Nitrous oxide emissions along a gradient of tropical forest disturbance on mineral soils in Sumatra

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Abstract

Forest conversion to oil palm and rubber plantations is a common land-use change in Jambi, Sumatra due to the high economic demand of forest border communities. The environmental effects of such conversions have raised global concerns due to the potential to increase nitrous oxide emissions (N_2O) to the atmosphere. To quantify this effect, we conducted a series of monthly N_2O flux measurements between July 2010 and August 2011 using a static chamber method in an undisturbed forest, a disturbed forest, a one year old rubber plantation, a twenty year old rubber plantation and an eight year old oil palm plantation. All plantations belonged to smallholders and were usually not fertilized. In order to understand the effect of management intensification on N_2O fluxes, we applied nitrogen (N) as urea (33.3 kg N ha^{-1}) in the oil palm plantation in April 2011 and monitored the emissions intensively until 28 days after fertilizer application. Nitrous oxide consumption was significant in these weathered soils, accounting for 30% of recorded flux data, although 17% of the negative fluxes fell below detection limits. Most of these happened in the oil palm plantation and undisturbed forest. Annual N_2O emission rates amounted to 1.73 ± 0.48, 1.22 ± 0.27, 1.34 ± 0.36, 1.02 ± 0.27 and 1.04 ± 0.39 kg N ha^{-1} y^{-1} in the undisturbed forest, disturbed forest, one year old rubber plantation, twenty year old rubber plantation and oil palm plantation, respectively. Forest disturbance and conversion to rubber and oil palm plantation did not significantly affect annual N_2O emission rates. However in the oil palm plantation, the amount of N emitted as N_2O was high (3.1 ± 1.2% of the fertilizer N applied), so at a typical fertilizer application rate of 141 kg N ha^{-1} y^{-1}, annual emissions would have amounted to 4.4 ± 1.6 kg N ha^{-1} y^{-1}, more than twice the emission rate in the undisturbed forest. Dry mass and nitrogen mass in standing litter, distance to the nearest termite nest, rainfall on the day of measurement and air temperature were the key factors that predicted annual N_2O fluxes across the land-use change transitions.

Keywords: N_2O, forest conversion, land-use change, fertilizer, oil palm, rubber

1. Introduction

Nitrous oxide (N_2O) is the third most important greenhouse gas (GHG) in the atmosphere, contributing 6% of the radiative forcing from long-lived GHGs (WMO, 2011). Although the concentration of N_2O in the atmosphere is lower than the concentration of carbon dioxide (CO_2), it has a long lifetime (114 years) and, over a 100 year time period, its global warming potential is 298 times greater than that of CO_2 (Foster, 2007). The atmospheric N_2O concentration has increased rapidly over the last 10 years by 0.75 ppb and human activities contribute 40% to total emissions (Vitousek *et al.*, 1997; Chapuis-Lardy *et al.*, 2007; WMO, 2011). Most of soil N_2O fluxes originate from the biological processes of nitrification and denitrification, with a smaller component generated by the chemical process of chemodenitrification (Bremner and Blackner, 1978; Davidson *et al.*, 1993). Nitrification is the aerobic oxidation of ammonium (NH_4^+) or ammonia (NH_3) to nitrite (NO_2^-) and nitrate (NO_3^-), with N_2O production occurring under either oxic or anoxic (nitrifier denitrification) conditions. Denitrification is the anaerobic reduction of NO_3^- and NO_2^- to N_2O and nitrogen gas (N_2) (Hergoualc'h, 2011). Variables known to control N_2O fluxes are mineral nitrogen (N) availability, land-use management, temperature, soil water content, pH, texture and carbon supplies (Parton *et al.*, 1996; Skiba and Smith, 2000).

There is growing evidence that land-use change and agriculture in the tropics may contribute substantially to N_2O emissions (Veldkamp and Keller, 1997; Itoh *et al.*, 2010). Forest conversion to more intensive agriculture leads to a change and a simplification in vegetation composition. It also changes both the quantity and quality of litter inputs, impacts nutrient cycling and modifies the micro-climate and other soil processes (Hairiah *et al.*, 2006; Verchot *et al.*, 2006). For instance, Hergoualc’h *et al.* (2008) reported that the vegetation composition affects the soil mineralization potential. Keller *et al.* (1993), Verchot *et al.* (1999) and Yashiro *et al.* (2008) all observed a temporary increase in N_2O fluxes following logging activities in tropical rainforests, which was due to the increase in available soil N. In addition, N fertilizer application in agricultural lands is known to increase N_2O emissions (Bouwman *et al.*, 2002; Van Groenigen *et al.*, 2010; Gundersen *et al.*, 2012).

Sumatra has been intensively deforested in the past 20 years at average annual rates of 0.8 Mha y^{-1} between 1990 and 2010 (Margono *et al.*, 2012) and 0.4 Mha y^{-1} between 2000 and 2010 (Miettinen *et al.*, 2011). Rubber was introduced in Sumatra at the beginning of the 20th century (Brockway, 1978; Stolle *et al.*, 2003) and plantations have expanded rapidly together with subsistence agriculture. In the early 80s, commercial logging created rapid changes, and lands with depleted concessions (that should no longer have been logged after the governmental logging permits expired) were usually illegally logged and then converted to oil palm or rubber plantations (Laumonier *et al.*, 2010). In 1999, most (67%) Indonesian oil palm plantations had been established by large companies, while rubber plantations had mainly (85%) been planted by smallholders (Miyamoto, 2006). However in 2010 nearly half (42%) of the overall oil palm plantation area was managed by smallholders (Ministry of Agriculture, 2011). These statistics do not make a distinction between nucleus estate smallholders (who get capital, technology and market support from oil palm companies) and independent smallholders (Obidzinski et al., 2012) whose production rates and practices differ substantially. Independent smallholders usually produce less than nucleus estate smallholders due to limited access to high quality seeds and low use of fertilizer to minimize the costs (Vermeulen and Goad, 2006). Both rubber and palm oil are important commodities (Miyamoto, 2006) and plantations have been widely established in Jambi (Murdiyarso and Wasrin, 1995; Ketterings *et al.*, 1999) due to their profitable price (Miyamoto, 2006; Wilcove and Koh, 2010).

Rubber plantations are generally managed extensively with no fertilizer applications. Recommended fertilizer application rates in oil palm plantations vary according to climatic conditions, soil type, age of palms and palm yield potential (Comte *et al.*, 2012). They range from 35 to 120 kg N ha^{-1} y^{-1} for immature (less than 3 year old) palms and from 35 to 245 kg N ha^{-1} y^{-1} for mature palms with a density of 140 palms ha^{-1}. From field surveys, Khasanah *et al.* (2011) estimated a time-averaged N fertilizer application rate in the oil palm plantations of Indonesia of 141 kg N ha^{-1} y^{-1}.

Research on N_2O emissions from mineral soils following forest degradation and land-use change in Southeast Asia has been limited (Ishizuka *et al.*, 2002; Ishizuka *et al.*, 2005a; Verchot *et al.*, 2006; Arai *et al.*, 2008; Veldkamp *et al.*, 2008; Yashiro *et al.*, 2008; Fowler *et al.*, 2011). Due to the different rainfall patterns in Asian tropical forests compared to other tropical rainforest regions, the rates of biogeochemical processes and N_2O fluxes might be expected to differ. Emissions of N_2O from rubber and oil palm plantations on mineral soils have not been sufficiently investigated (Ishizuka *et al.*, 2002; Ishizuka *et al.*, 2005a; Werner *et al.*, 2006; Fowler *et al.*, 2011). The objective of this work is therefore to strengthen the current understanding of how soil N_2O emissions are affected by forest degradation and conversion to rubber and oil palm plantations. We have measured N_2O emissions from undisturbed and disturbed forests, rubber and oil palm plantations in Sumatra, Indonesia, with the aim of (1) quantifying the effect of tropical forest disturbance on N_2O emissions, (2) evaluating seasonal variations in the fluxes, (3) assessing the timing and magnitude of the effect of mineral N fertilizer application on the N_2O fluxes in the oil palm plantation and (4) identifying the environmental factors that control N_2O fluxes. We expected that N_2O emissions would increase after forest conversion to N fertilized oil palm plantation. We also expected that N_2O emissions would be controlled by soil moisture and be higher during the wet than during the dry months.

2. Materials and methods

2.1. Site description

The research was conducted in Pasir Mayang, Jambi Province, Indonesia in five land-use systems (LUS): undisturbed forest (FR), disturbed forest (DF), one year old rubber plantation (RB1), twenty year old rubber plantation (RB20), and eight year old oil palm plantation (OP). The FR, DF and RB1 sites were located in the former BIOTROP research site (Murdiyarso and Wasrin, 1995; Ishizuka *et al.*, 2002; Ishizuka *et al.*, 2005a; Ishizuka *et al.*, 2005b). Lowland forests of Pasir Mayang have a primary production between 70 and 100 Mg ha^{-1} y^{-1}, are dominated by *Dipterocarpaceae* and present a multilayer canopy (Murdiyarso and Wasrin, 1995; Wasrin *et al.*, 1999). Land-use change is ongoing in the area as a result of the high demand for agricultural lands and areas for settlement.

The landscape in the research area is undulating (Wasrin *et al.*, 1999). The FR (102°6'0 3.6" E, 1°4' 36.3" S; 104 m above sea level (a.s.l.)) was characterized by a high population of trees (50 trees ha^{-1}) with diameter at breast height (DBH) > 30 cm. The DF site (102°6' 00.6"E, 01°04' 45.1" S; 117 m a.s.l.) was located near to the FR site and close to the forest studied by Ishizuka *et al.* (2002) (P1; 102°5'70.2" E, 1°05'16.4" S). The number of large trees with a DBH > 30 cm was less (36 trees ha^{-1}) than in FR. The RB1 site (102°06' 36.5" E, 01°05'04.0" S; 81 m a.s.l.) was close to the logged-forest site of the study of Ishizuka *et al.* (2002) (L2; 102°06' 58.6" E, 01°05'23.5" S) and the RB20 (102° 06' 58.4" E, 01°05. 27.1" S; 102 m a.s.l.) was located approximately 2.3 km from FR, DF and RB1. The RB1 site was planted at a density of 3000 trees ha^{-1}; all trees had a DBH < 5 cm. In RB20, the density of trees with a DBH < 5 cm was 75 trees ha^{-1} and that of trees with a DBH > 5 cm was 1031 trees ha^{-1}. The OP (102° 08' 21.0" E, 01° 01'10.4" S; 69 m.a.s.l) belonged to a smallholder, was planted at a density of about 145 palms ha^{-1} and was not fertilized. For the purposes of this research, fertilizer was applied in April 2011. All land uses were located within the same area with a distance of 0.3-8.3 km between sites.

The annual rainfall between 2007 and 2010 (Tujuh Koto Ilir weather station, BMKG Jambi 2011, unpublished data) varied between 2030-2986 mm with the highest value in 2010. During the monitoring period (July 2010 – August 2011, 14 months), total rainfall amounted to 2646 mm, average air temperature was 28.8°C, and the maximum and minimum air temperatures were 48.1°C and 19.2°C. The soil was classified as Oxisol suborder Xanthic Kandiudox in the FR, DF, RB1; Oxisol suborder Typic Hapludox in the RB20; and Inceptisol suborder Typic Dystrudept in the OP. The size of the plots where both flux and environmental parameter measurements were undertaken was 1 ha, except for the DF, which was slightly smaller (0.8 ha) because field condition (steeper slope and pathway) prohibited establishment of a 1 ha plot. The slope in FR and DF was steep (30-35%), RB1 was sloping (8-15%), RB20 and OP were gently sloping (3-6%).

2.2. Flux measurement

A static chamber method was used for measuring soil fluxes of N_2O (Verchot *et al.*, 2000; Verchot *et al.*, 2006; Hergoualc'h *et al.*, 2008). The chambers consisted of a round PVC base (0.045 m^2 in surface, 0.25 m in height) that was closed for measurements using a PVC lid equipped with a small central port for gas sampling and a 2 mm diameter vent. The sampling design in all LUS, except the OP, included nine replicated chambers that were placed in a 400 m^2 area following the method of Verchot *et al.* (2006) (Figure 1A). The distance between chambers was 5 m. To get an independent measurement of N_2O fluxes, the distance between sampling points should be at least 1 m (Röver *et al.*, 1999). In the oil palm plantation, twelve chambers were distributed between the fertilized zone (FZ; six chambers) and the unfertilized zone (NFZ; six chambers) (Figure 1B). For the FR, DF, RB1 and RB20 sites, the fluxes from the nine replicate chambers were averaged at each measurement date. For the OP site, the flux at the plot scale was calculated by weighting the average flux in the fertilized (FZ) and non-fertilized (NFZ) zones by the surface these areas represent (10% and 90%, respectively) and summing them. Annual emissions were calculated by linear interpolation between measurement dates.

Gas samples were taken from the chambers and injected into pre-evacuated glass vials at chamber closure and 10, 20 and 30 minutes afterward. The filled vials were transported to the laboratory for analysis by gas chromatography (Loftfield *et al.*, 1997). Nitrous oxide concentrations were analysed using a Shimadzu 14 A gas chromatograph (GC) with an electron capture detector.

Due to problems with the GC, measurements for October 2010 were omitted. Nitrous oxide fluxes were calculated from the change in the concentration by linear regression based on the four samples. Negative fluxes were treated as real and left in all calculations. The sampling and analytical precision of the method was determined by computing the average and coefficient of variation from 45 standards analysed with the GC and calculating the detection limit (DL) of N_2O fluxes following the method of Parkin *et al*. (2012). The average N_2O concentration of the standards measured by the GC was 335 ppb with associated standard deviation and coefficient of variation of 21 ppb and 0.063, respectively.

The N_2O flux measurements were separated into two schemes; monthly measurements and, in the OP site, additional intensive measurements following fertilizer application. In the monthly measurement scheme, N_2O fluxes were sampled over a 14 month period from July 2010 until August 2011. In the intensive measurements scheme, N_2O fluxes were measured -2, 0, 1, 2, 3, 4, 5, 6, 7, 10, 14, 17, 21 and 28 days after fertilizer application. Urea, potassium chloride (KCl) and triple super phosphate (TSP) were applied following typical farmer practices. The application rate was 500 g tree^{-1}, equivalent to 33.3 kg N: 46.4 kg K: 33.3 kg P ha^{-1}. The fertilizer was applied in a rainy month (April 2011) only in the fertilized zone (Figure 1B). It was spread evenly by hand within a 1 m radius around the stems of the palms. The amount of fertilizer applied to the FZ chambers was calculated as the surface ratio between the chamber and the FZ multiplied by the fertilizer application rate. The percentage of N applied that was emitted as N_2O was calculated by dividing the annual N_2O emissions by the fertilizer application rate.

2.3. Ancillary data

The contribution of organic matter to N inputs to the soil system was determined by measuring the standing litter, consisting of leaves, coarse litter (partly decomposed leaves), and dead woody branches/twigs on the soil surface at the beginning of the experiment (July 2010). For that purpose, a 5 x 40 m^2 transect was established in each LUS and leaves, coarse litter and twigs were collected from ten frames of 0.5 x 0.5 m^2 in size placed along the transect midline (Hairiah *et al*., 2010). In the OP, the midline was located in between two planting rows and half of the frames were positioned on the side where fronds were left to decompose, the other half on the other side. In addition, litterfall was monitored monthly by installing three 2 x 1 m^2 litter traps in the FR, DF and RB20 plantation. For the RB1 plantation, six 1 x 1 m^2 litter traps were installed around six young rubber trees selected at random. There was no litter trap in the OP plantation because fronds were cut as part of standard plantation management practice and left on the site. The annual litterfall rate was calculated using the average of monthly litterfall rates monitored over the 14 month period. The litterfall results from October were not included in the calculation as no gaseous emissions were monitored during that period. The annual N litterfall inputs were calculated by multiplying the annual input from each component (branches, leaves, fruits, flowers) by its N content and summing them up. All of the organic matter was oven-dried at 70 ^{o}C for three days and its C and N contents were analysed following the Walkley-Black and Kjeldahl methods respectively (Black, 1965; Hesse, 1971).

For climate monitoring of the area, a rain gauge (Delta Ohm type HD2013-D) was installed, and hourly rainfall and air temperature were measured. Concomitant with N_2O measurement and at each chamber, air temperature in the shade and soil temperature at a 5 cm depth were measured with a digital thermometer (GTH 1170). Soil was sampled from the top 10 cm with nine and twelve replicates per plot in non-OP and OP LUS, respectively, to analyse for gravimetric moisture content. The soil moisture was expressed as water-filled-pore space (WFPS) following the formula by Linn and Doran (1984).

2.4. Soil properties

In September 2010, nine replicate soil samples were taken from the top 10 cm using a 183 cm^3 ring. The soil was sampled every 20 m on a 40 × 40 m systematic grid that included the flux measurement plot**.** The bulk density was determined using the core method (Grimaldi *et al.*, 2003) and particle density by the method of kerosene (Henríquez and Cabalceta, 1999). The total N and C contents were determined by dry combustion using a Flash EA 1112 Series Elemental Analyser (Thermo Finnigan, Bremen, Germany). Soil pH was measured in water (1:5 soil:water ratio) using a standard pH electrode and particle size distribution by following the pipette method (Pansu and Gautheyrou, 2006). Phosporus was analysed by spectrophotometry using Bray 1 extractant (Bray and Kurtz, 1945). Due to the observation of high termite activity in all sites, the distance from the chamber to the nearest termite nest was recorded in January 2011 (for wet months) and August 2011 (for dry months); this is to account for any impact on nutrient dynamics of the nests which are rich with NH_3 as a result of the digestion process and may promote hot spots of N_2O fluxes in the surrounding areas (Ohkuma *et al.*, 1999; Ji and Brune, 2006; Ngugi and Brune, 2012).

To get an understanding of N availability in the soil, NH_4^+ content was analysed in wet (June 2011) and dry months (May 2011 and August 2011). The results of May in DF, RB1 and RB20 were omitted due to sample contamination. The soil was sampled close to the chambers to a depth of 10 cm. Three and four replicates were sampled each time in the non OP and OP LUS, respectively. In the OP, two of the four replicates were taken in the FZ and two in the NFZ. The concentrations were scaled up to the plot level in the OP following the same procedure as for soil N_2O emissions. Inorganic N was extracted by adding 100 ml of 2 M KCl to a 10 g subsample of fresh soil. The samples were shaken for 1 hour and allowed to settle for 24 hours. The supernatant was filtered using Whatman 42 filter paper and analysed for NH_4^+ concentration by spectrophotometry using an auto-analyser (Bran and Luebbe, Nordestedt, Germany).

3. Statistical analysis

Statistical analyses were done using the software SPSS 20 and Infostat, using a probability level of 0.05 to test the significance of the effects. Residual distributions of all variables were checked using P-P and Q-Q plots (Gan and Koehler, 1990; Conti *et al.*, 2005; Park, 2008). The residual values of both N_2O fluxes and their log-transformed values were not normally distributed according to the Kolmogorov-Smirnov and Shapiro Wilk tests; hence, the non-parametric test of Kruskall-Wallis was used to analyse the variance in results. The Kruskall-Wallis post hoc test was used to compare the mean rank between treatments. The ANOVA and Kruskall-Wallis tests were used for normally and non-normally distributed environmental variables monitored concomitantly with N_2O fluxes. The temporal variability of N_2O fluxes in each LUS was evaluated by averaging the coefficient of variation obtained for each chamber. The Pearson correlation coefficient was used to determine the correlation between daily N_2O fluxes and environmental variables (Karhu *et al.*, 2011). Linear and non-linear, simple and multiple regressions (Eyduran *et al.*, 2005; Alexopoulos, 2010), which aim to find the best set of independent variables to explain the dependent variable results, were tested to determine the influence of the environmental variables on annual N_2O fluxes along the gradient. The N_2O flux measurements following fertilizer application in the FZ and NFZ were analyzed in the same way as the monthly measurements.

Observations are reported as mean or cumulated values ± standard error. Gaussian error propagation (Lo, 2005; Malhi *et al.*, 2009) was used to determine the uncertainty in annual N_2O emission rates calculated from monthly measurements. This method has been widely used in analysis of C stocks and long-term land and atmospheric fluxes (Moncrieff *et al.*, 1996; Weymann *et al.*, 2008; Hergoualc'h and Verchot, 2011). It is also recommended when calculating cumulative flux estimates over long periods, especially where fluxes are negative (Cowan *et al.*, 2014).

4. Results

4.1. Monthly measurements over the 14 month period

4.1.1. Climate and soil properties

All soils texture classes were sandy clay loam except the soil in RB20, which was clay (Table 1). Soils were acidic with a low pH below 5, which increased along the transition gradient. Bulk density had increased as a result of FR conversion to RB1 and OP, but had decreased in the case of RB20. Total C in RB20 was significantly higher ($p < 0.01$) than total C in other LUS. Total N was significantly higher ($p < 0.01$) in RB20 and OP than that in other LUS. In the OP, the C:N ratio below 11 was significantly lower ($p < 0.01$) than that in other LUS, indicating a potential faster decomposition rate. The largest distance to the nearest termite's nest from the chambers was in the OP. In the other LUS, termite nests were distributed closer to the chambers than in the OP. The Xanthic Kandiudox suborder soils (FR, DF, RB1) were more acidic and displayed a significant lower soil N content than the other suborder soils (RB20 and OP). The Oxisols (FR, DF, RB1, RB20) had a significantly higher C:N ratio and lower phosphorous content than the Inceptisols (OP).

Neither the arithmetic nor log-transformed values of WFPS measurements were normally distributed. There was a significant difference between LUS ($p < 0.01$), the OP having the highest average WFPS and the RB20 having the lowest (Table 1). The WFPS in the RB1 site was significantly smaller than in the FR ($p < 0.01$). The WFPS also significantly differed across months ($p < 0.01$) (Figure 2B). The average of WFPS in November 2010, December 2010 and January 2011 were significantly higher ($p < 0.01$) than in other months. With the exception of RB20, near saturated WFPS conditions (> 60 % WFPS) were reached from November 2010 till February 2011. In March 2011, near saturated condition was reached in OP and RB1 only. There were five dry months (monthly rainfall < 100 mm) over the monitoring period: August 2010, October 2010, February 2011, May 2011, and August 2011 (Figure 3). Although February 2011 was considered as dry (rainfall 85 mm month^{-1}), the high monthly WFPS reflects the cumulated 59.8 mm rainfall during the ten days preceding the WFPS measurement. Indeed monthly averages of WFPS and cumulated rainfall were linearly related ($R = 0.17$, $p < 0.001$) albeit with a low coefficient of determination ($R^2 = 0.03$).

Air temperature was also not normally distributed, but the log transformation of air temperature was. There was a significant difference in air temperature between LUS ($p < 0.01$) and months ($p < 0.01$) (Figure 2C and Table 1). Air temperature in all converted lands was significantly higher than in the forest. The annual temperature was the highest in the one year old rubber plantation, approximately 18% higher than in the forest. As the plantation aged with a denser tree canopy, the air temperature fell by about 2°C compared to that in the young plantation. The highest average air temperature was in June 2011 (30.2 °C) and was significantly different to the average air temperature in other months ($p < 0.01$). The month of May 2011 was significantly cooler (27.0°C) than all other months ($p < 0.01$), except August 2010, November 2010 and February 2011. Soil temperature was normally distributed. There was a significant difference in soil temperature between LUS ($p < 0.01$) and months ($p < 0.01$) (Figure 2D). The soil temperature in the forest was significantly lower ($p < 0.01$) than in all other LUS (Table 1). Differences in soil temperature among LUS were overall similar to differences in air temperature.

Neither NH_4^+ concentration nor its log transformation was normally distributed. The measurements were done in May, June and August 2011 after the OP had been fertilized in April 2011. There was a significant difference in NH_4^+ between LUS ($p < 0.01$), the concentrations being the lowest in the RB1 and the highest in the RB20 (Table 2). The trend between dry and wet months was not clear with no difference detected over months. The NH_4^+ content in the OP was significantly higher in the FZ than in the NFZ ($p < 0.05$) with a clear effect of fertilizer application on the concentrations measured in May.

4.1.2.Standing litter and litterfall production

Standing litter consisted of leaves, coarse litter (more than half decomposed leaves) and twigs. It was significantly higher in FR than in RB20 ($p < 0.01$) (Table 3). Litter in the OP displayed a high spatial variability with high litter standing where fronds were left to decompose and less litter elsewhere. The N mass in standing litter in FR was also significantly higher ($p < 0.01$) than that in the RB20. The lower C:N ratio in the OP as compared to other land uses might be related to the type or diversity of vegetation in this LUS and the efficiency with which it absorbs soil nutrients.

The rate of monthly litter production did not display any seasonal variation. Litterfall rate was not measured in the OP as farmers manually collect the fronds from the palms and place them to decompose on the ground. There was a significant difference in annual litterfall between LUS ($p < 0.01$; Table 4). The FR generated significantly higher annual litterfall than the DF and RB1 ($p < 0.01$). The litterfall was mainly composed of branches (16%) and leaves (83%), while fruits accounted for 1% of litterfall only. Nitrogen content in branches and leaves were between 1.07-1.37% and 1.63-2.6%, respectively. Leaves had a significantly higher N content ($p < 0.01$) than all other components (branches, flowers, fruits). The C:N ratios of the litterfall were high (> 20), which suggests slow decomposition rates in all LUS. Nitrogen inputs from litterfall to the soil were significantly higher in FR than in all other LUS. After forest disturbance, N inputs dropped by 40% in DF and were the lowest in RB1. However, as the rubber trees aged, N inputs from litterfall increased gradually to a value similar to that in DF.

4.1.3. Soil fluxes of nitrous oxide

Nitrous oxide fluxes were observed at all sites over a 14 month period (Figure 2). The fluxes varied between -20.9 and 940.6 g N ha^{-1} d^{-1}. The daily N_2O fluxes were distributed 54%, between 0 and 10 g N ha^{-1} d^{-1}, 30 % below 0 and only 16% with values over 10 g N ha^{-1} d^{-1}. Negative flux values indicate higher N_2O consumption than production (Butterbach-Bahl *et al.*, 2013). Fluxes below the negative DL represented 17% of all negative fluxes and most of them (68%) were observed in the oil palm plantation and undisturbed forest. Low daily N_2O fluxes below 0.5 g N ha^{-1} d^{-1} were measured in all months except August 2010 and December 2010. The highest and lowest monthly average of N_2O fluxes were measured in the FR in December 2010 (30.2 ± 7.17 g N ha^{-1} d^{-1}) and July 2010 (-4.0 ±

1.7 g N ha^{-1} d^{-1}), respectively (Figure 2). The differences in emissions over time were highly significant ($p < 0.01$) indicating large temporal variations. The high coefficients of variation of 320, 191, 296, 232 and 263% in the FR, DF, RB1, RB20 and OP confirm the important temporal flux variability. Contrary to the initial hypothesis, overall N_2O fluxes in the wet and dry months were not significantly different from each other. Over the 14 month monitoring period there was no significant difference in the average of N_2O fluxes among LUS or between soil suborders. There was also no significant difference in N_2O fluxes between LUS systems before, during and after fertilizer application in the OP (Table 5) even though the OP emission rate after fertilization was substantially higher than the emission rate in the other LUS. Annual N_2O emission rates amounted to 1.73 ± 0.48, 1.22 ± 0.27, 1.34 ± 0.36, 1.02 ± 0.27 and 1.04 ± 0.39 kg N ha^{-1} y^{-1} in the FR, DF, RB1, RB20 and OP, respectively. In the OP, annual N_2O emissions amounted to 3.1 ± 1.2% of the N fertilizer dose applied (33.3 kg N ha^{-1} y^{-1}). Thus with a common fertilizer application rate of 141 kg N ha^{-1} y^{-1}, split into 2 to 3 applications as normal farmer practice, the emissions in the OP would have amounted to 4.4 ± 1.6 kg N ha^{-1} y^{-1}.

4.2. Intensive measurements over the fertilizer application period in the oil palm plantation

4.2.1.Climate and soil properties

The measurements following fertilizer application in the OP site are detailed by FZ and NFZ. All soil variables (WFPS, air and soil temperatures) were normally distributed during the intensive observation period following fertilizer application. During this period, 77 % of daily rainfall data were below 10 mm, 7% were in between 10-20 mm and 17 % were more than 20 mm. The high rainfall events happened 0, 6, 17, 21 days after fertilizer application (40, 73, 35 and 63 mm day^{-1}, respectively) (Figure 4A). Overall, the WFPS was higher than 70%. There was a significant difference in WFPS over time ($p < 0.01$) and between the fertilized (FZ) and non-fertilized (NFZ) zones ($p < 0.01$) (Figure 4B). The WFPS was significantly higher 4, 5, 6 and 10 days after fertilizer application than 2 days before it. The average of WFPS in the NFZ (100.5 ± 1.4 %) was significantly higher than that in the FZ (82.9 ± 1.7 %) ($p < 0.01$). Air and soil temperature followed a similar trend (Figure 4C and 4D). Both reached their maximum 14 days after fertilizer application (air and soil temperatures of 27.4 °C; 31.5 °C, respectively). Both were similar in the NFZ and FZ.

4.2.2. Soil fluxes of nitrous oxide

Negative fluxes (N_2O consumption), probably resulting from N_2O denitrification to N_2 in high WFPS anaerobic conditions, accounted for 24% of all observations during the period with 76% of the negative fluxes occurring in the NFZ. Nonetheless only 7% of these negative fluxes fell beyond the negative flux DL. There was a significant difference in emission rate between the NFZ and FZ ($p < 0.01$) and between measurement days in the FZ ($p < 0.01$) but not in the NFZ where the flux remained steady at an average rate of 2.0 ± 0.8 g N ha^{-1} d^{-1} (Figure 4A). Soil emission rates of N_2O started increasing in the FZ 5 days after fertilizer application, reaching a maximum 17 days after application (526.8 ± 115 g N ha^{-1} d^{-1}); this rate was significantly higher ($p < 0.01$) than the average rate before fertilization. Forty four days after fertilizer application (22th May 2011), the N_2O emission rate in the FZ was still twenty three times higher than pre-fertilization rates. When extrapolated at the plot scale, cumulated N_2O emissions from day one till forty four days after fertilizer application represented as much as 65 % of annual emissions.

4.3. Determinants of soil N_2O flux changes

In all LUS, daily N_2O fluxes were negatively correlated to soil pH ($R = -0.17$, $p < 0.01$) and soil total N ($R = -0.13$, $p < 0.01$) but positively correlated to soil C:N ratio ($R = 0.16$, $p < 0.001$) and litterfall ($R = 0.12$, $p < 0.001$). After fertilizer application in the OP, daily N_2O fluxes in the FZ were positively correlated to rainfall on the day of measurement ($R = 0.52$, $p < 0.01$) while in the NFZ, they were negatively correlated to the WFPS ($R = -0.23$, $p < 0.05$). However in all cases less than 30% in the variation of daily N_2O fluxes ($R^2 < 0.30$) was explained by these linear relationships. The regression analysis (Table 6) revealed that along the gradient annual N_2O fluxes were inversely proportional to the distance to the nearest termite nest, and increased linearly with increasing rainfall on the day of measurement. Increases in air temperature led to increase N_2O emissions in the LUS only when the air temperature exceeded 29.5˚C. When the fertilized OP was omitted from the analysis, the fluxes were positively related to dry mass and N mass in standing litter. Since the emissions in the OP mostly occurred after fertilizer application, annual emissions cannot be predicted based on standing litter alone.

5. Discussion

5.1. Comparison of nitrous oxide fluxes between land use systems

Agriculture, especially when practiced using high N fertilizer inputs, is widely recognised to be a major driver of N_2O emissions (Baumert *et al.*, 2005). On the other hand, the global impacts on atmospheric N_2O concentrations of forest conversion to agricultural land remain unclear, especially when fertilizer is not applied. In Sumatra, Verchot *et al.* (2006) observed an increase in N_2O emissions along a chronosequence of forest land converted to coffee gardens, most of them unfertilized. However, in Southwest China, Werner *et al.* (2006) measured significantly lower N_2O emissions in a 20 year old rubber plantation fertilized with 55 kg N ha^{-1} y^{-1} than in a nearby forest. This observation of reduced emissions following conversion is in agreement with the findings of Ishizuka *et al.* (2002), comparing a forest and a non-fertilized rubber plantation in Sumatra. By contrast, our analysis indicates no significant difference in N_2O fluxes between undisturbed forest, disturbed forest, and either young or old rubber plantations, suggesting that the absence of effects is sustained over a long period. Conversion to oil palm plantation also did not significantly affect the emissions; however, our results indicate that the application of a conventional fertilization dose would lead to a significant increase in emissions. This is in agreement with the observations of Fowler *et al.* (2011) in a forest and a fertilized oil palm plantation in Malaysia. The absence of a significant difference in N_2O emission between LUS may also be the result of other under-lying factors. Soil type and associated properties are known to influence nitrification and denitrification, for example, Ishizuka *et al.* (2005b) found different N_2O flux rates from Ultisols and Andisols in Sumatra. The soils of the RB20 and OP were both different from the soils of the FR, DF and RB1. The RB20's soil had a higher clay and C content than the soils in other LUS; this may have promoted denitrification and associated N_2O production (Skiba and Smith, 2000). On the other hand, the lower bulk density and therefore also lower WFPS of the soil at the RB20 site may have had the opposite effect on denitrification. The decreased bulk density in the RB20 as compared to that in the FR may have resulted from the dense root architecture of the old rubber trees. A negative relationship between rubber root density and soil bulk density was demonstrated by Samarappuli *et al.* (1996).

The soil in the OP also differed from the other soils essentially in its P content, C:N ratio and pH. A higher P content generally supports lower emissions (Hall and Matson, 1999), but a lower C:N ratio promotes increased N mineralization and N_2O emissions (Bouwman *et al.*, 1993). Finally, numerous studies have demonstrated that the N_2O:N_2 ratio decreases with increasing pH (Šimek and Cooper, 2002). Along the conversion gradient N_2O emissions were, however, not correlated to the intrinsic soil properties, which differed across sites (clay, carbon and P content); instead they displayed strong relationships with variables influenced by the land cover (dry mass and N mass in standing litter, air temperature, termites).

5.2. Determinants of fluxes

The rate of nitrogen cycling through terrestrial ecosystems is known to be an important determinant of soil N_2O production and other investigators (Davidson *et al.*, 2000; Erickson *et al.*, 2001) have found a direct relationship between N_2O fluxes and litterfall N input. Along the present land use gradient, litterfall N input was not a good index of N availability because the high standing litter mass in the RB1 resulting from recent slashing and burning of the forest was not reflected by its low litterfall rate and the litter was managed in the OP for soil fertility improvement and pest management. Although the organic matter and N litter pool size are not measures of N flows, their strong relationship to N_2O fluxes in non-fertilized LUS indicates that they may also be good indicators of the N cycle. We found a non-linear relationship between air temperature and annual fluxes of N_2O (Table 6) which is driven by the annual air temperature in RB1 significantly higher than that in the other LUS (Table 1) as a result of no canopy cover.

Termites are substantial components of biologically mediated response to land-use change in the tropics (Ackerman *et al*., 2007). Their nests can be considered as nutrient hotspots (Jouquet *et al.*, 2011) and have been observed to produce consistently higher N_2O emissions than the surrounding area (Khalil *et al.*, 1990; Brümmer *et al.*, 2009). The inverse relationship between N_2O emissions and distance to the nearest termite nest confirms that termite activity is an important driver for N_2O fluxes in this area; even though we made a qualitative measurement only and did not directly address the amount of N_2O produced by termite activities.

Along the gradient, we did not find any significant relationship between N_2O emissions and the WFPS; however, annual N_2O fluxes and the average amount of rainfall on the day of measurement were positively correlated. The WFPS was correlated to mean daily N_2O emissions only in the NFZ of the OP and after fertilization. The very low determination coefficient values of the correlations between daily N_2O fluxes and average environmental parameters support the statement of Groffman et al. (2000) suggesting difficulties in establishing strong predictive relationships at this temporal scale.

Negative N_2O flux values below the 6.1 g N ha-1 d-1 DL contributed about 17% to total negative fluxes and 5% to overall fluxes, which is significant. This provides confidence that uptake of N_2O in these LUS is a real phenomenon and not a random variation within the DL of the sampling and analytical methods. The processes responsible for N_2O uptake remain unclear. However, in anaerobic, near saturated WFPS (> 60 % WFPS) conditions (as was the case here), denitrification and nitrifier denitrification are recognized as the potential processes of N_2O consumption in soils (Chapuis-Lardy *et al.*, 2007). Despite significant uptake of N_2O, the annual rate of N_2O fluxes in the FR (1.73 kg N ha^{-1} y^{-1}) was higher than observed in an earlier work in the nearby area (0.13-0.39 kg N ha^{-1} y^{-1}; Ishizuka *et al.* (2002)). The N_2O fluxes in FR were within the ranges found by Stehfest and Bouwman (2006), Kim et al.(2013 a, b) and Dalal and Allen (2008), who calculated N_2O fluxes in tropical humid forests of 0.85, 1.9 and 4.76 kg N ha^{-1} y^{-1}, respectively. The N_2O fluxes observed in DF (1.22 kg N ha^{-1} y^{-1}) were comparable to the emissions reported by Ishizuka *et al.* (2002) in a logged forest nearby our site (0.56 and 1.41 kg N ha^{-1} $y^{-}1$) and by Werner *et al.* (2006) in a Chinese secondary forest (0.64 kg N ha^{-1} y^{-1}). Annual N_2O fluxes in RB1 and RB20 (1.02 and 1.34 kg N ha^{-1} y^{-1}) were higher than emissions in the same land use reported by Werner *et al.* (2006) (0.36 kg N ha^{-1} y^{-1}) and Ishizuka *et al.* (2002) (0.06 kg N ha^{-1} y^{-1}). A report by Fowler *et al.* (2011) suggested annual emissions in an old (12 m high canopy) oil palm plantation in Malaysian Borneo of 4.4 kg N ha^{-1} y^{-1}, which is more than fourfold our finding (1.04 kg N ha^{-1} y^{-1}). The difference may be attributed to the higher N fertilizer application in the study of Fowler *et al.* (2011) (81 kg N ha^{-1} y^{-1}) than here (33.3 kg N ha^{-1} y^{-1}). Ishizuka *et al.* (2005a) found higher emission rates in young (3-5 year old) plantations as compared to older (15 year old) ones. The difference was explained by higher N inputs brought by both N fertilization and N_2-fixing legume cover grown in the inter-rows of young plantations. Although the above mentioned studies on N_2O emissions in oil palm plantations are limited, they all point towards a chief control of the emissions by N inputs. Best management practices that limit the atmospheric impact without decreasing its productivity still need to be established. For this, experimental designs monitoring concomitantly palm production and N_2O emissions at varying rates of N inputs would be most appropriate.

5.3. Post-fertilization fluxes and N_2O emission factor

The timing and magnitude of fertilization impact on N_2O emissions depend on concomitant factors, such as plant demand and uptake, climatic conditions during and post fertilization, fertilizer type and application form or soil properties. The response of a same LUS can be expected to vary substantially between two fertilizer application events even if an identical dose is applied in a consistent manner, mainly due to the response to rainfall (Dobbie and Smith, 2003; Fermont et al., 2010). However, some recommendations can be provided in order to establish an experimental design able to capture appropriately the expected pulses in emissions. For that purpose, a stratified design, separating the fluxes from the fertilized and non-fertilized areas has proven to be effective in capturing emission peaks in the fertilized zone. The method, applied in this study and several previous GHG flux studies in fertilised soils (Veldkamp and Keller, 1997; Weitz *et al.*, 2001; Hergoualc'h *et al.*, 2008; Fowler *et al.*, 2011), allows emission rates to be extrapolated at plot scale, while limiting under or over-estimation of localized fertilization effects. Capturing temporal fluctuations requires intensive sampling regimes, which are particularly difficult to do in the remote regions of the tropics. Veldkamp and Keller (1997) measured elevated fluxes during the first two weeks following application in a banana plantation; this study and one by Hergoualc'h *et al.* (2008) found that forty four days after fertilization emission rates remained higher than pre-fertilization levels; and Fowler *et al.* (2011) and Weitz *et al.* (2001) measured sustained elevated emissions over two to three months after application. Within the post-fertilization period N_2O emissions can start to increase shortly (within hours) after fertilizer application (Weitz *et al.*, 2001) or a few days later such as in this study and in the one of Fowler *et al.* (2011). Given these considerations, monitoring of fluxes is recommended in the first month following application at least every three to four days starting on the fertilization day and every week in the subsequent two months. Most tropical studies (Veldkamp and Keller, 1997; Weitz *et al.*, 2001; Werner *et al.*, 2006) observed maximum peaked emissions in the range 250-500 g N ha^{-1} d^{-1}, but short term fluxes can be as large as 2000 g N ha^{-1} d^{-1} (Fowler *et al.*, 2011). Our measurements indicated that 3.1% of N applied in the OP was emitted as

N_2O. This is much higher than the 1% emission factor proposed by the IPCC Guidelines (IPCC, 2006), but lower than the 5.5% one suggested by Fowler *et al.* (2011). In an evaluation of the IPCC emission factor, Philibert *et al.* (2012) established that the emission response to increasing N input is exponential rather than linear. They also found that the emission factor based on an exponential model was lower than the 1% IPPC emission factor when the fertilizer applied was below 160 kg N ha^{-1}. This was not the case in the present study nor in the one of Fowler *et al.* (2011), demonstrating the need to further detailed investigation of the effect that N fertilizer application has on soil N_2O emissions in oil palm plantations.

6. Conclusion

Forest degradation and conversion to rubber and oil palm plantations did not significantly alter annual N_2O emissions at our research site located in Sumatra, Indonesia. Nonetheless the high emission factor of 3.1 % of N applied emitted as N_2O in the oil palm plantation indicates that if fertilized at a conventional rate of 141 kg N ha^{-1} y^{-1}, its annual emission would be more than twice that in the forest. Given the current expansion of oil palm plantations worldwide, it is critical to promote research on the impact of forest conversion to oil palm plantations on C stocks and trace gas emissions. This research, and the very few others studying oil palm plantations, indicate that management of N inputs, brought by fertilizer application and cultivation of a N_2-fixing legume cover crops in the inter-rows of young plantations, is likely to be a key factor controlling N_2O emissions. Along the forest conversion gradient, annual fluxes were in the range of previous studies in the humid tropics, even though soil N_2O uptake was significant. The temporal variation of the fluxes was very high, but did not display any clear pattern between dry and wet months, indicating that long term and frequent monitoring is needed to provide accurate estimates.

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References

Ackerman, I.L., Teixeira, W.G., Riha, S.J., Lehmann, J., Fernandes, E.C.M., 2007. The impact of mound-building termites on surface soil properties in a secondary forest of Central Amazonia. Applied Soil Ecology 37, 267-276.

Alexopoulos, E.C., 2010. Introduction to multivariate regression analysis. Hipokratia 14, 23-28.

Arai, S., Ishizuka, S., Ohta, S., Ansori, S., Tokuchi, N., Tanaka, N., Hardjono, A., 2008. Potential N_2O emissions from leguminous tree plantation soils in the humid tropics. Global Biogeochem Cy 22, n/a-n/a.

Baumert, K., Herzog, T., Pershing, J., 2005. Navigating the Numbers: Greenhouse Gas Data and International Climate Policy. World Resources Institute (WRI), USA.

Black, C.A., 1965. Methods of Soil Analysis, Part 2, Agronomy 9, American Society of Agronomy. Madison,Wis.

Bouwman, A.F., Boumans, L.J.M., Batjes, N.H., 2002. Emissions of N_2O and NO from fertilized fields: Summary of available measurement data. Global Biogeochem Cy 16, 6-1-6-13.

Bouwman, A.F., Fung, I., Matthews, E., John, J., 1993. Global analysis of the potential for N_2O production in natural soils. Global Biogeochem Cy 7, 557-597.

Bray, R.H., Kurtz, L.T., 1945. Determination of total, organic, and available forms of phosphorus in soils. Soil Science 59, 39-46.

Bremner, J.M., Blackner, A.M., 1978. Nitrous oxide: Emissions from soils during nitrification of fertilizer nitrogen. Science 199, 295-296.

Brockway, L.H., 1978. Science and colonial expansion: the role of the British Royal Botanic Gardens. American Ethnologis 6, 449-465.

Brümmer, C., Papen, H., Wassmann, R., Brüggemann, N., 2009. Termite mounds as hot spots of nitrous oxide emissions in South-Sudanian savanna of Burkina Faso (West Africa). Geophysical Research Letters 36 (9).

Butterbach-Bahl, K., Baggs, E.M., Dannenmann, M., Kiese, R., Zechmeister-Boltenstern, S., 2013. Nitrous oxide emissions from soils: How well do we understand the processes and their controls? Phil. Trans. R. Soc. B 368, 20130122.

Chapuis-Lardy, L., Wrage, N., Metay, A., Chotte, J.-L., Bernoux, M., 2007. Soils, a sink for N_2O? A review. Glob Change Biol 13, 1-17.

Comte, I., Colin, F., Whalen, J.K., Grünberger, O., Caliman, J.-P., 2012. Agricultural practices in oil palm plantations and their impact on hydrological changes, nutrient fluxes and water quality in Indonesia: A review. Advances in Agronomy 116, 71-124.

Conti, M.E., Iacobucci, M., Cecchetti, G., 2005. A statistical approach applied to trace metal data from biomonitoring studies. International Journal Environment and Pollution 23, 29-41.

Cowan, N.J., Famulari, D., Levy, P.E., Anderson, M., Reay, D.S., Skiba, U.M., 2014. Investigating uptake of N_2O in agricultural soils using a high-precision dynamic chamber method. Atmos. Meas. Tech. 7, 4455-4462.

Dalal, R. C., & Allen, D. E., 2008. TURNER REVIEW No. 18. Greenhouse gas fluxes from natural ecosystems. Australian Journal of Botany 56, 369-407.

Davidson, E.A., Keller, M., Erickson, H.E., Verchot, L.V., Veldkamp, E., 2000. Testing a conceptual model of soil emissions of nitrous and nitric oxides. BioScience 50, 667- 680.

Davidson, E.A., Matson, P.A., Vitousek, P.M., Riley, R., Dunkin, K.G., García-Méndez, J., Maass, M., 1993. Processes regulating soil emissions of NO and N_2O in a seasonally dry tropical forest. Ecology 74, 130-139.

Dobbie, K.E., Smith, K.A., 2003. Nitrous oxide emission factors for agricultural soils in Great Britain: the impact of soil water ‐filled pore space and other controlling variables. Glob. Change Biol. 9, 204–218.

Erickson, H., Keller, M., Davidson, E.A., 2001. Nitrogen oxide fluxes and nitrogen cycling during post agricultural succession and forest fertilization in humid tropics. Ecosystems 4, 67-84.

Eyduran, E., Özdemir, T., Alarslan, E., 2005. Important of diagnostics in multiple regression analysis. Journal of Applied Sciences 8, 1792-1796.

Fermont, A.M., Tittonell, P.A., Baguma, Y., Ntawuruhunga, P., Giller, K.E., 2010. Towards understanding factors that govern fertilizer response in cassava: lessons from East Africa. Nutr. Cycl. Agroecosyst. 86, 133–151. doi:10.1007/s10705-009-9278-3.

Foster, P., V. Ramaswamy, P. Artaxo, T. Berntsen, R. Betts, D.W. Fahey, J. Haywood, J. Lean, D.C. Lowe, G. Myhre, J. Nganga, R. Prinn, G. Raga, M. Schulz and R. Van Dorland., 2007. Changes in atmospheric constituents and in radiative forcing. In: Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M.Tignor and H.L. Miller (Ed.), Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge.

Fowler, D., Nemitz, E., Misztal, P., Di Marco, C., Skiba, U., Ryder, J., Helfter, C., Cape, J.N., Owen, S., Dorsey, J., Gallagher, M.W., Coyle, M., Phillips, G., Davison, B., Langford, B., MacKenzie, R., Muller, J., Siong, J., Dari-Salisburgo, C., Di Carlo, P., Aruffo, E., Giammaria, F., Pyle, J.A., Hewitt, C.N., 2011. Effects of land use on surface–atmosphere exchanges of trace gases and energy in Borneo: Comparing fluxes over oil palm plantations and a rainforest. Phil. Trans. R. Soc. B 366, 3196-3209.

Gan, F.F., Koehler, K.J., 1990. Goodness-of-fit test based on P-P probability plots. Technometrics 32, 289-303.

Grimaldi, M., Schroth, G., Teixeira, W.G., Huwe, B., 2003. Soil Structure. CABI Publishing, Bristol, United Kingdom, pp. 191–208.

Groffman, P.M., Brummer, R., Butterbach-Bahl, K., Dobbie, K.E., Mosier, A.R., Ojima, D., Papen, H., Parton, W. J., Smith, K. A., Wagner-Riddle, C., 2000. Evaluating annual nitrous oxide fluxes at ecosystems scale. Global Biogeochem Cy 14, 1061-1070.

Gundersen, P., Christiansen, J.R., Alberti, G., Brüggemann, N., Castaldi, S., Gasche, R., Kitzler, B., Klemedtsson, L., Lobo-do-Vale, R., Moldan, F., Rütting, T., Schleppi, P., Weslien, P., Zechmeister-Boltenstern, S., 2012. The response of methane and nitrous oxide fluxes to forest change in Europe. Biogeosciences 9, 3999-4012.

Hairiah, K., Dewi, S., Agus, F., Ekadinata, A., Rahayu, S., van Noordwijk, M., 2010. Measuring Carbon Stock Across Land Use Systems: A Manual. World Agroforestry Centre (ICRAF), SEA Regional Office, Bogor, Indonesia.

Hairiah, K., Sulistyani, H., Suprayogo, D., Widianto, Purnomosidhi, P., Widodo, R.H., Van Noordwijk, M., 2006. Litter layer residence time in forest and coffee agroforestry systems in Sumberjaya, West Lampung. Forest Ecology and Management 224, 45–57.

Hall, S.J., Matson, P.A., 1999. Nitrogen oxide emissions after N additions in tropical forests. Nature 400, 152-155.

Henríquez, C., Cabalceta, G., 1999. Guía práctica para el estudio introductorio de los suelos con un enfoque agrícola. ACCS (Asociación Costarricense de la Ciencia del Suelo), San Jose.

Hergoualc'h, K., 2011. Principles and methods for assessing climate change mitigation as an ecosystem service in agroecosystems. In: Rapidel, B., DeClerck, F., Le Coq, J.F., Beer, J. (Eds.), Ecosystem Services from Agriculture and Agroforestry: Measurement and Payment. Earthscan, pp. 19-36.

Hergoualc'h, K., Verchot, L.V., 2011. Stocks and fluxes of carbon associated with land use change in Southeast Asian tropical peatlands: A review. Global Biogeochem Cy 25, n/a-n/a.

Hergoualc’h, K., Skiba, U., Harmand, J.-M., Hénault, C., 2008. Fluxes of greenhouse gases from Andosols under coffee in monoculture or shaded by Inga densiflora in Costa Rica. Biogeochemistry 89, 329-345.

Hesse, P.R., 1971. A Textbook of Soil Chemical Analysis. Chemical Publishing Co.,Inc., New York.

IPCC, 2006. 2006 IPCC Guidelines for National Gas Inventory: Agriculture, Forestry, and Other Land Use. IPCC/IGES, Hayama, Japan.

Ishizuka, S., Iswandi, A., Nakajima, Y., Yonemura, S., Sudo, S., Tsuruta, H., Murdiyarso, D., 2005a. The variation of greenhouse gas emissions from soils of various land-use/cover types in Jambi province, Indonesia. Nutrient Cycling in Agroecosystems 71, 17-32.

Ishizuka, S., Iswandi, A., Nakajima, Y., Yonemura, S., Sudo, S., Tsuruta, H., Muriyarso, D., 2005b. Spatial patterns of greenhouse gas emission in a tropical rainforest in Indonesia. Nutrient Cycling in Agroecosystems 71, 55-62.

Ishizuka, S., Tsuruta, H., Murdiyarso, D., 2002. An intensive field study on CO_2, CH_4, and N_2O emissions from soils at four land-use types in Sumatra, Indonesia. Global Biogeochem Cy 16, 22(1)-22(11).

Itoh, M., Kosugi, Y., Takanashi, S., Hayashi, Y., Kanemitsu, S., Osaka, K., Tani, M., Nik, A.R., 2010. Temporal and spatial variations of soil carbon dioxide, methane, and nitrous oxide fluxes in a Southeast Asian tropical rainforest. Biogeosciences Discussions 7, 6847-6887.

Ji, R., Brune, A., 2006. Nitrogen mineralization, ammonia accumulation, and emission of gaseous NH_3 by soil-feeding termites. Biogeochemistry 78, 267-283.

Jouquet, P., Traoré, S., Choosai, C., Hartmann, C., Bignell, D., 2011. Influence of termites on ecosystem functioning: Ecosystem services provided by termites. European Journal of Soil Biology 47, 215-222.

Karhu, K., Mattila, T., Bergström, I., Regina, K., 2011. Biochar addition to agricultural soil increased CH_4 uptake and water holding capacity – Results from a short-term pilot field study. Agriculture, Ecosystems & Environment 140, 309-313.

Keller, M., Veldkamp, E., Weitz, A.M., Reiners, W.A., 1993. Effect of pasture age on soil trace-gas emissions from a deforested area of Costa Rica. Nature 36, 244-246.

Ketterings, Q.M., Tri Wibowo, T., van Noordwijk, M., Penot, E., 1999. Farmers' perspectives on slash-and-burn as a land clearing method for small-scale rubber producers in Sepunggur, Jambi Province, Sumatra, Indonesia. Forest Ecology and Management 120, 157-169.

Khalil, M.A.K., Rasmussen, R.A., French, J.R.J., Holt, J.A., 1990. The influence of termites on atmospheric trace gases: CH_4, CO_2, $CHCl_3$, N_2O, CO, H_2, and light hydrocarbons. J. Geophys. Res. 95, 3619-3634.

Khasanah, N., Ekadinata, A., Rahayu, S., van Noordwijk, M., Ningsih, N., Setiawan, A., Dwiyanti, E., Dewi, S., Octaviani, R., 2011. Carbon Footprint of Indonesian, Palm Oil Production. Oil Palm flyer no. 1. World Agroforestry Centre (ICRAF) Southeast Asia Program, Bogor, Indonesia.

Kim, D.-G., Giltrap, D., Hernandez-Ramirez, G., 2013a. Background nitrous oxide emissions in agricultural and natural lands: a meta-analysis. Plant and Soil 373, 17-30.

Kim, D.G., Giltrap, D., Hernandez-Ramirez, G., 2013b. Erratum to: Background nitrous oxide emission in agricultural and natural lands: A meta-analysis. Plant and Soil 373, 1007-1008.

Laumonier, Y., Edin, A., Kanninen, M., Munandar, A.W., 2010. Landscape-scale variation in the structure and biomass of the hill dipterocarp forest of Sumatra: Implications for carbon stock assessments. Forest Ecology and Management 259, 505-513.

Linn, D.M., Doran, J.W., 1984. Effect of water-filled pore spaces on carbon dioxide and nitrous oxide production in tilled and non-tilled soils. Soil Sci. Soc. Am. J. 48, 1267-1272.

Lo, E., 2005. Gaussian error propagation applied to ecological data: Post-ice-storm-downed woody biomass. Ecological Monograph 75, 451-466.

Loftfield, N., Flessa, H., Augustin, J., Beese, F., 1997. Automated gas chromatographic system for rapid analysis of the atmospheric trace gases methane, carbon dioxide, and nitrous oxide. Journal of Environmental Quality 26, 560–564.

Malhi, Y., AragÃO, L.E.O.C., Metcalfe, D.B., Paiva, R., Quesada, C.A., Almeida, S., Anderson, L., Brando, P., Chambers, J.Q., da Costa, A.C.L., Hutyra, L.R., Oliveira, P., PatiÑO, S., Pyle, E.H., Robertson, A.L., Teixeira, L.M., 2009. Comprehensive assessment of carbon productivity, allocation and storage in three Amazonian forests. Glob Change Biol 15, 1255-1274.

Margono, B.A., Turubanova, S., Zhuravleva, I., Potapov, P., Tyukavina, A., Baccini, A., Goetz, S., Hansen, M.C., 2012. Mapping and monitoring deforestation and forest degradation in Sumatra (Indonesia) using Landsat time series data sets from 1990 to 2010. Environmental Research Letters 7, 034010.

Miettinen, J., Shi, C., Liew, S.C., 2011. Deforestation rates in insular Southeast Asia between 2000 and 2010. Glob Change Biol 17, 2261-2270.

Miyamoto, M., 2006. Forest conversion to rubber around Sumatran villages in Indonesia: Comparing the impacts of road construction, transmigration projects and population. Forest Policy and Economics 9, 1-12.

Moncrieff , J.B., Malhi, Y., Leuning, R., 1996. The propagation of errors in long-term measurements of land-atmosphere fluxes of carbon and water. Glob Change Biol 2, 231-240.

Murdiyarso, D., Wasrin, U.R., 1995. Estimating land use change and carbon release from tropical forests conversion using remote sensing technique. Terrestrial Ecosystem Interaction with Global Change 2, 715-721.

Ngugi, D.K., Brune, A., 2012. Nitrate reduction, nitrous oxide formation, and anaerobic ammonia oxidation to nitrite in the gut of soil-feeding termites (*Cubitermes* and *Ophiotermes* spp.). Environmental microbiology 14, 860–871.

Obidzinski, K., Andriani, R., Komarudin, H., Andrianto, A., 2012. Environmental and social impacts of oil palm plantations and their implications for biofuel production in Indonesia. Ecol. Soc. 17. doi:10.5751/ES-04775-170125.

Ohkuma, M., Noda, S., Kudo, T., 1999. Phylogenetic diversity of nitrogen fixation genes in the symbiotic microbial community in the gut of diverse termites. Appl. Environ. Microbiol. 65, 4926-4934.

Pansu, M., Gautheyrou, J., 2006. Handbook of Soil Analysis. Mineralogical, Organic and Inorganic Methods. Springer, Berlin Heidelberg New York.

Park, M.H., 2008. Univariate Analysis and Normality Test Using SAS, Stata, and SPSS. The University Information Technology Services (UITS) Center for Statistical and Mathematical Computing, Indiana University, Indiana.

Parkin, T. B., Venterea, R. T., and Hargreaves, S. K. 2012. Calculating the detection limits of chamber-based soil greenhouse gas flux measurements..Journal of Environmental Quality,41 (3), 705-715.

Parton, W.J., Mosier, A.R., Ojima, D.S., Valentine, D.W., Schimel, D.S., Weier, K., Kulmala, A.E., 1996. Generalized model for N_2 and N_2O production from nitrification and denitrification. Global Biogeochem Cy 10, 401-412.

Philibert, A., Loyce, C., Makowski, D., 2012. Quantifying uncertainties in N_2O emission due to N fertilizer application in cultivated areas. PloS one 7, e50950.

Röver, M., Heinemeyer, O., Munch, J.C., Kauser, E.-A., 1999. Spatial heterogeneity within the plough layer: High variability of N_2O emissions rates. Soil Biology and Biochemistry 31, 167-173.

Samarappuli, L., Yogaratnam, N., Karunadasa, P., Mitrasena, U., 1996. Root development in *Hevea brasiliensis* in relation to management practices. Journal of the Rubber Institute of Sri Lanka 77, 93-111.

Šimek, M., Cooper, J.E., 2002. The influence of soil pH on denitrification: Progress towards the understanding of this interaction over the last 50 years. European Journal of Soil Science 53, 345-354.

Skiba, U., Smith, K.A., 2000. The control of nitrous oxide emissions from agricultural and natural soils. Chemosphere-Global Change Science 2, 379-386.

Stehfest, E., Bouwman, L., 2006. N_2O and NO emission from agricultural fields and soils under natural vegetation: Summarizing available measurement data and modeling of global annual emissions. Nutrient Cycling in Agroecosystems 74, 207-228.

Stolle, F., Chomitz, K.M., Lambin, E.F., Tomich, T.P., 2003. Land use and vegetation fire in Jambi Province, Sumatra, Indonesia. Forest Ecology and Management 179, 277-292.

Van Groenigen, J.W., Velthof, G.L., Oenema, O., Van Groenigen, K.J., Van Kessel, C., 2010. Towards an agronomic assessment of N_2O emissions: A case study for arable crops. European Journal of Soil Science 61, 903-913.

Veldkamp, E., Keller, M., 1997. Nitrogen oxide emissions from a banana plantation in the humid tropics. Journal of Geophysical Research: Atmospheres (1984–2012) 102 (D13), 15889-15898.

Veldkamp, E., Purbopuspito, J., Corre, M.D., Brumme, R., Murdiyarso, D., 2008. Land use change effects on trace gas fluxes in the forest margins of Central Sulawesi, Indonesia. Journal of Geophysical Research 113, 1-11.

Verchot, L.V., Davidson, E.A., Cattânio, J.H., Ackerman, I.L., 2000. Land-use change and biogeochemical controls of methane fluxes in soils of Eastern Amazonia. Ecosystems 3, 41-56.

Verchot, L.V., Davidson, E.A., Cattânio, J.H., Ackerman, I.L., Erickson, H.E., Keller, M., 1999. Land use change and biogeochemical controls of nitrogen oxide emissions from soils in eastern Amazonian. Global Biogeochem Cy 13, 31-46.

Verchot, L.V., Hutabarat, L., Hairiah, K., van Noordwijk, M., 2006. Nitrogen availability and soil N_2O emissions following conversion of forests to coffee in southern Sumatra. Global Biogeochem Cy 20, n/a-n/a.

Vermeulen, S., Goad, N., 2006. Towards Better Practice in Smallholder Palm Oil Production. International Institute for Environment and Development, London, UK. doi:10.1017/S001447970700542X.

Vitousek, P.M., Peter, M., Aber, J.D., Howard, R.W., Likens, G.E., Matson, P.A., Schindler, D.W., Schlesinger, W.H., Tilman, D.G., 1997. Human alteration of the global nitrogen cycle: sources and consequences. Ecol. Appl. 7, 737–750.

Wasrin, U.R., Arief, M., Setiabudhi, 1999. Spectral and textural characteristics of the lowland tropical rainforest of Jambi, Sumatra. Biotropica 13, 18-36.

Weitz, A.M., Linder, E., Frolking, S., Crill, P.M., Keller, M., 2001. N_2O emissions from humid tropical agricultural soils: effects of soil moisture, texture and nitrogen availability. Soil Biology and Biochemistry 33, 1077-1093.

Werner, C., Zheng, X., Tang, J., Xie, B., Liu, C., Kiese, R., Butterbach-Bahl, K., 2006. N_2O, CH_4 and CO_2 emissions from seasonal tropical rainforests and a rubber plantation in Southwest China. Plant and Soil 289, 335-353.

Weymann, D., Well, R., Flessa, H., von der Heide, C., Deurer, M., Meyer, K., Konrad, C., Walther, W., 2008. Groundwater N_2O emission factors of nitrate-contaminated aquifers as derived from denitrification progress and N_2O accumulation. Biogeosciences 5, 1215-1226.

Wilcove, D.S., Koh, L.P., 2010. Addressing the threats to biodiversity from oil-palm agriculture. Biodivers Conserv 19, 999-1007.

WMO, 2011. The state of greenhouse gas es in the atmosphere based on global observation through 2010. Greenhouse Gas Bulletin 7, 1-4.

Yashiro, Y., Kadir, W.R., Okuda, T., Koizumi, H., 2008. The effects of logging on soil greenhouse gas (CO_2, CH_4, N_2O) flux in a tropical rain forest, Peninsular Malaysia. Agricultural and Forest Meteorology 148, 799-806.

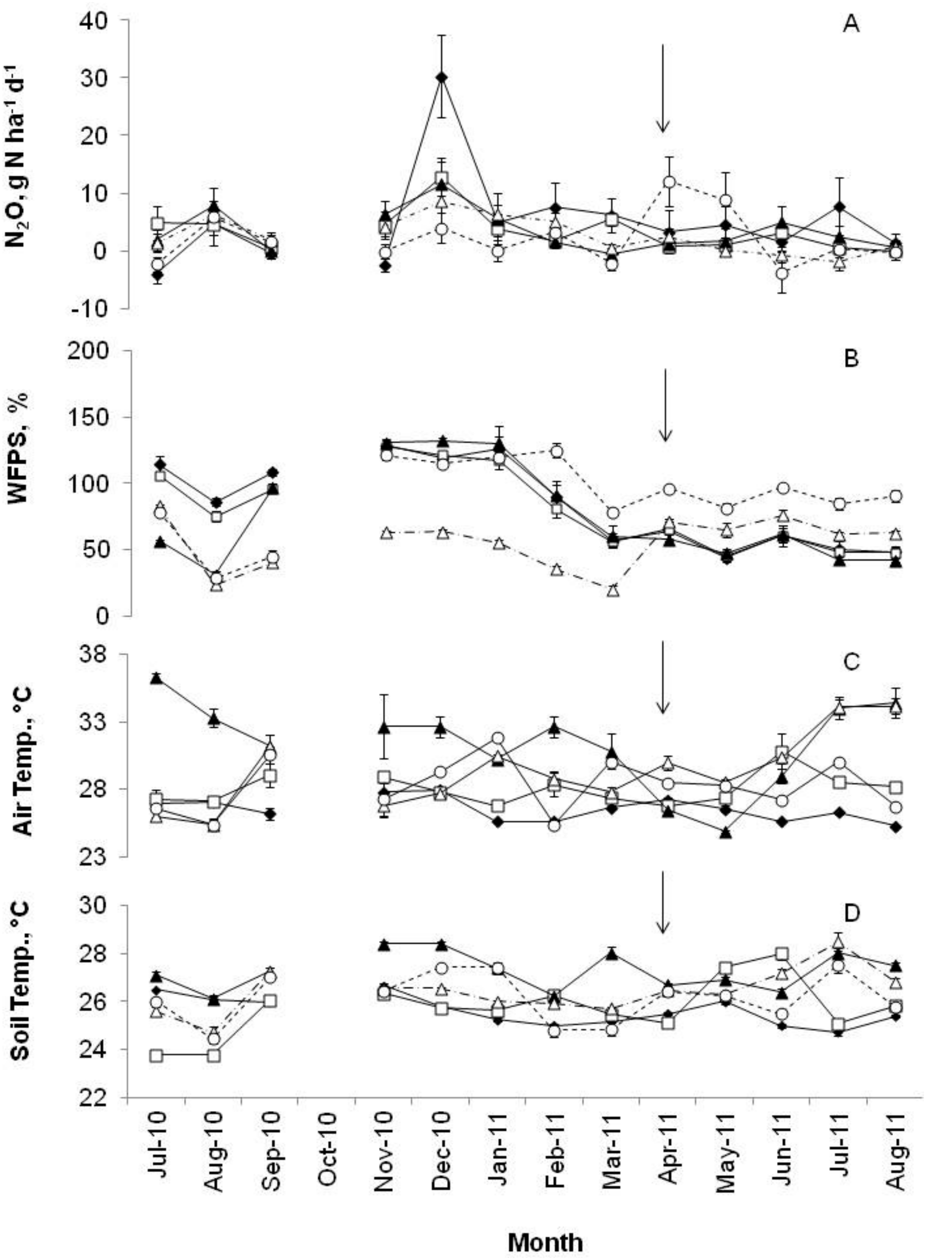

Figure 2. Average and SE of monthly N_2O fluxes (A), WFPS (B), air (C) and soil temperature (D) in the forest (FR= solid line, solid diamond), disturbed forest (DF= solid line, open square), one year old rubber plantation (RB1= solid line, solid triangle), twenty year old rubber plantation (RB20= dashed line, open triangle), and eight year old oil palm plantation (OP= dashed line, open circle) at Pasir Mayang, Jambi, Sumatra, Indonesia. The arrow indicates the fertilization in the OP.

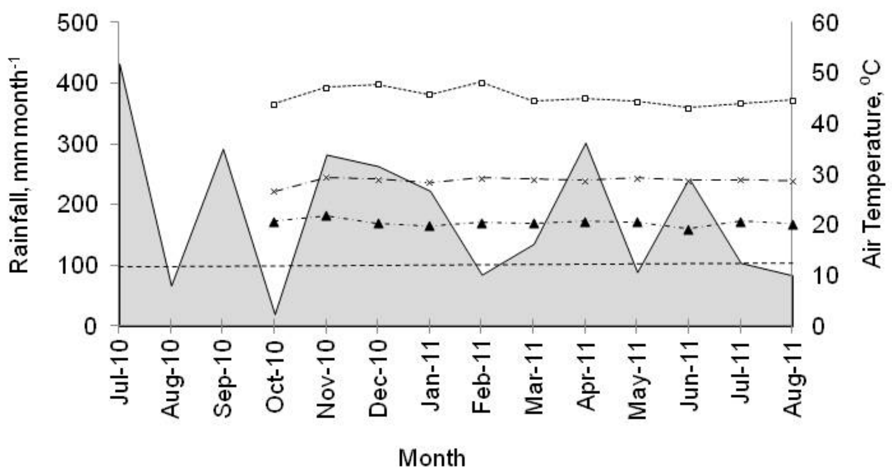

Figure 3. Monthly rainfall (grey area, mm), maximum (dashed line, open circle), minimum (dashed line, closed triangle) and average air temperature (dashed line, cross) at the study site in Pasir Mayang, Jambi, Sumatra, Indonesia. The dashed line without symbol indicates the limit above which a month is considered as wet.

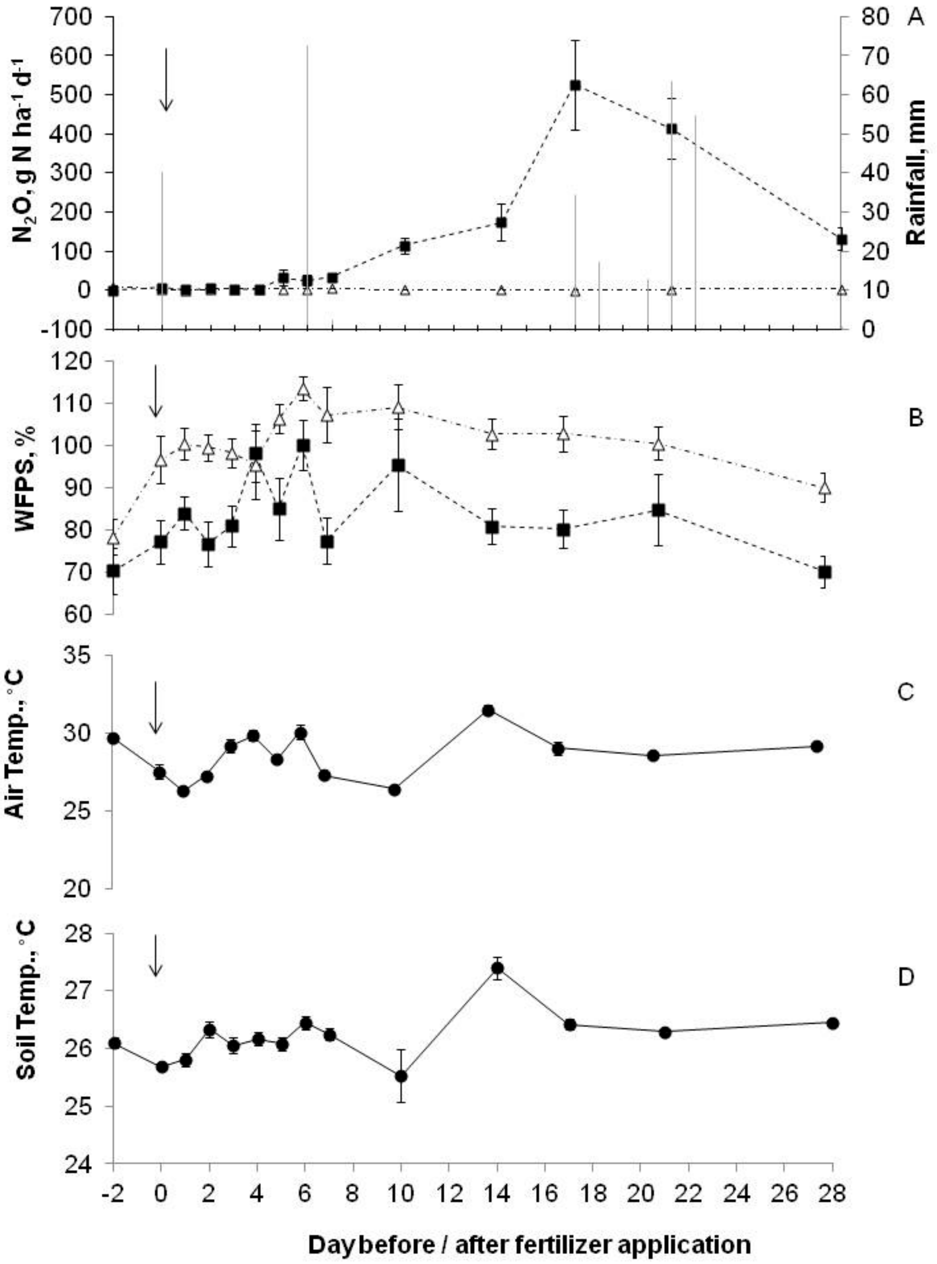

Figure 4. Daily N_2O fluxes following fertilizer application in the fertilized (FZ= dashed line, solid square) and non-fertilized (NFZ= dashed line, open triangle) zones and rainfall (A), WFPS in FZ and NFZ (B), air temperature (C) and soil temperature (D). For (C), (D) average values of the FZ and NFZ are presented. The arrow indicates the fertilization date in the OP.