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Conversion of the coastal Atlantic forest to pasture: Consequences for the nitrogen cycle and soil greenhouse gas emissions

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ABSTRACT

We evaluated how land use change affected the nitrogen cycle and greenhouse gas fluxes. We measured soil temperature and water-filled pore space (WFPS), as well as nitrate and ammonium concentrations, mineralization and nitrification rates, and flux of CH_4, CO_2 and N_2O in soils of pasture catchment and in soils of a forest catchment located on the north coast of the State of São Paulo. The main vegetation type in the forest catchment is primary Atlantic Montane Forest, while the pasture is an unfertilized 40-year old area planted with *Brachiaria humidicola*. Four plots were used for monthly sampling in each land use for an entire year. Soil temperature was always higher in pasture than in forest soils, while WFPS was lower in the pasture in relation to forest soils. Pasture soils were a weak source of CH_4 during the winter months and a less strong sink of methane than the forest soils during the rainy summer months of the year. The annual median CH_4 uptake was -1.8 ± 1.0 mg m^{-2} d^{-1} in the forest in contrast to -0.6 ± 0.9 mg m^{-2} d^{-1} in pasture soils. CO_2 emissions were similar in the winter, but higher in the summer months in pasture soils in relation to forest soils. The annual median flux in the forest was 4.2 ± 1.5 μmol m^{-2} s^{-1} and increased in the pasture to 6.5 ± 2.9 μmol m^{-2} s^{-1}. In contrast, N_2O fluxes were smaller in the pasture (0.3 ± 0.7 ng cm^{-2} h^{-1}) than in forest soils (0.5 ± 0.5 ng cm^{-2} h^{-1}).

1. Introduction

The single most important land cover in Brazil is pasture for livestock: almost 200 million ha of mostly forestlands were replaced by African grasses since Brazil was settled in the 1500s. Most of the major Brazilian biomes – Atlantic and Amazon forests, Cerrado, Caatinga and Pantanal – were affected by the forest-to-pasture conversion. The Amazon Forest and the Atlantic forest are of greatest concern due to the vast amount of acreage already converted and due to the large biodiversity of these two biomes. Besides the loss of biodiversity in the Amazon region, the replacement of forests by pastures alters the structure and the functioning of forest ecosystems. Although fewer studies have been done on the Atlantic forest, the same changes observed in the Amazon region seem to occur in the former biome (Coutinho et al., 2010).

Among these changes, biomass decreases abruptly, which also implies a loss of nutrients from the burned vegetation that are lost from pasture sites in a few years (Boone Kauffman et al., 1998). As pasture land ages, a general decrease in the nitrogen content in relation to its original cycle is observed in the tropics, especially in the Amazon region (Davidson et al., 2000; Erickson et al., 2001; Wick et al., 2005; Neill et al., 2005; Cerri et al., 2006). In general, a decrease is observed in the nitrogen transformation process such as mineralization and nitrification, which is followed by a shift in the nitrogen content in the soil, from nitrate–ammonium found in similar concentrations in the forest soils into ammonium dominance in pasture soils. As a consequence, over time the N gaseous emissions from pasture soils become lower than emissions observed in forest soils (Neill et al., 2005).

Although only 11–16% of the original Atlantic forest remains (Ribeiro et al., 2009), very little is known about changes in biogeochemical processes caused by the conversion of forests into pastures that occurred in this biome (Coutinho et al., 2010). Therefore, it is reasonable to question whether the changes observed in the Amazon region described above are similar to changes occurring in the Atlantic forest due to land use changes occurring in the past.

The main objective of this study is to evaluate how the replacement of the Atlantic forest by a 40-year old pasture affected the nitrogen cycle and the emissions of greenhouse gases. In order to

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achieve this objective, for one year, we measured indices monthly that represent the nitrogen cycle in both biomes, among them, inorganic nitrogen soil concentrations, potential mineralization and nitrification rates, and N_2O emissions. Additionally, fluxes of CO_2 and CH_4 were also measured.

2. Material and methods

2.1. Study sites

We conducted our sampling in four 1-ha plots in two small catchments located on the border between the Atlantic plateau and the scarps of the mountain chain of the *Serra do Mar*, which in turn reach the Atlantic Ocean coastline. In the first four months, we conducted our sampling in 1 ha plots located in the State Park of the *Serra do Mar* in the Santa Virgínia unit near the crest of the mountain chain at 23°17′–23°24′S and 45°03′–45°11′W. Due to logistical reasons, we had to move these initial sampling plots approximately 6 km southwest from this site in the same park to a site at the same altitude of approximately 1000 m. Both sites have the same type of soil and vegetation. However, we cannot exclude the possibility of differences in the parameters that we measured between these two sites. The new sampling area encompasses a forest catchment area of 11.5 ha located at 23°19′19″S; 45°05′56″W. The pasture catchment area is 4.7 ha in size and located approximately 18 km to the west (23°24′54″S; 45°15′04″W) from the forest catchment at an altitude of 886 m.

These forests are classified as Montane Ombrophylus Dense Forest according to the Brazilian classification. The average above-ground biomass of our studied site is approximately 280 Mg ha^{-1}, and the average above-ground carbon stock was 149 Mg ha^{-1}. There are approximately 1700 stems ha^{-1}, considering a diameter breast height of 4.8 cm, and the average height of the canopy is 15 m, with emergent trees reaching 30 m (Alves et al., 2010). The annual litterfall is 5.5 Mg ha^{-1} yr^{-1} and the litter stock is 4.8 Mg ha^{-1} yr^{-1} (Sousa Neto et al., 2011). A thick layer of fine roots, reaching a density of 684 g m^{-2} (live and dead fine roots) during the rainy season, and increasing to almost 2000 g m^{-2} during the dry season (Sousa Neto et al., 2011).

The local climate is humid subtropical with hot summers (Cfa type in Köppen) with a mean annual precipitation of 2500 mm and mean annual temperature ranging from 19.1 to 25.5 °C (Sentelhas et al., 1999). The rainy season occurs from October to March with maximum rainfall in summer from November to February and minimum in winter from June to August, although, precipitation is rarely absent in all months (Setzer, 1966; Tabarelli and Mantovani, 1999; Salemi, 2009; Trevisan, 2009). Bulk soil density was higher in the pasture soils than in the forest soils in the four depths (15, 30, 50 and 100 cm) where samples were taken (Salemi et al., submitted for publication).

Pasture was established in 1968 by replacing the forest with *Brachiaria decumbens* Stapf. There was no use of either heavy machinery or any type of fertilizer, and soil was not amended during this period. The mean cattle stocking density is about 2–3 heads per ha^{-1}.

Forest and pasture soils are young soils such Inceptsols (Typic Dystrudepts) formed under gneisses and granites. Both soils are of medium texture containing approximately 57% of sand and 23% of clay. The pH is very acid (3.5–3.8), with low P and base cation concentrations (Salemi et al., submitted for publication).

2.2. Experimental design and soil gas flux

Soil gas fluxes were measured once a month from May 2007 through April 2008, in each plot with a day of collection per land use, generally between 08:00 and 18:00 h local time. Fluxes of nitrous oxide (N_2O), carbon dioxide (CO_2), and methane (CH_4) were measured at random points along 30 m lines, a line per plot. Transects were initiated at randomized seed points in randomized directions each month with eight cylindrical PVC chambers (Comercial Hidráulica, Piracicaba, SP, Brazil), with volume of 0.01 m^3 (8 subsample chambers per plot) consisting of a pipe serving as a base (0.29 m diameter) and a cap fitting snugly on the base (Keller et al., 2005). Hutchinson and Livingston (2001) recommended an insertion depth in the soil of less than 5 cm in fine-textured soils, wet or compacted soils. Our soils are of medium texture, and they are wet in the forest and compacted in the pasture. Accordingly we chose an insertion depth of 2.5 cm in the soil in order to minimize root disturbance, which is always a concern in gas flux measurements with chambers (Davidson et al., 2002). Another concern when using chambers is the place and time of deployment. Keller et al. (2000) found that immediate installations made minutes before gas flux measurement, and permanent long-term installations made at least 30 days prior to measurement produced a lower error in flux measurements. Soil heterogeneity is always a problem for gas flux measurement (Davidson et al., 2002). In order to minimize potential errors in deploying a chamber in a site with a particularly large or small flux, we rotated the deployment of the chamber at each collection as explained above.

For N_2O and CH_4, four samples of 60 mL of the air from the chambers were withdrawn at intervals of 1, 10, 20 and 30 min after closing with 60 mL disposable syringes type BD (Cremer S.A., Blumenau, Santa Catarina, Brazil) and then transferred to previously evacuated glass vials sealed with 20 mm gas impermeable blue butyl rubber septum style stopper (Bellco Glass, Vineland, NJ, USA). Samples were analyzed by gas chromatography (SHIMADZU GC-14A Model – Columbia, MD, USA) within 5 days of collection. Laboratory tests showed that N_2O and CH_4 concentrations were unaffected by storage for up to 30 days. Gas concentrations were calculated by comparing peak areas for samples with those of commercially prepared standards (Scott-Marrin – Riverside, CA, USA) calibrated against standards prepared by the National Oceanic and Atmospheric Administration/Climate Monitoring and Diagnostic Laboratory (NOAA/CMDL – Bolder, CO, USA). Two cylinders were used to compare peaks, and their concentrations for N_2O and CH_4 were for cylinder #1: 313.4 ppb and 1.810 ppm; and for cylinder #2: 773.0 ppb and 0.920 ppm, respectively.

A dynamic flow system was used for measurements of CO_2. Air flowed from the soil enclosure through a Teflon-lined polyethylene sample line 5 m in length, and then it entered an infrared gas analyzer (Li-Cor 820 – Li-Cor Biosciences, Lincoln, NE, USA). Chambers were sealed closed for 5 min and the whole data sets from the beginning of the closure of the chambers were used in flux calculations.

Fluxes were calculated from the increase of concentration versus time adjusted for the ratio of chamber volume to area and the air density within the chamber (Keller et al., 2005). It has been reported that a linear regression is not always the best fit of the regression between time and gas concentration in the chamber (Kroon et al., 2008; Koehler et al., 2009). The failure to choose the right regression type could severely affect flux estimates (Kutzbach et al., 2007; Forbrich et al., 2010). Based on the Akaike coefficient, we tested all regression (697 chambers) to find the best data fit for flux calculations. For N_2O, approximately 89% followed a linear regression, 10% a quadratic regression and only 1% an exponential regression. Only 2% of CH_4 measurements found a quadratic regression, with 98% of the remaining data following a linear regression. Finally for CO_2 approximately 50% of all regressions followed a linear regression, and 50% a quadratic regression.

2.3. Soil water-filled pore space (WFPS) and N contents

Once a month during one year of collection, and after soil gas collection, the surface litter was removed from each chamber location and a soil core about 5 cm diameter and 10 cm length was collected. After collection, soil samples were transported on ice in an insulated cooler to the Laboratory of Isotope Ecology at CENA-USP and stored at ~4 °C until analysis. Before analysis procedures, soil samples were left for 24 h for acclimatization to recondition the microbial activity and to avoid artificially low N cycling rates possibly due to the cooling. Soil samples were sieved (sieve 2 mm mesh), and a 10-g-subsample was oven-dried at 105 °C for 24 h to determine water content gravimetrically and N contents (NH_4^+ and NO_3^-), and N-mineralization and N-nitrification processes as the procedures described by Piccolo et al. (1994). The preceding storage time and conditions until soil extraction may interfere with the inorganic N fraction content (Arnold et al., 2008; Turner and Romero, 2009). The ideal situation would be to perform soil extraction as soon as possible, preferably in the field. That was not the case in this study: extractions were made between a minimum of two days, and a maximum of five days after soil sampling. $NH4^+$ and NO_3^- concentrations tend to increase with storage time (Turner and Romero, 2009). Therefore, it could be that the inorganic nitrogen contents of this study are overestimated.

The processes of N_2O production are strongly influenced by soil moisture content (Firestone and Davidson, 1989; Davidson, 1993). In order to assess the relation between N_2O fluxes and soil moisture, we estimated the water-filled pore space (WFPS), which is thought to be an important factor controlling N-oxide emissions from soil. Thus, WFPS was evaluated from soil core samples collected once a month from each chamber location and calculated according to Carmo et al. (2007). Additionally, we recorded air and soil temperatures (2 cm depth) using digital thermometers, model Taylor 9842 (Taylor Precision Products, Las Cruces, NM, USA).

2.4. Statistical analyses

We tested the normality of distributions by applying the Kolmogorov–Smirnov test. The following parameter did not follow a normal distribution: soil nitrate and ammonium concentrations, net mineralization and net nitrification, and fluxes of methane, carbon dioxide and nitrous oxide. These parameters were transformed using Box–Cox transformations, but first, the parameters that had negative numbers were transformed into positive numbers by summing a certain value for each parameter. After the transformations, normality was checked again together with the homogeneity of the variance of the residues. The mineralization, nitrification and N_2O flux parameters did not follow a perfectly normal distribution. Therefore, we used non parametric tests to check the results obtained by parametric tests.

Due to the possible temporal correlations among data, in order to check for differences between land uses in the whole data set, we used mixed-effects ANOVA considering land use as a fixed effect and months as random effects. Seasonal and monthly differences were tested by applying repeated measures ANOVA followed by Tukey Honest Test for unequal difference. In cases where we checked the results obtained by parametric tests with non-parametric tests, we used Kruskal–Wallis ANOVA and median tests. For all cases, the parametric and non-parametric tests produced similar results. For regression analyses we used least-square methodology. All statistical analysis were made using STATISTICA version 10.

Table 1
Mean values and standard-deviation (stdev) for soil parameters measured in the forest and pasture sites of the Atlantic forest biome.

	Forest		Pasture	
	Mean	Std.dev	Mean	Std.dev
Air T (°C)	16.5	3.7	23.8	5.2
Soil T (°C)	15.1	2.1	19.7	2.7
Soil humidity (%)	41.6	7.5	21.7	4.6
WFPS (%)	56.7	10.3	55.5	11.7
Soil NH_4 ($\mu g\,g^{-1}$)	9.6	7.8	9.3	1.3
Soil NO_3 ($\mu g\,g^{-1}$)	12.7	15.9	7.8	12.7
N-mineralization ($\mu g\,g^{-1}\,d^{-1}$)	3.2	5.5	0.1	2.3
N-nitrification ($\mu g\,g^{-1}\,d^{-1}$)	3.4	5.1	0.4	0.3
N_2O fluxes ($ng\,cm^{-2}\,h^{-1}$)	0.5	0.5	0.3	0.7
CO_2 fluxes ($mg\,m^{-2}\,d^{-1}$)	4.0	1.4	6.9	2.9
CH_4 fluxes ($\mu mol\,m^{-2}\,s^{-1}$)	−1.8	1.0	−0.6	0.9

3. Results

The annual mean and standard-deviations for soil variables are summarized in Table 1.

3.1. Air and soil temperature

Air temperature does not reflect a long-term average, but simply the sampling day temperature. Even with this bias, the air temperature of the pasture area was higher than the forest (Fig. 1A) either considering the whole year data or the data grouped in dry and wet seasons (Table 2). Soil temperature increased from May-2007 to March-2008 and started to decrease again in winter in April-2008 in forest as well as pasture soils (Fig. 1B). Soil temperature in the dry season were lower than the wet season (Table 3) for both

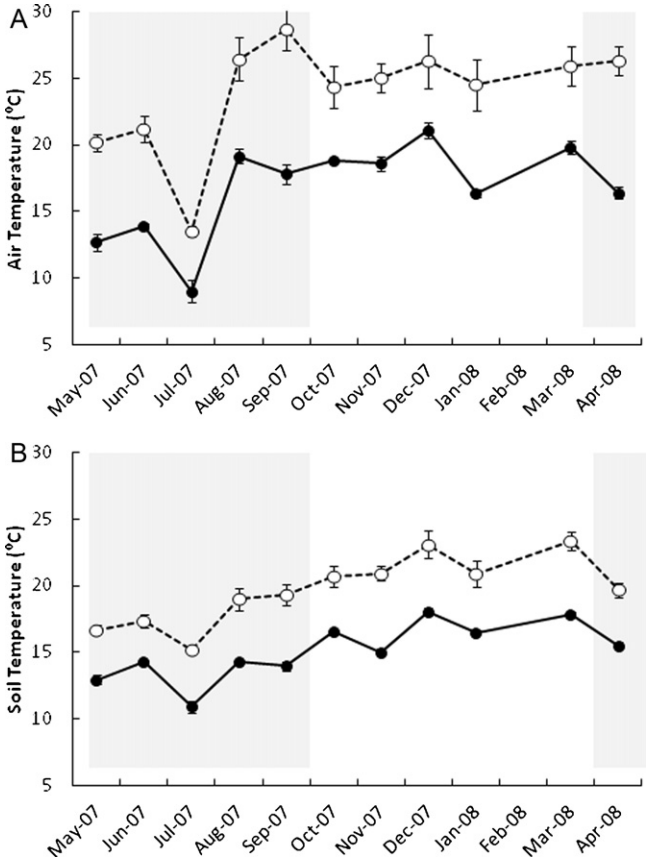

Fig. 1. Monthly variability of the mean soil humidity (A) and mean WFPS (B) in the forest (●) and pasture (○) sampling sites. Grey rectangles indicate dry season.

Table 2
Statistical comparisons of soil variables between forest and pasture for different time scales. Annual – *F* values and probabilities between brackets of mixing-models ANOVA. Seasonal – Tukey Honest Test probabilities after repeated measures ANOVA. Dry period: April–September; wet period: October–March. Monthly – results from Tukey Honest Test probabilities after repeated measures ANOVA showing months where differences between forest and pasture were significant. "ns" = not significant.

	Annual	Seasonal		Monthly
		Dry	Wet	
$T°C_{soil}$	75.03 (0.0001)	(<0.0001)	(<0.0001)	≠ All, except June
Humidity	161.86 (<0.0001)	(<0.0001)	(<0.0001)	≠ All, except December and April
WFPS	0.17 (ns)	(ns)	(ns)	≠ August
NH_4	0.02 (ns)	(ns)	(ns)	≠ December
NO_3	2.38 (ns)	(ns)	(ns)	≠ August
Miner.	6.00 (0.0343)	(0.0007)	(ns)	≠ June, July, and September
Nitrif.	5.02 (0.0489)	(0.0006)	(ns)	≠ June to September
N_2O	3.49 (ns)	(ns)	(ns)	≠ October
CO_2	17.09 (0.0020)	(0.0419)	(<0.0001)	≠ October to April
CH_4	20.74 (0.0010)	(<0.0001)	(ns)	≠ May to September

pasture and forest soils. Additionally, pasture soil temperatures were higher than forest soil temperatures either on an annual basis as well as in the wet and dry seasons (Table 2).

3.2. Soil humidity and water-filled pore space (WFPS)

Soil humidity did not change seasonally neither for forest or pasture soils (Fig. 2A). However, pasture soil humidity was lower than the forest soil both for the whole year and for dry and wet seasons (Table 2). There was no significant difference between the WFPS between the land uses, and no seasonal variation was observed in the WFPS in forest and pasture soils (Fig. 2B).

3.3. Soil ammonium and nitrate concentrations

Ammonium concentrations were relatively constant during the year with a sharp increase observed in December-2007 for both soils (Fig. 3A). For the whole annual data and for dry and wet periods, there was no statistical difference in ammonium concentration between forest and pasture soils (Table 2). However, considering month by month, the concentration peak that occurred in December-2007 in the pasture was higher than in the forest soil (Table 2).

There was also a sharp increase in the nitrate concentration in August-2007 in the forest soil and one month later in the pasture soil (Fig. 3B). From October-2007 to April-2008 the nitrate concentration was relatively constant (Fig. 3B). Similar to ammonium, the nitrate concentration in the forest soil was not different than the pasture soil considering the whole year or season (Table 2). But, the peak that occurred in August was significantly higher in the forest than in the pasture soils (Table 2).

Table 3
Probabilities of Tukey Honest test for soil variables between dry and wet periods for forest and pasture after repeated measures ANOVA. Dry period: April–September; wet period: October–March. "ns" = not significant.

	Pasture	Forest
$T°C_{soil}$	(<0.0001)	(<0.0001)
Humidity	(ns)	(ns)
WFPS	(ns)	(ns)
NH_4	(ns)	(ns)
NO_3	(ns)	(0.0190)
Miner.	(ns)	(0.0011)
Nitrif.	(ns)	(0.0005)
N_2O	(ns)	(ns)
CO_2	(0.0020)	(0.0419)
CH_4	(0.0010)	(<0.0001)

3.4. Mineralization and nitrification rates

The mineralization and nitrification rates (Table 3) of forest soils varied between dry and wet season but not for pasture soils. In the forest soils, both rates were higher in the dry season than in the wet season (Fig. 4A and B). Due to these higher rates during dry season months, the whole year data of mineralization and nitrification was also higher in the forest than in the pasture soils (Table 2).

3.5. Gas emissions

The N_2O fluxes were always positive (soil- atmosphere emission) with higher fluxes in the wet season in relation to the dry season (Table 3) (Fig. 5A). In the pasture soil, the N_2O

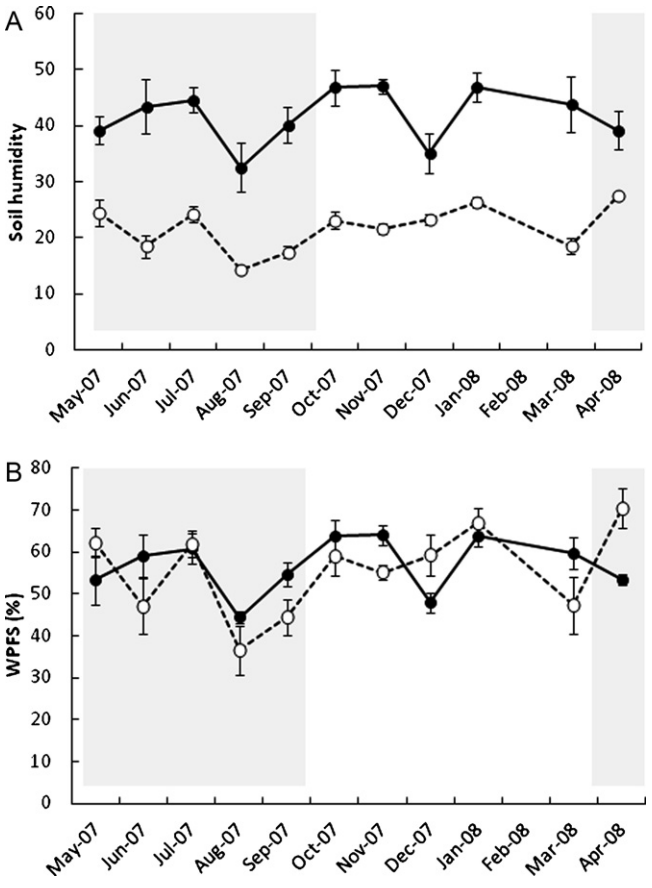

Fig. 2. Monthly variability of the mean air (A) and soil temperatures (B) in the forest (●) and pasture (○) sampling sites. Grey rectangles indicate dry season.

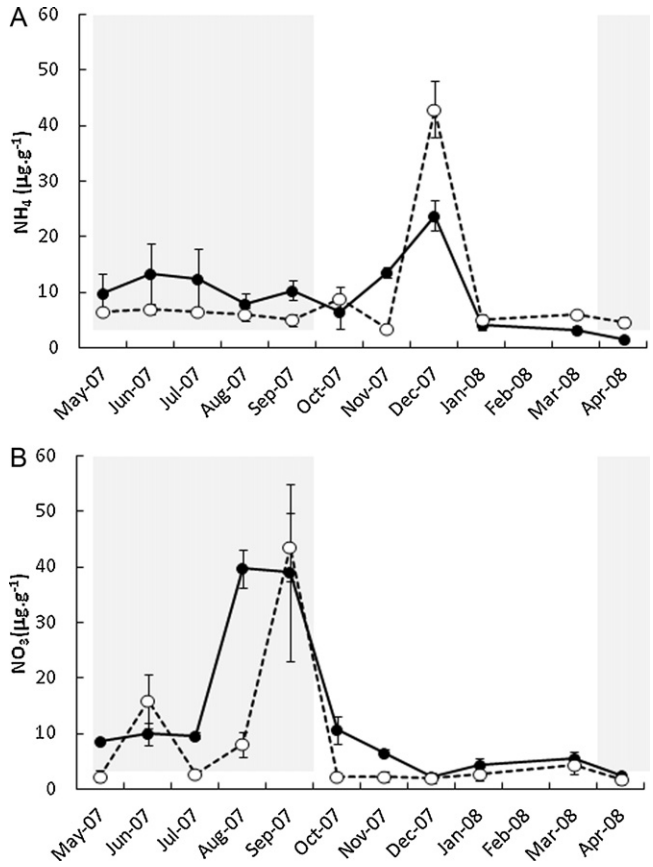

Fig. 3. Monthly variability of the mean soil ammonium concentration (A) and of the median nitrate concentration (B) in the forest (●) and pasture (○) sampling sites. Grey rectangles indicate dry season.

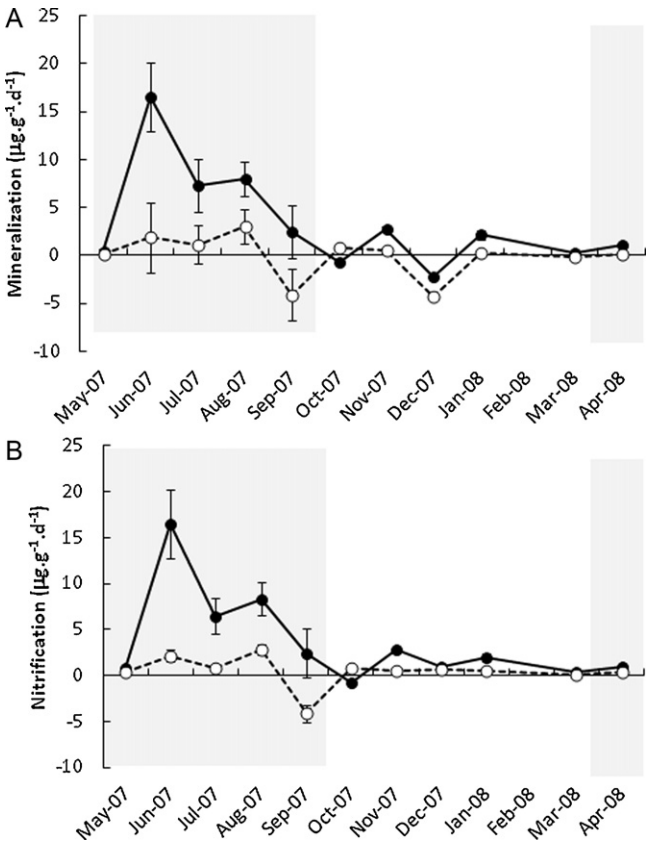

Fig. 4. Monthly variability of the mean of the net mineralization rate (A) and median of the net nitrification rate (B) in the forest (●) and pasture (○) sampling sites. Grey rectangles indicate dry season.

fluxes, however, showed an abrupt variation between October and November-2007 when the N_2O flux went from negative (soil consumption) to positive (soil emission), and a seasonal significant variation was not detected (Fig. 4A). With the exception of the month of October when the N_2O concentration was higher in the forest than in the pasture soils (Table 2), there were no seasonal or annual differences in N_2O fluxes between forest and pasture soils (Table 2).

The CO_2 fluxes of forest and pasture soils were always positive with higher emissions occurring in the wet season for both sites (Table 3) (Fig. 5B). The mean CO_2 flux throughout the sampling period was lower in the forest soils than in the pasture soils as well as in dry and wet seasons (Table 2) (Fig. 5B). We found a significant exponential correlation between soil temperature and CO_2 emission in the forest ($r^2 = 0.66$, $p < 0.0001$) and pasture soils ($r^2 = 0.74$, $p < 0.0001$). However, we did not find any significant correlation between CO_2 emission and WFPS.

Finally, the CH_4 fluxes in forest soils were always negative (soil consumption of atmospheric CH_4) with no clear seasonal variation (Fig. 5C). In pasture soils, fluxes went from slightly positive to neutral from May-2007 to September-2007 and became negative from October-2007 to April-2007 (Fig. 5C). As a consequence, pasture soils CH_4 fluxes were higher in the dry season than in the wet season (Table 3). In relation to the forest soils, the CH_4 fluxes of pasture soil were higher especially during the dry season and less so during the wet season (Table 3) (Fig. 5C).

The annual mean fluxes and the monthly variability observed in the forest sampling sites were similar to gas fluxes measured from September-2006 to August-2007 in a forest site located in the same state park, approximately 7 km northeast of our forest sampling site (Sousa Neto et al., 2011).

4. Discussion

There were important differences in the soil temperature between forest and pasture soils (Fig. 1). The pasture soil was heated by the removal of the forest canopy exposing pasture soil to direct sunlight. Additionally, bulk soil density was higher and soil hydraulic conductivity was lower in pasture than in forest soils (Salemi et al., submitted for publication). This fact increased overland water runoff in pasture in relation to forests, decreasing the soil water volume in the former (Salemi et al., submitted for publication). The same differential heating of pasture soils in relation to forest soils was also observed in the Atlantic forest region in a study conducted approximately 70 km northwest of our site (Coutinho et al., 2010). Also the same trend was found in a pasture site located near the city of Rio Branco, in the state of Acre, in the Amazon region (Salimon et al., 2004).

In our study, the annual CO_2 median flux was higher in pasture than in forest soils, and fluxes were higher in both land covers during the rainy-warmer season (Fig. 4B). The CO_2 fluxes were also higher in pasture soils located in the States of Acre and Rondônia in the Amazon region (Feigl et al., 1995; Salimon et al., 2004), but not for pastures located in the State of Pará (Davidson et al., 2000). We can argue that the CO_2 fluxes were higher in pasture than in forest soils only because the soil temperature in the pasture was always higher than in the forest, and although the soil was drier, some moisture was always available in pasture soils (Fig. 1). There is a great deal of evidence demonstrating the effect of soil

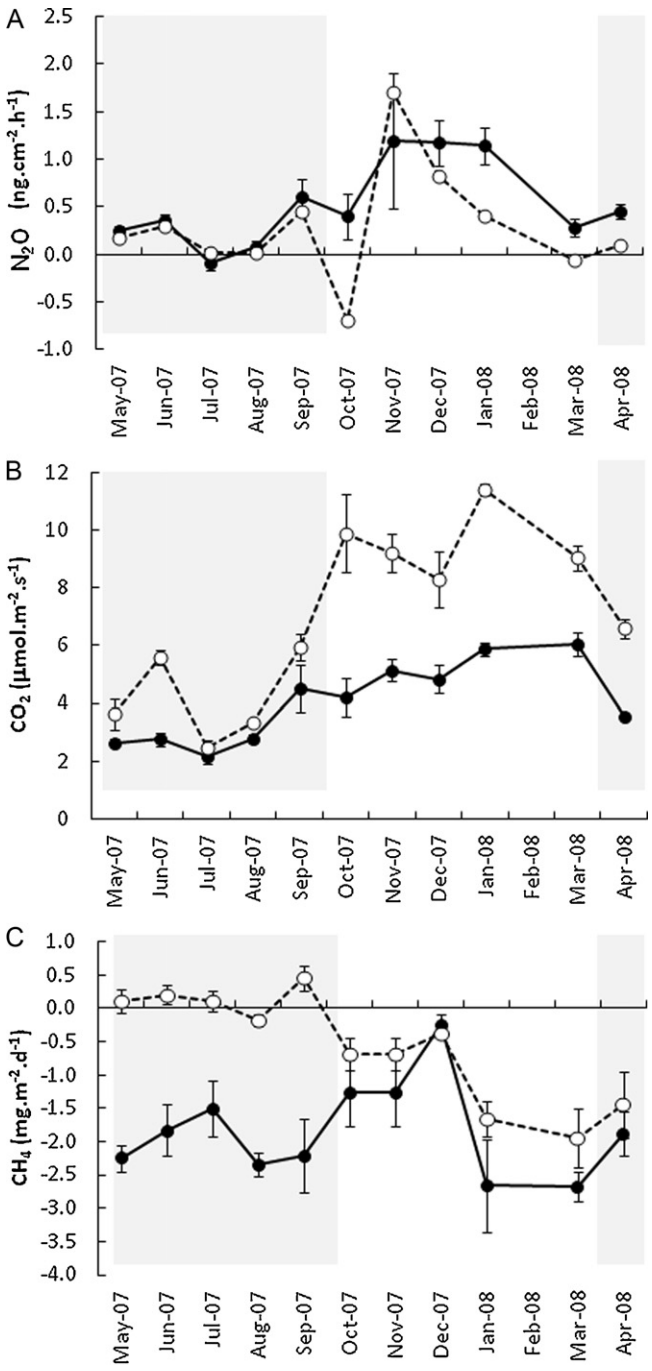

Fig. 5. Monthly variability of mean N_2O flux (A), median CO_2 flux (B), and median CH_4 flux (C) in the forest (●) and pasture (○) sampling sites Grey rectangles indicate dry season.

temperature on decomposition, especially of labile carbon (e.g. Davidson and Janssens, 2006; Dorrepaal et al., 2009).

Methane uptake by forest soils was observed, as has been observed in other tropical forest soils (Sousa Neto et al., 2011). On the other hand, the CH_4 uptake was lower in the pasture soils, and small emissions to the atmosphere were observed between May and July-2007. It is difficult to explain the behavior of CH_4 in pasture in relation to forest soils (Fig. 4C) since there was no significant difference in the WFPS between land uses (Fig. 1B), and no significant correlation was found between CH_4 flux and WFPS. Alternatively, it is possible that the increase in soil bulk density in the pasture coupled with a lower soil hydraulic conductivity could indicate a decrease in the soil pore connectivity that in turn would hinder diffusion of CH_4 into the soil leading to a lower uptake rate. On the other hand, it seems that the increase in CH_4 production in pasture soils after forest conversion is the rule in the tropics (Steudler et al., 1996; Verchot et al., 2000; Cerri et al., 2006). Verchot et al. (2000) found a significant direct relationship between CH_4 and CO_2 flux, and a significant inverse relationship between CH_4 and WFPS in pasture soils of the Amazon regions. These authors considered that an increase in the CO_2 production via soil respiration decreases the available O_2 and decreases CH_4 oxidation coupled with a decrease in the O_2 diffusivity caused by an increase in the WFPS. This combination of factors led to an increase in the CH_4 concentration. Observed changes in CH_4 oxidation with land use changes appear to be linked to changes in the methanotrophic community composition as observed in forest-pasture and forest-croplands conversions (Knief et al., 2005; Singh et al., 2007).

We found lower N_2O emission in the pasture soils than in the forest soils (Table 1). The same trend was found in a pasture area of the Atlantic forest located 70 km northwest of our study (Coutinho et al., 2010). Additionally, this seems to be a common future outcome in fallow unfertilized pastures of the Amazon region (Verchot et al., 2000; Melillo et al., 2001; Garcia-Montiel et al., 2001; Wick et al., 2005; Neill et al., 2005). Nitrogen availability is a key control agent for N_2O emissions (Erickson et al., 2001). The studies mentioned above have shown a strong decline in most indices of the N-cycle in old pastures in relation to forest soils. Probably this decline is a combined effect of lower N-input through biological nitrogen fixation coupled with export of nitrogen via cattle and progressive immobilization of nitrogen in the soil and plants (Wick et al., 2005). As a consequence, less nitrogen is available as nitrate to be denitrified. Additionally, several studies found significant correlations between inorganic soil nitrogen concentrations or N-cycle indices like mineralization and nitrification (e.g. Erickson et al., 2001; Wick et al., 2005; Cerri et al., 2006). Although, in our study we could not find any significant correlations as mentioned above, the same decline in most indices of the N-cycle was also observed in pasture soils, including lower N_2O emissions (Table 1). In addition, a larger nitrate-to-ammonium ratio in the forest than in the pasture soils suggests a more conservative soil N cycle in the former soils (Davidson et al., 2000). Overall such features in pasture soils of this study suggest that the nitrogen availability is lower in these soils than in forest soils of the Atlantic forest, explaining the higher N_2O emission in the latter (Table 1).

5. Conclusion

The conversion of the Atlantic forest to pasture resulted in pronounced change in several attributes of the soil, including a higher soil temperature coupled with lower soil water content than in forest soils. These changes were followed by changes in the carbon and nitrogen cycles. As observed in pastures of the Amazon region, the CH_4 uptake capacity of the pasture in the Atlantic forest decreased in relation to the forest soil. Also like some pastures in the Amazon region, pasture soil CO_2 emissions were higher than soils of the Atlantic forest, especially in the rainy and warm summer months. Finally, as in the Amazon region, a strong decline of the N-cycle indices was observed with an N-impoverishment of pasture soils that, in turn, lead to lower N_2O emissions in these soils in relation to soils of the Atlantic forest.

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