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Soil nitrogen oxide fluxes from lowland forests converted to smallholder rubber and oil palm plantations in Sumatra, Indonesia

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Abstract. Oil palm and rubber plantations cover large areas of former rainforest in Sumatra, Indonesia, supplying the global demand for these crops. Although forest conversion is known to influence soil nitrous oxide (N_2O) and nitric oxide (NO) fluxes, measurements from oil palm and rubber plantations are scarce (for N_2O) or nonexistent (for NO). Our study aimed to (1) quantify changes in soil-atmosphere fluxes of N-oxides with forest conversion to rubber and oil palm plantations, and (2) determine their controlling factors. In Jambi, Sumatra, we selected two landscapes that mainly differed in texture but both on heavily weathered soils: loam and clay Acrisol soils. Within each landscape, we investigated lowland forest, rubber trees interspersed in secondary forest (termed as *jungle rubber*), both as reference land uses, and smallholder rubber and oil palm plantations, as converted land uses. Each land use had four replicate plots within each landscape. Soil N_2O fluxes were measured monthly from December 2012 to December 2013, and soil NO fluxes were measured four times between March and September 2013. In the loam Acrisol landscape, we also conducted weekly to bi-weekly soil N_2O flux measurements from July 2014 to July 2015 in a large-scale oil palm plantation with four replicate plots for comparison with smallholder oil palm plantations. Land-use conversion to smallholder plantations had no effect on soil N-oxide fluxes (P = 0.58 to 0.76) due to the generally low soil N availability in the reference land uses that further decreased with land-use conversion. Over one-year measurements, the temporal patterns of soil N-oxide fluxes were influenced by soil mineral N and water contents. Across landscapes, annual soil N_2O emissions were controlled by gross nitrification and sand content, which also suggest the influence of soil N and water availability. Soil N_2O fluxes (μg N m^{-2} h^{-1}) were: 7 ± 2 to 14 ± 7 (reference land uses), 6 ± 3 to 9 ± 2 (rubber), 12 ± 3 to 12 ± 6 (smallholder oil palm), and 42 ± 24 (large-scale oil palm). Soil NO fluxes (μg N m^{-2} h^{-1}) were: -0.6 ± 0.7 to 5.7 ± 5.8 (reference land uses), -1.2 ± 0.5 to -1.0 ± 0.2 (rubber) and -0.2 ± 1.2 to 0.7 ± 0.7

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(smallholder oil palm). The low N fertilizer application in smallholder oil palm plantations (commonly 48 to 88 kg N ha^{-1} yr^{-1}) resulted in N-oxide losses of only 0.2–0.7 % of the applied N. To improve estimate of soil N-oxide fluxes from oil palm plantations in this region, studies should focus on large-scale plantations (which usually have two to four times higher N fertilization rates than smallholders) with frequent measurements following fertilizer application.

1 Introduction

Expansion of industrial forestry and agriculture has caused rapid deforestation in Sumatra, Indonesia, resulting in a total primary forest loss of 36 % between 1990 and 2010 (Margono et al., 2012). Nowadays, most accessible lowland rainforests have been converted (Laumonier et al., 2010) into economically important crops, such as oil palm (*Elaeis guineensis*) and rubber (*Hevea brasiliensis*), with an area of 9.2 million hectare (Mha) (BPS, 2016a). Indonesia is currently the principal oil palm producer and second largest rubber producer worldwide (FAO, 2016), and Sumatra is the most important contributor to the Indonesian production (BPS, 2016b). Despite the extent of land-use change in Sumatra, it is still uncertain how forest conversion will affect soil emissions of climate-relevant N-oxide gases, nitrous oxide (N_2O) and nitric oxide (NO). Only a few studies so far have reported soil N_2O fluxes from forest conversion to these rapidly increasing and economically important land uses, oil palm and rubber, on lowland mineral soils in Southeast Asia (Aini et al., 2015; Ishizuka et al., 2002, 2005; Yashiro et al., 2008) and no study exists on soil NO fluxes.

Tropical forest soils are major sources of N_2O and NO, emitting 1.3 Tg N_2O-N yr^{-1} (Werner et al., 2007) and 1.3 Tg NO-N yr^{-1} (Davidson and Kingerlee, 1997) to the atmosphere, whereby considerable amounts of NO are expected to get redirected in forest

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systems since NO is easily oxidized to NO_2 which, in turn, is absorbed by leaves (Jacob and Bakwin, 1991; Sparks et al., 2001). N_2O is a potent greenhouse gas (IPCC, 2013) and is projected to be the single most important ozone-depleting substance throughout the 21st century (Ravishankara et al., 2009). NO plays an important role in the formation of tropospheric ozone, which in itself is an important greenhouse gas (Lammel and Graßl, 1995). N_2O and NO are produced in soil by the microbial processes of nitrification and denitrification. The conceptual model of "hole-in-the-pipe" (HIP), which had been validated by studies in the tropics (Davidson et al., 2000), suggests that production and consumption of these gases in soils are influenced by two levels of control: first, the amount of soil available N, and second, the soil water content. HIP suggests that the higher the soil N availability, the higher are the soil N-oxide fluxes, and that well-aerated soil conditions (low moisture contents) favor for nitrification with NO as the main gaseous product while with increasing water content denitrification with increasing proportion of N_2O prevails (Davidson et al., 2000). Although there are other factors affecting soil N_2O and NO fluxes through their influence on nitrification and denitrification (e.g., soil pH, temperature, bioavailable carbon; Firestone and Davidson, 1989; Heinen, 2006; Skiba and Smith, 2000), landscape-scale investigations in tropical areas show the dominant role of soil N availability and water content (Corre et al., 2014; Koehler et al., 2009; Müller et al., 2015).

Conversion of tropical forests to agricultural land uses generally alters soil N-oxide fluxes through their effects on soil N availability and aeration as a consequence of management practices (e.g., fertilization, harvest, cultivation), which can add and export nutrients as well as compact or loosen the soil (Keller and Reiners, 1994; Veldkamp et al., 2008). In particular, the application of N-containing fertilizers can increase N-oxide emissions (Matson et al., 1996; Veldkamp et al., 1998) whereas agricultural land uses without fertilizer application lead to long-term reductions of soil N-oxide fluxes or to comparably low-level

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fluxes as those from previous forests (Ishizuka et al., 2005; Keller and Reiners, 1994; Verchot et al., 1999). In tropical regions, it has been shown that soil NO and N_2O emissions can be very high following fertilizer application, constituting 6.4–8.6 % of applied N fertilizer especially at high fertilizer application rates (Veldkamp and Keller, 1997; Veldkamp et al., 1998).

For lowland forests on highly weathered soils in Sumatra, Indonesia, where our present study was conducted, it has been shown that soil N availability (with gross rates of ammonium (NH_4^+) transformations as indices) is higher in the clay than loam Acrisol soils (Allen et al., 2015), suggesting that soil texture controls soil fertility which in turn affects plant productivity, soil water holding capacity, decomposition and ultimately soil-N cycling (Allen et al., 2015). Conversion of lowland forest and jungle rubber to oil palm and rubber on these Acrisol soils showed intermediate soil N availability in oil palm plantations, due to abatement of soil fertility decline by low to moderate applications of fertilizers and lime, whereas the unfertilized rubber plantations displayed the lowest soil N availability and fertility in general (Allen et al., 2015).

Our present study focuses on soil N_2O and NO fluxes from a region in Jambi, Sumatra where increased deforestation for rubber and oil palm production has occurred in the last two decades. We covered four different land uses within two landscapes on highly weathered soils that mainly differed in soil texture (clay and loam Acrisols): forest, rubber trees interspersed in secondary forest (hereafter called jungle rubber) as the reference land uses, and smallholder rubber and oil palm plantations as the converted land uses. Based on the above mentioned findings on soil N availability, we hypothesized that (1) soil N_2O and NO fluxes from the reference land uses will be higher in the clay than the loam Acrisol landscape, and that (2) forest and jungle rubber will have the highest soil N_2O and NO fluxes, followed by the fertilized oil palm plantations (fertilized at low to moderate rates), and with the lowest fluxes

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from the unfertilized rubber plantations. Our study aimed to (1) quantify changes in soil-atmosphere fluxes of N-oxides with forest conversion to smallholder oil palm and rubber plantations, (2) determine the temporal controls of soil N-oxide fluxes measured within one year, and (3) assess landscape-scale controlling factors of annual soil N_2O fluxes from converted lowland landscapes in Sumatra, Indonesia. We also investigated the effect of fertilizer application intensity in oil palm plantations on soil N_2O fluxes by comparing smallholder plantations with low to moderate N fertilizer input to a large-scale oil palm plantation with high N fertilizer input. Our study contributes to the much needed information on soil N-oxide fluxes from these economically and globally relevant tropical land uses.

2 Material and methods

2.1 Study area, experimental design and management practices

The study region is situated in Jambi province, Sumatra, Indonesia (2° 0’ 57” S, 103° 15’ 33” E, and elevation of 73 ± 3 m above sea level), where conversion of forest to rubber and oil palm plantations is widespread. The area has a mean annual temperature of 26.7 ± 0.1 °C and a mean annual precipitation of 2235 ± 385 mm (1991–2011; data from a climatological station at the Jambi Sultan Thaha Airport). During our study year (2013), annual rainfall in the study region was 3418-3475 mm (data from climatological stations at the Harapan Forest Reserve, Sarolangun and Lubuk Kepayang, approximately 10–20 km from our sites), which were higher than the long term average. Total dissolved N deposition via rainfall was between 12.9 ± 0.1 and 16.4 ± 2.6 kg N ha^{-1} yr^{-1}, measured at two locations in the study region during 2013 (Kurniawan, 2016).

We delineated the study region in two landscapes, which have the same highly weathered soil group but mainly differed in soil texture: clay and loam Acrisol soils. The clay Acrisol soil had larger pH (4.5 ± 0.0), base saturation (23 ± 6 %) and Bray-extractable P (1.4

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± 0.1 g P m^{-2}) and lower Al saturation (61 ± 3 %) in the top 10 cm depth compared to the loam Acrisol soil (4.3 ± 0.0 pH, 11 ± 1 % base saturation, 0.5 ± 0.1 g P m^{-2} and 80 ± 1 % Al saturation) (all $P \leq 0.05$; Allen et al., 2015). Within each landscape, we investigated four land-use types: lowland forest, jungle rubber, both as the reference land uses, and smallholder monoculture plantations of rubber and oil palm, as the converted land uses. Each land use within each landscape had four sites as replicates, and we laid out a 50 m × 50 m plot in each replicate site; in total we had 32 plots. Within each plot, a 10 × 10 grid was established and we randomly selected four subplots (5 m × 5 m each) per plot, each with one permanently installed chamber base for measurements of soil N-oxide fluxes. All measurements (see Sect. 2.2) were conducted in 2013. A more detailed description of the study sites and plot design was reported earlier by Allen et al. (2015) and Hassler et al. (2015).

In the loam Acrisol landscape, we conducted additional measurements in a large-scale oil palm plantation (called PTPN VI) from 2014 to 2015 in order to compare with the smallholder oil palm plantations within the same landscape. In the PTPN VI site, we selected four replicates at a distance of 50 m apart. At each replicate, we installed three permanent chamber bases at 0.8 m, 2.8 m and 4.8 m from the tree base, in order to characterize possible spatial variation caused by management practices within each replicate.

Based on our interviews with the smallholders, the monoculture plantations were established after clearing and burning of either forest or jungle rubber and hence these land uses served as the reference land uses with which the converted plantations were compared. Additionally, the comparability of initial soil conditions between the reference and converted land uses was tested based on a land use-independent soil characteristic, i.e., clay content at 0.5–2 m depth, which did not statistically differ among land uses within each landscape (Allen et al., 2015; Hassler et al., 2015). Thus, changes in soil N-oxide fluxes can be

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attributed to land-use change with its associated management practices. The plantations' ages ranged between 7 and 17 years, and tree density, tree height, basal area and tree species abundance were higher in the reference land uses than the monoculture plantations (all reported by Allen et al., 2015; Hassler et al., 2015; Kotowska et al., 2015).

Management practices in the plantations included manual harvest, weeding and fertilizer application. Harvesting of palm fruits was done every 2 weeks and collection of latex was done weekly. In the large-scale oil palm plantation PTPN VI, palm fruits were harvested weekly. Weeding in smallholder rubber and oil palm plantations was done both manually and with herbicides (2–5 L Gramaxone® or Roundup® ha^{-1} yr^{-1}) one to two times per year, and senesced oil palm fronds were regularly cut and piled on the inter-rows (Hassler et al., 2015). In PTPN VI, weeding was done with herbicides (1–1.5 L Glisat® ha^{-1} yr^{-1}) four times per year, combined with some manual hoeing, and senesced fronds were cut and partly piled on the inter-rows and partly taken out from the plot to use as fodder for cattle. Fertilizer application in the smallholder oil palm plantations was done one to two times per year and rates typically varied depending on cash capital of the smallholders. In 2013, fertilization rates ranged between 48–88 kg N ha^{-1} yr^{-1} (except two smallholders who applied 138 kg N ha^{-1} yr^{-1}), 21–38 kg P ha^{-1} yr^{-1} and 40–157 kg K ha^{-1} yr^{-1}, with the lower range in the clay Acrisol and the upper range in the loam Acrisol. The fertilizer sources were NPK complete, urea and KCl. One of the smallholders in the loam Acrisol landscape applied 200 kg dolomite ha^{-1} yr^{-1}. Fertilizers were applied around each palm tree at about 0.8–1 m from the stem base (Hassler et al., 2015). Rubber plantations were not fertilized. In the large-scale oil palm plantation PTPN VI, fertilizer application rates were typically higher than those in smallholder plantations; fertilizers were applied once in 2014 at the rates of 196-36-206 kg N, P, K ha^{-1} yr^{-1}, with also 602 kg dolomite ha^{-1} yr^{-1}, and once before the end of our measurements in July 2015 at the rates of 96-23-96 kg N, P, K ha^{-1} yr^{-1}. The fertilizer forms were NPK complete,

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urea, triple superphosphate and KCl. Application was done partly manually by applying the fertilizers at 1 m distance around each palm tree, and partly mechanically by broadcasting the fertilizer within 1–3 m distance from the palm rows. In 2015, fertilizers were mainly mechanically broadcasted within these inter-rows.

2.2 Soil N-oxide fluxes and supporting soil factors

In 32 plots, soil N_2O fluxes were measured monthly from December 2012 to December 2013, whereas soil NO fluxes were measured four times between March and September 2013, except in two forest sites and one jungle rubber site in the clay Acrisol landscape, where we were unable to measure soil NO fluxes due to difficulty in accessing these sites that did not allow us to stabilize the NO detector during transport in the field (i.e., using motorcycle on very rugged trails). Soil NO fluxes were not measured as frequently as N_2O fluxes and we decided to stop in September 2013 because NO fluxes were always very low at all sites. In the large-scale oil palm plantation PTPN VI within the loam Acrisol landscape, soil N_2O fluxes were measured more frequently (in congruent with its high fertilizer application rate): weekly to biweekly from July 2014 to July 2015, with the exception of September 2014 when we measured only once.

With our sampling strategy, where we used randomly installed chamber bases (with the distances to the tree base between 1.8 and 5 m) in combination with monthly measurements, we may have missed the N fertilizer-induced pulse of soil N-oxide emissions in the smallholder oil palm plantations. Therefore, we conducted more intensive measurements of soil N_2O fluxes during 3 to 8.5 weeks (with 6 to 11 samplings) following fertilizer application at three of the smallholder oil palm plantations within each landscape. These measurements served to characterize the short-term, N fertilizer-induced contribution

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(e.g., Koehler et al., 2009) to total N_2O fluxes. Soil NO fluxes were also measured during 6 to 8.5 weeks (with 9 to 10 samplings) following fertilizer application at one of the smallholder oil palm plantations within each landscape. In the clay Acrisol landscape, measurements in the three smallholder oil palm plantations were conducted during October–December 2013, February–March 2014, and February–April 2014; in the loam Acrisol, measurements were carried out during October–December 2013, January–March 2014, and March–April 2014. We applied the same fertilizer forms, rates and methods as used by the smallholders. Three oil palm trees were selected in each of the six sites. In the clay Acrisol landscape, each tree was applied with 2 kg complete NPK fertilizer (equivalent to 0.32 kg N $tree^{-1}$), whereas in the loam Acrisol, each tree was applied with 2 kg of combined complete NPK, ammonium sulfate and KCl fertilizers (equivalent to 0.26 kg N $tree^{-1}$). The fertilizer was applied within 0.8–1 m distance from the tree base. We installed three permanent chamber bases at various distances from the tree base: 0.3 m from the tree base (chamber location a), 0.8 m from the tree base that was on the fertilized area (chamber location b), and 4–4.5 m from the tree base that was in the middle of the inter-rows and served as the reference chamber without fertilizer application (chamber location c).

Soil N_2O fluxes were measured using the same methods employed in our earlier studies (e.g., Corre et al., 2014; Koehler et al., 2009). During gas sampling, the permanently installed chamber bases were covered with static vented, polyethylene hoods (chamber area of 0.05 m^2 and total volume of 12 L), and four gas samples (30 mL each) were taken at 1, 11, 21 and 31 min after chamber closure by connecting a syringe with a Luer-lock connection to the chamber sampling port. Gas samples were immediately injected into pre-evacuated 12 mL Labco Exetainers sealed with rubber septa (Labco Limited, Lampeter, UK), maintaining an overpressure; these exetainers have been tested by our group to be leak proof during extended period of storage (e.g., up to 6 months) (Hassler et al., 2015). Within 3–4 months the gas

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samples were transported by airfreight to Germany and were analyzed upon arrival using a gas chromatograph with an electron capture detector (GC 6000 Vega Series 2, Carlo Erba Instruments, Milan, Italy). For the measurements from March–July 2015 in the large-scale oil palm plantation PTPN VI, the gas samples were analyzed with another gas chromatograph (SRI 8610C, SRI Instruments Europe GmbH, Bad Honnef, Germany), which had been previously cross-calibrated using the same standards. For calibration, three standard gases were used with concentrations of 360, 1000 and 1600 ppb N_2O (Deuste Steininger GmbH, Mühlhausen, Germany).

Soil NO fluxes were measured (described in detail in our earlier works, e.g., Corre et al., 2014; Koehler et al., 2009) using the same chamber bases described above. During measurements, the chamber bases were covered with dynamic vented, polyethylene hoods (total volume of 12 L), and NO concentrations were measured in situ during 5–7 min following chamber closure using a Scintrex LMA-3 chemiluminescence detector (Scintrex, Ontario, Canada), in which NO is oxidized to NO_2 by a CrO_3 catalyst after which it reacts with a luminol solution. Calibration of the NO detector was carried out at each site prior to and after measurements using a two-point calibration of a standard gas with 3000 ppb NO (Deuste Steininger GmbH, Mühlhausen, Germany) which was diluted using dried ambient air. NO measurements were recorded every 5 seconds using a data logger (CR510, Campbell Scientific, Logan, USA).

Soil N_2O and NO fluxes were calculated from the linear increase of concentration over time adjusted for air temperature and atmospheric pressure, measured at each site and sampling day. Annual soil N_2O fluxes from the weekly or monthly sampling at each site were estimated using the trapezoidal rule on day intervals between measured flux rates, assuming constant flux rates per day (e.g., Hassler et al., 2015). Annual NO fluxes were not calculated,

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since we only conducted four measurement periods for each plot as explained above. To calculate the N fertilizer-induced pulse of soil N-oxide fluxes, we also used the trapezoidal rule on day intervals between measured flux rates to estimate the total flux during the entire period following fertilizer application, covering pre-fertilizer level, the peak, and the return to background levels of soil N-oxide fluxes. We calculated the percentage of combined soil NO and N_2O emissions from the applied N-fertilizer rate at each site as follows: % NO-N + N_2O-N of N applied yr^{-1} = NO-N + N_2O-N fluxes from the fertilized chamber locations a and b (µg N m^{-2} for the entire period of fertilizer effect) – NO-N + N_2O-N fluxes from the unfertilized chamber location c (µg N m^{-2} for the same period) * frequency of fertilization yr^{-1} * fertilized area (m^2 ha^{-1}) ÷ N fertilization rate (kg N ha^{-1} yr^{-1}* 10^9 µg/kg) * 100. In this calculation, we included fluxes from chamber location a in order to include any incidental fertilizer application to this area (possibly from previous applications by the smallholders and possible redistribution of applied nutrients within the soil) since N-oxide fluxes from chamber location a were often higher than those from unfertilized chamber location c (see Sect. 3.2).

Soil factors known to control soil N-oxide fluxes (i.e., temperature, water-filled pore space (WFPS), and extractable NH_4^+ and nitrate (NO_3^-) were measured for the top 0.05 m depth during each soil N-oxide flux measurement at all 32 sites. Soil temperature was measured close to each chamber base using a digital thermometer. Soil samples were taken at 1 m distance from the four chambers, pooled, mixed thoroughly, and subsampled for immediate extraction of mineral N in the field, using prepared extraction bottles containing 150 mL 0.5 M K_2SO_4. Upon arrival at the field station, extraction bottles were shaken for 1 h, filtered and extracts were frozen immediately. The remaining soil sample was used to determine the gravimetric moisture content (by oven-drying for at least 1 day at 105 °C), whereby WFPS was calculated using a particle density of 2.65 g cm^{-3} for mineral soil and the measured soil bulk density at our study sites (Allen et al., 2015). During the measurements

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following the fertilizer applications, soil was sampled close to each of the chamber locations a, b and c (described above) and was processed separately for mineral N extraction and WFPS determination. Frozen extracts were transported by airfreight to Germany and analyzed for NH_4^+ and NO_3^- concentrations using continuous flow injection colorimetry (SEAL Analytical AA3, SEAL Analytical GmbH, Norderstedt, Germany), as described in detail by Hassler et al. (2015).

2.3 Statistical analysis

We first tested each parameter for normal distribution (Shapiro-Wilk's test) and equality of variance (Levene's test), and a logarithmic transformation was applied when necessary. For analysis of differences in N-oxide fluxes among land uses or between soil landscapes, we used the means of the four chambers representing each replicate plot on a sampling day. Linear mixed-effect (LME) models (Crawley, 2007) were used to assess differences between landscapes for the reference land uses (i.e., clay vs. loam Acrisol; first hypothesis) or differences among land uses within each landscape (i.e., land-use change effect; second hypothesis). In the LME models, either landscape or land use was considered as fixed effect whereas replicate plots and sampling days were considered as random effects. For comparison of soil N_2O fluxes between the large-scale (PTPN VI) and smallholder oil palm plantations in the loam Acrisol landscape, we also used the means of the three chambers per replicate in the PTPN VI site on each sampling day as there were no significant differences between the chamber locations (based on LME models with chamber location as fixed effect and replicates as well as sampling days as random effects; $P = 0.70$). We then used the LME model with plantation types (i.e., large scale vs. smallholder) as a fixed effect and replicates and sampling days as random effects. For analysis of fertilization effects (i.e., as represented by the

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chamber locations a, b and c) on soil N-oxide fluxes from smallholder oil palm plantations, this was conducted for each site with oil palm trees as replicates. In the LME model for this experiment, chamber location was the fixed effect whereas replicate palm trees and sampling days were the random effects. To assess differences in N-oxide fluxes between landscapes following fertilization for chamber locations a and b, we also used LME models with landscape as fixed effect and with replicate plots (for N_2O) or replicate palm trees (for NO) and sampling days as random effects. In all LME models, we included (1) a variance function that allows different variances of the fixed effect, and/or (2) a first-order temporal autoregressive function to account for decreasing correlation between sampling days with increasing time difference, if these functions improved the relative goodness of the model fit based on the Akaike information criterion. Significant differences were based on the analysis of variance with Fisher's least significant difference test for multiple comparisons. We set the statistical significance at $P \leq 0.05$ and, only for a few specified parameters, we also considered marginal significance at $P \leq 0.09$ because our experimental design encompassed the inherently high spatial variability in our study area (e.g., Hassler et al., 2015).

To assess the temporal relationships between soil N-oxide fluxes and soil factors (temperature, WFPS, NO_3^- and NH_4^+), we used the means of the replicate plots per land use on each of the 12 monthly measurements and conducted Pearson's correlation test separately for the reference land uses (forest and jungle rubber, n = 48 (N_2O), n = 16 (NO)) and the converted land uses (rubber and oil palm, n = 48, (N_2O), n = 16 (NO)) across landscapes for the whole year. Similarly, for soil N_2O and NO fluxes following fertilizer application from smallholder oil palm plantations, we used the means of the three replicate trees per chamber location on each sampling day and conducted Pearson's correlation test for each site across the entire measurement period of fertilization effects (n = 6–11). To assess the spatial controls of soil biochemical characteristics (Appendix Table A1) on annual soil N_2O fluxes, we used

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the annual flux of each replicate plot and conducted Spearman's rank correlation test separately for the reference land uses and converted land uses across landscapes (n = 16) and within each landscape (n = 8). We did not assess the spatial control of soil biochemical characteristics on annual soil NO fluxes since we did not calculate annual flux from the four measurement periods (as explained in Sect. 2.2). Correlations were considered statistically significant at $P \leq 0.05$ and marginally significant at $P \leq 0.09$. All statistical analyses were conducted using R 3.2.2 (R Development Core Team, 2015).

3 Results

3.1 Soil N-oxide fluxes

In the reference land uses, N_2O was the dominant N-oxide emitted from soils; in the clay Acrisol landscape there was a net NO consumption in the soil of the jungle rubber (Table 1). Soil N_2O and NO fluxes from reference land uses were comparable between the two landscapes (P = 0.54–0.74; Table 1; Fig. 1a, b). These fluxes also exemplified high inherent spatial and temporal variations as indicated by their large standard errors.

In the converted land uses, soil N_2O fluxes were similar to the fluxes of reference land uses (P = 0.58–0.76; Table 1; Fig. 1a, b) within each landscape. However, in the loam Acrisol landscape, the large-scale oil palm plantation PTPN VI had on average 3.5 times higher soil N_2O fluxes than those from the smallholder plantations (Table 1), although this trend was not statistically different (P = 0.15) because of the large variation among replicate plots (as indicated by the large standard error) in this large-scale plantation. Soil NO fluxes, were not different either among land uses in the clay Acrisol landscape (P = 0.73; Table 1). However, in the loam Acrisol landscape, soil NO fluxes were marginally lower (P = 0.07) in rubber plantations (with net NO consumption in the soil) than in jungle rubber (with net NO emission), whereas they were intermediary in forests and oil palm plantations (Table 1).

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3.2 Fertilization effects on soil N-oxide fluxes from smallholder oil palm plantations

In comparison to the unfertilized area (chamber location c at 4–4.5 m from the tree base) soil N_2O fluxes were on average 442 times (clay Acrisol) and 22 times (loam Acrisol) higher within the small fertilized areas around the oil palms (chamber location b at 0.8–1 m from the tree base) during the 3 to 8.5 weeks following fertilizer applications (all $P < 0.01$–0.03; Table 2; Fig. 2c, d). In chamber location a, soil N_2O emissions were also 25 times higher compared to the reference chamber location c in the clay Acrisol landscape (all $P < 0.01$; Table 2; Fig. 2a). In the loam Acrisol landscape, we only detected such an effect in site 2 which displayed 16 times higher soil N_2O emissions in chamber location a compared to the reference chamber location c ($P = 0.03$; Table 2; Fig. 2b).

In the clay Acrisol landscape, soil N_2O emissions in chamber location b increased immediately after fertilizer application, reached a peak within 9 days following fertilizer application and stayed elevated for at most 2 months (Fig. 2c). In the loam Acrisol landscape, N_2O fluxes in chamber location b increased within the first 5 days, reached maximum fluxes within 5–21 days and remained elevated for at most 6.5 weeks (Fig. 2d). Soil N_2O fluxes in chamber location a displayed a similar but less pronounced pattern as those of chamber location b in both landscapes (Fig. 2a, b).

Considering the area coverage (4 % of the area in a hectare) and time span of fertilizer-induced N_2O emissions, their average contributions were 21 % to the annual fluxes in the clay Acrisol landscape (with its usual fertilizer application of once a year), and only 6 % to the annual fluxes in the loam Acrisol landscape (with its common fertilizer application of twice a year) (Table 1).

Compared to the unfertilized area (chamber location c), soil NO fluxes from the fertilized area (chamber location b) had on average 357 times (clay Acrisol) and 238 times

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(loam Acrisol) higher fluxes (both $P < 0.01$) during 6 to 8.5 weeks of measurements following fertilizer application (Table 2; Fig. 3c, d). No differences in soil NO fluxes were detected between chamber locations a and c ($P = 0.10–0.12$; Table 2; Fig. 3a, b). Soil NO fluxes in chamber location b peaked after 10 days in the loam Acrisol and after 3 weeks in the clay Acrisol landscape (Fig. 3c, d), and returned to the background fluxes after 6–8.5 weeks with a drastic drop after 3–5 weeks (Fig. 3c, d). In chamber location a, soil NO fluxes increased quickly and decreased to the background fluxes within at most 16 days following fertilizer application (Fig. 3a, b). As was the case for the monthly sampling, soil N_2O fluxes from chamber locations a and b were larger than soil NO fluxes for both landscapes, (Table 2; Fig. 2a–d and 3a–d). Comparing between landscapes, soil N_2O fluxes from chamber location b were higher in the clay than loam Acrisol soils ($P = 0.09$; Table 2; Fig. 2c, d) but were comparable for chamber location a ($P = 0.41$; Table 2; Fig. 2a, b) and for soil NO fluxes of both chamber locations ($P = 0.45–0.78$; Table 2; Fig. 3a–d).

Fertilizer-induced soil NO fluxes in the loam Acrisol landscape were 0.07 ± 0.02 kg NO-N ha^{-1} yr^{-1}, which was roughly the same as our extrapolated annual value of 0.06 ± 0.06 kg NO-N ha^{-1} yr^{-1} from the four measurement periods (Table 1). In the clay Acrisol landscape, fertilizer-induced soil NO fluxes were 0.12 ± 0.04 kg NO-N ha^{-1} yr^{-1}, which was a net emission compared to our extrapolated annual value with a net sink of -0.02 ± 0.11 kg NO-N ha^{-1} yr^{-1}, based on the four measurement periods (Table 1). The percentages of combined soil N_2O and NO fluxes to the applied N fertilizer rate were on average 0.73 % yr^{-1} in the clay Acrisol landscape and 0.20 % yr^{-1} in the loam Acrisol landscape.

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3.3 Temporal controls of soil N-oxide fluxes

In the reference land uses, soil N_2O and NO fluxes were both positively correlated with soil NO_3^- contents, while soil NO fluxes were also negatively correlated with WFPS and soil NH_4^+ contents (Table 3). In the converted land uses, soil N_2O fluxes were positively correlated with soil NO_3^- contents and temperature (Table 3). This latter correlation was influenced by one sampling period with high N_2O (fertilizer-induced) emissions; when this period was excluded in the analysis, we did not detect a significant correlation with soil temperature. There were no significant correlations observed between soil NO fluxes and soil factors in the converted land uses due to the very low NO emissions and even net NO uptake.

From the fertilizer application experiment in the smallholder oil palm plantations, the location directly receiving fertilizer (chamber location b) showed positive correlations of soil N_2O fluxes with soil NH_4^+ and/or NO_3^- contents in three of the six sites (Table 4). Here, also soil NO fluxes correlated positively with soil NO_3^- contents in the loam Acrisol but not in the clay Acrisol (Table 4). In chamber location a, positive correlations of soil N_2O fluxes with soil NH_4^+ and/or NO_3^- contents were observed in four of the six sites (Table 4). The correlations of soil N_2O fluxes with mineral N for chamber location a in site 2 of the clay Acrisol landscape were caused by one measurement period with very high flux, and exclusion of this observation resulted in a none significant correlation. For soil NO fluxes from chamber location a, we did not detect any significant correlation with soil factors (Table 4). A positive correlation of soil N_2O fluxes with WFPS was observed for chamber locations a and b in site 1 of the loam Acrisol landscape, whereas this correlation was negative for chamber location a in site 3 of the same landscape (Table 4). We also detected a negative correlation between soil NO fluxes and WFPS for chamber location b in site 3 of the clay Acrisol, whereas in the same site soil NO fluxes and WFPS were positively correlated for the unfertilized chamber location

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c (Table 4); however this latter correlation was caused by only one sampling time with a high flux and high WFPS.

3.4 Spatial controls of annual soil N_2O fluxes

The soil physical and biochemical characteristics used for this correlation analysis are reported in Table A1. For the reference land uses, annual N_2O fluxes were positively correlated with gross nitrification rates across landscapes (*Spearman's* ρ = 0.57, P = 0.02, n = 16). Within each landscape, annual soil N_2O fluxes correlated negatively with soil C:N ratio (ρ = -0.69, P = 0.07, n = 8) in the clay Acrisol, whereas in the loam Acrisol annual soil N_2O fluxes correlated positively with microbial C (ρ = 0.69, P = 0.07, n = 8). For the converted land uses, annual N_2O fluxes correlated negatively with sand content across landscapes (ρ = -0.57, P = 0.06, n = 12). There were no other correlations detected with any other soil biochemical parameters.

4 Discussion

4.1 Soil N_2O and NO fluxes from the reference land uses

N_2O fluxes from our forest soils (Table 1) fell at the lower end of those reported for humid tropical forests (9.8–85.1 µg N_2O-N m^{-2} h^{-1}; summarized by Castaldi et al., 2013). Compared to soil N_2O fluxes measured in Indonesia, our values were comparable to those from montane forests on Cambisol soil (at 1190 m elevation in Sulawesi) with similar temporal sampling scheme and spatial replication (12.7 µg N_2O-N m^{-2} h^{-1}; Purbopuspito et al., 2006) and to five lowland forest stands on Acrisol soil (at 0–180 m elevation in Jambi) measured once (11.6 µg N_2O-N m^{-2} h^{-1}; Ishizuka et al., 2005). However, soil N_2O fluxes from our forests were lower than those reported from submontane and montane forests on Cambisol soil (at 450–1160 m elevation in Sulawesi) with six monthly measurements and comparable spatial replication (25

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µg N_2O-N m^{-2} h^{-1}; Veldkamp et al., 2008) and from a lowland forest on Ferralsol soil (at 100 m elevation in Jambi) with 13 monthly measurements (19.8 µg N_2O-N m^{-2} h^{-1}; Aini et al., 2015). In contrast, our values were higher than those reported for two lowland forests on Ferralsol soil (at approximately 100 m elevation in Jambi) with nine monthly measurements (3.0 µg N_2O-N m^{-2} h^{-1}; Ishizuka et al., 2002). Since the studies from the montane forests were conducted on less weathered soils and the studies from the same region by Ishizuka et al. (2002, 2005) and Aini et al. (2015) have less temporal or spatial replication, their values should be carefully related to our measured fluxes.

Soil NO fluxes from Southeast Asian lowland forests are not reported so far. Our measured NO fluxes from the forest soils (Table 1) tended to be lower than those reported for lowland forests in Latin America with soils ranging from less weathered Cambisols to highly weathered Acrisols and Ferralsols (3.2 µg NO-N m^{-2} h^{-1}, Corre et al., 2014; 10.3 NO-N m^{-2} h^{-1}, Davidson et al., 2004; 88.0–90.0 µg NO-N m^{-2} h^{-1}, Keller et al., 2005; 17.4 µg NO-N m^{-2} h^{-1}, Verchot et al., 1999). There are only two studies that reported soil NO fluxes from montane forests on Cambisol soils in Sulawesi, Indonesia (Purbopuspito et al., 2006, Veldkamp et al., 2008). Our measured soil NO fluxes were comparable with the values reported for montane forests at ≥ 1800 m elevation (1.9–2.1 µg NO-N m^{-2} h^{-1}; Purbopuspito et al., 2006) but lower than those reported for (pre)montane forests at lower elevations (5.5 µg NO-N m^{-2} h^{-1} at 1190 m, Purbopuspito et al., 2006; 12.0 µg NO-N m^{-2} h^{-1} at 450–1160 m, Veldkamp et al., 2008). Although it is known that tropical forest soils are the largest natural source of N_2O and produce considerable amounts of NO, our measurements from these lowland forests in Jambi, Indonesia on highly weathered Acrisol soils showed generally low soil N-oxide fluxes.

In contrast to our first hypothesis, soil N-oxide fluxes from the reference land uses were comparable between loam and clay Acrisol landscapes. This is possibly due to the

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generally low soil N availability in these sites, as indicated by their lower gross N mineralization rates (Allen et al., 2015) compared, for example, to the less weathered Cambisol and Nitisol soils in a lowland forest of Panama (Corre et al., 2010). Soil N-oxide fluxes are largely controlled, first, by the magnitude of soil N availability, as depicted in the HIP conceptual model (Davidson et al., 2000). This influence of soil N availability on N-oxide fluxes was illustrated by the positive correlations of soil N-oxide fluxes with soil NO_3^- contents (Table 3). Across landscapes, this first level of control was also corroborated by the positive correlations of annual soil N_2O fluxes with gross nitrification rates, and within each landscape by the negative correlation with the soil C:N ratio (clay Acrisol landscape) and by the positive correlation with microbial C (loam Acrisol landscape) (see Sect. 3.4). Our findings were consistent with those from other tropical soils, illustrating that soil N-oxide fluxes across or within sites are controlled by soil N availability as expressed in various indexes such as soil NO_3^- contents (Keller and Reiners, 1994; Müller et al., 2015), nitrification rates (Davidson et al., 2000) and soil C:N ratio (Breuer et al., 2000).

Moreover, we attributed the low soil NO fluxes and the dominance of N_2O (Table 1) in our sites to the second level of control of N-oxide fluxes - soil aeration status (HIP model; Davidson et al., 2000). The ratio of N_2O to NO is expected to increase when WFPS exceeds 60 % as low soil aeration favors N_2O production by denitrification and nitrification processes (Davidson et al., 2000). WFPS in the reference land uses were ≥ 60 % (Appendix Table A2, except in jungle rubber of the loam Acrisol with 54 % WFPS). Hence, it was not surprising that our measured soil NO fluxes were close to zero or showed net consumption (Table 1); the high WFPS may have led to NO reduction to N_2O (Conrad, 1996; Pilegaard, 2013). This was supported by the negative correlation between soil NO fluxes and WFPS (Table 3). Furthermore, increased concentrations of NO in the atmosphere due to biomass burning in this region (Field et al., 2009; Levine, 1999) may have resulted in a net NO consumption (not

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only in the reference land uses but also in the converted land uses; Table 1) since increased ambient NO concentration could enhanced soil NO uptake (Conrad, 1994). In summary, soil NO fluxes from the reference land uses were of minor importance compared to soil N_2O fluxes. However, if droughts will occur more frequently or extremely in this region (Lestari et al., 2014), soil NO fluxes might become important.

4.2 Land-use change effects on soil N_2O and NO fluxes

Soil N_2O fluxes from our rubber plantations (Table 1) were comparable to fluxes from a rubber plantation on Ferralsol soil (at approximately 110 m elevation in Peninsular Malaysia) with eight measurements during 1.5-year period (7.8 μg N_2O-N m^{-2} h^{-1}; Yashiro et al., 2008) and slightly higher than fluxes reported from a rubber plantation on a lateritic soil (at 580 m elevation in Xishuangbanna, China) with only two months of sampling (4.1 μg N_2O-N m^{-2} h^{-1}; Werner et al., 2006). Studies from the same region (Jambi, Indonesia) report lower soil N_2O fluxes from one rubber plantation on Ferralsol soil (at approximately 100 m elevation) with nine monthly measurements (0.7 μg N_2O-N m^{-2} h^{-1}; Ishizuka et al., 2002) as well as higher fluxes from five rubber plantations on Acrisol soils (at 70–280 m elevation) with only one-time measurement (20.6 μg N_2O-N m^{-2} h^{-1}; Ishizuka et al., 2005) and from one rubber plantation on Ferralsol soil (at 100 m elevation) with 13 monthly measurements (11.6 μg N_2O-N m^{-2} h^{-1}; Aini et al., 2015). Soil N_2O fluxes from our oil palm sites were in the same order of magnitude as those reported from three oil palm plantations on Acrisol soils (at 70–110 m elevation) with only one-time sampling (15.1 μg N_2O-N m^{-2} h^{-1}; Ishizuka et al., 2005) and from one oil palm plantation on Cambisol soil (at 70 m elevation) with 13 monthly measurements (11.9 μg N_2O-N m^{-2} h^{-1}; Aini et al., 2015), whereby both studies were also conducted in Jambi, Indonesia. However, soil N_2O fluxes from our oil palm sites were higher

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compared to fluxes reported from one oil palm plantation on Ferralsol soil (at approximately 110 m elevation) in Peninsular Malaysia with eight measurements during 1.5-year period (-0.1 μg N_2O-N m^{-2} h^{-1}; Yashiro et al., 2008). Soil NO fluxes have never been reported from rubber or oil palm plantations. Our present study provides the first soil N-oxide flux measurements from these land uses with sufficient temporal coverage and spatial replications at the landscape scale.

In contrast to our second hypothesis, soil N-oxide fluxes were comparable among land uses (except for soil NO fluxes between rubber and jungle rubber in the loam Acrisol landscape as discussed below), even with the observed decreases in soil mineral N levels among land uses (i.e., generally lower NH_4^+ and NO_3^- levels in rubber plantations than in the reference land uses at both landscapes; Appendix Table A2). In the same study sites, Allen et al. (2015) found differences in other indices of soil N availability with land-use change, particularly in the clay Acrisol landscape: microbial C and N, gross N mineralization and NH_4^+ immobilization rates decrease with conversion of forest to rubber or oil palm plantations. N-oxide emissions generally account only a small fraction of soil available N (e.g., N_2O + NO emissions comprise 0.03 % of gross N mineralization rates in a lowland forest on Cambisol and Nitisols soils in Panama; Corre et al., 2014). In our present study, the reference land uses on highly weathered Acrisol soils have low soil N availability and their conversion to these plantations further decreases the soil N-cycling rates (Allen et al., 2015). Hence, we reason that we did not detect differences in N-oxide fluxes with land-use conversion to rubber and oil palm plantations because we started with low soil N availability and low N-oxide emissions and any changes were probably too small to detect statistically. The temporal pattern of soil N_2O fluxes in the converted land uses were also controlled by soil NO_3^- contents (Table 3), emphasizing the first level of control of soil N availability on soil N_2O fluxes (HIP model; Davidson et al., 2000). Across landscapes, the correlations of

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annual soil N_2O fluxes from these converted land uses with sand contents (see Sect. 3.4) also suggested the indirect influence of soil texture on water holding capacity, or conversely soil aeration status, which is the second level of control on soil N_2O fluxes (HIP model). Consequently, in terms of N-oxide emissions, this footprint of smallholder oil palm and rubber plantations was similar to the original land uses. However, this picture might change with increasing usage of N fertilizer (see Sect. 4.3).

The lower soil NO fluxes in rubber compared to jungle rubber in the loam Acrisol (Table 1) partly supports our second hypothesis. These differences might be related to the low WFPS and the higher soil NO_3^- contents in jungle rubber (Appendix Table A2), which could favor the relatively high soil NO emissions; this was also supported by the opposing correlations of soil NO with NO_3^- and WFPS (Table 3). Additionally, the low soil NO fluxes from rubber plantations could be the result of the effect of monoterpenes, produced by rubber trees, which reduce nitrification in soil (Wang et al., 2007; White, 1991). This is supported by low gross nitrification rates (measured in the same plots by Allen et al., 2015), low soil NO_3^- contents (Appendix Table A2) and consequently low soil NO fluxes in rubber plantations (Table 1).

4.3 Soil management effects on soil N_2O and NO fluxes from oil palm plantations

N fertilizer application, a commonly employed soil management in oil palm plantations (e.g., Allen et al., 2015; Hassler et al., 2015), increases N-oxide emission for a relatively short period (e.g., Koehler et al. 2009). Our findings show that these fertilizer-induced N-oxide emissions were mainly limited to the small area around the palm base where fertilizer is commonly applied (4 % of the area in a hectare) and that N-oxide emissions peaked within 3 weeks (Figs. 2 and 3). These N-fertilizer induced N_2O fluxes of 6–21 % of the annual soil N_2O fluxes were similar in magnitude as the standard errors of the annual fluxes (estimated

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from the monthly measurements; Table 1). Thus, inclusion of these N-induced emissions in our annual estimates did not result in statistically significant effects of land-use change.

The percentages of soil N_2O and NO fluxes to the applied N fertilizer rate were smaller than those reported from other agricultural land uses in humid tropical regions (6.4–8.6 %; Veldkamp and Keller, 1997; Veldkamp et al., 1998). Usually the percentage of soil N-oxide emissions to applied N fertilizer rate increases with increasing N fertilization rates (Hoben et al., 2011; Pennock and Corre, 2001). Since the fertilization rates in our studied smallholder oil palm plantations were lower compared to the fertilization rates in these other studies (with N fertilization rates ranging from 300–360 kg N ha^{-1} yr^{-1}), our quantified N-oxide loss from N fertilizer were also low. The higher soil N_2O fluxes in the large-scale oil palm plantation PTPN VI, although not statistically different from the smallholder plantations (Table 1), could be attributed to its high N fertilization rate (196 kg N ha^{-1} yr^{-1}). Summing the N-induced N-oxide fluxes and the annual soil N-oxide emissions based on the monthly measurements (Table 1), these values from the smallholder plantations were still lower than the annual flux from the large-scale plantation (Table 1). Based on our finding that soil N_2O fluxes following fertilizer application (chamber location b) were higher in the clay than loam Acrisol landscapes (most likely due to higher WFPS in the clay (61 ± 8 %) than loam Acrisol (27 ± 3 %) during this measurement period), soil N-oxide fluxes from large-scale plantations on clay soils could be even higher than what we measured here from a large-scale plantation on a loam soil. Our findings reinforced the need to quantify these climate-relevant N-oxide gases in large-scale plantations, which constitute ~50 % of the land area under oil palm plantation in whole of Sumatra (BPS, 2014).

Temporal patterns in soil N-oxide fluxes following fertilizer application were also controlled by soil N availability, as reflected by their positive correlations with soil NH_4^+ and/or NO_3^- contents (Table 4). The pulse application of N fertilizer provide temporary

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surplus of mineral N that was lost via gaseous emission and leaching (Kurniawan, 2016), and with time following fertilizer application such effect diminished as the mineral N is incorporated into the soil N-cycling processes (Allen et al., 2015). The positive correlation between soil N_2O fluxes and WFPS (i.e., chamber locations a and b in site 1 of the loam Acrisol; Table 4) and the negative correlation between soil NO fluxes and WFPS (i.e., chamber location b in site 3 of the clay Acrisol landscape; Table 4) again attested that when the first level of control (soil N availability) was favorable (i.e., high soil mineral N contents in these fertilized chamber locations) the control of soil moisture on aeration status was enhanced, as such correlation was not seen in the unfertilized area (chamber location c) or in the monthly measured fluxes (Tables 3 and 4). These correlations indicated that following fertilizer application soil NO fluxes decreased whereas soil N_2O fluxes increased with increases in WFPS. In site 3 of the loam Acrisol, the seemingly contradicting negative correlation of soil N_2O fluxes with WFPS (Table 4) was only because there was a decreasing WFPS following fertilizer application with concurrently increasing soil mineral N contents - the latter dominantly driving the increases in soil N_2O fluxes (i.e., positive correlations with NH_4^+ and NO_3^-; Table 4). In summary, the short-term effect of fertilization also depicted the two levels of controls on soil N-oxide fluxes as exemplified in the HIP model.

5 Conclusions

Our study provides the first spatially replicated study with a full year of measurements of soil N_2O fluxes and the first reported soil NO fluxes from this region of hotspot of land-use conversion for globally important tree cash crops. In contrast to our first hypothesis, soil texture, through its role on soil fertility, did not directly affect soil N-oxide fluxes (as shown by the comparable fluxes between landscapes with soil textural differences) but influenced the landscape-scale pattern of annual soil N_2O fluxes in the converted land uses (i.e., negative

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correlation between annual N_2O fluxes and sand content) most likely through its role on soil moisture availability. The generally low soil N-oxide fluxes from the reference land uses were due to the low soil N availability in these highly weathered Acrisol soils (Allen et al., 2015). Forest or jungle rubber conversion to rubber and oil palm by smallholders also did not show significant changes in soil N-oxide fluxes, except for the decrease in soil NO fluxes in rubber plantations and for the short-term pulse of soil N-oxide fluxes following fertilizer application in oil palm plantations. These partly support our second hypothesis. Using a conservative estimate of N-oxide (N_2O + NO) loss from the applied N fertilizer (average of 0.5 % from the loam and clay Acrisol landscapes), and a conservative average N fertilization rate across smallholder and large-scale plantations of 100 kg N ha^{-1} yr^{-1}, with the total land area of oil palm in Jambi province of 721000 ha (BPS, 2014), we estimated an annual soil N-oxide emission from N fertilization of 360500 kg N yr^{-1}. The N fertilization rates in our smallholder oil palm plantations were only about one-fourth to one-half of what is commonly practiced in large-scale industrial plantations (e.g., 130–260 kg N ha^{-1} yr^{-1} in Jambi, Indonesia; Pahan, 2010), and our measurements from a large-scale oil palm plantation PTPN VI showed high soil N-oxide fluxes. To improve estimate of soil N-oxide fluxes at regional level, future studies should focus on large-scale plantations (which constitute 38 % of oil palm land area in Jambi province; BPS, 2014) with frequent measurements during 2 months following fertilizer application, and particularly during wet season for N_2O flux measurements and during dry season for NO flux measurements.

Data availability

The underlying research data of this study is deposited at the EFForTS-IS data repository (https://efforts-is.uni-goettingen.de), an internal data exchange-platform, which is accessible for SFB 990 members only. Based on data sharing agreement within the SFB 990, these data

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are currently not publicly accessible but will be made available through a written request to the senior authors.

Competing interests

The authors declare that they have no conflict of interest.

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Table 1. Mean (±SE, n = 4 sites) soil N_2O and NO fluxes and annual soil N_2O fluxes from different land uses within each landscape in Jambi, Sumatra, Indonesia, measured monthly from December 2012 to December 2013. Means followed by different lowercase letters indicate significant differences among land uses within each landscape and different capital letters indicate significant differences between landscapes within each land use (linear mixed-effect models with Fisher's LSD test at $P \leq 0.09$). For soil NO fluxes in the clay Acrisol, forest was excluded in the comparison among land uses because its monthly measurements was only carried out in two sites due to road inaccessibility with the NO-measuring instrument in the other two sites. Annual soil N_2O fluxes were not statistically tested for differences between landscapes or land uses since these annual values are trapezoidal extrapolations. For smallholder oil palm plantations, values in italics are the fertilizer-induced annual soil N_2O fluxes (see Sect. 2.2). In the loam Acrisol landscape, soil N_2O fluxes were additionally measured in a large-scale oil palm plantation (mean±SE, n = 4 replicates) from July 2014 to July 2015; these fluxes did not differ from those of smallholder plantations in the same landscape (linear mixed-effect models with Fisher's LSD test at P = 0.15) due to large spatial variation (indicated by large SE).

Land-use type	N_2O fluxes ($\mu g\ N\ m^{-2}\ h^{-1}$)	NO fluxes ($\mu g\ N\ m^{-2}\ h^{-1}$)	Annual N_2O fluxes ($kg\ N\ ha^{-1}\ year^{-1}$)
clay Acrisol landscape			
Forest	$12.76 \pm 5.57^{a,A}$	(1.70 ± 0.32)	1.03 ± 0.41
Jungle rubber	$6.73 \pm 1.50^{a,A}$	$-0.56 \pm 0.69^{a,A}$	0.62 ± 0.14
Rubber	$5.56 \pm 2.47^{a,A}$	$-1.00 \pm 0.15^{a,A}$	0.46 ± 0.21
Oil palm (smallholder plantation)	$11.47 \pm 2.88^{a,A}$	$-0.20 \pm 1.23^{a,A}$	1.01 ± 0.25 *0.21 ± 0.04*
loam Acrisol landscape			

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Forest	9.77 ± 1.46[a,A]	1.87 ± 1.27[ab]	0.88 ± 0.15
Jungle rubber	14.01 ± 6.69[a,A]	5.68 ± 5.77[a,A]	1.19 ± 0.57
Rubber	8.61 ± 2.04[a,A]	-1.16 ± 0.49[b,A]	0.69 ± 0.17
Oil palm (smallholder plantation)	12.16 ± 6.08[a,A]	0.73 ± 0.67[ab,A]	1.13 ± 0.53 *0.07 ± 0.02*
Oil palm (large-scale plantation)	42.34 ± 24.22[a,A]	-	3.26 ± 1.73

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Table 2. Mean (±SE, *n* = 3 oil palm trees) soil N_2O and NO fluxes from three chamber locations during a fertilization in three (for N_2O) or one (for NO) smallholder oil palm plantation within each landscape, measured 6 to 11 times during 3–8.5 weeks following fertilization. Means followed by different letters indicate significant differences among chamber locations within each site (linear mixed-effect models with Fisher's LSD test at $P \leq 0.05$). Chamber locations a, b and c were placed at 0.3 m, 0.8 m, and 4–4.5 m, respectively, from each of the three trees in each oil palm plantation site. Smallholders fertilized around the base of each tree at about 0.8–1 m from the tree base, and thus chamber location b was on this fertilized area and chamber location c served as the reference chamber not receiving any fertilizer. The same fertilization rate and form were used as the smallholders applied in these plantations (see Sect. 2.2).

Oil palm site	Chamber location	N_2O fluxes ($\mu g\ N\ m^{-2}\ h^{-1}$)	NO fluxes ($\mu g\ N\ m^{-2}\ h^{-1}$)
clay Acrisol landscape			
1	a	156.66 ± 86.76^{b}	-
	b	910.11 ± 410.00^{a}	-
	c	6.93 ± 3.30^{c}	-
2	a	130.62 ± 34.62^{b}	-
	b	692.74 ± 144.10^{a}	-
	c	9.87 ± 3.01^{c}	-
3	a	45.49 ± 3.73^{b}	4.74 ± 1.74^{b}
	b	1280.95 ± 486.67^{a}	535.29 ± 194.46^{a}
	c	1.14 ± 1.64^{c}	1.50 ± 1.46^{b}

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Oil palm site	Chamber location	N_2O fluxes ($\mu g\ N\ m^{-2}\ h^{-1}$)	NO fluxes ($\mu g\ N\ m^{-2}\ h^{-1}$)
loam Acrisol landscape			
1	a	33.46 ± 9.76^{b}	-
	b	133.36 ± 34.90^{a}	-
	c	11.82 ± 6.08^{b}	-
2	a	129.74 ± 46.19^{a}	46.17 ± 19.63^{b}
	b	205.31 ± 24.17^{a}	157.12 ± 35.67^{a}
	c	7.89 ± 4.78^{b}	0.66 ± 0.30^{b}
3	a	5.17 ± 1.04^{b}	-
	b	104.53 ± 81.90^{a}	-
	c	3.68 ± 1.74^{b}	-

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Table 3. Pearson correlation coefficients between soil N_2O flux (n = 48; µg N m^{-2} h^{-1}), soil NO flux (n = 16; µg N m^{-2} h^{-1}), water-filled pore space (WFPS; %, top 0.05 m depth), soil temperature (°C, top 0.05 m depth) and extractable mineral N (mg N kg^{-1}, top 0.05 m depth) across landscapes for the reference and converted land uses. Correlation was conducted using the means of the four replicate plots per land use on each monthly measurement from December 2012 to December 2013 (soil N_2O fluxes) and March 2013 to September 2013 (soil NO fluxes).

Land-use type	Variable	WFPS	Soil temp.	NH_4^+	NO_3^-
Reference land uses (forest and jungle rubber)	Soil N_2O flux	-0.21	-0.09	-0.23	0.38[c]
	Soil NO flux	-0.74[c]	-0.15	-0.48[a]	0.69[c]
Converted land uses (rubber and oil palm)	Soil N_2O flux	0.11	0.30[b]	0.23	0.37[c]
	Soil NO flux	-0.05	0.09	-0.05	0.23

[a]$P \leq 0.09$, [b]$P \leq 0.05$, [c]$P \leq 0.01$.

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Table 4. Pearson correlation coefficients (n = 6–11 measurements following fertilization) between N-oxide fluxes (μg N m^{-2} h^{-1}), water-filled pore space (WFPS; %, top 0.05m depth) and extractable mineral N (mg N kg^{-1}, top 0.05 m depth), measured at different chamber locations (a, b and c were at 0.3 m, 0.8 m (fertilized area) and 4–4.5 m, respectively, from each of the three trees in each smallholder oil palm plantation). Correlation was conducted using the means of the three replicate trees per chamber location.

Oil palm plantation site	Chamber location	Variable	WFPS	NH_4^+	NO_3^-
clay Acrisol landscape					
1 (n = 6 measurements)	a	Soil N_2O flux	0.55	0.88[b]	0.46
	b		0.57	-0.22	-0.31
	c		0.37	-0.64	-0.44
2 (n = 11 measurements)	a	Soil N_2O flux	0.11	0.93[c]	0.95[c]
	b		0.08	0.05	-0.06
	c		0.09	-0.44	-0.45
3 (n = 10 measurements)	a	Soil N_2O flux	-0.19	0.10	0.09
	b		0.05	0.86[c]	0.85[c]
	c		-0.32	0.06	-0.44
3 (n = 10 measurements)	a	Soil NO flux	-0.34	0.44	0.48
	b		-0.61[a]	0.10	-0.04
	c		0.59[a]	-0.14	-0.13
loam Acrisol landscape					
1 (n = 6 measurements)	a	Soil N_2O flux	0.96[c]	-0.18	0.03
	b		0.78[a]	0.61	-0.40
	c		-0.06	-0.29	<0.01
2 (n = 9 measurements)	a	Soil N_2O flux	-0.55	0.71[b]	-0.03
	b		0.35	-0.20	0.89[c]
	c		0.34	<0.01	-0.35

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3 (n = 11 measurements)	a	Soil N_2O flux	-0.68[b]	0.67[b]	0.62[b]
	b		-0.27	-0.2	0.57[a]
	c		0.36	0.19	0.06
2 (n = 9 measurements)	a	Soil NO flux	-0.07	0.18	-0.27
	b		0.07	-0.11	0.96[c]
	c		-0.16	0.12	-0.23

[a]$P \leq 0.09$, [b]$P \leq 0.05$, [c]$P \leq 0.01$.

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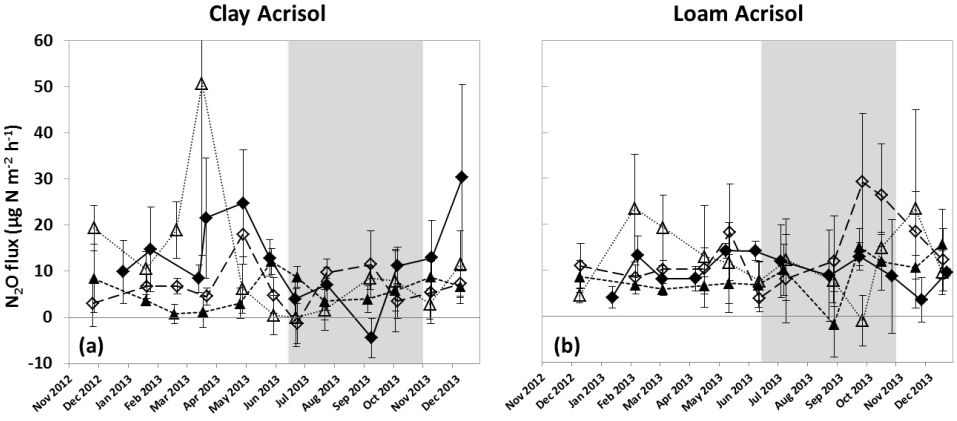

Figure 1. Mean (±SE, *n* = 4 sites) soil N_2O fluxes from forest (◆), jungle rubber (◇), rubber (▲) and oil palm (△), located within the clay (**a**) and loam Acrisol (**b**) landscapes in Jambi, Sumatra, Indonesia. Measurements were carried out monthly from December 2012 to December 2013; grey shadings mark the dry season.

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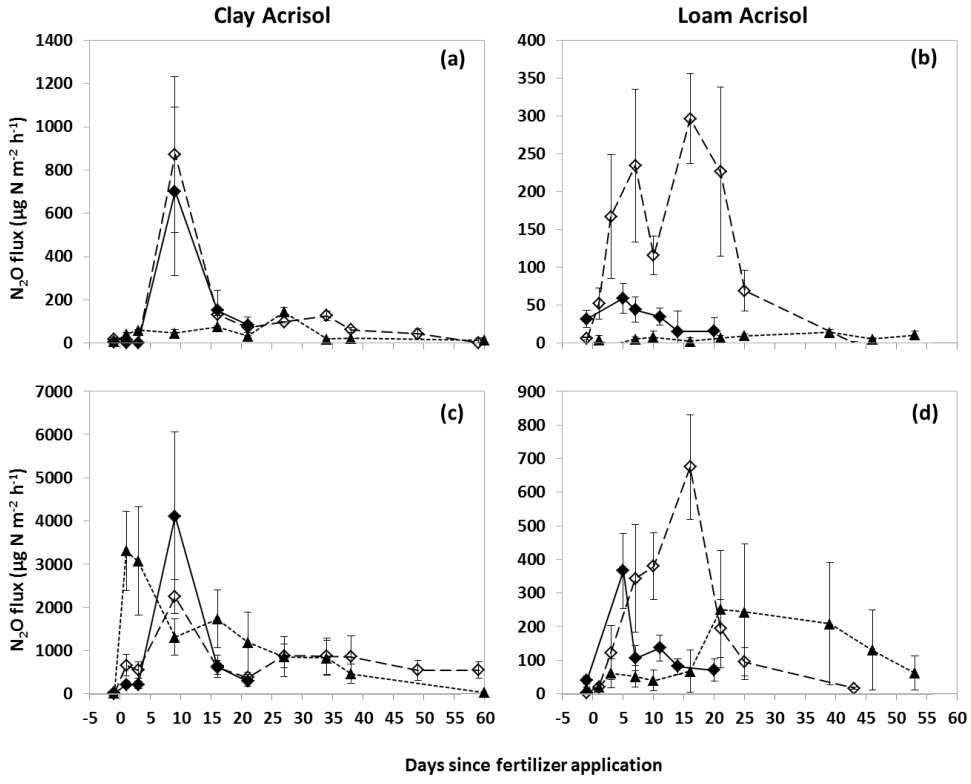

Figure 2. Mean (±SE, *n* = 3 oil palm trees) soil N_2O fluxes during a fertilization in smallholder oil palm plantations 1 (♦), 2 (◇) and 3 (▲) in the clay (**a** and **c)** and loam Acrisol (**b** and **d**) landscapes. Smallholders fertilized around the base of each tree at about 0.8–1 m from the tree base. Fluxes were measured at 0.3 m from the tree base (**a** and **b**) and at 0.8 m on the fertilized location (**c** and **d**) with the same rate and form that smallholders used (see Sect. 2.2).

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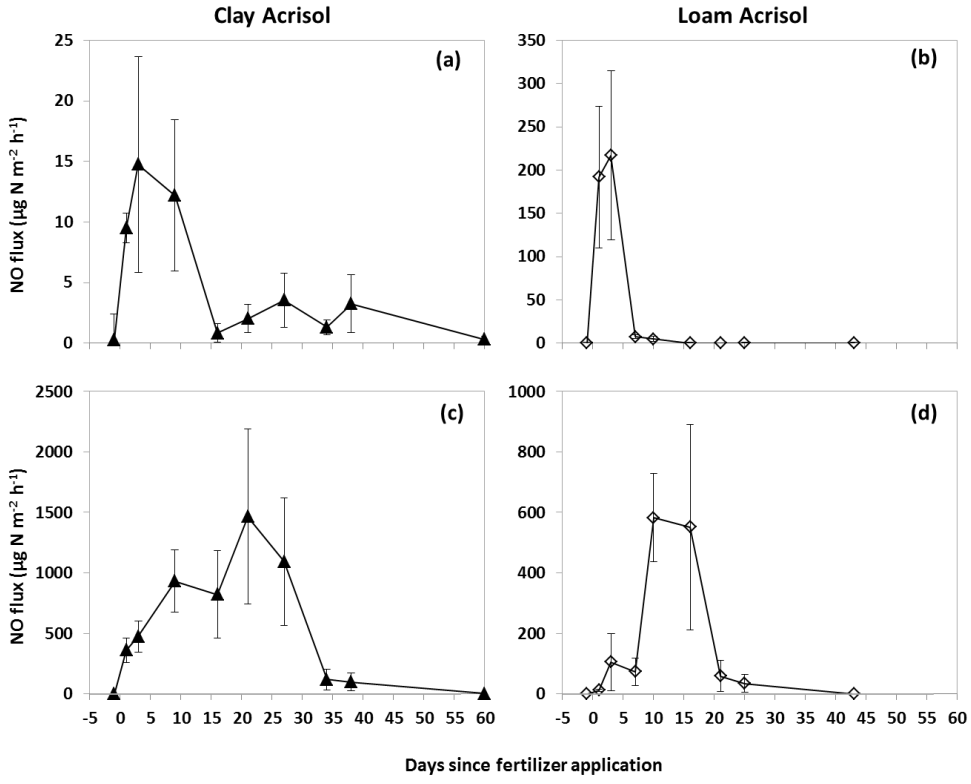

Figure 3. Mean (±SE, *n* = 3 oil palm trees) soil NO fluxes during a fertilization in a smallholder oil palm plantation in the clay (**a** and **c)** and loam Acrisol (**b** and **d**) landscapes. Smallholders fertilized around the base of each tree at about 0.8–1 m from the tree base. Fluxes were measured at 0.3 m from the tree base (**a** and **b**) and at 0.8 m on the fertilized location (**c** and **d**) with the same rate and form that smallholders used (see Sect. 2.2).

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Table A1. Mean (±SE, n = 4 sites) soil physical and biochemical characteristics in the top 0.10 m depth (except sand content with n = 3 sites) from different land uses within each landscape in Jambi, Sumatra, Indonesia. Means followed by different lowercase letters indicate significant differences among land uses within each landscape and different capital letter indicate significant differences between landscapes within each land use (linear mixed-effect models with Fisher's LSD test at $P \leq 0.05$ and marginally significant at $^{*}P \leq 0.09$). These soil characteristics were reported by Allen et al. (2015), except for the sand content (Allen et al., unpublished data).

Soil characteristics	Land-use type			
	Forest	Jungle rubber	Rubber	Oil palm
Clay Acrisol landscape				
Sand (%)	36 ± 11^{a}	27 ± 20^{a}	35 ± 7^{a}	$11 \pm 2^{a,B*}$
Soil C:N ratio	13.1 ± 1.3^{a}	13.0 ± 0.3^{a}	$14.3 \pm 0.6^{a,A}$	13.5 ± 0.2^{a}
Microbial C (mg C kg^{-1})	$1048 \pm 201^{a*,A}$	$922 \pm 223^{ab*}$	$561 \pm 61^{c*}$	$617 \pm 112^{bc*}$
Gross nitrification (mg N kg^{-1} day^{-1})	0.9 ± 0.3^{a}	1.0 ± 0.2^{a}	0.7 ± 0.2^{a}	2.0 ± 0.8^{a}
Loam Acrisol landscape				
Sand (%)	39 ± 8^{a}	42 ± 19^{a}	26 ± 13^{a}	$43 \pm 14^{a,A*}$
Soil C:N ratio	14.3 ± 0.2^{a}	13.7 ± 0.8^{a}	$11.7 \pm 0.7^{b,B}$	12.5 ± 0.5^{ab}
Microbial C (mg C kg^{-1})	$514 \pm 48^{a,B}$	578 ± 45^{a}	461 ± 58^{a}	403 ± 24^{a}
Gross nitrification (mg N kg^{-1} day^{-1})	1.9 ± 0.4^{a}	0.9 ± 0.2^{a}	0.9 ± 0.2^{a}	1.2 ± 0.5^{a}

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Table A2. Mean (±SE, *n* = 4 sites) soil water-filled pore space (WFPS) and extractable mineral N in the top 0.05 m depth for different land uses within each landscape in Jambi, Sumatra, Indonesia, measured monthly from December 2012 to December 2013. Means followed by different lowercase letters indicate significant differences among land uses within each landscape and different capital letters indicate significant differences between landscapes within each land use (linear mixed-effect models with Fisher's least significant difference (LSD) test at $P \leq 0.05$). These soil characteristics were reported by Hassler et al. (2015).

Land-use type	WFPS (%)	NH_4^+ (mg N kg^{-1})	NO_3^- (mg N kg^{-1})
clay Acrisol landscape			
Forest	$72.97 \pm 12.31^{a,A}$	$6.99 \pm 1.03^{a,A}$	$2.15 \pm 0.36^{a,A}$
Jungle rubber	$86.74 \pm 5.93^{a,A}$	$7.33 \pm 0.21^{a,A}$	$0.23 \pm 0.06^{b,B}$
Rubber	$61.49 \pm 7.41^{a,A}$	$4.25 \pm 0.23^{b,A}$	$0.05 \pm 0.01^{b,B}$
Oil Palm	$74.03 \pm 7.28^{a,A}$	$5.80 \pm 0.64^{a,A}$	$0.81 \pm 0.49^{b,}$
loam Acrisol landscape			
Forest	$63.97 \pm 3.30^{a,A}$	$5.94 \pm 0.40^{a,A}$	$0.61 \pm 0.15^{ab,B}$
Jungle rubber	$53.86 \pm 3.70^{a,B}$	$5.64 \pm 0.28^{a,B}$	$1.25 \pm 0.63^{a,A}$
Rubber	$72.58 \pm 5.73^{a,A}$	$4.14 \pm 0.57^{b,A}$	$0.12 \pm 0.02^{b,A}$
Oil Palm	$59.04 \pm 6.74^{a,A}$	$4.20 \pm 1.10^{b,B}$	$0.60 \pm 0.36^{ab,B}$