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THE EFFECTS OF LITTER LAYER AND SOIL PROPERTIES ON THE SOIL-ATMOSPHERE FLUXES OF GREENHOUSE GASES IN KARST FOREST, SOUTHWEST CHINA

ABSTRACT: Temporal variation is a major source of the uncertainty in estimating the fluxes of the greenhouse gases (GHGs) in terrestrial ecosystems, and the GHG fluxes and its affecting factors in the karst region of southwest China remains weakly understood. Using the static chamber technique and gas chromatography method, the CO2, CH4 and N2O fluxes were carried out between 9 and 11 a.m. at 15 day intervals from June 2008 to May 2009 in a Pinus massoniana forest. Two treatments were chosen for this study: undisturbed (soil with litter layer) and disturbed (surface litter removal). Both treatments were found to be the net source of atmospheric CO2 and N2O, but a sink of atmospheric CH4. The seasonality of soil CO2 emission coincided with the seasonal climate pattern, with high CO2 emission rates in the hot-wet season and low rates in the cool-dry season. In contrast, seasonal patterns of CH4 and N2O fluxes were not clear, although higher CH4 uptake rates were often observed in autumn and higher N2O emission rates were often observed in spring (dry-wet season transition). The litter was active in GHG fluxes, and removal of the litter layer reduced soil CO2 emission (17%) and increased CH4 uptake (24%) whereas N2O fluxes were not affected distinctly in the pine forest, indicating that litter layer had an important effect on C exchanges. In the pine forest, soil CO2 emissions and CH4 uptakes correlated significantly with soil temperature (r² = 0.87, P < 0.01; r² = 0.34, P < 0.05, respectively), but had no significant relationship with soil moisture. And there was a significant correlation between CH4 flux and NH4+ -N (r² = 0.39, P < 0.05) and soil inorganic N (r² = 0.48, P < 0.05), but no significant correlation was found between CH4 flux and NO3- -N. Moreover, we found a significant negative logarithmic correlation between N2O flux and soil NO3- -N concentration (r² = 0.41, P < 0.05), and the relationship between CO2 emission and soil inorganic N content (r² = 0.35, P < 0.05). These results suggested that soil temperature and mineral N dynamics largely affected the temporal GHG exchanges between forest soil and atmosphere.

KEY WORDS: NO3- -N, NH4+ -N, seasonal variation, litter layer, karst area, greenhouse gases fluxes

1. INTRODUCTION

Since the 19th century, global surface temperature has been increasing, partly due to increasing concentration of Greenhouse gases (GHGs) in the atmosphere. CO2 is the most important individual GHG, and its efflux from soil to the atmosphere, generally defined as soil respiration, is the key part of carbon cycle in terrestrial ecosystems (Raich and Schlesinger 1992). Moreover, CH4 and N2O are also important GHGs whose concentrations in the atmosphere have increased
due to human activities. Although the absolute quantities of gases (CH$_4$ and N$_2$O) are small compared with that of CO$_2$, they are 25 and 298 times more effective per molecule as a GHG than CO$_2$ in a period of 100 years, respectively (IPCC 2007). In addition, the CH$_4$ and N$_2$O increases account for 20 and 6% of the increased greenhouse warming potential (GWP) of the atmosphere, respectively (Ramswamy et al. 2001). Therefore, a complete inventory of all three key GHG fluxes is necessary to determine net GHG budgets and for estimating the net benefit in GWP in forests for the purpose of C and N cycling.

GHGs fluxes of forest soils are all produced or consumed as a result of microbiological processes in the soil, and the exchanges between the soil and the atmosphere depends heavily on many environmental factors, such as soil temperature, soil moisture, soil pH and texture, the distribution of O$_2$, root activities, decomposition of organic matter as well as C and N substrate availability (Konda et al. 2008, Nkongolo et al. 2010). Whereas, CO$_2$ efflux is generally tightly correlated to soil temperature, C and N substrate availabilities (Kicklighter et al. 1994, Vose and Bolstad 2006), CH$_4$ and N$_2$O fluxes are more dependent on a number of parameters that may affect soil aeration and gas diffusivity, including soil moisture, bulk density and litter layer characteristics, besides substrate control (Dong et al. 1998, Smith et al. 2003, Mosier et al. 2004). Therefore, studies about GHG sources, sinks and their relationships with environmental factors have always been the focus of modeling or predicting C and N exchanges between the atmosphere and forest soils.

Previous study showed that litter layer is an integrated response between biological processes and the influence of environmental factors (Pedersen and Bille-Hansen 1999). Therefore, litter layer is important to understand nutrient and organic matter dynamics as well as an indicator of trace gas fluxes at the forest soil (Yan et al. 2005). Generally, the removal of litter causes a reduction in CO$_2$ emission and an increase in CH$_4$ uptake (Dong et al. 1998, Rey et al. 2002, Li et al. 2004, Liu et al. 2008). However, Borken and Beese (2006) found that litter layer removal caused a significant decrease in CH$_4$ uptake in mixed spruce and beech forests but had no effect on CH$_4$ uptake in a spruce forest. And no distinct changes in CH$_4$ fluxes were also found after the litter layer removal in a hilly area of south China (Liu et al. 2008). They thought that the majority of CH$_4$ oxidation occurred in the mineral soil rather than in the surface litter in the pine plantation and orchard. In N$_2$O fluxes, no distinct changes were found after the litter layer was removed in three subtropical forest ecosystems in southern China (Tang et al. 2006). On the other hand, some studies showed that litter removal reduced N$_2$O effluxes (Xiao et al. 2004, Liu et al. 2008). So far, different researches got different or even contrary conclusions, and the mechanisms of litter layer on soil GHG fluxes have not been extensively studied.

China is one of the countries featuring extensive karst areas, with nearly 2.0 million km$^2$, about 23% of its land area, covered by open and covered karst rocks, which are mainly concentrated in the southwest, including Guizhou, Yunnan, and Guangxi provinces, etc (Zhang et al. 2006). On the other hand, there is a large human population in southwest karst regions, and the population density was 33.3% higher than the national average. For hundreds of years, forests in karst regions have been impacted by human activities, including timber and intensive biofuel harvesting (Brown et al. 1995). It is a universal phenomenon to harvest layer litter for fuel sources in the karst region because of relatively backward economy and growing population. However, to our knowledge, the impact of litter layer exclusion on exchanges of GHG fluxes from such a typical karst forest is poorly understood, especially in southwest China. Due to the specificity of karst environment, the removal of litter may have different effects on GHG exchanges. Therefore, it is necessary to carefully study GHG exchanges before and after litter exclusion from karst forests.

In order to improve the database of GHG exchanges from a Pinus massoniana forest in the typical karst areas of southwest China, we carried out this study. The objectives of this study were to: (1) estimate the contribution of litter to GHG fluxes; and (2) determine the soil environmental factors controlling GHG exchanges between soil and atmosphere.
2. MATERIALS AND METHODS

2.1. Site description

The research site is located in a Pinus mas-soniana forest (26°32’N and 106°46’E) at altitude of 1200 m in Longdongbao areas, approximately 10 km north of Guiyang city, Guizhou province, which is located in the heart of the southwest karst, China (Fig. 1). The study region is a humid subtropical climate, with fairly mild winters and warm summers. The mean annual temperature is 14.8°C with the lowest mean temperature in January at 3.9°C and the highest mean temperature in July at 23.2°C. The mean annual precipitation is about 1118 mm with the lowest mean precipitation (21.4 mm) in December and the highest mean precipitation (203.1 mm) in July, and most rainfall (more than 86%) is in the wet season (from April to October). Our field campaign was conducted from June 2008 to May 2009, spanning a transition from wet seasons into dry seasons. Seasonal precipitation, soil temperature (5 cm), WFPS (0–10 cm) between undisturbed and disturbed pine forests were presented in Fig. 2.

The stand density (number of pine trees above breast height) of the forest is 2000 trees ha⁻¹. The average DBH (diameter at breast height) is 15 cm and the average tree height is 13 m. The forest floor is covered by shrubby vegetations including Camellia oleifera and Itea yunnanensis, etc., and some low-rise herbs (Artemisia sp.) under the shrub layer. In our study site, the forest soil is a limestone soil developed from limestone, with a pH of about 6.0. Some soil properties are listed in Table 1.

2.2. Experimental design

To determine the effect of litter on CO₂, CH₄ and N₂O fluxes, two treatments were applied to the field: (1) undisturbed: litter (litter layer was intact with thickness of 2–3 cm). (2) disturbed: without litter, chambers were placed in the plots from which the litter layer was removed in advance and keep clear of fresh litter. In each treatment, three replicate polyvinyl chambers were randomly installed on the surface soil. The CO₂, CH₄ and N₂O fluxes were carried out between 9 and 11 a.m. at 15 day intervals, with a total of twenty-four times in a year.

The gas fluxes were measured using a static chamber method, the chamber assembly consisted of a permanently installed stainless steel base (30 cm×30 cm×10 cm) with a U-shaped groove at the top edge to hold a mobile polyvinyl chloride (PVC) (30 cm×30 cm×50 cm) that was equipped with an electronic fan. Once the cover was placed onto the base, the groove was filled with water, which acted as an air seal. At 0, 5, 10, 15 and 20 minutes after the groove was sealed, a 30-mL air sample inside the chamber was taken into a 15-mL glass bottle vacuum-sealed with a butyl rubber stopper and a plastic cap. We brought the glass vials back to laboratory and analyzed completely the gas concentrations within three days.

Fig. 1. The study area and its adjacent districts in the city of Guiyang, Guizhou province.
2.3. Gas flux rate determinations

The gas concentrations were analyzed by HP 6890 gas chromatography with a flame-ionization detector (FID) for CH$_4$ and an electron capture detector (ECD) for N$_2$O. CO$_2$ was reduced by a nickel catalyst and thus analyzed by FID. The gas chromatography configurations for analyzing concentrations of GHG were according to the method of Wang and Wang (2003). And the calculation of GHG flux followed the description of Yan et al. (2005). Negative flux values indicate GHG uptakes from the atmosphere, and positive flux values represent GHG emissions to the atmosphere.

2.4. Soil properties measurements

Soil samples close to the chamber positions were taken approximately every 15 days and three replicate soils (0–10 cm) were collected from each sampling site at the time of gas flux measurement. The soil samples used for C and N analysis were maintained for 12 h on 10% hydrochloric acid to dissolve carbonates and dried at 70°C for 48 h, then milled in a Tecator 1093 Cyclotec Sample Mill, using a 0–5 mm screen. The soil organic C and total N concentrations were determined using elemental analyser (Model: PE 2400Ⅱ, Perkin Elmer, USA). Soil NH$_4^+$-N and NO$_3^-$-N concentrations were determined by extracting mineral N in the soils with 2M KCl solution (1:5 soil to KCl solution), the suspension was shaken for 1 hour and filtered in Watmann #42 filter paper. NH$_4^+$-N and NO$_3^-$-N concentrations in the extracts were analyzed in an UV-VIS spectrophotometer. Soil pH was determined using a pH meter (Metrohm, Switzerland). The soil physical-chemical properties are shown in Table 1.

Table 1. Physical-chemical properties of the soils (0–10 cm depth) in the pine forest. Mean values ± standard deviation (SD) are given.

<table>
<thead>
<tr>
<th>Soil type</th>
<th>Soil texture</th>
<th>Soil Organic C (mg g$^{-1}$)</th>
<th>Total N (mg g$^{-1}$)</th>
<th>C/N (molar)</th>
<th>NH$_4^+$-N (mg kg$^{-1}$)</th>
<th>NO$_3^-$-N (mg kg$^{-1}$)</th>
<th>WFPS (%)</th>
<th>pH ($^\text{H}_2\text{O}$) (1:2.5)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Limestone soil</td>
<td>Sand silt</td>
<td>36.1±6.1</td>
<td>2.9±0.3</td>
<td>14.3±1.3</td>
<td>22.9±16.0</td>
<td>2.8±1.0</td>
<td>76.0±7.2</td>
<td>6.0±0.4</td>
</tr>
</tbody>
</table>

Table 2. Annual means, minimum and maximum of CO$_2$, CH$_4$ and N$_2$O fluxes.

<table>
<thead>
<tr>
<th>Fluxes</th>
<th>Undisturbed</th>
<th>Disturbed</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Means</td>
<td>Min.</td>
</tr>
<tr>
<td>CO$_2$ flux</td>
<td>491.8±58.9</td>
<td>107.2</td>
</tr>
<tr>
<td>(mg m$^{-2}$ h$^{-1}$)</td>
<td></td>
<td>(24)</td>
</tr>
<tr>
<td>CH$_4$ flux</td>
<td>-124.1±21.0</td>
<td>-354.8</td>
</tr>
<tr>
<td>(μg m$^{-2}$ h$^{-1}$)</td>
<td></td>
<td>(24)</td>
</tr>
<tr>
<td>N$_2$O flux</td>
<td>25.1±4.7</td>
<td>0.7</td>
</tr>
<tr>
<td>(μg m$^{-2}$ h$^{-1}$)</td>
<td></td>
<td>(24)</td>
</tr>
</tbody>
</table>
Effects of litter layer and soil properties on the greenhouse gases fluxes measured in 1:2.5 soil/water solution using a pH meter with a glass electrode. The gravimetric water content was measured by drying a sample at 105°C for 24 h to constant weight, which was transformed to WFPS: WFPS = gravimetric water content × bulk density/(1 – bulk density/2.65).

2.5. Statistical analyzes

The differences in CO₂, CH₄ and N₂O fluxes among the seasons and treatments were compared using a two-way ANOVA followed by the least significant differences (LSD) test, the relationships between CO₂, CH₄ and N₂O fluxes and soil variables (temperature, moisture and mineral N) were examined in a way of linear or nonlinear regression models fitting. Multiple regression analysis was also to examine the relationship between topsoil GHG fluxes, soil temperature and soil moisture. R² of the model parameters were used to determine goodness of fit. All statistical considerations were based on P <0.05 significant level. Statistical analyses were conducted using SPSS 13.0 software package (SPSS Inc., Chicago, USA).

3. RESULTS

3.1. The fluxes patterns of GHG over one year field measurements

In the pine forest, CO₂ emission rates were significantly higher in the summer season (June–August, 2008) than in the winter season (December, 2008–February, 2009) between two treatments (P <0.01) (Fig. 3B). Maximum CO₂ release took place in June 2008 when soil temperature was relatively high and moisture was moderate, while minimum emissions occurred in the winter when both soil temperature and moisture were low (Fig. 2A; 2B). Overall, seasonal difference of CO₂ emissions was more pronounced in the disturbed soil than in the undisturbed soil.
CH₄ measurements indicated a consistent net soil consumption of CH₄ (i.e. negative flux) in both treatments (Fig. 3A). Generally, CH₄ consumption fluxes of two treatments are both relatively low from November 2008 to February 2009, particularly in winter, the CH₄ consumption ability is the weakest, but relatively high in the rest of the year, and the largest consumption occurred in September 2008 (transition period between summer and autumn). Although higher uptake rates were observed in the warm-wet season, ANOVA indicated no significant seasonal difference in CH₄ uptake between both treatments (Un-disturbed: P = 0.221; Disturbed: P = 0.208) (Fig. 3A).

During the period of experiment, the basic pattern of N₂O emissions was similar between both treatments. The trend of seasonal variations of N₂O emission presents roughly as rising gradually from May to August, reaching the maximum in July and declining gradually until early January of next year, and then it rises rapidly from February to April, reaching the secondary high value in March (Fig. 3C). In a word, two obvious peaks of N₂O emissions occurred during the whole measurement period between two treatments in the pine forest of karst region.

### 3.2. Means fluxes of GHG during dry and wet seasons

The highest mean CO₂ emission rate (1409.9 ± 133.5 mg m⁻² h⁻¹) was recorded in the disturbed forest during the wet season, and the lowest mean CO₂ emission rate (105.3 ± 4.6 mg m⁻² h⁻¹) came from the disturbed forest soil during the dry season (Table 2). In the wet season, the CO₂ emissions were about twice as high as those in the dry season between two treatments, and significant differences of CO₂ fluxes were observed between two seasons in both treatments (Undisturbed: P = 0.000; Disturbed: P = 0.002) (Table 3). Compared to the undisturbed forest, CO₂ emission from the disturbed soil decreased both in wet and dry seasons, but no significant differences between two treatments in both seasons (P > 0.05) (Table 3). The CO₂ emissions from the undisturbed and disturbed soils in the wet season were higher by 142 and 152% than those in the dry season, respectively. However, the coefficient of variation (CV) value of CO₂ fluxes for the undisturbed forest (35%) in the wet season was close to that (34%) in the dry season. The results showed that a similar temporal variability of CO₂ emission between wet and dry seasons in the pine forest.

The average annual CH₄ absorption in the disturbed forest (153.9 ± 25.4 μg m⁻² h⁻¹) was higher than that in the undisturbed forest (124.1 ± 21.0 μg m⁻² h⁻¹), but there was no significant difference between two treatments (P = 0.370) (Table 2). Compared to the undisturbed forest, the CH₄ absorption in the disturbed forest increased both in wet and dry seasons (Table 3). The CH₄ absorptions from the undisturbed and disturbed forests in the wet season were higher by 41 and 63% than those in the dry season, respectively. The CV values of CH₄ fluxes for the undisturbed forest (32%) in the wet season were close to that in the dry season (31%). However, in the disturbed forest, the CV values of CH₄ fluxes (36%) in the wet season were lower than that (57%) in the dry season.

The average N₂O flux over the sampling period was 25.1 ± 4.7 μg m⁻² h⁻¹ with a range of 0.7 to 84.7 μg m⁻² h⁻¹ for the undisturbed forest and 26.1 ± 3.8 μg m⁻² h⁻¹ with a range of −6.5 to 67.2 μg m⁻² h⁻¹ for the disturbed forest. There was no significant difference between

<table>
<thead>
<tr>
<th>Seasons</th>
<th>CO₂ (mg m⁻² h⁻¹)</th>
<th>CH₄ (μg m⁻² h⁻¹)</th>
<th>N₂O (μg m⁻² h⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wet</td>
<td>Undisturbed</td>
<td>Disturbed</td>
<td>Undisturbed</td>
</tr>
<tr>
<td></td>
<td>(14)</td>
<td>(14)</td>
<td>(10)</td>
</tr>
<tr>
<td>Average</td>
<td>651.1±87.2</td>
<td>545.0±101.0</td>
<td>268.9±41.5</td>
</tr>
<tr>
<td></td>
<td>(14)</td>
<td>(14)</td>
<td>(10)</td>
</tr>
<tr>
<td>Average</td>
<td>-141.2±16.9</td>
<td>-183.5±24.6</td>
<td>-100.1±13.7</td>
</tr>
<tr>
<td></td>
<td>(14)</td>
<td>(14)</td>
<td>(10)</td>
</tr>
<tr>
<td>Average</td>
<td>28.6±6.4</td>
<td>24.3±3.8</td>
<td>20.2±11.6</td>
</tr>
<tr>
<td></td>
<td>(14)</td>
<td>(14)</td>
<td>(10)</td>
</tr>
</tbody>
</table>

Different superscripts indicate significant (P ≤ 0.05) in the fluxes. The number of observation is in the parenthesis below the mean. Negative values indicate uptake, and positive values indicate emission.
Effects of litter layer and soil properties on the greenhouse gases fluxes

4. DISCUSSION

4.1. Effects of litter layer on GHG fluxes

Many studies indicate that litter removal reduces CO$_2$ emissions significantly (Dong et al. 1998, Rey et al. 2002, Li et al. 2004). After the removal of litter layer from the forest floor, a distinct reduction of CO$_2$ emission from the bare soil was observed in most cases of this study (Fig. 3B; Table 2), which was mainly due to the role of the litter layer as a large reservoir of organic material ready for microbial decomposition. About 17% of soil CO$_2$ emission was contributed by aboveground litter layer at our site, which was similar to published estimates by 19% in a temperate old-growth forest (Sulzman et al. 1989), 22% in the deciduous forests in Germany (Dong et al. 1998) and 21.9% in a Mediterranean mixed oak forest (Rey et al. 2002). However, it was lower than values from a wet tropical forest in Puerto Rico (54–68%) (Li et al. 2004). In a word, the impact of litter exclusion in our study was at the lower end of the mean value (20–30%) derived from forests worldwide (Raich and Nadelhoffer 1989). This could be due to an association with the obvious decrease of litter layer fall during the experiment period.

We estimated that a 24% increase in CH$_4$ uptake following the removal of the litter layer in the subtropical pine forest (Fig. 3A; Table 2). Yan et al. (2008) reported a 29% increase in CH$_4$ uptake following the removal of the layer in a tropical rain forest, similar to our results. The effect of the litter layer on CH$_4$ exchanges occurs mainly through its control on a diffusion barrier for gases diffusion into and out of the soil. Therefore, the increased CH$_4$ consumption in the disturbed soils after litter removal may be mainly due to a higher permeation rate, which promotes CH$_4$ oxidation through an enhanced contact between atmospheric CH$_4$, O$_2$ and the biologically active of soil layer (Dong et al. 1998).

In contrast to CO$_2$ and CH$_4$ fluxes, no distinct change in N$_2$O fluxes was found after the litter removal (Table 3). Generally, N$_2$O two treatments ($P = 0.864$), where slightly higher N$_2$O fluxes were observed in the disturbed treatment compared to the undisturbed treatment (Table 2). The highest mean N$_2$O emission rate ($84.7 \pm 24.0$ μg m$^{-2}$ h$^{-1}$) was recorded in the undisturbed forest during the wet season, and a relatively small N$_2$O uptake rate ($6.5 \pm 5.1$ μg m$^{-2}$ h$^{-1}$) was mentioned in the disturbed forest during the wet season, indicating that the soil is able to be a sink for N$_2$O by reducing its atmospheric N$_2$O. However, in both treatments, no significant difference was observed between dry and wet seasons ($P > 0.05$) (Table 3). The CV values of N$_2$O fluxes for the undisturbed (59%) and disturbed (41%) forests in the wet season were lower than those in the dry season (128%; 79%, respectively). The results showed that a higher temporal variability of N$_2$O emission in the dry season than the wet season between two treatments.

3.3. Relationships between GHG fluxes and environmental factors

In the present work, soil temperatures at 5cm were the main factors regulating the temporal variations of CO$_2$ fluxes in the undisturbed and disturbed forests ($r^2 = 0.87$ and 0.80, $P < 0.01$, respectively) (Fig. 4). In addition, statistic analysis with logarithmic regression identified that the soil inorganic N concentration with a significant influence on the temporal variability of CO$_2$ emissions in the undisturbed forest ($r^2 = 0.35$, $P < 0.05$) (Fig. 5). Similarly, the CH$_4$ uptake rates were significantly and positively correlated to soil temperature under layer litter ($r^2 = 0.34$, $P < 0.05$) and litter removal ($r^2 = 0.37$, $P < 0.05$) (Fig. 6), and soil temperature could explained 34% and 37% of the temporal variation in the CH$_4$ uptakes in the undisturbed and disturbed forests, respectively. Furthermore, a moderately strong positive relationship ($r^2 = 0.48$, $P < 0.05$) was found between CH$_4$ fluxes and the soil inorganic N concentration (Fig. 7). And there was also a significantly positively logarithmic correlation between CH$_4$ flux and soil NH$_4^+$-N concentration ($r^2 = 0.39$, $P < 0.05$) (Fig. 7). For N$_2$O flux, a significantly negatively logarithmic correlation with soil NO$_3^-$-N concentration was found ($r^2 = 0.41$, $P < 0.05$) (Fig. 8). However, other factors as measured in this study were not significantly correlated with temporal GHG fluxes of surface soils.
emission between soil and the atmosphere are largely determined by soil moisture (Davidson et al. 1993, Kiese and Butterbach-Bahl 2002). In our study, soil temperature and soil moisture levels changed little after litter removal in most cases, and the annual averages of WFPS were 76.0% and 69.2% in undisturbed and disturbed forests, respectively (Fig. 2B), suggesting that minor changes in soil moisture showed no obvious influence on microbial activities. It also suggested that the majority of the activities of nitrification and denitrification happen in the mineral soil rather than in the litter layer of surface soil in the pine forest (Tang et al. 2006). However, the litter layer had the positive effect in the wet season, resulting in more N\(_2\)O emissions (15%), while it showed a more negative effect on N\(_2\)O emissions (42%) in the dry season (Fig. 3C, Table 3). Therefore, the litter layer had totally different roles in N\(_2\)O production in different seasons. The reason might be that nitrification and denitrification carried out in different ways under different precipitation conditions (Yan et al. 2005).

4.2. Effects of soil temperature and moisture on GHG exchanges

Significant exponential regressions were analyzed between soil temperature at 5 cm and the undisturbed forest (\(F_{CO_2} = 88.10e^{0.099T}\), \(r^2 = 0.87, P < 0.01\)), and the disturbed forest (\(F_{CO_2} = 72.48e^{0.097T}\), \(r^2 = 0.80, P < 0.01\)) (Fig. 4), with estimated Q\(_{10}\) values (Q\(_{10}\): increased of the flux with a temperature increase of 10°C) of 2.69 and 2.64 for the undisturbed and disturbed forests, respectively. These values were lower than Q\(_{10}\) values of 3.9 for an ambient plot and 5.7 for a drought plot in temperate forest soils (Borken et al. 1999), but similar to the global median value of 2.4 reported in a literature review when soil temperatures were used in the evaluation (Raich and Schlesinger 1992). However, soil moisture did not have a strong effect on CO\(_2\) emissions (Undisturbed: \(r^2 = 0.17, P > 0.05\); Disturbed: \(r^2 = 0.08, P > 0.05\)), which are consistent with results reported in a temperate mixed hardwood forest (Davidson et al. 1998), young ponderosa pine plantation (Xu and Qi 2001) and coppice oak forest (Rey et al. 2002). In our study, soil WFPS showed a positive rather than negative relations with temperatures (Fig. 2). This is partly caused by the fact that the soil moisture measurements were often lower than the soil field capacity (Fig. 2B) and not high enough to reach the point when mineralization gets limited by reduced oxygen diffusion into the soil. Moreover, because of the covariation of soil moisture and temperature driven by the simultaneous seasonal patterns of precipitation and air temperature, it is difficult, if not impossible, to distinguish the relative importance of moisture and temperature in controlling CO\(_2\) emission rates (Tang et al. 2006). By combining soil temperature at 5 cm depth with soil moisture at 0–10 cm in a multiple linear regression model to explain the temporal variations in CO\(_2\) fluxes resulted in a significant relationship between two treatments and we could explain 77% (Undisturbed) and 63% (Disturbed) of the temp-
Effects of litter layer and soil properties on the greenhouse gases fluxes

poral variation in measured CO₂ emissions at the pine forest in karst region.

When soil temperature is relatively low, the CH₄ and O₂ diffusion potentials are higher than soil CH₄ and O₂ consumptions rates due to weak soil microbial activity (Nedwell and Watson 1995). In this case, soil temperature is the limiting factor for CH₄ oxidation. A significant positive correlation between soil temperature and CH₄ uptake rates was observed (Undisturbed: r² = 0.34, P < 0.05; Disturbed: r² = 0.37, P < 0.05) (Fig. 6), which is consistent with other results (Crill 1991, King 1997). However, soil moisture did not correlate with CH₄ fluxes significantly (Undisturbed: r² = 0.02, P > 0.05; Disturbed: r² = 0.00, P > 0.05). Decreasing CH₄ consumption with increasing soil moisture has been reported in other studies (Castro et al. 1994, Rosenkranz et al. 2006), but Savage et al. (1997), as well as this study, did not observe any response of CH₄ fluxes to soil moisture, and soil moistures throughout the dry season (WFPS during this period ranging from 62.6 to 79.3%, and from 54.0 to 76.9% in the undisturbed and disturbed forests, respectively) were often lower than the water-holding field capacity. Hence, soil water contents in our research site probably did not reach the critical values needed to affect the activities of CH₄ consuming microbes during the experiment.

In the pine forest, the lack of any soil temperature and soil moisture effects on N₂O exchange (P > 0.05, data not shown) contrasts with other studies which showed strong dependencies of N₂O fluxes on both controls (Smith et al. 2003, Ball et al. 2007), but it is consistent with results reported in agricultural and forest soils (Crill et al. 2000, Kiese and Butterbach-Bahl 2002, Ullah et al. 2009). Previous studies showed that soil from tropical agriculture ecosystem to pasture with low nitrogen content (4.0–5.3 mg g⁻¹) had a weak or none dependency of N₂O fluxes

Fig. 5. Relationship of CO₂ flux with soil (NH₄⁺-N+NO₃⁻-N) concentrations of surface soil (0-10 cm depth) in the pine forest. Regression curve is fitted for values averaged at each month. Error bars indicate standard deviations of fluxes and environmental factors.

Fig. 6. Relationships of CH₄ fluxes to soil temperature at a depth of 5 cm between undisturbed and disturbed pine forests. Regression line is fitted for values averaged at each month. Error bars indicate standard deviations of fluxes and environmental factors.

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from increasing soil moisture (Dobbie et al. 1999, Weitz et al. 2001). Our study also indicated that N$_2$O emission was insensitive to soil temperature and moisture, possibly resulting from the lower soil nitrogen content (2.9 ± 0.3 mg g$^{-1}$) (Table 1). Therefore, it is not uncommon to encounter a lack of relationship between soil temperature and moisture and N$_2$O fluxes measured at plot scale within the same forest stand (Peichl et al. 2010).

4.3. Effects of soil mineral N on soil GHG fluxes

Previous studies showed that CO$_2$ emissions from agricultural soil or long-term fertilization soil decrease logarithmically with soil inorganic N (Koos and Nemeth 2007, Inselsbacher et al. 2011). A similar phenomenon would be observed in the pine forest in our study and a moderately strong negative relationship ($r^2 = 0.35$, $P < 0.05$) was found between CO$_2$ emissions and soil inorganic N (Fig. 5). These results suggested that the increasing depletion of soil inorganic N pools due to root uptake together with a simultaneous increase in root respiration then a direct influence of soil dissolved inorganic N concentrations on CO$_2$ emissions (Inselsbacher et al. 2011), which is in accordance with Raich and Tufekcioglu (2000) who found that root respiration contributed 12–38% to the total soil respiration.

Increased N availability has been shown to reduce the soil CH$_4$ sink (Steudler et al. 1989). (Keller et al. 1990) also suggested that a pulse of N availability after deforesta-
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In the pine forest, litter layer removal resulted in a decrease in CO$_2$ emission (17%) and an increase in CH$_4$ uptake (24%), while it had no distinct influence on N$_2$O fluxes. This study confirmed temperature exerts a controlling influence on CO$_2$ emission and CH$_4$ uptake, and there was a significant correlation between CH$_4$ flux and NH$_4^+$-N (r$^2 = 0.39$, P < 0.05). Furthermore, we also found a significant negative logarithmic correlation between N$_2$O flux and soil NO$_3^-$-N concentration (r$^2 = 0.41$, P < 0.05) and the relationship between CO$_2$ emission and soil inorganic N (r$^2 = 0.35$, P < 0.05). These results indicated that litter, soil temperature and mineral N dynamics largely affected the soil GHG fluxes in the karst forest, southwest China.

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6. REFERENCES


Bouwman A.F. 1990 – Exchange of greenhouse gases between terrestrial ecosystems and the atmosphere. (In: Soils and the greenhouse
Morishita T., Hatano R., Nagata O., Sakai K., Koide T., Nakahara O. 2004 – Effect
Effects of litter layer and soil properties on the greenhouse gases fluxes


Vose J.M., Bolstad P.V. 2006 – Biotic and abiotic factors regulating forest floor CO₂ flux across a range of forest age classes in the southern Appalachians–Pedobiologia, 50: 577–587.


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