

ORIGINAL ARTICLE

Temporal variations in soil–atmosphere methane exchange after fire in a peat swamp forest in West Siberia

Tomoko NAKANO¹, Wataru TAKEUCHI², Gen INOUE³,
Masami FUKUDA⁴, and Yoshifumi YASUOKA²¹Department of Geography, Tokyo Metropolitan University, Hachioji 192-0397, ²Institute of Industrial Science, University of Tokyo, Tokyo 153-8505, ³National Institute for Environmental Studies, Tsukuba 305-8506 and ⁴Institute of Low Temperature Science, Hokkaido University, Sapporo 060-0819, Japan

Abstract

Temporal variations in methane (CH₄) exchange between the soil and the atmosphere during a period of 3 years after a forest fire were estimated by combining field measurements of CH₄ flux with an analysis of satellite images. The study area was located in a boreal peat swamp forest in the West Siberian plain that experienced a severe fire in the summer of 1998. The surface of the burned area was classified into bare soil, open water and recovered vegetation. In the summers of 1999 and 2000, CH₄ fluxes, using a closed-chamber method, and environmental variables, such as soil temperature and soil water content, for each of the three surface types in the burned area and in the unburned area were measured. In general, CH₄ fluxes were controlled by the surface moisture in the burned area and by the temperature in the unburned forest. Temporal changes in the areal coverage of soil, water and vegetation in the burned area were investigated using NOAA AVHRR (Advanced Very High Resolution Radiometer) data with subpixel land-cover characterization. Based on the satellite information, temporal changes in the moisture conditions of the burned surface were estimated and temporal variations in the CH₄ fluxes for the entire burned area were calculated. The cumulative CH₄ emission rates from the entire burned area during the summer (from June to August) were estimated to range from 0.39 to 0.48 g C m⁻² during the period from 1999 to 2001. In contrast, unburned forest soils were consistently net CH₄ consumers. The cumulative consumption rate during the summer was calculated to be approximately 0.4 g C m⁻² based on a reanalysis air temperature dataset. As the surface soil had become extremely wet since the fire, the soil had become a net emitter of CH₄ after the fire disturbance, although CH₄ oxidation predominated in the unburned forest.

Key words: forest fire, NOAA AVHRR, soil–atmosphere methane exchange, spectral mixture analysis, western Siberia.

INTRODUCTION

Soils are major global sources and sinks of methane (CH₄) and play an important role in regulating atmospheric CH₄ concentrations. Natural wetlands are large sources of CH₄ emission to the atmosphere (e.g. Christensen *et al.* 1995; Nakano *et al.* 2000; Nykänen *et al.* 2003; Whalen and Reeburgh 1992), whereas aerobic

environments such as upland forest soils consume CH₄ owing to microbial oxidation (e.g. Borken *et al.* 2000; Castro *et al.* 1995; Priemé and Christensen 1997; Smith *et al.* 2000). The direction and rate of the net CH₄ exchange between soils and the atmosphere are commonly highly variable both spatially and temporally and are controlled by various environmental factors such as soil moisture, soil temperature and chemical properties (e.g. pH, nitrogen availability). Among these factors, soil moisture is the major determinant of whether soils are net CH₄ emitters or consumers because it directly influences the availability of oxygen.

Fire is ubiquitous in the boreal region and is extremely common in boreal forests (Kasischke 2000). Fire is associated with important consequences for carbon

Correspondence: Dr T. NAKANO, Department of Geography, Tokyo Metropolitan University, 1-1 Minami-Osawa, Hachioji 192-0397, Japan. Email: nakanot@comp.metro-u.ac.jp

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storage in boreal forests and is linked to immediate pulses of trace gases to the atmosphere through combustion. Fire also causes long-term surface disturbances and changes in the gas exchange between soil and the atmosphere. In many studies, carbon emissions during burning and post-fire release of CO₂ have been investigated using satellite imagery, historical statistics, modeling and/or field soil surveys (Auclair and Carter 1993; Cahoon *et al.* 1994; Conard *et al.* 2002; Dixon and Krankina 1993; French *et al.* 2000; Isaev *et al.* 2002; Kasischke and Bruhwiler 2003; Kasischke *et al.* 1995; Pitkänen *et al.* 1999; Turetsky and Wieder 2001; Zoltai *et al.* 1998). In addition, changes in the surface CH₄ flux before and after controlled fires (Kim and Tanaka 2003; Levine *et al.* 1990) have been investigated using a chronosequential method (Burke *et al.* 1997). However, continuous variations in the soil-atmosphere CH₄ exchange over several years at a particular burned site have been examined in only a few studies. Another consideration is the need to scale up the CH₄ flux measured at points or in a small area to an entire burned area to evaluate the impact of fire on the soil-atmosphere CH₄ exchange.

Remote sensing is an appropriate technique for monitoring the land surface over a wide area. High-resolution images such as LANDSAT TM, SPOT HRV and Terra ASTER images have been used to map forest fire extent and damage and to estimate carbon emissions from burned areas (Garcia-Haro *et al.* 2001). However, high-resolution images may not cover a target area frequently because of their narrow swath width. Coarse spatial resolution data, such as NOAA AVHRR (Advanced Very High Resolution Radiometer) data, have the advantages of wide coverage and high temporal resolution, although this sensor may not detect fine spatial structures in mixtures of vegetation, soil and water in land ecosystems, which results in the existence of mixed pixels. However, a technique for extracting subpixel cover type reflectance levels from the mixed pixels of a coarse spatial resolution image has been developed (Oleson *et al.* 1995; Takeuchi *et al.* 2003).

The study area was located in a boreal forest in the West Siberian plain that experienced a forest fire in 1998. The surface environment of the burned area consisted of a mosaic of small ponds (open water), bare soil and invading herbaceous plants, whose distribution had varied temporally over the years following the fire. The purpose of the present study was to estimate temporal variations in the CH₄ exchange between soil and the atmosphere after fire disturbance for the entire burned area. To achieve this objective, first we conducted flux measurements at the burned sites and the unburned site and revealed relationships between the CH₄ flux and environmental variables. Second, temporal variations in

the surface environment were analyzed based on NOAA AVHRR images covering the study area. Finally, we estimated temporal variations in CH₄ flux from the entire burned area by combining the results from the field measurements with the results from an analysis of the remotely sensed data.

MATERIALS AND METHODS

Study sites

The study area was located in a boreal forest (latitude 56°52'N, longitude 83°17'E) near Plotnikovo, Tomsk Oblast, in the southern part of the West Siberian plain. This plain is an extremely paludinous area characterized by very large carbon stocks resulting from peat accumulation (Yefremov and Yefremova 2001). The study area was covered with a thick organic soil layer derived from peat. In the area, the annual mean air temperature was -1.1°C, mean monthly temperatures ranged between -18.6°C and -17.6°C throughout the year, and the annual precipitation was approximately 530 mm (Lapshina *et al.* 2001). Measurements were conducted in an unburned forest and in an adjacent area that was burnt in August 1998. Based on a comparison between satellite images before and after the fire, the burned area was estimated to extend over 13.8 km² (Takeuchi *et al.* 2002).

In the unburned area, the forest consisted of white birch (*Betula pendula*) with occasional Scots pines (*Pinus sylvestris*). The ground vegetation included shrubs (*Ledum palustre*, *Chamaedaphne calyculata*, *Rubus chamaemorus*) and herbaceous plant communities. A 3-cm layer of decomposing litter (L horizon) was underlain by a thick (approximately 90 cm), black, organic layer (H horizon), which overlaid a silty, mineral horizon (B horizon).

In general, forest fires in West Siberia burn organic-rich soils and tree roots, not tree canopies. Thus, most trees in the burned areas fall down during the fire and leave innumerable root scars on the soil surface. The soil surface in the burned study area was uneven (approximately 50-cm high undulation) and was entirely covered with fallen trees (Fig. 1). The burned area was covered with snow until the beginning of May. The surface was flooded after snowmelt and gradually dried out owing to evaporation during the summer. Although most plants were completely burnt in August 1998, the ground surface was gradually invaded by plants in the summer of the following years: first liverwort (*Marchantia* Sp.) and successively other herbaceous plants (mainly *Epilobium angustifolium*). The soil had no litter layer, but the black organic H horizon was approximately 30 cm thick and was underlain by a silty, mineral horizon (B horizon). Some charcoal fragments were included in the upper part of the H horizon. The surface environment in the burned area



Figure 1 The burned area in August 1999; almost the entire area was covered with fallen trees.

was classified into open water, unvegetated bare soil and plant-covered soil; the distribution and areal coverage of each component varied across seasons and years.

Methodology of field observations

Measurements of net CH_4 flux were conducted at the following sites: unburned forest (site U), open water (site W), bare soil (site S) and vegetation (site V) sites in the burned area using a closed-chamber method. The system comprised a 0.4-m-tall transparent chamber, stainless-steel collars with an enclosure of 0.152 m^2 , a portable CH_4/CO_2 analyzer equipped with a semiconductor CH_4 detector, and a 12-V lead-acid battery. The chamber was made of acrylic plastic and was equipped with sampling ports, a Tedlar bag to equalize pressure, a thermometer and a battery-operated fan. Before the initiation of the measurements, two collars were embedded into the soil to a depth of 10 cm at site U, and they remained throughout the study. At sites W, S and V, two or three collars were inserted into the soil at each site on the first day of each study period and remained only for the duration of the period, because the surface environment at the burned sites changed with time. The chamber was closed for 10 min by fitting a water-filled groove into the collar for the measurements. Sample air was continuously pumped at a rate of approximately $1 \text{ dm}^3 \text{ min}^{-1}$ from the chamber through a polyethylene tube to the CH_4 analyzer and back to the chamber. Details of the analysis have been described by Nakano *et al.* (2004). The CH_4 concentration in the chamber was recorded at 5 s intervals with a data logger (NR-1000, Keyence Co., Osaka, Japan). CH_4 flux at the drier unburned site was determined from the rate constant of the exponential curve fit to the concentration changes over a 10 min sampling period, whereas the flux was calculated using a linear regression of the

concentration changes over the first 3 min at the wetter burned site that was more prone to ebullition and disturbances. The minimum r^2 value for significance was 0.90, and any data that fell below this value were rejected. The detection limit was approximately $\pm 0.03 \text{ mg C m}^{-2} \text{ day}^{-1}$. A positive CH_4 flux represents a transfer from the soil surface to the atmosphere and a negative flux indicates CH_4 uptake by the soil.

Flux measurements were carried out during 1-week periods in August 1999, June 2000 and September 2000. At site U, flux was measured four times per day for each of two collars to examine temporal variations in methane oxidation by boreal forest soil (Nakano *et al.* 2004), and the daily mean flux was calculated based on these data. At sites W and S, flux measurements were conducted three to five times at each site for each period. Flux at site V was measured only in September 2000 because herbaceous plants invaded our observation area after the measurement of June 2000.

Environmental data were collected concurrently with the flux measurements. At sites U, S and V, the soil temperature at a depth of 10 cm and volumetric water content (VWC) at the time of the flux measurements were determined using a handheld digital thermometer (ST-920, Testoterm GmbH & Co., Berlin, Germany) and time domain reflectometry (TDR) probes (0–12 cm depth; HydroSense, Campbell Scientific Inc., Logan, UT, USA) positioned vertically from the surface into the soil, respectively. At site W, the depth of standing water was measured at the time of the flux measurements. Measurements of the soil temperature, soil moisture and water depth were carried out at 5 points around each collar immediately after the flux measurement was recorded and averaged. In addition, air temperatures at 1.5 m above ground level at both the burned and unburned areas were measured every 30 min using data loggers with thermistor probes (SK-L200T, Sato Keiryoki Mfg. Co., Tokyo, Japan). The spatial distribution of soil moisture and the depth of standing water in the burned area were also surveyed every 5 m along two 50-m sampling transects in August 1999, June 2000, September 2000 and August 2001.

Surface soil samples at both the burned and unburned sites were collected in triplicate and brought back to Japan in September 2000. The relative gas diffusion coefficient (D/D_0) was determined using Osozawa's (1987) diffusion chamber method with 100-cm^3 undisturbed soil cores. Soil core porosity was measured with an actual volume meter (DIK-1000, Daiki Rika Kogyo Co., Tokyo, Japan) and the bulk density was determined gravimetrically. The soil cores were also used for cross-checking the TDR results using the gravimetric method. Total carbon and nitrogen contents were determined by dry combustion using a CN analyzer (Vario-EL, Elementar Analysensysteme GmbH,

Hanau, Germany) equipped with a thermal conductivity detector. Soil pH and electrical conductivity (EC) were determined with a glass electrode pH meter and an EC meter, respectively, in a supernatant suspension of 1:10 soil : deionized water mixture.

Analysis of NOAA AVHRR imagery

A total of 715 AVHRR images from the summers of 1998–2001 were acquired. NOAA AVHRR channels 1 and 2 were calibrated to reflectance values using coefficients based on the NOAA KLM User's Guide (Goodrum *et al.* 1999), and cloud-free areas were selected carefully with visual interpretation. Corrections of the atmospheric conditions and bidirectional reflectance distribution function (BRDF) were not carried out. Pixels over 45° of the sensor scan angle were considered unacceptable for processing. The AVHRR data were geometrically corrected based on ground control point (GCP) matching using PaNDA software, and the registration error over the image was less than 1 pixel. PaNDA is a free software package for NOAA data analysis (Shimoda *et al.* 1998). After screening out inappropriate images, 24 images were selected for analysis (Table 1).

The NOAA AVHRR is one of the few space-borne sensors currently capable of acquiring radiometric data over a broad range of view angles. However, the relatively coarse spatial resolution of AVHRR (1.1–4.3 km)

most often results in measurements of mixed land covers and pixel unmixing is required (Asner *et al.* 1997). A linear spectral mixture model based on three end members, vegetation, soil and water (Yamagata *et al.* 1997), was defined by

$$\text{AVHRR}_{\text{CH1}} = a_{11}V + a_{12}S + a_{13}W \quad (1)$$

and

$$\text{AVHRR}_{\text{CH2}} = a_{21}V + a_{22}S + a_{23}W \quad (2)$$

where $\text{AVHRR}_{\text{CH1}}$ and $\text{AVHRR}_{\text{CH2}}$ are the reflectance of channels 1 and 2, respectively, and V, S and W are the fractional coverage of vegetation, soil and water; a_{ij} represents the end members and ranges from 0 to 1. The constraint equation (Eq. 3) requires that the sum of V, S and W in each pixel equal 1.

$$V + S + W = 1 \quad (3)$$

By solving the three equations with the AVHRR reflectance levels of channels 1 and 2 simultaneously, we were able to calculate the fractional coverage of V, S and W within a pixel. Results of the linear spectral mixture model were verified based on the field survey.

RESULTS

Field measurements

Summary statistics of the CH_4 flux and environmental variables for the entire study period are presented in Table 2. Soil–atmosphere exchanges of CH_4 were

Table 1 The NOAA AVHRR data used in this study

No.	Date	Julian Day	Time (GMT)	Satellite
1	24 May 1998	144	0856	NOAA-14
2	4 June 1998	155	0835	NOAA-14
3	17 June 1998	168	0744	NOAA-14
4	9 July 1998	190	0850	NOAA-14
5	25 July 1998	206	0914	NOAA-14
6	10 August 1998	222	0719	NOAA-14
7	23 May 1999	143	0818	NOAA-14
8	6 June 1999	157	0901	NOAA-14
9	22 June 1999	173	0923	NOAA-14
10	7 July 1999	188	0812	NOAA-14
11	30 July 1999	211	0855	NOAA-14
12	10 August 1999	222	0831	NOAA-14
13	19 May 2000	140	1002	NOAA-14
14	19 June 2000	171	0900	NOAA-14
15	25 June 2000	177	0939	NOAA-14
16	9 July 2000	191	1017	NOAA-16
17	31 July 2000	213	0914	NOAA-16
18	7 August 2000	220	0934	NOAA-16
19	26 May 2001	146	1042	NOAA-16
20	24 June 2001	175	1000	NOAA-16
21	6 July 2001	187	0712	NOAA-16
22	19 July 2001	200	0635	NOAA-16
23	3 August 2001	215	0721	NOAA-16
24	17 August 2001	229	0820	NOAA-16

Table 2 Summary statistics of CH_4 flux and environmental variables at the burned sites and the unburned site

Site [†]	<i>n</i>	Mean	Median	SD	Maximum	Minimum
CH_4 flux ($\text{mg C m}^{-2} \text{d}^{-1}$)						
W	17	11	6.5	8.5	30	2.6
S	14	3.4	0.24	5.1	16	−0.48
V	9	0.27	0.04	0.64	0.07	−0.41
U	43	−3.9	−3.9	1.0	−2.2	−6.8
Soil temperature at 10-cm depth (°C) [‡]						
S		12.5	11.5	1.9	15.5	10.5
V		11.2	11.0	0.4	11.8	10.9
U		13.4	13.9	1.2	14.9	10.3
Water depth (cm) [‡]						
W		10.5	10.0	5.5	20.5	3.0
Volumetric water content (%) [‡]						
S		57.9	50.5	14.8	82.9	43.0
V		25.4	25.0	3.5	32.8	21.7
U		11.2	10.8	2.5	19.7	7.4

[†]Burned sites: W, open water; S, bare soil; and V, vegetated. U, unburned site. [‡]Data collected concurrently with flux measurements. SD, standard deviation.

Table 3 Summary of air temperature data (°C) recorded automatically at 30-min intervals at the burned and unburned areas

	August 1999		June 2000		September 2000	
Site	Burned	Unburned	Burned	Unburned	Burned	Unburned
Mean	10.7	10.3	18.0	17.2	11.5	11.2
Maximum	21.3	21.0	32.1	30.8	26.5	26.1
Minimum	-1.0	-0.4	2.0	3.1	-0.4	0.3

always negative at site U, indicating that forest soil before the fire was a net consumer of CH₄. In contrast, CH₄ flux was always positive (net emission) at site W. Both emission and consumption of CH₄ were observed at sites S and V. The flux varied widely at site S, ranging from -0.48 to 16 mg C m⁻² d⁻¹, whereas it was close to zero at site V. Soil temperatures at a depth of 10 cm showed slight variations (approximately 10–15°C) at all sites. The VWC of the soils differed markedly among the sites: at site S the values were very high and variable, whereas the soil at site U experienced drought conditions (VWC < 20%) for the entire period.

Air temperature recorded at 30-min intervals was significantly different between the burned area and the unburned area for all the study periods (paired *t*-test, *P* < 0.0001; Table 3). Mean temperatures for each period at the burned area were 0.3–0.8°C higher than those at the unburned area. Temperature variations were also significantly different between the areas; the range of temperature variations was wider in the burned area than in the unburned area.

Table 4 shows the results of the transect measurement of the water depth of pools and VWC of bare and plant-covered soils at the burned area. Measurements were carried out during each period at 22 points every 5 m along the two 50-m transects. The number and depth of the pools were greatest in June 2000, immediately after the snowmelt. Soil moisture level in the bare soil was also highest in June 2000. The soil gradually dried out during summer, and herbaceous plants started to invade the measurement lines in the summer of 2000. The VWC of the plant-covered soils was approximately 25% in both September 2000 and August 2001.

Soil properties of the surface H horizon at the burned area and the unburned area are presented in Table 5. Significant differences in means between the unburned area and the burned area were found for the thickness of the H horizon, relative gas diffusion coefficient (D/D₀), pH and EC (two-sample *t* test, *P* < 0.05). The thickness of the H horizon decreased by approximately 60 cm after the fire, and D/D₀ also decreased considerably. The values of the pH and EC showed a slight increase and decrease, respectively.

Spearman's rank correlation coefficients (*r_s*) for the relationship between individual measurements of

Table 4 Water depth of pools and volumetric water content of bare and plant-covered soils measured along two 50-m transect lines in the burned area

Period	<i>n</i>	Mean	SD
Water depth (cm)			
August 1999	9	6.5	2.1
June 2000	14	13.8	4.5
September 2000	8	10.2	3.6
August 2001	3	4.0	1.4
Volumetric water content (%) of bare soil			
August 1999	13	48.0	7.2
June 2000	8	75.9	6.8
September 2000	8	53.5	5.1
August 2001	4	39.1	5.3
Volumetric water content (%) of plant-covered soil			
August 1999	0	–	–
June 2000	0	–	–
September 2000	6	25.2	3.8
August 2001	15	23.7	4.1

–, no data. SD, standard deviation.

Table 5 Soil properties of the surface organic layer (H horizon) at the burned and unburned areas

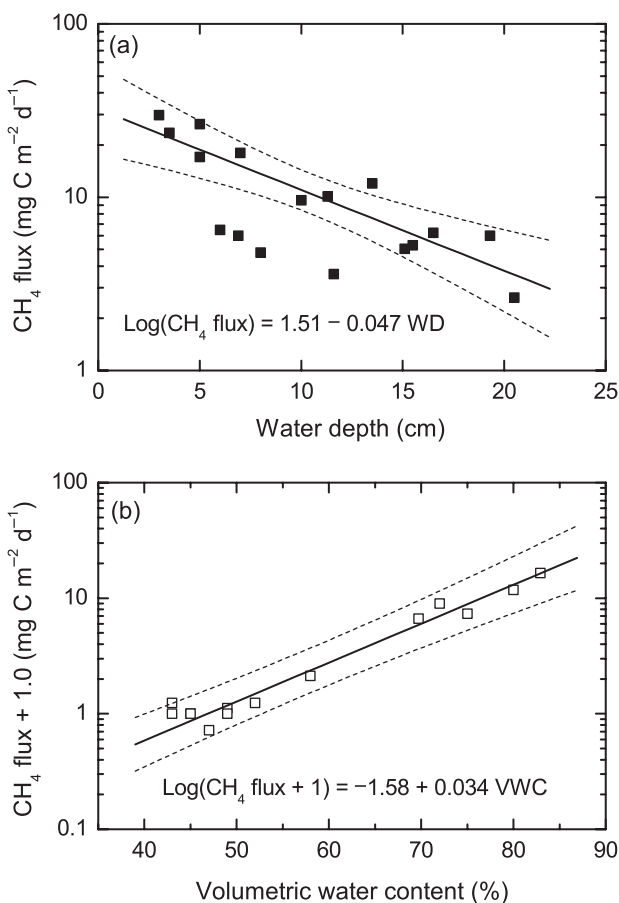
	Unburned	Burned
Thickness (m)	0.89 ± 0.06	0.31 ± 0.16
Bulk density (Mg m ⁻³)	0.19 ± 0.02	0.22 ± 0.02
Porosity (%)	90 ± 2	89 ± 3
D/D ₀	0.37 ± 0.05	0.010 ± 0.001
Total C (g g ⁻¹)	0.43 ± 0.02	0.45 ± 0.01
Total N (g g ⁻¹)	0.025 ± 0.002	0.025 ± 0.002
pH (H ₂ O)	4.9 ± 0.1	5.5 ± 0.2
EC	17 ± 1	12 ± 2

Values represent mean ± SD. D/D₀, relative gas diffusion coefficient in soil; EC, electrical conductivity.

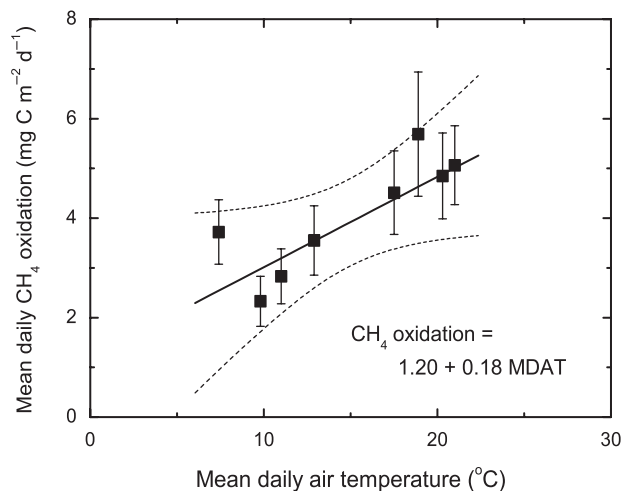
the CH₄ flux and environmental variables are given in Table 6. At site W, the CH₄ flux showed a negative correlation with water depth. The flux was positively correlated with air and soil temperatures and VWC at site S. There were no significant correlations at site V. Relationships of the CH₄ flux versus water depth and

Table 6 Spearman's rank correlation coefficients (r_s) for the relationships between CH_4 flux and environmental variables for the burned and unburned sites

	Site W	Site S	Site V	Site U
Air temperature	NS	0.74	NS	-0.40
Soil temperature at 10-cm depth	-	0.83	NS	NS
Water depth	-0.60	-	-	-
Volumetric water content	-	0.87	NS	NS

NS, not significant ($P > 0.05$). -, not determined.**Figure 2** Relationships between CH_4 flux and (a) water depth (WD) at the open water site and (b) volumetric water content (VWC) at the bare soil site. Lines indicate linear regression (solid line) and the 95% confidence bands (broken lines) for the regression.

VWC are presented in Fig. 2. CH_4 fluxes after logarithmic transformation were approximated by linear functions of the water depth and VWC. CH_4 flux in the unburned forest showed a negative correlation with air temperature (Table 6), indicating that CH_4 oxidation by soil increased as temperature increased. The relationship between mean

**Figure 3** Relationship between mean daily CH_4 oxidation rate and mean daily air temperature (MDAT) in the unburned birch forest site. Error bars indicate standard deviations. Lines indicate linear regression (solid line) and the 95% confidence bands (broken lines) for the regression.

daily CH_4 oxidation and mean daily air temperature is shown in Fig. 3. The oxidation rate was approximated by a linear function of air temperature.

Monitoring of surface environments by satellite images

Spectral mixture analysis of AVHRR imagery revealed the presence of temporal variations in the areal fractions covered with water, soil and vegetation in the study area (Fig. 4). As each plot represented a value on each day listed in Table 1, this figure showed 4-year variations in the surface coverage. In every year, the vegetation and water fractions showed seasonal variations: the vegetation fraction increased as the season progressed from spring to summer, whereas the water fraction decreased during the same period. Inter-annual variations were also evident (Fig. 4). The annual maximum value for the vegetation decreased from 80% in 1998 to 40% in 1999 owing to the fire, and then gradually increased in the following years. In contrast, the water fraction increased after the fire and decreased thereafter.

Areal flux estimates

The relationships between the areal fractions of open water deduced from AVHRR imagery and the water depth of the pools obtained from field measurements, and that between the fractions of water and VWC in bare soil are illustrated in Fig. 5a,b, respectively. Both water depth and VWC increased as the fractional value of open water increased, and both relationships were approximated by linear functions.

Figure 4 Temporal variations in areal fractions of vegetation, bare soil and open water in the study area. The dates of the records for the data plots are listed in Table 1.

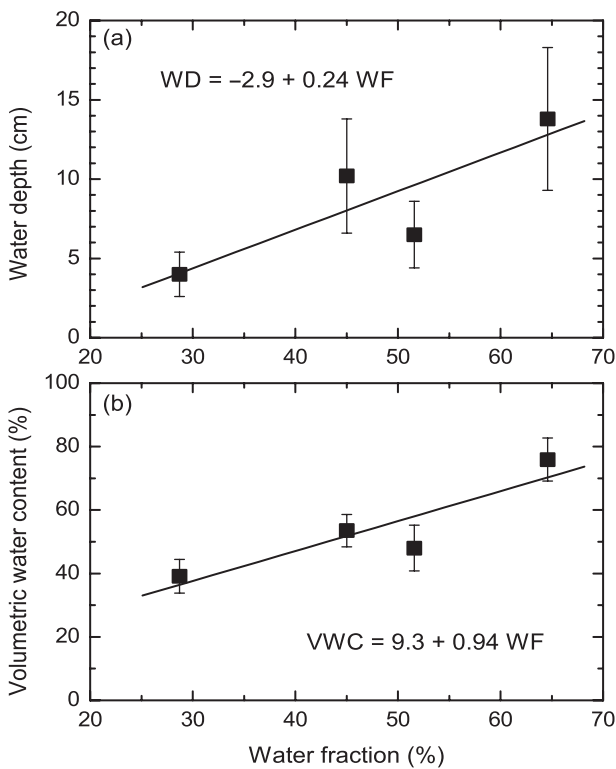
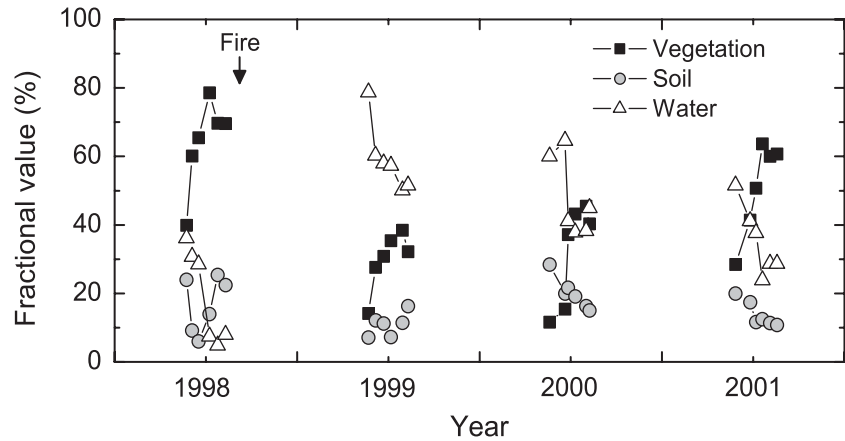


Figure 5 Relationships between areal fractions of open water derived from AVHRR imagery and (a) water depth (WD) of ponds and (b) volumetric water content (VWC) of bare soil. Points represent mean \pm standard deviations of the values for each measurement period (August 1999, June 2000, September 2000, August 2001). The data for the water fraction (WF) were obtained from four images acquired on 10 August 1999, 19 June 2000, 7 August 2000 and 3 August 2001.

We first estimated the values for water depth of pools and VWC of bare soils from the areal fraction of open water on days for which AVHRR images were available (Table 1) using the linear functions in Fig. 5. Both water depth and VWC decreased with time (Fig. 6a). Second,

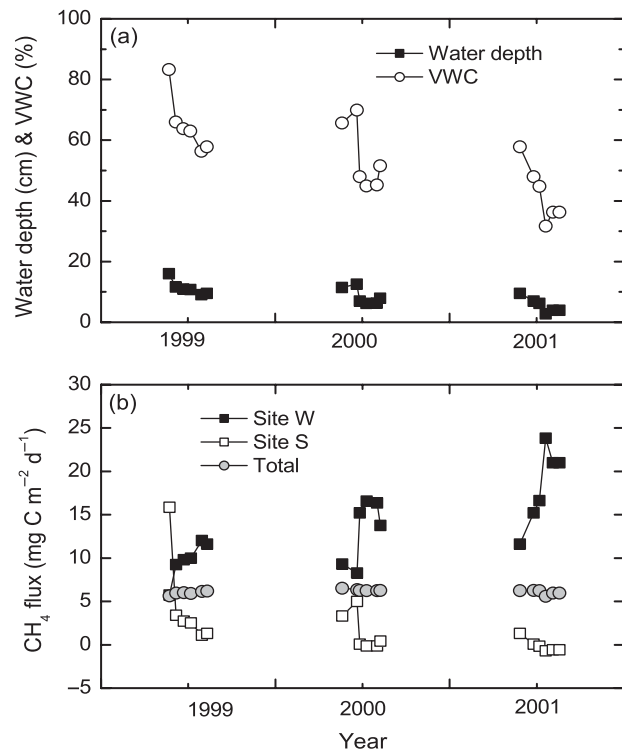


Figure 6 Estimates of temporal variations in (a) water depth of pools and volumetric water content (VWC) of bare soil after the fire, and (b) CH_4 fluxes from open water, bare soil and the entire burned area.

CH_4 fluxes at sites W and S were calculated by substituting the water depth and VWC given above into the linear functions shown in Fig. 2 (Fig. 6b). The CH_4 flux at site W increased from spring to summer and with time, whereas that at site S decreased over the years and became almost nil in 2001. The values of the CH_4 flux from the entire burned area were also computed by weighting the CH_4 flux at sites W and S with areal coverage of each site and by assuming that the CH_4 flux

at site V was always zero (Fig. 6b). The integrated CH₄ flux for the entire burned area showed slight variations within and between years and was approximately 5–6 mg C m⁻² day⁻¹ throughout the study period. Cumulative CH₄ emission rates during the 3-month period from June to August were estimated to be 0.39 g C m⁻² in 1999, 0.45 g C m⁻² in 2000 and 0.48 g C m⁻² in 2001.

Estimates of CH₄ oxidation in the unburned forest

Using the relationship between CH₄ oxidation and air temperature shown in Fig. 3, seasonal variations in CH₄ oxidation by unburned forest soils were estimated based on the surface air temperature data of the reanalysis dataset of the National Centers for Environmental Prediction (NCEP)/National Center for Atmospheric Research (NCAR) (Kalnay *et al.* 1996). Horizontal resolution of this dataset is latitude 2.5° × longitude 2.5°. Mean daily air temperature data at four grids encompassing Plotnikovo (55.0°N, 82.5°E; 55.0°N, 85.0°E; 57.5°N, 82.5°E; 57.5°N, 85.0°E) from 1998 to 2001 were downloaded from the website (<http://www.cdc.noaa.gov/cdc/reanalysis/>). We averaged temperatures for the four grids and substituted them into the equation in Fig. 3 to represent the temperature at Plotnikovo. Mean daily CH₄ oxidation rates were calculated, and the cumulative rates during the 3-month period from June to August were estimated to be 0.41 ± 0.11 g C m⁻² (mean ± 95% confidence interval) in 1998, 0.39 ± 0.11 g C m⁻² in 1999, 0.40 ± 0.10 g C m⁻² in 2000 and 0.38 ± 0.10 g C m⁻² in 2001.

DISCUSSION

Changes in the surface and soil environment after fire

Forest fires result in various changes in the physical, chemical and biological aspects of the surface environment. One important effect of fire is the destruction of the thick organic layer (Lucarotti 1980). Soil organic matter (SOM) consumption associated with wildfires has been studied using peat cores (Pitkänen *et al.* 1999; Turetsky and Wieder 2001), remote sensing analysis of postfire landscapes (Kasischke *et al.* 1995) and modeling (Zhuang *et al.* 2003). The amount of SOM lost during a single fire event has been estimated to be in the range of 2.5–3.0 kg C m⁻² in the boreal forest zone of Alaska (Kasischke *et al.* 1995), 2.5 kg C m⁻² in a peatland of eastern Finland (Pitkänen *et al.* 1999), and 2.2 kg C m⁻² in a peatland of western Canada (Turetsky and Wieder 2001). In the present study area, the thickness of the soil organic layer decreased by approximately 60 cm (Table 5). The amount of SOM, which

was calculated as the product of the thickness, bulk density and total carbon content of the H horizon, was estimated to be 73 kg C m⁻² and 31 kg C m⁻² at the unburned and burned areas, respectively. Assuming that the difference between these amounts corresponds to the loss of soil carbon during the fire, the area lost is 42 kg C m⁻². This value is considerably larger than the estimates given above, probably because the West Siberian plain is an extremely paludinous area characterized by very large carbon stocks from peat accumulation (Yefremov and Yefremova 2001). Moreover, forest soils in this plain are currently well drained and therefore burn easily.

Our results indicated that soil moisture drastically increased after the disturbance by fire. Desyatkin (1993) reported that, in eastern Siberia, thermokarsts (also called “*alas*”) were formed after severe forest fires by subsidence associated with permafrost thawing. The center of each *alas* was very wet, with the formation of ponds from 10 to 100 m in diameter, because water was supplied from the thawed permafrost. In contrast, a model simulation for black spruce forests in Alaska by Zhuang *et al.* (2003) suggested that the soil moisture of the burned stands would be much lower than that in the unburned stands during approximately the first 50-year period after a fire because of higher evaporation because more radiation reached the soil surface with a reduced canopy and greater drainage was associated with a thicker active layer after the fire. Lucarotti (1980) revealed that soil moisture was unaffected by fire in the *Picea* woodlands of eastern Canada. The present study area became wet after the fire, although it was not located in a permafrost region. A possible explanation is that in the West Siberian plain, surface soil moisture is determined by a balance between precipitation and evapotranspiration because the plain is extremely flat and it is difficult for water to flow laterally in and on the soil. The fire burned trees down and caused a remarkable decrease in transpiration by trees, hence the soil surface became wet.

In several studies, it was noted that soil temperatures at burned sites exceeded those at unburned sites throughout the growing season, and that fire effects may take several years to influence the soil temperature (Burke *et al.* 1997; O'Neill *et al.* 2003; Zhuang *et al.* 2003). Our results indicated that the mean air temperature for each study period in the burned area was 0.3–0.8°C higher than that in the unburned area, although we could not compare soil temperatures because the measurement times differed among the sites. The range of air temperatures was also greater in the burned area than at the unburned site. This higher and more variable temperature in the burned area was attributed to the loss of shading provided by the plant canopy (Fig. 1).

A decrease in latent heat led to a decrease in transpiration because of plant losses and also resulted in a higher air temperature in the burned area.

Control of CH₄ flux

Studies to date have indicated that factors controlling CH₄ emission from aquatic environments include soil moisture, soil temperature (e.g. Bellisario *et al.* 1999; Bubier *et al.* 1995; Christensen *et al.* 2003), substrate type (Crill *et al.* 1991; Svensson and Sundh 1992) and vegetation (Whiting and Chanton 1992). The most important among these factors is soil moisture because it affects the degree of anaerobicity in the soil profile. In many studies, the relationship between the CH₄ flux and water-table position was reported based on field measurements (Bubier 1995; Dise *et al.* 1993; Fiedler and Sommer 1999; Heikkinen *et al.* 2002; Liblik *et al.* 1997; Nykänen *et al.* 1998; Sebachner *et al.* 1986) and laboratory experiments (Daulat and Clymo 1998; MacDonald *et al.* 1998). Correlation of CH₄ flux and soil VWC has also been reported (Christensen *et al.* 1995; Morishita *et al.* 2003). These studies showed that a higher water-table level and water content led to a higher CH₄ emission. Our results (Fig. 2b) are in agreement with these previous findings.

When the water-table position was above the soil surface, however, the higher water level was associated with a lower CH₄ flux (Fig. 2a). Similar results have been reported previously (Juutinen *et al.* 2001; Kelley *et al.* 1995; Moore *et al.* 1990; Otter and Scholes 2000; Weyhenmeyer 1999). In general, methane-oxidizing bacteria consume a large fraction of the methane formed in methanogenic habitats before it reaches the atmosphere (Kiene 1991). The CH₄ oxidation potential is likely to increase by the longer water-column pathway, resulting in lower diffusive CH₄ fluxes across the sediment–water interface.

The CH₄ flux was almost zero at the plant-invaded site in the burned area and no relationship was found between the flux and environmental variables. Soil moisture at this site was relatively low (approximately 25%) and consistent over time. This low moisture level suppressed the CH₄ emission from the soil to the atmosphere. The limited variations in soil moisture and soil temperature during the study periods may account for the lack of a correlation.

Effects of soil temperature and moisture on the CH₄ oxidation rate at the unburned site were analyzed by Nakano *et al.* (2004). In the present study, a relationship between air temperature and CH₄ oxidation rate was demonstrated (Fig. 3) and temporal variations in CH₄ oxidation were estimated based on the air temperature data from the NCEP/NCAR reanalysis. Although few studies have focused on air temperature as a

controlling factor of CH₄ oxidation rate, we consider that air temperature is a good predictor of CH₄ flux because long-term and global-scale air temperature data, unlike soil temperatures, can be obtained by using datasets such as the NCEP/NCAR reanalysis dataset.

CH₄ fluxes before and after a fire

In the present study, temporal variations in CH₄ flux during the 3-year period after a fire were investigated based on field measurements and analysis of NOAA AVHRR imagery. In the study area, CH₄ oxidation in the dry soils of the unburned forest predominated and the cumulative CH₄ oxidation rate during the summer (from June to August), which was nearly consistent during the 4-year period (1998–2001), amounted to approximately 0.4 g C m⁻². The surface soil had become extremely wet after the fire and was covered with numerous small pools. As a result, the soil became a net emitter of CH₄ after the fire disturbance.

The soil gradually dried out and the pools became shallower and then disappeared from spring to summer and over the years. This moisture change in the burned area was apparent from both the field survey and the remote sensing analysis (Table 4, Fig. 6). CH₄ emission per unit area from the pools increased with time because CH₄ emission rate shows an inverse relationship with the depth of standing water. In contrast, the CH₄ flux from bare soils decreased as soil moisture decreased. Areal coverage of the pools and bare soil decreased over the years, as these cover types were replaced by herbaceous plants, whose CH₄ flux was almost zero. By weighting CH₄ fluxes at the pools, bare soils and invaded-plant sites according to their areal fractions and adding these values, we could estimate the soil–atmosphere CH₄ exchange in the entire burned area. The total CH₄ flux showed slight seasonal and interannual variations and amounted to approximately 5–6 mg C m⁻² day⁻¹ throughout the study period (Fig. 6). The cumulative CH₄ emission rates during the summer were calculated to be in the range of 0.39–0.48 g C m⁻², values within the same order of magnitude as the CH₄ oxidation rate in the unburned forest. However, because the soil is becoming still drier and the pools are disappearing, we predict that the burned soils will return to CH₄ consumer status in several years.

We are not aware of any published studies on the temporal changes in CH₄ flux after a fire to compare our results with. However, in a few studies, changes in the CH₄ exchange between soils and the atmosphere before and after forest fires were examined (Burke *et al.* 1997; Kim and Tanaka 2003). Burke *et al.* (1997) carried out CH₄ and CO₂ flux measurements at several sites with different fire histories (fires had occurred from 0 to 7 years before the measurements) in black

spruce stands and jack pine stands in Canada. All sites were net sinks of atmospheric CH₄, and the burned sites tended to be slightly stronger CH₄ sinks than the unburned controls after a few years of recovery. They observed a net CH₄ emission shortly after a fire (a few weeks) and suggested that this emission may have been of pyrogenic origin. Kim and Tanaka (2003) measured the fluxes of CO₂, CH₄ and N₂O before and after a prescribed fire in the boreal forest of interior Alaska. They reported that most CH₄ after the fire was oxidized by soil, but that some was emitted to the atmosphere above the burned stand. In addition, the CH₄ flux from soil increased from 7% to 142% after the fire, presumably because thawing of the frozen soil was accelerated by fire, and CH₄ may have been released from permafrost. Unlike these studies, our results indicated that the surface soil changed from a net CH₄ oxidizer to an emitter after the fire.

A key parameter of surface CH₄ exchange before and after the fire was soil moisture. Soil moisture in the burned area varied temporally and from site to site depending on the soil type, microrelief and vegetation cover. Remotely sensed satellite images are appropriate for monitoring surface moisture conditions over broad areas. Combining field measurements with the analysis of satellite images is a good strategy for evaluating the soil-atmosphere CH₄ exchange after fire disturbance.

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