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Moderate swidden agriculture inside dense evergreen ombrophilous forests can sustain soil chemical properties over 10–15 year cycles within the Brazilian Atlantic Forest

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ABSTRACT

Swidden agriculture as a production system has been practiced for thousands of years, based on ecological processes of forest ecosystems. While this cultivation system has often been framed as contributing to environmental degradation, it can also be argued to be compatible with sustainable agricultural practices. Its sustainability strongly depends on the length of the recovery period, the period of secondary forest succession, after swidden agriculture. In this study we focused on soil fertility maintenance and recovery during secondary succession after swidden agriculture, compared to old-growth forests. Smallholders practiced a moderate form of swidden agriculture, abandoned the production field after one or two production cycles, and returned to the field after 10–15 years. Sampling took place in two traditional communities situated in the southeast portion of the Atlantic Rainforest at São Paulo state, Brazil. Soil samples were collected from forest fragments belonging to seven different *Caiçara* families (descendants from Amerindians, African Brazilian and European colonizers). Our chronosequence spanned a period of sixty years, while the old-growth forests were 100 years old. In all 28 sites were sampled, of which 19 were from ombrophilous forests on clay and 9 from *restinga* vegetation on sandy soil. Soils were analyzed for texture, macro- and micronutrient availability, bulk density, pH and nitrogen and carbon stocks. Linear regression, with clay content and fallow period as main factors, showed that successional age of the forest stand had only a significant effect on pH and Mn. Clay content of the soil influenced availability of nutrient bases and the CEC. Other soil properties, including stocks of carbon and nitrogen were not influenced by fallow period. Our data indicate that swidden agriculture might not lose key soil nutrients over time and hence can be considered an ecologically sustainable system in the region, challenging the negative perception of swidden agriculture.

1. Introduction

Swidden agriculture, or shifting cultivation, is a system in which parts of forest land are cultivated temporarily and allowed to revert after harvest to their natural vegetation (Altieri, 1991; Ribeiro Filho et al., 2013; Siahaya et al., 2016). After cultivation for a few years, smallholders usually move to the next piece of land due to the declining yields and / or increased labor investments to maintain yields. By moving to that next piece of land, farmers can achieve sustainable land use and maintenance of soil fertility (Peters and Neuenschwander, 1988). Managing fire and all vegetation stages that the swidden agricultural system involves, is highly contextual. Large differences in swidden agricultural practices have been reported between regions and countries, as well as between farming communities or even families (Condominas, 2009; Cramb et al., 2009). Therefore, temporal and spatial differences of impacts from swidden agriculture make it hard to

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generalize about the broader effect of swidden agriculture on the surrounding environments in which it is practiced. Yet, we can attempt to add to the discussion about the sustainability of such systems via case studies.

Three basic phases can be distinguished in swidden agricultural systems. These phases include: (a) conversion, (b) cultivation and (c) fallow, which together form the swidden cycle (Kleinman et al., 1995; Ribeiro Filho et al., 2013). Conversion of forest to agricultural land may result in an acceleration of nutrient flows out of the ecosystem (Hölscher et al., 1997; Obale-Ebanga et al., 2003; Béliveau et al., 2009;). The conversion phase is strongly related to soil dynamics (Peters and Neuenschwander, 1988). Depending on the fire technique used by the smallholder, nutrients are released from biomass, plant debris on the soil, soil aggregates and sometimes even from parent material. Some nutrients, like nitrogen (N) are partially lost through volatilization. Ashes are key for the increases in soil phosphorus (P), potassium (K), calcium (Ca) and magnesium (Mg) after burning (Farella et al., 2006; Aboim et al., 2008).

The length of the cultivation period generally ranges between one to three years and is always much shorter than the fallow period afterwards during which secondary succession takes place and soil fertility recovers (Kleinman et al., 1995). Often several shorter cycles of cultivation occur with shorter periods of secondary succession, then followed by a long period during which old forest can develop (Pedroso et al., 2009). The number of cultivation cycles before this long fallow determines the degree of negative impacts on soil fertility (Lessa et al., 1996; Hölscher et al., 1997; Farella et al., 2006). Too many cultivation – short fallow cycles, or too short recovery time after long fallow period, are thus likely to be unsustainable (Jakovac et al., 2015; 2016). For Peters and Neuenschwander (1988), sustainable regrowth of forests depends upon a combination of ecological, economic, socio-cultural and political factors.

The tropical world is quickly changing and population pressure and exploitation of its forests affects the practice of swidden cultivation, which threatens its sustainability (Peters and Neuenschwander, 1988; van Vliet et al., 2012; Heinimann et al., 2017). Institutions try to abolish swidden agriculture by promoting intensification towards permanent agriculture or by bringing forests under permanent protection by increasing reserves – placing vulnerable populations in an illegal situation. However, research has pointed out that swidden cultivation cannot be abolished as smallholders rely on these ancient techniques for their subsistence (Beckerman, 1983; Mertz, 2009; Lawrence et al., 2010).

In this study, we assessed the effects of swidden agriculture practiced by local communities (*Caiçaras*) from the Atlantic Forest in Southeast São Paulo state, Brazil. We aimed to analyze (chemical) soil fertility and a number of key nutrients, both available pools and total stocks, in areas of secondary succession in comparison to old (>100 years) preserved forest. By doing this, we may shed some light on the consequences of human action on these lands and to address the issues of sustainability in an area designated for forest conservation (in a protected area) and land sharing (as local inhabitants are allowed to continue swidden agriculture). We hypothesized that i) post-cultivated soils (+3 years) show recovery of soil nutrients, and ii) that the amount of available nutrients in the soil would reach a steady state with increasing fallow age.

2. Material and methods

2.1. Site description

Cananéia (São Paulo state, Brazil; 25° 00′ 52.99′S, 47° 55′ 36.01 W) is located in the Southeast portion of Atlantic Forest and is practically surrounded by protected nature reserves, ecological stations and state parks (e.g. *Lagamar de Cananéia* and *Ilha do Cardoso* state park, Fig. 1). The municipality is located about 360 km south from São Paulo. The native population includes many social groups, of which *Caiçara* is one of the biggest (Hanazaki et al., 2007, Alarcon et al., 2016;). *Caiçaras* are a mixed blood of descendants from Amerindians, African Brazilian and European colonizers (Hanazaki et al., 2007).

The climate in the area is subtropical humid with a wet and a dry

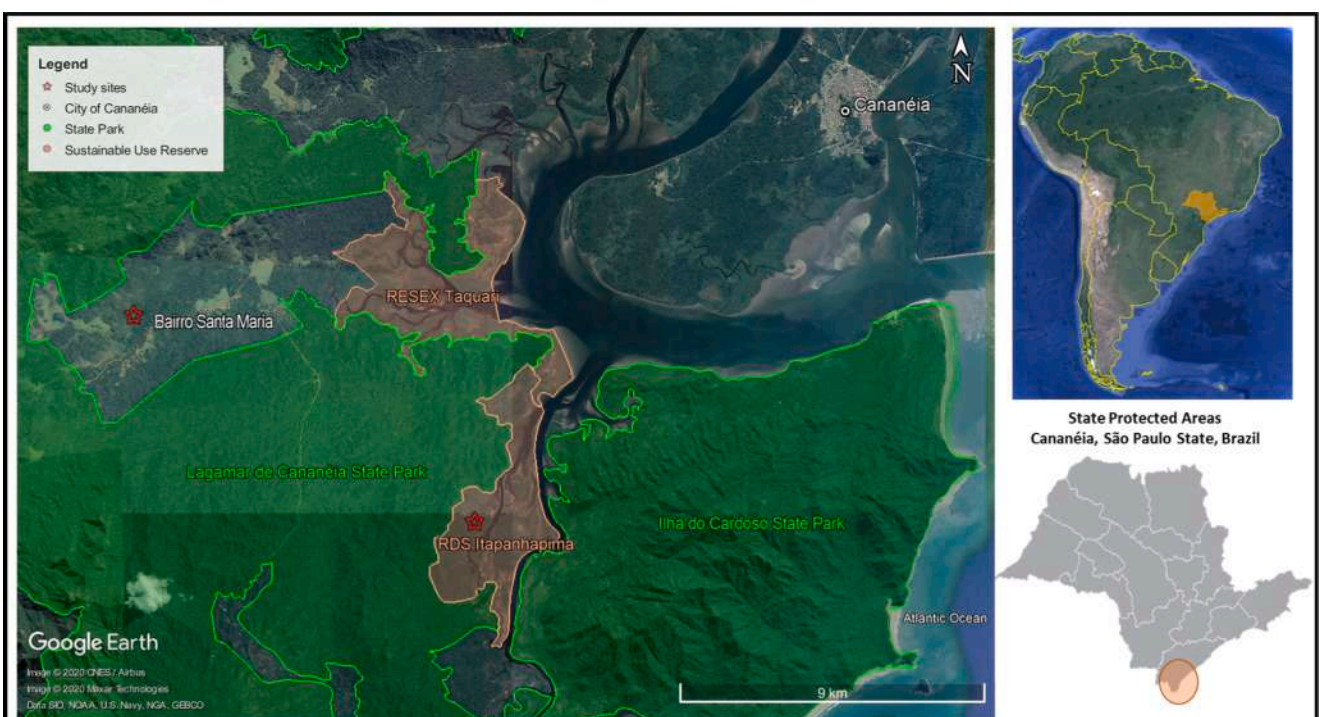

Fig. 1. Lay-out sampling area in Cananéia, Sao Paulo, Brazil. Sampling took place at the area of the red stars. The ecosystem surrounding Bairro Santa Maria is characterized by ombrophilous forest formations and around RDS Itapanhapima extractive reserve by restinga forest formations. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

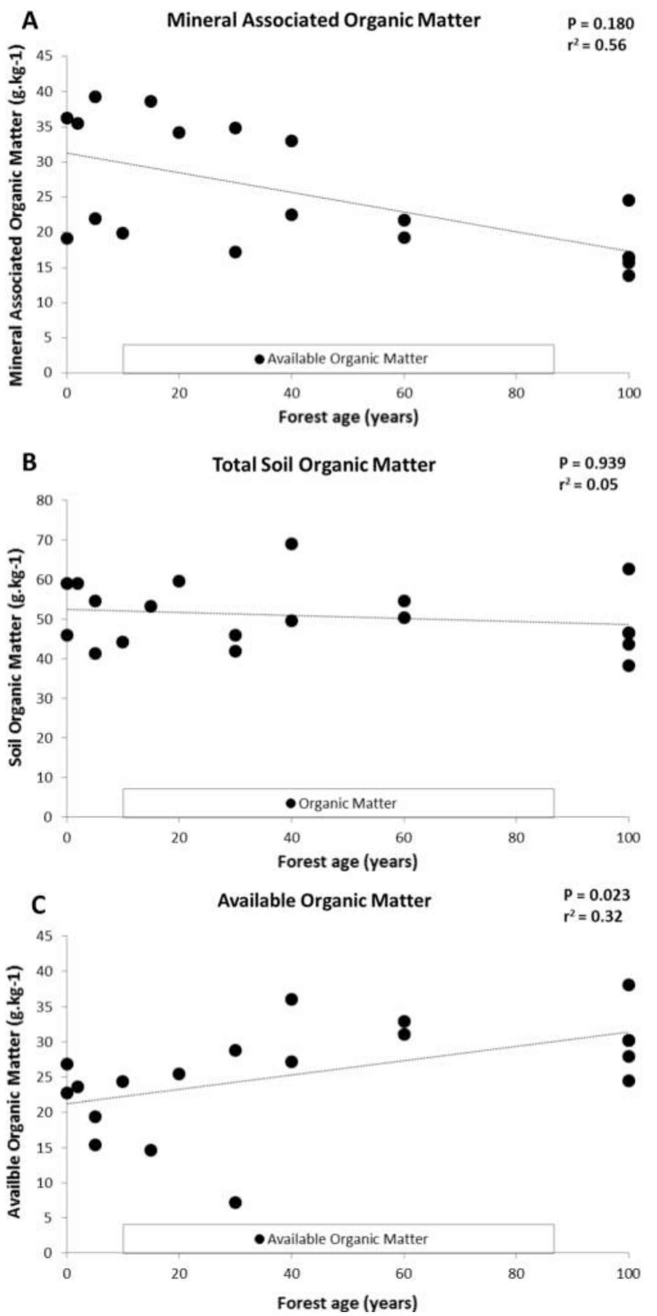

Fig. 2. Soil organic matter dynamics in the first soil layer (0–10 cm) from secondary ombrophilous forest plots of which fallows varied in age. Mineral associated organic matter rates (MAOM; A), total soil organic matter rates (SOM; B) and available organic matter rates (AOM; C) are being displayed. Linear regression showed significant interaction between succession age and the amount of available organic matter available. Sampling occurred in the period of December 2018 – April 2019 in the Atlantic Forest of southeast Brazil.

season (Alvares et al., 2013). The average temperature for the last 15 years is about 18 °C in winter and 27 °C in summer; average temperature throughout the year is 22 °C. Precipitation averages between 1700 and 1900 mm year^{-1} falling mainly between September and March (CIIAGRO, 2019). The main natural elements in the landscape of Cananéia are lagoons bounded by islands, coastal streams occupied by restinga vegetation and mangroves, as well as fluvial-marine plains. These natural elements are surrounded by coastal massifs that are cut by freshwater streams (Ab'Sáber, 2000; Oliveira, 2004). Most of the municipality's territory contains sedimentary environments formed by Entisols such as Fluvents, Aquents and Quartzipsamments. The areas derived from rocky formations contain Ultisols, Inceptisols and small portions of Oxisols (Oliveira, 2004; Rossi, 2017).

Samples from our study were taken in *Itapanhapima* Sustainable Development Reserve (RDS Itapanhapima) and *Bairro Santa Maria* community where swidden agriculture is still practiced by local farmers. The exact locations are shown in Fig. 1. All samples were collected at approximately the same altitude (<100 m). The main soil type of the land belonging to Bairro Santa Maria is Ultisols (USA taxonomy), the landscape of this zone is a plain between the sea and the rocky formation (Rossi, 2017). Bairro Santa Maria is consequently characterized by a lowland dense ombrophilous (evergreen) forest formation with soils that can contain more clay minerals. Samples taken from RDS Itapanhapima were mainly Fluvents soils (Entisols; USA taxonomy), which are more susceptible to changes due to floods. RDS Itapanhapima is characterized by high *Restinga* forests, a vegetation type characteristic for sandy, nutrient-poor soils along the Atlantic coast (Monteiro and Cesar, 1995; Guedes et al., 2006; Magnago et al., 2010; Lima et al., 2011).

Only a few families keep small agricultural plots, performing swidden agriculture of secondary vegetation, rotation of small plots, and semi-sedentary home gardens near the house (Hanazaki et al., 2007). These *Caiçaras* are part of the last groups to protect the ancient traditions of maintaining biodiversity and swidden agriculture in the region (Peroni and Martins, 2000). Main cultivars produced by these swidden agriculturalists are: rice (*Oryza sativa* L.), banana (*Musa paradisiaca* L.), sugarcane (*Saccharum officinaruman* L.) and other horticultural crops, like peach palm (*Bactris gasipaes* Kunth), cassava (*Manihot esculenta* Crantz), beans (*Phaseolus vulgaris* L.), maize (*Zea mays* L.), sweet potatoes (*Ipomoea batatas* (L.) Lam.) and yam (*Dioscorea* spp.; Peroni and Hanazaki, 2002). Smallholders usually cultivate for one or two production cycles and then abandon the plot. The swidden cycle is then repeated after about 10–15 years after they have observed the regrowth of certain woody species such as *Miconia cinnamomifolia* (DC.) Naudin, *Pera glabrata* (Schott) Baill., *Alchornea triplinervia* Müll. Arg. or *Myrsine coriacea* (Sw.) Roem. & Schult.

2.2. Sampling approach

Soil samples were collected from forest patches belonging to seven *Caiçara* families. Two families were from the *Itapanhapima* reserve and the other five families had land in *Bairro Santa Maria*. All families practiced swidden agriculture and owned preserved or secondary forest units, varying in fallow period. This time-length difference between the plots examined allowed for post-cultivation analyses with the establishment of a chronosequence (Abreu et al., 2009).

Chronosequences were established after talking to the smallholders who owned the plots, to get to know the field conditions (clearing strategies, fire management practices, age of fallow period, cropping management etc.). Later we conducted exploratory field visits with the smallholders, to explore the areas physically. Patches were selected if they were abandoned after the last cultivation cycle and not reused during the fallow period. Smallholders explained to us that none of the selected zones were cropped using fertilizers or pesticides. A total of 28 forest units were sampled for soil properties with fallow periods ranging from 0 (in use) to 60 years plus preserved areas with no record of use (>100 years). The age of 100 years was adopted based on information collected from the local population (elderly - age > 60 years), which indicated that these areas have not changed in the last 100 years - neither them, nor their parents had managed them. This minimum age was therefore used as a reference. Soil sampling occurred in rectangular shaped plots (30x10m) randomly distributed within each selected forest unit. In each forest unit three plots were established. The rectangular plots in each forest unit were constructed using a randomized approach. Soil sampling at RDS Itapanhapima took place in September 2018 and samples from the Santa Maria community were obtained in various field visits in the period of December 2018 - April 2019.

A standardized sampling method was used for collecting soil data (Soil Science Division Staff, 2017). The topmost layer of the soil is normally most fertile and simultaneously most affected by (anthropogenic) disturbances making its analysis essential to assess possible changes caused by swidden agriculture (Mareddy, 2018). A soil probe was used with a Dutch auger (mud auger) to take soil samples at a depth of 0–10 cm. Soil sampling to greater depths was desired, yet not possible due to financial and time constraints. We consider that sampling the uppermost 10 cm layer of the soil does not invalidate the findings of this study. Studies on soil C in tropical systems show more or less an equal number of studies where 0–10 and 0–30 cm layers have been sampled (Marín-Spiotta and Sharma, 2013). These authors argued that in most cases soil C follows a pattern of exponential decline, and therefore our pool estimates can be compared to other studies where larger sampling depths have been applied by using that scaling relation. Their conclusions were further supported in a recent review that showed no significant differences in fractional changes in soil C (and correlated nutrients) after deforestation and use as cropland when soil depths of 0–10 and 10–50 cm were compared (Veldkamp et al., 2020).

In each plot fifteen soil samples were collected using the soil probe. The litter and debris laying on the soil surface were removed before digging. Samples were placed in a bucket, mixed, and then a composite sample (~0.5 kg) was taken. For bulk density (BD) determination a metal ring (5.1 D x5.3H cm) was inserted into an undisturbed soil (0–10 cm depth) in the center of each subplot. BD samples were taken to the Forest Ecophysiology and Silviculture laboratory at *Escola Superior de Agricultura "Luiz de Queiroz"* (ESALQ; Piracicaba, Brazil) and oven-dried at 40 °C until constant weight. Dry soil weight (g) was divided by soil volume (cm^3) to calculate BD ($g.cm^{-3}$).

2.3. Soil parameters

The dry soils samples were sieved to 2 mm and analyzed at the Laboratory of Soil Fertility at the Center for Nuclear Energy in Agriculture (CENA; Piracicaba, Brazil). Soil analyses included soil texture (silt, clay and sand content), pH, soil organic matter (SOM), cation exchange capacity (CEC), macro- (P_{resin}, exchangeable K, S, Mg and Ca) and micronutrients (potential availability of Fe, Mn, Cu, Zn) and the metal Al, according to methods outlined by Raij et al. (2001). Mineral associated organic matter (MAOM) was estimated based on soil and clay content with a slightly modified regression for calculating MAOM in tropical soils (Six et al., 2002):

$$\text{MAOM } (g.\ kg^{-1}) = 2 * (0.92 + 0.30(\text{clay}\% + 0.50*\text{silt}\%))$$

Available organic matter (AOM) was then calculated by subtracting MAOM from the total amount of SOM.

Additionally, two clay-based chronosequences were established for total carbon and nitrogen analyses at the Soil Organic Matter laboratory at ESALQ (Piracicaba, Brazil). Samples for these analyses were selected based on their clay content, establishing chronosequences of high (±40%) or medium (±25%) clay contents. From each designated soil sample 5 g was ground to a particle size of 150 μm. These analyses followed the methods outlined by Nelson and Sommer (1982) in which soil carbon is determined by dry combustion. C and N stocks ($Mg.ha^{-1}$) were calculated by multiplying the content (%) of each element, the width of the soil layer (cm), and soil layer density ($g.cm^{-3}$).

2.4. Management and site information

To better understand swidden practices of the different families and the political ecology in the area, personal histories, land use history, and fire and agricultural management data were collected through direct observation and informal farmer interviews. Farmers were asked a common set of questions how they practice the swidden cycle (family activity planning, control of fire etc.), about the criteria they use to choose a suitable area to burn, how they till the land after the post-burning phase, and what crops they prefer to cultivate and why. They were also encouraged to expand on some topics when items of interest or questions arose. Informants remained anonymous to protect their integrity, yet we wanted to add information about the complex social-political and ecological constraints these smallholder farmers are frequently dealing with.

2.5. Statistical analyses

Data exploration was executed using box plots to visualize general patterns and possible outliers. Possible outliers were identified as data points located outside the 1st and 4th quartile. However, we decided to keep all outliers (2.7% of the data) in the analyses because after double checking we concluded that these data unlikely presented some sort of error. Natural log or square root transformations were applied as needed to meet assumptions of homoscedasticity and normality. All analyses were carried out using JMP pro 14.0.1 software package (SAS Institute, 2018).

Two data sets were created. Data set A comprised soil properties taken in the ombrophilous forest patches ($n = 53$). The results for data set A, were averaged out per sampled forest site to avoid pseudoreplication (data set A; $n = 19$). Pearson's correlation test was carried out and these data showed (see Results) that most soil properties were (highly) correlated with clay content. We decided for this reason to use multiple linear regressions, for testing for significant sources of variation as a function of fallow period for data set A. Multiple linear regressions were carried out to test whether fallow period (t) might have influenced the different soil properties, meanwhile avoiding noise caused by variation in clay content (c) of the soil samples. We also tested for the fallow period by clay content interaction. Hence, for data set A we studied the probability of a statistical difference in the average values between the different fallow ages.

Soil carbon and nitrogen stocks were calculated for two chronosequences under ombrophilous forest formations, with a medium (±25%) or high (±40%) clay content. For these data a separate data set (B) was created ($n = 12$). For data set B likewise, multiple linear regressions were carried out with fallow period and clay content as explanatory factors. Likewise, in data set B we considered the probability of a statistical difference in the values between the different fallow ages.

Additionally, with data set A, a principal component analysis (PCA) was performed to understand if clay content of the soil or fallow period played a more significant role on the variance of the nutrient availability. In the PCA, indicators of nutrient availability that are correlated cluster together, demonstrating their correlation either with clay content or fallow period.

3. Results

3.1. Relationships between soil properties in dense evergreen ombrophilous forest areas

Pearson correlation coefficients were calculated to assess relationships between soil chemical properties across 19 ombrophilous forest plots varying in fallow period (Fig. 4). The clay fraction was positive and highly significantly correlated with CEC ($r = 0.583$, $P = 0.009$), Mg ($r = 0.831$, $P < 0.001$), Ca ($r = 0.749$, $P < 0.001$) and micronutrients such as Zn ($r = 0.586$, $P = 0.008$), Cu ($r = 0.704$, $P = 0.001$) and Mn ($r = 0.839$, $P < 0.001$). CEC values are generally higher when soils contain higher clay contents (Fig. 3). As it is implausible that either swidden agriculture or secondary succession modifies soil texture, we consider variation in clay content (and hence variation in macro and micronutrient content) as caused by spatial variability. We used clay as covariate in our analyses to prevent this spatial variability to dominate over any indication of secondary-succession impacts on nutrients. After this correction, pH was

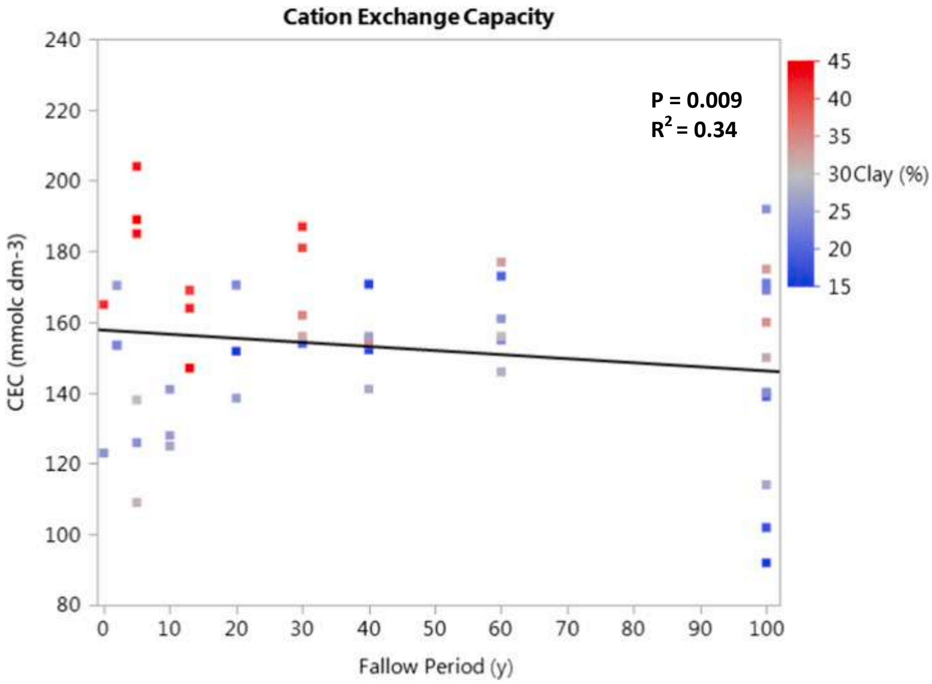

Fig. 3. Cation exchange capacity (CEC) values from the first soil layer (0 – 10 cm) taken from secondary ombrophilous forest plots varying in age. Clay content in these forest patches varied and is account for by using a color scale, blue marks represent lower clay contents and red marks represent higher clay contents. The *P*-value and R^2 are given for the significant CEC by clay correlation. Sampling occurred in the period of December 2018 – April 2019 in the Atlantic Forest of southeast Brazil. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

positively correlated with K (r = 0.786, $P < 0.001$) and P (r = 0.599, $P = 0.007$) and was negatively correlated with Fe (r = -0.702, $P = 0.008$). Al was positively correlated with P (r = 0.629, $P = 0.004$), while SOM displayed a positive correlation with P (r = 0.812, $P < 0.001$), S (r = 0.695, $P = 0.001$) and Al (r = 0.629$P = 0.004$). The importance of clay for the provision of key nutrients was not observed under Restinga formations.

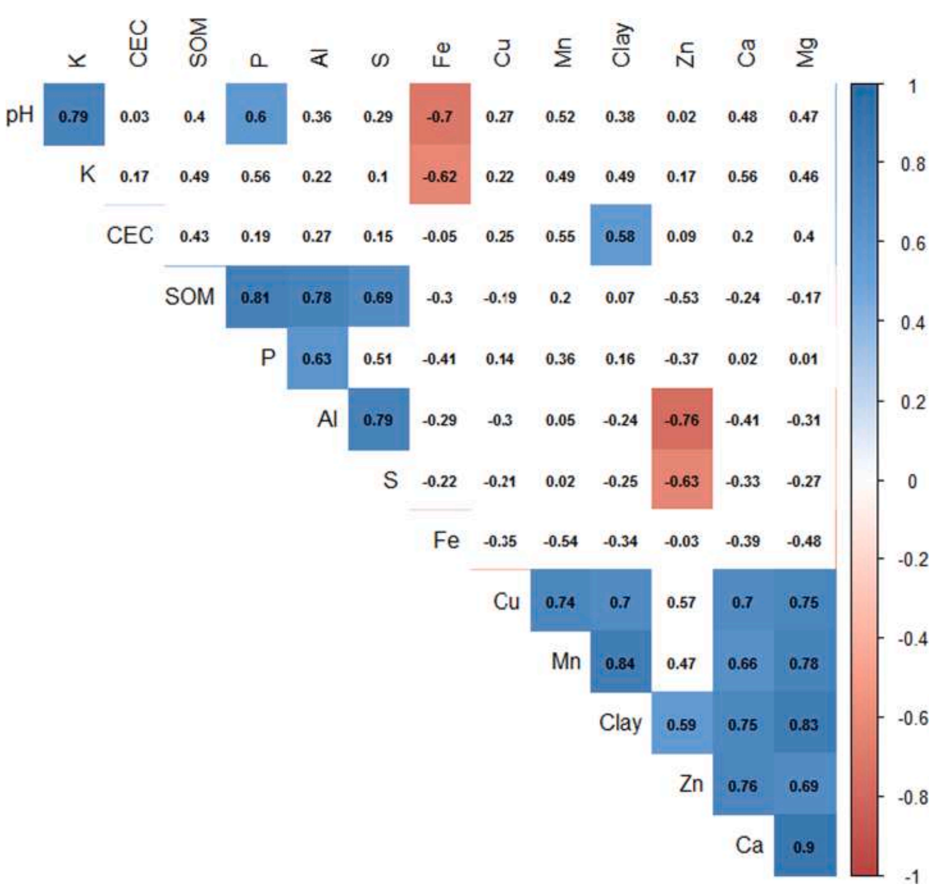

Fig. 4. Pearson correlation test of soil chemical properties until 10 cm depth (n = 19) under ombrophilous forest formations in the Atlantic Forest of southeast Brazil. Blue and red circles represent significant positive or negative correlation between properties in the soil. Blank grids are left out as these correlations were insignificant. Variables included are: cation exchange capacity (CEC), soil organic matter (SOM), pH, Resina-P (P), exchangeable K, Ca, Mg, S, potential available Cu, Fe, Mn, Zn and clay content. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

3.2. Effect of fallow period on nutrient provision of post swidden soils under ombrophilous vegetation

Fallow period had overall little significant effects on the availability of nutrients in both forest formations examined (Table 1). The samples of the first soil layer (0–10 cm) under ombrophilous vegetation were high in clay content, which ranged from 17 − 44%. These soils had an average bulk density of 0.82 g.cm^{-3}. Overall, SOM was high (50 g.kg^{-1}), SOC around 30 g.kg^{-1}, with no effect of fallow age (Table 1). Mineral-associated organic matter (MAOM) was on average 25 g.kg^{-1} and hence about 50% of total SOM was available organic matter (AOM; Fig. 2C). MAOM did not change over the chronosequences, yet AOM was significantly affected by fallow age ($P = 0.023$). The amount of AOM increased when the secondary-forest plots were more aged.

There was one significant interaction for Mn ($P = 0.016$) between clay and fallow period, in which Mn was higher with higher clay content in younger fallows. In ombrophilous forests fallow period had a significant effect on pH ($P = 0.047$). pH was on average 3.7 and was slightly higher in earlier successional stages than in older forest stands (Table 1). Other important soil nutrients like P ($P_{resin;}$ avg. 8.6 mg.kg^{-1}), K (avg. 1.25 mmol$_c$ dm^{-3}), Ca (avg. 3.5 mmol$_c$ dm^{-3}) and Mg (avg. 3.8 mmol$_c$ dm^{-3}) did not change over the chronosequences. Exchangeable Al was not affected by fallow age and was overall high in all plots, on average 18.4 mmol$_c$.dm^{-3} (Table 1).

3.3. Are nutrient stocks mostly influenced by clay content or fallow period within ombrophilous forest areas?

Soil C and N stocks were calculated for two chronosequences, under ombrophilous secondary forest formations, with a medium (±25%) or high (±40%) clay content (Table 2). In the uppermost 10 cm, C-stocks were on average 25.7 Mg.ha^{-1} while N-stocks were 2.0 (Mg.ha^{-1}). C:N ratios were on average 11.5. Linear regression did not show a significant effect of fallow period on the C and N stocks ($P > 0.05$).

The chemical soil properties together with clay content and fallow age, were considered in a principal component analysis (PCA) and grouped in components. Each principal component (PC) was interpreted according to the magnitude of their eigenvalues (Fig. 5). The first principal component (PC1) explained 38.7% of the variance in the data and was more associated with clay (0.86) compared to fallow period (-0.70; Fig. 5). PC1 is mainly associated with cations such as Mn (0.89), Mg (0.86) and Ca (0.82). Clay content mainly clusters in the biplot with indicators such as CEC, Zn, Cu, pH, K, Ca, Mg and Mn. These former indicators also seem to be lower over time as they mainly associate with early stage samples compared to the intermediate and late-succession samples. Fallow period on the other hand clusters mainly together with Fe. The second principal component (PC2) explained 30.5% of the variance and was mainly associated with Al (0.94), SOM (MAOM; 0.88) and S (0.82).

4. Discussion

4.1. Fallow period had limited impacts on nutrient provision

Our main finding was that fallow period had little influence on the availability of nutrients in ombrophilous forest formations. This contradicts our (first) hypothesis. It is also different from earlier literature reports on the effect of secondary succession on soil properties (e.g from processes described by Powers and Marín-Spiotta 2017). One possible explanation for lack of (strong) effects of the swidden cycle on dynamic

Table 1
Mean values for soil chemical properties of chronosequences sampled, at a depth of 0–10 cm, in the ombrophilous forest formations between December 2018 and April 2019 in the Bairro Santa Maria, Cananéia municipality of the Atlantic Forest in Southeast Brazil. Numbers in bold are mean values, numbers between parentheses are standard errors, linear regression (n = 16) *P*-values are given for the impact of fallow age (t) and the clay content (c) by fallow age interaction (t × c). *P*-values highlighted are significant interactions ($P < 0.05$).

			Ombrophilous forest Fallow period (years)								*P*-value	
Soil properties	Units	Adequate range*	5	10	15	20	30	40	60	Preserved forest (100)	Time (t)	t × c
pH		> 5	**3.9** (0.04)	**3.7** (0.00)	**3.9** (0.00)	**4.0** (0.07)	**3.7** (0.08)	**3.8** (0.05)	**3.7** (0.07)	**3.6** (0.04)	**0.047**	0.770
SOM	g kg^{-1}	>20	**48.0** (3.54)	**44.3** (1.67)	**53.3** (3.18)	**59.7** (6.12)	**44.0** (1.61)	**59.3** (5.24)	**52.5** (2.64)	**48.1** (2.33)	0.939	0.587
$_{resina}$P	mg kg^{-1}	> 13	**7.3** (1.26)	**8.3** (0.67)	**9.7** (1.45)	**11.7** (1.20)	**6.8** (0.60)	**10.8** (2.41)	**7.0** (0.77)	**8.2** (0.90)	0.279	0.618
K	mmol$_c$ dm^{-3}	> 1.6	**1.9** (0.12)	**0.8** (0.12)	**1.6** (0.24)	**1.5** (0.00)	**0.8** (0.13)	**1.4** (0.19)	**1.3** (0.17)	**0.9** (0.09)	0.123	0.945
Ca	mmol$_c$ dm^{-3}	>4	**9.5** (0.50)	**2.0** (0.58)	**5.7** (0.67)	**1.0** (0.00)	**5.5** (2.08)	**1.7** (0.21)	**2.8** (0.31)	**2.4** (0.39)	0.379	0.357
Mg	mmol$_c$ dm^{-3}	>5	**7.8** (0.65)	**2.7** (0.33)	**7.7** (1.20)	**1.3** (0.33)	**6.3** (1.91)	**1.8** (0.31)	**4.2** (0.70)	**2.5** (0.38)	0.931	0.807
Al	mmol$_c$ dm^{-3}	<5	**13.0** (2.18)	**12.3** (0.67)	**13.7** (0.67)	**39.0** (2.08)	**15.0** (0.89)	**25.0** (5.29)	**14.5** (1.12)	**15.5** (2.29)	0.719	0.556
CEC	mmol$_c$ dm^{-3}	–	**158.5** (15.95)	**131.3** (4.91)	**160.0** (6.66)	**153.6** (9.28)	**165.7** (5.97)	**154.8** (3.87)	**161.3** (4.78)	**142.5** (7.32)	0.061	0.061
S	mg kg^{-1}	>5	**7.2** (0.70)	**4.3** (0.33)	**5.0** (0.000	**10.0** (0.58)	**7.2** (1.56)	**9.8** (1.30	**5.5** (0.50)	**7.5** (1.11)	0.665	0.852
Cu	mg kg^{-1}	> 0.3	**0.9** (0.19)	**0.2** (0.02)	**1.1** (0.34)	**0.2** (0.00)	**1.1** (0.46)	**0.2** (0.00)	**0.3** (0.05)	**0.5** (0.12)	0.703	0.074
Fe	mg kg^{-1}	>5	**103.9** (7.80)	**194.1** (5.09)	**94.4** (11.24)	**101.7** (11.20)	**188.1** (37.86)	**162.7** (18.70)	**143.2** (31.45)	**189.90** (21.66)	0.105	0.519
Zn	mg kg^{-1}	>0.6	**2.6** (0.22)	**1.5** (0.13)	**2.0** (0.21)	**0.8** (0.06)	**1.6** (0.16)	**1.2** (0.19)	**1.6** (0.16)	**1.6** (0.13)	0.225	0.073
Mn	mg kg^{-1}	>1.3	**25.2** (8.35)	**1.4** (0.24)	**34.8** (1.67)	**13.3** (3.55)	**13.9** (5.90)	**4.8** (1.78)	**2.3** (0.43)	**1.9** (0.49)	**0.049**	**0.016**
BD	g.cm^{-3}	<1.6	**0.6** (0.06)	**0.8** (0.05)	**1.1** (0.04)	**0.6** (0.01)	**0.7** (0.03)	**0.6** (0.07)	**0.8** (0.07)	**1.0** (0.08)	0.898	0.079

* Adequate ranges of soil nutrients to support production in conventional agriculture (Raij et al. 1996).

Table 2
Soil nutrient stocks (Mg.ha-1) and ratios of two chronosequences high (±40%) and medium (±25%) in clay content on the first soil layer (0–10 cm). Soil samples were collected in the period of December 2018 – April 2019 under ombrophilous forest formations in the Atlantic Forest of southeast Brazil. linear regression *P*-values are given for the impact of fallow age (*t*) on the different nutrient stocks.

	fallow period years	Texture* %	N-stock $Mg.ha^{-1}$	C-stock	C:N ratio –
High clay content	0	(42/37/21)	2.84	23.67	8.3
	5	(43/40/16)	1.69	17.02	10.1
	10	(40/41/18)	4.23	41.16	9.7
	30	(42/38/20)	1.15	16.55	14.4
	40	(35/12/52)	2.09	30.15	14.4
	preserved forest (100)	(34/18/48)	2.02	17.65	8.7
Medium clay content	5	(25/18/57)	1.37	13.12	9.6
	10	(26/26/58)	1.60	19.55	12.2
	30	(23/66/70)	1.88	18.45	9.8
	40	(26/13/60)	1.39	17.65	12.7
	60	(23/16/60)	1.58	20.09	12.8
	preserved forest (100)	(24/9/67)	2.49	24.87	10.0
P-values (t)		–	*0.827*	*0.885*	*0.964*

* (clay/silt/sand)

soil properties is the large textural variation between different patches. As some soil properties covaried with clay content (Figs. 4 and 5), it is possible that variability overrode successional patterns. However, by applying linear regression with clay content as additional explanatory factor we tried to correct for that variation. Still there were only minor effects in ombrophilous forests and no effects in restinga. We will therefore continue to focus our discussion on ombrophilous forests.

Our findings showed that Mn was affected by fallow period. This is not an uncommon finding, as discussed by Ribeiro Filho et al. (2013), who reviewed the impact of swidden agriculture on tropical soils based on 19 studies and showed that the increase in pH in young fallows reduced Mn availability. Fallow age also influenced pH after the burning

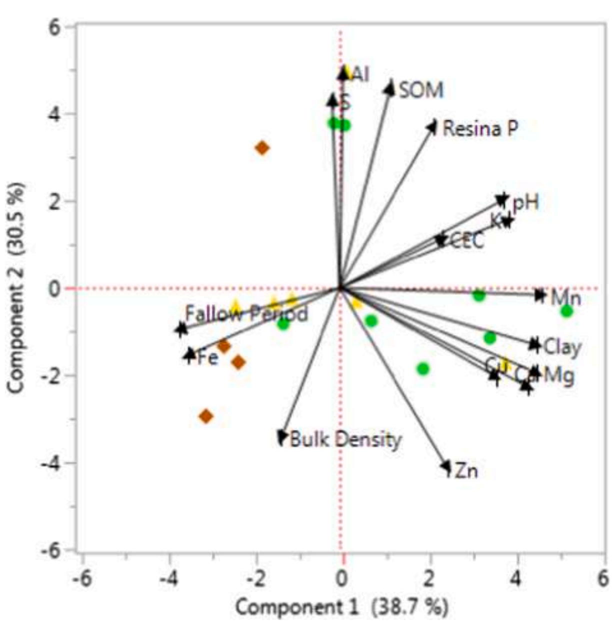

Fig. 5. Bi-plot showing how fallow period and clay content relate to different soil chemical properties. Samples were taken at a depth of 10 cm from ombrophilous forest plots varying in fallow period in the Atlantic Forest of southeast Brazil. Principal components 1 and 2 explain 38.7% and 30.5% of total variation, respectively. See table 1 for additional description of abbreviations. *Green circles* are samples of early succession stages (<20 years), *yellow triangles* are samples from moderate succession stages (20–60 years) and *brown squares* are samples from late successional stages (>60 years). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

phase, however the change in pH was minor when comparing fallow of different ages, with only slightly higher pH values for younger compared to older fallows. Higher pH in younger stands might be explained by fire that produces carbonate-containing ash, during combustion. However, the pH effect is often short-lasting (Peters and Neuenschwander, 1988). Our soil remained rather acidic (pH < 4) immediately even after combustion took place, suggesting low-intensity fires with limited impact on soil pH and on most nutrients whose availability is pH-dependent. The rise in pH had a marginally positive influence on the availability of P as also reported before (Giardina et al., 2000; Sommer et al., 2004). Lower pH causes more P-sorption to metal (hydr-)oxides and clay edges (Smeck, 1985). Lower pH can also increase exchangeable Al, however, considering the low pH, irrespective of successional age, Al levels were high (Ma et al., 2001). Lessa et al. (1996), Hölscher et al. (1997) and Farella et al. (2006) also reported this effect. Our results are also similar to what was described in the *meta*-analyses of Ribeiro Filho et al. (2015) on soil chemistry dynamics after swidden agriculture. Their analysis specified that P or K may be more sensitive to changes in pH after combustion (Sparks et al., 1996; McKenzie, 2003).

Total SOM levels were overall high (avg. 50 $g.kg^{-1}$) in the ombrophilous ecosystem irrespective of fallow age (Table 1). These high levels may seem surprising as it is often considered textbook knowledge that tropical soils are low in SOM (around 20 $g.kg^{-1}$). It is likely that many of such reports come from intensively used systems where the amounts of AOM have been reduced, as most SOM is mineral-associated. Our amounts of MAOM (25 $g.kg^{-1}$) are more comparable with these reports of SOM levels of 20 $g.kg^{-1}$. After estimating the MAOM, we calculated that about 50% of the OM is available and hence subject to fairly rapid turnover which then would release nutrients for plant growth. This is not an uncommon finding, as generally MAOM represents the vast majority of SOM in agricultural soils (Cotrufo et al., 2013, 2015; Lehmann and Kleber, 2015). Due to low pH and abundance of iron oxides, the soil is strongly positively charged and will therefore strongly adsorb and protect SOM. We therefore expect that this forest ecosystem maintains its soil organic matter content, after combustion, for a combination of different reasons.

Firstly, the study area is part of the Atlantic Forest, an old biome where trees and understory species may be highly adapted to local growth conditions (Morellato and Haddad, 2000; da Silva and Casteleti, 2003; George et al., 2012). High availability of aluminum is usually toxic for plants and inhibits microbial growth (Wright, 1989), yet the studied forest ecosystem supports biomass growth and high total SOM levels (Table 1). We expect that part of the extractable aluminum was not available in the toxic ionic form in the soil solution but was adsorbed to dissolved organic matter and hence plant-unavailable in the first 10 cm. Support for this explanation is provided by the strong positive correlation between the SOM and Al content in the soil. That would imply that perhaps Al toxicity mainly affects deeper soil layers where SOM contents are lower, therefore soil samples of deeper soil layers are needed in future investigations.

Secondly, we suppose that the forest soil draws its high fertility from nutrient cycling (Foster and Bhatti, 2006). Young secondary forests quickly recover litter production at rates equal to that of primary forests, especially in the tropics (Ewel, 1976; Barlow et al., 2007). Litter adds nutrients to the soil solution and can thus function as a fertilizer (Loranger-Merciris et al., 2007, Visscher et al., 2020). In tropical zones, decomposition rates are usually high because of favorable conditions for soil microbes, like high temperatures and humidity during most of the year (Berg, 2014) resulting in low amounts of AOM. By taking this and the previous argument into consideration on adaptions of species to local condition, we hypothesized, like Appanna et al. (1995) that the soil microbial community in this forest ecosystem is highly adjusted to acidic soil conditions. Due to their adaptation microbes may continue contributing substantially to the decomposition rates and corresponding SOM contents and stabilization of it (Chander et al., 1998, Kallenbach et al., 2015). However, more research is needed to fully understand this effect.

Finally, by removing the vegetative cover the bare soil surface would usually be directly exposed to precipitation and radiation (Kendawang et al., 2004; Hattori et al., 2005), it may therefore increase the soil temperature, which could potentially increase organic matter decay (Mendoza-Vega and Messing, 2005; Okore et al., 2007). We think that due to the humid conditions and the moderate burning practices of the smallholders in this area, substantial soil heating and concomitant SOM losses were prevented. Fire-induced soil hydrophobicity is commonly observed after moderate forest burning (Huffman et al., 2001). Specifically for acidic soils, hydrophobicity is known to increase the structural stability of soil aggregates and pore size (Mataix-Solera and Doerr, 2004). Since SOM is generally strongly correlated with aggregate stability, fire-induced soil hydrophobicity could help preserve SOM rates after moderate burning events as observed in our study area (Chenu et al. 2000).

In the interviews the smallholders explained to us that they usually only burn once after slashing, and then leave the unburned residues on the cropping fields during the subsequent cultivation phase. Even with the selective logging of larger trees before burning, they reported that most of the felled vegetation remains in the area and that some trunks are often only partially burned. That practice contrasts with other swidden systems where the incompletely burned larger woody debris is subsequently piled up for a second, higher-temperature burn. This lack of a second burn was also visible during our field visits where we discovered still trees from the felling phase. This type of observation was also reported by Peters and Neuenschwander (1988) as a contributing factor to the sustainability of swidden systems. In their book they study various outcomes and its causes after the combustion of vegetation, within the swidden cycle in tropical forests.

In our study clay played an important role in the availability of Mg, Ca, Zn, Cu, and the CEC, regardless of fallow period in the sampled plots under ombrophilous formations. The importance of clay for the nutrient provision was under Restinga formation not observed. We suppose that this might be due to the sandy character of the Restinga soil. Sandy soils show generally less fertility compared to soils with a higher clay content (Brady and Weil, 2008).

We calculated Carbon and Nitrogen stocks for two chronosequences, from ombrophilous formations, with medium or high clay content, in an attempt to account for the impact clay content has on nutrient provision. We conclude that again fallow age did not impact C and N stocks, hence swidden practices in our study may not directly result in soil degradation. This is an important finding to keep in mind as nutrient stocks are often negatively impacted as a consequence of a more conventional agriculture (often associated with intense plowing and pesticide use; Reganold and Wachter, 2016). Swidden agriculture, as performed in this study area, might therefore not be as bad for the environment as often negatively implied in related literature.

Our results on nutrient dynamics, especially the trends observed under ombrophilous formations, agree with earlier studies (Ribeiro Filho et al., 2013; 2018). The conversion of forest biomass releases rapid nutrients into the soil solution, a common positive effect of swidden agriculture (Kleinman et al., 1995). We however could not statistically demonstrate with our analyses that the availability of nutrients in the soil lowers to steady state over time in post-cultivated soils, as we hypothesized. This result should be interpreted with caution as we might have not been sufficiently able to demonstrate increases in the soil solution directly after combustion as our focus was on post-cultivated soils. Despite the latter, the lack of impacts of the swidden practice on soil nutrient dynamics is noteworthy and shows that swidden agriculture, when studied in combination with the local practice, is not as negative as often claimed.

4.2. *The threat of 'conservation' politics on the swidden livelihood in Cananéia*

Along with local drivers (inherent; clay content or manageable; fallow period) that impact swidden agriculture, there are also external drivers such as policies that influence the smallholders in their agricultural practices. The municipality of Cananéia is recognized as national cultural heritage because it is one of the oldest settlements of Brazil. The region contains also a large number of protected areas, making it an important territory for biodiversity conservation in the Atlantic Forest (Dias and Oliveira, 2015). This recognition restricts real estate and agricultural expansion into natural areas but is also imperiling local communities' livelihoods and limits their access to zones in the forest. With that, rural communities are getting more isolated (Hanazaki et al., 2007).

Preservation of biodiversity in local territories is largely prioritized, with some areas (e.g. Sustainable Development Reserves) designated for continued anthropic use (e.g. agriculture by local inhabitants; Dias and Oliveira, 2015). This hinders *Caiçara* smallholders in the countryside of Cananéia to continue practicing swidden agriculture as they did for centuries (Jung et al., 2017). The protected status of the area, which makes sustainable swidden agriculture difficult to maintain (e.g. by reduction of fields available for rotation), paradoxically, contributes to fast land degradation. This process likely happens globally but is a fact that many policymakers tend to overlook (Shiferaw et al., 2009). Next to land degradation the process likely also accelerates rural impoverishment (Lawrence et al., 2010; van Vliet et al., 2012).

Conservation policies should be in-line with the local context in order to have the desired outcome (Jung et al., 2017). Otherwise, we risk blaming innocent smallholders for forest degradation, which they, in fact, frequently do not commit. Community farmers are in this specific case just trying to adapt themselves to the new established political ecology (van Vliet et al., 2012). Under restrictive laws and with agricultural products being more accessible in local markets, locals have turned to other economic alternatives abandoning traditional agriculture. In this way the region is suffering a great cultural loss increasing the disconnection with the rest of the nature.

5. Conclusions

Fallow period only marginally influenced chemical properties such as pH and Mn, or did have no effect at all on other nutrients, contrary to our hypothesis. After closer investigation we found that time-length difference of secondary succession showed no depletion of the soil nutrients, showing no decrease due to human activities. The impact of swidden agriculture seems therefore limited, suggesting that the system that we observed to date is still sustainable. We further consider chronosequences a valuable method in the analyses of soil properties.

However, chronosequences are sensitive to misinterpretation due to spatial differences (such as clay content) in sampling plots. The study site exhibits large spatial variability in texture, which makes demonstrating effects on soils nutrient dynamics more difficult.

External drivers, such as conservation and biodiversity policies, also influence the practice of swidden agriculture in smallholder communities in the countryside of Cananéia. As a consequence, *Caiçaras* often abandon this form of agriculture and enter into more intensive and permanent agricultural practices or illegal forest exploitation. This tendency is likely not only restricted to southeast Brazil, suggesting we can learn from these social-political conditions globally to sustainably improve smallholder livelihoods. With this case study we want to contribute to changing the negative paradigm of swidden agriculture, by studying smallholder practices in their environment in the pursuit of better understanding and adjusting old technologies such as the fallow duration and cropping cycles.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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References

Aboim, M.C., Coutinho, H.L., Peixoto, R.S., Barbosa, J.C., Rosado, A.S., 2008. App. Soil Ecol. 38, 100–108. https://doi.org/10.1016/j.apsoil.2007.09.004.

Abreu, Z., Llambí, L.D., Sarmiento, L., 2009. Restor. Ecol. 17, 619–627. https://doi.org/10.1111/j.1526-100X.2008.00406.x.

Ab'Sáber, A. N., 2000. Fundamentos da geomorfologia costeira do Brasil inter e subtropical. Rev. Bras. Geomorf. 1, 27-43. https://doi.org/10.20502/rbg.v1i1.67.

Alarcon, G.G., Fantini, A.C., Salvador, C.H., 2016. Ambient. Soc. 19, 87–112. https://doi.org/10.1590/1809-4422ASOC136361V1932016.

Altieri, M.A., 1991. Outlook Agr. 20, 15–23. https://doi.org/10.1177/003072709102000105.

Alvares, C.A., Stape, J.L., Sentelhas, P.C., de Moraes Gonçalves, J.L., Sparovek, G., 2013. Köppen's climate classification map for Brazil. Meteorol. Z. 22, 711–728. https://doi.org/10.1127/0941-2948/2013/0507.

Appanna, V.D., Huang, J., Prusak-Sochaczewski, E., Pierre, M.S., 1995. Microbial adaptation to aluminum. Biotechnol. Prog. 11, 159–163. https://doi.org/10.1021/bp00032a007.

Barlow, J., Gardner, T.A., Ferreira, L.V., Peres, C.A., 2007. For. Ecol. Manag. 247, 91–97. https://doi.org/10.1016/j.foreco.2007.04.017.

Beckerman, S., 1983. Does the swidden ape the jungle? Hum. Ecol. 11, 1–12. https://doi.org/10.1007/BF00891227.

Béliveau, A., Lucotte, M., Davidson, R., do Canto Lopes, L. O., Paquet, S., 2009. Early Hg mobility in cultivated tropical soils one year after slash-and-burn of the primary forest, in the Brazilian Amazon. Sci. Total Environ. 407, 4480-4489. https://doi.org/10.1016/j.scitotenv.2009.04.012.

Berg, B., 2014. Soil Biol. Biochem. 78, 222–232. https://doi.org/10.1016/j.soilbio.2014.08.005.

Brady, N.C., Weil, R.R., 2008. The Nature and Properties of Soils. Fourteenth, Prentice Hall/Pearson Education.

Chander, K., Goyal, S., Nandal, D.P., Kapoor, K.K., 1998. Biol. Fertil. Soils 27, 168–172. https://doi.org/10.1007/s003740050416.

Chenu, C., Le Bissonnais, Y., Arrouays, D., 2000. Soil Sci. Soc. Am. J. 64, 1479–1486. https://doi.org/10.2136/sssaj2000.6441479x.

CIIAGRO, 2019. Portal Agrometeorológico e Hidrológico do Estado de São Paulo. Centro integrado de informações agrometeorológicas, Instituto Agronômico. http://www.ciiagro.org.br (August 6, 2020).

Cotrufo, M.F., Wallenstein, M.D., Boot, C.M., Denef, K., Paul, E., 2013. Glob. Chang. Biol. 19, 988–995. https://doi.org/10.1111/gcb.12113.

Cotrufo, M.F., Soong, J.L., Horton, A.J., Campbell, E.E., Haddix, M.L., Wall, D.H., Parton, W.J., 2015. Nat. Geosci. 8, 776–779. https://doi.org/10.1038/ngeo2520.

Condominas, G., 2009. Hum. Ecol. 37, 265–267. https://doi.org/10.1007/s10745-009-9248-z.

Cramb, R.A., Colfer, C.J.P., Dressler, W., Laungaramsri, P., Le, Q.T., Mulyoutami, E., Peluso, N.L., Wadley, R.L., 2009. Hum. Ecol. 37, 323–346. https://doi.org/10.1007/s10745-009-9241-6.

da Silva, J.M.C., Casteleti, C.H.M., 2003. Status of the biodiversity of the Atlantic Forest of Brazil. CABS and Island Press, Washington, pp. 43–59.

Dias, R.L., Oliveira, R.C.D., 2015. Soc. Nat. 27, 111–123. https://doi.org/10.1590/1982-451320150108.

Ewel, J.J., 1976. J. Ecol. 293–308 https://doi.org/10.2307/2258696.

Farella, N., Lucotte, M., Davidson, R., Daigle, S., 2006. Sci. Total Environ. 368, 19–29. https://doi.org/10.1016/j.scitotenv.2006.04.025.

Foster, N.W., Bhatti, J.S., 2006. Forest ecosystems: nutrient cycling. Encyclopedia of soil science 1, 718–721. https://doi.org/10.1081/E-ESS-120001709.

George, E., Horst, W. J., Neumann, E., 2012. Adaptation of plants to adverse chemical soil conditions. In Marschner's mineral nutrition of higher plants, Academic Press 409-472. https://doi.org/10.1016/B978-0-12-384905-2.00017-0.

Giardina, C.P., Sanford, R.L., Døckersmith, I.C., 2000. Soil Sci. Soc. Am. J. 64, 399–405. https://doi.org/10.2136/sssaj2000.641399x.

Guedes, D., Barbosa, L.M., Martins, S.E., 2006. Acta Bot. Bras. 20, 299–311. https://doi.org/10.1590/S0102-33062006000200006.

Hanazaki, N., Castro, F.D., Oliveira, V.G., Peroni, N., 2007. Ambient. Soc. 10, 121–136. https://doi.org/10.1590/S1414-753X2007000100008.

Hattori, D., Sabang, J., Tanaka, S., Kendawang, J.J., Ninomiya, I., Sakurai, K., 2005. Malaysia. Soil Sci. Plant Nutr. 51, 231–241. https://doi.org/10.1111/j.1747-0765.2005.tb00027.x.

Heinimann, A., Mertz, O., Frolking, S., Egelund Christensen, A., Hurni, K., Sedano, F., Hurtt, G., 2017. PLoS One 12. https://doi.org/10.1371/journal.pone.0184479.

Huffman, E.L., MacDonald, L.H., Stednick, J.D., 2001. Hydrol. Process. 15, 2877–2892. https://doi.org/10.1002/hyp.379.

Hölscher, D., Ludwig, B., Möller, R.F., Fölster, H., 1997. Agric. Ecosyst. Environ. 66, 153–163. https://doi.org/10.1016/S0167-8809(97)00077-7.

Jakovac, C.C., Peña-Claros, M., Kuyper, T.W., Bongers, F., 2015. J. Ecol. 103, 67–77. https://doi.org/10.1111/1365-2745.12298.

Jakovac, C.C., Peña-Claros, M., Mesquita, R.C., Bongers, F., Kuyper, T.W., 2016. Agric. Ecosyst. Environ. 218, 116–125. https://doi.org/10.1016/j.agee.2015.11.013.

Jung, S., Rasmussen, L.V., Watkins, C., Newton, P., Agrawal, A., 2017. Ecol. Econ. 136, 53–61. https://doi.org/10.1016/j.ecolecon.2017.02.004.

Kallenbach, C.M., Grandy, A.S., Frey, S.D., Diefendorf, A.F., 2015. Soil Biol. Biochem. 91, 279–290. https://doi.org/10.1016/j.soilbio.2015.09.005.

Kendawang, J. J., Tanaka, S., Ishihara, J., Shibata, K., Sabang, J., Ninomiya, I., ... Sakurai, K., 2004. Effects of shifting cultivation on soil ecosystems in Sarawak, Malaysia: I. Slash and burning at Balai ringin and Sabal experimental sites and effect on soil organic matter. Soil Sci. Plant Nutr. 50, 677-687. https://doi.org/10.1080/00380768.2004.10408524.

Kleinman, P.J., Pimentel, D., Bryant, R.B., 1995. Agric. Ecosyst. Environ. 52, 235–249. https://doi.org/10.1016/0167-8809(94)00531-I.

Lawrence, D., Radel, C., Tully, K., Schmook, B., Schneider, L., 2010. Biotropica 42, 21–30. https://doi.org/10.1111/j.1744-7429.2009.00599.x.

Lehmann, J., Kleber, M., 2015. The contentious nature of soil organic matter. Nature 528, 60–68. https://doi.org/10.1038/nature16069.

Lessa, A.S., Anderson, D.W., Moir, J.O., 1996. Agric. Ecosyst. Environ. 59, 191–202. https://doi.org/10.1016/0167-8809(96)01044-4.

Lima, R.A.F., Oliveira, A.A., Martini, A.M.Z., Sampaio, D., Souza, V.C., Rodrigues, R.R., 2011. Acta Bot. Bras. 25, 633–645. https://doi.org/10.1590/S0102-33062011000300017.

Loranger-Merciris, G., Imbert, D., Bernhard-Reversat, F., Ponge, J.F., Lavelle, P., 2007. Biol. Fertil. Soils 44, 269–276. https://doi.org/10.1007/s00374-007-0199-5.

Ma, J.F., Ryan, P.R., Delhaize, E., 2001. Trends Plant Sci. 6, 273–278. https://doi.org/10.1016/S1360-1385(01)01961-6.

Magnago, L.F.S., Martins, S.V., Schaefer, C.E.G.R., Neri, A.V., 2010. Acta Bot. Bras. 24, 734–746. https://doi.org/10.1590/S0102-33062010000300017.

Mareddy, A. R., 2018. Impacts on soils and land environment. In Mareddy, A. R., Shah, A., and Davergave, N. (eds.), Environmental impact assessment: theory and practice. Butterworth-Heinemann, pp. 249-296 https://doi.org/10.1016/C2015-0-06055-5.

Marín-Spiotta, E., Sharma, S., 2013. Glob. Ecol. Biogeogr. 22, 105–117. https://doi.org/10.1111/j.1466-8238.2012.00788.x.

Mataix-Solera, J., Doerr, S.H., 2004. Geoderma 118, 77–88. https://doi.org/10.1016/S0016-7061(03)00185-X.

McKenzie, R. 2003. Soil pH and plant nutrients. AGRI-FACTS, Practical information for Alberta's agriculture industry. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/agdex6607/$file/soilph.pdf?OpenElement (March 29, 2019).

Mendoza-Vega, J., Messing, I., 2005. Catena 60, 279–292. https://doi.org/10.1016/j.catena.2004.12.002.

Mertz, O., 2009. Curr. Opin. Environ. Sustain. 1, 156–160. https://doi.org/10.1016/j.cosust.2009.10.002.

Monteiro, R., Cesar, O., 1995. Spatial patterns or trees in the Picinguaba coastal sandy forest (state of Sao-Paulo, Brazil)., Arquivos de Biologia E Tecnologia, 533-540. http://hdl.handle.net/11449/32992.

Morellato, L.P.C., Haddad, C.F., 2000. Introduction: The Brazilian Atlantic Forest. Biotropica 32, 786–792. https://doi.org/10.1111/j.1744-7429.2000.tb00618.x.

Nelson, D. W., Sommers, L., 1982. Total carbon, organic carbon, and organic matter. Methods of soil analysis: Part 2 chemical and microbiological properties, 9, 539-579. https://doi.org/10.2134/agronmonogr9.2.2ed.c29.

Obale-Ebanga, F., Sevink, J., de Groot, W.D., Nolte, C., 2003. Soil Use Manage. 19, 83–86. https://doi.org/10.1111/j.1475-2743.2003.tb00284.x.

Okore, I.K., Tijani-Eniola, H., Agboola, A.A., Aiyelari, E.A., 2007. Agric. Ecosyst. Environ. 120, 250–258. https://doi.org/10.1016/j.agee.2006.09.011.

Oliveira, T. A., 2004. Compartimentação fisiográfica aplicada à avaliação de terrenos - subsídio ao planejamento territorial do município de Cananéia. Universidade Estadual Paulista, Instituto de Geociências e Ciências Exatas. http://hdl.handle.net/11449/92834.

Pedroso Jr., N.N., Adams, C., Murrieta, R.S.S., 2009. In: Current Trends in Human Ecology. Cambridge Scholars Press, Newcastle upon Tyne, pp. 12–34.

Peroni, N., Hanazaki, N., 2002. Agric. Ecosys. Environ. 92, 171–183. https://doi.org/10.1016/S0167-8809(01)00298-5.

Peroni, N., Martins, P. S., 2000. Influência da dinâmica agrícola itinerante na geração de diversidade de etnovariedades cultivadas vegetativamente. Interciencia. 25, 22-29. ISSN: 03781844.

Peters, W.J., Neuenschwander, L.F., 1988. Slash and burn: farming in the Third World forest. University of Idaho Press.

Powers, J.S., Marín-Spiotta, E., 2017. Annu. Rev. Ecol. Evol. Syst. 48, 497–519. https://doi.org/10.1146/annurev-ecolsys-110316-022944.

2001. Instituto Agronômico, Campinas, p. 285.

1996. InstitutoAgronômico & Fundação IAC, Campinas.

Reganold, J.P., Wachter, J.M., 2016. Organic agriculture in the twenty-first century. Nat. Plants 2, 1–8. https://doi.org/10.1038/nplants.2015.221.

Ribeiro Filho, A.A., Adams, C., Murrieta, R.S.S., 2013. Bol. Mus. Para. Emílio Goeldi. Ciênc. Hum. 8, 693–727. https://doi.org/10.1590/S1981-81222013000300013.

Ribeiro Filho, A.A., Adams, C., Manfredini, S., Aguilar, R., Neves, W.A., 2015. Soil Use Manag. 31, 474–482. https://doi.org/10.1111/sum.12224.

Ribeiro Filho, A.A., Adams, C., Manfredini, S., Munari, L.C., da Silva Junior, J.A., Ianovali, D., Neves, W.A., 2018. Bol. Mus. Para. Emílio Goeldi. Ciênc. Nat. 13, 79–106. https://boletimcn.museu-goeldi.br/bcnaturais/article/view/370.

Rossi, M., 2017. Mapa Pedológico Do Estado de São Paulo: Revisado e Ampliado. Instituto Florestal. http://iflorestal.sp.gov.br/2017/09/26/mapa-pedologico-do-estado-de-sao-paulo-revisado-e-ampliado/ (March 28, 2019).

SAS Institute, 2018. JMP pro 14.0.1. SAS institute, Cary, North Carolina, USA.

Shiferaw, B.A., Okello, J., Reddy, R.V., 2009. Environ. Dev. Sustain. 11, 601–619. https://doi.org/10.1007/s10668-007-9132-1.

Siahaya, M.E., Hutauruk, T.R., Aponno, H.S., Hatulesila, J.W., Mardhanie, A.B., 2016. Int. J. Biodivers. Sci. Ecosyst. Serv Manag. 12, 14–23. https://doi.org/10.1080/21513732.2016.1169559.

Six, J., Callewaert, P., Lenders, S., De Gryze, S., Morris, S. J., Gregorich, E. G., ... Paustian, K., 2002. Measuring and understanding carbon storage in afforested soils by physical fractionation. Soil Sci. Soc. Am. J. 66, 1981-1987. https://doi.org/10.2136/sssaj2002.1981.

Smeck, N.E., 1985. Geoderma 36, 185–199. https://doi.org/10.1016/0016-7061(85)90001-1.

Soil Science Division Staff, 2017. Soil survey manual. C. Ditzler, K. Scheffe, and H.C. Monger (eds.). USDA Handbook 18. Government Printing Office, Washington, D.C. https://www.nrcs.usda.gov/wps/portal/nrcs/detailfull/soils/ref/?cid=nrcs142p2_054262.

Sommer, R., Vlek, P.L., de Abreu Sá, T.D., Vielhauer, K., Coelho, R.D.F.R., Fölster, H., 2004. Nutr. Cycling Agroecosyst. 68, 257–271. https://doi.org/10.1023/B:FRES.0000019470.93637.54.

Sparks, D. L., Page, A. L., Helmke, P. A., Loeppert, R. H., Soltanpour, P. N., Tabatabai, M. A., ... Sumner, M. E., 1996. Soil pH and soil acidity. Methods of soil analysis. Part, 3, 475-490.

Veldkamp, E., Schmidt, M., Powers, J.S., Corre, M.D., 2020. Nat. Rev. Earth Environ. 1, 590–605. https://doi.org/10.1038/s43017-020-0091-5.

Visscher, A. M., Vanek, S., Meza, K., de Goede, R. G., Valverde, A. A., Ccanto, R., ... Fonte, S. J., 2020. Eucalyptus and alder field margins differ in their impact on ecosystem services and biodiversity within cropping fields of the Peruvian Andes. Agric. Ecosyst. Environ. 303, 107107 https://doi.org/10.1016/j.agee.2020.107107.

van Vliet, N., Mertz, O., Heinimann, A., Langanke, T., Pascual, U., Schmook, B., ... Castella, J. C., 2012. Trends, drivers and impacts of changes in swidden cultivation in tropical forest-agriculture frontiers: a global assessment. Global Environ. Chang. 22, 418-429. https://doi.org/10.1016/j.gloenvcha.2011.10.009.

Wright, R.J., 1989. Soil aluminum toxicity and plant growth. Commun. Soil Sci. Plant 20, 1479–1497. https://doi.org/10.1080/00103628909368163.