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Greenhouse Gases Exchange at the Forest Floor of a Dominant Forest in South China

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Abstract

We used static chambers and the gas chromatography method to measure greenhouse gases (GHG) exchange and collected ancillary data (soil moisture, soil temperature and litterfall) in a dominant mixed coniferous and broad-leaved forest (MF) in the subtropical region of South China. Data were collected from May 2003 to April 2004. The objectives of the study were to examine temporal variations of GHG fluxes at the forest floor and factors controlling GHG exchange. It was unclear whether or not there was a diurnal peak in GHG exchange. Field data strongly suggested good correlations between average values from 9:00 A.M. to 12:00 P.M. and the diurnal mean values of GHG exchange. As found in other research, the seasonal dynamic patterns of GHG fluxes revealed a significant difference between the wet and dry season.

Consumption of CH₄ showed a negative relation to water-filled pore space that was partly offset by CH₄ production as soil neared saturation. CO₂ fluxes showed fairly well described exponential relationships with soil temperature and Q₁₀ values in three treatments and were similar to the median value of 2.4 found in a review of relevant literature. A single linear function applies to the relationship of N₂O to water-filled pore space and the r² values obtained were in a range from 0.51 to 0.55 for different treatments.

In addition, the relationships between litterfall and GHG fluxes suggested that litterfall has an important effect on GHG exchange. These findings need to be tested against a greater range of forest types and climatologies.

Key Words: Mixed coniferous and broad-leaved forest; greenhouse gases; chamber system; soil temperature; water-filled pore space; litterfall

Introduction

After years of industrial development the scientific community has concluded that the world's temperature is rising unusually fast. Climate models used by the Intergovernmental Panel on Climate Change (IPCC) project that the global climate will warm by from 1.4°C to 5.8°C by the end of the 21st century (IPCC 2001). A central tenet of Earth System Science is the accumulation of radiatively active gases in the atmosphere will lead to the absorption of infrared radiation by these gases which in turn will cause an increase in average atmospheric temperature (Houghton 1997). Gases of principal concern are CH₄, CO₂, N₂O, chlorofluorocarbons, and O₃ (Houghton *et al.* 1996). Each of these gases has different sources, spectral behavior and sinks (Cicerone 1989, Bouwman 1998). Through a delicate balance between absorption and efflux, floors of terrestrial ecosystems, and in particular forests, are thought to emit a significant portion of CO₂ and N₂O to the atmosphere (Moren and Lindroth 2000, Reiners *et al.* 2002), but absorb CH₄ emissions from the atmosphere (Keller and Reiners 1994). However, the location, magnitude, and vulnerability of GHG absorption and efflux of the floors of terrestrial biota are still uncertain. GHG occurrences and losses often show a strong seasonal variability and a relationship with physical factors. The influence of climate and

phenology can in some cases shift a forest floor from being a sink to a source of CH₄ (Keller and Reiners 1994).

Human activity, which has caused extensive changes in the Earth's land cover for millennia, has vastly accelerated over recent decades (Meyer and Turner 1992, Reiners *et al.* 2002). These activities have affected terrestrial GHG sources and sinks (Canadell 2002). The Chinese government had made great efforts to rehabilitate its degraded forest lands in the subtropical region at the end of the last century. The project 'Restoring degraded hills in five years, greening Guangdong in ten years' for ecological benefits, achieved its target. To further reinforce these achievements, the provincial government accelerated the construction of forests for ecological benefits. An important achievement of forest rehabilitation activities has been that pioneer forests (Masson pine forests) cover increased from 19% to 43% of the land area (Zhou and Yan 2000). By the subtropical forest succession and human intervention, the pioneer forests are changing in the mixed coniferous and broad-leaved forests (MF). Based on practice and succession theory (Peng and Wang 1995), MF may be a permanent feature of the subtropical landscape in South China. As a result of the changes in land cover there may be large changes in the soil-atmosphere fluxes of CH₄, CO₂ and N₂O

(Keller *et al.* 1993). It is therefore appropriate to discuss GHG exchange at the forest floor of a MF in the subtropical zone in South China.

Soil-atmosphere exchange of gases is the result of a number of complex processes, including production, transport, and interactions between physical and biological factors within soil. Many studies have indicated that soil temperature is a good predictor of CO₂ exchange (Raich and Schlesinger 1992, Kicklighter *et al.* 1994, Reiners *et al.* 2002) and water-filled pore space has a close relation to CH₄ and N₂O exchange at the forest floor (Keller and Reiners 1994, Dobbie *et al.* 1999). Our work was carried out at the Dinghushan Biosphere Reserve at a sample site that is typical of a MF in the subtropical region in South China. GHG in this region might increase quickly because of industrial development in this area. It is an important part of the government's management of the environment to understand the sources and sinks of GHG.

In this paper we address the following questions: (1) What are the patterns of diurnal variation of GHG exchange and why are they unclear? (2) What are the seasonal patterns in GHG fluxes from soil to atmosphere at the study site? (3) How do soil temperature and soil moisture affect the GHG exchange at the forest floor? (4) What are the relationships between GHG fluxes and litterfall?

The study area

Dinghushan Biosphere Reserve (23° 09' 21" ~ 23° 11' 30" N, 112° 30' 39" ~ 112° 33' 41" E) is located in central Guangdong Province, South China, about 84 km west of Guangzhou. The total area of the reserve is 1,156 ha. The rock formations of Dinghushan are composed of sandstone and shale belonging to the Devonian Period. Most of the Dinghushan area is covered with rolling hills and low mountains, with the altitude ranging from 100 to 700 m. Jilongshan is the highest point with an altitude of 1,000 m. This reserve has a typical, subtropical monsoon, humid climate with an average annual temperature of 20.9°C. The highest and lowest monthly mean temperatures are 28.0°C in July and 12.0°C in January, and the highest and lowest extreme temperatures are 38.0°C and -0.2°C, respectively. The average annual precipitation is 1,956 mm, of which more than 80% falls in the wet season (April to September) and less than 20% comes in the dry season (October to March). Annual mean relative humidity is 82%.

The predominant soil type in Dinghushan is lateritic red-earth, between the elevations of 400 to 500 m, followed by yellow earth which is found between the elevations of 500 to 800 m. The pH of the soil is 4.5 to 6.0 and a rich humus layer is common. The Biosphere Reserve is covered by a MF. The age of the stand of the MF used in this study is about 100 years old, dominated by *Pinus massoniana*, *Schima superba* and *Castanopsis chinensis*. The flora includes 260 families, 864 genera, and 1,740 species of wild plants.

Methods

GHG fluxes measurement

Nine permanent sampling points were randomly selected on the MF floor. During the measurement period, three points were measured for GHG fluxes on bare ground (both litter and shrub removal treatment: S), another three were measured with litter cover (shrub removal treatment only: S+L), and the rest were measured with an intact forest floor (both litter and shrub retention: S+L+P). We measured the GHG fluxes 4 days per month from 9:00 A.M. to 12:00 P.M. local time and another day once a month to measure diurnal variation. This experiment commenced in March 2003 and measurements continued from May 2003 through April 2004.

GHG were measured in the field using static, vented field chambers following the method of Hutchinson and Mosier (1981). The chamber system was made up of a pedestal with a trough and a top chamber (Length, width and height were all 50 cm). The chambers were constructed from stainless steel and when in use the top chamber was covered by a cotton pad to prevent heat exchange. The pedestals were always inserted to a depth of 10 cm (the depth of the litter layer is about 5-7 cm) as a permanent measurement point and established at least 4 weeks prior to the first sampling. One tube was installed on the upper section of the top chamber to allow the collection of gas samples, another tube was installed at one of the sides of the chamber to equalize air pressure between the interior and exterior, and two mini electric fans promoted mixing gases in the chamber. When collecting samples, water was poured into the trough of the pedestal and then the top chamber was inserted into the trough, forming an airtight seal. An airtight syringe was immediately inserted through a septum-covered access port to collect the initial gas sample. Four successive samples were collected at 10-minute intervals to determine the rate of change in chamber gas concentration, and exact times of collection were recorded. After 100-150ml of gas was removed with the airtight syringe the remaining chamber air was rapidly expelled via a three-port valve before 80-100ml mixed headspace air was collected for analysis. Filled syringes were sealed and stored for later analysis. Simultaneously, the air temperature and the air pressure were measured inside the chamber. The gas samples were analyzed in the laboratory within 6 hours of collection using a HP4890D gas chromatograph, equipped with flame ionization detectors (FID), to ascertain gas concentrations. The flux rates were calculated from the linear increase gas concentrations in the headspace of the chambers. Corrections were made for air temperature and pressure. Flux data were declared valid if the regression coefficient (which was calculated from the linear increase with time in gas concentrations in the chamber headspace) was significantly different from zero ($P < 0.05$). All the results of this paper were based on valid flux data. Fluxes were calculated from linear regressions of the gas concentration versus time curves according to:

$$F_c = \rho \frac{V}{A} \frac{dc}{dt} \quad (1)$$

The flux F_c ($\text{mg m}^{-2} \text{h}^{-1}$) is equal to the linear change in gas concentration within the chamber multiplied by the density ρ (mg m^{-3}) of chamber air and the ratio of chamber volume V (m^3) to soil surface area A (m^2). Positive fluxes indicate an emission of gases by soil to the atmosphere. Negative fluxes indicate a net consumption of gases by soil from the atmosphere.

Temperature and moisture measurements

Soil moisture (Trime Portable TDR, MESA Systems Co.) in 0-10 cm and temperatures (Thermistor, TES-1300/1302, Taiwan) at 2 m above ground, at the land surface and at the depth of 5 cm were recorded together with each chamber measurement. In an adjacent MF stand, the dry bulk density was calculated. Five replicate cores were taken using a plexiglass piston corer (diameter 9 cm, length 10 cm) once a month. The cores were taken to the lab and weighed,

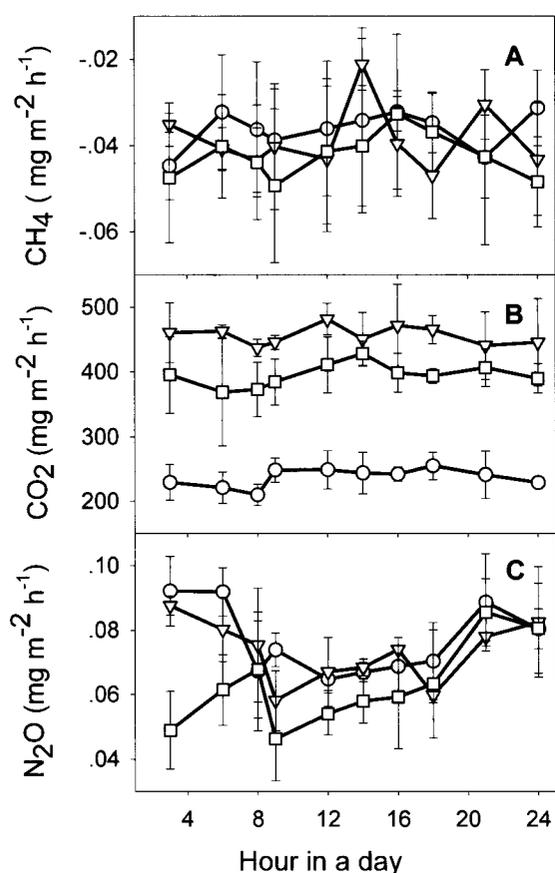


Fig. 1. Mean values ($\text{mg m}^{-2} \text{h}^{-1}$) sampling one day per month over a period of one year and standard error for diurnal variation of CH_4 (A), CO_2 (B) and N_2O (C). The symbols used are circles for S, triangles for S+L and squares for S+L+P.

and then dried at 105°C and weighed again. The water content of the cores and dry bulk density were calculated. From soil moisture (determined by TDR) and dry bulk density, the percentage of water-filled pore space was then calculated.

A seven-layer 38 m flux tower was established close to the site. The open-path technique was applied to measure eddy covariance at the second layer (under the canopy) and the 5th layer (above the canopy). Solar radiation, rainfall, and PAR data were collected at the top layer. The relevant routine meteorological data, including temperature, humidity and wind speed, and direction were collected at all seven layers. Meanwhile, soil moisture and soil temperature were also measured at different depths.

Litterfall sampling

Fifteen litterfall traps were placed to best represent the state of the stand canopy for the whole community. The traps were made of 1 m^2 plastic nets to allow throughfall to pass through easily but retain litter particles. The traps were suspended 50 cm off the ground. Litterfall was collected once a week from March 2003 to May 2004. The litter was first air-dried, and then separated into leaves, branches, bark, and flower/fruit. The unidentified fine litter particles were added to flower/fruit. All the components were weighed in their air-dried state, and then dried at 65°C until constant weights were obtained and the dry weight of each component in all samples was recorded.

Results

Diurnal (24 h) variations of GHG

Diurnal variation patterns of CH_4 , CO_2 and N_2O fluxes are shown in Figure 1. The data were derived from 24-hourly flux measurements from May 2003 through April 2004. The diurnal variation patterns of GHG were not apparent. A diurnal peak in GHG exchange was very unclear. Statistical analysis also showed that the differences between daytime and nighttime for any of the GHG were not significant ($p = 0.05$, $n=36$) with the calculated t values (0.802, 1.044 and 1.879 for CH_4 , CO_2 and N_2O , respectively) being less than the value given in the T -test table.

Diurnal mean values of CH_4 exchange were $-0.036 \text{ mg m}^{-2} \text{h}^{-1}$, $-0.038 \text{ mg m}^{-2} \text{h}^{-1}$ and $-0.042 \text{ mg m}^{-2} \text{h}^{-1}$ for S, S+L and S+L+P, respectively. The values showed consumption of CH_4 in S+L and S+L+P were greater than in S, which demonstrated that litter and shrubs could be a net sink for CH_4 at the forest floor of a MF in South China.

Diurnal mean values of CO_2 exchange were $236.4 \text{ mg m}^{-2} \text{h}^{-1}$, $456.2 \text{ mg m}^{-2} \text{h}^{-1}$ and $394.8 \text{ mg m}^{-2} \text{h}^{-1}$ for S, S+L and S+L+P, respectively. The plots with S+L showed the greatest CO_2 efflux among the three treatments, suggesting that litter respiration made an important contribution to the increase in CO_2 exchange. The lower CO_2 exchange rate in the S+L+P plot compared to the S+L one can be attributed to less litter covering the ground under the shrubs.

Diurnal mean values of N_2O exchange were $0.076 \text{ mg m}^{-2} \text{h}^{-1}$, $0.073 \text{ mg m}^{-2} \text{h}^{-1}$ and $0.063 \text{ mg m}^{-2} \text{h}^{-1}$ for S, S+L

and S+L+P, respectively. Litter and shrubs seemed to reduce the transport of N_2O from the forest floor to the atmosphere. A possible explanation may be that soil covered with litter and shrubs contains more organisms and microbes than bare ground, which should encourage microbe activity. To improve understanding of the variations in N_2O at the forest floor, this process should be studied further.

Analysis of the changes of diurnal GHG fluxes using the data collected one day each month revealed that average values from 9:00 A.M. to 12:00 P.M. were very similar to the whole day means (Fig. 2). The same result has been found in a previous study (Davidson *et al.* 1998, Zheng *et al.* 1998). In that study measurements were made between 9:00 A.M. and 12:00 P.M., which was when the average flux of the diurnal cycle occurred. They were, therefore, able to use the morning measurements to make daily estimates. We studied this phenomenon further and found that observation of the physical factors from 9:00 A.M. to 12:00 P.M., such as soil temperature and soil moisture, explained most of the variations in GHG exchange at the forest floor (Fig 3). The data used were from May 2003 to April 2004 (No data from July 22 to 25, 2003), recorded by a flux station near the experimental site. For both soil temperature and moisture, the average values from 9:00 A.M. to 12:00 P.M. were virtually the same as the daily mean values.

Seasonal dynamics of GHG

According to the results shown in Figures 2 and 3, it was possible for us to assume that morning measurements (from 9:00 A.M. to 12:00 P.M.) were reasonable estimates of the daily mean values. We calculated the monthly values of GHG exchange from the five observation days each month, and applied them to further investigate seasonal variations in GHG as shown in Figure 4.

The forest floor of the MF consumed CH_4 throughout the year at different treatment conditions. The seasonal dynamics of CH_4 exchange indicated a slight difference between the wet season and the dry season though soil moisture and temperature in the wet season were much higher than those in the dry season. CH_4 consumption in the dry season was slightly greater than that in the wet season. CH_4 consumption in the dry season was 55% of the total annual consumption for S, 54% for L+S and 51% for L+S+P. Litter and shrubs had a negative effect on CH_4 consumption in the dry season.

The seasonal dynamic pattern of CO_2 flux revealed a significant difference between the wet season and the dry season. In the Dinghushan area, the wet season coincides with high temperatures, which results in improved biological activity. The combination of water and heat inputs causes much greater CO_2 flux in the wet season than in the dry season. CO_2 efflux in the dry season was 40% of that in the wet season for S, 43% for L+S and 40% for L+S+P. The plot with litter cover aggravated the difference between the wet season and the dry season.

N_2O emissions from soil were inherently variable with changes in the season. N_2O emissions in the wet

season were $0.076 \text{ mg m}^{-2} \text{ h}^{-1}$, $0.058 \text{ mg m}^{-2} \text{ h}^{-1}$ and $0.051 \text{ mg m}^{-2} \text{ h}^{-1}$ for S, L+S and L+S+P, respectively, while those in the dry season were $0.026 \text{ mg m}^{-2} \text{ h}^{-1}$, $0.043 \text{ mg m}^{-2} \text{ h}^{-1}$ and $0.030 \text{ mg m}^{-2} \text{ h}^{-1}$ for S, L+S and L+S+P, respectively. Comparing different treatments, litter and shrubs had a negative effect in the wet season that resulted in more N_2O emissions, while in the dry season showed a more positive effect on N_2O emissions. Nitrification and denitrification are the microbial processes that dominate N_2O production in soil (Dobbie and Smith 2001). Litter and shrubs have totally different roles in N_2O production depending on the season. This might be explained by the different ways nitrification and denitrification occur under different soil moisture conditions.

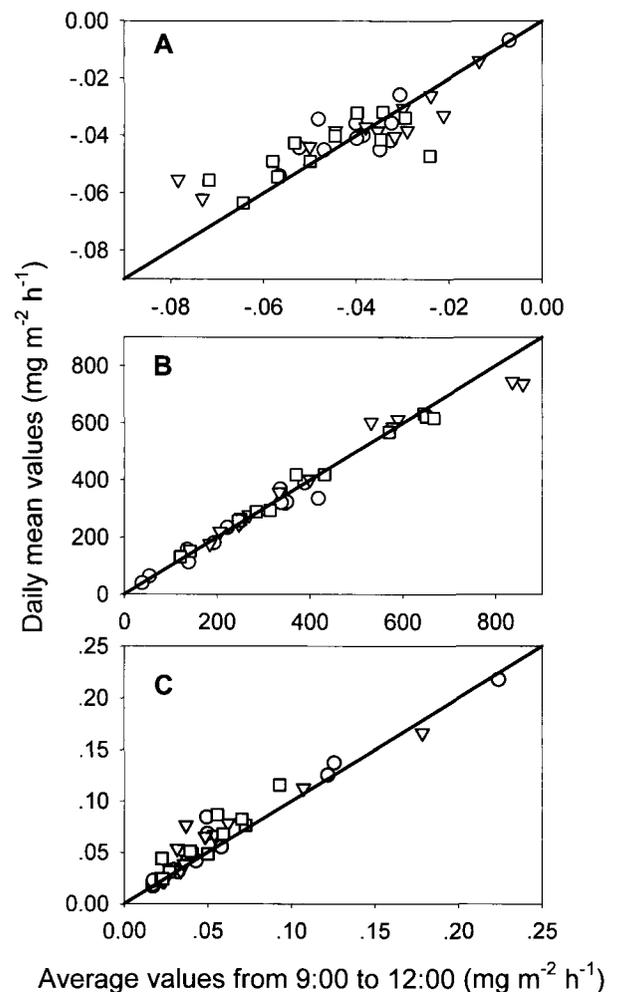


Fig. 2. A comparison between average values ($\text{mg m}^{-2} \text{ h}^{-1}$) from 9:00 A.M. to 12:00 P.M. and daily mean values ($\text{mg m}^{-2} \text{ h}^{-1}$), for CH_4 (A), CO_2 (B) and N_2O (C). The symbols used are circles for S, for triangles S+L and squares for S+L+P.

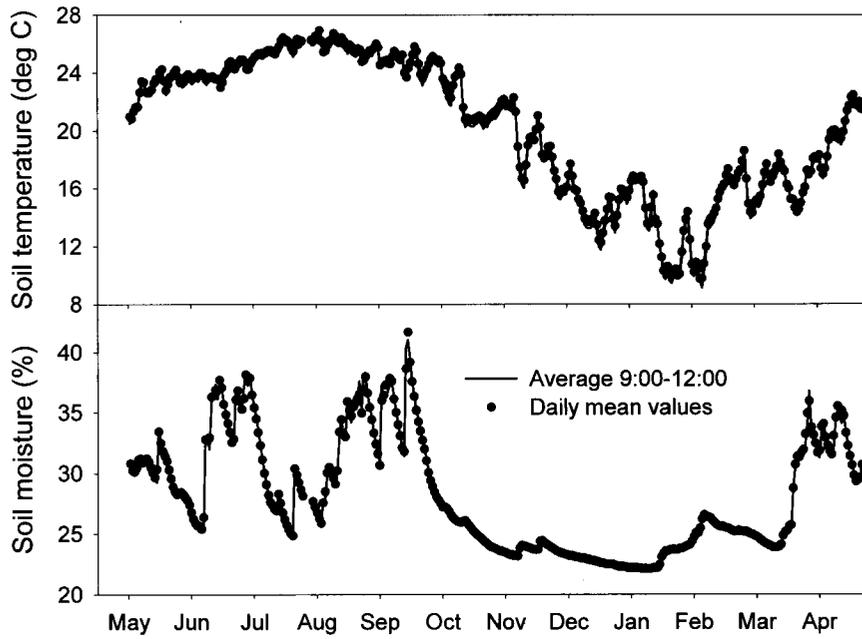


Fig. 3. A comparison between average values from 9:00 A.M. to 12:00 P.M. and daily mean values for 361 days in 2003-2004, for temperature (deg C) at 5 cm soil depth and volumetric soil moisture (%) at 0-10 cm depth.

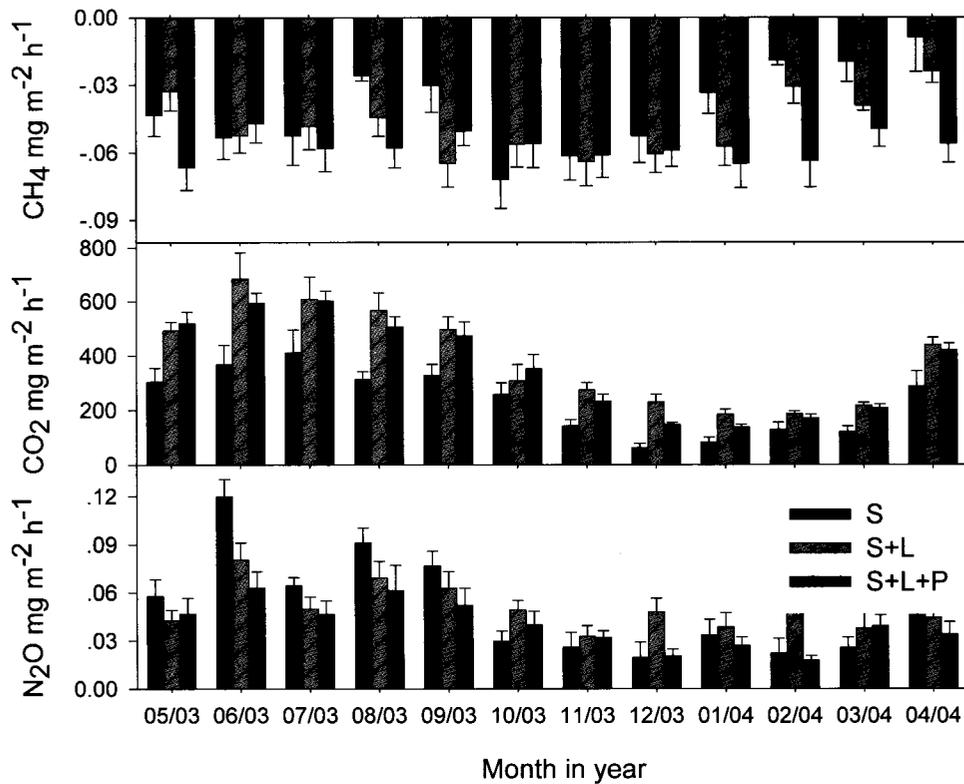


Fig. 4. Monthly dynamics of flux rates for greenhouse trace gases at the floor of MF.

Discussion

Temporal variations in GHG exchange

Although normally soil temperature peaks after noon and soil moisture peaks around sunrise (Yan *et al.* 2000), no significant differences in GHG exchange between daytime and nighttime were observed in our experiment, and a diurnal peak in GHG exchange was also very unclear. It is not directly related to soil temperature or soil moisture. The effects of soil moisture and temperature might be offset by other factors and result in a relative steady-state physical condition for GHG exchange at the forest floor.

This result was also observed in CO₂ flux in S+L+P during the day in spite of the existence of shrubs. A likely explanation for this was that the chambers used were made of stainless steel and covered by white cotton. They were not transparent. Therefore, the diurnal variation did not include shrub photosynthetic uptake during daytime measurement. Moren and Lindroth (2000) pointed out that peak soil respiration in a boreal forest might often occur near sunset. Similar results were found in an earlier study suggesting that soil respiration exhibits a burst at dusk, which often was greater than the highest respiration rates measured in the afternoon (Baldocchi *et al.* 1986). We were not convinced that this was the normal pattern of diurnal variation for CO₂ exchange at the forest floor. In 1991, Baldocchi and Meyers suggested that their earlier measurements were not reliable, because of non-steady-state conditions caused by a rapid build-up of carbon concentration in the trunk space.

In reviews of the literature, we discovered that there was no agreement about diurnal variation in N₂O emissions because of the complicated processes of N₂O soil-atmospheric exchange. Dong (2000) made a study on the floor of a grassland ecosystem and reported that the rates of N₂O emissions during nighttime were greater than those in daytime. However, the result did not fit the diurnal variation of N₂O emissions at the

floor of a forest ecosystem. For a MF, the rates of N₂O emissions in daytime were a little bit higher than those during nighttime.

At the forest floor, the physical factors of diurnal patterns varied relatively little but there were great changes in seasonal dynamics. Therefore, many scientists have focused on differences between dry and wet seasons, or dry and wet conditions, in recent papers (Dobbie and Smith 2001, Borken *et al.* 2002, Smith *et al.* 2003, Sotta *et al.* 2004). On a diurnal time scale, soil moisture tended to be negatively correlated with soil temperature so their effects might cancel each other out and lead to relatively constant GHG fluxes, but monthly variations in GHG fluxes were influenced by seasons, positively correlated to temperature and moisture variations. In the Dinghushan area, more than 80% of annual rainfall falls during April-September, creating distinct wet and dry seasons. Figure 3 also shows that soil temperature varied clearly throughout the year and the higher temperature period coincided with the season of the most soil moisture. Both temperature and soil moisture in the subtropical region increased markedly during the wet season from April through September (Zhou and Yan 2001). Therefore, it is important to talk to other scientists in this research field about seasonal dynamics.

In Figure 4, the seasonal dynamic patterns of CO₂ and N₂O fluxes revealed a strong difference between the wet season and the dry season because soil temperature and soil moisture have been frequently identified as dominant factors in controlling soil CO₂ and N₂O fluxes. However, no significant differences of CH₄ fluxes between the wet and dry seasons were observed (Fig 4). This might be relative to the processes of CH₄ exchange between the atmosphere and soil on the forest floor.

The lifetime of CH₄ in the atmosphere is quite short, approximately 10 years. Most atmospheric CH₄ is destroyed through oxidation by UV-created hydroxyl

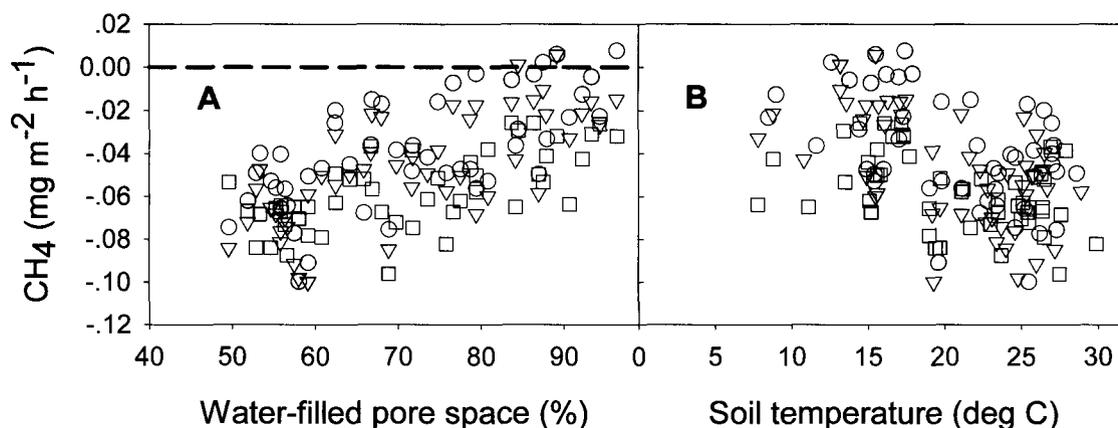


Fig. 5. CH₄ flux (mg m⁻² h⁻¹) changes with water-filled pore space (%) at 0-10 cm soil depth (A) and soil temperature (deg C) at 5 cm soil depth (B). The symbols used are circles for S, triangles for S+L and squares for S+L+P.

radicals (OH), while 5-10% is removed from the atmosphere by diffusion into aerobic topsoils where it is oxidized by soil microorganisms (Prather *et al.* 1995). CH₄ is also formed in soil under anaerobic conditions and can migrate to the surface where it is emitted into the atmosphere. Under certain conditions soil can change from a net sink to a net source of CH₄. The floor of the MF in this study was a net sink for CH₄, no matter if it was the dry season or the wet season. Keller and Reiners (1994) reported that the conversion of forest to pasture resulted in a net change from a sink to a source of CH₄ fluxes in the Atlantic lowlands of Costa Rica.

Effects of soil temperature and moisture on GHG exchange

Natural wetlands and flooded rice fields, as well as lake sediments, contribute about 40% of the total CH₄ emissions (Prather *et al.* 1995). CH₄ emissions have been found to vary considerably with the type of vegetation present, but the greatest differences in fluxes appear to be brought about by the depth of the water table (Liblik *et al.* 1997, Smith *et al.* 2003). In general, at the forest floor the variation of soil-atmosphere fluxes of CH₄ closely follows the pattern of water-filled pore space (Keller and Reiners 1994). Consumption of CH₄ at the MF floor in this study showed a negative relation to water-filled pore space across all treatments (Fig. 5A). Although this explanation relies solely on the assumption that soil moisture was lacking, we did not exclude the possibility that consumption of CH₄ in the soil was partly offset by CH₄ production as soil neared saturation. This result is consistent with the assertion that soil CH₄ consumption is limited by the effective diffusivity of CH₄ in the soil, which was modeled using Fick's first equation (Dorr *et al.* 1993). CH₄ consumption also presented degressive trends when soil temperature increased for all treatments (Fig. 5B). It was the essential reason that lower soil moisture always coincided with higher soil temperature.

Soil temperature and soil moisture explain most of the variation in soil respiration but additional factors such as forest type, soil fertility and soil texture could also be important for variations in soil respiration (Jiang 1997, Borken *et al.* 2002). Some empirical models have been developed to describe the combination of moisture and temperature dependency of soil respiration (Howard and Howard 1993, Savage and Davidson 2001, Broken *et al.* 2002). A function (2) was established between CO₂ fluxes (R) and soil temperature at a depth of 5 cm (T), where a and b are fitting parameters: a is the base respiration rate and b is related to Q₁₀, the factor by which a reaction increases for an increase of 10°C in soil temperature (Formula 3).

$$R = a \exp^{bT} \quad (2)$$

$$Q_{10} = \exp^{10b} \quad (3)$$

The values of r² were 0.68, 0.64 and 0.67 for S, S+L and S+L+P, respectively. CO₂ fluxes were fairly well described by the regression (2), which explained more than 80% of total variation in the measured CO₂ fluxes over a period of 12 months for three treatments (Fig. 6).

Q₁₀ values obtained of 2.4, 2.1 and 2.1 for S, S+L and S+L+P, respectively, which were similar to the median value of 2.4 reported in a literature review of soil respiration studies (Raich and Schlesinger 1992) and lower than Q₁₀ values of 3.9 for the ambient plot and 5.7 for a drought plot in temperate forest soil (Borken *et al.* 1999).

Soil moisture was one of the essential physical factors controlling soil respiration and the regression (2) could be normalized by soil moisture (Law *et al.* 1999). However, the Dinghushan region has a wet and a dry season, and air temperature is higher during the wet season than in the dry season, so monthly soil temperature and moisture at a 5 cm depth are positively correlated. Using normalized regression is therefore

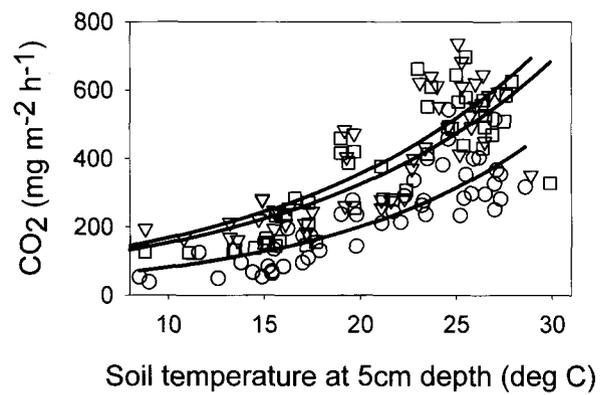


Fig. 6. Functions applied to CO₂ flux (mg m⁻² h⁻¹) changes with soil temperature at 5 cm soil depth (deg C). The symbols used are circles for S, triangles for S+L and squares for S+L+P.

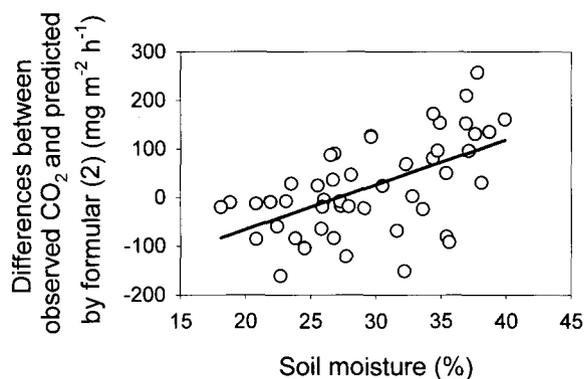


Fig. 7. Analysis of volumetric soil moisture (%) effects in terms of the residuals from the temperature function in the plot of S+L.

unlikely to lead to a significant increase in the correlation coefficient as compared with using equation (2) alone. As a test of this assumption, we examined the relation between soil moisture and the residuals from the regression (2) for S+L treatment (Fig. 7). The degree of scatter in Figure 7 supports this conclusion.

In a review of the literature, the key factors affecting N_2O emissions from the soil have been shown to be water-filled pore space, temperature, mineral N concentration, vegetation types and rainfall (Keller and Reiners 1994, Reiners *et al.* 2002, Dobbie and Smith 2001, 2003, Izaurralde *et al.* 2004). Exponential relationships were found between water-filled pore space and fluxes, and there was a significant linear relationship between soil temperature and N_2O fluxes (Dobbie and Smith 2001). Different physical factors effect N_2O emissions in different ways, so the analysis

of N_2O fluxes response to individual controlling variables has been proven to be more useful than the employment of multiple regression techniques. Clayton *et al.* (1997) could only account for 28% of the variance when using multiple variables in the regression, whereas using the individual variable of water-filled pore space or soil temperature could account for 54% or 41% of the variation of N_2O fluxes, respectively (Dobbie *et al.* 1999). Dobbie and Smith (2003) indicate that any one of the controlling variables could become limiting at different times. For example, if the soil is too dry, fluxes will be low, no matter how much N concentration is in the soil.

A single linear function (Fig. 8A) used to describe the relation of N_2O emissions to water-filled pore space and r^2 values was from 0.51 to 0.55 for all treatments, which was very close to the results reported by Dobbie

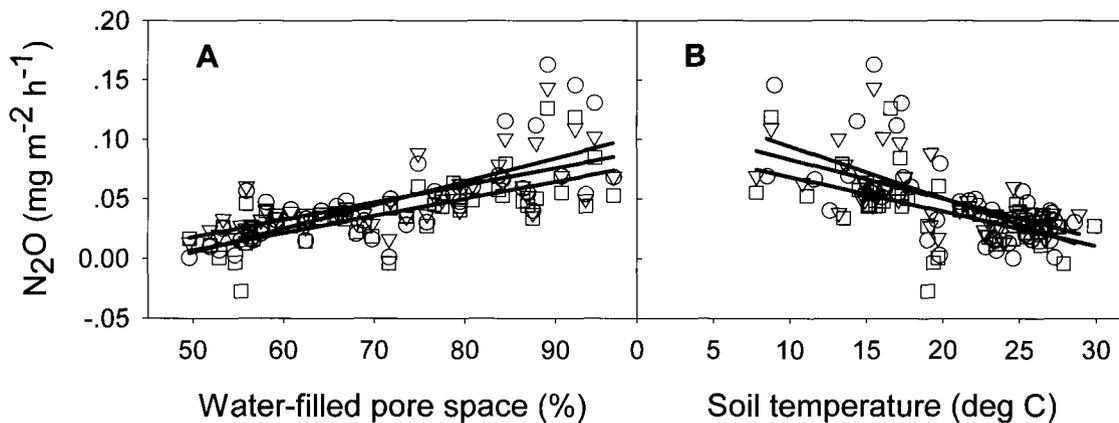


Fig. 8. Linear regressions applied to N_2O flux ($mg\ m^{-2}\ h^{-1}$) changes with water-filled pore space (%) at 0-10 cm soil depth (A) and soil temperature (deg C) at 5 cm soil depth (B). The symbols used are circles for S, triangles for S+L and squares for S+L+P.

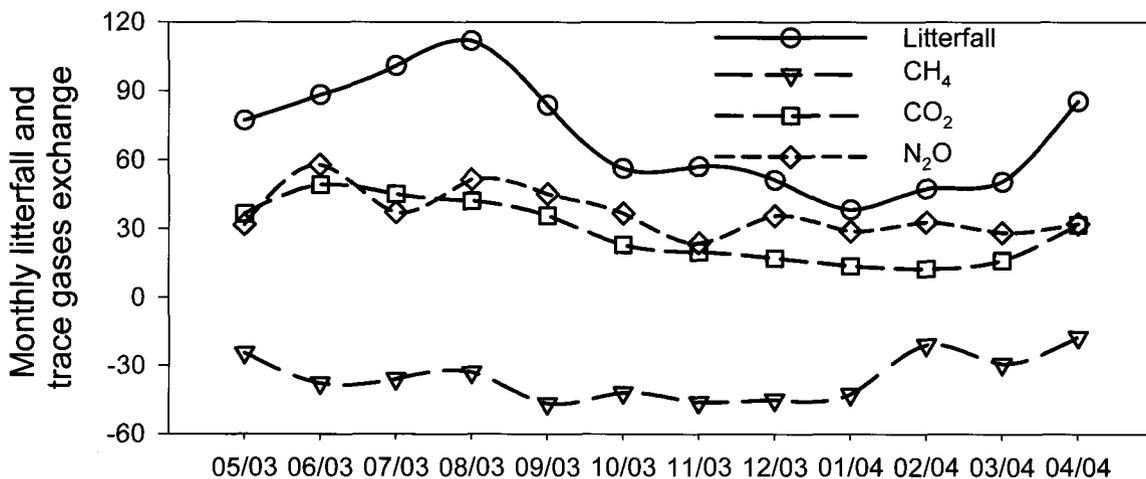


Fig. 9. Comparing the monthly dynamics of litterfall ($g\ m^{-2}$ per month), CH_4 ($mg\ m^{-2}$ per month), CO_2 ($10\ g\ m^{-2}$ per month) and N_2O ($mg\ m^{-2}$ per month). The values of CH_4 , CO_2 and N_2O derived from the plot of S+L.

et al. (1999). N₂O emissions were also strongly driven by soil water-filled pore space in the model of Conen *et al.* (2000). As soil temperature and moisture were positively correlated in the Dinghushan region, we could also describe the relation of N₂O emissions to soil temperature using linear regression (Fig. 8B). r^2 values were 0.41, 0.42 and 0.33 for S, S+L and S+L+P, respectively.

Litterfall contribution to GHG exchange

Litterfall dynamics contained lots of information including biological properties and environmental factors (Kozłowski *et al.* 1990, Hart *et al.* 1992, Finer 1996). Litterfall is, in fact, an integrated response between biological processes and the influence of environmental fluctuations (Pedersen and Hansen 1999). And thus it is important in understanding nutrient and organic matter dynamics as well as an indicator of trace gas fluxes at the forest floor. The main emphasis in the earliest litterfall studies was placed on fluctuations and litterfall composition, as well as distribution. Recent emphasis has been placed upon ecosystem analysis with litterfall playing a central part, particularly in nutrient cycling and organic matter accumulation. Comparing trace gas fluxes of S to those of S+L (fig. 1, 4), litterfall had significant effects on GHG fluxes in the forest ecosystems.

Figure 9 illustrates the monthly dynamic patterns of trace gases and litterfall. They had a similar trend as seasons changed. The relationships between litterfall and trace gases were obtained when using the lagging procedure to maximize the correlations and r^2 values were 0.56 of CH₄ (lagging 3 months), 0.87 of CO₂ (lagging 6 months) and 0.44 of N₂O (in the same month), respectively. Those results should lead us to pay more attention to litterfall effects on GHG exchange. At the same time, litterfall was also an important parameter in modeling GHG fluxes. Chen *et al.* (1999) reported CO₂ efflux rate of litter decomposition in grassland and showed in one of their results that the litter layer on the soil surface could slow down the emissions of CO₂ from soil to atmosphere. There was an inverse result at the forest floor in our study. Litter layer was not only the source of CO₂ itself but also aggravated CO₂ efflux by indirect effects on biological processes under soil.

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