Dynamics of methane fluxes from two peat bogs in the Ore Mountains, Czech Republic

L. Bohdálková[1,2], J. Čuřík[1], A.A. Kuběna[3], F. Bůzek[1]

[1]*Czech Geological Survey, Prague, Czech Republic*
[2]*Global Change Research Centre, Academy of Sciences of the Czech Republic, Brno, Czech Republic*
[3]*Faculty of Science, Charles University in Prague, Prague, Czech Republic*

ABSTRACT

Methane fluxes were studied at two high-elevation oligotrophic peat bogs in the Ore Mts., Czech Republic. The Bukova dolina Bog was drained 15 years ago and 2 years ago was partly restored, whereas the Brumiste Bog is an intact peatland. Draining led to a change of vegetation structure, dominated by *Molinia caerulea*, *Carex* sp., and forestation by Norway spruce. Methane fluxes were measured monthly from April to November 2011 using a closed chamber. Temperature and presence of *Carex* were significant controls on methane fluxes. Peat depth, water table and the presence of other plant species had no significant effect on CH_4 emissions. Methane emissions ranged from 9 to 2700 mg/m^2/day at the degraded and from 3 to 260 mg/m^2/day at the intact bog. In general, the degraded peat bog emitted three times more methane compared to the intact peat bog, likely due to vegetation changes after long-term artificial draining.

Keywords: restoration; peatland; climate change; *Sphagnum*; *Carex*

Globally, peatlands represent a reservoir of 455 Gt of carbon (Gorham 1991) that accumulated due to a relatively small imbalance between the production and decay of organic matter in anoxic conditions (Laiho 2006). The predicted climate change, leading to higher temperatures, lower water table levels (Roulet et al. 1992) and irregular precipitation with abrupt flooding (IPCC 2007) could influence carbon exchange between soil and atmosphere. These parameters play a key role in the production of greenhouse gases, mainly methane and CO_2, the final products of microbial decomposition of organic matter in peatlands (Whalen 2005). Wetlands could change from sink to source of carbon and by elevated emissions of methane and CO_2 start to accelerate climate warming (Laiho 2006 and references therein). An invasion of vascular plants into wetlands may follow (Minkkinen et al. 1999), in turn affecting emissions of CH_4. In wetlands, methane is produced under anaerobic conditions by methanogenic bacteria via aceticlastic methanogenesis and CO_2 reduction with hydrogen and subsequently transported by molecular diffusion, bubble ebullition and emergent plants (Whalen 2005). Part of the methane is oxidized by methanotrophic bacteria and the rest is released to the atmosphere. Vascular plants with aerenchymatic tissue could enhance methane emissions by conduction of methane from roots to the atmosphere, bypassing the oxidation zone (Whalen 2005). Spatial variation in methane fluxes is connected to temperature, water table level, pore solute quality, microtopography, organic matter quality, microbial characteristics and plant species composition (Whalen 2005, Lai 2009).

Land use management of wetlands could also affect carbon cycling and cooling effect of peatlands. In the Czech Republic, many peatlands were drained and forested in order to increase wood production. Long-term artificial drainage of

Supported by the Czech Geological Survey, Project No. 332600, CzechGlobe (CZ.1.05/1.1.00/02.0073) and by the Czech Science Foundation, Grant No. GAP504/12/1782.

peatlands substantially changes hydrogeochemical conditions (Prévost et al. 1999) and vegetation composition (Minkkinen et al. 1999). Changes in plant community influence substrate quality (Hahn-Schöfl et al. 2011), and microbial community structure (Straková et al. 2011). In addition, carbon exudates from plant roots could act as labile substrate for methanogenesis (Bellisario et al. 1999). The range of degradation due to drainage and forestry vary among the sites. Some peatlands underwent extensive changes that do not allow restoring them, other sites can be successfully revitalized. From the beginning of the twenty-first century, drained peatlands in the Czech Republic started to be intensively restored (Urbanová et al. 2012). We selected two nearby peatlands, one intact and one degraded due to 15-year long draining, with following two-year restoration to compare methane emissions. Little is known about *in situ* methane fluxes from Central European peat bogs and only a few studies compared fluxes between natural, degraded and restored bogs (Hahn-Schöfl et al. 2011, Urbanová et al. 2012). Our objective was (i) to evaluate seasonal variability in methane fluxes at the two contrasting peat bogs, and (ii) to study controls on methane fluxes.

MATERIAL AND METHODS

Our study sites are situated in the Ore Mts. on granitic bedrock, and, originally, covered by similar vegetation. Mean annual temperature is 4.5°C and mean annual precipitation is 1080 mm. The Brumiste Bog (BR; 50°22'18N, 12°45'26E, 930 m a.s.l.) is an intact peatland dominated by *Sphagnum fallax*, *Eriophorum vaginatum, Eriophorum angustifolium, Oxycoccus palustris, Carex limosa, Andromeda polifolia, Pinus × pseudopumilio, Carex nigra*. Lawn microhabitat is frequent, however hummock and hollows are also presented. The bog area is 17 ha, with one natural outflow.

The Bukova dolina Bog (BU; 50°24'28N, 12°36'40E, 980 m a.s.l.) was artificially drained 15 years ago, with discharge via two draining channels, 100 cm deep and up to 200 cm wide. Draining led to peatland degradation followed by vegetation changes with spreading of *Molinia caerulea*, *Carex* sp., and forestation by *Picea abies*. BU is dominated by *Sphagnum fallax*, *Molinia caerulea, Pinus × pseudopumilio, Picea abies, Eriophorum vaginatum, Carex canescens* and *Carex nigra*. Two years before sampling, BU was partially restored by damming of the channel, and the water table level

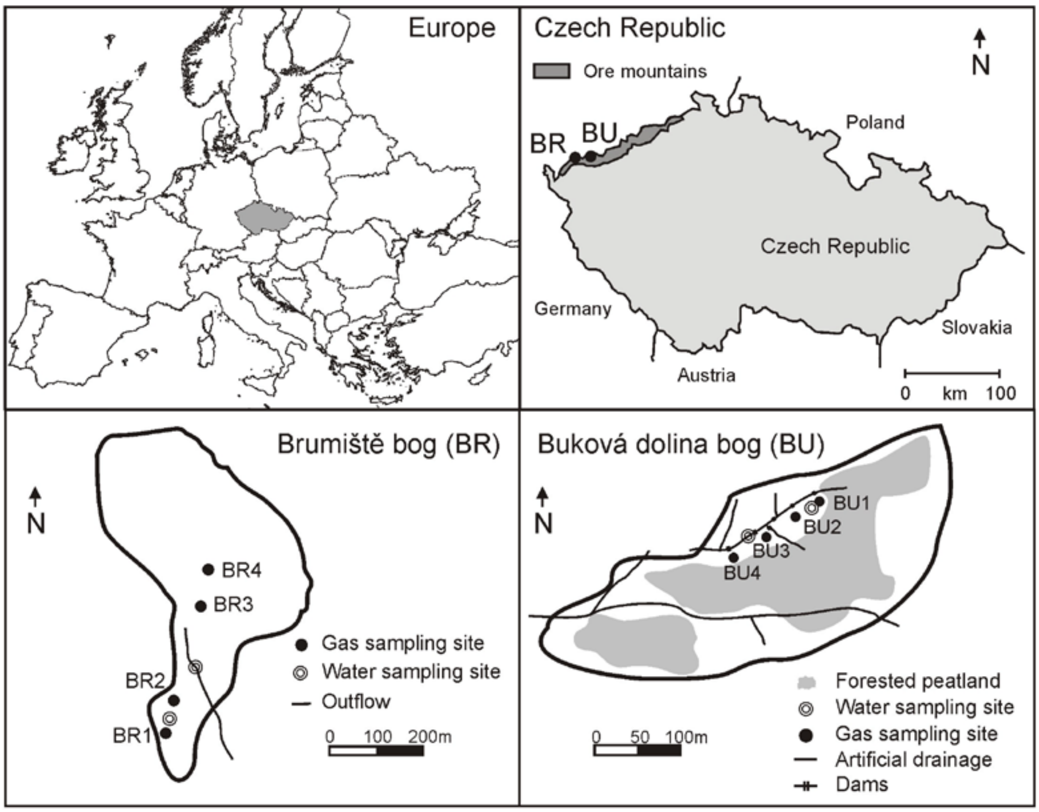

Figure 1. Study sites

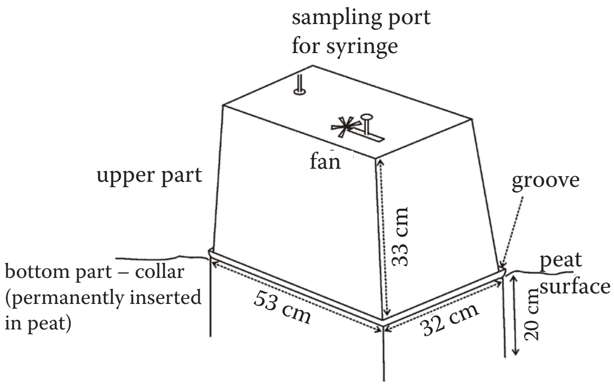

Figure 2. Close chamber diagram

has locally increased. Lawn microhabitat is less common, hummock and hollows prevail. The bog area is 7.2 ha. Both wetlands are oligotrophic, with relatively high atmospheric loads of pollutants, due to nearby industry (Fottová 2003).

Gas samples were collected *in situ* each month between April and November 2011, using static close chambers. We defined a transect at each site (Figure 1), 102 m long at BU and 345 m long at BR. Along each transect four chambers were inserted in lawn microhabitats (Figure 1, BR1-BR4 at BR and BU1-BU4 at BU) that prevail at BR. At BU, distances between close chambers BU1, BU2, BU3 and BU4 were 33, 35 and 34 m, respectively. They were from 7 m to 11 m far from the restored channel. At BR, distances between BR1, BR2, BR3 and BR4 were 101, 180 and 64 m, respectively. Each PVC chamber consisted of two parts (Figure 2). The bottom part (collar with surrounding groove) was permanently inserted 20 cm into the peat. Before each sampling, the upper part (pot-shaped vessel of total volume of 53 L) was placed into the groove filled with water. Gases emitted from the peat surface were trapped into the chamber and mixed by a fan (Figure 2). One gas sample were taken after 0, 1, 1.5 and 2 h from each close chamber by a syringe and transferred to Tedlar bags. Methane concentrations were determined by gas chromatography (CHROM 5, Prague, Czech Republic) using a flame ionization detector. Fluxes were calculated using the linear portion of the gas concentration change over the 2-h time step (mean R^2 = 0.89). Rare ebullition events were deleted from the data (Yrjälä et al. 2011). Water table level (WTL) and temperature of pore water from the surface was recorded (Figures 3b,d,f). Three replicates of precipitation water at each site were collected together with one replicate of water from the stream at BR and from the channel at BU (Figure 1). Pore water at each site was sampled by piezometers between BR1 and BR2 at BR, and between BU1 and BU2 at BU, one replicate per site each month at the depth of 0, 25 and 50 cm under WTL (Figure 1).

Statistical analysis was performed using the SPSS (Chicago, USA). Non-homogeneous flux data were log-transformed to achieve normality of errors. Plant species presented inside each collar (Table 1) were evaluated as binary variables (present vs. absent) and their control on methane fluxes was evaluated using a regression tree to exclude insignificant plant species. The *Carex* group was shown as a significant predictor of CH_4 flux, and therefore was used as a predictor in a LME

Table 1. Peat depth, mean water table level (WTL) ± standard deviation (SD), wetness and vegetation cover at the sampling sites

Sampling site	Peat depth (cm)	Mean WTL ± SD (cm)	Wetness	*Sphagnum* type	Other vascular plants
BU1	120	0 ± 4	wet	*fallax*	*Carex nigra*
BU2	150	–7 ± 5	dry	*fallax*	*Eriophorum vaginatum, Molinia caerulea, Oxycoccus palustris*
BU3	125	–15 ± 10	dry	*fallax*	*Polytrichum commune, Carex canescens*
BU4	115	–6 ± 10	wet	*cuspidatum*	*Eriophorum vaginatum*
BR1	175	–11 ± 12	dry	*fallax*	*Polytrichum commune, Eriophorum angustifolium*
BR2	115	–7 ± 10	dry	*fallax*	*Polytrichum commune, Carex nigra, Oxycoccus palustris*
BR3	> 200	–5 ± 4	dry	*papillosum*	*Eriophorum vaginatum, Andromeda polifolia*
BR4	> 200	2 ± 2	wet	*fallax*	*Eriophorum vaginatum*

BU – Bukova dolina Bog; BR – Brumiste Bog

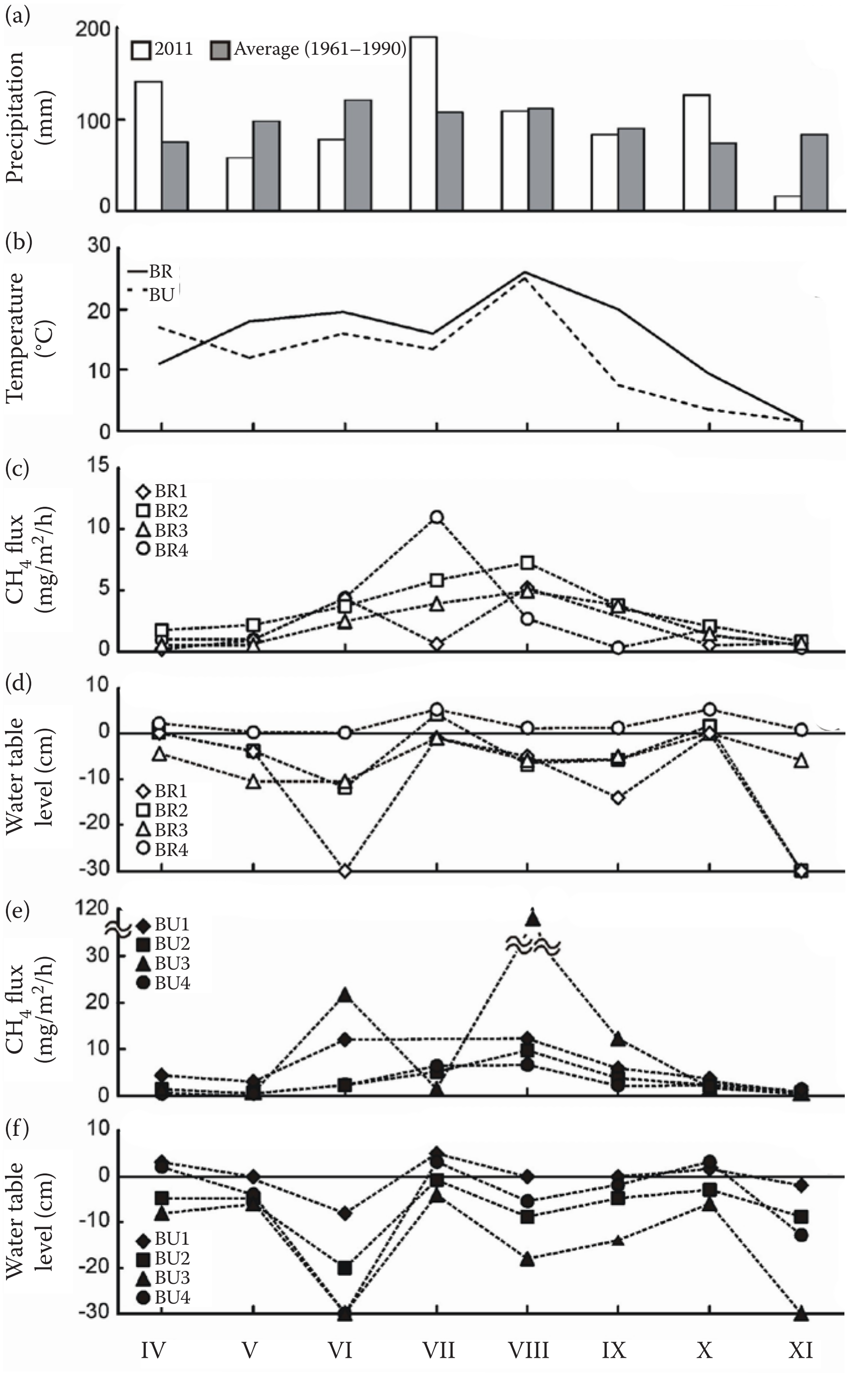

Figure 3. Mean precipitation at Bukova dolina Bog (BU) and Brumiste Bog (BR) in 2011 compared to long-term average (a), temperature (b), methane fluxes at BR (c), water table level at BR (d), methane fluxes at BU (e) and water table level at BU (f)

model. Repeated measurement of fluxes at the same sites resulted in non-independent data. A linear mixed model (LME) was used to avoid pseudoreplication. Logarithm of methane flux was the response variable, explanatory terms include the continuous variables temperature, water table level and peat depth. The factorial response variables represented site effect (BR vs. BU), *Carex* sp. (present vs. absent) and wetness (wet vs. dry; predictor 'wetness' was classified by vegetation occurrence inside of each collar, typical for drier or wetter environment and water table fluctuation during the year at each of the eight sites (Table 1, Figures 3d,f). Finally, GLM-RM (General linear model – repeated measures) enables to describe percentage of variability by eta-squared values and

Table 2. Pore water quality and the results of Mann-Whitney test (comparison of water-quality between sites)

		pH	TOC (mg/L)	Mg^{2+}	K^+	Fe	As	NH_4^+	$(NO_3)^-$	$(SO_4)^{2-}$
				(μg/L)				(mg/L)		
	n	9	9	9	9	9	8	8	8	8
BR	mean	4.2	12.3	227	741	591	1.7	0.04	0.3	5.9
	SD	0.2	5.2	158	774	849	1.3	0.03	0.3	6.2
	n	8	8	8	8	8	7	7	7	7
BU	mean	3.9	50.6	698	196	438	4.6	0.03	0.1	0.5
	SD	0.1	12.0	179	345	225	1.6	0.02	0.0	0.3
Mann-Whitney test		0.001*	0.001*	0.001*	0.101	0.564	0.005*	0.157	0.449	0.006*

BU – Bukova dolina Bog; BR – Brumiste Bog; *n* – number of samples; SD – standard deviation; TOC – total organic carbon; $^*P < 0.05$

assess Cohen's conventions for small, medium, and large effects.

Pore water quality data (Table 2) in individual bogs were compared using Mann-Whitney U test.

RESULTS AND DISCUSSION

Fluxes of methane were calculated for each site (Figures 3c,e). Temperature ($P < 0.001$) and *Carex* occurrence ($P = 0.003$) controlled significantly methane fluxes (Figures 4a,b,c), whereas water table level ($P = 0.7$), peat depth ($P = 0.318$) and wetness within the lawn microhabitat ($P = 0.7$) were insignificant. Methane fluxes at BU bog and BR bog were different ($P = 0.003$), BU emitted 3 times more methane than BR (Table 3). Temperature explained 32% of the variability in methane fluxes, showing a large effect size. In contrast, *Carex* occurrence and site effect explained 12% and 11% of variability in methane fluxes, respectively (medium effect size). The classification tree for evaluating vegetation control on methane fluxes is in Figure 4a. *Carex canescens* enhanced methane fluxes six times, *Carex nigra* about twice. Possible interaction between temperature and *Carex* presence that would be seen in Figure 4c from different slopes of linear regression, was not statistically significant (data not shown). Generally, for evaluating the interactions between variables more data are

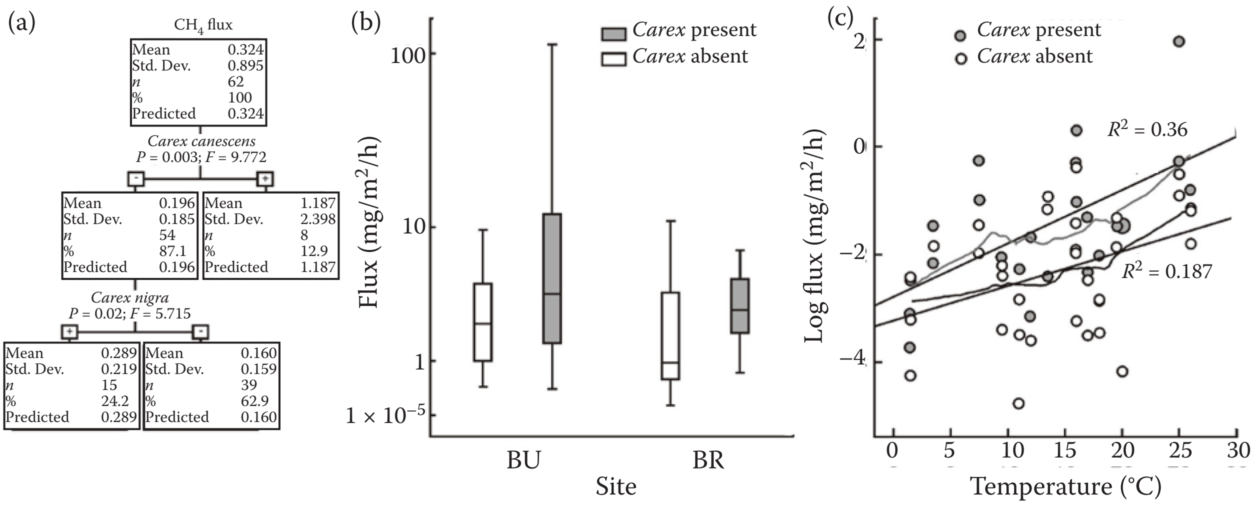

Figure 4. Classification tree evaluating vegetation control on methane fluxes (*n* = number of samples) (a), box plots of methane fluxes at both sites (b), temperature dependence of methane fluxes, linear regression and loess curves (c); BU – Bukova dolina Bog; BR – Brumiste Bog

Table 3. Logarithm of methane fluxes of Brumiste Bog and Bukova dolina Bog. Controls on methane fluxes were evaluated using linear mixed model (LME) model

Response	Log (CH_4 flux)		Flux ratio	S.E.	95% confidence interval	
	F	significance				
Temperature	44.6	< 0.001	2.4[a]	0.01[a]	1.9[a]	3.2[a]
Carex occurance	10.8	0.003	2.3[b]	0.26[b]	1.4[b]	4.0[b]
Peatland	10.4	0.003	2.5[c]	0.29[c]	1.4[c]	4.6[c]
Wet/dry conditions	0.16	0.70				
Water table level	0.17	0.69				
Peat depth	1.03	0.32				

[a]methane flux increase under 10°C temperature increase; [b]*Carex*-present/*Carex*-absent methane flux ratio; [c]BU/BR methane flux ratio; S.E. – standard error

needed. Within-site variability of methane fluxes and water table fluctuation during the season was higher at the degraded BU than at the intact BR. At BU, methane flux varied from 9 to 2700 mg/m^2/day with median of 57 mg/m^2/day. At BR, methane flux varied from 3 to 260 mg/m^2/day with median of 42 mg/m^2/day. Saarnio et al. (2007) reviewed published average annual net methane fluxes in mires of southern Finland from 4 to 150 mg CH_4 mg/m^2/day. Methane fluxes from bogs and poor fens in Canada and USA range from 0.27 to 423 mg/m^2/day (Lai 2009). Methane emissions at the studied sites were comparable with other northern peatlands. In the Bohemian Forest, Czech Republic, Urbanová et al. (2012) measured methane fluxes in bogs and fens with changed hydrological regime and described average seasonal fluxes of 0–83 mg/m^2/day CH_4.

The *Carex* vascular plant group and cotton-grass (*Eriophorum vaginatum)* were found to be predictors of methane fluxes in previous studies (Bellisario et al. 1999, Tuittila et al. 2000, Whalen 2005). Yavitt et al. (2000) found correlation between methane production and fresh lignin derived from *Carex* sedges.

Gas sampling sites at the degraded BU with the lack of *Carex* vascular plants exhibited also higher methane fluxes compared to the intact BR (Table 1, Figures 3c,e). Site-specific predictor of methane fluxes could be connected to land use changes at BU. In general, drained peatlands usually emitted less methane in contrast to intact peatlands (Urbanová et al. 2011). The reason is thinning of the substrate zone hosting methanogens and aeration of the upper part of the peat. On the other hand, increased water table level after revitalization could be the reason

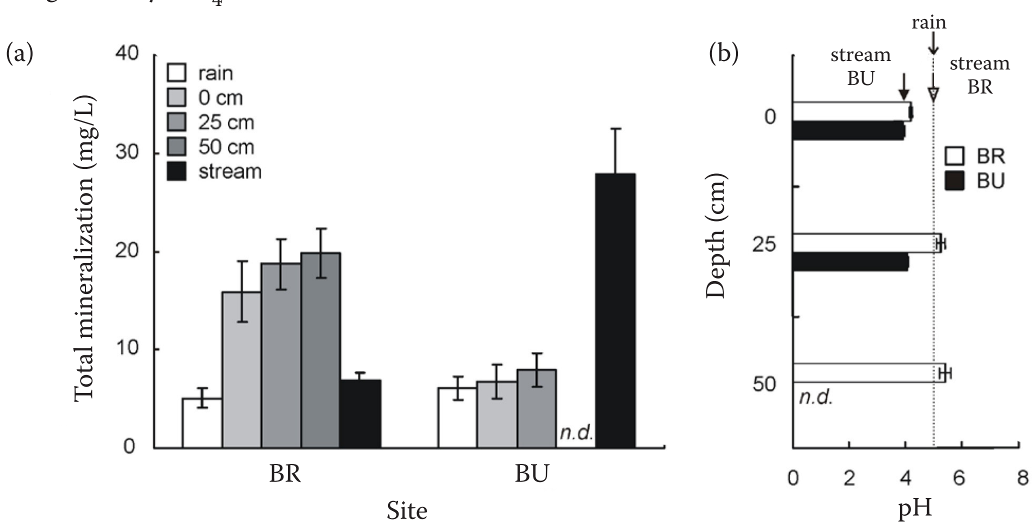

Figure 5. Total mineralization – inorganic compounds (a) and pH values (b) of rain, pore water at the depth of 0, 25 and 50 cm under water table level, stream water from Brumiste Bog (BR) and water from the draining channel at Bukova dolina Bog (BU)

of higher methane emissions (Tuittila et al. 2000). After blocking ditches, the water regime stabilizes almost immediately, but it requires years to achieve ecological stabilization (Price et al. 2003). Urbanová et al. (2012) neither found vegetation changes, nor methane flux increase during the first year after water regime restoration. Although methane emissions may remain low over long periods of time after rewetting, we cannot exclude possible effect of restored hydrological regime on methane fluxes. We observed the highest methane emissions at a site BU3 with the lowest water table level (Table 1, Figures 3e,f). We suggest that, more likely, vegetation changes followed by a change in litter quality and stimulation of microbial activities via root exudates after drainage influenced methane emissions. Decomposition of *Carex* species and other spreading vascular plants is relatively faster compared to moss litter (Laiho 2006). This easily degradable litter type could explain higher concentration of TOC in pore water at BU (Table 2) and partially influence methane fluxes at the degraded site. However, to verify this explanation for enhanced CH_4 fluxes, further study is needed.

Pore water quality could also influence methane fluxes. Both methanogens and methanotrophs showed optimum pH at least 2 units higher than *in situ* values (pH range of pore water measured at the sites was 3.7–6.1). Higher methane emissions would be expected at the intact BR with higher pH value (Table 2, Figure 5b). Analyses of bulk precipitation, stream and pore water in different peat depth were performed to clarify possible effect of different nutrient status at the sites on CH_4 emissions. Figure 5 show that pore water at BU is not influenced by groundwater at least to a depth of 25 cm under the water table where methanogens are active. Higher mineralization of pore water and slightly increasing pH values with the depth at BR indicate that our sampling site in the lagg area was partially influenced by groundwater. However, samples from the stream at BR in deeper part of the peat bog confirmed that both sites are mostly rain-fed. Surface water from 1.5 m deep draining channel at BU is mixed with groundwater but pore water data show that this could not influence the fluxes of emitted gases.

Thermodynamically, a threefold amount of sulfate in pore water could cause that sulfate reducers outcompete methanogens (Vile et al. 2003) and thus reduce methane emissions at BR. Similarly, higher amounts of magnesium at BU could support methane fluxes, because Mg^{2+} is one of the elements needed by methanogens (Sprott and Jarell 1981).

We conclude that the enhanced methane emissions at BU remain unexplained. Different nutrient status at the sites was excluded from possible flux controls. Sulfate reducers at the intact site BR could outcompete methanogens and thus reduce methane emissions. Most likely, an invasion of easily degradable vascular plants contributed to the enhanced methane fluxes. Composition of peatland plant communities has been considered as one of the most important drivers also in previous studies.

Acknowledgements

We thank V. Melichar, J. Váňa, V. Chlupáčková, P. Bohdálek, O. Myška and M. Novák for assistance.

REFERENCES

Bellisario L.M., Bubier J.L., Moore T.R. (1999): Controls on CH_4 emissions from a northern peatland. Global Biogeochemical Cycles, *13*: 81–91.

Fottová D. (2003): Trends in sulphur and nitrogen deposition fluxes in the Geomon network, Czech Republic, between 1994 and 2000. Water, Air, and Soil Pollution, *150*: 73–87.

Gorham E. (1991): Northern peatlands: role in the carbon cycle and probable responses to climatic warming. Ecological Applications, *1*: 182–195.

Hahn-Schöfl M., Zak D., Minke M., Gelbrecht J., Augustin J., Freibauer A. (2011): Organic sediment formed during inundation of a degraded fen grassland emits late fluxes of CH_4 and CO_2. Biogeosciences Discussions, *7*: 9273–9303.

IPCC (2007): Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Brussels.

Lai D.Y.F. (2009): Methane dynamics in northern peatlands: A review. Pedosphere, *19*: 409–421.

Laiho R. (2006): Decomposition in peatlands: Reconciling seemingly contrasting results on the impacts of lowered water levels. Soil Biology and Biochemistry, *38*: 2011–2024.

Minkkinen K., Vasander H., Jauhiainen S., Karsisto M., Laine J. (1999): Post-drainage changes in vegetation composition and carbon balance in Lakkasuo mire, Central Finland. Plant and Soil, *207*: 107–120.

Prévost M., Plamondón A.P., Belleau P. (1999): Effects of drainage of a forested peatland on water quality and quantity. Journal of Hydrology, *214*: 130–143.

Price J.S., Heathwaite A.L., Baird A.J. (2003): Hydrological processes in abandoned and restored peatlands: An overview of management approaches. Wetlands Ecology and Management, *11*: 65–83.

Roulet N., Moore T., Bubier J., Lafleur P. (1992): Northern fens: Methane flux and climatic change. Tellus, *44B*: 100–105.

Saarnio S., Morero M., Shurpali N.J., Tuittila E.-S., Mäkilä M., Alm J. (2007): Annual CO_2 and CH_4 fluxes of pristine boreal mires as a background for the lifecycle analyses of peat energy. Boreal Environmental Research, *12*: 101–113.

Sprott G.D., Jarrell K.F. (1981): K^+, Na^+, and Mg^{2+} content and permeability of *Methanospirillum hungatei* and *Methanobacterium thermoautotrophicum.* Canadian Journal of Microbiology, *27*: 444–451.

Straková P., Niemi R.M., Freeman C., Peltoniemi K., Toberman H., Heiskanen I., Fritze H., Laiho R. (2011): Litter type affects the activity of aerobic decomposers in a boreal peatland more than site nutrient and water table regimes. Biogeosciences, *8*: 2741–2755.

Tuitilla E.-S., Komulainen V.-M., Vasander H., Nykänen H., Martikainen P.J., Laine J. (2000): Methane dynamics of a restored cut-away peatland. Global Change Biology, *6*: 569–581.

Urbanová Z., Picek T., Bárta J. (2011): Effect of peat re-wetting on carbon and nutrient fluxes, greenhouse gas production and diversity of methanogenic archaeal community. Ecological Engineering, *37*: 1017–1026.

Urbanová Z., Picek T., Hájek T., Bufková I., Tuittila E.-S. (2012): Vegetation and carbon gas dynamics under a changed hydrological regime in central European peatlands. Plant Ecology and Diversity. DOI: 10.1080/17550874.2012.688069 (In press)

Vile M.A., Bridgham S.D., Wieder R.K., Novák M. (2003): Atmospheric sulfur deposition alters pathways of gaseous carbon production in peatlands. Global Biogeochemical Cycles, *17*: 1058–1064.

Whalen S.C. (2005): Biogeochemistry of methane exchange between natural wetlands and the atmosphere. Environmental Engineering Science, *22*: 73–94.

Yavitt J.B., Williams C.J., Wieder R.K. (2000): Controls on microbial production of methane and carbon dioxide in three *Sphagnum*-dominated peatland ecosystems as revealed by a reciprocal field peat transplant experiment. Geomicrobiology Journal, *17*: 61–88.

Yrjälä K., Tuomivirta T., Juottonen H., Putkinen A., Lappi K., Tuitilla E.-S., Penttila T., Minkkinen K., Laine J., Peltoniemi K., Fritze H. (2011): CH_4 production and oxidation processes in a boreal fen ecosystem after long-term water table drawdown. Global Change Biology, 17: 1311–1320.

Received on May 16, 2012
Accepted on November 26, 2012

Corresponding author:

Mgr. Leona Bohdálková, Ph.D., Česká geologická služba, Praha, Česká republika
phone/fax: + 420 251 085 260, e-mail: leona.bohdalkova@geology.cz