

## N<sub>2</sub>O, CH<sub>4</sub> and CO<sub>2</sub> emissions from seasonal tropical rainforests and a rubber plantation in Southwest China

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**Abstract** The main focus of this study was to evaluate the effects of soil moisture and temperature on temporal variation of N<sub>2</sub>O, CO<sub>2</sub> and CH<sub>4</sub> soil-atmosphere exchange at a primary seasonal tropical rainforest (PF) site in Southwest China and to compare these fluxes with fluxes from a secondary forest (SF) and a rubber plantation (RP) site. Agroforestry systems, such as rubber plantations, are increasingly replacing primary and secondary forest systems in tropical Southwest China and thus effect the N<sub>2</sub>O emission in these regions on a landscape level. The mean N<sub>2</sub>O emission at site PF was  $6.0 \pm 0.1$  SE  $\mu\text{g N m}^{-2} \text{h}^{-1}$ . Fluxes of N<sub>2</sub>O increased from  $<5 \mu\text{g N m}^{-2} \text{h}^{-1}$  during dry season conditions to up to  $24.5 \mu\text{g N m}^{-2} \text{h}^{-1}$  with re-wetting of the soil by the onset of first rainfall

events. Comparable fluxes of N<sub>2</sub>O were measured in the SF and RP sites, where mean N<sub>2</sub>O emissions were  $7.3 \pm 0.7$  SE  $\mu\text{g N m}^{-2} \text{h}^{-1}$  and  $4.1 \pm 0.5$  SE  $\mu\text{g N m}^{-2} \text{h}^{-1}$ , respectively. The dependency of N<sub>2</sub>O fluxes on soil moisture levels was demonstrated in a watering experiment, however, artificial rainfall only influenced the timing of N<sub>2</sub>O emission peaks, not the total amount of N<sub>2</sub>O emitted. For all sites, significant positive correlations existed between N<sub>2</sub>O emissions and both soil moisture and soil temperature. Mean CH<sub>4</sub> uptake rates were highest at the PF site ( $-29.5 \pm 0.3$  SE  $\mu\text{g C m}^{-2} \text{h}^{-1}$ ), slightly lower at the SF site ( $-25.6 \pm 1.3$  SE  $\mu\text{g C m}^{-2} \text{h}^{-1}$ ) and lowest for the RP site ( $-5.7 \pm 0.5$  SE  $\mu\text{g C m}^{-2} \text{h}^{-1}$ ). At all sites, CH<sub>4</sub> uptake rates were negatively correlated with soil moisture, which was also reflected in the lower uptake rates measured in the watering experiment. In contrast to N<sub>2</sub>O emissions, CH<sub>4</sub> uptake did not significantly correlate with soil temperature at the SF and RP sites, and only weakly correlated at the PF site. Over the 2 month measurement period, CO<sub>2</sub> emissions at the PF site increased significantly from  $50 \text{ mg C m}^{-2} \text{h}^{-1}$  up to  $100 \text{ mg C m}^{-2} \text{h}^{-1}$  (mean value  $68.8 \pm 0.8$  SE  $\text{mg C m}^{-2} \text{h}^{-1}$ ), whereas CO<sub>2</sub> emissions at the SF and RP site were quite stable and varied only slightly around mean values of  $38.0 \pm 1.8$  SE  $\text{mg C m}^{-2} \text{h}^{-1}$  (SF) and  $34.9 \pm 1.1$  SE  $\text{mg C m}^{-2} \text{h}^{-1}$  (RP). A dependency of soil CO<sub>2</sub> emissions on changes in soil water

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content could be demonstrated for all sites, thus, the watering experiment revealed significantly higher CO<sub>2</sub> emissions as compared to control chambers. Correlation of CO<sub>2</sub> emissions with soil temperature was significant at the PF site, but weak at the SF and not evident at the RP site. Even though we demonstrated that N and C trace gas fluxes significantly varied on subdaily and daily scales, weekly measurements would be sufficient if only the sink/ source strength of non-managed tropical forest sites needs to be identified.

**Keywords** Carbon dioxide · Greenhouse gases · Methane · Nitrous oxide · Soil moisture · Soil temperature · SE Asia · Tropical rainforest · Rubber plantation

## Introduction

Tropical rainforest soils have been identified as a major source of the radiatively active trace gases nitrous oxide (N<sub>2</sub>O) and carbon dioxide (CO<sub>2</sub>), whilst concurrently acting as a significant sink for methane (CH<sub>4</sub>) (e.g., Mosier et al. 1998; Kroeze et al. 1999; Prather and Ehhalt 2001; Potter et al. 1996; Raich et al. 2002). Humid tropical forests have been estimated to cover  $1,150 \pm 54 \times 10^6$  ha (11.8%) of the global land surface (Achar et al. 2002) and the soils of these ecosystems are thus, despite large annual deforestation losses, the dominant source for N<sub>2</sub>O from natural ecosystems, contributing 2.2–3.7 Tg N yr<sup>-1</sup> (24% of the total source strength) to the global atmospheric N<sub>2</sub>O budget (Prather and Ehhalt 2001). Despite their significance, detailed studies of the soil-atmosphere trace gas exchange in tropical ecosystems are still scarce. Empirical up-scaling of the global source strength of tropical ecosystems is thus based on a very limited database that is unable to take account of the spatial and temporal variability of the soil-atmosphere exchange of trace gases (Serca et al. 1994; Breuer et al. 2000; Kiese et al. 2005). The high variability in trace gas flux rates is generally caused by the underlying biogeochemical processes (e.g., mineralization, nitrification and denitrification), which are controlled by environmental factors such as soil

moisture, soil temperature and nutrient availability (e.g., Firestone and Davidson 1989; Davidson 1992; Conrad 1996; Smith et al. 2003). As these environmental factors vary in time and space, site specific estimates of annual trace gas exchange are likely to contain large uncertainties.

In order to quantify the source strength, empirical equations are now commonly used to up-scale flux measurements to regional or global trace gas budgets (Prather and Ehhalt 2001), but, being a generic approach, do not account for regional variations and controlling factors. In recent years, up-scaling approaches which link detailed geographic information systems (GIS) to mechanistic biochemical models like CASA (Potter et al. 1993), CENTURY (Parton et al. 1988) or PnET-N-DNDC (Li et al. 2000; Stange et al. 2000) to calculate regional or global emission inventories (Potter et al. 1996; Kesik et al. 2005; Kiese et al. 2005) are seen as a promising option to improve the current estimates and reduce the associated uncertainties (e.g., Davidson et al. 1998; Li et al. 2000; Butterbach-Bahl et al. 2001, 2004a). These models generally operate in daily time steps, therefore, to adequately calibrate and validate them trace gas flux measurements of a high temporal resolution, from a variety of ecosystems are required (Kiese et al. 2005). For financial and logistical reasons, annual observation periods are often not feasible in remote tropical ecosystems and thus it is common practice to either record fluxes in transition periods to cover a range of hygric conditions, or to manually sample on a weekly or monthly basis with the risk of missing short-term peak fluxes.

In this study we report about a field campaign to measure the soil-atmosphere exchange of N<sub>2</sub>O, CH<sub>4</sub> and CO<sub>2</sub> in a tropical primary forest, a secondary forest, and a rubber plantation in the Xishuangbanna region in Southwest China. This region is representative of many mountainous regions in Southeast Asia, including large parts of Vietnam, Laos, Myanmar and Thailand (Fox and Vogler 2005). While our main focus was to evaluate the controlling factors and seasonal variation of the soil-atmosphere exchange of C and N trace gases from natural rainforest soils, we also recorded fluxes in a secondary forest and a rubber plantation. In many parts of Southeast

Asia, agroforestry systems, such as rubber plantations, are increasingly replacing primary and secondary forest systems. For instance, the extent of rubber plantations in the Xishuangbanna region increased from 6130 to 136782 ha between 1963 and 1998 (Xu et al. 2005), while the area covered by tropical forests decreased from 30% to approximately 10% of the total land area during the same period.

The aim of the study was to (i) measure the magnitude and spatial-temporal variation of soil trace gas fluxes during the transition from a dry to wet season in different forest ecosystems of the seasonal tropics in mountainous Southeast Asia, and to (ii) investigate the effect of environmental driving forces such as soil moisture and soil temperature on trace gas fluxes.

## Methods

### Site description

The climate in the Xishuangbanna region is strongly seasonal. Between May and October about 80% of the total annual precipitation of 1,493 mm occurs via the tropical Southwest Monsoon, whereas dry and cold air masses of the subtropical jet streams dominate the regional climate from November to April. The dry period is further divided into a cool-dry period lasting from November to February with frequent heavy radiation fog at night and in the morning, and a hot-dry period with heavy radiation fog only during the morning hours (Cao et al. 2006). Water deposition from fog equates to  $0.52 \text{ mm day}^{-1}$  (49%) and  $0.41 \text{ mm day}^{-1}$  (33%) during cool-dry and hot-dry periods, respectively, which contributes substantially to the total water input below the canopy during these months. The long-term mean temperature at Menglun town, Menla County, is  $21.8^\circ\text{C}$  (Cao et al. 2006).

Due to the monsoonal climate and the rough topography, four general vegetation types can be found in this region: tropical seasonal rainforest, tropical mountain rainforest, evergreen broad-leaved forest, and monsoon forest over limestone (Shanmughavel et al. 2001). The primary forest

(PF) site in our investigation is defined as tropical seasonal rainforest with an average canopy height of about 40 m. Zhu et al. (2006) stratifies this forest type into three to four indistinct tree layers, where top layer trees generally exceed 30 m (up to 30% crown coverage) and the second/main canopy layer contains the greatest stem density and reaches up to 30 m (70–80% crown coverage). The third layer, containing small trees and juveniles, generally reaches 5–18 m (up to 40% coverage) and can be further divided into sub-layers. The above-ground biomass at the site was estimated by Zheng et al. (2006b) to be  $536 \text{ t ha}^{-1}$ , while the root mass amounted to  $147 \text{ t ha}^{-1}$ . In contrast to the tropical lowland evergreen rain forest, this forest type contains deciduous trees that drop leaves during the dry season (Shanmughavel et al. 2001). Total annual litterfall is  $6.3 \text{ t ha}^{-1} \text{ yr}^{-1}$ ,  $1.9 \text{ t ha}^{-1} \text{ yr}^{-1}$  and  $1.9 \text{ t ha}^{-1} \text{ yr}^{-1}$  for leaf, wood and reproduction litter, respectively (Zheng et al. 2006a). As a direct effect of intensive agroforestry practices in the area, most primary forest is now restricted to steep and mountainous terrain, not suitable for agricultural management.

Consequently, the PF site is situated in a small valley in the upper parts of a mountain range at 770 m altitude (see Table 1 for site characteristics). The average slope of the terrain varies between 5 and  $35^\circ$  and has a southerly aspect. The secondary forest (SF) and rubber plantation (RP) sites are located at 550 m altitude on the riverbanks of the Loso River, a tributary to the Mekong. The latest burning and logging events of the SF site occurred in the late 1980's. Since then the site has been undisturbed and the forest was allowed to re-grow. The secondary forest now reaches a canopy height of approximately seven meters and dense shrubs dominate the undergrowth at present. Our rubber plantation site (RP) is approximately 2 km from the secondary forest, and both the RP and SF sites are located approximately 10 km southeast of the PF site. Approximately 20 years ago, the natural slope ( $10\text{--}15^\circ$ ) of the RP site was terraced during the establishment of the rubber plantation. Furthermore, in agreement with local farming practice, the RP site is fertilized annually with a multiplex (NPK) fertilizer of synthetic compounds and

**Table 1** Main characteristics of the investigated sites (DW: dry weight, oven dried at 105°C; ± standard error)

	Primary forest (PF)	Secondary forest (SF)	Rubber plantation (RP)
Location	21°57'49" N 101°12'01" E	21°55'11" N 101°16'04" E	21°54'31" N 101°16'19" E
Altitude [m. a.s.l.]	770	550	580
Slope [°]	5–35	<5	15 (terraced)
Aspect	South	None	West
<i>Litter layer</i>			
Mass [g DW m <sup>-2</sup> ]	345 ± 32	487 ± 57	505 ± 7
PH	5.2 ± 0.2	5.3 ± 0.1	6.4 ± 0.1
C:N-ratio	31.7 ± 1.7	28.2 ± 1.7	34.0 ± 0.8
C <sub>org</sub> [%]	42.2 ± 1.2	42.0 ± 1.6	43.1 ± 1.4
<i>Mineral soil layer [0–20 cm]</i>			
Bulk density [g cm <sup>-3</sup> ]	1.30 ± 0.02	1.18 ± 0.01	1.24 ± 0.01
Soil texture	Sandy loam	Clay loam	Silty clay
Sand [%]	59	26	8
Silt [%]	23	40	51
Clay [%]	18	34	41
<i>Mineral soil [0–10 cm]</i>			
PH	5.1 ± 0.1	3.5 ± 0.1	4.7 ± 0.1
C:N-ratio	10.0 ± 0.3	11.0 ± 0.4	10.3 ± 0.3
C <sub>org</sub> [%]	1.9 ± 0.2	3.0 ± 0.2	2.5 ± 0.2
<i>Mineral soil [10–20 cm]</i>			
pH	4.6 ± 0.2	3.5 ± 0.0	4.5 ± 0.1
C:N-ratio	8.5 ± 0.3	10.7 ± 0.5	9.2 ± 0.4
C <sub>org</sub> [%]	1.1 ± 0.1	2.0 ± 0.1	2.3 ± 0.1

organic matter at a rate of approximately 55 kg N ha<sup>-1</sup> yr<sup>-1</sup>. The fertilizer is usually applied in the middle of the dry season. At our site, the rubber plantation was fertilized only two weeks prior to the start of the flux measurements. During these two weeks no precipitation was recorded at the nearest climate station. The fertilizer was point-applied by digging and mixing into 20–30 cm deep holes between rubber trees, at a dose of 0.4–0.6 kg N per hole. The stand density was in a range of 108–124 trees ha<sup>-1</sup>.

The majority of tropical rain forests in the Xishuangbanna region are located on yellow lateritic soils developed from granite and gneiss parent material, or lateritic red soils derived from sandstone substrates (Cao et al. 2006). Soil texture, as well as other soil properties such as pH, bulk density and soil organic carbon (SOC), varied significantly among the sites. The PF site had a

sandy loam texture, whereas the SF and RP sites were classed as clay loam and silty clay, respectively. The mineral soil pH at the sites varied in a range of 3.5–5.1, with most acidic conditions observed at the SF site and least acidic conditions at the PF site (Table 1). SOC was found to be highest at the SF site (3% in the uppermost 10 cm) and lowest at the PF site (1.9%).

#### Measurement of soil N<sub>2</sub>O, CH<sub>4</sub> and CO<sub>2</sub> exchange fluxes

The soil-atmosphere exchange of N<sub>2</sub>O, CH<sub>4</sub> and CO<sub>2</sub> was measured with the static chamber technique. At the PF site an automated measuring system was used (Breuer et al. 2000), whereas at the SF and RP sites trace gas fluxes were calculated from manual sampling. The fully automated gas sampling system installed at the PF site allowed us to measure short-term changes in N<sub>2</sub>O, CH<sub>4</sub> and CO<sub>2</sub> fluxes, as ten flux measurements were taken per day. The automated gas sampling system consisted of:

- (1) Eight chambers (three chambers with the size of 70 × 70 × 30 cm; width × length × height and five chambers 50 × 50 × 15 cm). The lids of the chambers were opened and closed automatically with pneumatic devices. The chambers were fixed on stainless steel frames, which were driven approximately 15 cm into the soil.
- (2) An automated gas sampling system, including a pump, valves and mass flow rate regulators. It allows to suck air samples from each chamber alternately and to inject the air samples towards the analytical devices. A set of four chambers was closed for 74 min and re-opened for the following 74 min. During the closing period five chamber air samples were taken and automatically analysed for concentrations of N<sub>2</sub>O, CH<sub>4</sub> and CO<sub>2</sub>.
- (3) Analytical devices: a gas chromatograph (Texas Instruments SRI 8610C, Torrance/USA) equipped with a <sup>63</sup>Ni electron capture detector (ECD) for N<sub>2</sub>O analysis and a flame ionisation detector (FID) for CH<sub>4</sub> analysis. In addition, an infrared gas analyser

(LI-COR 820, LICOR, Lincoln/USA) was installed to allow measurements of CO<sub>2</sub> concentrations in air samples.

- (4) A laptop for data storage (N<sub>2</sub>O and CH<sub>4</sub> chromatograms, CO<sub>2</sub> data and auxiliary data such as soil temperature and soil moisture) and system control.

The gas chromatograph was calibrated automatically on an hourly basis using standard gases (Messer Griesheim/Germany). Further details on the automated system and analytical conditions applied for gas analyses are found in Breuer et al. (2000) and Kiese and Butterbach-Bahl (2002).

Flux measurements at the RP and SF sites were manually performed. Eight (SF) or ten (RP) semi-translucent plastic chambers (30 × 40 × 22 cm) were inserted approximately 10 cm into the soil. Rubber seals ensured that chamber lids could be gas-tightly closed. Gas sampling from the chamber headspace was done with syringes via a 1-m long teflon tube (1/8 inches) connected to the chambers. For determination of each flux rate, five 30 ml air samples were taken from the chamber at 15-min intervals following closure. These gas samples were analysed within 12 h by gas chromatography (Agilent 4860D, Agilent Technologies, Palo Alto/USA) utilizing a <sup>63</sup>Ni ECD for N<sub>2</sub>O analysis and a FID for CH<sub>4</sub> and CO<sub>2</sub> analysis. Prior to detection, CO<sub>2</sub> was reduced to CH<sub>4</sub> using a methanizer. For further details on analytical conditions for the manual sampling system see Wang and Wang (2003). Chamber positions at all sites were chosen in such a way that the variety of aspects, topography and vegetation were covered.

As the onset of the rains could not be predicted or guaranteed, a watering experiment was performed to investigate the influence of soil rewetting on the magnitude of C and N trace gas exchange. At each site, daily rainfall events of 20 mm were simulated for four (PF and SF) or five (RP) chamber locations by evenly watering an area of approximately 2 m<sup>2</sup> around each chamber. At the RP site, two chambers were placed over fertilizer holes and three chambers on the surrounding terrace.

All watered chambers were located down-slope of the control chambers to avoid influences

on the control chambers by lateral movement or surface flow of irrigated water. Daily watering took place between March 2–12 and April 11–21 at the PF site, between March 9–22 and April 11–17 at the SF site and between March 14–26 at the RP site. All non-watered chambers are referred to as control chambers.

#### Auxiliary measurements

Soil temperatures (litter, 5 cm, 10 cm) were recorded automatically and in hourly resolution at the PF site. At the other sites, soil temperature (5 cm) was recorded daily while chamber air samples were taken (between 9 and 11 am). Soil moisture was measured individually for each chamber with a portable TDR probe on a daily basis. During the watering experiment, this was done prior to the daily watering procedure. Volumetric soil moisture values were converted into values of water filled pore space (WFPS) by the following formula:

$$\text{WFPS}[\%] = \frac{\text{Vol}[\%]}{1 - \frac{\text{bd}[\text{g cm}^{-3}]}{2.65[\text{g cm}^{-3}]}}$$

where bd is bulk density, Vol is volumetric water content and 2.65 is the density of quartz.

#### Statistical analysis

In order to examine the relationship between soil parameters and the measured N<sub>2</sub>O, CH<sub>4</sub> and CO<sub>2</sub> fluxes, linear regression analysis was performed using SPSS 8.0 statistical analysis software (SPSS Inc., Chicago/United States) and Origin 7.0 (OriginLab Corporation, Northampton/USA). The non-parametric, pair-wise Wilcoxon test was used without any data transformation for the comparison of control and water treatments.

## Results

### N<sub>2</sub>O emission

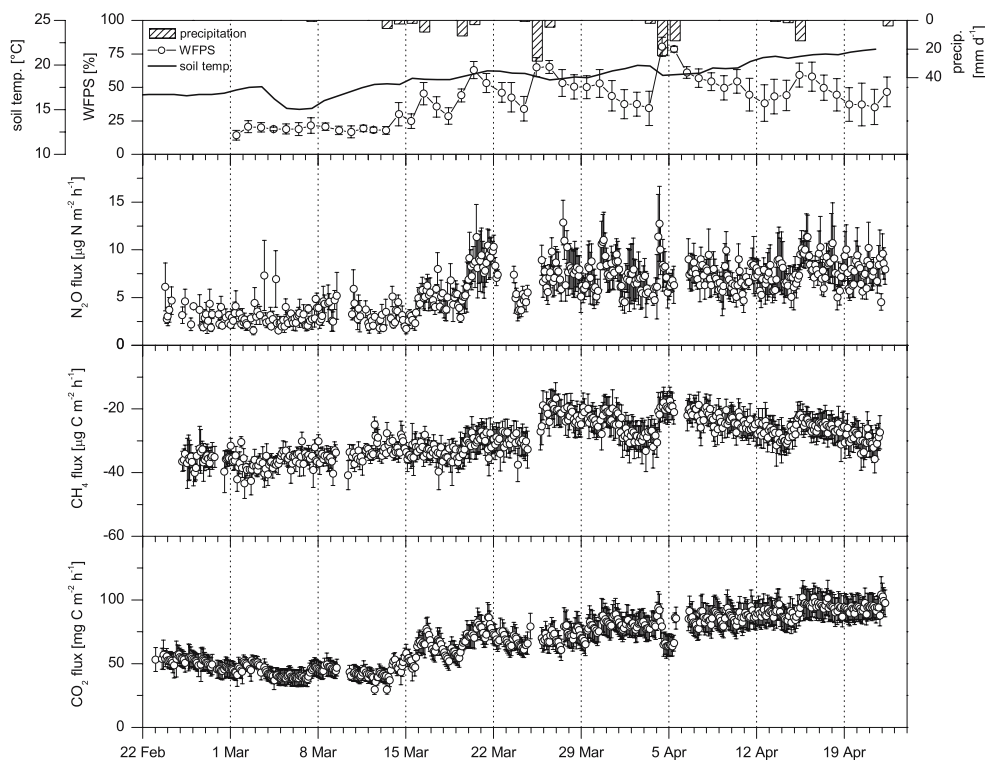
At the PF site, N<sub>2</sub>O emissions from the control chambers ranged from 0.5 μg N m<sup>-2</sup> h<sup>-1</sup> to

$24.5 \mu\text{g N m}^{-2} \text{h}^{-1}$ . During the first three weeks of measurements, without significant rainfall events and WFPS below 20%, average  $\text{N}_2\text{O}$  emissions were generally less than  $5 \mu\text{g N m}^{-2} \text{h}^{-1}$ . With the onset of the first rainfall event,  $\text{N}_2\text{O}$  emissions from the PF site increased and were generally greater than  $5 \mu\text{g N m}^{-2} \text{h}^{-1}$  (Fig. 1). Peak  $\text{N}_2\text{O}$  emissions with flux rates  $>10 \mu\text{g N m}^{-2} \text{h}^{-1}$  were observed after major rainfall events ( $>20 \text{ mm rainfall day}^{-1}$ ), which occasionally increased WFPS values up to 75%. The mean  $\text{N}_2\text{O}$  emission at the PF site during the entire observation period was  $6.0 \pm 0.1 \mu\text{g N m}^{-2} \text{h}^{-1}$  (Table 2).

At the SF site,  $\text{N}_2\text{O}$  emissions of the control chambers varied between  $1.0 \mu\text{g N m}^{-2} \text{h}^{-1}$  and  $28.5 \mu\text{g N m}^{-2} \text{h}^{-1}$ , which is comparable with the range observed at the PF site (Table 2). Lowest  $\text{N}_2\text{O}$  emissions were recorded at the beginning of the measurement campaign. As at the PF site, the  $\text{N}_2\text{O}$  emissions at the SF site increased following the first rainfall events and corresponding

increases in soil moisture, to values generally greater than  $6 \mu\text{g N m}^{-2} \text{h}^{-1}$  (Fig. 3). With  $7.3 \pm 0.7 \mu\text{g N m}^{-2} \text{h}^{-1}$  as the mean  $\text{N}_2\text{O}$  emission for the entire observation period, the  $\text{N}_2\text{O}$  emissions from this site (SF) were slightly greater.

At the RP site,  $\text{N}_2\text{O}$  emissions of the non-fertilized control chambers were below  $5 \mu\text{g N m}^{-2} \text{h}^{-1}$ , except for the last three measuring days when  $\text{N}_2\text{O}$  emissions increased above  $7 \mu\text{g N m}^{-2} \text{h}^{-1}$ . However,  $\text{N}_2\text{O}$  emissions of the fertilized control chambers increased continuously during the entire measuring period, from  $2.3 \mu\text{g N m}^{-2} \text{h}^{-1}$  at the beginning, up to  $1,570 \mu\text{g N m}^{-2} \text{h}^{-1}$  towards the end of the measuring campaign (Fig. 4). By considering the ratio of fertilized area ( $13.9 \text{ m}^2 \text{ ha}^{-1}$ , conservatively calculated by multiplying 116 fertilized holes  $\text{ha}^{-1}$  with  $0.12 \text{ m}^2$  area per chamber) versus non-fertilized area ( $9,986.1 \text{ m}^2 \text{ ha}^{-1}$ ), a weighted mean  $\text{N}_2\text{O}$  emission for the RP site was calculated as  $4.1 \pm 0.5 \mu\text{g N m}^{-2} \text{h}^{-1}$  (see Table 2). This value



**Fig. 1** Measured  $\text{N}_2\text{O}$ ,  $\text{CH}_4$  and  $\text{CO}_2$  net emission, soil moisture (water filled pore space, WFPS), soil temperature and precipitation at the primary forest site. Given are

mean fluxes of the four replicate control chambers (error bars are standard errors)

**Table 2** Mean ( $\pm$ SE), minima and maxima of nitrous oxide, methane and carbon dioxide emissions at the primary forest, secondary forest and rubber plantation sites

	Nitrous oxide [ $\mu\text{g N m}^{-2} \text{h}^{-1}$ ]		Methane [ $\mu\text{g C m}^{-2} \text{h}^{-1}$ ]		Carbon dioxide [ $\text{mg C m}^{-2} \text{h}^{-1}$ ]	
	Control	Watered	Control	Watered	Control	Watered
<i>Primary forest</i>						
Mean	6.0 $\pm$ 0.1	6.8 $\pm$ 0.1**	-29.5 $\pm$ 0.3	-25.4 $\pm$ 0.4**	68.8 $\pm$ 0.8	83.9 $\pm$ 0.9**
Minimum	0.5	0.7	-52.3	-57.8	18.1	32.4
Maximum	24.5	12.7	-4.3	-12.8	131.6	217.7
<i>n</i>	1813	1846	1958	1905	2097	2068
<i>C<sub>v</sub></i> [%]	40	32	19	31	28	25
<i>Secondary forest</i>						
Mean	7.3 $\pm$ 0.7	8.2 $\pm$ 0.8*	-25.6 $\pm$ 1.3	-23.6 $\pm$ 1.1	38.0 $\pm$ 1.8	43.7 $\pm$ 1.6**
Minimum	1.1	1.5	-44.5	-50.0	10.1	26.4
Maximum	28.5	14.2	-0.4	-3.2	56.9	68.5
<i>n</i>	75	77	79	75	78	77
<i>C<sub>v</sub></i> [%]	41	45	22	21	21	16
<i>Rubber plantation</i>						
Mean	4.1 $\pm$ 0.5 <sup>a</sup>	5.4 $\pm$ 0.3 <sup>a*</sup>	-5.7 $\pm$ 0.5	-3.7 $\pm$ 0.3**	34.9 $\pm$ 1.1	44.4 $\pm$ 0.9**
Minimum	0.4 (8.3)	1.8 (62.3)	-16.0	-10.3	20.9	31.6
Maximum	6.5 (1926.3)	10.2 (899.7)	1.7	6.1	67.7	63.9
<i>n</i>	81	80	80	81	83	85
<i>C<sub>v</sub></i> [%]	54	25	39	47	13	8

Mean values and coefficients of variation ( $c_v$ ) were calculated from chamber averages and minima and maxima nitrous oxide emissions of fertilized chambers at the rubber plantation site are given in brackets (*n*: total number of flux rates; significant differences between control and water treatments: \*  $P < 0.05$ /\*\* $P < 0.01$ )

<sup>a</sup> Weighted area flux including fertilized area of  $13.9 \text{ m}^{-2}$  (116 point applications  $\text{ha}^{-1} * 0.12 \text{ m}^{-2}$  chamber area) and unfertilized area of  $9986.1 \text{ m}^{-2}$

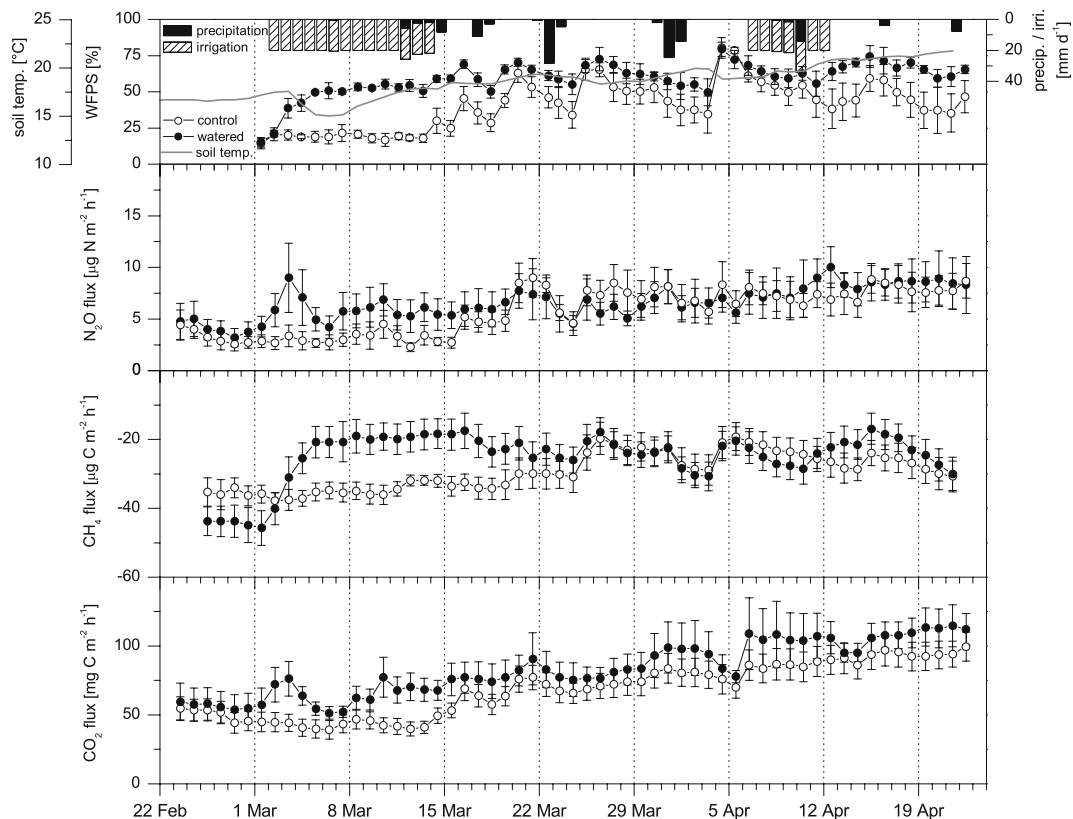
is approximately 25% higher than the mean value,  $3.3 \pm 0.4 \mu\text{g N m}^{-2} \text{h}^{-1}$ , obtained by taking into account only observations from non-fertilized control chambers.

At all sites, re-wetting of the soil significantly increased  $\text{N}_2\text{O}$  emissions during the first weeks of measurements, when no, or little, rainfall occurred and WFPS at all sites was  $<30\%$  (Figs. 1, 2, 3 and 4). From the end of March onwards, the differences in WFPS between watered and control chambers had diminished and were no longer significant.

For the measurement period as a whole, the mean  $\text{N}_2\text{O}$  emission rates of watered chambers for the PF, SF and RP sites were higher (13%, 12% and 31%, respectively) than those of the control chambers (see Table 2). Figure 5 shows in detail the difference in  $\text{N}_2\text{O}$  emissions (12-h means) between watered and control chambers, especially so for the PF site. In order to account for the natural variation in  $\text{N}_2\text{O}$  source strength between watered and un-watered (control) treat-

ments, evident from the  $\text{N}_2\text{O}$  emissions measured prior to the watering experiment 1 (February 23–March 1), all  $\text{N}_2\text{O}$  emissions of the watered chambers were normalized by this natural deviation.

On average, the  $\text{N}_2\text{O}$  emissions during the watering experiment 1 were 72% higher than  $\text{N}_2\text{O}$  emissions from the control chambers, with a maximum difference of 169% at March 4th. However, after the artificially induced emission pulse, the  $\text{N}_2\text{O}$  emissions of the watered chambers were generally 20% (March 15 until the end of the measuring campaign) lower than  $\text{N}_2\text{O}$  emissions from the controlled chambers. During the watering experiment 2, greater  $\text{N}_2\text{O}$  emissions were only detected from the watered chambers within the first 48 h, but the increase was small (14%). For the watering experiment 2 period as a whole,  $\text{N}_2\text{O}$  emissions from the control chambers were actually 11% greater than those from the water chambers. The integration of the differences between watered and control chambers



**Fig. 2** Daily mean net emission fluxes of  $\text{N}_2\text{O}$ ,  $\text{CH}_4$  and  $\text{CO}_2$  of the watered and control chamber groups, daily mean soil moisture (water filled pore space, WFPS) and

soil temperature, precipitation and irrigated water amount at the primary forest site. Given are means of four replicates (error bars are standard errors)

over the entire observation period revealed that  $\text{N}_2\text{O}$  emissions from the watered chambers were slightly smaller ( $-6\%$ ) than those for the control chambers, even though the mean WFPS of the watered chambers was approximately 50% greater than that in the control chambers.

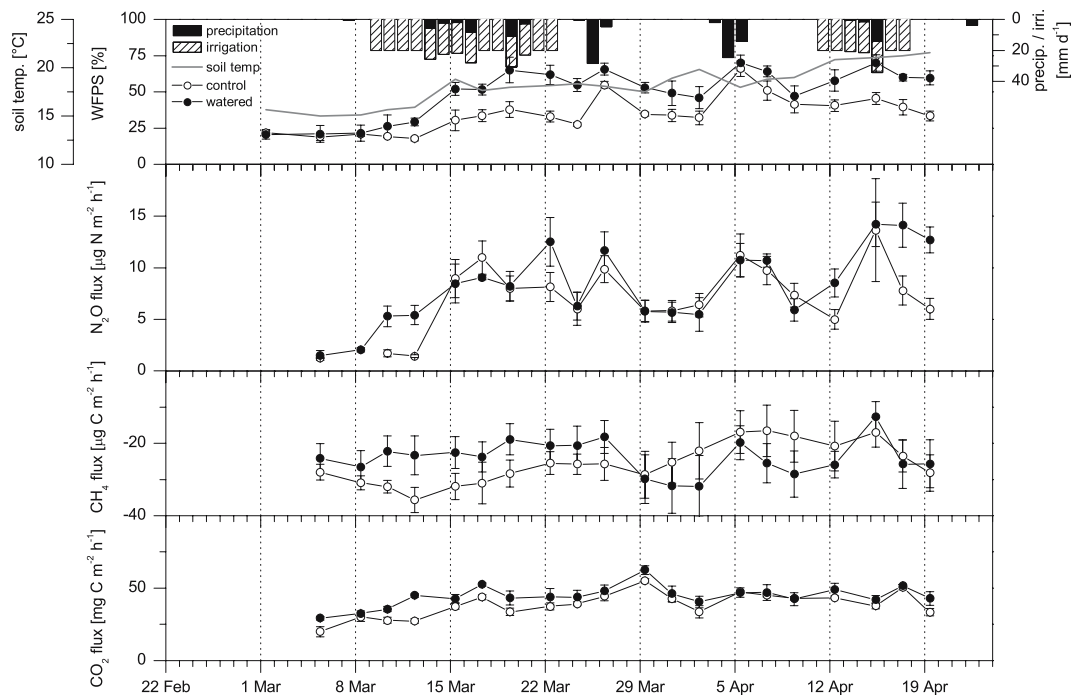
For all sites, a significant positive correlation existed between  $\text{N}_2\text{O}$  emissions and WFPS and between  $\text{N}_2\text{O}$  emissions and soil temperature (Table 3, Fig. 6). Changes in WFPS explained up to 48% of the temporal variability in  $\text{N}_2\text{O}$  emissions at the PF site, 61% at the SF site, but only 30% of the variability at the RP site. Regression analysis between soil temperature and  $\text{N}_2\text{O}$  emissions explained 50%, 35% and 27% for PF, SF and RP, respectively. Multiple linear regression models explained up to 69% of temporal variations in  $\text{N}_2\text{O}$  emissions through changes in temperature and soil moisture (Table 3). Combining the data from all three

sites did not improve the coefficients of determination from the multiple linear regression model.

#### $\text{CH}_4$ uptake

Soils were acting as a net sink for atmospheric  $\text{CH}_4$  at all three sites (negative flux rates indicate soil uptake of atmospheric  $\text{CH}_4$ ). Mean  $\text{CH}_4$  uptake rates were highest at the PF site ( $-29.5 \pm 0.3 \mu\text{g C m}^{-2} \text{h}^{-1}$ ), slightly lower at the SF site ( $-25.6 \pm 1.3 \mu\text{g C m}^{-2} \text{h}^{-1}$ ) and lowest for the RP site ( $-5.7 \pm 0.5 \mu\text{g C m}^{-2} \text{h}^{-1}$ ) (Table 2). At site PF,  $\text{CH}_4$  uptake was highest at the beginning of the measurement period and was reduced by approximately 50% after the first rainfall event and associated increases in soil moisture. Only during the first weeks of measurements until the end of March, when WFPS in the control chambers were still smaller than 30%,  $\text{CH}_4$  uptake in the watered chambers was





**Fig. 3** Measured  $\text{N}_2\text{O}$ ,  $\text{CH}_4$  and  $\text{CO}_2$  net emission of the watered and control chamber groups, simultaneously measured soil temperature and moisture, precipitation

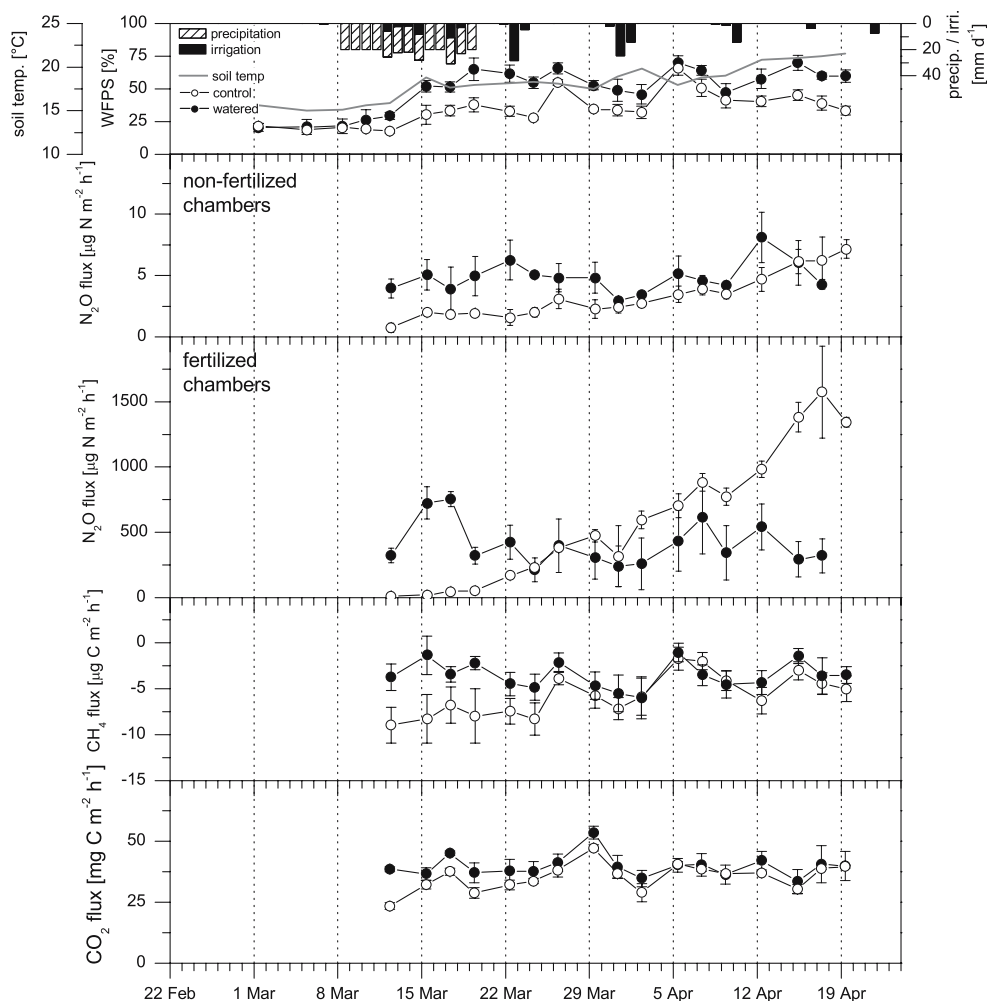
and irrigated water amount at the secondary forest site. Given are means of four replicates (error bars are standard errors)

significantly less than that in the control chambers (Fig. 2). Thereafter, i.e., with the increase of soil moisture in the control chambers due to the onset of the wet season, the difference between  $\text{CH}_4$  uptake in control and watered chambers disappeared. Variations in  $\text{CH}_4$  uptake were significantly correlated with changes in WFPS (Table 3, Fig. 6). Figure 5 illustrates that in comparison with the control chambers the watering experiments at the PF site lead to a relative drop in soil  $\text{CH}_4$  uptake of 72% (experiment 1) and 54% (experiment 2) within 48–72 h of watering, and that these uptake rates remained at this lower level until the end of the experiments. In contrast to the observed  $\text{N}_2\text{O}$  fluxes, the reduction in  $\text{CH}_4$  uptake following watering gradually diminished so that  $\text{CH}_4$  uptake had returned to initial uptake levels before the start of the watering experiment 2. At the PF and RP sites, changes in WFPS explained more than 65% of the temporal variation in  $\text{CH}_4$  uptake (Table 3). A significant correlation between soil temperature and  $\text{CH}_4$  exchange was only observed at the PF site

( $P < 0.05$ ). However, the coefficient of determination was low ( $r^2 = 0.05$ ). Combining the data from all three sites for a multiple linear regression did not significantly improve the explanation capabilities.

#### $\text{CO}_2$ emission

Considerable temporal variations in  $\text{CO}_2$  emission could only be found at the PF site (Fig. 1), whereas  $\text{CO}_2$  emissions at the SF and RP sites (Figs. 3, 4) showed only minor variations. From late February until mid of March, the  $\text{CO}_2$  emissions at the PF site were approximately  $50 \text{ mg C m}^{-2} \text{ h}^{-1}$ . Thereafter, the  $\text{CO}_2$  emissions at this site increased constantly up to  $100 \text{ mg C m}^{-2} \text{ h}^{-1}$  until the end of measurements at the end of April (Fig. 2). The soil  $\text{CO}_2$  emissions from the SF ( $38.0 \pm 1.8 \text{ mg C m}^{-2} \text{ h}^{-1}$ ) and RP ( $34.9 \pm 1.1 \text{ mg C m}^{-2} \text{ h}^{-1}$ ) sites were significantly lower (51–55%) than those at the PF site ( $68.8 \pm 0.8 \text{ mg C m}^{-2} \text{ h}^{-1}$ ) (Figs. 1–4, Table 2).



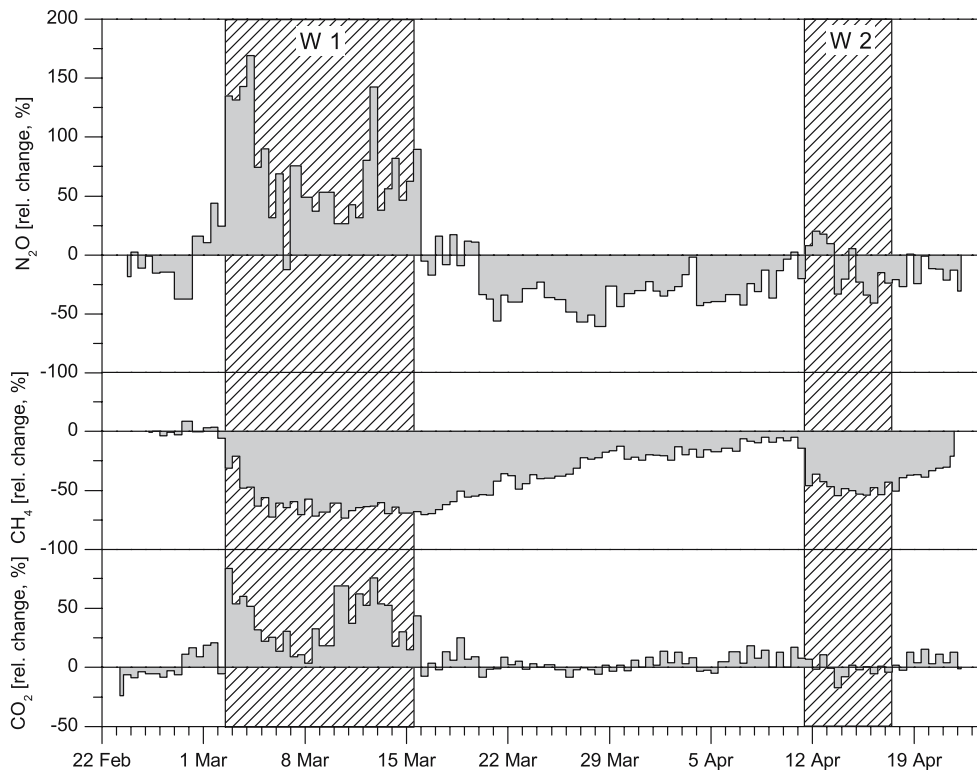
**Fig. 4** Measured N<sub>2</sub>O, CH<sub>4</sub> and CO<sub>2</sub> net emission of watered and control chamber groups, simultaneously measured soil temperature and moisture, precipitation and irrigated water amount at the rubber plantation site.

Watering stimulated soil CO<sub>2</sub> emissions over the entire observation period at all sites, leading to a generally greater CO<sub>2</sub> emissions from watered chambers (see Table 2). In comparison with the control chambers, mean soil CO<sub>2</sub> emissions from the watered chambers increased by 18% (PF), 13% (SF) and 21% (RP). As shown in Fig. 5, the relative changes due to watering in CO<sub>2</sub> emissions during the watering experiment 1 and 2 were similar to those in N<sub>2</sub>O emissions. However, immediately after the watering experiment 1, the CO<sub>2</sub> emission fluxes from either watered or control chambers declined to the initial levels measured before watering com-

Nitrous oxide emissions are divided into fertilized (two chamber mean) and non-fertilized (three chamber means) groups. Methane and CO<sub>2</sub> fluxes are given as five chamber means (error bars are standard errors)

menced. Thereafter no further differences were evident between the watered and control chambers. During the watering experiment 2, no increase in CO<sub>2</sub> emission fluxes due to watering was measured.

Soil CO<sub>2</sub> emissions at the PF site correlated strongest with changes in soil temperature ( $r^2 = 0.71$ ), but correlations with WFPS were also significant ( $r^2 = 0.49$ ). At the other sites, CO<sub>2</sub> emissions were better correlated with changes in WFPS, as compared to soil temperature (Table 3, Fig. 6). Consequently, the multiple linear regression incorporating both soil moisture and soil temperature improved the explanation



**Fig. 5** Relative changes in  $\text{N}_2\text{O}$ ,  $\text{CH}_4$  and  $\text{CO}_2$  net emissions due to the watering treatments at the primary forest site. Shown are the relative changes of 12-hourly mean fluxes. The data was normalized by the mean

difference prior to the watering experiment 1 in order to remove spatial emission deviations between the two chamber groups (W1, watering period 1; W2, watering period 2)

of  $\text{CO}_2$  variability significantly for the PF site ( $r^2 = 0.85$ ), while no improvements could be achieved for the two other sites.

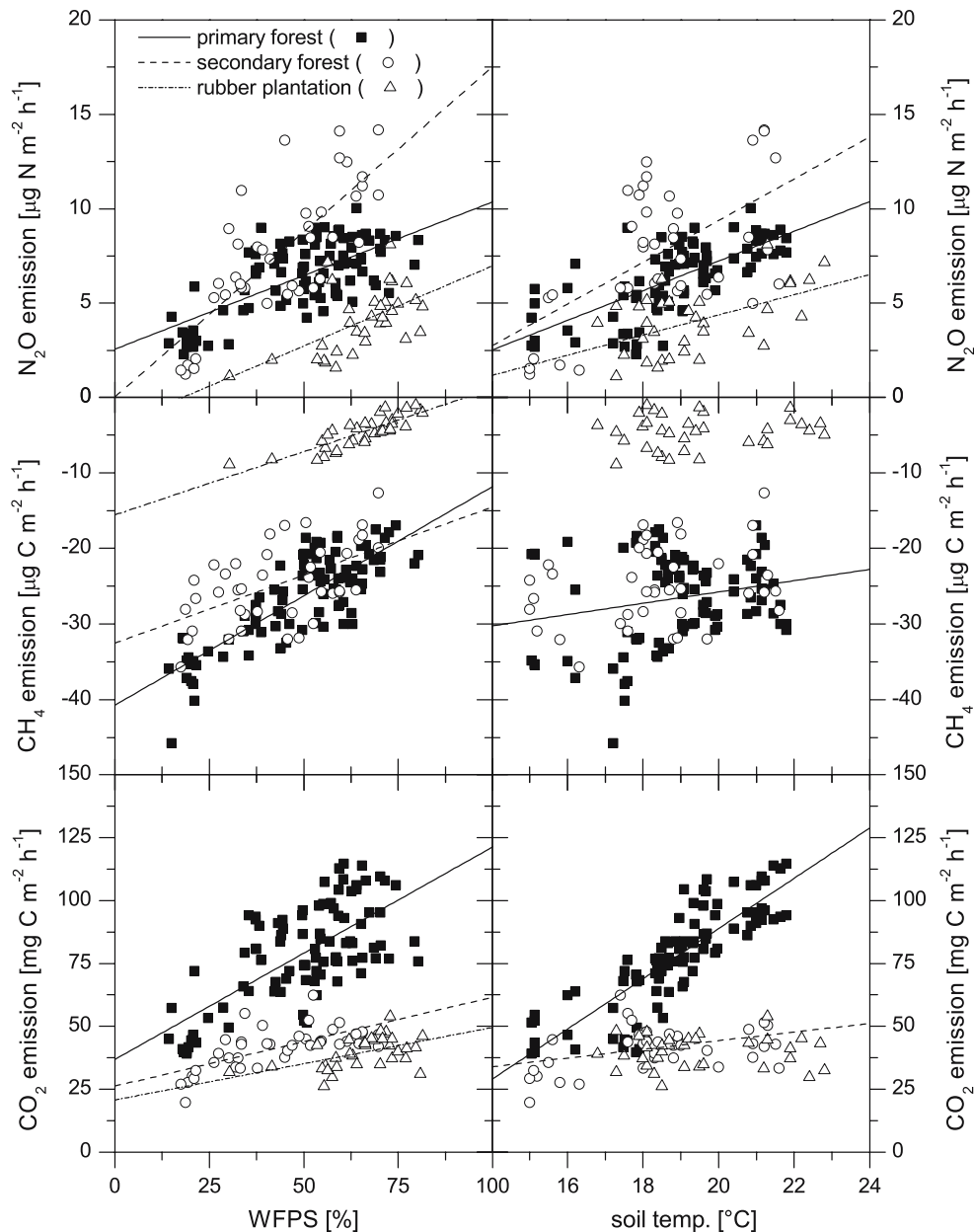
## Discussion

Our measurements provide the first set of simultaneously measured soil  $\text{N}_2\text{O}$ ,  $\text{CH}_4$  and  $\text{CO}_2$  fluxes for three different forest cover types in continental, tropical Southeast Asia. So far, only measurements of C and N trace gas fluxes for tropical forest ecosystem in Indonesia and Borneo have been published (Hall et al. 2004; Ishizuka et al. 2002, 2005a, b, see also Table 4). The observed  $\text{N}_2\text{O}$  emissions during the transition from dry to wet season at our PF site was  $6.0 \pm 0.1 \mu\text{g N m}^{-2} \text{h}^{-1}$  (ranging from  $0.5 \mu\text{g N m}^{-2} \text{h}^{-1}$  to  $24.5 \mu\text{g N m}^{-2} \text{h}^{-1}$ ) and, thus, within the lower to medium range of reported

$\text{N}_2\text{O}$  emissions from other tropical rainforest ecosystems under comparable hygric conditions (Breuer et al. 2000; Melillo et al. 2001; Kiese and Butterbach-Bahl 2002; Garcia-Montiel et al. 2003; Kiese et al. 2003). However,  $\text{N}_2\text{O}$  fluxes reported in this study are significantly higher than reported  $\text{N}_2\text{O}$  emissions for a primary forest site in the Jambi Province, Indonesia (mean value  $1.5\text{--}4.4 \mu\text{g N m}^{-2} \text{h}^{-1}$ ; Ishizuka et al. 2002). Hall et al. (2004) reported  $\text{N}_2\text{O}$  fluxes of  $1\text{--}7.5 \mu\text{g N m}^{-2} \text{h}^{-1}$  for sites of differing substrates and altitudes at Mt. Kinabalu, Borneo (Table 4). Based on earlier studies (Keller and Reiners 1994; Verchot et al. 1999), one would have expected that  $\text{N}_2\text{O}$  emissions at the SF site should be lower than at the PF site, due to nutrient losses following burning and/or selected cutting. However, our observation revealed that the  $\text{N}_2\text{O}$  emissions at the SF site ( $7.3 \pm 0.7 \mu\text{g N m}^{-2} \text{h}^{-1}$ ) tended to be 22% higher than those at the PF site.

**Table 3** Regression analysis between mean daily nitrous oxide, methane and carbon dioxide fluxes and changes in mean daily soil moisture and soil temperature ( $P < 0.05^*$ ,  $P < 0.01^{**}$ )

	Primary forest ( $n = 94$ )	Secondary forest ( $n = 40$ )	Rubber plantation ( $n = 34$ )	All sites ( $n = 168$ )
<i>Soil moisture (M [WFPS, %])</i>				
Nitrous oxide	$f(M)$ $r^2$ 0.48** 2.563 + 0.078M	0.61** 0.030 + 0.175M	0.30** -1.521 + 0.085M	0.11** 3.616 + 0.051M
Methane	$R^2$ 0.66** $f(M)$ -40.722 + 0.289M	0.30** -32.539 + 0.180M	0.72** -15.573 + 0.167M	0.49** -42.461 + 0.407
Carbon Dioxide	$R^2$ 0.49** $f(M)$ 36.861 + 0.844M	0.45** 26.245 + 0.353M	0.26** 20.746 + 0.288M	0.07** 42.529 + 0.376M
<i>Soil temperature (T [°C])</i>				
Nitrous oxide	0.50** -8.508 + 0.787T	0.35** -12.764 + 1.107T	0.27** -6.325 + 0.535	0.21** -6.778 + 0.688T
Methane	$f(T)$ $r^2$ 0.05* -40.719 + 0.748	0.08 -38.919 + 0.770T	0.02 -7.874 + 0.166T	0.06** -46.853 + 1.331T
Carbon Dioxide	$f(T)$ $r^2$ 0.71** -110.992 + 9.992T	0.16* 9.855 + 1.721T	0.00 38.433 + 0.062T	0.17** -44.005 + 5.588T
<i>Multiple linear regression analysis (soil temperature and soil moisture)</i>				
Nitrous oxide	$F(T,M)$ $r^2$ 0.69** -6.847 + 0.561T + 0.053M	0.66** -8.25 + 0.521T + 0.145M	0.58** -11.829 + 0.533T + 0.085M	0.24** -6.105 + 0.579T + 0.027M
Methane	$f(T,M)$ $r^2$ 0.69** -30.934-0.584T + 0.315M	0.30** -32.529 - 0.001T + 0.180M	0.74** -18.642 + 0.158T + 0.167M	0.48** -35.349-0.427T + 0.426M
Carbon Dioxide	$f(T,M)$ $r^2$ 0.85** -95.642 + 7.903T + 0.494M	0.46** 21.735 + 0.288T + 0.335M	0.26 19.815 + 0.048T + 0.288M	0.18** -39.791 + 4.944T + 0.156M



**Fig. 6** Correlations of daily mean  $\text{N}_2\text{O}$ ,  $\text{CH}_4$  and  $\text{CO}_2$  net emissions with soil temperature and soil moistures (water filled pore space, WFPS) at the primary forest, secondary forest and rubber plantation sites. Soil moisture was recorded daily for each chamber. Soil temperature was

acquired automatically at the primary forest site and manually at the secondary forest and rubber plantation sites (linear regression only shown when significant [ $P < 0.05$ ]; see Table 3 for details)

This discrepancy is most likely due to the differences in texture and SOC (clay loam and 3% SOC at SF versus sandy loam and 1.9% SOC at PF), as a more loamy soil with higher SOC generally supports greater  $\text{N}_2\text{O}$  emissions (Li et al. 2005). Forest composition was also shown to

affect the N trace gas release in tropical forest ecosystems (Erickson et al. 2002). As compared to the PF site, littermass was 40% greater at the SF site (see Table 1). The C:N ratio of the SF litter was slightly smaller than that at the PF site. As a large proportion of  $\text{N}_2\text{O}$  originates from the

**Table 4** Nitrous oxide (N<sub>2</sub>O) flux measurements from soils of tropical Southeast Asian ecosystems (BD: bulk density, MAP: mean annual precipitation; ± standard errors)

Site	N <sub>2</sub> O emission [μg N m <sup>-2</sup> h <sup>-1</sup> ]	<i>n</i>	MAP [mm]	C [%]	N [%]	pH	BD [g cm <sup>-3</sup> ]
<i>Primary forest</i>							
Borneo, Mt. Kinabalu <sup>b</sup>	1.0–7.5	6	2,509	n.d.	n.d.	3.5–4.0	1.1
Borneo, Mt. Kinabalu <sup>b</sup>	2.0–4.0	6	2,714	n.d.	n.d.	3.4–4.6	0.4–0.7
Jambi Prov., Indonesia <sup>c</sup>	1.5 ± 0.2	9	2,060	3.0	0.19	4.2	1.12
Jambi Prov., Indonesia <sup>c</sup>	4.4 ± 1.4	9	2,060	3.0	0.19	4.2	1.12
Xishuangbanna, China	6.0 ± 0.1	453	1,530	1.9	0.19	5.12	1.3
<i>Secondary forest</i>							
Jambi Prov., Indonesia <sup>c</sup>	7.9 ± 2.5	5	2,060	3.5	0.24	4.8	0.81
Jambi Prov., Indonesia <sup>c</sup>	6.4 ± 2.2	5	2,060		0.65	4.0	0.88
Xishuangbanna, China	7.3 ± 0.7	18	1,530	3.0	0.27	3.5	1.18
<i>Rubber plantation</i>							
Jambi Prov., Indonesia <sup>c</sup>	0.7 ± 0.1*	9	2,060	1.6	0.12	4.7	0.98
Xishuangbanna, China	4.1 ± 0.5 <sup>a</sup>	17	1,530	2.5	0.24	4.7	1.24

\* Mean flux; non-fertilized

<sup>a</sup> Mean flux for measurement period; fertilized (55 kg N ha<sup>-1</sup>)

<sup>b</sup> Hall et al. (2004)

<sup>c</sup> Ishizuka et al. (2002), all soil data for 0–10 cm

litter layer and smaller C:N ratios support higher N emissions (e.g., Erickson et al. 2002; Kiese and Butterbach 2002), the difference in N<sub>2</sub>O emissions could also be explained by variations in litter composition and mass.

The meteorological data for our sites indicate that our measurements started at the end of a pronounced dry season. After prolonged dry periods, it has been shown that re-wetting of the soil can be accompanied by high emissions of N trace gases (Davidson et al. 1993; Garcia-Montiel et al. 2003; Butterbach-Bahl et al. 2004b). Such pulse fluxes are associated with the rapid microbial consumption of ammonium or nitrite accumulated during the dry period (Davidson et al. 1993), which are in turn partly oxidized (ammonium) or reduced (nitrite) to N<sub>2</sub>O by nitrifying bacteria (Poth and Focht 1985). As shown e.g., in Figs. 1, 2, we also detected significant N<sub>2</sub>O pulse emissions due to re-wetting not only in the watered chambers but also in the control ones. Thereby, Fig. 1 illustrates the detailed sub-daily fluxes at the PF site showing a significantly higher temporal dynamic (e.g., subdaily reaction time of N<sub>2</sub>O emission on precipitation at 28th of March and 4th/5th of April) as compared to the daily aggregated flux rates presented in Fig. 2. Maximum event-based

emissions presented in Fig. 1 are as high as 12 μg N m<sup>-1</sup> h<sup>-1</sup>, whereas due to daily averaging, maximum daily flux rates are only 7 μg N m<sup>-1</sup> h<sup>-1</sup>. Using the entire data set, we calculated mean N<sub>2</sub>O emissions based on randomly selected data sets assuming daily, weekly or monthly measuring intervals. This analysis revealed that the reduction of the subdaily measuring frequency would reduce the accuracy of the calculated mean N<sub>2</sub>O-emission up to 8% (daily), 25% (weekly) and 63% (monthly), respectively. From these figures it can be concluded that weekly measurements of trace gas fluxes would have provided a reliable basis for estimating the N<sub>2</sub>O exchange at our site. However, it has to be stressed that for further improvement and validation of mechanistic models it is of utmost importance to provide detailed data such as daily or subdaily trace gas fluxes in order to test our process understanding and to develop suitable algorithms describing processes involved in trace gas production, consumption and emission. Such detailed data were e.g., used to test the anaerobic balloon concept of the Forest-DNDC model for predicting simultaneously production (consumption) of N<sub>2</sub>O by anaerobic (denitrification) and aerobic (nitrification) microbial processes (Li et al. 2000).

The magnitude of  $N_2O$  pulse emissions in our study was less pronounced than those reported in previous studies cited above. The reason for this discrepancy remains unclear, but is most likely related to a lower accumulation of substrates (e.g., ammonium and nitrate) for N turnover and associated N trace gas emission within nitrification and denitrification processes. As already demonstrated in earlier studies, soil moisture changes are the main driver for temporal variations of  $N_2O$  emissions in tropical forest ecosystems (e.g., Davidson 1993; Steudler et al. 1996; Breuer et al. 2000; Kiese and Butterbach 2002). Changes in soil moisture ultimately control soil aeration and effect nutrient availability. This finally feeds back into the spatial and temporal differences in the occurrence and magnitude of the oxidative nitrification and reductive denitrification processes and associated N trace gas emissions (e.g., Smith 1980; Conrad 1996). In unsaturated soil, higher moisture values are generally expected to support higher  $N_2O$  emissions. However, this was clearly not the case for the PF site. In our watering experiment at the PF site,  $N_2O$  emissions were only stimulated for a few days. But after the end of the artificial watering, the enhanced  $N_2O$  emissions of the watered chambers diminished and even dropped below those of the control chambers. This means that simulating rainfall during the transition from dry to wet season influenced the timing of  $N_2O$  emissions, but not the total amount of  $N_2O$  released over our entire observation period. This may be attributed to the fact that after the rapid mineralization of easily decomposable organic litter and/or rapid microbial consumption of ammonium and nitrite accumulated during the dry period (Davidson et al. 1993), microbial N turnover processes, such as mineralization, nitrification and denitrification, and associated  $N_2O$  emissions decreased due to substrate limitations regardless of the suitable soil moisture condition.

At all three investigated sites, we observed a linear relationship between WFPS (range from 15% to 85%) and  $N_2O$  emission fluxes. Butterbach-Bahl et al. (2004b) also found a positive linear correlation between WFPS (ranging from 10% to 50%) and  $N_2O$  emissions for tropical forest soils in Northern Queensland, Australia,

while Keller and Reiners (1994) described an exponential correlation in primary and secondary forests of Costa Rica for a WFPS range of 50–90%. Kiese et al. (2002) reported a linear correlation between  $N_2O$  emissions and WFPS less than 60%, but noted a decline in  $N_2O$  emissions at higher moisture levels, which is most likely due to the increasing formation of  $N_2$ , rather than  $N_2O$ , as the denitrification process begins to dominate, as has been shown elsewhere (e.g., Nömmik 1956; Butterbach-Bahl et al. 2002).

The temporal variability of  $N_2O$  emissions was pronounced in our study. At the PF and SF site, the coefficients of variation ( $c_v$ ) for  $N_2O$  emissions of the entire measurement period (daily aggregated and analysed for any single chamber) ranged from 35% to 62% (Table 2). The observed variations in  $N_2O$  emissions in this study are within the  $c_v$ -range of 14–125% reported for tropical rainforest soils in Queensland, Australia (Breuer et al. 2000; Kiese and Butterbach-Bahl 2002; Butterbach-Bahl et al. 2004b), but lower than the range of 94–195% reported for tropical rain forest soils in Amazonia (Vitousek et al. 1989; Verchot et al. 1999). Despite the fact that the rubber plantation was fertilised with approximately  $55 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ , weighted mean  $N_2O$  emission fluxes of the RP site were significantly lower than those observed at the nearby SF site or at the PF site. This result is in agreement with Ishizuka et al. (2002) for rubber and rainforest sites in Sumatra, Indonesia. If we compare the neighbouring sites SF and RP with the soil at the RP site, it appears that the RP soil was depleted in total C and N even though the textural composition of the soil was similar to the soil at the SF site (see Fig. 1). Moreover, the RP site was characterised by the highest C:N ratio of above-ground litter among our three sites (Table 1). This supports the explanation of Ishizuka et al. (2002) that the relatively low N trace gas emissions of rubber sites is due to the loss of labile carbon and nitrogen stocks following land-use conversion, which is supposed to feed back on reduced mineralization and nitrification activities (Neill et al. 1995). However, the field campaign was short (2 months) and extrapolation of the observed mean  $N_2O$  emission ( $4.1 \pm 0.5 \mu\text{g N m}^{-2} \text{ h}^{-1}$ ) to an annually based flux rate ( $0.36 \text{ kg N ha}^{-1}$ ) is likely

to underestimate the  $\text{N}_2\text{O}$  source strength at the RP site since we conservatively estimated the area ( $13.9 \text{ m}^2$ ) affected by fertilizer application. For comparison, we calculated the annual  $\text{N}_2\text{O}$  loss following the IPCC guidelines for agricultural soils (IPCC 1997). By applying the recommended emission factor of  $1.25 \pm 1.0\%$ , the annual  $\text{N}_2\text{O}$  emission for the RP site estimated to be  $0.68 \text{ kg N ha}^{-1}$ . Taking into account the high levels of uncertainty associated with both methods, some credence is given to the fact that the calculated annual  $\text{N}_2\text{O}$  loss rates extrapolated from the two months measuring period was of the same order of magnitude as the IPCC emission factor approach.

Methane exchange at the soil-atmosphere interface is the result of simultaneously occurring consumption and production of  $\text{CH}_4$  by methanotrophic and methanogenic bacteria (Conrad 1996). The soil methanogens usually occur under strictly anaerobic conditions (Conrad 2002) and  $\text{CH}_4$  oxidation by methanotrophs does require the presence of molecular  $\text{O}_2$  in the soil air. The magnitude of  $\text{CH}_4$  oxidation in soils has been found to depend on the effective diffusivity of the soil medium to  $\text{O}_2$ , which is generally negatively correlated with soil moisture (e.g., Dörr et al. 1993; Keller and Reiners 1994; Brumme and Borken 1999; Verchot et al. 2000). During the observed transition period from dry to wet season, the soils of the PF and SF sites predominantly functioned as a significant sink for atmospheric  $\text{CH}_4$ . Only in the watered chambers, we sporadically observed a weak net emission of  $\text{CH}_4$  from the soils. This can be interpreted that under water saturated conditions anaerobic micro-sites developed within the soil profile and methanogenic processes were initiated, such that net  $\text{CH}_4$  production resulted. The mean  $\text{CH}_4$  net uptake fluxes of  $21.0 \pm 0.3 \mu\text{g C m}^{-2} \text{ h}^{-1}$  (PF) and  $28.1 \pm 0.8 \mu\text{g C m}^{-2} \text{ h}^{-1}$  (SF) are in general agreement with previously recorded values for tropical rainforest soils from Kiese et al. (2003) and Ishizuka et al. (2005a, b). The temporal variations of daily fluxes from the control chambers, as expressed by  $c_v$ , were at all sites up to two fold higher for  $\text{N}_2\text{O}$  emission than for  $\text{CH}_4$  uptake. This indicates that the dynamics of  $\text{CH}_4$  exchange, controlled by environmental changes,

were less pronounced than those for  $\text{N}_2\text{O}$  emissions, which is in agreement with previous findings for a lowland tropical rain forest site in Australia (Kiese et al. 2003).

The average net  $\text{CH}_4$  uptake of soils at the RP site was  $5.7 \mu\text{g C m}^{-2} \text{ h}^{-1}$ , which is only 20–25% of the observed uptake rates at the PF and SF sites. In contrast to the other sites, the soil of the RP regularly turned into a weak  $\text{CH}_4$  source after heavy rainfall events or watering activities (Table 3, Fig. 5). On a regional scale, the conversion of forest ecosystems into rubber plantations therefore leads to a significant reduction in atmospheric  $\text{CH}_4$  uptake by soils and, thus, has a negative effect on the greenhouse gas budget of the Xishuangbanna region.

The reduction in the  $\text{CH}_4$  oxidation capability of the RP site may be attributed to (a) soil disturbance and compaction as a consequence of land use/cover change and terracing during the establishment of the rubber plantations and (b) nitrogen fertilization providing  $\text{NH}_4^+$  which is discussed to reduce the  $\text{CH}_4$  oxidation capacity of methanotrophs (Bodelier and Laanbroek 2004). The explanation that nitrogen fertilization and soil physical disturbances strongly reduces  $\text{CH}_4$  uptake is in good agreement with the findings of MacDonald et al. (1996) and Mosier et al. (2004). These authors found that, after re-establishment of forests on agricultural land, it may take several decades to regain the  $\text{CH}_4$  uptake dynamics as observed for undisturbed forest sites. As illustrated by Brumme and Borken (1999), the physical climate parameters temperature and precipitation are responsible for the short-term control of  $\text{CH}_4$  oxidation processes and are thus controls of diurnal and seasonal variability. Combining all daily net  $\text{CH}_4$  exchange fluxes from control and watered chambers, we found that soil moisture variations were able to explain 66% (PF), 30% (SF) and 72% (RP) of the temporal variability at our sites. This is in contrast to the work of Ishizuka et al. (2002), who found no correlation between soil moisture and  $\text{CH}_4$  exchange fluxes at the rainforest sites in Sumatra and only a weak correlation for the investigated rubber plantation. However, this could be due to the fact that these authors only measured  $\text{CH}_4$  net exchange fluxes on a monthly basis ( $n = 12$  per



year), which may not be sufficient to explain short-term changes at hourly to daily time scales as was possible with our data ( $n = 498$  at PF, 20 at SF, and 17 at RP for the 2-month investigation period). Based upon year-round diurnal measurements for soils of the wet tropics in Australia, Kiese et al. (2003) reported a significant correlation between  $\text{CH}_4$  net uptake rates and soil moisture, similar to the results of our study.

Carbon dioxide emission fluxes at the PF site ranged from  $18.1 \text{ mg C m}^{-2} \text{ h}^{-1}$  to  $131.6 \text{ mg C m}^{-2} \text{ h}^{-1}$ . The peak flux was up to  $217.7 \text{ mg C m}^{-2} \text{ h}^{-1}$  in a watered chamber. Significantly lower fluxes were recorded at the SF site ( $10.1\text{--}56.9 \text{ mg C m}^{-2} \text{ h}^{-1}$ ), though the soil organic carbon content at this site was higher and the C:N ratio slightly lower than that at the PF site. The lower  $\text{CO}_2$  emissions at the SF site may be explained by a smaller contribution of plant root autotrophic respiration and a shift in carbon quality towards more resistant carbon pools as a consequence of human management, e.g., logging and burning (inert carbon). The same arguments hold also for the significantly lower  $\text{CO}_2$  emissions at the RP site. The magnitude of soil  $\text{CO}_2$  emissions at the PF site is in good agreement with previous measurements for other tropical rainforest soils. La Scala et al. (2000), Kiese and Butterbach-Bahl (2002), Ishizuka et al. (2002) and Raich (1998) reported  $\text{CO}_2$  emissions from tropical forest floors as  $54.4\text{--}107.2$ ,  $24.0\text{--}247.7$ ,  $51.3\text{--}93.7$  and  $41.7\text{--}125.0 \text{ mg C m}^{-2} \text{ h}^{-1}$ , respectively. The temporal  $c_v$  of our sites, 29–31% at PF and 21–27% at SF, are similar to those reported by La Scala et al. (2000) and Kiese and Butterbach-Bahl (2002). Ishizuka et al. (2002) found no clear relationship between  $\text{CO}_2$  emissions and soil moisture. In contrast, we observed a significant positive correlation at all sites ( $r^2 = 0.26\text{--}0.49$ , see Table 3). It is worthwhile to mention that a strong positive relationship between soil temperature and  $\text{CO}_2$  emissions ( $r^2 = 0.71$ ) only appeared at the PF site which is situated at a higher altitude, and is thus affected by a more pronounced diurnal temperature regime. At the PF site, we even observed temperature driven diurnal variations of  $\text{CO}_2$  emissions ( $\delta\text{CO}_2$  app.  $20 \text{ mg C m}^{-2} \text{ h}^{-1}$ ) at days without significant

changes in soil moisture (Fig. 1; e.g., 7th–10th of April). This trend was less pronounced for  $\text{N}_2\text{O}$  emissions and not evident or  $\text{CH}_4$  fluxes.

## Conclusion

Our study demonstrates that short term changes in  $\text{N}_2\text{O}$  emissions from tropical seasonal rainforest soils closely follow changes in soil moisture, whereas changes in temperature were of minor importance. Artificial increases in soil moisture have not significantly changed total  $\text{N}_2\text{O}$  emissions during our observation period, but did control the timing of emissions. This indicates that substrate availability ultimately controls the magnitude of  $\text{N}_2\text{O}$  emissions in tropical seasonal rainforests. Differences in the magnitude of  $\text{N}_2\text{O}$  emissions for the other investigated forest systems (secondary forests and rubber plantations) were most likely driven by differences in soil properties (SOC, texture) and litter quality and quantity. All sites were significant sinks for atmospheric  $\text{CH}_4$ . However,  $\text{CH}_4$  uptake at the intensively managed and terraced rubber plantation site was only 25% of the observed uptake rates at the primary and secondary forest sites. This demonstrates the long-term negative effect of soil disturbances for atmospheric  $\text{CH}_4$  uptake by soils. At all sites, soil  $\text{CO}_2$  emissions were positively correlated with changes in soil moisture. In contrast to  $\text{N}_2\text{O}$ , artificial irrigation increased total  $\text{CO}_2$  emissions substantially over our observation period. However, it is possible that this was mainly an effect of increased root respiration.

For a better understanding of trace gas exchange in seasonal tropical rainforest ecosystems and in view of the observed pronounced temporal variability, long term measurements of trace gas fluxes in at least weekly resolution at multiple sites are required to further clarify the role of these ecosystems as sources ( $\text{N}_2\text{O}$ ,  $\text{CO}_2$ ) and sinks ( $\text{CH}_4$ ) of greenhouse gases.

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