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ORIGINAL ARTICLE

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Abstract

Nitrous oxide (N$_2$O) fluxes from tropical peatland soils were measured at a grassland, three croplands, a natural forest, a burned forest and a regenerated forest in Central Kalimantan, Indonesia. Only croplands received fertilization (665–1278 kg N ha^{-1} year^{-1}). Mean annual N$_2$O emissions from croplands were 21–131 kg N ha^{-1} year^{-1} in 2002–2003 and 52–259 kg N ha^{-1} year^{-1} in 2003–2004, and were significantly higher than the emissions from other comparable sites. Cropland N$_2$O emissions were among the highest values reported from cultivated tropical, temperate and boreal organic soils. Mean annual N$_2$O emissions were 7.1 (2002–2003) and 23 (2003–2004) kg N ha^{-1} year^{-1} from grassland, and were significantly higher than in natural, regenerated and burned forests (0.62, 0.40 and 0.97 kg N ha^{-1} year^{-1} in 2002–2003 and 4.4, 4.0 and 1.5 kg N ha^{-1} year^{-1} in 2003–2004, respectively). Annual N$_2$O emissions did not differ significantly between forests in 2002–2003, but were significantly lower in burned forest in 2003–2004. Annual N$_2$O emission was significantly correlated between years. Regression analysis revealed that annual N$_2$O emissions in 2003–2004 were 1.9-fold the corresponding 2002–2003 value (annual precipitation of 2339 and 1994 mm, respectively). N$_2$O fluxes were higher during the rainy season than during the dry season at all sites except the regenerated forest. N$_2$O fluxes in cropland and grassland were significantly lower when the water-filled pore space (WFPS) was less than 60–70%, and increased with an increase in soil NO$_3$–N concentration when WFPS exceeded this threshold. Thus, changes in soil moisture were important in controlling seasonal changes in N$_2$O emission. Our results suggest that changing land use from forestry to agriculture will increase N$_2$O production. The effect of forest fires on N$_2$O emission from these soils was not clear.

Key words: closed-chamber method, forest fire, land-use type, N$_2$O, tropical peatland.

INTRODUCTION

Nitrous oxide (N$_2$O) is a major greenhouse gas that contributes approximately 6% of the total global warming (Intergovernmental Panel on Climate Change 2001) and leads to the destruction of the ozone layer in the stratosphere (Crutzen 1981). N$_2$O is produced in soils through nitrification and denitrification. The estimated N$_2$O emissions from tropical wet forest soils (3 Tg N year^{-1}) and agricultural soils (3.5 Tg N year^{-1}) account for a large proportion of total global emissions (16.2 Tg N year^{-1}; Intergovernmental Panel on Climate Change 1996). Tropical peatlands contain a large amount of organic C and N under waterlogged conditions and could, therefore, be an important source of N$_2$O and carbon dioxide (CO$_2$) after drying (Bouwman 1990; Kasimir-Klemedtsson *et al.* 1997; Mosier *et al.* 1998) owing to accelerated decomposition of the peat. The total area of tropical peatlands is estimated at 33–50 Mha, which is 8.6–12% of the global peatland area (Maltby and Immirzi 1993). Most tropical peatlands are located in Indonesia, where they cover 17–27 Mha (Maltby and Immirzi 1993). A large area of tropical peatland in Indonesia has been affected

by forest fires (Page *et al.* 2002) and agricultural exploitation (Muhanmad and Rieley 2002). Page *et al.* (2002) estimated that 27.6% of the total area of peat swamp forest (1.37 million ha) burned in Central Kalimantan during the 1997 El Niño Southern Oscillation event using satellite images of a 2.5 million ha study area. In 1995, more than 1 million ha of tropical peatland in Central Kalimantan was reclaimed by the "Mega Rice Project" (Muhanmad and Rieley 2002).

The cultivation of tropical peatlands has been considered to be a large source of N$_2$O emissions and the Intergovernmental Panel on Climate Change (2000) estimated that direct N$_2$O emissions from cultivated organic soils, totaling 16 kg N ha^{-1} year^{-1}, resulted from the mineralization of soil organic nitrogen. However, this value was determined using emission data collected only from boreal and temperate regions. There have been few reports on N$_2$O emissions from tropical peatlands (e.g. Hadi *et al.* 2000; Inubushi *et al.* 2003; Melling *et al.* 2005). Therefore, there is large uncertainty in estimating N$_2$O emissions from tropical peatlands. Moreover, the factors that control N$_2$O emission from these sites are not well understood. We conducted the present study to: (1) quantify the amount of N$_2$O emissions, (2) investigate the factors that control N$_2$O emissions from these sites, (3) investigate the effects of forest fire and agriculture on N$_2$O emissions from tropical peatlands.

MATERIALS AND METHODS

Site description

The study sites were located at Kalampangan village, near Palangka Raya (2°S, 114°E) in Central Kalimantan, Indonesia (Fig. 1). We established experimental plots at three forest sites and four agricultural sites. The forest sites consisted of a natural forest (NF: 2°21′S, 114°02′E) and two fire-degraded forests designated as regenerated forest (RF: 2°21′S, 114°02′E) and burned forest (BF: 2°19′S, 114°01′E). The natural forest had not been affected by forest fire. The other two forests (regenerated and burned forest) were degraded by forest fire in 1997. The trees and the surface peat soils (maximum at a depth of 30 cm) in the burned and regenerated forests burned during the fire. The regenerated forest regenerated naturally after the fire, but the burned forest did not regenerate a forest cover. Both the regenerated and burned forests were burned again by forest fire in July and September 2002, respectively. As of March 2004, no trees could be found at either site. Both forest fires in 1997 and 2002 were caused naturally during the extreme dry season owing to the El Niño Southern Oscillation event. In the natural forest, the depth of the peat soil ranges from 3.50 to 4.85 m (Tuah *et al.* 2001). This forest consists mainly of deciduous
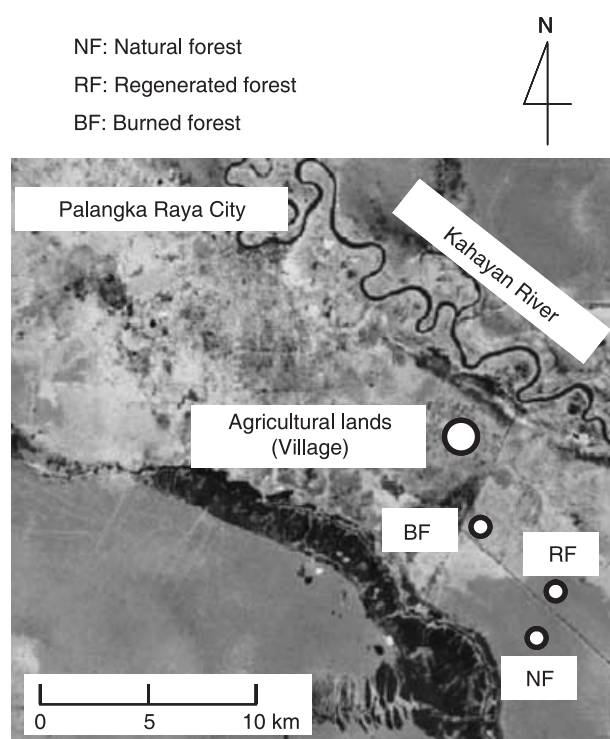

Figure 1 Locations of the study sites in Kalampangan, Central Kalimantan. "Agricultural lands" represent the grassland site (GL) and the croplands (CL) A, B and C.

trees such as *Tetramerista glabra* Miq., *Calophyllum* sp., *Shorea* sp., *Combretocarpus rotundatus* (Miq.) Danser, *Palaquium* sp., *Buchanania sessifolia* Bl., *Syzygium* sp., *Dactylocladus stenostachys* Oliv., in Hook, *Dyera costulata* (Miq.) Hook.f., *Ilex cymosa* Bl., *Tristaniopsis obovata* R. Br. and *Dyospyros* sp. (Tuah *et al.* 2003). The height of the forest canopy is approximately 26 m (Hirano *et al.* 2005). In the regenerated forest, the peat soil depth ranges from 3.25 to 5.20 m. The dominant vegetation of the regenerated forest was also various tree species (*C. rotundatus*, *Camnosperma coriaceum* (Jack.) Hall.f., *Palaquium* sp., *T. glabra*, *Calophyllum* sp. and *D. costulata*) before the forest fire in 2002 (Tuah *et al.* 2001). After the fire, this area was dominated by ferns (*Stenochlaena palustris* (Burm.f.) Bedd. and *Nephrolepis* sp.). The burned forest was also dominated by ferns before and after the fire in 2002.

The agricultural sites (2°17′S, 114°01′E) consisted of one grassland (GL) and three croplands, designated as cropland A (CL-A), cropland B (CL-B) and cropland C (CL-C). These four sites were adjacent to each other. The grassland site was dominated by dwarf grasses and was grazed by cows every 2 months. The main crops in the croplands were maize (*Zea mays* L.), spinach (*Spinacia oleracea* L.) and cassava (*Manihot esculenta*

Table 1 Amount of nitrogen applied at the cropland sites

| | Nitrogen application rate (kg N ha^{-1} year^{-1}) | | | | | | | | |
| | Cropland A | | | Cropland B | | | Cropland C | | |
Year	Urea	Manure	Total	Urea	Manure	Total	Urea	Manure	Total
2002–2003	818	180	998	760	40	800	1,098	180	1,278
2003–2004	545	120	665	853	100	953	665	120	785
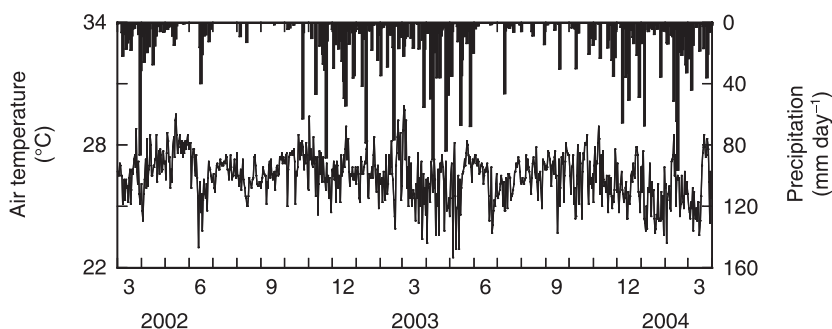
Figure 2 Daily air temperature and precipitation recorded near the natural forest (at 41.7 and 41 m above the ground, respectively).

Crantz.). There was no fertilizer applied in the grassland. The amount of N fertilization of the croplands ranged from 665 to 1278 kg N ha^{-1} year^{-1} (Table 1) according to the farmer. Of this total, 82–95% of the total N was applied in the form of chemical fertilizer (urea). Ash from plants and peat was also added to the cropland soils to supply mineral elements to the peat soil. Manure (mainly cow manure) was also added to the peat soil. The major soil chemical properties at each site are shown in Table 2. The peat soils at the study sites were all classified as Typic Tropofibrists (USDA Soil Taxonomy).

The mean daily air temperature ranged from 22 to 28°C (Fig. 2). In this region, the dry season normally begins in June and ends in October (Takahashi *et al.* 2002). Precipitation during the dry season was greater in 2003 (349 mm) than in 2002 (274 mm). Total annual precipitation was 2339 mm from April 2003 to March 2004 and 1994 mm from April 2002 to March 2003 (Fig. 2).

The soil temperature at a depth of 4 cm in the natural, regenerated and burned forests ranged from 25.6 to 30.2°C, 25.7 to 31.9°C and 28.4 to 34.1°C, respectively (Fig. 3). The mean soil temperature at a depth of 4 cm during the rainy season was significantly ($P < 0.05$) higher in both the burned forest (30.6 ± 1.4°C) and the regenerated forest (29.7 ± 1.6°C) than in the natural forest (27.0 ± 0.9°C) (Table 3). The mean monthly value for water-filled pore space (WFPS) (Fig. 3, Table 3) at the natural forest site showed a clear seasonal change, and was significantly ($P < 0.05$) lower in the dry season (28 ± 16%) than in the rainy season (46 ± 5.5%). The WFPS in natural forest during the 2002 dry season was lower (<20%) than in 2003 because of the lower precip-

itation in 2002 (Fig. 3). During the 2003 rainy season, WFPS in the regenerated forest increased rapidly and remained high compared with levels in the natural forest (Fig. 3). The WFPS values in the regenerated and burned forests were significantly ($P < 0.05$) higher than in the natural forest in the rainy season (Table 3). This may have resulted from decreased water uptake by plants because of the absence of trees on these sites. The regenerated and burned forests were sometimes flooded late in the rainy season (Fig. 3). The mean monthly soil temperature at a depth of 4 cm at the four agricultural sites (Fig. 4) was almost constant and had a similar pattern. The mean soil temperatures during the dry and rainy season ranged from 28.2 to 29.9°C and 29.5 to 32.1°C, respectively (Table 3). The monthly WFPS in the four agricultural lands also showed a similar pattern, and was significantly ($P < 0.05$) lower during the dry season (Fig. 4, Table 3). The average depths to the groundwater table at the agricultural sites (croplands A, B and C were 68, 72 and 76 cm, respectively, *vs* 62 cm for the grassland) were lower than at the forest sites (natural, regenerated and burned forest were 51, 23 and 52 cm, respectively).

Gas sampling for flux measurement

Nitrous oxide flux measurements were carried out once every 3 months from March to August 2002, and monthly from August 2002 to March 2004. During the late rainy season we were unable to conduct N$_2$O flux measurements on several occasions because of flooding in the regenerated forest (March, April and December 2003 as well as January 2004) and in the burned forest

Table 2. Soil chemical and physical properties at the study sites

Site	Natural forest		Regenerated forest				Burned forest				Grassland	Cropland A	Cropland B	Cropland C
			Before[†]		After[‡]		Before[†]		After[‡]					
Depth (cm)	0–5[§]	5–10	0–5[§]	5–10	0–3	3–10	0–5	5–10	0–3	3–10	0–10	0–10	0–10	0–10
pH (1:20 H_2O)	5.2	3.6	5.3	3.8	5.9	3.9	3.8	3.4	4.6	3.8	4.4	5.3	4.6	6.0
CEC ($cmol_C$ kg^{-1} dry soil)	93	152	104	147	85	119	139	159	94	136	110	120	76	89
Exchangeable cations ($cmol_C$ kg^{-1} dry soil)														
Na	0.28	0.36	0.24	0.42	0.44	0.51	0.22	0.35	0.52	0.82	0.27	0.24	0.90	0.23
K	3.4	2.6	3.6	1.3	3.5	3.4	0.82	0.73	4.0	1.4	1.0	0.71	4.4	1.7
Ca	18.7	0.67	25.9	1.5	35.5	9.4	14.6	3.0	39.0	1.7	27.9	62.6	31.5	62.0
Mg	16.9	4.8	11.2	5.4	21.1	10.0	6.9	5.8	11.4	4.0	27.9	16.1	20.7	23.4
Base saturation (%)	42	5.5	39	5.8	72	20	16	6.2	59	5.9	52	66	76	98
NH_4-N (mg kg^{-1} dry soil)	129	511	106	371	471	155	163	22	319	125	67	20	19	17
NO_3-N (mg kg^{-1} dry soil)	0.43	91	2.7	5.9	1.5	0.79	93	127	1.5	1.1	292	236	287	225
C content (g kg^{-1} dry soil)	577	578	575	579	662	594	609	619	649	650	556	530	632	547
N content (g kg^{-1} dry soil)	14.5	18.5	12.8	16.8	14.1	18.0	19.2	14.4	14.3	10.7	14.2	13.0	14.2	14.3
C/N ratio	40	31	45	35	47	33	32	43	46	61	39	41	44	38
Bulk density (Mg m^{-3}) [¶]	0.14						0.22				0.30	0.40	0.41	0.42
Porosity (%)[¶]	90					84					73	76	76	77

[†]Before the forest fire in 2002. [‡]After the forest fire in 2002. [§]Litter layer. [¶]0 to 5 cm depth, measured in February 2005 (Bulk density and porosity of regenerated forest wasn't measured because of flooding).
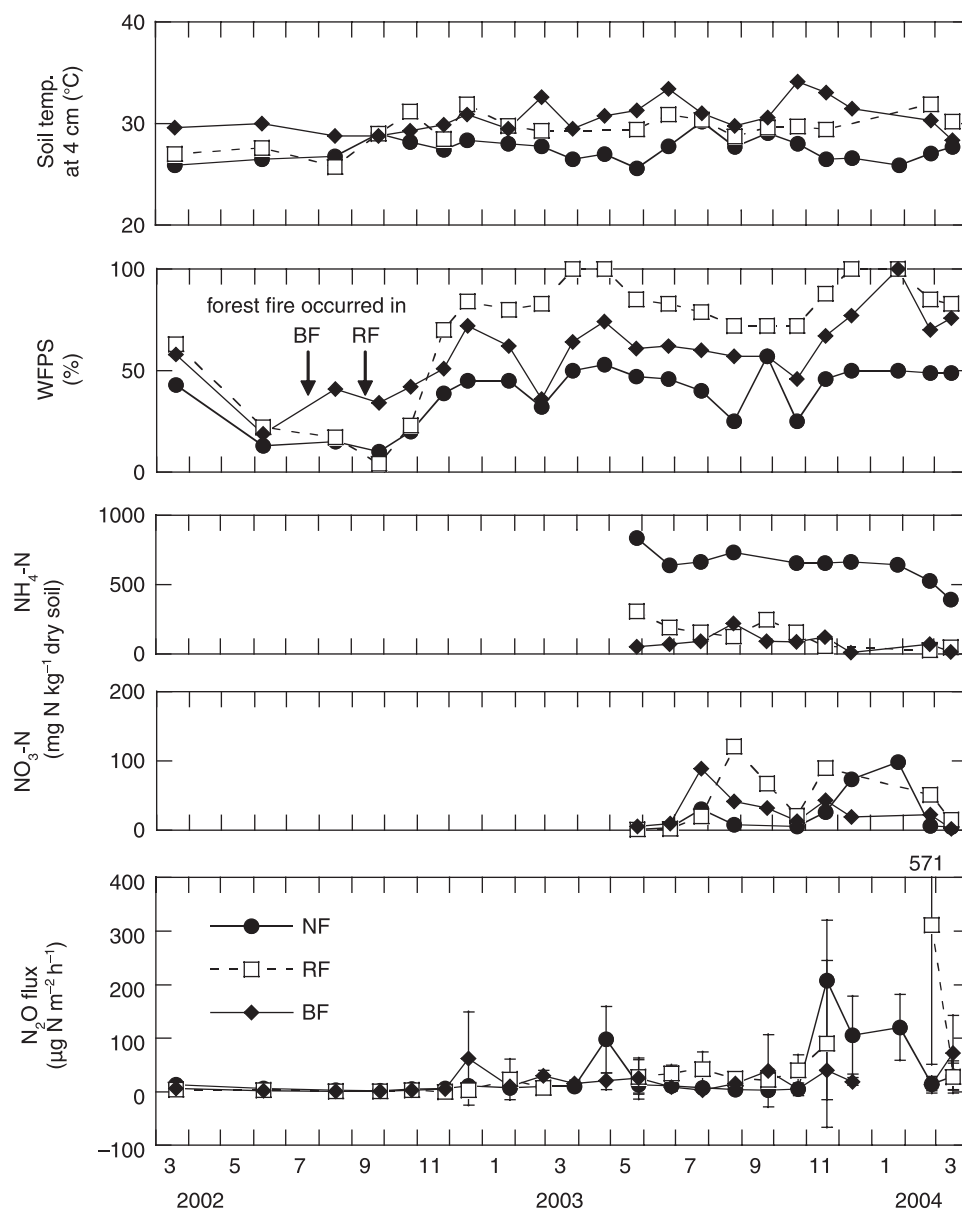
Figure 3 Seasonal changes in soil temperature, water-filled pore space (WFPS), soil NH$_4$–N, soil NO$_3$–N and N$_2$O fluxes at the three forest sites. When the regenerated and burned forests were flooded, WFPS was assumed to reach its maximum value of 100%. Arrows indicate the timing of forest fires. Positive N$_2$O fluxes indicate net emissions; negative values indicate net uptake. Error bars show standard deviations. BF, burned forest; NF, natural forest; RF, regenerated forest.

(January 2004). N$_2$O fluxes from the soil surface were measured using a closed-chamber method (Morishita *et al.* 2003). To facilitate transportation of the chambers to the field, the chambers were constructed so that six chambers with slightly different sizes could fit within each other (Morishita *et al.* 2003). That is, the height of each chamber was the same (25 cm), but the diameters differed slightly (18.5, 19.0, 19.5, 20.0, 20.5 and 21.0 cm). We assumed beforehand that differences in the chamber size did not affect our flux measurements. Vegetation

inside the chamber was cut at ground level and removed and the chambers were installed to a depth of 3 cm below the soil surface 1 day before the measurements were conducted. At 0, 10, 20 and 40 min after the top of the chamber was closed with a lid, a 20-mL gas sample was taken into a 10-mL vacuum vial sealed with a butyl rubber stopper (SVF-10, Nichiden-Rika, Kobe, Japan). The air temperature at a height of 1 m was measured with a thermistor thermometer (Checktemp 1C, Hanna Instruments, Woonsocket, USA) during the flux

Table 3 Mean soil temperatures, water-filled pore space (WFPS) values and N$_2$O fluxes at the study sites during the dry (June to October) and rainy (November to May) seasons

Site	$n^†$	Soil temperature (°C)		WFPS (%)		N$_2$O flux (µg N m^{-2} h^{-1})	
		Dry	Rainy	Dry	Rainy	Dry	Rainy
Natural forest	(9, 13)	28.1 ± 1.1a	27.0 ± 0.9a*	28 ± 16a	46 ± 5.5a*	5.0 ± 2.8a	49 ± 63a*
Regenerated forest	(9, 9)	29.2 ± 1.7a	29.7 ± 1.6b	49 ± 32ab	80 ± 8.3cd*	19 ± 17a	55 ± 100ab
Burned forest	(9, 12)	30.6 ± 1.9a	30.6 ± 1.4b	46 ± 14ab	64 ± 12b**	8.5 ± 12a	26 ± 22a*
Grassland	(9, 13)	29.8 ± 1.3a	29.5 ± 1.8b	43 ± 14ab	66 ± 8.3bc**	50 ± 55ab	257 ± 335bc*
Cropland A	(9, 13)	29.9 ± 1.3a	30.1 ± 1.5b	52 ± 5.9ab	69 ± 8.3bc**	163 ± 219c	4,298 ± 4303d**
Cropland B	(9, 13)	28.2 ± 1.2a	30.1 ± 2.7b*	52 ± 7.1ab	63 ± 6.0b**	154 ± 188bc	637 ± 642cd*
Cropland C	(9, 13)	29.7 ± 1.1a	32.1 ± 2.2b**	61 ± 11b	81 ± 6.0d**	94 ± 62abc	2,441 ± 3553cd*

†Numbers in parentheses represent the sample size in the dry and rainy seasons, respectively. Numbers are means and standard deviations of soil temperature at a depth of 4 cm, WFPS at a depth of 0–6 cm depth and N$_2$O fluxes. Numbers for a parameter followed by different letters differ significantly among sites (Steel–Dwass test, $P < 0.05$). Asterisks (* and **) indicate significant differences between the dry and rainy season ($P < 0.05$ and $P < 0.01$, respectively).

measurements. There were six replications for the forest sites and three replications for the agricultural lands. In general, gas flux measurements were conducted between 09.00 hours and 15.00 hours.

Soil temperature, soil moisture and meteorological data

The soil temperature at a depth of 4 cm, the volumetric soil water content from 0 to 6 cm in depth, and the depth of the water table were measured during the gas flux measurements. The soil temperature was measured using a thermistor thermometer (Checktemp 1C, Hanna Instruments). An amplitude domain reflectometry (ADR, ML2 Theta Probe Delta-Y Devices, Cambridge, UK) was used to measure the volumetric soil water content. These values were calibrated using the volumetric soil water content measured in 200-mL soil core samples taken from a depth of 0–10 cm. The calibration curves for each site were fitted using third-order equations ($r^2 = 0.97$ to 1.00, $P < 0.01$). The resulting volumetric soil water content was converted into a value for WFPS, which represents the ratio of the volumetric water content to the total porosity of the soil, by assuming that the porosity measured in February 2005 was consistent throughout the measurement period. The measurement of porosity is described in the "Soil sampling and analysis" section. The porosity of the regenerated forest was not measured because of flooding, and was replaced by the value for the burned forest site in the WFPS calculation. There were three replications per chamber for the soil temperature and volumetric soil water content measurements. To measure the water table depth, perforated PVC pipes (4 cm in diameter) were inserted into the peat soil at each site.

Air temperature and precipitation were measured every half hour using a 50-m-tall micrometeorological tower established inside the forest (Hirano et al. 2005).

The air temperature was measured at 41.7 m above the ground with a platinum resistance thermometer (HMP45, Vaisala, Helsinki, Finland) and the precipitation was measured at a height of 41 m with a tipping-bucket rain gauge (TE525, Campbell Scientific. Inc., Logan, UT, USA) (Hirano et al. 2005).

Soil gas sampling

The following procedure was used to measure N$_2$O concentrations in the soil gas (Morishita et al. 2003). Three stainless-steel pipes (8 mm in diameter) were installed at each of five depths (5, 10, 20, 40 and 80 cm). A 50-mL sample of air was removed from each pipe, and all gas samples collected from the same depth were mixed in a 1-L sampling bag (Tedlar bag, OMI Odorair Service, Shiga, Japan). Simultaneously, air was sampled at a height of less than 5 cm above the soil surface with a sampling bag to provide a 0-cm (surface) sample. A 10-mL vacuum vial with a butyl rubber stopper was used to sample 20 mL of each of these gas samples.

When only water was collected from the pipes, the dissolved N$_2$O concentrations in the water were measured using a modified version of the method of Sawamoto et al. (2002). Dissolved N$_2$O concentrations were measured using the head-space method (Sawamoto et al. 2002), with a single equilibration using ambient air, and were then converted into N$_2$O concentrations in the soil gas, which were equilibrated to the sampled water.

Analysis of gas concentration and calculation of flux

The N$_2$O concentrations were analyzed with a gas chromatograph (GC-14B, Shimadzu, Kyoto, Japan) equipped with an electron capture detector (ECD). The minimum detectable concentration of N$_2$O was 0.007 p.p.m.v determined by the deviation obtained by repeatedly analyzing the concentration of an N$_2$O standard
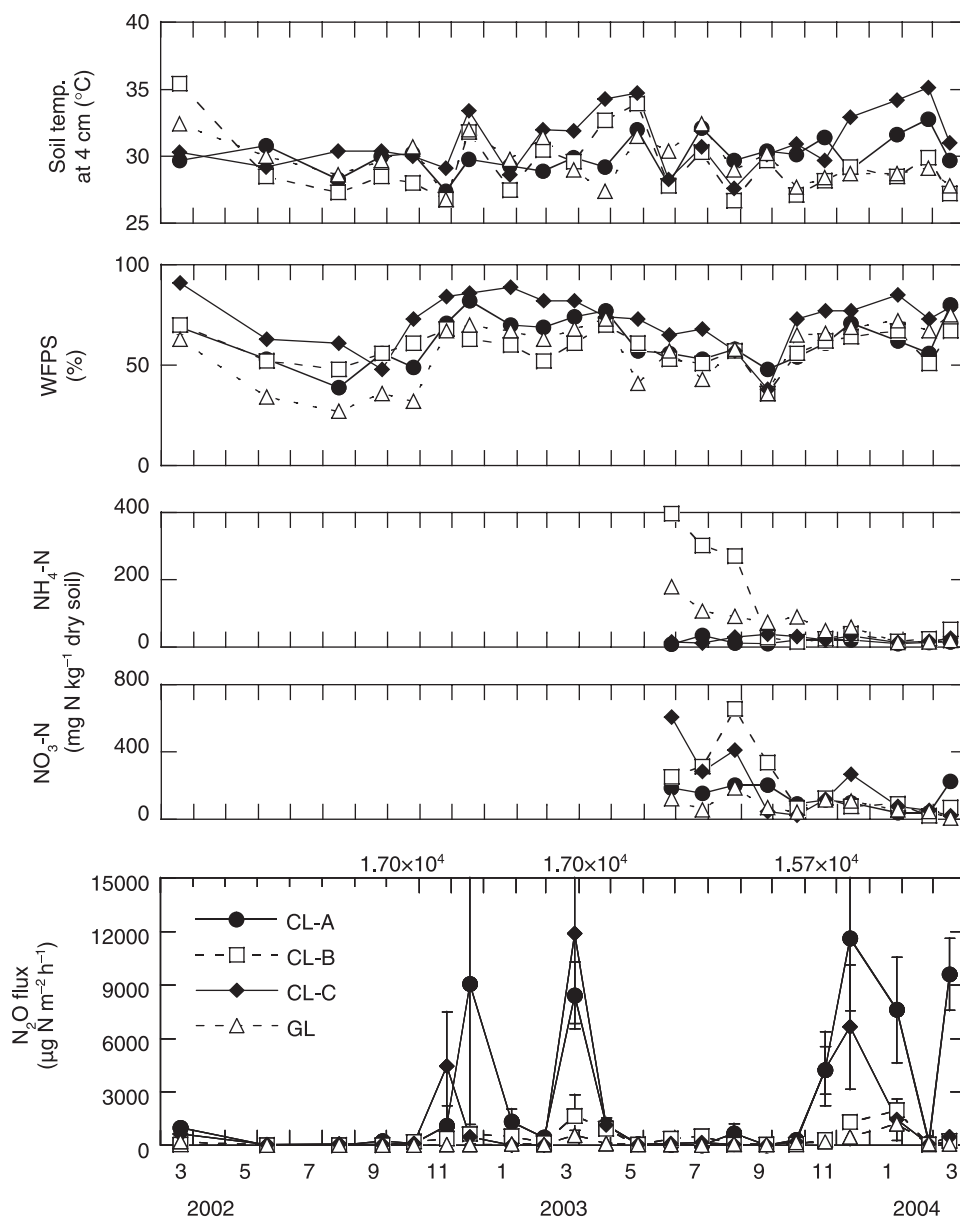
Figure 4 Seasonal changes in soil temperature, water-filled pore space (WFPS), soil NH$_4$–N, soil NO$_3$–N and N$_2$O fluxes at the agricultural sites. Error bars show standard deviations. CL, cropland; GL, grassland.

gas (atmospheric level). Gas fluxes were calculated as follows:

$$F = \rho \times V/A \times \Delta c/\Delta t \times 273/(273 + T) \times \alpha$$

where F is the flux (µg N m^{-2} h^{-1}); ρ is the gas density of N$_2$O (1.978×10^9 µg m^{-3}); V is the volume of the chamber (m^3); A is the cross-sectional area of the chamber (m^2); $\Delta c/\Delta t$ is the ratio of change in the gas concentration (c) inside the chamber per unit time (t) during the sampling period (m^3 m^{-3} h^{-1}); T is the air temperature (°C) and α is a conversion factor from N$_2$O to N (=28/44).

The annual N$_2$O emissions were calculated from the monthly mean flux values as follows:

$$\text{Annual N}_2\text{O emission} = \sum_{i=1}^{n-1} F_i \times D_i$$

where F_i is the mean gas flux (kg N ha^{-1} day^{-1}) between two sampling times (i.e. for time interval i), D_i is the number of days in the sampling interval and n is the number of sampling times. Missing values in the regenerated and burned forests because of flooding were

replaced with flux data from the water surface, which was measured using a tall chamber (30 cm × 30 cm × 60 cm darkened acryl chamber) in February 2005. The mean N$_2$O fluxes (flooding water depth) from the regenerated and burned forests in February 2005 were 4.7 µg N m^{-2} h^{-1} (at a depth of 20 cm in flooding water) and −1.6 µg N m^{-2} h^{-1} (at a depth of 10 cm in flooding water), respectively.

Soil sampling and analysis

General soil properties

Disturbed soil samples used for chemical analysis were taken from a depth of 0–10 cm at each site in 2002. The forest soils were divided into a litter layer and a peat layer. In the regenerated and burned forests, soil samples were taken both before and after the forest fire in 2002.

Sieved (2 mm), air-dried soil samples were used for the chemical analysis, except for NH$_4$–N content. Soil pH (H$_2$O basis) was measured with a glass electrode pH meter (pH meter F-22, Horiba, Kyoto, Japan) in a 1:20 soil : deionized water mixture. The concentrations of NO$_3$–N were also measured in this suspension using ion chromatography (Dionex QIC Analyzer, Dionex Japan, Osaka, Japan).

Sieved (2 mm), air-dried soil samples were extracted with 1 mol L^{-1} ammonium acetate (pH 7), washed with 80% ethanol (pH 7) and subsequently extracted with 1 mol L^{-1} KCl. Cation exchange capacity (CEC) was determined by analyzing NH$_4$–N concentrations using a 1 mol L^{-1} KCl extraction. Exchangeable cations were determined by analyzing their concentrations in a 1 mol L^{-1} ammonium acetate (pH 7) extraction (Kamewada 1997). Concentrations of Ca^{2+} and Mg^{2+} were determined using atomic-absorption spectrometry and those of K$^+$ and Na$^+$ were determined using atomic-emission spectrometry (Z5010, Hitachi, Tokyo, Japan). Base saturation was calculated by dividing the sum of the exchangeable cations by the CEC.

After grinding the air-dried samples using an agate pestle and mortar, total C and N content were determined using an N/C analyzer (NC-1000, Sumika Chemical Analysis Service, Osaka, Japan).

Fresh samples that had not been allowed to dry were extracted with 2 mol L^{-1} KCl solution (1:20 dried soil : water) and NH$_4$–N concentrations in the extracted solution were determined using colorimetry (the indophenol-blue method) with a UV-VIS spectrophotometer (UV mini 1240, Shimadzu, Kyoto, Japan) following the method of Hidaka (1997).

Undisturbed 100-cm^3 core samples were collected in February 2005 for measurement of the air ratio using a soil volume analyzer (DIK-1110, Daiki Rika Company, Saitama, Japan). The cores were then oven-dried at 105°C for 48 h and reweighed to determine their bulk density and volumetric water content. Porosity equaled the sum of the air ratio and the volumetric water content.

Temporal changes in soil NH$_4$–N and NO$_3$–N

Disturbed soil samples were taken from a depth of 0–3 cm at each site simultaneously with the gas flux measurements from June 2003 to March 2004. Soil NH$_4$–N and NO$_3$–N contents were measured as described above.

Statistical analysis

One-way ANOVA followed by a Steel–Dwass test was used to compare the means of soil temperature, WFPS, soil NH$_4$–N, soil NO$_3$–N and N$_2$O flux at each site. An unpaired t-test was used to compare the means of soil temperature, WFPS and N$_2$O flux between the dry and rainy seasons. A least significant difference (LSD) test was used to compare the annual N$_2$O emissions between sites. All statistical analyses were conducted using Excel statistics ver. 5.0 (Esumi Company, Tokyo, Japan).

RESULTS

Soil properties

General soil properties

The pH values in the peat layer of the agricultural sites (4.4–6.0) were higher than those in the soils of the forest sites (3.4–3.9). Soil NO$_3$–N levels at the agricultural sites (225–292 mg N kg^{-1} dry soil) were higher than those at the forest sites (0.43–127 mg N kg^{-1} dry soil). In contrast, soil NH$_4$–N levels at the agricultural sites (17–67 mg N kg^{-1} dry soil) were lower than those at the forest sites (22–511 mg N kg^{-1} dry soil) (Table 2). The soil bulk density was similar for all croplands (0.40–0.42 Mg m^{-3}) and was higher than that in the grassland (0.30 Mg m^{-3}), natural forest (0.14 Mg m^{-3}) and burned forest (0.22 Mg m^{-3}) (Table 2). Porosity values for the soils of the natural forest, burned forest, grassland and croplands were 90%, 84%, 73% and 76–77%, respectively (Table 2).

Seasonal changes in soil NH$_4$–N and NO$_3$–N

The soil NH$_4$–N contents (0–3 cm depth, mean ± SD) in the natural forest (643 ± 117 mg N kg^{-1} dry soil) were significantly ($P < 0.01$) higher than those at other sites (Figs 3,4). The mean soil NH$_4$–N contents in the regenerated forest, burned forest, grassland and croplands A, B and C were 146 ± 94, 83 ± 60, 70 ± 51, 17 ± 7, 118 ± 146 and 24 ± 9 mg N kg^{-1} dry soil, respectively (Figs 3,4).

The soil NO$_3$–N contents (0–3 cm depth, mean ± SD) were significantly ($P < 0.05$) higher in the croplands
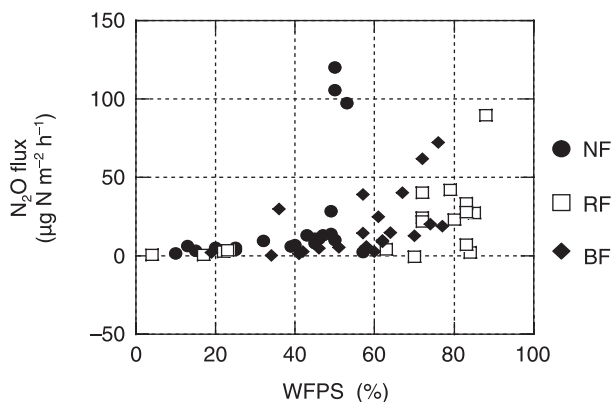
Figure 5 The relationships between water-filled pore space (WFPS) and N$_2$O flux from the forest sites. BF, burned forest; NF, natural forest; RF, regenerated forest.

Figure 6 The relationships between water-filled pore space (WFPS) and N$_2$O flux from the agricultural sites. CL, cropland; GL, grassland.

(croplands A, B and C, 136 ± 68, 201 ± 195 and 191 ± 197 mg N kg^{-1} dry soil) than in the natural forest (26 ± 34 mg N kg^{-1} dry soil) and burned forest (28 ± 26 mg N kg^{-1} dry soil) (Figs 3,4). The soil NO$_3$–N contents in the regenerated forest and grassland were 43 ± 42 and 81 ± 52 mg N kg^{-1} dry soil, respectively (Figs 3,4). The soil NO$_3$–N contents in the grassland were constantly low, whereas those in the three croplands during the dry season were initially high and decreased during the rainy season (Fig. 4).

N$_2$O fluxes

N$_2$O fluxes from forest sites

The highest N$_2$O fluxes from forest sites were mainly found during the rainy season (Fig. 3). The dry season N$_2$O fluxes (mean $\pm$ SD) from the natural, regenerated and burned forests were 5.0 ± 2.8, 19 ± 17 and 8.5 ± 12 μg N m^{-2} h^{-1}, respectively (Fig. 3, Table 3). While the rainy season N$_2$O fluxes (mean $\pm$ SD) from the natural, regenerated and burned forests were 49 ± 63, 55 ± 100 and 26 ± 22 μg N m^{-2} h^{-1}, respectively (Fig. 3, Table 3). The mean N$_2$O fluxes in the natural and burned forests were significantly ($P < 0.05$) higher in the rainy season than in the dry season (Table 3).

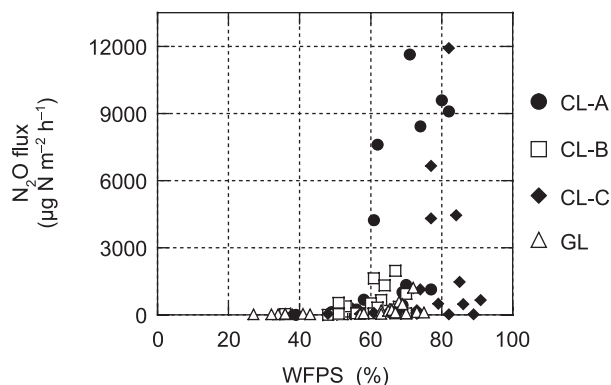The relationships between WFPS and N$_2$O flux from the forest sites are shown in Fig. 5. High N$_2$O fluxes

(>50 μg N m^{-2} h^{-1}) were found when WFPS levels in the natural, regenerated and burned forests were 45–55, 85–90 and 70–80%, respectively.

N$_2$O fluxes from agricultural sites

N$_2$O fluxes (mean $\pm$ SD) from all croplands and from the grassland were significantly ($P < 0.05$) higher during the rainy season than during the dry season (grassland 257 ± 335 *vs* 50 ± 55 μg N m^{-2} h^{-1}; cropland A $4.30 \times 10^3 \pm 4.30 \times 10^3$ *vs* 163 ± 219 μg N m^{-2} h^{-1}; cropland B 637 ± 642 *vs* 154 ± 188 μg N m^{-2} h^{-1}; cropland C $2.44 \times 10^3 \pm 3.55 \times 10^3$ *vs* 94 ± 62 μg N m^{-2} h^{-1}) (Fig. 4, Table 3).

The relationships between WFPS and N$_2$O flux from the agricultural sites are shown in Fig. 6. Significantly ($P < 0.05$) higher N$_2$O fluxes were found when WFPS was greater than a threshold value of 60% for grassland and croplands A and B, and greater than 70% for cropland C. However, N$_2$O fluxes decreased at the highest levels of WFPS (approximately 90%) in cropland C (Fig. 6).

The relationships between the soil NO$_3$–N concentration (0–3 cm depth) and the N$_2$O flux from agricultural sites at WFPS values below and above the threshold value (60% for grassland and croplands A and B and 70% for cropland C are shown in Fig. 7). N$_2$O fluxes from all croplands and the grassland were significantly ($P < 0.05$) lower when WFPS was below the threshold value, regardless of the soil NO$_3$–N concentration (Fig. 7).

Table 4 Annual N$_2$O emissions from the study sites

Year	N$_2$O emission (kg N ha^{-1} year^{-1})						
	Natural forest	Regenerated forest	Burned forest	Grassland	Cropland A	Cropland B	Cropland C
2002–2003	0.62 ± 0.11a	0.40 ± 0.32a	0.97 ± 0.65a	7.1 ± 1.2b	131 ± 59d	21 ± 5.4c	83 ± 26d
2003–2004	4.4 ± 1.2b*	4.0 ± 1.9b*	1.5 ± 0.70a	23 ± 9.8c*	259 ± 44f*	52 ± 8.2d*	151 ± 36e

Numbers represent the cumulative annual value and standard deviation. Numbers within a row followed by different letters differ significantly among sites ($P < 0.05$, least significant difference [LSD] test). Asterisks (*) denote significant differences between the 2002–2003 value and the 2003–2004 value ($P < 0.05$, LSD test).
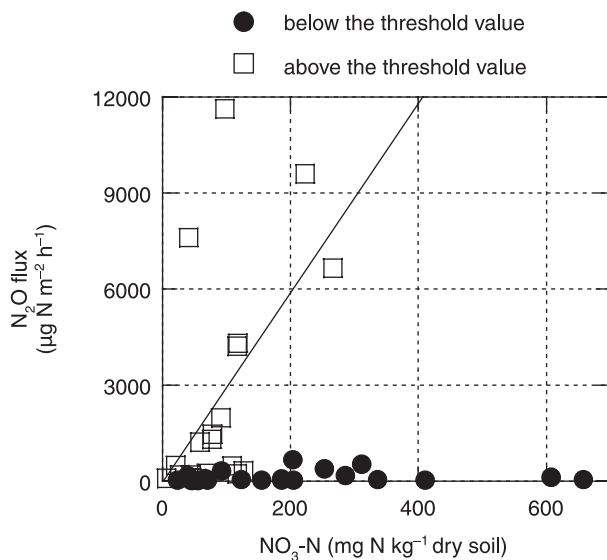
Figure 7 The relationship between soil NO_3–N (0–3 cm in depth) and N_2O flux from soils of agricultural lands below and above the threshold value of water-filled pore space (WFPS) (60% for grassland and croplands A and B, and 70% for cropland C; Fig. 6). A significant correlation was found ($n = 20$, $r = 0.536$, $P < 0.05$) for the values above the threshold value of WFPS.

A significant correlation was found ($r = 0.536$, $P < 0.05$) between soil NO_3–N and N_2O flux above the threshold value of WFPS (Fig. 7).

N_2O concentration in the soil gas

The N_2O concentration was higher at greater depths in the soil, and the maximum N_2O concentration was found deeper in the soil (Fig. 8), indicating that there was a source of N_2O in deeper soil horizons. The maximum N_2O concentration was higher in soils of the agricultural sites ($988–5.01 \times 10^3$ p.p.m.v) than in the soils of the forest sites (31.3–842 p.p.m.v). In both groups of sites, N_2O concentration increased during the rainy season and decreased during the dry season.

Annual N_2O emissions

Annual N_2O emissions from croplands (mean $\pm$ SD) were 131 ± 59 kg N ha⁻¹ year⁻¹ for cropland A, 21 ± 5.4 kg N ha⁻¹ year⁻¹ for cropland B and 83 ± 26 kg N ha⁻¹ year⁻¹ for cropland C in 2002–2003, and were 259 ± 44, 52 ± 8.2 and 151 ± 36 kg N ha⁻¹ year⁻¹, respectively, in 2003–2004. These values were significantly ($P < 0.05$) higher than those from all other sites (Table 4). Annual N_2O emissions from unfertilized grassland were 7.1 ± 1.2 (2002–2003) and 23 ± 9.8 (2003–2004) kg N ha⁻¹ year⁻¹, and these values were also significantly ($P < 0.05$) higher than those from the forest sites (Table 4). There were no significant differences in annual N_2O emissions among

the forest sites (0.62 ± 0.11 kg N ha⁻¹ year⁻¹ for natural forest, 0.40 ± 0.32 kg N ha⁻¹ year⁻¹ for regenerated forest and 0.97 ± 0.65 kg N ha⁻¹ year⁻¹ for burned forest) in 2002–2003. However, the annual N_2O emission from burned forest (1.5 ± 0.70 kg N ha⁻¹ year⁻¹) was significantly lower than that from natural forest (4.4 ± 1.2 kg N ha⁻¹ year⁻¹) and regenerated forest (4.0 ± 1.9 kg N ha⁻¹ year⁻¹) in 2003–2004 (Table 4). There was a significant difference in annual N_2O emission between years for all sites except for burned forest and cropland C (Table 4). As a whole, annual N_2O emission was significantly correlated between years for each site ($r = 0.998$, $P < 0.01$; Fig. 9). From the slope of the regression line, the annual N_2O emission in 2003–2004 was 1.9-fold the value in 2002–2003.

DISCUSSION

Temporal variation in N_2O emissions

N_2O fluxes were significantly higher during the rainy season (November to May) than during the dry season (June to October) at all sites except the regenerated forest site (Table 3). N_2O emission from both forests and agricultural lands was strongly stimulated by high soil moisture, expressed in terms of the WFPS value (Figs 5,6).

N_2O fluxes in all croplands and the grassland were significantly lower when WFPS was below the threshold values (60–70%), regardless of soil NO_3–N concentration (Figs 6,7). Consequently, changes in soil moisture conditions could be an important factor controlling seasonal changes in N_2O emission from tropical peat soils at these study sites.

However, low N_2O emission was also found when WFPS was higher than the threshold value, and particularly low N_2O emissions were found for the soils of cropland C at WFPS values of approximately 90% (Fig. 6). These low N_2O emissions could have been caused by the predominance of N_2 as a product of denitrification processes under anaerobic conditions. This hypothesis is supported by the suggestion of Davidson et al. (2000) that N_2 production becomes higher than N_2O production through denitrification at WFPS values >80%. In addition, the results with air-dried samples showed a significant relationship between soil NO_3–N and N_2O flux when WFPS was above the threshold value (Fig. 7). This result suggests that soil NO_3–N could be a controlling factor for temporal changes in N_2O flux at high soil moisture levels at these study sites.

Inubushi et al. (2003) investigated seasonal changes in N_2O emission over a year in tropical peatlands in South Kalimantan. N_2O emissions from abandoned agricultural land and secondary forest were low (ranging from −40 to 30 μg N m⁻² h⁻¹), and they found no clear seasonal changes in N_2O emission. They explained this result as inhibition of N_2O emission by floodwater.
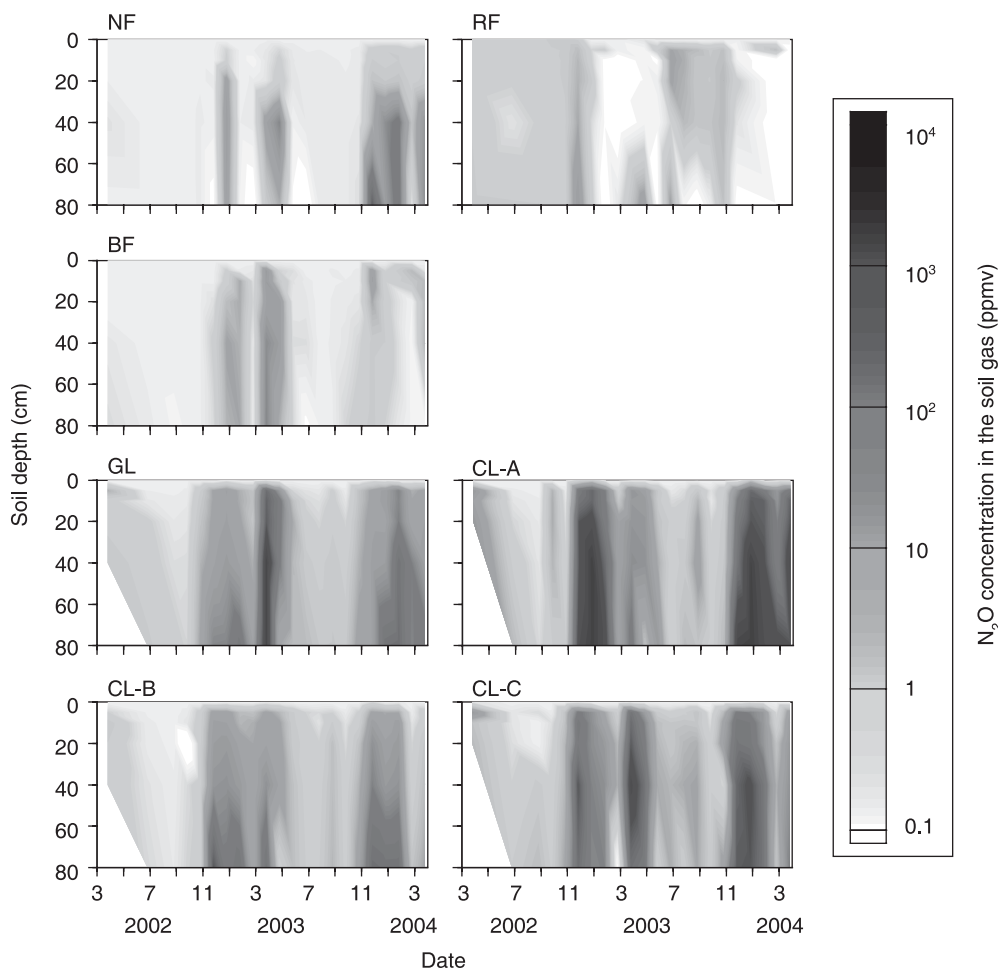
Figure 8 Seasonal changes in N_2O concentration in the soil gas (p.p.m.v). BF, burned forest; CL, cropland; GL, grassland; NF, natural forest; RF, regenerated forest.

Our study sites had a greater depth to the groundwater table and drier soil conditions than was the case at Inubushi *et al.*'s (2003) study sites (where the depth of the groundwater table ranged from −2.1 cm [above the soil surface] to 15 cm [below the soil surface]). Consequently, soil moisture fluctuations could become a major factor that controls N_2O fluxes from tropical peatlands.

Because soil moisture is controlled by precipitation, annual N_2O emissions should have been significantly higher in 2003–2004, with a total annual precipitation of 2339 mm, than in 2002–2003, with a total annual precipitation of 1994 mm. In fact, this is exactly what we observed (Table 4, Fig. 9).

Effect of forest fire on N_2O emissions

Ishizuka *et al.* (2002) reported that N_2O emissions from primary forests on Ultisols in Sumatra, Indonesia, ranged from 0.13 to 0.39 kg N ha^{-1} year^{-1}, and increased after clear cutting and burning in the forest.

In our study, annual N_2O emission from natural forest (0.62–4.4 kg N ha^{-1} year^{-1}; Table 4) was higher than that reported by Ishizuka *et al.* (2002). There was no significant difference in annual N_2O emission among the natural, regenerated and burned forests in 2002–2003. However, the annual N_2O emission from burned forest was significantly lower than the emission from the natural and regenerated forests in 2003–2004. Consequently, the effect of forest fire on N_2O fluxes at our forest sites was neither consistent nor clear.

Effects of agriculture on N_2O emission

Soil pH at the agricultural sites became higher than that in forest sites owing to the application of ash. Murakami *et al.* (2005) reported that lime addition to tropical peat raised the soil pH, and subsequently enhanced the decomposition of organic matter. The organic matter decomposition rates at our agricultural sites could, thus, have been higher than those at the forest sites.
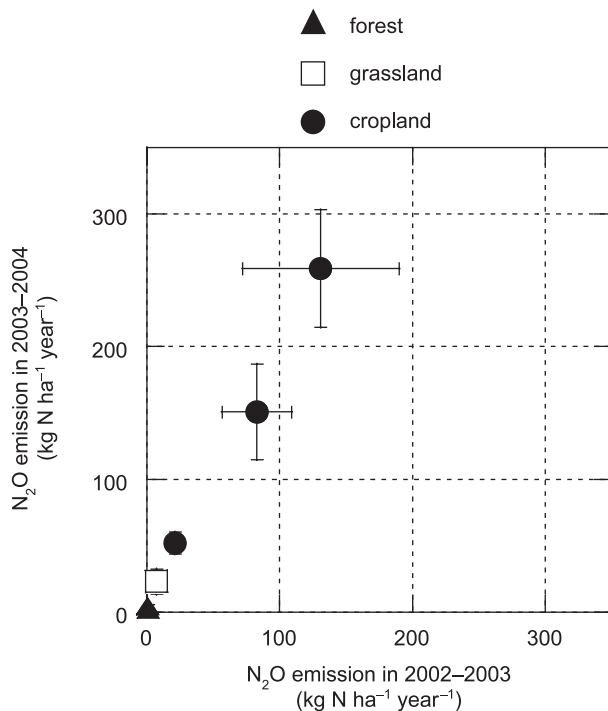
Figure 9 The relationship between N$_2$O emission in 2002–2003 and 2003–2004 at each site. Error bars are standard deviations. A significant correlation was found ($y = 1.90x + 4.16$, $r = 0.998$, $P < 0.01$) between the years.

N$_2$O emission was significantly higher in the croplands than in the unfertilized grassland, and was significantly higher in the grassland than at the three forest sites (Table 4). This indicates that agricultural practices could increase N$_2$O emissions from tropical peatland. The N$_2$O emissions from cropland sites were also higher than those in cultivated boreal and temperate organic soils (8.3–11.0 kg N ha^{-1} year^{-1}, Maljanen *et al*. 2004; 2–38 kg N ha^{-1} year^{-1}, Kasimir-Klemedtsson *et al*. 1997) and those in tropical peatland soils in South Kalimantan (−1.1 to −0.37 kg N ha^{-1} year^{-1}, Inubushi *et al*. 2003) and Sarawak, Malaysia (1.2–3.3 kg N ha^{-1} year^{-1}, Melling *et al*. 2005). Annual N$_2$O emission from the unfertilized grassland (average 15 kg N ha^{-1} year^{-1}) was close to the emission factor proposed by the Intergovernmental Panel on Climate Change (2000) for N$_2$O emission from cultivated organic soils (16 kg N ha^{-1} year^{-1}). Only a weak and non-significant correlation was found between applied N (Table 1) and N$_2$O emissions (Table 4) in all croplands and the grassland ($y = 0.065x + 46.5$, $r = 0.348$, $P = 0.398$). This suggests that there are high variations in fertilizer-induced N$_2$O emission. We calculated the fertilizer-induced N$_2$O emission factor, which was expressed as a percentage of N$_2$O emission against applied N by assuming that N$_2$O emission from the grassland represented the value for an unfertilized soil. The emis-

sion factor in croplands ranged from 1.8 to 36%, and these values were considerably higher than the value of 1.25% for N$_2$O emission induced by nitrogen fertilizer that was proposed by the Intergovernmental Panel on Climate Change (2000). However, high N$_2$O emission from the unfertilized grassland indicated that N$_2$O emission owing to peat decomposition could be high and also might cause this high variation in N$_2$O emission from these agricultural soils. Therefore, quantification of N$_2$O emission owing to peat decomposition is needed to accurately estimate the N$_2$O emission factor in these soils. It is also important to elucidate the reason for the high emission factors that we observed and the factors responsible for the high variation in future research.

The relationship between WFPS and N$_2$O flux suggested that denitrification might be an important process for N$_2$O emission from these agricultural land sites (Fig. 6). While inorganic nitrogen was supplied to these soils as NH$_4$–N by urea fertilization and organic matter decomposition. Therefore, nitrification also played an important role in N$_2$O emission from these soils as a direct N$_2$O source and a NO$_3$–N supply for denitrification. Nitrifier denitrification (Wrage *et al*. 2001) might also contribute to high N$_2$O emission from these soils. Further study is required to clarify the mechanism of considerably high N$_2$O emission associated with agricultural land-use change in tropical peatland.

Our results suggest that changing land use from forestry to agriculture and agricultural practices, such as drainage, fertilization, manuring and liming, may change the microenvironmental conditions in the tropical peat soil to promote N$_2$O production.

Conclusions

In our study area, N$_2$O emission from tropical peatland soils was mainly controlled by a combination of soil moisture conditions and land use (i.e. forestry *vs* grassland *vs* agriculture). The effect of forest fires on N$_2$O emission from these soils was not clear. Agricultural practices may increase N$_2$O emission from tropical peatland soils as a result of nitrogen application and changes in the N$_2$O production potential of the soil.

ACKNOWLEDGMENTS

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