the genus *Cecropia* and for the species *Elaeis guineensis*

References	Category/species	Equation
Chave et al. 2005	DBH ≥ 2 cm	$(AGB)_{EST} = 0.0509 * \rho D^2 H$
Nelson et al. 1999	*Cecropia sp.* DBH ≥ 2 cm	$\mathrm{DW} = \mathrm{EXP}(2.5118 + 2.4257 \times \mathrm{Ln}(\mathrm{DBH}))$
Corley and Tinker 2003	*Elaeis guineensis* DBH ≥ 1 cm	$Bt = \pi \times (r \times \mathrm{Z})^2 \times 100 \times h \times \rho$ $\rho = \frac{Id \times 0.0076 + 0.083}{1000}$

AGB, estimated biomass; *ρ*, wood density; *DW*, calculated biomass for the genus *Cecropia*; *Bt*, biomass of the trunk (stipe) of *Elaeis guineensis; r*, trunk radius; *Z*, diameter of the base, estimated in 0.777; *h*, total height; *ρ*, trunk density.

Landscape context and tree diversity

As the data were not normally distributed (Shapiro–Wilk test; $W = 0.71$, $P < 0.05$) we used the non-parametric Kruskal–Wallis test with 95% confidence intervals followed by the Dunn's test to test for significant pair-wise differences between land cover types for species richness and carbon stocks. To test for the effect of landscape configuration on tree species diversity and carbon stocks we visualised the mean distance from each plot to the nearest primary forest edge. All fieldwork took place on private properties and landowners were visited prior to fieldwork to introduce the project and secure permissions for surveys.

Results

We recorded 5,696 individuals with a DBH ≥ 2 cm, of 413 species in 68 families, of which 381 species were recorded in PF (303 exclusive PF species). Of these 381 species, 93 were recorded in other land cover types, 77 in SF, 13 in Past and four in OP.

Floristic composition of the tree communities

The highest mean richness of tree species and families was recorded in PF, meanwhile OP had the lowest values (Table 2). Species accumulation curves indicated an asymptotic pattern in PF and OP, while for SF and Past the curves showed a tendency of continued increase, indicating insufficient sampling effort (Figure S1).

Table 2. Summary table of taxonomic richness and aboveground biomass carbon stocks in primary forest, secondary forest, oil palm plantation and pasture, Moju region, Pará, Brazil. PF, primary forest; SF, secondary forests; OP, oil palm plantation; and PAST, cattle pasture.

Land cover type	Plots	Sample area (ha)	Individual	Species	Family	Biomass (Mg ha^{-1})	Carbon stock (Mg ha^{-1})
DBH ≥ 10 cm							
PF	17	4.25	2173	273	48	306.75	153.67
SF	4	1.00	359	54	25	28.12	14.06
OP	7	1.75	363	2	2	58.86	29.43
>2cm DBH<10cm							
PF	85	0.85	2135	283	53	81.61	40.80
SF	20	0.20	612	87	30	92.49	46.24
OP	5	0.05	13	3	3	0.52	0.26
PAST	10	0.10	41	15	10	6.10	3.05

At the species level, the taxonomic composition of the land cover types showed major differences (Table 3). The upper strata (DBH > 10 cm) of PF were dominated by Eschweilera coriacea (IVI = 21.5) and Lecythis idatimon (IVI = 14.2) and Lecythidaceae and Fabaceae were the most common tree families. Fabaceae was also the dominant family in SF in the canopy and sub-canopy strata. Small trees such as Rinorea guianenis and R. flavescens (Violaceae) were common in the lower strata of PF while Poecilanthe effusa (Fabaceae) dominated in SF. In OP, trees were essentially represented by the planted E. guineensis and four shade tree species. In Past, we did not record any trees with a DBH > 10 cm.

Table 3. The top tree species ordered by their Importance Value Index (IVI) values in PF, primary forest; SF, secondary forest; OP, Oil palm plantations; and PAST, pasture, Moju region, Pará state, Brazil.

Species	Family	IVI	LAND COVER
DBH (> 10 cm)			
Eschweilera coriacea (DC.) S.A.Mori	Lecythidaceae	21.5	PF
Lecythis idatimon Aubl.	Lecythidaceae	14.23	
Rinorea guianensis Aubl.	Violaceae	11.77	
Vouacapoua americana Aubl.	Fabaceae	10.83	
Eschweilera grandiflora (Aubl.) Sandwith	Lecythidaceae	8.83	
Cecropia sciadophylla Mart.	Urticaceae	6.53	
Sterculia pruriens (Aubl.) K.Schum.	Malvaceae	5.60	
Inga alba (SW.) Willd.	Fabaceae	5.04	
Inga thibaudiana DC.	Fabaceae	5.02	
Licania canescens Benoist	Chrysobalanaceae	4.29	
Inga alba (SW.) Willd.	Fabaceae	34.79	SF
Cecropia palmata Willd.	Urticaceae	16.79	
Inga rubiginosa (Rich.) DC.	Fabaceae	15.06	
Annona exsucca DC.	Annonaceae	14.76	
Vismia guianensis (Aubl.) Choisy	Hypericaceae	13.84	
Stryphnodendron guianense (Aubl.) Benth.	Fabaceae	13.03	
Eriotheca globosa (Aubl.) A.Robyns	Malvaceae	12.92	
Schizolobium parahyba var. amazonicum (HuberexDucke) Barneby	Fabaceae	12.63	
Nectandra cuspidata Nees&Mart.	Lauraceae	12.13	
Cordia scabrifolia A.DC.	Boraginaceae	9.98	
Elaeis guineenses Jacq.	Arecaceae	276.66	OP
Cecropia distachya Huber	Urticaceae	13.77	
DBH (2cm <10cm)			
Rinorea guianensis Aubl.	Violaceae	16.50	PF
Rinorea flavescens (Aubl.) Kuntze	Violaceae	11.13	
Eschweilera coriacea (DC.) S.A.Mori	Lecythidaceae	9.43	
Dodecastigma integrifolium (Lanj.) Lanj. & Sandwith	Euphorbiaceae	8.45	
Protium trifoliolatum Engl.	Burseraceae	6.55	
Vouacapoua americana Aubl.	Fabaceae	6.06	
Sagotia racemosa Baill.	Euphorbiaceae	5.88	
Lecythis idatimon Aubl.	Lecythidaceae	5.50	
Pouteria decorticans T.D.Penn	Sapotaceae	5.03	
Fusaea longifólia (Aubl.) Saff.	Annonaceae	4.72	
Poecilanthe effusa (Huber) Ducke	Fabaceae	22,84	SF
Ocotea glomerata (Nees)Mez	Lauraceae	19,81	
Banara guianensis Aubl.	Salicaceae	19,44	
Vismia guianensis (Aubl.) Choisy	Hypericaceae	14,41	
Cecropia palmata Willd.	Urticaceae	14,31	
Annona exsucca DC.	Annonaceae	13,66	
Inga alba (SW.) Willd.	Fabaceae	12,38	
Cupania scrobiculata Rich.	Sapindaceae	8,86	
Cordia exaltata Lam.	Boraginaceae	8,22	
Byrsonima crispa A.Juss.	Malpighiaceae	6,76	
Cecropia palmata Willd.	Urticaceae	192.90	OP
Inga thibaudiana DC.	Fabaceae	37.46	
Banara guianensis Aubl.	Salicaceae	37.46	
Vismia guianensis (Aubl.) Choisy	Hypericaceae	55,52	
Banara guianensis Aubl.	Salicaceae	34,62	
Guatteria poeppigiana Mart.	Annonaceae	28,75	

Inga alba (SW.) Willd.	Fabaceae	24,90	
Inga thibaudiana DC.	Fabaceae	21,32	
Cordia scabrifolia A.DC.	Boraginaceae	16,70	PAST
Tapirira guianensis Aubl.	Anacardiaceae	16,64	
Eschweilera grandiflora (Aubl.) Sandwith	Lecythidaceae	15,56	
Lacunaria crenata (Tul.) A.C.Sm.	Quiinaceae	15,15	
Xylopia nítida Dunal	Annonaceae	14,56	

Species richness and carbon stock

We found significant differences among land cover types for both species richness (H = 41.5, df = 3, P < 0.05 Figure 2(a)) and carbon stocks (H = 41.1, df = 3, P < 0.05, Figure 2(b)), with PF having the highest mean ± SE species richness (93 ± 1.9) followed by SF (44 ± 1.2) and Past (7 ± 4). Between-group differences were significantly different between PF and Past and PF and OP for both species richness and carbon stock. There were significant differences in the composition of tree species between PF and the other land cover types (F = 12.901, P < 0.001; Figure 3). Species accumulation curves (Figure S1) only approached asymptotes in OP, although sample sizes for Past with enough woody biomass were low.

Figure 2. Box plots of species richness (a) and carbon stock (b) in primary forest (PF), secondary forest (SF), oil palm (OP), and cattle pasture (Past) in the Moju region, Pará, Brazil.

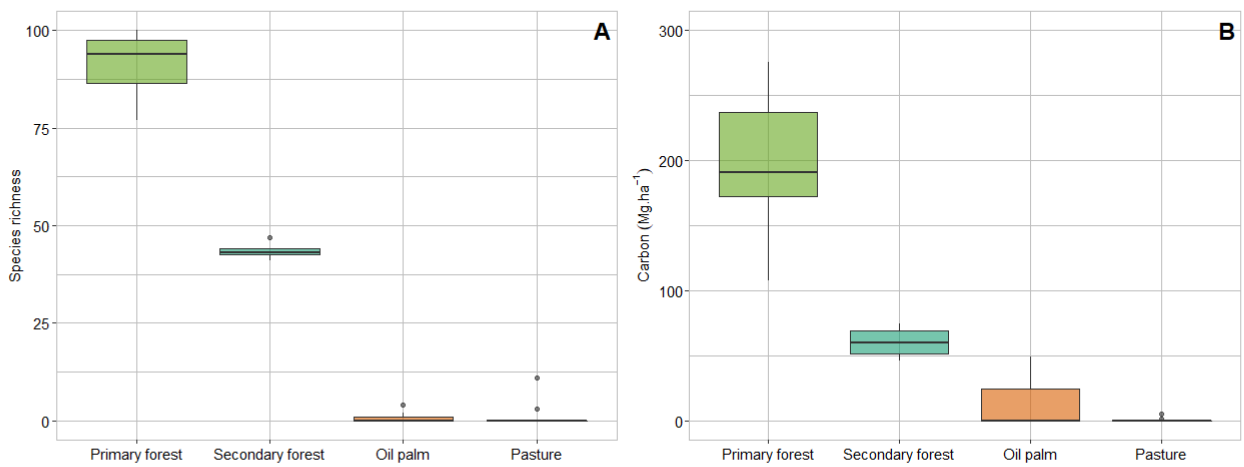
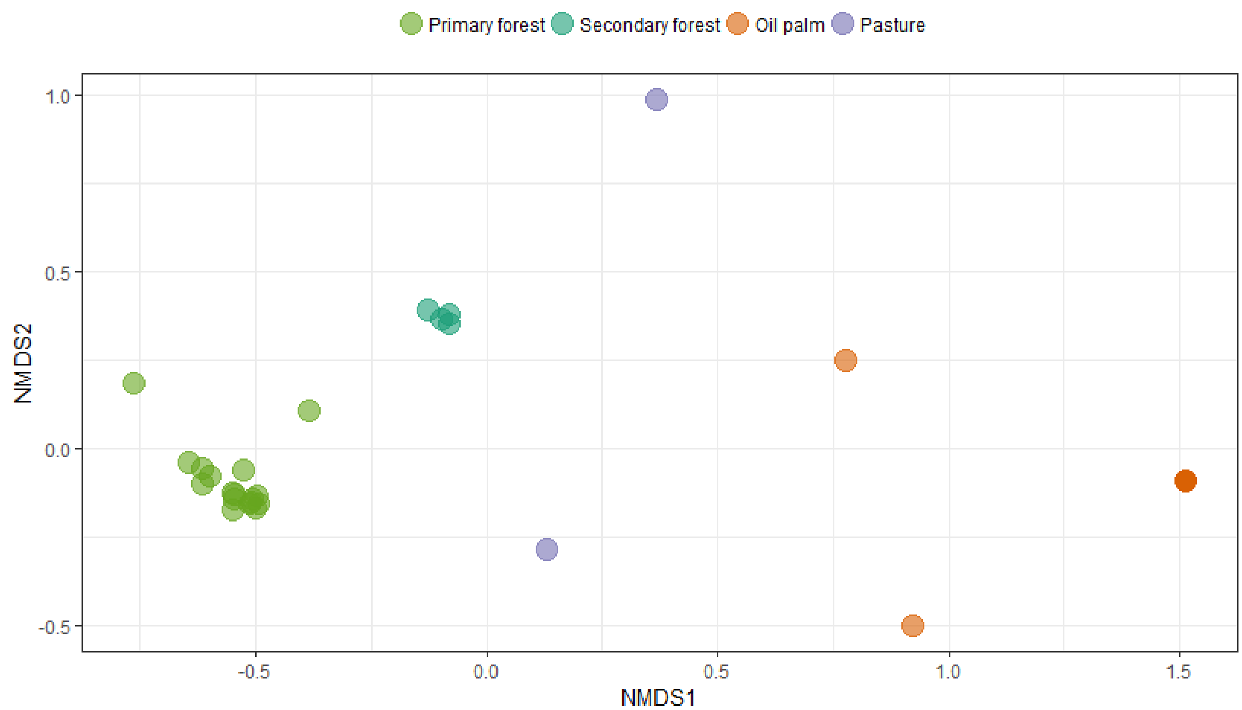


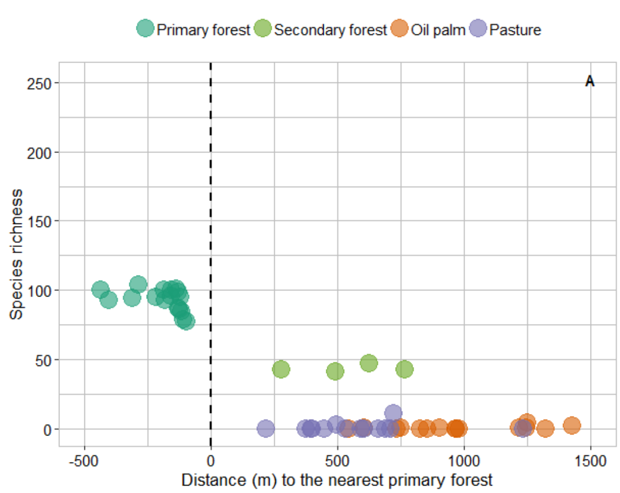
Figure 3. MDS plots of tree community composition between land-cover classes: primary forest, secondary forest, oil palm and cattle pasture in the Moju region, Pará, Brazil, showing strong clustering of

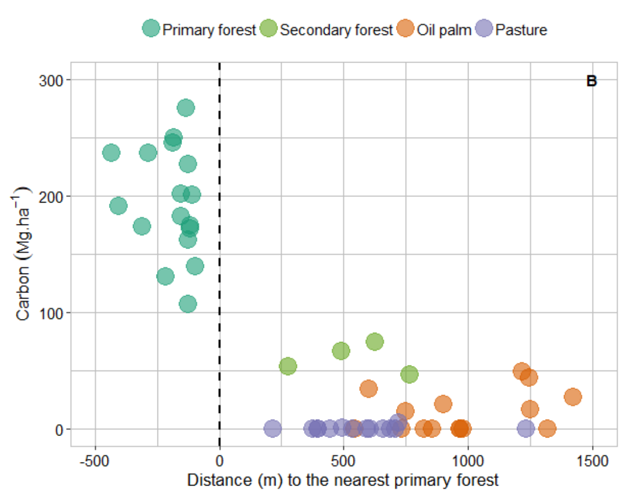
primary forest and secondary forest stands whilst remaining land covers are more dissimilar.



The assemblages were rather distinct in their composition, especially PF; there was some compositional overlap between OP and Past. Secondary forest had its own discrete assemblage composition and its species richness was intermediate between that of PF and OP and Past (Figure 4(a)). We did not find evidence for strong effects of distance from the primary forest edges in mediating either carbon stock or species richness for SF, OP and Past (Figure 4(b)); note that none of the sampled secondary forests were > 700 m from the nearest PF.

Figure 4. Species richness (a) and carbon stock (b) of land cover types in the Moju region, Pará, Brazil demonstrating little variation in response to distance to the nearest primary forest edge.





Discussion

Our study provides, for the first time, real data about tree diversity in Neotropical oil palm plantations. The results confirmed that these plantations have very low species richness, which is related to management to suppress regeneration to avoid competition for water, light and nutrients, by other woody species. Management was probably responsible for cancelling out any potential effect on species richness of proximity to primary forest. Despite a pervasive history of disturbance, remnant PF within OP plantations in the Moju region still retains relatively high tree diversity and biomass carbon, and support a significant proportion of the species assemblage of continuous forest. Lecythidaceae and Fabaceae were the most abundant families in these forests, and are almost invariably present in native forests in the Amazon basin (Ter Steege et al. 2000). Secondary forests are typically characterised by an abundance of fast-growing taxa of early successional stages and even 70-year-old secondary forests in eastern Amazonia have a conspicuously different taxonomic composition from that of primary forests of comparable stature (Vieira et al. 2003; Vieira 2019). The ecological and ecosystem service values of secondary forests are lower in comparison with those of primary forests, however secondary forests are important for conservation in fragmented oil palm dominated landscapes as the carbon stocks and biodiversity in older secondary forests approach 80% of that found in undisturbed primary forests (Lennox et al. 2018).

The role of secondary forests, and their potential to contribute to restoring soil functions, carbon sequestration and biodiversity conservation, especially in heavily impacted regions is considered important (Chazdon et al. 2009; Poorter et al. 2016). Secondary forests are rarer in the landscape than disturbed primary forests and pastures in the Moju region, and almost all are quite young and often return to production (slash-and-burn) after few years of fallowing. They have thus not been afforded the time to acquire much biodiversity or biomass, reflected in their hosting of on average less than half of the species associated with primary forests.

The predictable loss of tree diversity and above-ground carbon stocks with increasing land use intensity (conversion to plantation and agricultural) matches the pattern found in various other studies (e.g. Almeida and Vieira 2001; Berenguer et al. 2014). These non-forest land cover types retained no more than 30 Mg ha^{-1} carbon stock in our study, well below that found in primary forest (194.5 Mg ha^{-1}) (Table 2). This result underscores the message that oil palm is no substitute, either in terms of biodiversity conservation or carbon sequestration and storage, for native vegetation and protection areas and legal reserves should not be planted with oil palm (Lees and Vieira 2013). If legal reserves were formed by intact primary forest, covering 80% of the rural properties of Moju region it should allow the preservation of most of biodiversity and the maintenance of important ecosystem functions.

The role of oil palm in sequestering and storing carbon during its lifetime (25–30 years) is similar to that recorded in young secondary forests. Our estimated above-ground carbon stock of 33 Mg ha^{-1} at 12 years is comparable to the 30 Mg ha^{-1} found by Filho (2012) for an 11-year-old plantation in north-east Pará. Palm trees reaching around 25–30 years of age are cut down, zeroing the standing carbon stock. By comparison, late-successional secondary forests in the region can reach 70 Mg ha^{-1} of carbon after 70 years (Almeida and Vieira 2001). Our carbon stock values in secondary forests in the current study are conservative estimates limited by the relative scarcity of old secondary forests and consequent low sample sizes.

Most areas identified as suitable for oil palm are located along the Amazonian Arc of Deforestation in southern Amazonia where forest loss and fragmentation are ongoing and potentially increasing after a recent decrease (Lees and Peres 2006; Freitas et al. 2018). Previous work has shown that oil palm expansion in Amazonia had occurred mostly at the expense of secondary forests, albeit affecting some areas of primary forest, thus oil palm expansion here has resulted so far in less loss of tropical forest than elsewhere in the tropics (Benami et al. 2018). Secondary forests have received legal protection in recent years in the state of Pará after establishing quantitative criteria that deemed when early-successional secondary forests should be preserved in rural properties (Vieira et al. 2014). Avoiding any conversion of primary or secondary forest in advanced stage of succession into plantations and restricting expansion of new plantations onto land with low carbon stocks ought to result in carbon-neutral production (Agus et al. 2013). Maintaining biodiversity and conserving carbon stocks almost always incur a substantial financial opportunity cost but the environmental and land use value trade-offs associated with oil-palm expansion can be potentially handled by using adequately planned and spatially explicit development strategies (Butler and Laurence 2009; Gilroy et al. 2015). Future work in Amazonia should explore displacement of smallholder agriculture resulting in further forest loss or negative implications for local food security.

Conclusions

We demonstrate that oil palm plantations support very low tree diversity and modest carbon stocks, the latter comparable to that of young secondary forests. Conversely, remnant primary forest in the legal reserve portion of plantations maintains substantial tree diversity and associated biota and carbon stocks. Regional development plans need to consider the conflicting interests of different social actors, institutions and companies, monitoring the health of primary and secondary forests to see that environmental quality is not jeopardised by oil palm or other large-scale monocultures. Integration needs to occur across development policy, company strategies and with local farmers and is a fundamental criterion for sustainable expansion production of oil palm in Amazonia. Given the role that diverse plant communities have in supporting species in other groups, future work should focus on the diversity of plant communities in understorey vegetation. Future research and policy discussions also need to focus on a broader view of environmental management of oil palm plantations integrating an approach of valuating ecosystem services and biodiversity conservation in oil palm landscapes that can provide a basis for sustainable land use policies in the Amazon region.

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