



Spatial variations of nitrogen trace gas emissions from tropical mountain forests in Nyungwe, Rwanda

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Abstract. Globally, tropical forest soils represent the second largest source of N₂O and NO. However, there is still considerable uncertainty on the spatial variability and soil properties controlling N trace gas emission. Therefore, we carried out an incubation experiment with soils from 31 locations in the Nyungwe tropical mountain forest in southwestern Rwanda. All soils were incubated at three different moisture levels (50, 70 and 90 % water filled pore space (WFPS)) at 17 °C. Nitrous oxide emission varied between 4.5 and 400 μg N m⁻² h⁻¹, while NO emission varied from 6.6 to 265 μg N m⁻² h⁻¹. Mean N₂O emission at different moisture levels was 46.5 ± 11.1 (50 % WFPS), 71.7 ± 11.5 (70 % WFPS) and 98.8 ± 16.4 (90 % WFPS) μg N m⁻² h⁻¹, while mean NO emission was 69.3 ± 9.3 (50 % WFPS), 47.1 ± 5.8 (70 % WFPS) and 36.1 ± 4.2 (90 % WFPS) μg N m⁻² h⁻¹. The latter suggests that climate (i.e. dry vs. wet season) controls N₂O and NO emissions. Positive correlations with soil carbon and nitrogen indicate a biological control over N₂O and NO production. But interestingly N₂O and NO emissions also showed a positive correlation with free iron and a negative correlation with soil pH (only N₂O). The latter suggest that chemo-denitrification might, at least for N₂O, be an important production pathway. In conclusion improved understanding and process based modeling of N trace gas emission from tropical forests will benefit from spatially explicit trace gas emission estimates linked to basic soil property data and

differentiating between biological and chemical pathways for N trace gas formation.

1 Introduction

Nitrogen oxide emissions from soil are of major concern because of their significant impact on atmospheric chemistry and as a driver for global climate (Crutzen, 1979; WMO, 2006; Mosier et al., 1998; Breuer et al., 2000). Nitrous oxide (N₂O) is one of the main contributors to radiative forcing and is becoming the main cause of stratospheric ozone destruction (Meehl et al., 2007). Nitric oxide (NO) acts as a catalyst in the synthesis of tropospheric ozone (Delmas et al., 1997; Holland and Lamerque, 1997) an important component in ecosystem and human health issues. Nitrous oxide and NO are produced in soils by microbial processes of nitrification, denitrification, nitrifier-denitrification (e.g. Davidson et al., 2000). Besides agricultural soils, tropical forest soils are considered as important sources for atmospheric N₂O (Bouwman et al., 1993; Matson et al., 1990; Mosier et al., 1998; Breuer et al., 2000; Butterbach-Bahl et al., 2004; Kiese et al., 2003; Werner et al., 2006). Similarly, NO from tropical forest soils represents a significant source within the global atmospheric budget (Butterbach-Bahl et al., 2004; Gut et al., 2002). However, current estimates of global sources are still highly uncertain because detailed measurements, in

particular, for tropical forest soils are scarce. Kroeze et al. (1999) and Mosier et al. (1998) estimated the contribution of N_2O from tropical forest soils to be in the range of 2.2–3.7 Tg $\text{N}_2\text{O-N yr}^{-1}$. The mean estimate of 3.0 Tg $\text{N}_2\text{O-N yr}^{-1}$ accounts for about 18 % of total N_2O sources (Prather and Ehhal, 2001). Werner et al. (2007a), using a GIS coupled mechanistic biogeochemical model (ForestDNDC-tropica), provided a revised estimate of 1.34 Tg $\text{N}_2\text{O-N yr}^{-1}$ (0.88–2.37 Tg $\text{N}_2\text{O-N yr}^{-1}$).

On the basis of limited available field measurements, the global contribution of tropical forest soils to the global NO budget is estimated at 1.1 Tg NO-N yr^{-1} (Davidson and Kingerlee, 1997). However, a detailed study of Butterbach-Bahl et al. (2004) indicates that NO emission from tropical rain forest soils might be as high as 3 Tg NO-N yr^{-1} .

Further, simulation results, using an N isotopic coupled mechanistic biogeochemical model, show that total gaseous losses, including N_2 , from tropical rain forest soils in Hawaii contributed for ~26–48 % of total N losses from natural ecosystems (Bai and Houlton, 2009).

So far current research and policy programs for tropical forests mainly focus on C storage and emission (Stickler et al., 2009; Defries et al., 2010). Though, tropical forest ecosystems play an important role in the global C balance, also non- CO_2 greenhouse gases contribute to the net greenhouse gas balance from tropical forest ecosystems. For this reason, N_2O and NO emission inventories in tropical forest are required as an additional decision tool for sustainable forest management and closing global trace gas budgets.

Despite the importance of tropical rain forest soils as source of atmospheric N_2O and NO, there are only few datasets available (Serca et al., 1994; Breuer et al., 2000; Kiese et al., 2005; Butterbach-Bahl et al., 2004; Gut et al., 2002; Werner et al., 2007b). The majority of N_2O measurements were conducted in the Amazon, Central America (e.g. Keller and Reiners, 1994; Verchot et al., 1999) and in tropical regions of Australia (e.g. Kiese et al., 2003; Butterbach-Bahl et al., 2004). The only datasets of N_2O emissions from tropical rain forest soils of Africa was reported by Serca et al. (1994) and Werner et al. (2007b), who worked in the Mayombe forest in the DR Congo and the Kakamega forest in Kenya, respectively. Furthermore, most studies on NO were carried out in the Amazon (Gut et al., 2002) and Queensland, Australia (Butterbach-Bahl et al., 2004). To our knowledge except for Serca et al. (1994), no reports on soil NO emissions from African tropical forest soils are available to date, indicating a need for an improved data availability of N-trace gas exchange for African forest soils in particular.

It is widely accepted that the magnitude of forest soil N_2O and NO emissions is highly variable and strongly influenced by environmental conditions, like soil properties and soil moisture (e.g. Davidson, 1993; Breuer et al., 2000). Additionally, knowledge of how soil properties and changes in soil moisture regulate the emission of these N oxide gases

is essential for realistic predicting soil-atmosphere N trace gas exchange. Even though variations in soil moisture are considered as the main driver of temporal variations in N oxide emissions, it does not explain observed spatial variations from site to regional scales. Based on current knowledge it is obvious that observed spatial variations in N trace gas emissions must be closely linked to variations in soil properties, though effects of soil properties on N trace gas emissions are largely unexplored in the tropics. So far, only a few studies are available where variations in soil properties have been explicitly linked to spatial variation of N_2O and NO emissions from tropical soils (e.g. Breuer et al., 2000; Ishizuka et al., 2005; Keller et al., 2005; van Haren, 2010).

However, more detailed information on the linkage between soil properties and N oxide emissions are needed to better understand spatial variability of N trace gas emissions in tropical forest regions. Furthermore, this information may be used for improving the parameterization and for validating predictions of mechanistic biogeochemical model such as the ForestDNDC-tropica (Werner et al., 2007a), which have recently been used to estimate global N_2O emissions from tropical rainforest soils.

In-situ N-trace gas fluxes for plot-based studies are usually derived with static (N_2O) or dynamic (NO) chamber measurements (see e.g. Kiese and Butterbach-Bahl, 2002). However, for regional scale flux estimates this approach is not feasible (especially not for remote areas) since it concentrates measurements only on a few sites with limitations to cover all climatic, soil and vegetation characteristics of a given region. Soil incubation studies in the laboratory have the advantage to allow for measuring N-trace gas emissions from a high number of samples, thereby allowing to covering regional variability of soils. Several previous studies reported N-trace gas fluxes, which were derived from soil incubation experiments and which agreed well with field-derived fluxes determined via chambers (Gut et al., 1999; Otter et al., 1999; Ludwig et al., 2001, van Dijk et al., 2002).

In view of the fact that few results on N_2O and NO fluxes from African tropical rain forest soils have been reported and the limited knowledge on their spatial variability and control factors, we present results of N_2O and NO emissions from an incubation experiment using 31 different soil samples from the Nyungwe tropical mountain forest in southwestern Rwanda. Our aims were (a) to quantify the magnitude and spatial variability of soil N_2O and NO emissions, and (b) to determine the importance of different soil properties for explaining N_2O and NO emissions from this central African tropical mountain forest.

2 Methods

2.1 Site description and soil sampling

The study was conducted in the Nyungwe National Park covering an area of about 1000 km² of tropical mountain forest, located between 2°17'–2°49' S and 29°03'–29°29' E at altitudes of 1485–2925 m in southwestern Rwanda (Fig. 1). Nyungwe is one of the largest mountainous rainforest areas remaining in Africa. It divides the Nile and the Congo river basins. The soils have been developed mainly from schists, micaschists, quartzitic schists and granites (UGent/Minagri, 2000a, b, c, d). The eastern part of the forest, with an altitude exceeding 2000 m, is dominated by micaschists, whereas the western area featuring lower altitudes (<2000 m) shows schists as dominant parent material. Figure 1 illustrates the difference in elevation and parent material from the western to eastern part of the forest.

The forest contains various ecosystems ranging from dense forest, bamboo groves to marshes, and contains approximately 1105 plant species, as well as high biodiversity of fauna. Many species are endemic for the area and the central African Highlands (Graham et al., 1995; Sun et al., 1996; Masozera and Alavalapati, 2004; Plumptre et al., 2007; Fischer and Killmann, 2008).

The average annual precipitation from 1974 to 1989 as measured by the seven weather stations located in the vicinity of the Nyungwe forest (Fig. 1a) is 1660 mm (ranging from 1308 to 2071 mm). In the dry seasons (June to August) monthly precipitation is below 80 mm per month and generally above 130 mm during the other months. The average monthly minimum and maximum temperature is 11 and 23 °C, respectively. The average annual temperature was 17 °C, with small seasonal variations (monthly mean range: 17.0 °C in June to 17.6 °C in April) (Minagri and CTB/BTS, 1993a, b). Weather data was not available for more recent years, but an automatic climate station has been established inside the Nyungwe forest (2°28'–43.3" S, 29°12'00" E) on February 2007 (Fig. 1a). The average annual mean temperature and precipitation at this location is 14.7 °C and 1706 mm, respectively (Nsabimana, 2009).

Soil sampling in the Nyungwe forest has been performed during September 2009 at 31 different locations (Fig. 2) covering all major soil types and climate conditions in the region. Soil core samples were taken from 0 to 10 cm depth. The soil samples were composite samples consisting of 5 different subsamples per location, which were taken in an X-shaped pattern (in each leg head and in the center of the X) of two meters from each head with a length of ~2.8 m for each leg. All soil samples were immediately air-dried to reduce microbial activity during transportation to laboratory facilities in Belgium.

Total soil C and N content were measured by an elemental analyzer coupled to an isotope ratio mass spectrometer (EA-IRMS) (20–20, SerCon, Crewe, UK). Soil texture

was determined by the Andreasen pipette method by sieving (Allen, 1975). The pH was measured by suspending dried soil samples in water (1:10 soil/water) and a glass electrode. Free iron (Fe oxides outside the silicate lattices) was measured upon sodium dithionite extraction (Mehra and Jackson, 1960) and measured via optical emission spectrometer (Varian ICP-OES) (720 ES, Mulgrave VIC 3170, Australia).

2.2 Laboratory experimental conditions for soil N₂O and NO measurements

After air-drying the soil was homogenized by sieving (2 mm mesh size). The soil samples in two replicate were incubated in the laboratory in tubes of 2.6 cm diameter, 9 cm height and 47.78 cm³ volume. The mass of dry soil needed was calculated from the measured field bulk density (ρ_b) with the known volume ($\rho_b = \frac{\text{mass of dry soil}}{47.78 \text{ cm}^3}$). The soil moisture content was brought to three levels (50, 70 and 90 %) of water-filled pore space (WFPS) and kept constant during the incubation. The lowest water content in this study was 50 %, which is e.g. equivalent to lowest WFPS for Kakamega forest soils in Kenya (~40 % WFPS with daily precipitation only by <5 mm) (Werner et al., 2007b). The air temperature during the incubation was kept constant at 17 °C. To avoid that flux measurements are biased by the exponential increase of microbial activity upon remoistening the soil samples, all the soil incubation tubes were first pre-incubated at the three moisture levels at 17 °C for 7 days.

2.3 Measurements of soil N₂O and NO production

The incubation tubes were placed into airtight sealed glass containers with a volume of 1200 ml. Measurements of N oxide gases were performed at day 1, 3 and 5 upon 7 days pre-incubation. The headspace concentration of NO of each incubation tube sample was measured four times (0, 40, 80 and 120 min after closing the glass containers) during each measuring day.

NO concentration of the headspace was determined using a NO analyzer (CLD 77AM, Eco Physics, Switzerland) with a continuous flow rate of 55 ml min⁻¹. The detection is based on the chemoluminescence of the oxidation of NO to NO₂ in the presence of ozone (O₃).

The NO analyzer was calibrated using an NO reference gas, with a known concentration of 9.8 ppmv ± 0.5 in N₂. Following measuring NO concentrations at 0, 40, 80 and 120 min, the same containers were used to sample N₂O gas. Immediately after the NO measurements, 12 ml gas sample was withdrawn from the headspace by a syringe. The gas sample was transferred to a 12 ml vacutainer tube and stored for later measurements of N₂O concentration. Since less than 10 % of the headspace was used during the measurement the effect of change in pressure has been ignored in view of the large variability of observed NO and N₂O fluxes.

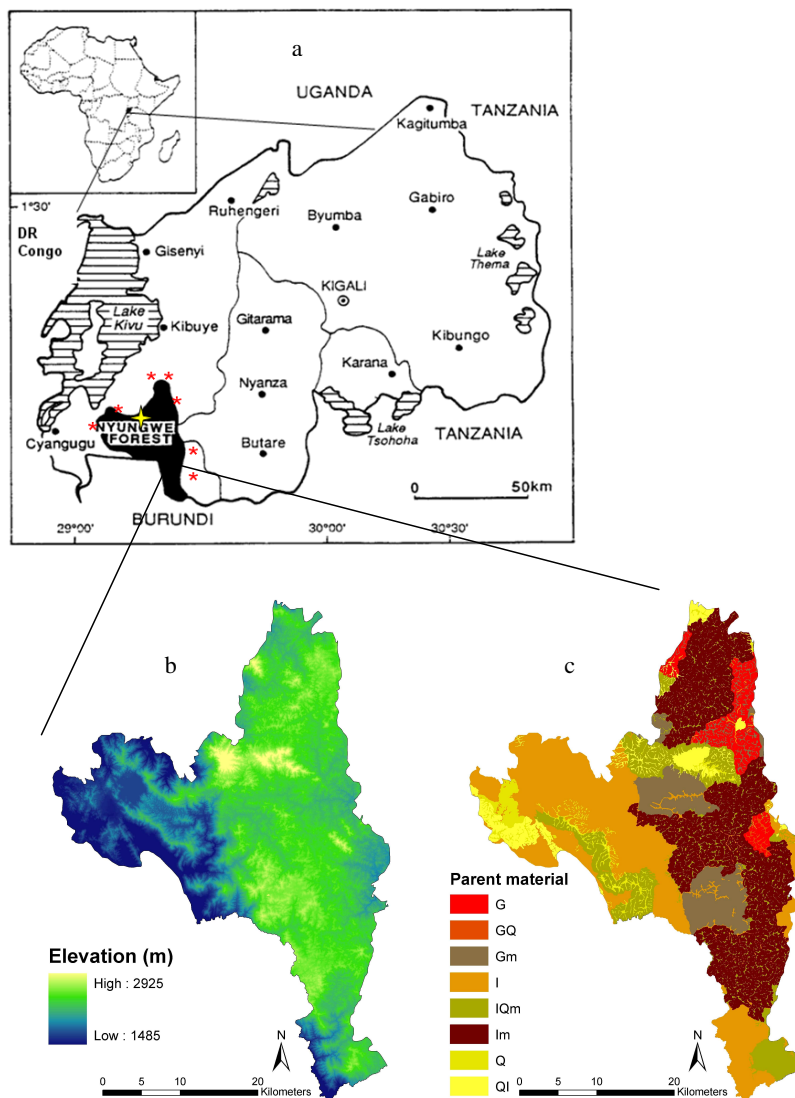


Fig. 1. Location of the Nyungwe forest in southwestern Rwanda; seven climate stations around the forest are shown by red stars; the recent climate station in the Nyungwe forest is shown by a yellow star (a), elevation map (b) and parent material map of the Nyungwe forest: Q = quartzite; QI = quartzite intercalated with schists; IQm = quartzite intercalated with micaschists; GQ = granitic and quartzitic rocks; G = acid rocks (granite); Gm = micaceous acid rocks (granitoides); Im = micaschists; I = schists (c).

As a check on the reliability of NO measurements at each moment in time containers with known initial NO concentrations (24.5, 49.8 and 99.7 ppb) were measured during the experiment to determine the leakiness of the glass containers, and the soil measurements were corrected accordingly with a correction magnitude of less than ca. 13%. The N₂O concentration was determined via a Gas Chromatograph (14B, Shimadzu, Japan) equipped with an electron capture detector ECD). One ml from the 12 ml vacutainer was injected into the GC with a Hamilton airtight syringe. The GC analysis of the samples was always accompanied by measurements of varying amounts of a reference gas with a known

concentration of 2.46 ± 0.12 pmv in order to construct a calibration curve for N₂O.

The ideal gas law in combination with the molecular weight of N₂O and NO was used to calculate headspace gas concentrations in $\mu\text{g l}^{-1}$. Those values were then converted to gas fluxes ($\mu\text{g kg}^{-1}$ dry soil h^{-1}) by multiplying with the headspace of the incubation jar, dividing with the amount of soil used and linear regression of changing headspace concentrations over time. The NO and N₂O fluxes were finally recalculated to $\mu\text{g m}^{-2} \text{h}^{-1}$ using a known exposed soil surface area.

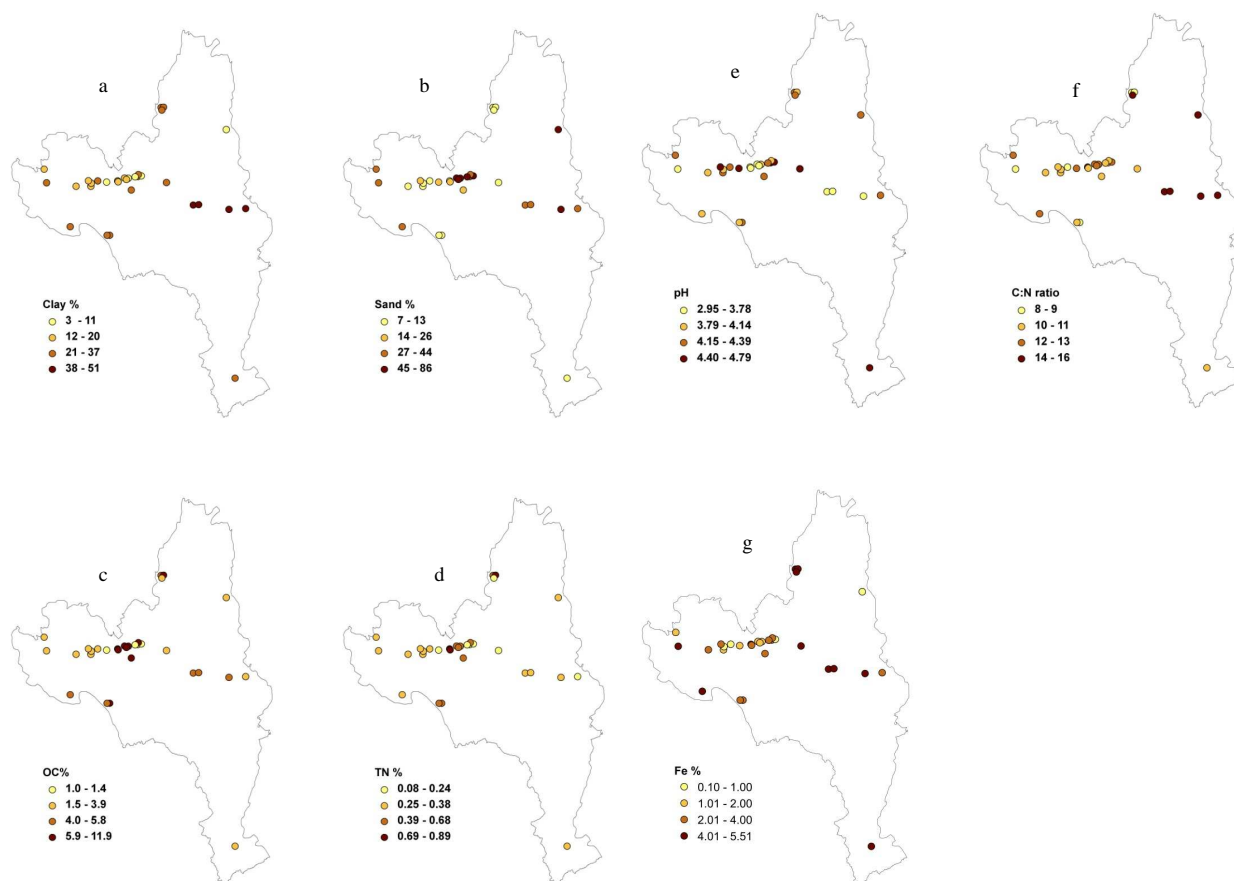


Fig. 2. Spatial pattern of the soil characteristics of the samples sites in the Nyungwe forest: clay (a), sand (b), organic carbon (OC) (c), total nitrogen (TN) (d), pH (e), C:N ratio (f), free iron (Fe) (g).

2.4 Statistical analysis

Statistical analysis was performed using the statistical package SPSS 16 (SPSS inc., 2007). Using the Kolmogorov-Smirnov goodness-of-fit test, each N oxide gas emission rate showed a log-normal distribution. Therefore, log-transformed data were used to make comparisons between groups and all data were tested at the 5 % significance level. All data given in tables and figures are accompanied by the standard error (error bars).

3 Results

3.1 N₂O and NO production rates

All N₂O and NO fluxes showed significant effects for soil moisture. Multiple comparisons of all N₂O fluxes (log-transformed data) from all sites showed a significant difference (Table 1) between the three applied WFPS levels. In all cases the highest fluxes were observed for 90 % WFPS. A significant positive correlation was observed between N₂O fluxes and WFPS for the entire dataset (Fig. 3). The N₂O

emission rate averaged over the 31 locations for day1, day3 and day5 were 40.2, 43.2 and 56.3 $\mu\text{g N m}^{-2} \text{h}^{-1}$, respectively for 50 % WFPS, 70.7, 70.9 and 73.6 $\mu\text{g N m}^{-2} \text{h}^{-1}$ for 70 % WFPS and 88.3, 109.6 and 98.3 $\mu\text{g N m}^{-2} \text{h}^{-1}$ for 90 % WFPS (Table 1). In contrast to N₂O emissions, NO emissions decreased with increasing soil moisture, with significant differences between the three applied WFPS levels. Consequently, the NO emission rates were significantly negatively correlated with WFPS (Fig. 3). The mean NO emission rate for day1, day3 and day5 were 59.9, 61.3 and 86.6 $\mu\text{g N m}^{-2} \text{h}^{-1}$, respectively for 50 % WFPS, 40.3, 43.3 and 57.6 $\mu\text{g N m}^{-2} \text{h}^{-1}$ for 70 % WFPS and 33.9, 33.3 and 41.0 $\mu\text{g N m}^{-2} \text{h}^{-1}$ for 90 % WFPS (Table 1).

3.2 Variation of soil properties and N₂O and NO gas emission rates

The soil textural classes at the sampled sites were classified (USDA classification system) as clay (C), clay loam (CL), sandy loam (SL), silt loam (SiL), loamy sand (LS), silty clay (SiC) and silty clay loam (SiCL). Table 1 and Fig. 2 show summary statistics and distribution for soil characteristics

Table 1. Range, median, mean, standard error (SE), skewness and kurtosis of soil characteristics^a, N₂O and NO flux rates (measured at day 1, 3, and 5 after pre-incubation). The letters indicate significant differences ($P < 0.05$) for mean N trace gas fluxes between different water-field pore space (WFPS).

		Range	Median	Mean	SE	Skewness	Kurtosis
N₂O ($\mu\text{g N m}^{-2}\text{h}^{-1}$)		<i>n</i> = 31					
WFPS 50 %	Day 1	5.8–226.6	21.5	40.2	10.4	2.5	5.7
	Day 3	4.5–249.2	19.0	43.2	11.4	2.2	3.9
	Day 5	7.7–244.2	34.1	56.3	12.7	1.9	2.5
WFPS 70 %	Day 1	38.2–297.5	46.7	70.7	11.7	2.5	5.6
	Day 3	31.1–245.9	51.2	70.9	11.1	1.9	2.5
	Day 5	15.9–276.6	56.4	73.6	12.7	1.8	2.3
WFPS 90 %	Day 1	45.5–400.5	55.6	88.3	16.4	2.9	8.2
	Day 3	41.0–400.5	81.8	109.6	17.0	2.3	4.7
	Day 5	31.8–400.0	80.3	98.3	17.0	2.2	4.2
average 1st, 3rd, 5th day							
WFPS 50 %	CV ^b 133.3	8.8–238.4	23.6	46.5a	11.1	2.1	3.2
WFPS 70 %	CV 89.9	36.6–266.6	47.0	71.7b	11.5	2.2	3.4
WFPS 90 %	CV 92.5	46.2–400.2	66.0	98.8c	16.4	2.5	5.3
NO ($\mu\text{g N m}^{-2}\text{h}^{-1}$)		<i>n</i> = 31					
WFPS 50 %	Day 1	0.7–260.8	55.7	59.9	8.2	2.8	12.1
	Day 3	0.8–231.1	52.1	61.3	7.9	1.9	6.2
	Day 5	2.6–265.5	64.5	86.4	13.5	1.2	0.3
WFPS 70 %	Day 1	2.6–118.6	33.4	40.3	5.1	1.2	1.0
	Day 3	4.3–113.7	36.2	43.3	5.1	0.99	0.4
	Day 5	2.3–173.5	46.0	57.6	8.1	1.3	1.3
WFPS 90 %	Day 1	3.4–127.5	28.8	33.9	4.7	1.9	4.9
	Day 3	2.8–72.1	28.7	33.3	3.0	0.6	−0.2
	Day 5	0.9–144.5	41.4	41.0	5.7	1.7	4.1
average 1st, 3rd, 5th day							
WFPS 50 %	CV 75.5	1.3–252.5	53.3	69.3a	9.3	1.6	3.7
WFPS 70 %	CV 68.8	4.6–128.8	36.4	47.1b	5.8	1.2	1.0
WFPS 90 %	CV 65.2	2.9–108.9	31.7	36.1c	4.2	1.5	2.8
Soil properties							
Clay (%)	CV 47.4	3.0–51.0	28.0	26.1	2.2	−0.1	−1.0
Sand (%)	CV 72.3	7.0–86.0	26.0	30.4	3.9	0.7	−0.3
OC (%)	CV 50.9	1.0–11.9	3.9	4.5	0.4	1.0	2.1
TN (%)	CV 51.8	0.08–0.89	0.35	0.4	0.4	0.6	−0.2
C:N	CV 11.2	7.7–16.3	11.1	11.5	0.4	0.4	−0.5
pH	CV 9.1	2.9–4.89	4.2	4.1	0.6	−0.92	0.82
Fe (%)	CV 52.1	0.1–5.5	3.04	3.03	0.3	−0.24	−1.02

^aOC = organic carbon, TN = total nitrogen, C:N = organic carbon to total N ratio, Fe = free iron (Fe oxides from outside the silicate lattices)

^bCV = coefficient of variation in %

of the studied sites. The sites in the eastern and northeastern corners of the forest (Fig. 2a) showed high clay content (>37%). The highest silt contents were found at sites in the northern part of the forest with values above 50%. The sites in the northwestern corners had a very low fraction of sand (<14%). The forest is dominated by strongly leached acid soils (pH ranging from 2.9 to 4.8). Areas with high soil organic carbon (OC) contents (>4%, Fig. 2e), corresponding with high total nitrogen (TN) contents, were found in

the northern and northeastern parts of the forest. OC content were in general high and ranged from 3.6 to 11.9%, total N contents ranged from 0.08 to 0.89% and C:N ratio ranged from 7.7 to 16.3 in the analyzed samples. Free iron ranged from 0.1 to 5.5% and high free iron contents (>3%, Fig. 2g) were mostly present in sites of the eastern part of the forest, where altitude is higher and micaschists dominate the soilscape (Fig. 1). The variation in silt, sand, clay content and free iron (formed through weathering) reflects the

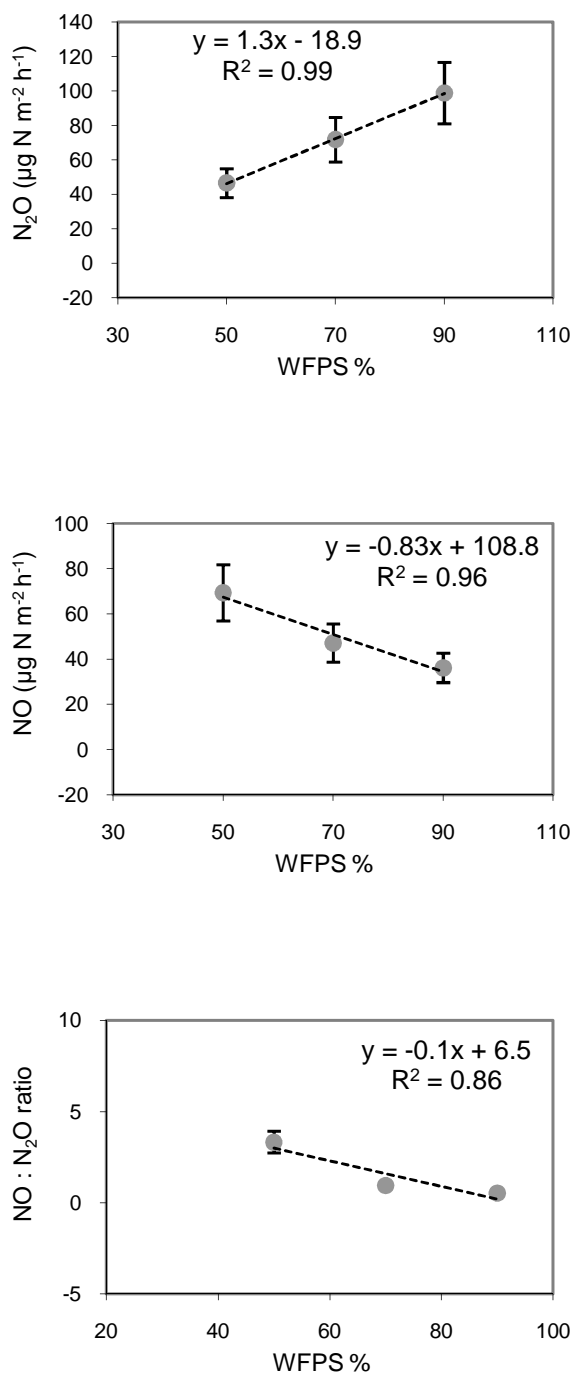


Fig. 3. Correlation of water-filled pore space (WFPS) and N₂O, NO emission rates and NO:N₂O ratio (mean of 31 sites, error bars indicate plus minus standard error).

lithological variation in the forest. Coefficients of variation (CV) ranged between 45 and 72 % for all soil variables except for pH (CV = 9 %) and C:N ratio (CV = 11 %).

The N₂O and NO emission rates showed a large spatial variation and a highly skewed distribution. Coefficients of

variation for the N₂O emission rates were 133.3, 89.9 and 92.5 %, respectively for 50, 70 and 90 % WFPS. Coefficients of variation of the NO emission rates for 50, 70 and 90 % WFPS were 75.5, 68.8 and 65.2 %, respectively. The N₂O emission rates ranged from 4.5 to 400.5 µg N m⁻² h⁻¹ and the NO emission rates from 6.6 to 265.5 µg N m⁻² h⁻¹. The range and skewness of the N₂O emission rates was higher than the NO emission rates. The majority of soil samples showed relatively low N₂O emission rates, with the exception of a few sites (large skewness). Figure 4 shows the spatial variation of the N₂O and NO emission rates for the three WFPS levels considered. It also illustrates that measured N₂O emissions varied substantially over the study region. Compared to N₂O, the spatial variability of NO emission rates was more or less opposite (Fig. 4).

The NO:N₂O emission ratio (NO:N₂O) varied on average from 3.3 to 0.9 and 0.5, respectively for 50, 70 and 90 % WFPS. The NO:N₂O ratio showed no marked trend for soil samples taken across the study region (data not shown). Increased moisture levels resulted in a decrease of the measured NO:N₂O ratio (Fig. 3).

3.3 Correlation between N₂O and NO gas emission rates and soil properties

For co-variation analysis we only used N trace gas data at 50 % WFPS, since spatial differences between sampling sites were most pronounced for this treatment. The variation in N₂O emissions was positively correlated with clay and free iron content and negatively with pH and silt content (Table 2). No significant correlation was observed for OC and TN contents. In contrast, NO emission rates were significantly positively correlated with OC, TN and free iron content and showed a weak significant negative correlation with C:N ratio.

These contradictory results could be due to the large variance in N₂O and NO emission. Nitrous oxide emission rates for sites at low altitude (<2000 m) in the western part of the forest (where schists dominate the soilscape) were relatively low (e.g. 8.8–130 µg N m⁻² h⁻¹ for 50 % WFPS) compared to higher N₂O emission rates (e.g. up to 238 µg N m⁻² h⁻¹ for 50 % WFPS) for samples taken from higher altitude in the eastern part (where micaschists dominate the soilscape). High NO emissions were distributed among sampling sites with high OC and TN content.

This result indicates that variation in parent material and topography (Fig. 1b, c), which largely control variation in soil properties, exerts a primary control on especially N₂O emission. Therefore, we separately investigated how correlation between N₂O and NO emissions and soil properties varied for low altitude, where schists dominate and high altitude, where micaschists dominate. We did this by creating a different sub dataset according to variation in parent material and altitude: data subset 1: locations at high altitude and micaschists as dominant parent material; and data

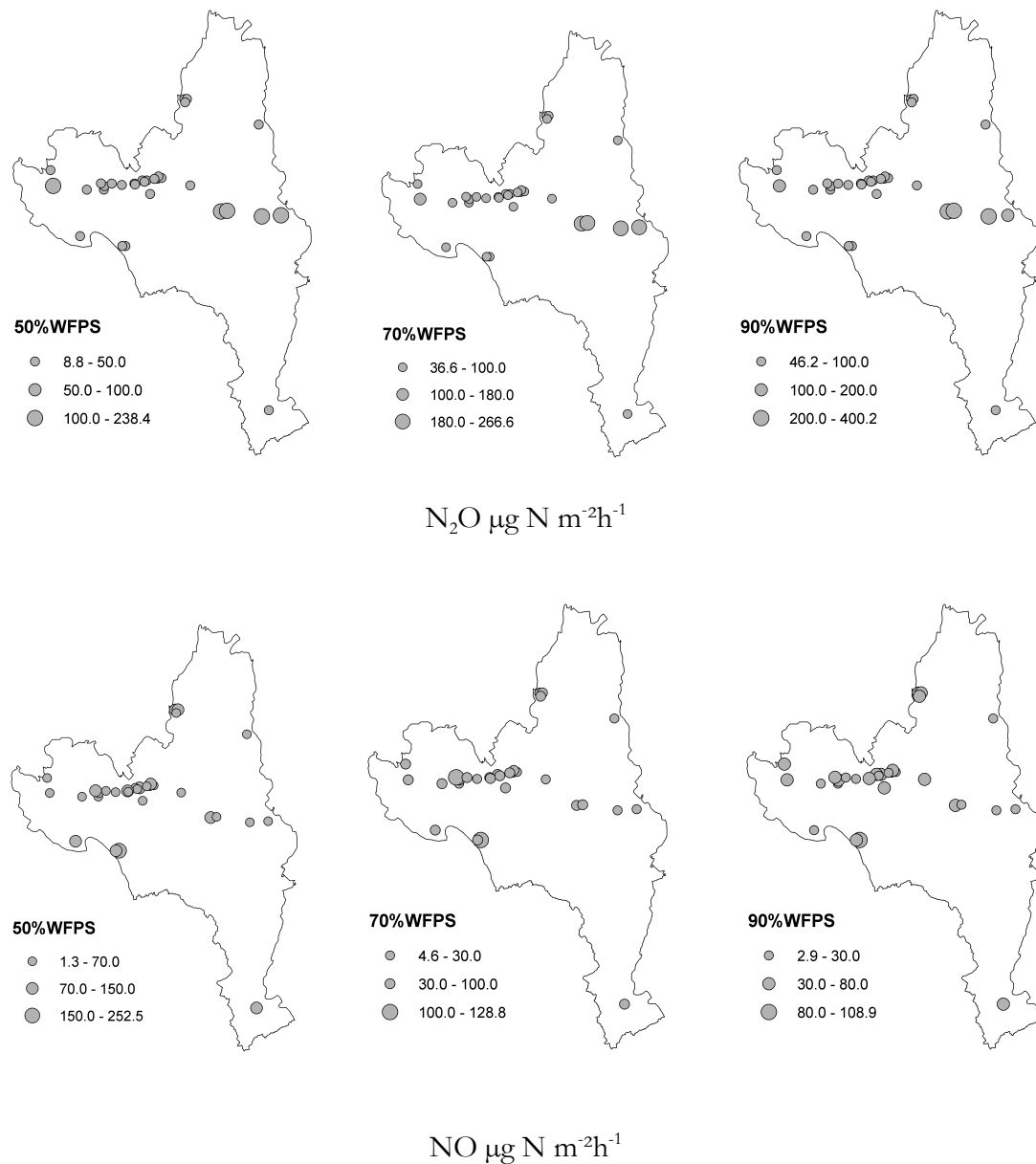


Fig. 4. Spatial distribution of N_2O (top) and NO (bottom) emission rates in the Nyungwe forest.

subset 2: locations at low altitude and schists as dominant parent material.

For data subset 1, the N_2O emission rate was significantly and positively correlated with OC and clay content and negatively correlated with pH. The NO emission rate showed significantly negative correlation with C:N ratio ($p < 0.05$) and positive correlation with OC and TN. Both N_2O and NO correlated positively with free iron content but not significant.

For data subset 2, N_2O emissions were significantly and positively correlated with the OC, TN clay and free iron content. NO emission was positively, significantly correlated

with TN and free iron. The correlation was also positive with OC but not significant.

4 Discussion

To date, there exist very few coarse-scaled (large area as to exhibit great spatial variability of soil parameters), data-intensive N-oxide flux data for tropical forest ecosystems. However, spatial variability is essential for extrapolation and mechanistic understanding of N_2O and NO emission (Breuer et al., 2000; Kiese and Butterbach-Bahl, 2002; Werner et

Table 2. Correlation coefficients for N₂O and NO emission rates and soil properties (silt and clay content; organic carbon (OC), total nitrogen (TN), C:N ratio pH, and free iron (Fe)) for two different data subsets data subset 1: locations at high altitude and micascists as dominant parent material; and data subset 2: locations at low altitude and schists as dominant parent material. In this analysis we only include N oxide emission data from the 50 % WFPS; **indicates significant correlations at $P < 0.01$ and *indicates significant correlations at $P < 0.05$.

		Clay	Silt	Sand	OC	TN	C:N	pH	Fe
Data Subset1	N ₂ O	0.59*	-0.57	0.07	0.68*	0.25	0.65	-0.77*	0.26
	NO	0.19	0.58	-0.56	0.61*	0.81**	-0.63*	-0.09	0.42
Data Subset2	N ₂ O	0.51*	-0.09	-0.14	0.50*	0.63**	-0.12	-0.35	0.50*
	NO	0.31	-0.07	-0.08	0.39	0.49*	-0.07	0.21	0.53*
Entire dataset	N ₂ O	0.58*	-0.52*	0.16	0.02	-0.15	0.45	-0.59**	0.38*
	NO	0.13	0.27	-0.33	0.49**	0.63**	-0.36*	0.02	0.40*

al., 2007b). Understanding spatial N trace gas emission estimates further requires constrained information of soil and environmental properties. To the best of our knowledge, our N₂O and NO flux estimate for 31 different locations in the Nyungwe tropical forest in southwestern Rwanda represent the first large, coarse scale, spatial dataset of N₂O and NO emission on the African continent.

The N₂O emission rates we measured during laboratory incubations were comparable in magnitude with N₂O fluxes from field measurements at the Kakamega forest sites in Kenya (1.1–324.8 $\mu\text{g N m}^{-2} \text{h}^{-1}$; Werner et al., 2007b). With mean N₂O emission rates ranging from 46.3 to 98.8 $\mu\text{g N m}^{-2} \text{h}^{-1}$ the Nyungwe soil emitted comparable levels of N₂O as reported by Serca et al. (1994) for the Mayombe forest soils in Democratic Republic of Congo (mean flux in the rainy season: 19.6 $\mu\text{g N m}^{-2} \text{h}^{-1}$ and mean flux at the end of rainy season: 207 $\mu\text{g N m}^{-2} \text{h}^{-1}$). Furthermore, with an average clay, silt and OC content of 26 %, 44 % and 4.4 %, respectively, the Nyungwe forest soils are similar to the Kakamega (Kenya) rainforest soils (Werner et al., 2007b), (34 % clay, 23 % silt and 3.5 % C content) and the Mayombe forest soils (1.7–4.5 % C content).

Only Serca et al. (1994) measured NO fluxes from African tropical forest. Their results for the Mayombe forest in DR Congo: 14.4 $\mu\text{g N m}^{-2} \text{h}^{-1}$ at the end of a rainy season (mean flux rainy season 4.1 $\mu\text{g N m}^{-2} \text{h}^{-1}$) were in the lower range of our NO emissions. In our study we measured individual N₂O fluxes up to 558.5 $\mu\text{g N m}^{-2} \text{h}^{-1}$, which is of similar magnitude as the reported maximum N₂O fluxes of 324.8 $\mu\text{g N m}^{-2} \text{h}^{-1}$ for Kakamega rainforest soils following periods of intensive rainfalls (Werner et al., 2007b). The coefficient of variation between different sampling sites in the Nyungwe for N₂O emission rates at three WFPS were in good agreement with results from Werner et al. (2007b), who reported values of 52.8–147.9 % for the spatial variation of N₂O fluxes for the Kakamega forest in Kenya.

Nitrogen trace gas emissions in tropical environments are predominantly governed by WFPS (Davidson, 1991;

Kiese and Butterbach-Bahl, 2002; Werner et al., 2007b). The observed decrease of the NO:N₂O ratio with increasing WFPS (Fig. 3) indicates that denitrification or nitrifier-denitrification processes was possibly the main pathway of N₂O gas production rather than nitrification. This assumption is in good agreement with previous observations in rain forest ecosystems in Australia (Kiese and Butterbach-Bahl, 2002; Butterbach-Bahl et al., 2004) and Kenya (Werner et al., 2007b). Based on previous studies, maximum N₂O emission rates are reported at WFPS values between 50 and 80 %, though other studies suggest slightly higher ranges of 60 to 90 % (Davidson, 1991; Kiese and Butterbach-Bahl, 2002; Werner et al., 2007b). In our study maximum N₂O emission occurred at 90 % WFPS, conditions whereby the denitrification process normally favors N₂ formation. The relationship between N₂O emissions and WFPS was suggested to be linear for Australian rain forest soils (WFPS ranging from 10 to 50 % (Butterbach-Bahl et al., 2004) and exponential for Kakamega rain forest soils in Kenya (WFPS ranging from 40 to 75 % (Werner et al., 2007b)). For our data, using a linear function (Fig. 3), 99 % of the variability of the N₂O emissions could be explained by changes in soil moisture content. Keller et al. (2005) also found significant relationship between N₂O emissions and soil moisture for tropical forest site in Brazil.

Maximum NO emissions can be expected at WFPS <50 % depending on soil physical properties. A quadratic function was reported for the relation between NO emissions and WFPS for Australian rain forest soils (Butterbach-Bahl et al., 2004) (WFPS levels ranging from <5 % to 50 %).

For our data a linear function was able to describe 96 % of the variation between NO emission rates and WFPS (Fig. 3). The NO fluxes at low moisture content (50 % WFPS) were higher than those at medium (70 % WFPS) and pronounced higher than those at high soil moisture (90 % WFPS). These results are supportive to several other studies which were showing a strong reduction in soil NO

production if soil moisture is increasing to values $> 50\%$ WFPS (e.g. Butterbach-Bahl et al., 2004).

In contrast to NO emission, correlation between N_2O emission and OC and TN only appeared in the data subsets. In the western part of the forest, where altitude is lower and schists dominate the soilscape, both OC, but also TN showed a correlation with N_2O emissions. In the eastern part of the forest (higher altitude and micascists) both OC and TN were correlated with NO emissions. Further, pH seems to be an important controlling factor as it appeared in the entire data set and sub-dataset 1.

Positive correlations of N_2O emissions with OC and TN are in agreement with Booth et al. (2005), who showed a direct effect OC and TN on N_2O emission for soils from a wide range of ecosystems. It is well known that low pH decreases the activity of the N_2O -reductase, thereby increasing production of N_2O from denitrification (Nömmik, 1956; Weier and Gillam, 1986; Granli and Bockman, 1994). For nitrification, it has also been demonstrated that low pH values favor N_2O production (Sitaula and Bakken, 1993; Martikainen and De Boer, 1993; Kesik et al., 2006). Furthermore, it has been shown by Kiese and Butterbach-Bahl (2002) that low pH was a crucial factor driving high N_2O emissions from coastal lowland soils in an Australian rainforest.

The importance of soil N for explaining N_2O and NO fluxes may be explained by the fact that nitrogen availability is one of the primary controlling parameters for organic matter mineralization, whereby mineralization is delivering inorganic N for nitrifying and denitrifying bacteria. Nitrogen content, as an important controlling parameter for NO emission, was also clearly indicated in the study of Pilegaard et al. (1999) analyzing NO emission rates from temperate forest soils across a wide range of sites in Europe. The negative effect of the C:N ratio on the NO emission rate is similar to that reported by Stark et al. (2002), indicating that the C:N ratio affected the NO emission rate from forest soils of Western North America by controlling N mineralization.

Breuer et al. (2000), investigating different rain forest sites in tropical Queensland, Australia, reported that nitrate concentration and WFPS content at Kauri Creek; soil pH and nitrate concentration at Lake Eacham; and for all sites CO_2 emission, C:N ratio and WFPS content were the most influential factors driving small scale (< 100 m) spatial variations in soil N_2O emissions. Furthermore, Werner et al. (2007b) explained site differences in N_2O emissions for different sites in the tropical rainforest of Kakameka, Kenya, by differences in the C:N ratio and clay content. Mapanda et al. (2010) also indicated that variability of N_2O emissions from Miombo woodland in Zimbabwe is controlled by changes of soil moisture, mineral N and pH. Further, a recent study by Wolf et al. (2011) show that increment of tree basal area can be promising proxy to predict soil N-oxide fluxes in nitrogen limited tropical montane forest in southern Ecuador. It must be pointed out that the latter results of N_2O and NO

emissions from tropical rain forest were field measurements at a small scale, while we analyzed the spatial variability of incubation fluxes along an approx. 40 km long transect with 31 sampling sites and resulted in high coefficient of variation for both N_2O and NO emissions.

The acidic soils of the Nyungwe forest in combination with high free iron contents could favor chemo-denitrification (auto-decomposition of nitrites) for NO and N_2O production in addition to microbial processes (Van Cleemput and Baert, 1984). All the favorable conditions for this phenomenon to occur are present: very low pH, abundance of clay and reduced metals (e.g. Fe^{2+}) (Nelson and Bremner, 1970; Wullstein and Gilmour, 1964). Positive correlations of N_2O and NO emissions with clay content (for both data subsets) and free iron for the entire data and data subsets and negative correlations with soil pH (N_2O only) (Table 2) give further supporting evidence that chemo-denitrification might play a role. Serca et al. (1994) also found that chemo-denitrification in acid Mayombe forest soils is a potentially important cause of N oxide gases production. Again, variation in parent material and topography caused average free iron content ($4.1 \pm 0.2\%$) to be higher in the eastern part of the forest, where micascists dominate the soilscape, than the average free iron content ($2.5 \pm 0.2\%$ free iron) at western part, where chists dominate the soilscape.

Of course, our incubation study cannot unambiguously improve N trace gas source strength estimates for tropical forests since all results are based on incubations for a few days only. Nonetheless, our study reveals clear aspects of spatial variability and control of N_2O and NO emissions that may assist to the development of baseline information required for reducing emissions from deforestation and degradation activities designed to limit greenhouse gas emissions from tropical forests from developing countries ((UN-REDD program, Stickler et al., 2009). Finally, an additional benefit of the obtained results is their potential to validate the mechanistic biogeochemical (e.g. ForestDNDC-tropica, Werner et al., 2007a)) models and close gaps in global trace gas budgets.

5 Conclusions

This laboratory study provided a unique and large spatial explicit data set of N_2O and NO fluxes for tropical rain forest soils. Nitrous oxide and NO emissions rates were in the range with those reported for African tropical rain forests, but showed a large spatial variation. Soil water content was found to affect NO and N_2O emission differently. The relationship between soil properties and N_2O and NO emission rates was somewhat scale-dependent and spatial information on topography and parent material distribution helped to determine biochemical (OC, TN) and geological (pH, free iron) controls on N_2O and NO emission. Positive correlations with soil carbon and nitrogen

indicate a biological control over N₂O and NO production. But, a negative correlation (only N₂O) with soil pH and a positive correlation with free iron and clay content suggest that chemo-denitrification might be an important alternative production pathway. It would be interesting and useful to define and distinguish chemo-denitrification from related pathway processes in acidic soils of the Nyungwe forest. We, therefore, recommend that future research on N trace gas research in tropical forests include efforts to differentiate biological and chemical pathways for N trace gas emission.

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