



## Collapse of ecosystem carbon stocks due to forest conversion to soybean plantations at the Amazon-Cerrado transition

Isabelle Bonini<sup>a</sup>, Ben Hur Marimon-Junior<sup>a,\*</sup>, Eraldo Matricardi<sup>b</sup>, Oliver Phillips<sup>c</sup>, Fabiano Petter<sup>d</sup>, Bianca Oliveira<sup>a</sup>, Beatriz Schwantes Marimon<sup>a</sup>

<sup>a</sup> Universidade do Estado de Mato Grosso (UNEMAT), Programa de Pós-Graduação em Ecologia e Conservação, Campus de Nova Xavantina, PO Box 08, 78690-000 Nova Xavantina, MT, Brazil

<sup>b</sup> Universidade de Brasília, Department of Forestry, 70910-900 Brasília, DF, Brazil

<sup>c</sup> School of Geography, University of Leeds, Leeds, UK

<sup>d</sup> Instituto de Ciências Agrárias e Ambientais, Universidade Federal de Mato Grosso, 78557-267 Sinop, MT, Brazil



### ARTICLE INFO

#### Keywords:

Deforestation  
Agriculture  
Soybean  
Rubber plantation

### ABSTRACT

Deforestation to establish monocrops in the tropics is causing massive reductions in ecosystem C stocks. Amazonia is a principal target of this process, owing to the expansion of the agribusiness frontier throughout the transition with the Cerrado biome, the zone known as the “Arc of Deforestation”. In this vast contact region between the two largest South American biomes, the conversion of primary forest to soybean and pasture systems has led to the deforestation of nearly five million hectares since 1980. Despite this, we lack precise understanding of the effects of land use on ecosystem C stocks and pools in this region. Addressing this knowledge gap is crucial to improve predictions and fit models for different land use scenarios in Amazonia. To reduce uncertainty regarding the magnitude of the impacts of deforestation on the C cycle, we evaluated ecosystem C stocks in contrasting land-use systems across a topographically, climatically, and edaphically near-homogeneous landscape in southern Amazonia. We investigated the soil, litter, fine root and aboveground biomass (AGB) C stocks of soybean plantations and compared them to those of remnant native forests and rubber plantations; the latter is considered *a priori* as a cropping system with low impact on the C cycle. We found that the conversion of native forest to soybean plantation caused a 130.5 Mg C ha<sup>-1</sup> loss, about threefold higher than the loss when forest is converted to rubber plantations, 48.5 Mg C ha<sup>-1</sup>. While 30-year old rubber plantations had recovered 84% of forest carbon stocks, all plantation types induced sustained losses of at least one-third of the original soil carbon. Fine root allocation changed sharply in the two crops following conversion, indicating an alteration in plant nutrient dynamics. Our results show that perennial and annual monocrops have very different impacts on the C cycle, which need to be accounted for in carbon-climate models as well as in public policies regulating land use in Amazonia. Our results show that while silviculture has the potential to restore most of the above-ground C stocks of previously forested areas, but neither silviculture nor conventional agriculture may ever restore Amazon soil C stocks once they become vulnerable and oxidized after deforestation. If such conversion-driven soil carbon losses were scaled across the Amazon they would induce a cumulative loss of more than 5 Pg in soil carbon by 2050.

### 1. Introduction

One of the greatest uncertainties surrounding land use in the tropics is the impact of agriculture on Amazon Forest C stocks. Additionally, we are unsure about the sustainability threshold of traditional agricultural systems established in the savannas of central Brazil once applied in Amazonia (Petter et al., 2017). Brazilian agriculture has worldwide importance; it is mainly based on cereal crops and covers nearly 59

million hectares (CONAB, 2016), 60% of which is located in the Cerrado biome (IBGE, 2016). Amazonia borders this agribusiness belt from northwest to northeast, and, due to the scarcity of new agricultural land, is being pressured by the intense conversion of native forests to pasture and annual cropping systems, especially that of soybean (Persson et al., 2014; Farias et al., 2016). The largest supply of land is located exactly at the transition zone between Amazonia and Cerrado, the world's largest ecotonal complex. The confluence of the two floras

\* Corresponding author.

E-mail address: [bhmarimon@unemat.br](mailto:bhmarimon@unemat.br) (B. Hur Marimon-Junior).

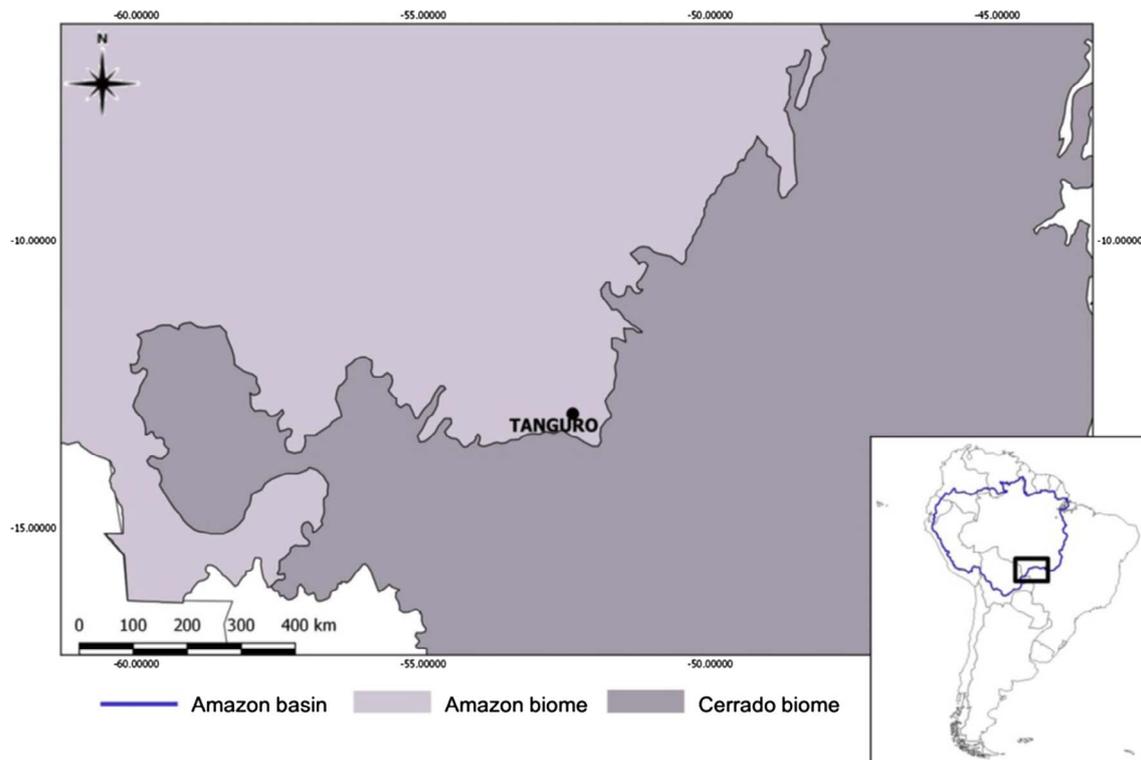


Fig. 1. Location of the Tanguro Farm, Querência, MT.

in addition to the largest and most dynamic agricultural frontier in the tropics result in a vast area of agribusiness occupation extending for nearly six thousand kilometers and known as the “Arc of Deforestation”.

The native vegetation remnants in this ecotonal complex contain greater species diversity than each biome separately in unique physiognomies such as *cerradão* and transitional forests. However, the expansion of agribusiness in this contact zone has led to a series of massive environmental impacts (Costa & Pires, 2010), such as a rapid loss of biodiversity (Wearn et al., 2012) and soil and water degradation (Maia et al., 2010; Neill et al., 2013). Another impact of regional land use is substantial C transfer from soil-plant systems to the atmosphere (Friedlingstein et al., 2010; Coe et al., 2013), initially one of the results of deforestation and fire and, later, intensive land use, especially in the traditional soybean cropping system.

This scenario of land-use change is reflected on the C cycle and may lead to significant alterations in ecosystem structure, composition and functioning (Ojima et al., 2013). The conversion of native vegetation to agricultural and pasture lands compromises nutrient cycling (Davidson et al., 2004), alters organic matter dynamics (Don et al., 2011), and consequently has negative effects on the integrity of biogeochemical cycles, especially that of C. As estimated by Buckeridge (2008), the conversion of Brazilian forests in crop/pasture systems over the past 140 years has led to a net release of 121 Pg C to the atmosphere. This C increase alters not only the climate but also biodiversity patterns (Strassburg et al., 2010), considerably modifying ecosystems.

However, uncertainty still predominates in estimating the effects of land use on biogeochemical cycles and their consequences on the regional C balance. Current knowledge gaps exist regarding the impacts of native vegetation conversion to soybean plantations on CO<sub>2</sub> emissions in the Arc of Deforestation. Moreover, there has been no detailed research on the ways in which the substitution of native ecosystems by agroecosystems could reduce and/or redistribute C stocks in the biosphere. In this case, seasonal patterns of biomass deposition and allocation may also change (Yang et al., 2004), thus affecting the balance of nutrients, water and C (Wilson et al., 2000; Kotowska et al., 2016).

These modifications are the result of a reduction in vegetation structure and the simplification of trophic interactions (Moore et al., 2005). In addition, land-use change can alter C residence time in the system, increasing the necessity for repetitive fertilization (Drinkwater & Snapp, 2007), thus leading to instability and unsustainability.

Nevertheless, the magnitude of the effects of those changes can vary greatly among crops (Kotowska et al., 2015) and management systems (Petter et al., 2017). Annual and perennial monocrops, for instance, differ in C storage and cycling patterns (Baldotto et al., 2015). Successive harvests in short-cycle crops are responsible for significant C exportation (Zeri et al., 2013) and a consequent reduction in soil stocks, even when direct planting is used. This results mainly from the traditional agricultural management system based on soybean planting during the summer followed by corn during the interharvest season (Petter et al., 2017). On the other hand, perennial crops such as rubber tree can positively influence this scenario by incorporating large amounts of organic matter into the substrate, which consequently protects against erosion and leaching (Suddick et al., 2013) and favors biogeochemical cycles by maintaining C stocks. As a result, perennial crops need lower nutrient additions and interventions compared to annual crops.

Because cropping systems show distinct behaviors, we aimed to assess the impacts of native forest conversion to soybean plantations (high-impact annual monocropping systems), also comparing with rubber tree plantations (low-impact perennial monocropping systems), in terms of C stocks and dynamics in distinct ecosystem compartments in the Arc of Deforestation. To achieve this goals, we generate prior knowledge on the impacts of such conversion by addressing these questions:

- (1) What is the magnitude of the difference in native forest conversion impacts on C stocks between annual (soy) and perennial (rubber) monocrops?
- (2) In the conversion of native forest to rubber tree monocrops, do C stocks tend to become similar to the original stocks of native forest over time?

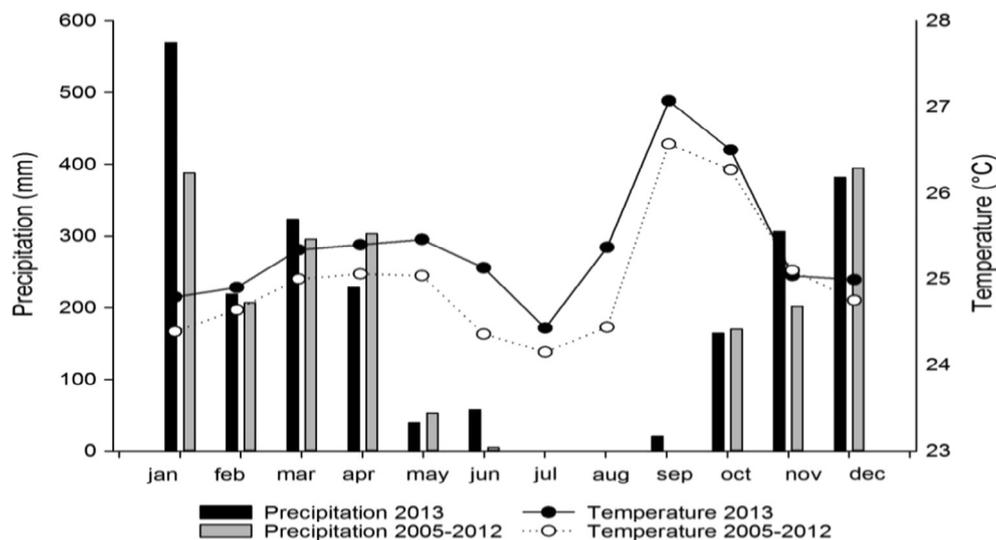


Fig. 2. Monthly mean precipitation and temperature from 2005 to 2012 and in 2013 (sampling year) in the region where the four study sites are located, Tanguro Farm, Querência, MT.

- (3) Does the conversion of native forest to soy and rubber monocrops change the annual dynamics of biomass deposition and allocation and consequently the system's nutrient dynamics?

## 2. Materials and methods

### 2.1. Study sites

This study was conducted at Tanguro Farm (80,000 ha), located in Querência, Mato Grosso State, Brazil, southeastern Amazon basin (Fig. 1). The property, which belongs to the Amaggi Group, is located in the transition zone between the Amazon and Cerrado biomes, where Semideciduous Seasonal Forests predominate (Veloso et al., 1991). Nearly 50% of the farm was cleared for pasture land during the early 1980s, but after 2003, most of the pastures were converted into monocropping systems, especially those of cereals.

The mean annual rainfall is 1770 mm, with an intense dry season between May and September, and the mean annual temperature is 25 °C (Neill et al., 2013). During the sampling year, we recorded slightly atypical rainfall, with some rain events until early June, although the other months were within  $\pm 50\%$  of the normal means (Fig. 2). Additionally, temperatures were slightly above average, consistent with the long-term warming trend in southern Amazonia (Jiménez-Muñoz et al., 2013). Soils in the study site are mainly deep, dystrophic, acidic, nutrient-poor Oxisols and have a sandy-clay texture, and the groundwater level is between 12 and 15 m depth (Balch et al., 2008).

Unfortunately, data regarding the soil and native vegetation prior to the conversion to pasture and crops are not available. To our knowledge, no previous botanical or soil data are available for any area within the Arc of Deforestation in southern Amazonia covered by forest for the past thirty years. However, the landscape of the study sites is very uniform; the sites all belong to the same physiographic unit, occur at nearly the same altitude, and have similar topography (plain) and soil physico-chemical characteristics (RadamBrasil, 1981). Furthermore, the intact forest remnants at the study sites are very similar in terms of species composition, basal area and biomass. Such conditions were verified in a ForestPlots analysis ([www.forestplots.net](http://www.forestplots.net)), where three forest plots appeared to be very similar to one another in 2015: 154, 164, 153t AGB ha<sup>-1</sup> and trees  $\geq 10$  cm diameter at breast height (DBH) (biomass calculation based on the model in Chave et al., 2014).

To address our research questions, we chose four sampling sites with distinct land-use systems and histories (Table 1; Fig. 3). The land was cleared (deforested) to plant soybeans during the early 1980s, initially converted to a pasture of the African grass *Urochloa brizantha*

(syn. *Brachiaria brizantha*) for beef production until 2002 and then converted to soybean cultivation (Fig. 3). The oldest rubber tree stand was established in the late 1970s by directly clearing the forest (Fig. 3). The sites were chosen in a way to be closest to one another to minimize edaphoclimatic variability. At each site, we sampled 25 contiguous 20 m  $\times$  20 m plots (1 ha), totaling 4 ha of sampling. Within the plots, we sampled the following ecosystem compartments: soil, litter, roots and aboveground biomass. We sampled the areas four times: twice during the dry season (June and September) and twice during the rainy season (February and December).

### 2.2. Soil sampling

To determine the soil organic matter (SOM) contents, we collected samples during the peak of the rainy season (February) and the peak of the dry season (September). We randomly collected one sample per plot at both 0–10 and 10–20 cm soil depths for a total of 50 samples per study site. Each sample was labeled and air dried for physico-chemical analyses according to Embrapa (1997).

The soil variables assessed were pH, using the electrode method, P and K, extracted with diluted concentrations of strong acids [0.5 N HCl + 0.025 N H<sub>2</sub>SO<sub>4</sub> (Mehlich I)], and Ca, Mg and Al, extracted in 1 N KCl. We determined the Phosphorus concentration by the colorimetric method, Ca and Mg by atomic spectroscopy, K by flame emission spectrometry, and the potential acidity (H + Al) by extraction with a 0.5 mol L<sup>-1</sup> calcium acetate solution at pH 7.1–7.2 and titration with 0.025 mol L<sup>-1</sup> NaOH, using 10 g L<sup>-1</sup> phenolphthalein as an indicator. We obtained the cation exchange capacity using the sum of Ca, Mg and K. In Brazil, the levels of Na<sup>+</sup> in the study region (Amazon biome) are very low, and for this reason, it is not included in the CEC calculation. We calculated the base saturation using the following equation:

$$\text{Base saturation (\%)} = 100 \times \left\{ \frac{(Ca + Mg + K)}{Ca + Mg + K + (H + Al)} \right\}.$$

To determine the total organic carbon (TOC) we used oxidation Walkley-Black method as described in Nelson and Sommers (1996). This method employs organic matter oxidation by potassium dichromate in sulfuric acid. We estimated the organic carbon content in the soil by titration of excess dichromate following oxidation.

To obtain the apparent density of the soil (ADS), we collected intact soil cores at the same sampling points and depths using a volumetric cylinder (97 cm<sup>3</sup>). The samples were oven dried at 105 °C until they reached a constant dry weight to obtain the soil dry mass. The ADS (g cm<sup>-3</sup>) was calculated according to Blake & Hartge (1986).

**Table 1**  
Land-use systems selected for this study at Tanguro Farm, Querência, MT.

| Land use   | Description  |
|--|--|
| Native forest (reference area)<br>12°82'63.56" S 53°35'29.89" W                      | Extensive and well-preserved fragment of Evergreen Seasonal Forest. No record of previous use.   |
| Soybean plantation ( <i>Glycine max</i> L. Merrill) 12°49'40.39" S<br>52°22'06.73" W | Used for cereal crops since 2003. This site was amended with limestone (~1500 kg ha <sup>-1</sup> ) during the conversion of pasture to soybeans in 2002. Additionally, it receives yearly additions of KCl (granular potassium chloride) and P (phosphate, simple superphosphate and triple superphosphate) at 80 kg ha <sup>-1</sup> year <sup>-1</sup> (K) and 50 kg ha <sup>-1</sup> year <sup>-1</sup> (P). Current management includes direct planting where soil is no longer tilled. |
| Rubber plantation of 30 years in age<br>12°52'31.56" S 52°22'32.62" W                | Site used for rubber monocropping ( <i>Hevea brasiliensis</i> (Willd. ex A. Juss.) Müll. Arg, clone RRIM 600) since 1970, although there is no precise information on the type of soil management employed at that time. Spacing is 7.5 m × 2.0 m.   |
| Rubber plantation of 5 years in age<br>12°52'34.66" S 52°22'31.69" W                 | This recent stand also used clone RRIM 600. Land was prepared by plowing and harrowing, and dolomitic limestone was applied (Ca:Mg 3:1) to increase the soil base saturation by 50%. One month after planting, 30 g N per plant was applied, which was repeated twice as cover fertilization during the first year after planting. Currently, N, P <sub>2</sub> O <sub>5</sub> and K <sub>2</sub> O fertilization occurs at appropriate doses for this clone. Spacing is 7.5 m × 2.5 m.      |

2.3. Litter biomass sampling

We collected litter samples during the four field data collection efforts. For this procedure, we used the Marimon-Hay device (Marimon Junior & Hay, 2008), which provides thickness, density and volume data for the collected material. We randomly collected samples at one point in each plot for a total of 25 samples per study site. The collected material was oven dried at 70 °C until it reached a constant weight based on a precision scale.

2.4. Fine root biomass sampling

We sampled fine roots at 0–10 and 10–20 cm depths during the same sampling times and at the same sampling points where litter was collected. We collected core samples using a cylindrical root borer with known volume (Fugiwara et al., 1994). All live roots (clear and flexible) with a diameter ≤2 mm were manually separated in the laboratory, whereas dead roots (dark and rigid) were discarded (Hendricks et al., 2006). The selected roots were oven dried at 70 °C until a constant weight was reached, determined by a precision scale.

2.5. Aboveground biomass sampling

To estimate the aboveground biomass of the native forest, we used the method proposed by Baker et al. (2004) using a multispecific allometric equation based on individual diameter and wood density (Eq. (1)). For the rubber plantation sites, we used the equation proposed by Nelson et al. (1999) adjusted for the diameter and height of the individuals (Eq. (2)). In the native forest and 30-year-old rubber plantation, we sampled all individuals with a diameter at breast height (DBH) ≥ 10 cm, while in the 5-year-old rubber plantation, we sampled all standing individuals. Wood densities were obtained from the ForstPlots.net dataset (Lopez-Gonzalez et al. 2011).

$$BA = \frac{\rho_i}{0.67} \exp(0.33[InD] + 0.933[InD]^2 - 0.122[InD]^3 - 0.37) \tag{1}$$

$$In(BA) = -2.5202 + 2.14*In(D) + 0.4644*In(H) \tag{2}$$

where BA = aboveground biomass (kg tree<sup>-1</sup>), ρ<sub>i</sub> = wood density (g cm<sup>-3</sup>), D = diameter at breast height (cm) and H = height (m).

To quantify the aboveground biomass in the soybean plantation, we randomly selected 10 quadrats of 0.25 m<sup>2</sup>, from which all plants (whole individuals) at the peak of their physiological development (R6 stage) were collected at the ground level. Samples were oven dried at 70 °C until a constant weight was reached as, determined by a precision scale. We estimated the dry biomass per hectare according to Arevalo et al. (2002).

2.6. Carbon stock calculation

We estimated the soil C stocks for the top 20 cm since this represents the soil depth of greatest influence for the root zone of soybean and pasture. Greater depths are influenced only over the short term in native forests or perennial plantations, such as those of rubber. Therefore, short-term comparisons at depths greater than 20 cm for annual soybean monocrops or even new rubber plantations with native forests and mature rubber plantations may lead to conclusion errors.

To calculate the C stocks in the soil at all sites we used the Eq. (3), proposed by Veldkamp (1994):

$$EC_{soil} = \frac{CO * DAS * e}{10} \tag{3}$$

where EC<sub>soil</sub> = organic C stock at the considered depth (Mg ha<sup>-1</sup>); CO = total organic C at the sampled depth (g kg<sup>-1</sup>); DAS = apparent density of the soil (kg dm<sup>-3</sup>) and e = depth of the considered layer (cm).

Next, we applied Eq. (4) as a correction for EC<sub>soil</sub>, as proposed by Sisti et al. (2004). This prevents estimation errors from changes in ADS due to the conversion of native vegetation to pasture and later to soybean and rubber plantations.



**Fig. 3.** Land-use history of the four study sites at Tanguro Farm, Querência, MT. FOR: Native forest; SOY: Soy plantation; RUB30: 30-year-old rubber plantation; RUB5: 5-year-old rubber plantation.

$$Cs = \sum_{i=1}^{n-1} Cti + \left[ Mtn - \left( \sum_{i=1}^n Mti - \sum_{i=1}^n Msi \right) \right] * Ctn \quad (4)$$

where  $Cs$  = total organic C stock, corrected as a function of the soil mass of a reference area;  $\sum_{i=1}^{n-1} Cti$  = sum of  $EC_{soil}$  from the first to the second-to-last sampled layer ( $Mg\ ha^{-1}$ );  $Mtn$  = soil mass of the last layer in the treatment ( $Mg\ ha^{-1}$ );  $\sum_{i=1}^n Mti$  = sum of the total soil mass under the treatment ( $Mg\ ha^{-1}$ );  $\sum_{i=1}^n Msi$  = sum of the total mass of the soil sampled in the reference area ( $Mg\ ha^{-1}$ ) and  $Ctn$  = soil organic C content in the last layer sampled ( $Mg\ C\ Mg^{-1}$  soil).

For the other compartments – litter, fine root mat and aboveground biomass – we calculated the C stocks by multiplying the values found for biomass by the factor for the C percentage found, which is 0.39 for litter and 0.485 for root and aboveground biomass, according to Higuchi (1998).

### 2.7. Data analysis

Initially, we assessed the significance of the spatial structure for the entire dataset through correlograms with Moran’s I index (Legendre & Fortin, 1989) and local spatial autocorrelation maps (LISA maps) for each distance class (Anselin, 1995). To test the global significance of the correlograms, we used Bonferroni’s sequential correction (Fortin & Dale, 2005). Since the residuals of the models showed significant spatial structure, we chose to calculate and add spatial autovectors (filters) as covariates in the multiple comparison tests. These filters were obtained using a connectivity matrix calculated with the maximum distance criterion (Griffith, 2013), completely eliminating the spatial autocorrelation in the residuals.

To compare the responses of the variables for each ecosystem compartment, we tested the assumptions of the covariance analysis. Normality and homoscedasticity were verified using D’Agostino’s and Levene’s tests, respectively (Zar, 2010). The homogeneity of the regression slopes was tested using interaction models (Sullivan & D’Agostino, 1996), while linearity between covariables and response variables was confirmed with Pearson’s correlation coefficients (Zar, 2010). Since the assumptions of normality and homoscedasticity were not met, we performed nonparametric covariance analyses according to Marôco (2011). As suggested for designs with multiple factors and response variables, we performed multivariate covariance analyses, also according to Marôco (2011). Significance in post hoc comparisons among groups was indicated by Tukey’s HSD test at the 5% probability level (Zar, 2010).

To compare the simultaneous responses of all collected variables, we performed a permutational multivariate analysis of variance (PERMANOVA) based on the Euclidian distance (Anderson, 2005). To test for homogeneity of dispersion between sites, we performed permutational tests of multivariate dispersion (PERMDISP) with 9999 randomizations (Anderson, 2004). Significance of variances was indicated by Tukey’s HSD post hoc test.

### 3. Results

Native forest conversion to soybean plantation led to a reduction in the total organic C stocks of  $130.5\ Mg\ C\ ha^{-1}$  and, for the rubber tree monocrop,  $48.5\ Mg\ C\ ha^{-1}$  ( $F_{(1,98)} = 174.16$ ;  $p = 0.00$ ). These decreases were 72 and 27%, respectively, which means that the conversion to rubber tree cultivation is 2.7-fold less impactful. This difference results from a sharp reduction in post-conversion aboveground biomass in the soybean plantation. The perennial crop, on the other hand, presents aboveground biomass replacement over time. This condition was clear in the 5-year-old rubber plantation, which showed similar C contents as those in soybean, revealing an important initial loss of C stocks during its establishment (Fig. 4). In this case, the similarity was evident through the results of the permutational tests of dispersion, which not only noted differences among sites ( $F_{(3,96)} = 24.728$ ;

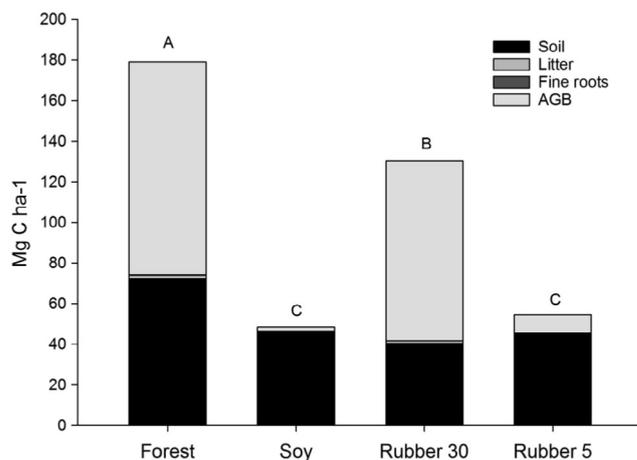


Fig. 4. Organic C stocks of the four study sites at Tanguro Farm, Querência, MT. Total values with the same letter do not statistically differ.

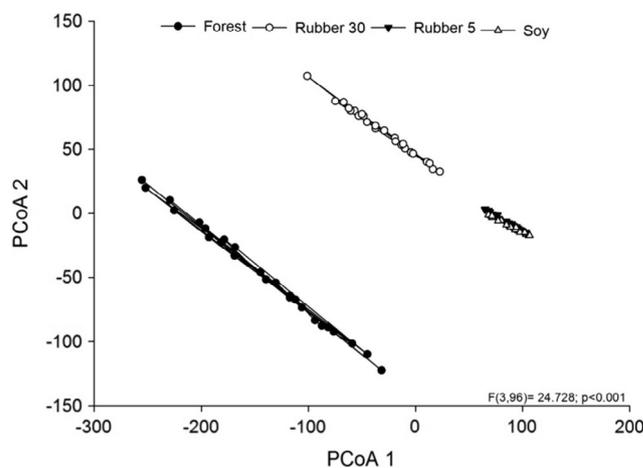


Fig. 5. Differences in C stocks indicated by bidimensional ordination of variances of the dataset with respect to the centroids of the four study sites at Tanguro Farm, Querência, MT.

$p < 0.001$ ) but also revealed the similarity between the soybean and 5-year-old rubber monocrops, which is associated with habitat homogenization by recent management practices (Fig. 5).

Our results also reveal that the response of each ecosystem compartment to the conversion of the native vegetation varied by monocrop type and season. Moreover, the respective intensity of this response also significantly varied between ecosystem compartments within each study site. This variation within and across sites highlights the complexity of the measured variables as a function of the intense transformation of the environments, which shifted from well-preserved primary forest to homogeneous agricultural systems. Thus, we describe the results obtained for each ecosystem compartment at each site below.

#### 3.1. Soil

The soils at all sites are acidic, clayey and present low cation exchange capacity (CEC) in addition to differing in all physico-chemical attributes ( $F_{MANCOVA\ nonparametric(3,192)} = 720.66$ ;  $p = 0.00$ ) except for textural class (only slightly for soy). The soils under the monocrops had lower CEC values than those under native forest since the latter undergoes the constant production and decomposition of litter (organic matter and nutrients). However, we found low aluminum saturation and a high sum of bases and base saturation in soils under crops compared to forest soils, which is related to the effect of the successive

**Table 2**

Soil physico-chemical attributes (mean  $\pm$  standard deviation) in both layers sampled at the four study sites at Tanguro Farm Querência, MT. Means followed by the same lower-case letter in a row do not statistically differ.

|  | Native Forest         |                       | Soybean                |                        | 5-Year-Old Rubber      |                      | 30-Year-Old Rubber     |                        |
|--|-----------------------|-----------------------|------------------------|------------------------|------------------------|----------------------|------------------------|------------------------|
|  | 0–10 cm               | 10–20 cm              | 0–10 cm                | 10–20 cm               | 0–10 cm                | 10–20 cm             | 0–10 cm                | 10–20 cm               |
| <i>Chemical attributes</i>                                 |                       |                       |                        |                        |                        |                      |                        |                        |
| pH H <sub>2</sub> O  | 4.25c<br>$\pm$ 0.16   | 4.28c<br>$\pm$ 0.26   | 6.24 a<br>$\pm$ 0.27   | 5.87 a<br>$\pm$ 0.41   | 5.79 ab<br>$\pm$ 0.29  | 5.56b<br>$\pm$ 0.25  | 4.74c<br>$\pm$ 0.32    | 4.58c<br>$\pm$ 0.27    |
| pH CaCl <sub>2</sub>                                       | 3.56c<br>$\pm$ 0.15   | 3.61c<br>$\pm$ 0.25   | 5.61 a<br>$\pm$ 0.27   | 5.11 ab<br>$\pm$ 0.40  | 5.03b<br>$\pm$ 0.28    | 5.79 a<br>$\pm$ 0.26 | 3.99c<br>$\pm$ 0.31    | 3.85c<br>$\pm$ 0.26    |
| CEC <sub>pH7.0</sub> (cmol <sub>c</sub> dm <sup>-3</sup> ) | 11.89 a<br>$\pm$ 1.77 | 8.32b<br>$\pm$ 1.25   | 6.91 bc<br>$\pm$ 1.10  | 5.47 d<br>$\pm$ 0.88   | 6.75c<br>$\pm$ 0.64    | 5.77 d<br>$\pm$ 0.43 | 6.54c<br>$\pm$ 0.68    | 5.39 d<br>$\pm$ 0.56   |
| SumBase (cmol <sub>c</sub> dm <sup>-3</sup> )              | 0.72b<br>$\pm$ 0.14   | 0.55b<br>$\pm$ 0.17   | 4.65 a<br>$\pm$ 1.07   | 2.54 a<br>$\pm$ 0.85   | 3.11 a<br>$\pm$ 0.67   | 2.23 a<br>$\pm$ 0.51 | 0.72b<br>$\pm$ 0.38    | 0.45b<br>$\pm$ 0.22    |
| Base sat. (%)  | 6.00c<br>$\pm$ 1.18   | 6.60c<br>$\pm$ 1.87   | 65.13 a<br>$\pm$ 10.71 | 48.44 a<br>$\pm$ 13.04 | 45.63 ab<br>$\pm$ 8.25 | 38.53b<br>$\pm$ 7.75 | 11.35c<br>$\pm$ 5.24   | 8.94c<br>$\pm$ 4.04    |
| Al sat. (%)  | 72.31 a<br>$\pm$ 6.00 | 76.37 a<br>$\pm$ 6.51 | 0.00b<br>$\pm$ 0.00    | 3.62b<br>$\pm$ 6.76    | 2.55b<br>$\pm$ 4.82    | 6.82b<br>$\pm$ 8.36  | 55.93 a<br>$\pm$ 16.98 | 68.21 a<br>$\pm$ 13.76 |
| <i>Physical attributes</i>                                 |                       |                       |                        |                        |                        |                      |                        |                        |
| Sand (g kg <sup>-1</sup> )                                 | 463.4 $\pm$ 60.7      |                       | 309.8 $\pm$ 18.1       |                        | 376.4 $\pm$ 7.6        |                      | 420.0 $\pm$ 13.7       |                        |
| Silt (g kg <sup>-1</sup> )                                 | 124.6 $\pm$ 10.8      |                       | 155.0 $\pm$ 7.8        |                        | 137.8 $\pm$ 3.3        |                      | 133.6 $\pm$ 3.1        |                        |
| Clay (g kg <sup>-1</sup> )                                 | 412.0 $\pm$ 50.1      |                       | 535.2 $\pm$ 11.8       |                        | 485.8 $\pm$ 5.3        |                      | 446.4 $\pm$ 12.0       |                        |
| Textural Class   | Clay                  |                       | Clay                   |                        | Clay                   |                      | Clay                   |                        |

additions of limestone and fertilizers to those managed soils (Table 2).

The conversion of native forest to both monocrops reduced soil organic C stocks (C<sub>org</sub>) by 39% on average in the top 20 cm ( $F_{\text{ANCOVA nonparametric}(3,192)} = 454.23$ ;  $p = 0.00$ ). The 30-year-old rubber plantation was the system that had the highest soil C deficit, with a surprising reduction of 31.9 Mg C ha<sup>-1</sup> compared to the forest. We did not find significant variation in C stocks between the two soil depths at any of the sites except the oldest rubber plantation, where a 33% decrease in the 10–20 cm soil layer was found when compared to the superficial layer (Table 3).

### 3.2. Litter

Carbon stocks significantly decreased in the litter layer ( $F_{\text{MANCOVA nonparametric}(3,192)} = 2395.8$ ;  $p = 0.00$ ), where a greater difference was found between the native forest and the youngest rubber plantation in contrast to the mature plantation (Table 4). We recorded 4.2- and 3.7-fold reductions in the litter layers of the soybean and 5-year rubber plantations, respectively, revealing the high impact of these types of land use on the soil organic layer. No differences in seasonality were observed for biomass or C stocks, which reveals the lack of a seasonal effect on those variables. The forest and soybean litter layers showed a higher thickness and volume during the rainy season, while the density

**Table 3**

Soil organic C contents, apparent density (ADS) and C stocks (mean  $\pm$  SD) at the four study sites at Tanguro Farm, Querência, MT. Means followed by the same lower-case letter in a row for the 0–10 and 10–20 depths and the same upper-case letter in a column for the total depth (0–20 cm) do not statistically differ.

| Site               | Depth (cm) | C Content (g C dm <sup>-3</sup> ) | ADS (g cm <sup>-3</sup> ) | C <sub>org</sub> stock (Mg C ha <sup>-1</sup> ) |                     |
|--------------------|------------|-----------------------------------|---------------------------|---|---------------------|
|                    |            |                                   |                           | Calculated                                      | Corrected           |
| Forest             | 0–10       | 27.94                             | 1.40                      | 39.11 a $\pm$ 7.92                              | 39.11 a $\pm$ 7.92  |
|                    | 10–20      | 19.51                             | 1.70                      | 33.17 a $\pm$ 7.50                              | 33.17 a $\pm$ 7.50  |
|                    | 0–20       |                                   |                           | 72.28 A $\pm$ 7.71                              | 72.28 A $\pm$ 7.71  |
| Soybean            | 0–10       | 15.87                             | 1.73                      | 27.45b $\pm$ 7.77                               | 25.30b $\pm$ 6.90   |
|                    | 10–20      | 11.75                             | 1.94                      | 22.79b $\pm$ 6.74                               | 20.48b $\pm$ 6.33   |
|                    | 0–20       |                                   |                           | 50.25B $\pm$ 7.25                               | 45.78B $\pm$ 6.61   |
| 5-Year-old rubber  | 0–10       | 15.64                             | 1.64                      | 25.65b $\pm$ 6.94                               | 25.64b $\pm$ 6.9    |
|                    | 10–20      | 12.54                             | 1.55                      | 19.44 bc $\pm$ 4.26                             | 19.38 bc $\pm$ 4.18 |
|                    | 0–20       |                                   |                           | 45.08B $\pm$ 5.60                               | 45.02B $\pm$ 5.54   |
| 30-Year-old rubber | 0–10       | 15.03                             | 1.31                      | 19.69 bc $\pm$ 2.21                             | 24.16b $\pm$ 3.97   |
|                    | 10–20      | 11.34                             | 1.26                      | 14.29c $\pm$ 2.69                               | 16.22c $\pm$ 2.81   |
|                    | 0–20       |                                   |                           | 33.98C $\pm$ 2.45                               | 40.38C $\pm$ 3.39   |

was higher in the dry season for soybean (Table 4).

### 3.3. Fine roots

Our results indicate large alterations in the C stocks in this compartment, not only among sites but also between depths and seasons ( $F_{\text{ANCOVA nonparametric}(3,784)} = 5.45$ ;  $p = 0.00$ ). In the oldest rubber plantation, the fine root biomass was nearly four-fold lower than in the forest, while the other sites showed nearly a 20-fold decrease. We also found a greater amount of root biomass in the 0–10 cm layer for all land uses. We observed that the forest presented a higher quantity of fine roots during the dry season, while the opposite was found for soybean and the oldest rubber plantation (Table 5).

Considering the fine root total C stocks at the 20 cm soil depth (total layer), the impacts of native forest conversion to monocrops were also significant ( $F_{\text{ANCOVA nonparametric}(3,192)} = 716.12$ ;  $p = 0.00$ ). Land-use change to soybean and rubber plantations reduces C stocks by over 90% in this compartment and changes their allocation patterns (Table 6), with possible modifications to biogeochemical cycling.

### 3.4. Aboveground biomass

Forest conversion caused a loss of 102.8 Mg C ha<sup>-1</sup> of aboveground

**Table 4**

Thickness, volume, density, total biomass and C stocks of the litter layer (mean  $\pm$  SD) at the four study sites during both seasons at Tanguro Farm, Querência, MT. Means followed by the same lower-case letter in a row and the same upper-case letter in a column do not statistically differ.

| Site  | Dry                  | Rainy               |
|---|----------------------|---------------------|
| <i>Thickness (cm)</i>                         |                      |                     |
| Forest  | 1.11 Ab $\pm$ 0.38   | 1.31 Aa $\pm$ 0.29  |
| Soybean                                       | 0.41 Bb $\pm$ 0.37   | 0.92 Ba $\pm$ 0.45  |
| 5-year-old rubber                             | 0.31 Ba $\pm$ 0.46   | 0.16 Ca $\pm$ 0.17  |
| 30-year-old rubber                            | 1.07 Aa $\pm$ 0.35   | 1.08 Aa $\pm$ 0.45  |
| <i>Volume (m<sup>3</sup> ha<sup>-1</sup>)</i> |                      |                     |
| Forest  | 111 Ab $\pm$ 38.4    | 131 Aa $\pm$ 29.04  |
| Soybean                                       | 41 Bb $\pm$ 36.8     | 92 Ba $\pm$ 45.35   |
| 5-year-old rubber                             | 30.5 Ba $\pm$ 46.64  | 15.5 Ca $\pm$ 17.68 |
| 30-year-old rubber                            | 107 Aa $\pm$ 35.41   | 108 Aa $\pm$ 44.71  |
| <i>Density (g cm<sup>-3</sup>)</i>            |                      |                     |
| Forest  | 0.035 Aa $\pm$ 0.012 | 0.03 Aa $\pm$ 0.008 |
| Soybean                                       | 0.035 Aa $\pm$ 0.05  | 0.01 Bb $\pm$ 0.008 |
| 5-year-old rubber                             | 0.045 Aa $\pm$ 0.04  | 0.05 Aa $\pm$ 0.05  |
| 30-year-old rubber                            | 0.034 Aa $\pm$ 0.01  | 0.03 Aa $\pm$ 0.01  |
| <i>Biomass (Mg ha<sup>-1</sup>)</i>           |                      |                     |
| Forest  | 3.78 Aa $\pm$ 1.32   | 4.16 Aa $\pm$ 1.39  |
| Soybean                                       | 1.23 Ba $\pm$ 0.65   | 0.98 Ba $\pm$ 0.37  |
| 5-year-old rubber                             | 0.85 Ba $\pm$ 0.45   | 1.13 Ba $\pm$ 0.69  |
| 30-year-old rubber                            | 3.53 Aa $\pm$ 1.15   | 3.21 Aa $\pm$ 0.66  |
| <i>C Stock (Mg ha<sup>-1</sup>)</i>           |                      |                     |
| Forest  | 1.47 Aa $\pm$ 0.51   | 1.62 Aa $\pm$ 0.54  |
| Soybean                                       | 0.48 Ba $\pm$ 0.64   | 0.38 Ba $\pm$ 0.52  |
| 5-year-old rubber                             | 0.33 Ba $\pm$ 0.45   | 0.44 Ba $\pm$ 0.68  |
| 30-year-old rubber                            | 1.38 Aa $\pm$ 0.44   | 1.25 Aa $\pm$ 0.65  |

**Table 5**

Fine root biomass in Mg ha<sup>-1</sup> (mean  $\pm$  SD) at different depths and during different seasons at the four study sites at Tanguro Farm, Querência, MT. Means followed by the same upper-case letter in a column (same depth), the same lower-case letter in a row and subscripted in a column (between depths) do not statistically differ.

| Site               | Dry                               | Rainy                             |
|--------------------|-----------------------------------|-----------------------------------|
| <i>0–10 cm</i>     |                                   |                                   |
| Forest             | 0.855 Aa <sub>a</sub> $\pm$ 0.571 | 0.593 Ab <sub>a</sub> $\pm$ 0.244 |
| Soybean            | 0.012 Cb <sub>a</sub> $\pm$ 0.016 | 0.055 Ca <sub>a</sub> $\pm$ 0.067 |
| 5-year-old rubber  | 0.034 Ca <sub>a</sub> $\pm$ 0.051 | 0.046 Ca <sub>a</sub> $\pm$ 0.058 |
| 30-year-old rubber | 0.094 Bb <sub>a</sub> $\pm$ 0.072 | 0.205 Ba <sub>a</sub> $\pm$ 0.147 |
| <i>10–20 cm</i>    |                                   |                                   |
| Forest             | 0.258 Aa <sub>b</sub> $\pm$ 0.164 | 0.166 Ab <sub>b</sub> $\pm$ 0.091 |
| Soy                | 0.003 Cb <sub>b</sub> $\pm$ 0.004 | 0.007 Ca <sub>b</sub> $\pm$ 0.010 |
| 5-year-old rubber  | 0.018 Ca <sub>b</sub> $\pm$ 0.028 | 0.007 Cb <sub>b</sub> $\pm$ 0.036 |
| 30-year-old rubber | 0.090 Bb <sub>b</sub> $\pm$ 0.155 | 0.098 Ba <sub>b</sub> $\pm$ 0.113 |

**Table 6**

Organic C stocks in Mg ha<sup>-1</sup> (mean  $\pm$  SD) in fine roots at 20 cm depth at the four sites studied at Tanguro Farm, Querência, MT, during the rainy and dry seasons. Means followed by the same upper-case letter in a column and the same lower-case letter in a row do not statistically differ.

| Site               | Dry                  | Rainy                |
|--------------------|----------------------|----------------------|
| Forest             | 0.540 Aa $\pm$ 0.248 | 0.368 Ab $\pm$ 0.136 |
| Soybean            | 0.008 Cb $\pm$ 0.006 | 0.030 Ca $\pm$ 0.026 |
| 5-year-old rubber  | 0.026 Ca $\pm$ 0.020 | 0.036 Ca $\pm$ 0.023 |
| 30-year-old rubber | 0.090 Bb $\pm$ 0.057 | 0.148 Ba $\pm$ 0.078 |

biomass in the soybean plantation, while only 16.3 Mg C ha<sup>-1</sup> was lost in the 30-year-old rubber monocrop (Table 7). These results indicate decreases in aboveground biomass and C by nearly 98% for soybean and only 15% for the mature rubber plantation. Reductions in C stocks in the aboveground biomass were also high in the youngest rubber plantation, and there were significant differences among all of the types

**Table 7**

Aboveground biomass and C stocks in Mg ha<sup>-1</sup> (mean  $\pm$  SD) at the four study sites at Tanguro Farm, Querência, MT. Means followed by the same upper-case letter in a column do not statistically differ.

| Site               | Aboveground biomass  | C Stock              | C Loss |
|--------------------|----------------------|----------------------|--------|
| Forest             | 216.26 A $\pm$ 93.07 | 104.88 A $\pm$ 45.14 | –      |
| Soybean            | 4.39 D $\pm$ 1.40    | 2.13 D $\pm$ 0.67    | 102.75 |
| 5-year-old rubber  | 18.64 C $\pm$ 4.52   | 9.04 C $\pm$ 2.19    | 95.84  |
| 30-year-old rubber | 182.75 B $\pm$ 39.36 | 88.63 B $\pm$ 19.09  | 16.25  |

of land use and management ( $F_{\text{ANCOVA nonparametric}(3,95)} = 593.17$ ;  $p = 0.00$ ), showing the following order: native forest > mature rubber plantation > young rubber plantation > soybean plantation.

#### 4. Discussion

We investigated the carbon losses related to the land-use in the “Arc of Deforestation” in the Southern Amazon. Our results indicate a collapse in the carbon stocks in the ecosystem caused by the conversion of native forests to soybean and rubber plantations. This effect is mainly due to the reduction of aboveground biomass, a continuous decrease of the carbon content in the soil and biomass losses of fine root mat. On the other hand, the recovery of C stocks in the aboveground biomass of rubber trees reached nearly 84% after 30 years, indicating this crop’s strong capacity to mitigate the effects of deforestation on CO<sub>2</sub> emissions. For example, if we consider the total area destined for soybean and rubber monocropping in Amazonia (Abiove, 2012; IBGE, 2013), we would infer a minimum biomass C loss of  $274 \cdot 10^6$  Mg in the conversion to soybean in contrast to only  $1.49 \cdot 10^6$  Mg in that to rubber tree monocropping. However, it is necessary to adjust the calculations and predictions to other sites within Amazonia owing to important variation in species composition, wood density and aboveground biomass. In addition, some areas in Amazonia are not suitable for rubber trees because of pathogens (e.g., *Microcyclus ulei*), which thus must be excluded from calculations. Even so, those values indicate significant C loss as a result of the substitution of native vegetation by arable lands, mainly in the current scenario of deforestation in southern Amazonia.

Other studies have also found reductions in soil C<sub>org</sub> stocks following a shift from native vegetation to soybean plantations (Bayer et al., 2002; Calonego et al., 2011; Perrin et al., 2014), by nearly 16 to 79%. Nevertheless, we did not expect the soil under the soybean crop to present a higher C<sub>org</sub> stock than that under the mature rubber plantation. Several authors (Moraes & Moreira, 2003; Alvarenga & Carmo, 2006; Patricio, 2014) have stressed that rubber plantations are systems with higher potential to store C<sub>org</sub> in the soil compared to annual monocrops, as no intensive tilling occurs, and the amount of litterfall is similar to that in areas with native vegetation. In addition, previous studies have shown post-conversion soil C losses in rubber plantations of between 14 and 35% (Schroth et al. 2002; Yang et al. 2004; Blécourt et al. 2013), that is, a slight loss compared to that observed in conversions to soybean or other annual crops. However, this trend was not detected in our study, and estimates of loss by conversion to rubber (44.13%) surpassed those found in the literature.

In contrast, the soil C<sub>org</sub> stock in the 5-year-old rubber crop was higher than that estimated for the oldest crop, which can be related to the land-use history. Before the 5-year-old rubber plantation was established, that land had been used for pasture for approximately 26 years. The capacity of pastures to store C in the soil (Assad et al., 2013), the time during which pasture was the type of land use, and the establishment of the rubber crop contributed to the higher C accumulation in that soil. A similar result was found by Saengruksawong et al. (2012), who associated the large amount of C stored in soils under young rubber plantations to the history of land use and management employed until 10 years following establishment (e.g., plowing and fertilization).

Our results reveal that all crops have much lower soil C stocks than forest soils, suggesting a near-permanent loss of soil C – i.e., soil C stocks have no resilience in terms of their ability to recover from land-use-change-induced losses, at least over the period of years to decades. Moraes et al. (1995) estimated that 45% of all soil C in Amazonia is concentrated in the top 20 cm, which may add up to 21 Pg C in the entire basin. Based on our results, the mean reduction in soil  $C_{org}$  stocks in the 20 cm depth would be 35%, suggesting C losses to the atmosphere of approximately 13.65 Pg C if our values were extrapolated to all of Amazonia. If we consider the 40% deforestation estimate for Amazonia by 2050 (Soares-Filho et al., 2006), this would potentially transfer 5.46 Pg C to the atmosphere only from the soil. However, to confirm this estimate, new studies must be conducted across distinct soils and management systems. Similar studies should be conducted globally, since high rates of deforestation are also observed in other tropical forests, such as in Asia and Africa, where the average above-ground carbon losses are estimated at 26.6 and 16.6 Mg C ha<sup>-1</sup>, respectively (Baccini et al., 2017).

No differences in C stocks in the litter layer were found between native forest and the 30-year-old rubber crop, which indicates the capacity of *Hevea brasiliensis* clone RRIM 600 to maintain this ecosystem function comparably to that in the original vegetation or even increase it, as shown by Ren et al. (1999) in rubber plantations in southwest China. In certain cases, this situation can be reversed, with lower foliar input, as recorded by Kindel et al. (2006) in clone RRIM 600 compared to primary forests in Brazil. Regardless, the similar amount of litter between the rubber crop and forest does not enhance the soil C content. Such a condition indicates that recalcitrant soil C stocks operate on particular time scale in relation to agricultural and silvicultural management. The apparent lack of recovery after three decades suggests that the readjustment of soil C might require a century or more. Thus, for all practical carbon management purposes, this soil C has been permanently lost from the ecosystem.

Apart from the implications in terms of C stocks, the differences in the amount of litter accumulated across the study sites may affect the soil C balance. Since soybean and 5-year-old rubber plantations have lower amounts of litter, we may infer that C incorporation into the soil at both sites is lower than that in the forest and at the mature rubber plantation. This means that there is low input and high losses of particulate  $C_{org}$  due to soil tillage and therefore low storage potential in that fraction. The aforementioned nutritional imbalance is most likely the greatest change in biogeochemical cycles caused by the conversion of primary forest to agricultural land, which leads to mandatory annual fertilization considering the dystrophic condition of the soil.

Similarly, the fine root biomass was lower in both the soybean and 5-year-old rubber plantations. This quantity is not significant at the ecosystem level, but the importance to cycling and nutritional balance dynamics is crucial, especially in ecosystems with dystrophic soils (Vitousek & Sanford, 1986). Nonetheless, the seasonal behavior of roots is still controversial. Our results for primary forest, for instance, differ from those found in other studies (Green et al., 2005; Metcalfe et al., 2008; Jiménez et al., 2009), which found higher fine root biomass during the rainy season. The pattern described by those authors was recorded here for soy and mature rubber, evincing discrepancies in site nutritional dynamics. Generally, an increase in fine root production is expected during the peak of the rainy season to enhance water and nutrient absorption during the period of highest availability (Green et al., 2005). However, it must be taken into account that such production is more intense at the soil-litter interface, where nutrients are released in greater quantities (Vitousek, 1984). Therefore, during the rainy season, the majority of fine root biomass may be strategically concentrated above the soil level, in contact with the litter layer rather than in the hemiorganic layers. This nutrient absorption strategy is commonly found in tropical forests, systems with higher root competition, niche exploitation and strong trade-offs.

The unexpected root biomass accumulation during the dry season

may be strongly related to the timing of plant responses to soil moisture (Espeleta & Clark, 2007). This is possible because canopy water interception and transpiration, among other factors, can delay soil water replacement so that the highest moisture content of the soil may occur during the mid (or late) rainy season. This scenario enhances root production during that period, which can be extended until the moisture content begins to decline during the early dry season. Moreover, the topsoil water holding capacity as a function of texture and porosity may also enhance production and lead to higher root phenological activity during the rainy-dry interseason. In our soils, the clay texture and the atypical pluviometric behavior during the sampling year enabled longer water retention and consequently a great accumulation of live roots during the dry season. Therefore, it is clear that climate change may interfere with mechanisms of root renewal as well as the conversion of native vegetation to agroecosystems.

It is noteworthy to mention that there is no literature on fine root C stocks in the Amazon-Cerrado transition. This highlights the importance of new regional studies on this compartment since mechanisms of turnover and biomass allocation in the superficial root mat may indicate important aspects in the dynamics of C and nutrient cycles. Thus, this information can be very useful for the elaboration of strategies for sustainable land use and management.

Even though the conversion of native forest to soybean and rubber tree crops may have caused reductions in soil, litter and fine root C stocks, the aboveground biomass was undoubtedly the compartment that quantitatively most contributed to the C loss. The establishment of soybean led to a reduction in C storage by 98% in that compartment, while the youngest rubber plantation showed a 91% decrease. Nevertheless, the rubber tree aboveground biomass C stock tends to increase considerably over time without reaching that found in the native vegetation. Furthermore, the annual harvesting of soybeans constantly removes massive amounts of C from the system, which is disadvantageous when compared to rubber plantations.

We emphasize, however, that despite the longer rotation cycles of rubber plantations (30 years on average), there are also massive C losses during crop renovation. Hence, although these systems reach a stationary state equilibrium, the removal of trees to renew the rubber plantation and use the wood will result in substantial C loss and therefore large structural and functional changes that will later be offset by planting new rubber trees. Even so, the resident C stock in the agroecosystem between planting, harvesting and reforming cycles will still be considerable at a large scale, as planting and renovation are not concurrent.

We can consider that the rubber tree plantation is not the best solution for carbon emissions and biodiversity losses in Amazon. However, the management of rubber tree stands of various ages can be considered a viable alternative to mitigate C stock losses compared to annual crops since the impact of harvesting rubber trees for renovation is minimized by the size of the stock. Furthermore, rotational management of distinct stand ages guarantees continuous income to the farmer. That is, such activities can be economically and environmentally sustainable in terms of C emissions for southern Amazonia, possibly minimizing the impacts of agribusiness frontier expansion in the Arc of Deforestation.

Additionally, our results reveal that primary forest conversion to perennial and annual monocrops in Amazonia contributes, distinctly, to changes in the global C cycle. This condition must be duly considered in the construction of carbon-climate models and the elaboration of public policies for all tropical forests, not only in South America. However, new studies should test the effects of primary forest conversion to agricultural and pasture land based on soil C stocks to a depth of 4 m. Furthermore, it is necessary to test for conversion impacts on nutrient cycling and other ecosystem functions, such as the regulation of the hydrological cycle. These ecosystem functions are key in maintaining tropical forests on nutrient-poor soils (Oliveira et al., 2017). Lastly, we suggest economic viability studies addressing alternatives to cereal

crops in Amazonia, such as rubber trees, which occupy less territory and can be managed by small farmers.

## 5. Conclusions

The conversion of native forest to a soybean plantation caused a 2.7 times greater loss in total C stocks compared to forest-to-rubber tree conversion. This can be related to the low aboveground biomass cover and litter enrichment of annual crops. The mean reduction in soil C<sub>org</sub> stocks by 35% in the top 20 cm in both crops suggests a permanent loss of soil C and low resilience of this component to environmental alterations. The conversion of native forests to soybean or rubber tree plantations could represent soil C losses to the atmosphere of up to 5.46 Pg considering the 40% deforestation estimate for all of Amazonia by 2050.

Even though both monocrops present similar changes in the annual pattern of root biomass allocation, rubber trees have a more developed litter layer with a 4.2-fold greater C stock. This fact in addition to well-established nutrient cycling contribute to rubber tree cropping as being less impactful in terms of agroecosystem function and production. Therefore, rubber tree cropping may be considered to be an ecological and economical alternative to extensive soybean cultivation, thereby contributing to a reduction in global C emissions. In addition, rubber plantations are likely contributing to other key ecosystem services, such as recharging of atmospheric moisture and reducing local temperature. Therefore, if this and other tree crops were widely adopted as an alternative to soybean cultivation could minimize the impacts of land use in Southern Amazonia would be substantially reduced.

On the other hand, as observed in any monoculture, the conversion of native forests into rubber plantations results in high losses of biodiversity and some important ecosystem functions. We emphasize that more detailed analyses about trophic connectivity, turnover processes and C fluxes among the various ecosystem compartments are necessary and can contribute to concomitantly reducing the costs of crop establishment and management, making the system ecologically and economically more sustainable.

## Acknowledgements

We thank the Brazilian National Council of Science and Technology (Conselho Nacional de Desenvolvimento Científico e Tecnológico - CNPq) for the PVE project #401279 (Professor Oliver L. Phillips, University of Leeds-UK) and CNPq/PPBio project (#457602). B. H. Marimon-Junior and B. S. Marimon received productivity grants (PQ-2) from the CNPq. This study was also supported by PELD (LTER)/CNPq (#403725/2012-7), and USA-NAS/PEER (#PGA-200005316).

## References

Abiove, 2012. Moratória da Soja: 5º ano do mapeamento e monitoramento do plantio de soja no bioma Amazônia. Grupo de Trabalho da Soja, p. 23. Disponível em: < <http://www.abiove.org.br> > (Acesso em: 22 dez. 2015).

Alvarenga, A.P., Carmo, C.A.F.S., 2006. Sequestro de Carbono: quantificação em seringais de cultivo e na vegetação natural. Embrapa Solos 338.

Anderson, M.J., 2004. PERMDISP: a FORTRAN Computer Program for Permutational Analysis of Multivariate Dispersions (For Any Two-Factor ANOVA Design) Using Permutation Tests. Department of Statistics, University of Auckland.

Anderson, M.J., 2005. PERMANOVA: a FORTRAN Computer Program for Permutational Multivariate Analysis of Variance. Department of Statistics, University of Auckland.

Anselin, L., 1995. Local indicators of spatial association – LISA. *Geogr. Anal.* 2, 93–115. <http://dx.doi.org/10.1111/j.1538-4632.1995.tb00338.x>.

Arevalo, L.A., et al., 2002. Metodologia para estimar o estoque de carbono em diferentes sistemas de uso da terra. Embrapa Florestas (Documentos, 73), 41.

Assad, E.D., et al., 2013. Changes in soil carbon stocks in Brazil due to land use: paired site comparisons and a regional pasture soil survey. *Biogeosci. Discuss.* 10, 5499–5533. <http://dx.doi.org/10.5194/bg-10-6141-2013>.

Baccini, A., et al., 2017. Tropical forests are a net carbon source based on aboveground measurements of gain and loss. *Science* 358 (6360), 230–234. <http://dx.doi.org/10.1126/science.aam5962>.

Baker, T.R., et al., 2004. Variation in wood density determines spatial patterns in Amazonian forest biomass. *Glob. Change Biol.* 10 (5), 545–562. <http://dx.doi.org/10.1111/j.1365-2486.2004.00751.x>.

Balch, et al., 2008. A negative fire feedback in a Transitional Forest of Southeastern Amazonia. *Glob. Change Biol.* 14 (10), 2276–2287. <http://dx.doi.org/10.1111/j.1365-2486.2008.01655.x>.

Baldotto, M.A., et al., 2015. Estoque e frações de carbono orgânico e fertilidade de solo sob floresta, agricultura e pecuária. *Rev. Ceres* 62 (3), 301–309.

Bayer, C., et al., 2002. Carbon stocks in organic matter fractions as affected by land use and soil management, with emphasis on no-tillage effect. *Ciência Rural* 32 (3), 401–406. <http://dx.doi.org/10.1590/50103-84782002000300006>.

Blake, G.R., Hartge, K.H., 1986. Bulk density. In: Klute, A. (Ed.), *Methods of soil analysis. Part 1. Physical and mineralogical methods*, second ed. American Society of Agronomy, Soil Science Society of America, Wisconsin, pp. 363–375.

Blécourt, M., et al., 2013. Soil carbon stocks decrease following conversion of secondary forests to rubber (*Hevea brasiliensis*) plantations. *PLoS One* 8 (7), e69357. <http://dx.doi.org/10.1371/journal.pone.0069357>.

Buckeridge, M.S., 2008. *Biologia e Mudanças Climáticas no Brasil*. RiMa, p. 316.

Calonego, J.C., et al., 2011. Estoques de carbono e propriedades físicas de solos submetidos a diferentes sistemas de manejo. *Revista Caatinga* 5 (2), 128–135.

Chave, J., et al., 2014. Improved allometric models to estimate the aboveground biomass of tropical trees. *Glob. Change Biol.* 20, 3177–3190. <http://dx.doi.org/10.1111/gcb.12629>.

Coe, M.T., et al., 2013. Deforestation and climate feedbacks threaten the ecological integrity of south – southeastern Amazonia. *Philos. Trans. R Soc. Lond. B* 368, 20120155. <http://dx.doi.org/10.1098/rstb.2012.0155>.

CONAB – Companhia Nacional de Abastecimento. 2016. Disponível em: < <http://www.conab.gov.br/conteudos.php?a=1252> > (Acesso em: 11 jan. 2016).

Costa, M.H., Pires, G.F., 2010. Effects of Amazon and Central Brazil deforestation scenarios on the duration of the dry season in the arc of deforestation. *Int. J. Climatol.* 30 (13), 1970–1979. <http://dx.doi.org/10.1002/joc.2048>.

Davidson, E.A., 2004. Loss of nutrients from terrestrial ecosystems to streams and the atmosphere following land use change in Amazonia. In: DeFries, R. (Ed.), *Ecosystem and Land Use Change*, Geophysical Monograph Series 153. American Geophysical Union, pp. 147–158.

Don, A., et al., 2011. Impact of tropical land-use change on soil organic carbon stocks – a meta-analysis. *Glob. Change Biol.* 17 (4), 1658–1670. <http://dx.doi.org/10.1111/j.1365-2486.2010.02336.x>.

Drinkwater, L.E., Snapp, S.S., 2007. Nutrients in agroecosystems: rethinking the management paradigm. *Adv. Agron.* 92, 163–186. [http://dx.doi.org/10.1016/S0065-2113\(04\)92003-2](http://dx.doi.org/10.1016/S0065-2113(04)92003-2).

Embrapa. Centro Nacional de Pesquisa de Solos, 1997. Manual de métodos de análise de Solo. 2. ed. Embrapa Solos, p. 212.

Espeleta, J.F., Clark, D.A., 2007. Multi-scale variation in fine roots biomass in a tropical rain forest: a seven-year study. *Ecol. Monogr.* 77 (3), 377–404. <http://dx.doi.org/10.1890/06-1257.1>.

Farias, J., et al., 2016. Survival and growth of native *Tachigali vulgaris* and exotic *Eucalyptus urophylla* x *Eucalyptus grandis* trees in degraded soils with biochar amendment in southern Amazonia. *Forest Ecol. Manage.* 368, 173–182. <http://dx.doi.org/10.1016/j.foreco.2016.03.022>.

Fortin, M.J., Dale, M.R.T., 2005. *Spatial Analysis. A Guide for Ecologists*. Cambridge University Press, pp. 450.

Friedlingstein, P., et al., 2010. Update on CO<sub>2</sub> emissions. *Nat Geosci* 3, 811–812. <http://dx.doi.org/10.1038/ngeo1022>.

Fuguiwara, M. et al. 1994. A técnica de estudo de raízes pelo método do trado. Instituto Agronômico de Campinas, p. 9 (Boletim Técnico, 153).

Green, J.J., et al., 2005. Fine root dynamics in a tropical rain forest is influenced by rainfall. *Plant Soil* 276, 23–32. <http://dx.doi.org/10.1007/s11104-004-0331-3>.

Griffith, D., 2013. *Spatial Autocorrelation and Spatial Filtering: Gaining Understanding Through Theory and Scientific Visualization*. Springer Science & Business Media, pp. 250.

Hendricks, J.J., et al., 2006. Assessing the patterns and control of fine roots dynamics: an empirical test and methodological review. *J. Ecol.* 94 (1), 40–57. <http://dx.doi.org/10.1111/j.1365-2745.2005.01067.x>.

Higuchi, N., et al., 1998. Biomassa da parte aérea da vegetação de floresta tropical úmida de terra firme da Amazônia Brasileira. *Acta Amazonica* 28 (2), 153–165.

IBGE, 2013. Produção Agrícola Municipal: culturas temporárias e permanentes. PAM 39, 1–101.

IBGE – Instituto Brasileiro de Geografia e Estatística. 2016. Mapas: Produção Agrícola e Vegetação Natural. Disponível em: < <http://mapas.ibge.gov.br/tematicos/agricultura> > (Acesso em: 01 jan. 2016).

Jiménez, E.M., et al., 2009. Fine roots dynamics for forests on contrasting soils in the Colombian Amazon. *Biogeosciences* 6, 2809–2827. [0.5194/bg-6-2809-2009](http://dx.doi.org/10.5194/bg-6-2809-2009).

Jiménez-Muñoz, J.C., et al., 2013. Spatial and temporal patterns of the recent warming of the Amazon forest. *J. Geophys. Res. Atmos.* 118 (11), 5204–5215. <http://dx.doi.org/10.1002/jgrd.50456>.

Kindel, A. et al., 2006. Ciclagem de nutrientes e carbono na serapilheira de seringais e fragmentos de Mata Atlântica. In: Alvarenga, A.P., et al. (Eds.), *Sequestro de Carbono: Quantificação em Seringais de Cultivo e na Vegetação Natural*. Embrapa Solos, pp. 135–158.

Kotowska, M.M., et al., 2015. Quantifying above and belowground biomass carbon loss with forest conversion in tropical lowlands of Sumatra (Indonesia). *Glob. Change Biol.* 21 (10), 3620–3634. <http://dx.doi.org/10.1111/gcb.12979>.

Kotowska, M.M., et al., 2016. Conversion of tropical lowland forest reduces nutrient return through litterfall, and alters nutrient use efficiency and seasonality of net primary production. *Oecologia* 180 (2), 601–618. <http://dx.doi.org/10.1007/s00442-015-3481-5>.

Legendre, P., Fortin, M.J.F., 1989. Spatial pattern and ecological analysis. *Vegetatio* 80,

- 107–138. <http://dx.doi.org/10.1007/BF00048036>.
- Lopez-Gonzalez, G. et al., 2011. ForestPlots.net Database. 2011. Disponível em: < <http://www.forestplots.net> > (Acesso em: 30 dez. 2013).
- Maia, S.M.F., et al., 2010. Soil organic carbon stock change due to land use activity along the agricultural frontier of the southwestern Amazon, Brazil, between 1970 and 2003. *Glob. Change Biol.* 16 (10), 2775–2788. <http://dx.doi.org/10.1111/j.1365-2486.2009.02105.x>.
- Marimon-Junior, B.H., Hay, J.D., 2008. A new instrument for measurement and collection of quantitative samples of the litter layer in forests. *Forest Ecol. Manage.* 255, 2244–2250. <http://dx.doi.org/10.1016/j.foreco.2008.01.037>.
- Marôco, J., 2011. Análise estatística com o SPSS Statistics. fifth ed. Report Number. pp. 356–361.
- Metcalfe, D.B., et al., 2008. The effects of water availability on root growth and morphology in an Amazon rainforest. *Plant Soil* 311, 189–199. <http://dx.doi.org/10.1007/s11104-008-9670-9>.
- Moore, J.C., et al., 2005. Modeling trophic pathways, nutrient cycling, and dynamic stability in soils. *Pedobiologia* 49 (6), 499–510.
- Moraes, V.H.F., Moreira, A., 2003. Potencial de Contribuição da Cultura da Seringueira para a Conservação do Ambiente no Trópico Úmido. Embrapa Amazônia Ocidental, p. 18. <http://doi.org/10.1016/j.pedobi.2005.05.008>.
- Moraes, J.L., et al., 1995. Soil carbon stocks of the Brazilian Amazon basin. *Soil Sci. Soc. Am. J.* 59 (1), 244–247. <http://dx.doi.org/10.2136/sssaj1995.03615995005900010038x>.
- Neill, C., et al., 2013. Watershed responses to Amazon soya bean cropland expansion and intensification. *Philos. Trans. R Soc. Lond. B* 368, 20120425. <http://dx.doi.org/10.1098/rstb.2012.0425>.
- Nelson, B.W., et al., 1999. Allometric regressions for improved estimate of secondary forest biomass in the Central Amazon. *Forest Ecol. Manage.* 117, 149–167. [http://dx.doi.org/10.1016/S0378-1127\(98\)00475-7](http://dx.doi.org/10.1016/S0378-1127(98)00475-7).
- Nelson, D.W., Sommers, L.E., 1996. Total carbon, organic carbon, and organic matter. In: Page, A.L. et al., (Ed.), *Methods of Soil Analysis, Part 2, second ed.*, Agronomy. 9:961–1010. Am. Soc. of Agron., Inc. Madison, WI.
- Ojima, D., et al., 2013. Ecosystem sustainability through strategies of integrated carbon and land-use management. pp. 523–538. In: Brown, D.G. (Ed.), *Advances in Integrated Science*. Cambridge University Press, Management and Policy, pp. 564.
- Oliveira, B., et al., 2017. Unraveling the ecosystem functions in the Amazonia-Cerrado transition: evidence of hyperdynamic nutrient cycling. *Plant Ecol.* 218, 225–239. <http://dx.doi.org/10.1007/s11258-016-0681-y>.
- Patricio, J.H.P., 2014. How much soil organic carbon is there in agricultural lands? A case study of a prime agricultural province in Southern Philippines. *AES Bioflux* 6 (3), 194–208.
- Perrin, A.S., et al., 2014. Conversion of forest to agriculture in Amazonia with the chop-and-mulch method: does it improve the soil carbon stock? *Agric. Ecosyst. Environ.* 184, 101–114. <http://dx.doi.org/10.1016/j.agee.2013.11.009>.
- Persson, U.M., et al., 2014. A method for calculating a land-use change carbon footprint (LUC-CFP) for agricultural commodities – applications to Brazilian beef and soy, Indonesian palm oil. *Glob. Change Biol.* 20, 3482–3491. <http://dx.doi.org/10.1111/gcb.12635>.
- Petter, F.A., et al., 2017. Carbon stocks in oxisols under agriculture and forest in the southern Amazon of Brazil. *Geoderma Regional* 11, 53–61.
- Radambrasil, 1981. Levantamento de Recursos Naturais. Ministério das Minas e Energia 25, Folha SD-22/Goiás.
- Ren, Y.H., et al., 1999. A comparative study on litterfall dynamics in a seasonal rain forest and a rubber plantation in Xishuangbanna, SW China. *Acta Phytoecol. Sin.* 23 (5), 418–425.
- Saengruksawong, C., et al., 2012. Growths and carbon stocks of rubber plantations on Phonpisai Soil Series in Northeastern Thailand. *Rubber Thai J.* 1, 1–18.
- Schroth, G., et al., 2002. Conversion of secondary forest into agroforestry and monocrop plantations in Amazonia: consequences for biomass, litter and soil carbon stocks after 7 years. *Forest Ecol. Manage.* 163, 131–150. [http://dx.doi.org/10.1016/S0378-1127\(01\)00537-0](http://dx.doi.org/10.1016/S0378-1127(01)00537-0).
- Sisti, C.P.J., et al., 2004. Change in carbon and nitrogen stocks in soil under 13 years of conventional or zero tillage in southern Brazil. *Soil Till Res.* 76, 39–58. <http://dx.doi.org/10.1016/j.still.2003.08.007>.
- Soares-Filho, B.S., et al., 2006. Modelling conservation in the Amazon basin. *Nature* 440, 520–523. <http://dx.doi.org/10.1038/nature04389>.
- Strassburg, B.B.N., et al., 2010. Global congruence of carbon storage and biodiversity in terrestrial ecosystems. *Conserv. Lett.* 3, 98–105. <http://dx.doi.org/10.1111/j.1755-263X.2009.00092.x>.
- Suddick, E.C., et al., 2013. Monitoring soil carbon will prepare growers for a carbon trading system. *Calif Agric.* 67 (3), 162–171. <http://dx.doi.org/10.3733/ca.v067n03p162>.
- Sullivan, L.M., D'Agostino, R.B., 1996. Robustness and power of analysis of covariance applied to data distorted from normality by floor effects: homogeneous regression slopes. *Stat. Med.* 15 (5), 477–496. [10.1002/\(SICI\)1097-0258\(19960315\)15:5<477::AID-SIM217>3.0.CO;2-R](http://dx.doi.org/10.1002/(SICI)1097-0258(19960315)15:5<477::AID-SIM217>3.0.CO;2-R).
- Veldkamp, E., 1994. Organic carbon turnover in three tropical soils under pasture after deforestation. *Soil Sci. Soc. Am. J.* 58, 175–180. <http://dx.doi.org/10.2136/sssaj1994.03615995005800010025x>.
- Veloso, H.P. et al., 1991. Classificação da vegetação brasileira adaptada a um sistema universal. IBGE, p. 124.
- Vitousek, P.M., 1984. Litterfall, nutrient cycling, and nutrient limitation in tropical forests. *Ecology* 65, 285–298. <http://dx.doi.org/10.2307/1939481>.
- Vitousek, P.M., Sanford, R.L., 1986. Nutrient cycling in moist tropical forest. *Ann. Rev. Ecol. Syst.* 17, 137–167. <http://dx.doi.org/10.1146/annurev.es.17.110186.001033>.
- Wearn, O.R., et al., 2012. Extinction debt and windows of conservation opportunity in the Brazilian Amazon. *Science* 337 (6901), 228–232. <http://dx.doi.org/10.1126/science.1219013>.
- Wilson, K.B., et al., 2000. Factors controlling evaporation and energy partitioning beneath a deciduous forest over an annual cycle. *Agric. For. Meteorol.* 102, 83–103. [http://dx.doi.org/10.1016/S0168-1923\(00\)00124-6](http://dx.doi.org/10.1016/S0168-1923(00)00124-6).
- Yang, J.C., et al., 2004. Long-term impacts of land use change on dynamics of tropical soil carbon and nitrogen pools. *J. Environ. Sci.* 16 (2), 256–261.
- Zar, J.H., 2010. *Biostatistical Analysis*, fifth ed. Pearson Prentice-Hall, Upper Saddle River, pp. 944.
- Zeri, M., et al., 2013. Water use efficiency of perennial and annual bioenergy crops in central Illinois. *J. Geophys. Res. Biogeosci.* 118, 581–589. <http://dx.doi.org/10.1002/jgrg.20052>.