Impacts of prescribed burning on soil greenhouse gas fluxes in a suburban native forest of south-eastern Queensland, Australia

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Abstract

Prescribed burning is a forest management practice that is widely used in Australia to reduce the risk of damaging wildfires. Prescribed burning can affect both carbon (C) and nitrogen (N) cycling in the forest and thereby influence the soil-atmosphere exchange of major greenhouse gases, i.e. carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). To quantify the impact of a prescribed burning (conducted on 27 May 2014) on greenhouse gas exchange and the potential controlling mechanisms, we carried out a series of field measurements before (August 2013) and after (August 2014 and November 2014) the fire. Gas exchange rates were determined in 4 replicate plots which were burned during the combustion and in another 4 adjacent unburned plots located in green islands, using a set of static chambers. Surface soil properties including temperature, pH, moisture, soil C and N pools were also determined either by in situ measurement or by analysing surface 10 cm soil samples. All of the chamber measurements indicated a net sink of atmospheric CH_4, with mean CH_4 uptake ranging from

1.15 to 1.99 mg m^{-2} d^{-1}. Prescribed burning significantly enhanced CH_4 uptake as indicated by the significant higher CH_4 uptake rates in the burned plots measured in August 2014. In the following 3 months, the CH_4 uptake rate was recovered to the pre-burning level. Mean CO_2 emission from the forest soils ranged from 2721.76 to 7113.49 mg m^{-2} d^{-1}. The effect of prescribed burning on CO_2 emission was limited within the first 3 months, as no significant difference was observed between the burned and the adjacent unburned plots in both August and November 2014. The CO_2 emissions showed more seasonal variations, rather than the effects of prescribed burning. The N_2O emission in the plots was quite low, and no significant impact of prescribed burning was observed. The changes in understory plants and litter layers, surface soil temperature, C and N substrate availability and microbial activities, following the prescribed burning, were the factors that controlled the greenhouse gas exchanges. Our results suggested that the low intensity prescribed burning would decrease soil CO_2 emission and increase CH_4 uptake, but this effect would be present within a relative short period. Only slight changes in the surface soil properties during the combustion and very limited impacts of prescribed burning on the mineral soils supported the rapid recovery of the greenhouse gas exchange rates.

1 Introduction

As a result of continuously increasing greenhouse gas emissions, global climate change studies have predicted a drier future with high probability of increasing temperatures, lower average rainfall and increase in the frequency and severity of droughts and extreme weather events (Sherwood and Fu, 2014; Fu et al., 2015). In Australia, climate changes were also identified as key drivers of the increases in days with high fire risk weather and probability of severe wildfires (Murphy and Timbal, 2008; Fest, 2013). In response to these predictions, the use of prescribed burning is increased in Australia forest management to protect both native and plantation forests from the risk of damaging wildfires (Wang et al., 2014). Prescribed burns are generally targeted at the understorey vegetation and surface litters, while aiming for minimum damage to overstorey trees. Despite the controlled burning conditions, prescribed burning can still have significant effects on soil water content and soil temperature. The combustion event would also result in amounts of charcoal and dying tree roots (Kim et al., 2011; Sullivan et al., 2011) and therefore alter root activities, organic matter decomposition, availability of substrate and soil N dynamics (Weber, 1990; Certini, 2005; Livesley et al.,

2011; Wang et al., 2014). All these parameters are closely related to three major greenhouse gas exchanges at soil-atmosphere interface, namely carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). Studies have paid special attention to these greenhouse gas fluxes, not only because of the warming effect caused by CO_2, CH_4 and N_2O in the atmosphere globally (Sherwood and Fu, 2014), but also because of their use as very effective indicators for evaluating soil C and N pools and soil microbial activities (Weber, 1990). Many studies have been conducted to quantify CO_2, CH_4 and N_2O exchanges at forest soil-atmosphere interface and the impact of intensive wildfires in different climate regions, but very few works have reported the effects of prescribed burning on soil greenhouse gas emissions, especially in Australia.

The CO_2 is the primary greenhouse gas and account for a major part of anticipated global warming (Sommerfeld et al., 1993). Within the forest ecosystem, soil is the major C reservoir while soil respiration is an important mechanism that releases the fixed C into atmosphere (Seidl et al., 2014). Forest fires are generally reported to decrease soil CO_2 efflux (Weber, 1990; Burke et al., 1997; Kim et al., 2011; Livesley et al., 2011). The key driving factors of fires on CO_2 efflux are the changes in soil temperature, moisture and fine root activities. As the altered temperature and moisture could change the amplitude of seasonal variations in CO_2 emissions, reduced fine root activities after fires are more responsible for the decreased CO_2 (Kim et al., 2011; Sullivan et al., 2011). Sullivan et al. (2011) also concluded that reduced microbial respiration indicated by microbial biomass after prescribed burning could further contribute to the decreased CO_2 efflux. Unlike with this “decrease” effect, Fest et al. (2015) also reported that low intensity burning slightly increased soil CO_2 flux in temperature eucalypt forest systems. This is attributed to the higher inputs of easily decomposable compounds, higher surface temperature and soil nutrient depletion after burning treatments (Fest et al., 2015).

The CH_4 is the second most important greenhouse gas, with a global warming potential of 25 times greater than CO_2. Aerated forest soils are one of the most important biological sink of CH_4 as the results of oxidation of atmospheric CH_4 by methanotrophic bacteria. Studies have reported both positive (Livesley et al., 2011; Sullivan et al., 2011; Fest, 2013) and no significant impacts of fires on forest soil CH_4 uptake (Kim et al., 2011). Since soil gas diffusivity is one of the key regulators of soil CH_4 uptake in the forest soil system, prescribed burning altered soil moisture condition, and removal of litter layer and soil O horizon would

weaken or eliminate the barrier effect of the surface soil and thereby increase diffusion of CH_4 into soil profile (Sullivan et al., 2011; Fest, 2013).

The N_2O emission is less reported in forest soil studies, despite the greater climate warming potential of N_2O than CO_2 and CH_4 and the large contribution of forest soil N_2O emission to the global atmospheric N_2O budget (Kiese and Butterbach-Bahl, 2002; Kiese et al., 2003). Fires would affect soil N_2O emission by altering the rates of nitrification and denitrification processes but there are only few published studies and their results are inconsistent. Fires could change forest soil N storages but the mechanism of driving N_2O emission changes is unclear (Certini, 2005; Nave et al., 2011). Available N substrate (Kiese et al., 2003), soil moisture and temperature (Fest et al., 2009), water filled pore space (Kiese and Butterbach-Bahl, 2002) and stand conditions (Butterbach-Bahl et al., 1997) are potential factors which could affect N_2O emission rates.

The eucalypt dominated natural forest ecosystem studied in this experiment is quite typical across most of Australia's forest areas, with a total of 92 million hectares or 74% of Australia's forest area (Department of Agriculture, http://www.agriculture.gov.au/abares/forestsaustralia/profiles/eucalypt-forest). These forests hold important C storage over the country and also provide important ecosystem services such as biodiversity, recreation, water resource and wood products (Fest et al., 2009). Prescribed burning is one of the most important management tools in Australia to protect these forests from fires and maintain their functions, including forest regeneration, site preparation, fuel reduction and habitat management (Guinto et al., 2000; Bai et al., 2012; Wang et al., 2014). **S**ome studies have reported the greenhouse gas emissions from Australian forest soils (Kiese and Butterbach-Bahl, 2002; Dalal et al., 2003), but the impact of prescribed burning on Australian eucalyptus forests are rarely studied. Therefore, we have limited understanding about the magnitude and direction of the effect of prescribed burning on the greenhouse gas exchange which is critical to understand the interaction between burned ecosystem and the atmosphere. In this study, we setup four sampling plots which had similar stand conditions to address the following questions: (1) would prescribed burning affect greenhouse gas emissions at the soil atmosphere interface? (2) How long would these effects last? And (3) what would be the controlling factors? To address these questions, we conducted a series of field measurements of CH_4, CO_2 and N_2O exchange at surface soil before and after prescribed burning. To examine the potential driving factors and mechanisms

we also collected surface soil samples for analysing biological, chemical and physical variables which might be altered by prescribed burning.

2 Methods

2.1 Site description

The study was carried out in Toohey Forest (27°30′S, 135°02′E), located 10 km south of Brisbane in south-eastern Queensland, Australia (Figure 1). This forest accounts for about 600 hectares dominated by different species of eucalypt and contains some 460 species of vascular plants. About half of this forest is a local government conservation reserve and surrounded by suburban areas (Catterall and Wallace, 1987; Catterall et al., 2001; Farmer et al., 2004). The climate for the region and around is characterized as subtropical with a dominant weather pattern of hot, wet summers and cool, dry winters. The mean annual rainfall is about 1000 mm, with the majority received during summer months. Temperatures generally ranged between approximately 30 and 35 ℃ in summer and 10 and 15 ℃ in winter.

Patterns of burning prior to the 1950s are unknown, but from the 1950s to the early 1970s, individual fires probably burned across a large proportion of the area. From the early 1970s individual fires were confined to more localized areas and created a spatially heterogeneous fire regime. Since 1993, 27 blocks within Toohey Forest have been conducted with regular prescribed burnings (Wang et al., 2015). Prescribed burnings are generally low intensity cool burns, and usually occur at the end of the dry season in winter. Generally, every burning event would last for several hours (usually overnight) from ignition to extinguishing of any small fires. The fire was monitored and controlled during the burning.

The prescribed burning of this study was conducted on 27 May 2014. Last recorded burning in this block was in 18 June1999. Before the prescribed burning, we selected 4 plots (around 30 m between each other, Figure 1) with similar stand conditions for sampling. The understory of these plots was burned out during the recent prescribed burning, left a layer of wood charcoal on the ground. After the burning, these 4 plots were measured repeatedly at three months (August 2014) and six months (November 2014). Detailed sampling dates and weather conditions for the selected sampling events were listed in Table 1. Briefly, the sampling events were conducted under clear weather condition and there were no major precipitation events either 30 days or 90 days before the sampling events. Furthermore,

another 4 unburned plots adjacent to the burned plots as paired plots were selected to further examine the impacts of prescribed burning. These unburned plots were located in the adjacent green islands of those not touched by the recent prescribed burning.

(Figure 1)

2.2 Sampling method

As previously described, 4 plots was selected for sampling. We treated these 4 plots as replicates and a series of experiments, including impacts of burning on soil greenhouse gas emission, N transformation and litter decomposition, were carried out in each plot, a total of 11 chambers (rings) were established for specific purposes through introducing different treatments (e.g. water or solution application, N isotope enrichment), only the chamber for soil gas sampling was left untreated and the results were presented here.

We adopted a static chamber method to measure CO_2, CH_4 and N_2O emissions simultaneously. A PVC chamber (cylinder with a diameter of 245 mm, wall thickness approximately 4 mm and length 300 mm) was installed in each of the 4 plots. The chambers were sharpened at the bottom edge and were pushed 10 cm into the soil using a hammer. Weed control was performed within and around the chamber to prevent the impacts of grasses. Sampling was conducted a week later after the chambers were set. Gas fluxes across the soil-atmosphere interface were determined daily by sampling air in the headspace of PVC chambers during the 4-day field measurement. The internal volume of a chamber was 9.4 L when placed 10 cm deep into forest floor. There were 4 holes evenly distributed on the chamber (10 cm above ground) to help the inner atmosphere fully mixed with outside, on all sampling occasions prior to each gas sampling, these holes were sealed with a set of rubber plugs. The top of the chamber was also covered with a cap fastened with black rubber band to prevent any gas exchange between the inner headspace and the outside. After covering the cap, 15 ml gas samples were taken from the sampling port at the centre of the chamber top at 0 and 60 min after chamber deployment. A 25 ml syringe was attached to the sampling port and the plunger of the syringe was pumped up and down several times to mix the gases in the chamber before taking a sample. Gas samples were immediately injected into pre-evacuated 15-ml tubes capped with butyl rubber stoppers and prepare for analysis of CO_2, CH_4 and N_2O. Gas sampling was conducted for 4 consecutive days to capture consistent emission patterns from the 4 plots and the data was pooled for statistical analysis (Lennon, 2011).

All gas samplings were conducted during daylight hours from approximately 8:00 to 12:00. However, considering of the great spatial heterogeneity of soil gas fluxes reported by existing studies (Kim et al., 2011; Sullivan et al., 2011), the acquired results in the present study might be tangled with burning induced impacts and spatial related variabilities due to the relative large distance among the plots (Prosser, 2010). We considered the similarity of soil and stand conditions (eucalypt forest with moderate density) and plot location (middle slope) when selecting the plots, which would partly offset this potential impact.

2.3 Soil property analysis

The top 10 cm soil in the chamber was collected with a shovel. Collected soil samples were thoroughly mixed and passed through a 2 mm sieve. Soil moisture was measured gravimetrically after drying at 105 ℃ for 48 hours. pH was measured with a 1:5 aqueous solutions after shaking for 30 min. Soil inorganic N concentrations were extracted with 2 *M* KCL and measured using a modified micro-diffusion method (Wang et al., 2015) and a Discrete Chemistry Analyser (Westco Smartchem SC 200, Discrete Wet Chemistry Analyser).

To determine water soluble organic C (WSOC) and total N (WSTN), 7 g fresh soil was added to 35 ml distilled water in a 50 ml plastic centrifuge vial, the suspension was then shaken by an end-over-end shaker for 5 min followed by centrifuging at 10000 rpm for 10 min. The suspension was then filtered through a Whatman 42 filter paper and a 33 mm Millex syringe-driven 0.45 μm filter successively before analysed with a Shimadzu TOC-VCSH/CSN TOC/N analyser. Similarly, hot water extractable organic C (HWEOC) and total N (HWETN) were also measured, while the only difference was, 1:5 soil water solution, was incubated in a capped and sealed tube at 70 ℃ for 18 hours.

Soil microbial biomass C (MBC) and N (MBN) were determined using the fumigation-extraction method described by Vance et al. (1987) and Brookes et al.(1985). Briefly, fumigated and non-fumigated soils (5 g dry weight equivalent) were extracted with 25 ml of 0.5 *M* K_2SO_4 (soil/extractant ratio 1:5). The fumigation lasted for 16 hours. Samples were shaken for 30 minutes and then filtered through a Whatman 42 filter paper. Soluble organic C and total soluble N (TSN) in the fumigated and non-fumigated samples were determined using a Shimadzu TOC-VCSH/CSN TOC/N analyser. MBC and MBN were calculated using conversion factors of 2.64 and 2.22 for C (Vance et al., 1987) and N (Brookes et al., 1985), respectively.

2.4 Gas analysis

Collected gas samples were sent to laboratory for gas chromatography (GC) analysis for CO_2, CH_4 and N_2O concentrations shortly after the field sampling. The concentrations of CO_2 and CH_4 were measured using a GC system (GC-2010 PLUS Shimadzu) with Flame Ionization Detector and N_2O was measured using the same GC system with Electron Capture Detector. The standards (0.5 ppm for CH_4, 400.5 ppm for CO_2 and 0.3 ppm for N_2O) were run before and after each set of samples to ensure the reproducibility of measurements. Gas fluxes for CO_2, CH_4 and N_2O were determined from a regression analysis with gas concentration change within the chamber versus time.

2.5 Statistical analysis

All statistical analyses were performed using IBM SPSS STATISTICS (version 20) software. One-way ANOVA was introduced to examine statistically significant differences between soil gas fluxes measured before and after the burning in the burned plots. This analysis was also applied to compare the fluxes between burned and unburned plots in Aug 2014 and Nov 2014, respectively. Collected soil properties and gas fluxes at the four replicate plots during the three sampling events were also pooled together for Pearson correlation analysis to detect possible effects of soil environmental variables on soil CO_2, CH_4 and N_2O fluxes.

3 Results

3.1 Greenhouse gas exchange rates before and after prescribed burning

Average CH_4, CO_2 and N_2O emissions rates of the 4 replicate plots for each sampling event were listed in Table 2. While temporal patterns of gas exchange for the 4-day sampling of the 3 sampling periods were shown in Figure 2. Coefficient of variance (CV, ratio of the standard deviation to the mean) among the plots during the 4 sampling days ranged from 14%-68% (mean 32%), 9%-15% (mean 10%) and 10%-28% (mean 16%), for Aug 2013, Aug 2014 and Nov 2014, respectively. All the sampling plots showed negative CH_4 emissions rates during the three sampling events, or uptake atmospheric CH_4. In the burned plots, mean CH_4 uptake was significantly increased by 64% three months after the prescribed burning ($p < 0.001$), while during the third sampling period, CH_4 uptake rate became similar to that before the burning ($p = 0.843$). In the unburned plots, CH_4 uptake was relatively stable during the dates

of the sampling periods and also showed less variation between August 2014 and November 2014. The significant difference in mean CH_4 uptake rate in August 2014 ($p < 0.001$) but similar rates in November 2014 ($p = 0.921$) also confirmed that the CH_4 uptake increased at the first three months but was recovered to the pre-burning level about six months after prescribed burning.

Soil CO_2 flux showed relative higher variance as indicated by the higher standard deviations (Figure 2) and CVs (ranged from 43% to 50% during the three sampling periods). Mean CO_2 emission from all burned plots was significantly decreased by 41% in August 2014 ($p < 0.001$). In November 2014, CO_2 efflux rates had exceeded that before the burning by 28% but the difference was not significant ($p = 0.392$). Similar CO_2 emission rates between the burned and adjacent unburned plots during the sampling dates of August 2014 ($p = 0.549$) and November 2014 ($p = 0.218$) were also observed. This might indicated that the temporal dynamics detected at the burned plots reflected more natural variations rather than burning induced impacts.

The lower N_2O emission rates, compared to that in August 2013, were found both in August 2014 ($p = 0.003$) and November 2014 ($p < 0.001$). During the three sampling periods, the study plots were not solely performed as a source of atmospheric N_2O, on 27 August 2013, 6 August 2014 and most days of November 2014, but the plots also took up N_2O from the atmosphere. No significant difference in N_2O emission was observed between the burned and unburned plots in both August and November 2014.

(Table 2)

(Figure 2)

3.2 Soil basic properties and their relationships with gas exchange rates

After the prescribed burning, soil moisture showed no significant difference between burned and unburned plots ($p = 0.804$), although most of the sampling plots (5 out of 8 for the two sampling events in 2014) had relative higher values. Soil temperature was slightly higher for most sampling dates at in the burned plots, but no significant difference was found in August 2014 ($p = 0.644$) and November 2014 ($p = 0.751$). The pH in the surface soil was higher in 2014 than in 2013, and the values in all burned plots were slightly higher than those of unburned sites ($p = 0.293$). NO_3-N was quite low both before and after the prescribed burning, but NH_4-N was significantly increased after the prescribed burning.

When relating these soil parameters to greenhouse gas emissions from the soil surface, soil temperature showed a positive correlation with the CH_4 uptake ($R = 0.232$, $p = 0.044$) and CO_2 efflux ($R = 0.47$, $p < 0.000$) and a negative correlation with N_2O emission ($R = -0.284$, $p = 0.011$). pH was negatively correlated with the CH_4 uptake ($R = -0.595$, $P = 0.006$) and CO_2 ($R = -0.591$, $p = 0.006$) emission. NH_4-N was negatively correlated with N_2O emission ($R = -0.533$, $p = 0.015$).

(Table 3)

3.3 Soil C and N dynamics before and after burning

There was no significant difference in WSOC in the burned plots between August 2013 and August 2014, and only slightly decreased WSOC was observed in August 2014. However, WSOC was significant higher in November 2014 ($p = 0.034$). No significant difference was found between any sampling periods for WSTN, despite higher WSTN in some burned plots than those before the prescribed burning and in the unburned plots. HWEOC was significantly increased in August 2014 than that in August 2013 ($p < 0.001$) and in November 2014 it was recovered to the level before the prescribed burning ($p = 0.929$). The difference in HWEOC between the burned and adjacent unburned plots were also significant in August 2014 ($p = 0.0361$) but insignificant in November 2014. The situation was similar for HWETN.

MBC in the burned plots in August 2014 was 378.94 mg kg^{-1}, which was lower than that in August 2013 (522.45 mg kg^{-1}, $p = 0.069$), and this value did not change much in November 2014 (380.37 mg kg^{-1}).

The correlation analysis between soil C or N pools and gas emissions showed that CH_4 uptake was negatively correlated with WSOC ($R = 0.523$, $p = 0.018$). CO_2 efflux had negative correlation with HWEOC ($R = -0.690$, $p = 0.001$) and HWETN ($R = -0.730$, $p < 0.001$). N_2O emission was positively correlated with MBN ($R = 0.565$, $p = 0.009$).

4 Discussions

4.1 Impacts of prescribed burning on soil properties

Prescribed burning resulted in a slightly increase in the surface soil temperature of this study, which is in the agreement with most of the literature (Burke et al., 1997; Certini, 2005). The burning of the understory vegetation cover, together with the consumption of fuel loads,

removal of litter layer and increased charred materials on the soil surface would all affect soil temperatures by intercepting direct sunlight and moderating the loss of soil heat by radiation. However, the controlled burning condition or low fire intensity limited this difference at an insignificant level. Meanwhile, the 4 selected plots in the study did not show consistent fluctuations in the surface soil moisture before and after the prescribed burning, which would generally be expected to decrease after a fire (Burke et al., 1997; Kim et al., 2011; Sullivan et al., 2011). Generally, increased soil temperature, combined with the reduced shade, would result in higher evaporation rates and therefore restrict the movement of water into soil profile (Burke et al., 1997; Certini, 2005). This be might attributed to the representativeness of the measurements and one measurement for each sampling period might not be adequate to represent the physical state of water in the soil (Weber, 1990).

An increase in soil pH values was found in the burned plots in August 2014 and it was returned to a similar level in November 2014. Although no significant difference was found between the burned and adjacent unburned plots in 2014, pH values for the burned plots were still higher than those at the unburned plots. The increased pH after the prescribed burning would be probably due to the release of extractable basic cations from the deposited ashes during the burning. Several studies also reported increased pH after the fire (Guinto et al., 1999; Certini, 2005; Kim et al., 2011; Xue et al., 2014) and the increased pH would either be recovered to the pre-burning level within a year (Rhoades et al., 2004; Xue et al., 2014) or last for longer periods (Arocena and Opio, 2003; Ponder Jr et al., 2009; Granged et al., 2011), depending on the site condition and burning intensities.

NH_4-N was significantly increased after the prescribed burning, but no significant changes were observed for NO_3-N, since NH_4-N was a direct product of combustion and NO_3^- was formed from NH_4^+ some weeks or months later as a result of nitrification (Covington and Sackett, 1992; Diaz-Raviña et al., 1992; Wang et al., 2015). Hence, the increase in NH_4-N was probably due to the transformation of organic N during the combustion. Also the deposition of organic N in ash and enhanced ammonification would also contribute to the increased NH_4^+ (Knoepp and Swank, 1993; Wan et al., 2001). This was also supported by the similar findings of Nardoto and Bustamante (2003) in savannas of Central Brazil and Covington and Sackett (1992) in a ponderosa pine forest in USA.

The prescribed burning significantly reduced MBC in the surface soil and it showed no apparent sign of recovery six months after the prescribed burning. Decreased MBC after

prescribed burning or wildfires have been reported and it would normally last for several years (Prieto-Fernández et al., 1998). As for the other two soil liable organic C pool indicators, WSOC showed no significant change before and after burning while HWEOC was significantly increased in August 2014 and returned to the pre-burning level in November 2014. While the low intensity of prescribed burning may only cause volatilization of organic C to a limited extent, soil microbes might be decreased due to their sensitiveness to temperature (Hernández et al., 1997; Neary et al., 1999). This microbial lysis, as well as the heat-induced alterations of soil organic matter, contributed to the release of carbohydrates which were reflected by the initial increase in HWEOC.

4.2 Variations in greenhouse gas exchanges and their driving factors

4.2.1 CH_4 uptake

The CH_4 uptake rates before prescribed burning and six months after the burning from the burned plots and all fluxes from the unburned plots fall in the range of CH_4 fluxes reported by Kiese et al. (2003) (from 0.84-1.63 mg m^{-2} d^{-1}) and a recent study by Rowlings et al. (2012) which were conducted in a similar forest ecosystem in Australia. While the high uptake rate of CH_4 three months after the burning was also comparable to the results obtained in Australia forests under extreme dry conditions (Fest et al., 2009; Rowlings et al., 2012). The prescribed burning increased CH_4 uptake in this study. The same effect has also been reported by Burke et al. (1997) and Sullivan et al. (2011). However, unlike most studies reporting that the enhanced CH_4 uptake may last for several years, our results indicated that CH_4 uptake rate was returned to the pre-burning level within six months after the prescribed burning. We obtained this conclusion from the similar CH_4 uptake rates in November 2014 when compared to the CH_4 uptake in the unburned plots and the rates before the burning in the burned plots. The low fire intensity of the prescribed burning in this study might cause less impact on the system and therefore shorten the required time to recover to the pre-burning conditions. Studies have found that fire intensity has significant effect on forest soil CH_4 consumption and CO_2 emissions while severe wildfires always impact gas exchange rates for the subsequent several years (Burke et al., 1997; Neary et al., 1999; Sullivan et al., 2011). Kim et al. (2011) also found a quick recovery of CH_4 uptake that after 2 years of low intensity burnings in a Japanese forest.

Soil moisture has been shown to be a key parameter controlling CH_4 consumption by the soils through limiting the transport of atmospheric CH_4 to microbial communities living at depth in the soil (Burke et al., 1997). However, we observed no significant relationships between soil moisture and CH_4 uptake as reported by other studies (Sommerfeld et al., 1993; Kiese et al., 2003; Livesley et al., 2011). This was probably due to the low intensity burning, and hence the prescribed burning did not affect soil water conditions in the soil horizons relevant to the CH_4 oxidation, or the soil moisture was partly recovered to the pre-burning level. Castro et al. (1994) found that moisture control was strongest when between 60% and 100% of available soil pore space was water filled. Meanwhile, surface soil temperature appeared to show more significant influence on CH_4 uptake in this study. We also found weak but significant relationship between CH_4 uptake and soil pH. The mechanism of how increased pH would affect soil CH_4 uptake after fire is not clear, and Jaatinen et al. (2004) estimated that the increased pH after fire caused any change in the methanotroph community and would not be directly responsible for the increased uptake rates. Therefore, the increased pH in our study would probably indirectly affect CH_4 uptake together with other fire introduced changes.

The relative high and significant correlation between CH_4 uptake and WSOC indicates that the decreased soil C might have increased CH_4 diffusion into the soil profile. Removal of the C rich O horizon caused by the prescribed burning eliminated a barrier for CH_4 diffusion. This is also supported by the decreasing CH_4 uptake and recovered WSOC in November 2014, combined with recovered litter deposit and ground plant regrowth. This effect of prescribed burning reduced thickness of organic layer to CH_4 uptake was also found in similar forest ecosystems (Saari et al., 1998; Steinkamp et al., 2001). Another possible reason for the observed characteristics of CH_4 uptake is the physical changes in the surface soil. Although not measured in the current study, literature has shown that low to moderate fires would increase soil structure stability due to the formation of the hydrophobic film on the external surface of aggregates (Mataix-Solera and Doerr, 2004). With lack of the protection of ground plants and litter layers, surface soil was more likely to loose more fine fractions and lead to soil coarsening by the increased erosion (Certini, 2005). These physical changes in the surface soil would all form channels ideal for diffusion of atmosphere CH_4 into the soil profile and thereby increase CH_4 oxidation rates.

4.2.2 CO_2 effluxes

The studied plots acted as a persistent source of atmospheric CO_2 before and after the prescribed burning, while the CO_2 emission rates, either before or after the burning, were similar to the results obtained by Carlyle and Than (1988) in a native forest with low soil moisture (about 5520 mg m^{-2} d^{-1}) and by Rowlings et al. (2012) in an Australian subtropical rainforest (around 3600 mg m^{-2} d^{-1}). However, the CO_2 emission values were much lower than the reported high soil respirations (over 20000 mg m^{-2} d^{-1}) in various Australian forest ecosystems with high soil moisture and temperature (Carlyle and Than, 1988; Fest et al., 2009; Rowlings et al., 2012). A number of studies have reported reduced CO_2 emissions after the fire due to diminished root activity and lower root respirations (Czimczik et al., 2006; Kim et al., 2011; Sullivan et al., 2011), and the effects would last for various length from less than 2 years (Weber, 1990; Irvine et al., 2007) to longer periods (Burke et al., 1997). However, in this study, it seems that CO_2 emission started recovering three months after the burning. This is supported by the similar CO_2 emission rates in August 2014 ($p = 0.218$) and November 2014 ($p = 0.549$) between the burned and adjacent unburned plots. The decreased CO_2 flux three months after the burning, which is only 41% of that before burning, might be attributed to the reduced root respiration and decomposition activities related to soil microbial communities. The observed high HWEOC value in August 2014 probably indicated increased dead fine roots after the combustion of ground biomass, which was in agreement with the estimation that after removing the aboveground biomass, most fine roots would die within 2 months by Fahey and Arthur (1994). Meanwhile, studies have reported that root respiration could contribute up to 50% of the total soil respiration (Irvine and Law, 2002; Zerva and Mencuccini, 2005). Decomposition of the dead fine roots could lead to the flush of C substrate but this was limited by the decreased microbial activities in this study, suggested by the significant lower MBC and MBN values in August and November 2014. Meanwhile, lower MBC in the burned plots compared to the unburned plots probably also suggested a reduced heterotrophic contribution to total soil respiration. Similar finding was also reported by Sullivan et al. (2011) who concluded that MBC explained a large proportion of the variation in soil CO_2 flux at the burned site than at the unburned sites in a ponderosa forest in south-western USA. Considering the positive relationship between CO_2 efflux and WSOC, which was an important part of soil labile C, microbial biomass was likely to be limited by the amount of labile C available for assimilation into microbial biomass. Maheswaran and Attiwill (1989) and Zerva and Mencuccini (2005) also reported reduced CO_2 emission which was related to reduced microbial populations limited by an available source of C after the fire.

Such explanation can be further supported by the subsequent recovery of CO_2 flux six months after the burning, combined with recovered MBC to the near pre-burning level and higher WSOC levels than before the burning. It was also reasonable that higher temperature in November 2014 had stimulated the surface soil respiration and therefore could contribute to the higher CO_2 emission rate.

4.2.3 N_2O emissions

The soil-atmosphere fluxes of N_2O measured in the study were very small (-0.21 to 0.54 mg m^{-2} d^{-1} before the burning and -0.18-0.11 mg m^{-2} d^{-1} after the burning). These low fluxes were similar to the small N_2O emission reported by Fest et al. (2009) and Livesley et al. (2011), but was much lower than the range of 0.75-8.19 mg m^{-2} d^{-1} recorded by Kiese et al. (2003) in a tropical rainforest and the range of 0.62-1.57 mg m^{-2} d^{-1} by Rowlings et al. (2012) in a subtropical rainforest. No significant effect of prescribed burning was observed on the N_2O emission. Since forest soils were generally accepted as a source of atmospheric N_2O (Butterbach-Bahl et al., 1997), the negative values we measured might be attributed to the changes in N_2O concentration were quite low during the chamber employment, and these changes were below the detection limit of the GC system. Even though the dry and well aerated soil of the sampled plots would make it prone to nitrification rather than denitrification, the observed small inorganic N pool (NH_4^+ < than 10 mg N kg^{-1} while NO_3^- < 0.1mg N kg^{-1}) dominated by NH_4^+ also limited the nitrification processes. Although there was a significant increase in NH_4^+ three months ($p = 0.009$) and six months ($p = 0.009$) after the burning, nitrification was still negligible. This could be attributed to low soil water availability and dry conditions after prescribed burning, since the removal of understory plants and litter layers and increased evapotranspiration rates would limit the activities of soil nitrifiers (Livesley et al., 2011). Also the prescribed burning induced charcoal at the soil surface which would also supress N_2O exchange rates as reported in a recent controlled experiment in Japan (Kim et al., 2011). However, accumulated N substrate, either NH_4^+ or NO_3^-, might cause further high N_2O emissions under appropriate conditions, for example, wet after precipitations.

4.3 Contribution of the gas emissions to the burning introduced greenhouse gas effect

Although consistently consuming atmospheric CH_4, the forest soil in Toohey Forest still acted as a net C source to the atmosphere, due to the greater CO_2 emission rates during the studied period. However, the burning induced lower CO_2 emission and higher CH_4 uptake rates could significantly reduce the amount of C released into atmosphere, especially when extending these effects to the first several months after the burning. This reduced C emission could partly compensate the greenhouse gas effect during the operation of the burning - prescribed burning could cause eruption of CO_2 into the atmosphere by combusting photosynthetic fixed C embedded in understory plants, litter layers, surface soil organic C and also the consumption of fossil fuels. Data on the C loss due to prescribed burning, fuel consumed and continuous measurement of soil gas exchanges are required to quantify the burning-caused greenhouse effect in future studies.

5 Conclusion

The low intensity prescribed burning in Toohey Forest caused changes in both soil properties and greenhouse gas exchange rates. Soil CH_4 uptake was significantly enhanced due to the increased CH_4 diffusivity into the soil profiles. The removal of litter layer and surface soil organic materials and the altered soil physical structural caused by the prescribed burning were the major factors contributing to the increased CH_4 diffusion. The CO_2 emission was largely decreased but it was largely caused by natural annual variations. Changes in root respiration and soil microbial community were the two controlling factors related to the effect of prescribed burning on CO_2 emission. Due to the controlled condition of prescribed burning, both CH_4 uptake and CO_2 emission started to recover about three months after the burning and it would appear that the gas exchange rates were recovered to the pre-burning level about six months after the burning. This rapid recovery was closely related to the limited effect of prescribed burning on the soil. However, the decreased CO_2 emission and increased CH_4 uptake during this period could still partly compensate the greenhouse gas effect caused by the combustion of C during the burning. The N_2O emission was quite low in the studied plots and showed no obvious impacts of prescribed burning.

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References

Arocena, J. M., and Opio, C.: Prescribed fire-induced changes in properties of sub-boreal forest soils, Geoderma, 113, 1-16, 2003.

Bai, H. S., Sun, F., Xu, Z., Blumfield, T., Chen, C., and Wild, C.: Appraisal of ^{15}N enrichment and ^{15}N natural abundance methods for estimating N_2 fixation by understorey Acacia leiocalyx and A. disparimma in a native forest of subtropical Australia, J Soils Sediments, 12, 653-662, 2012.

Brookes, P. C., Landman, A., Pruden, G., and Jenkinson, D. S.: Chloroform fumigation and the release of soil nitrogen: a rapid direct extraction method to measure microbial biomass nitrogen in soil., Soil Biol Biochem, 17, 837-842, 1985.

Burke, R. A., Zepp, R. G., Tarr, M. A., Miller, W. L., and Stocks, B. J.: Effect of fire on soil-atmosphere exchange of methane and carbon dioxide in Canadian boreal forest sites, J Geophys Res-Atmos, 102, 29289-29300, 1997.

Butterbach-Bahl, K., Gasche, R., Breuer, L., and Papen, H.: Fluxes of NO and N_2O from temperate forest soils: Impact of forest type, N deposition and of liming on the NO and N_2O emissions, Nutr Cycl Agroecosys, 48, 79-90, 1997.

Carlyle, J. C., and Than, U. B.: Abiotic controls of soil respiration beneath an eighteen-year-old pinus radiata stand in south-eastern Australia, J Ecol, 76, 654-662, 1988.

Castro, M. S., Melillo, J. M., Steudler, P. A., and Chapman, J. W.: Soil moisture as a predictor of methane uptake by temperate forest soils, Can J Forest Res, 24, 1805-1810, 1994.

Catterall, C. P., and Wallace, C. J.: An Island in Suburbia: the Natural and Social History of Toohey Forest., Institute of Applied Environmental Research, Griffith University., Brisbane, 1987.

Catterall, C. P., Piper, S. D., Bunn, S. E., and Arthur, J. M.: Flora and fauna assemblages vary with local topography in a subtropical eucalypt forest, Austral Ecol, 26, 56-69, 2001.

Certini, G.: Effects of fire on properties of forest soils: a review, Oecologia, 143, 1-10, 2005.

Covington, W. W., and Sackett, S. S.: Soil mineral nitrogen changes following prescribed burning in ponderosa pine, Forest Ecol Manag, 54, 175-191, 1992.

Czimczik, C. I., Trumbore, S. E., Carbone, M. S., and Winston, G. C.: Changing sources of soil respiration with time since fire in a boreal forest, Glob Change Biol, 12, 957-971, 2006.

Dalal, R. C., Wang, W., Robertson, G. P., and Parton, W. J.: Nitrous oxide emission from Australian agricultural lands and mitigation options: a review, Aust J Soil Res, 41, 165-195, 2003.

Diaz-Raviña, M., Prieto, A., Acea, M. J., and Carballas, T.: Fumigation-extraction method to estimate microbial biomass in heated soils, Soil Biol Biochem, 24, 259-264, 1992.

Fahey, T. J., and Arthur, M. A.: Further-studies of root decomposition following harvest of a northern hardwoods forest, For Sci, 40, 618-629, 1994.

Farmer, D., Catterall, C. P., and Piper, S. D.: Abundance patterns across months and locations, and their differences between migrant and resident landbirds in lowland subtropical eucalypt forest, Emu, 104, 283-296, 2004.

Fest, B.: The impact of fire disturbance and simulated climate change conditions on soil methane exchange in eucalypt forests of south-eastern Australia, Department of Forest and Ecosystem Science, Melbourne School of Land and Environment, University of Melbourne, 2013.

Fest, B. J., Livesley, S. J., Drösler, M., van Gorsel, E., and Arndt, S. K.: Soil–atmosphere greenhouse gas exchange in a cool, temperate Eucalyptus delegatensis forest in south-eastern Australia, Agr Forest Meteorol, 149, 393-406, 2009.

Fest, B. J., Livesley, S. J., Fischer, J. C., and Arnadt, S. K. Repeated fuel reduction burns have little long-term impact on soil greenhouse gas exchange in a dry sclerophyll eucalypt forest. Agr Forest Meteorol, 201, 17-25, 2015.

Fu, L., Zhao, Y., Xu, Z. H., and Wu, B. F.: spatial and temporal dynamics of forest aboveground carbon stocks in rsponse to climate and environmental changes, J Soils Sediments, 15, 249-259, 2015.

Granged, A. J. P., Jordán, A., Zavala, L. M., Muñoz-Rojas, M., and Mataix-Solera, J.: Short-term effects of experimental fire for a soil under eucalyptus forest (SE Australia), Geoderma, 167–168, 125-134, 2011.

Guinto, D. F., Xu, Z. H., Saffigna, P. G., House, A. P. N., and Perera, M. C. S.: Soil nitrogen mineralisation and organic matter composition revealed by ^{13}C NMR spectroscopy under repeated prescribed burning in eucalypt forests of south-east Queensland, Aust J Soil Ress, 37, 123-136, 1999.

Guinto, D. F., Xu, Z., House, A. P. N., and Saffigna, P. G.: Assessment of N_2 fixation by understorey acacias in recurrently burnt eucalypt forests of subtropical Australia using ^{15}N isotope dilution techniques, Can J Forest Res, 30, 112-121, 2000.

Hernández, T., García, C., and Reinhardt, I.: Short-term effect of wildfire on the chemical, biochemical and microbiological properties of Mediterranean pine forest soils, Biol Fert Soils, 25, 109-116, 1997.

Irvine, J., and Law, B. E.: Contrasting soil respiration in young and old-growth ponderosa pine forests, Glob Change Biol, 8, 1183-1194, 2002.

Irvine, J., Law, B. E., and Hibbard, K. A.: Postfire carbon pools and fluxes in semiarid ponderosa pine in Central Oregon, Glob Change Biol, 13, 1748-1760, 2007.

Jaatinen, K., Knief, C., Dunfield, P. F., Yrjälä, K., and Fritze, H.: Methanotrophic bacteria in boreal forest soil after fire, FEMS Microbiol Ecol, 50, 195-202, 2004.

Kiese, R., and Butterbach-Bahl, K.: N_2O and CO_2 emissions from three different tropical forest sites in the wet tropics of Queensland, Australia, Soil Biol Biochem, 34, 975-987, 2002.

Kiese, R., Hewett, B., Graham, A., and Butterbach-Bahl, K.: Seasonal variability of N_2O emissions and CH_4 uptake by tropical rainforest soils of Queensland, Australia, Glob Biogeochem Cy, 17, n/a-n/a, 2003.

Kim, Y. S., Makoto, K., Takakai, F., Shibata, H., Satomura, T., Takagi, K., Hatano, R., and Koike, T.: Greenhouse gas emissions after a prescribed fire in white birch-dwarf bamboo stands in northern Japan, focusing on the role of charcoal, Eur J Forest Res, 130, 1031-1044, 2011.

Knoepp, J. D., and Swank, W. T.: Site preparation burning to improve southern Appalachian pine-hardwood stands: nitrogen responses in soil, soil water, and streams, Can J Forest Res, 23, 2263-2270, 1993.

Lennon, J. T.: Replicatoin, lies and lesser-known truths regardsing experimental design in environmental microbiology, Environ. Microbiol, 13, 1383-1386, 2011.

Livesley, S. J., Grover, S., Hutley, L. B., Jamali, H., Butterbach-Bahl, K., Fest, B., Beringer, J., and Arndt, S. K.: Seasonal variation and fire effects on CH_4, N_2O and CO_2 exchange in savanna soils of northern Australia, Agr Forest Meteorol, 151, 1440-1452, 2011.

Maheswaran, J., and Attiwill, P. M.: Soil respiration in eucalypt forests of southeastern Australia, Biol Fert Soils 8, 154-159, 1989.

Mataix-Solera, J., and Doerr, S. H.: Hydrophobicity and aggregate stability in calcareous topsoils from fire-affected pine forests in southeastern Spain, Geoderma, 118, 77-88, 2004.

Murphy, B. F., and Timbal, B.: A review of recent climate variability and climate change in southeastern Australia, Int J Climatol, 28, 859-879, 2008.

Nardoto, G. B., and Bustamante, M. M. C.: Effects of fire on soil nitrogen dynamics and microbial biomass in savannas of Central Brazil, Pesqui Agropecu Bras, 38, 955-962, 2003.

Nave, L. E., Vance, E. D., Swanston, C. W., and Curtis, P. S.: Fire effects on temperate forest soil C and N storage, Ecol Appl, 21, 1189-1201, 2011.

Neary, D. G., Klopatek, C. C., DeBano, L. F., and Ffolliott, P. F.: Fire effects on belowground sustainability: a review and synthesis, Forest Ecol Manag, 122, 51-71, 1999.

Ponder Jr, F., Tadros, M., and Loewenstein, E. F.: Microbial properties and litter and soil nutrients after two prescribed fires in developing savannas in an upland Missouri Ozark Forest, Forest Ecol Manag, 257, 755-763, 2009.

Prieto-Fernández, A., Acea, M. J., and Carballas, T.: Soil microbial and extractable C and N after wildfire, Biol Fert Soils, 27, 132-142, 1998.

Prosser, J. I.: Replicate or lie, Environ. Microbiol, 12, 1806-1810, 2010.

Rhoades, C. C., Meier, A. J., and Rebertus, A. J.: Soil properties in fire-consumed log burnout openings in a Missouri oak savanna, Forest Ecol Manag, 192, 277-284, 2004.

Rowlings, D. W., Grace, P. R., Kiese, R., and Weier, K. L.: Environmental factors controlling temporal and spatial variability in the soil-atmosphere exchange of CO_2, CH_4 and N_2O from an Australian subtropical rainforest, Glob Change Biol, 18, 726-738, 2012.

Saari, A., Heiskanen, J., and Martikainen, P. J.: Effect of the organic horizon on methane oxidation and uptake in soil of a boreal Scots pine forest, FEMS Microbiol Ecol, 26, 245-255, 1998.

Seidl, R., Schelhaas, M.-J., Rammer, W., and Verkerk, P. J.: Increasing forest disturbances in Europe and their impact on carbon storage, Nature Climate Change, 4, 806-810, 2014.

Sherwood, S., and Fu, Q.: A Drier Future?, Science, 343, 737-739, 2014.

Sommerfeld, R. A., Mosier, A. R., and Musselman, R. C.: CO_2, CH_4 and N_2O flux through a Wyoming snowpack and implications for global budgets, Nature, 361, 140-142, 1993.

Steinkamp, R., Butterbach-Bahl, K., and Papen, H.: Methane oxidation by soils of an N limited and N fertilized spruce forest in the Black Forest, Germany, Soil Biol Biochem, 33, 145-153, 2001.

Sullivan, B. W., Kolb, T. E., Hart, S. C., Kaye, J. P., Hungate, B. A., Dore, S., and Montes-Helu, M.: Wildfire reduces carbon dioxide efflux and increases methane uptake in ponderosa pine forest soils of the southwestern USA, Biogeochemistry, 104, 251-265, 2011.

Vance, E. D., Brookes, P. C., and Jenkinson, D. S.: An extraction method for measuring soil microbial biomass C, Soil Bio. Biochem, 19, 703-707, 1987.

Wan, S., Hui, D., and Luo, Y.: Fire effects on nitrogen pools and dynamics in terrestrial ecosystems: a meta-analysis, Ecol Appl, 11, 1349-1365, 2001.

Wang, Y., Xu, Z., and Zhou, Q.: Impact of fire on soil gross nitrogen transformations in forest ecosystems, J. Soils Sediments, 14, 1030-1040, 2014.

Wang, Y., Xu, Z., Zheng, J., Abdullah, K., and Zhou, Q.: $\delta^{15}N$ of soil nitrogen pools and their dynamics under decomposing leaf litters in a suburban native forest subject to repeated prescribed burning in southeast Queensland, Australia, J Soils Sediments, 15, 1063-1074, 2015.

Weber, M. G.: Forest soil respiration after cutting and burning in immature aspen ecosystems, Forest Ecol Manag, 31, 1-14, 1990.

Xue, L., Li, Q., and Chen, H.: Effects of a wildfire on selected physical, chemical and biochemical soil properties in a pinus massoniana forest in south China, Forests, 5, 2947-2966, 2014.

Zerva, A., and Mencuccini, M.: Short-term effects of clearfelling on soil CO_2, CH_4, and N_2O fluxes in a Sitka spruce plantation, Soil Biol Biochem, 37, 2025-2036, 2005.

Table 1. Weather conditions for the three sampling events, precipitation was recorded by the Mt Gravatt Alert weather station (27.55° S, 153.07° E, ~2 km from the sampling plots) and the data were collected at the website of Bureau of Meteorology (http://www.bom.gov.au).

Sampling events	Sampling dates	Antecedent precipitation* (30 days, mm)	Antecedent precipitation (3 months, mm)	Rainfall during sampling period	Temperature measured on-site
Aug 2013	27-30	4	71	0	24.76
Aug 2014	5-8	10[1]	22	0	23
Nov 2014	10-13	11	85[2]	0	29.88

* Total rainfall for the indicated periods

1 highest daily rainfall was 7 mm recorded on 27 Jul 2014

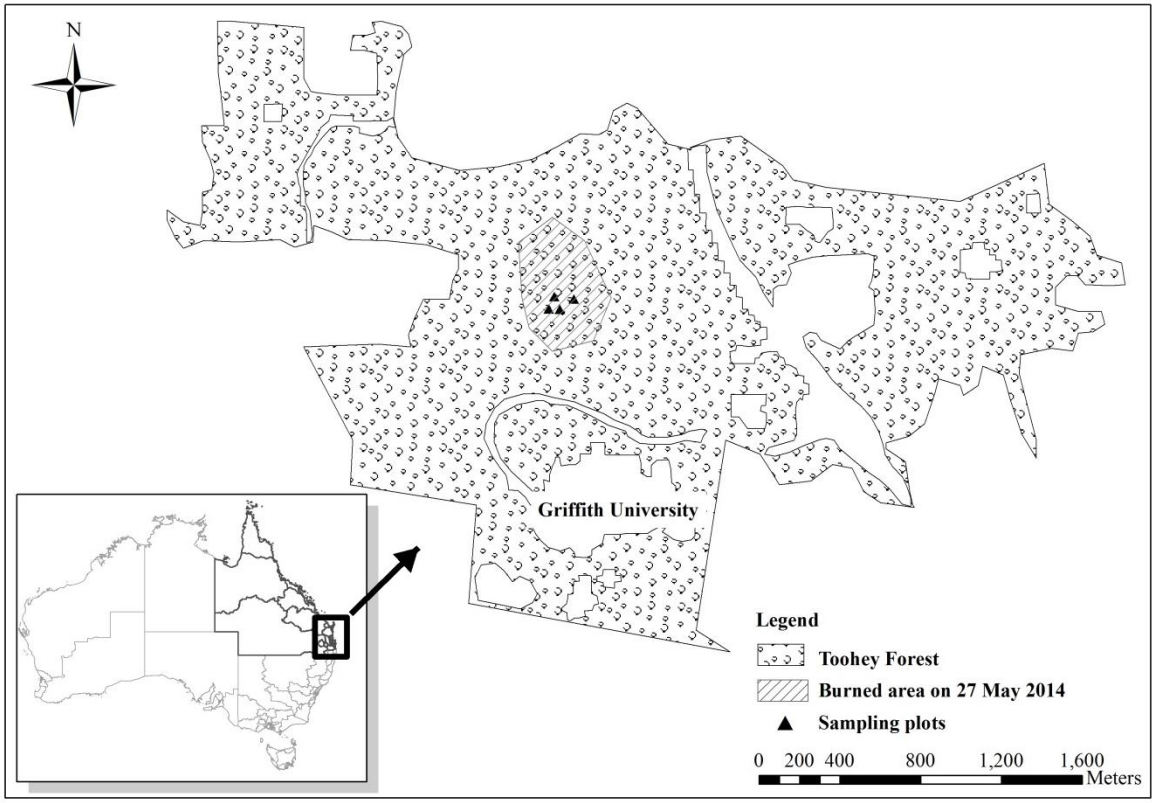
2 most rainfall for this period was recorded on 17 August 2014 (69 mm)

Table 2. Average gas exchange rates from surface soil in Toohey Forest before and after the prescribed burning. Values in parentheses indicate standard errors for the 4 replicate plots of each sampling period. Significant differences between measurements before and after the burning in the burned plots presented in lowercase letters. Significant differences between burned and adjacent unburned plots presented in uppercase letters. Mean values followed by the same letter are not significantly different (one-way ANOVA, $p \geq 0.05$).

Sites	**Dates**	**CH_4 mg m^{-2} d^{-1}**	**CO_2 mg m^{-2} d^{-1}**	**N_2O mg m^{-2} d^{-1}**
Burned	Aug 2013	-1.21 (0.42)a	5009.17 (2657.67)a	0.21 (0.24)a
	Aug 2014	-1.99 (0.51)bA	2974.24 (895.78)bA	0.00 (0.10)aA
	Nov 2014	-1.17 (0.78)aC	5835.69 (2639.99)aB	-0.04 (0.07)bB
Unburned	Aug 2014	-1.28 (0.26)B	2721.76 (1360.24)A	0.02 (0.11)A
	Nov 2014	-1.15 (0.16)C	7113.49 (3086.07)B	-0.01 (0.09)B

Table 3. Surface soil properties in Toohey Forest before and after the prescribed burning. Values in parentheses indicate standard errors for the 4 replicate plots of each sampling period. Soil moisture is presented in %. Other parameters (except pH) are presented in mg kg^{-1}

Sites	Date	Moisture	pH	NH_4-N	NO_3-N	WSOC	WSTN	HWEOC	HWETN	MBC	MBN
Burned	Aug 2013	12.3 (4.4)	4.33 (0.10)	1.78 (0.55)	0.02 (0.03)	88.83 (13.54)	7.10 (0.83)	875.44 (180.32)	67.80 (10.38)	522.45 (76.18)	56.37 (14.2)
	Aug 2014	10.3 (2.7)	5.76 (0.17)	6.76 (2.30)	0.09 (0.05)	80.00 (20.20)	6.81 (2.16)	2809.99 (479.18)	183.75 (39.10)	378.94 (103.58)	35.77 (10.00)
	NOV 2014	10.3 (3.0)	4.88 (0.24)	10.63 (4.18)	0.10 (0.12)	148.09 (38.25)	7.38 (2.97)	893.47 (310.29)	54.08 (19.32)	444.68 (45.27)	48.02 (7.15)
Unburned	Aug 2014	10.1 (2.0)	5.44 (0.24)	3.79 (1.67)	0.03 (0.04)	97.27 (28.36)	7.22 (1.19)	3638.29 (571.01)	203.91 (17.70)	493.62 (73.81)	45.26 (8.19)
	NOV 2014	9.8 (2.3)	4.67 (0.16)	6.44 (1.83)	0.04 (0.07)	114.35 (28.30)	5.14 (1.29)	942.56 (254.30)	51.94 (13.02)	406.07 (249.90)	42.76 (14.56)



1

2 Figure 1. Map of the study site in Toohey Forest located in south-eastern Queensland, Australia.

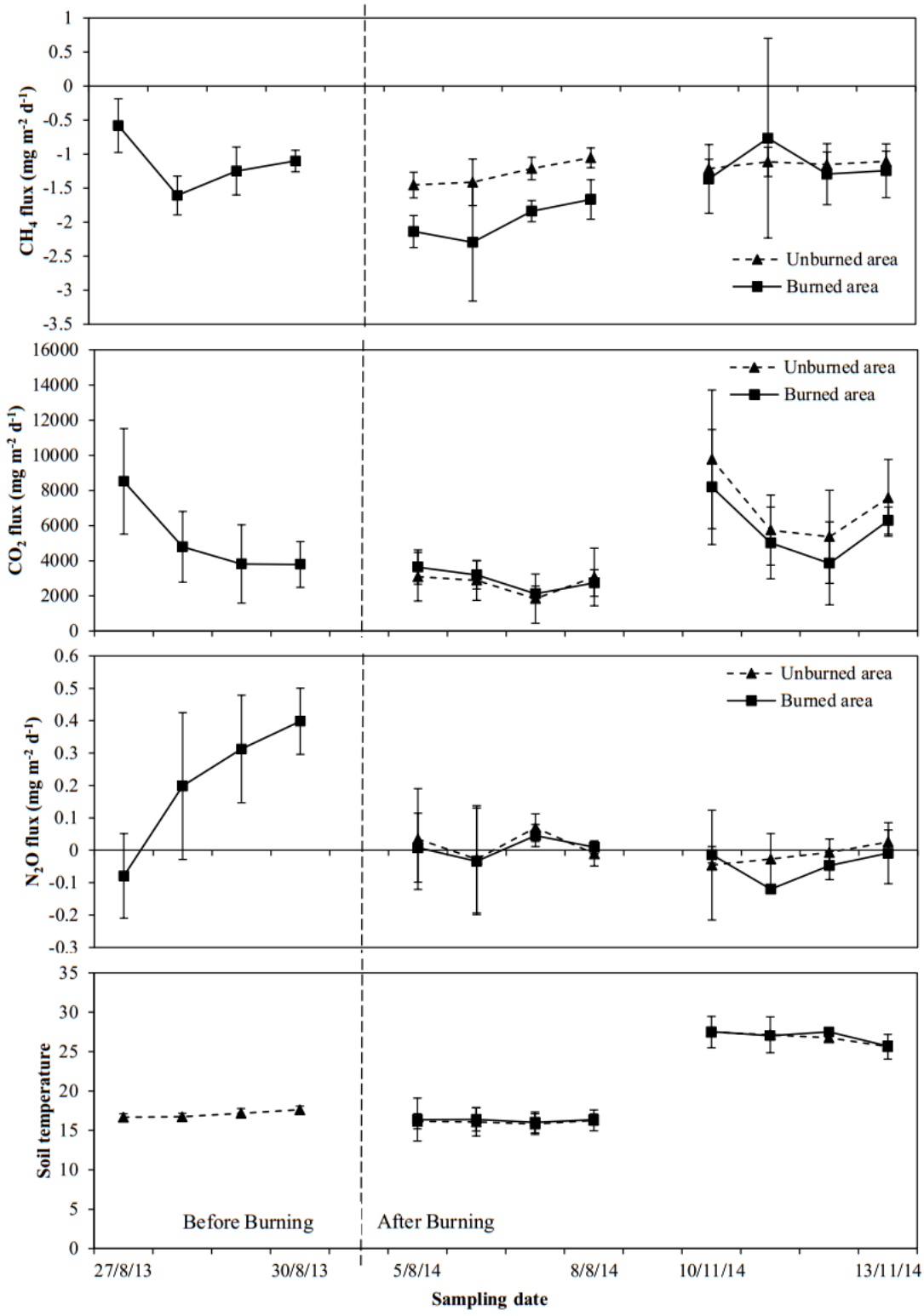


Figure 2. CH_4, CO_2 and N_2O exchange rates and on-site measured soil temperature before and after the prescribed burning. The dash line indicated the date of burning conducted on 27 May 2014. Each sampling period lasted for 4 days and the values were averaged from the 4 selected plots each day. The vertical bars indicated the standard error of the mean.