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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. It 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 at 4 replicate sites which were burned during the combustion and another 4 adjacent unburned sites 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} day^{-1}. The burning significantly enhanced CH_4 uptake as indicated by the significant higher CH_4 uptake rates at the burned sites measured in August 2014. While within the next 3 months the CH_4 uptake rate was recovered to pre-burning levels. Mean CO_2 emission from forest soils ranged from 2721.76 to 7113.49 mg m^{-2} day^{-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 sites in both August and November 2014. The temporal dynamics of the CO_2 emission presented more seasonal variations, rather than burning effects. The N_2O emission at the studied sites was quite low, and no significant impact of burning was observed. The changes in understory plants and litter layers, surface soil temperature, C and N substrate availability and microbial activities, resulting from the 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, however, this effect would be present within a relative short period. Only

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slight changes in the surface soil during the combustion and very limited damages in the mineral soils supported the quick recovery of the greenhouse gas exchange rates.

1 Introduction

As the 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 (Zhao et al., 2013; Sherwood and Fu, 2014; Fu et al., 2015). As for 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). The 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 altering environmental factors including 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 altering root activities, decomposition of organic matters, 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 attentions to soil CO_2, CH_4 and N_2O fluxes, not only because of the warming effect caused by CO_2, CH_4 and N_2O in the atmosphere globally (Zhao et al., 2013; 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 of

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different climate regions, but very few works have reported the effects of prescribed burning on soil greenhouse gas emission, 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 reported driving factors of fires on CO_2 efflux are the changes in soil temperature, moisture and fine root activities. As temperature and moisture reflect the 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 burning could further contribute to the decreased CO_2 efflux.

The CH_4 is the second most important greenhouse gas forcing climate change, 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, the burning altered soil moisture condition, 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 denitrificaiton processes but few published studies have reported inconsistent results. Fires could change forest soil N storages but the mechanism of

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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 forest ecosystem studied in this experiment is quite typical across most of Australia's forest areas, with a total of 92 million ha 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 firestorm 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). Some studies have reported the greenhouse gas emissions from Australia 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 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 sites which had similar stand conditions to address the following questions: (1) would prescribed fire affect greenhouse gas emissions at the soil atmosphere interface? (2) And if so, how long would these effects last? (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 a 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 the burning.

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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 (Fig. 1). This forest accounts for about 600 ha 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 winter months. Temperatures generally ranged between approximately 30 and 35 °C in summer and 10 and 15 °C 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 has been conducted regular 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 burning related with this study was conducted on 27 May 2014. Before the burning, we selected 4 sites with similar stand conditions for sampling. The understory of these sites was burned out during the burning, left a layer of wood charcoal on the ground. After the burning, these 4 sites were measured repeatedly at three months (August 2014) and six months (November 2014) later. Additionally, another 4 unburned sites were selected to further examine the impacts of burning. These sites were located in some adjacent green islands of those not touched by the burning and were near the existing four burned sites.

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2.2 Sampling method

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 sites. 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. All gas samplings were conducted during daylight hours from approximately 08:00 to 12:00 (UTC/GMT +10 hours).

2.3 Soil properties 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 °C for 48 h. pH was measured with a 1 : 5 aqueous solutions after shaking for 30 min. Soil inorganic N concentrations were extracted with 2

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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 vials, the suspension was then shaken by an end-over-end shaker for 5 min followed by centrifuging at 10 000 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 by 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 °C for 18 h.

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 h. Samples were shaken for 30 min 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 right after the field campaign. The concentration of CO_2 and CH_4 was 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

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a regression analysis with gas concentrations change within chamber vs. time (Zhao et al., 2013).

2.5 Statistical analysis

All statistical analyses were performed using IBM SPSS STATISTICS (version 20) software. Repeated measures ANOVA was used to examine statistically significant differences and changing patterns of soil gas fluxes and soil variables following the burning with measurement date as the repeated factor. Correlation analysis was tested for 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 burning

Average CH_4, CO_2 and N_2O emissions rates of the 4 replicate sites for each sampling event were listed in Table 1. While temporal patterns of gas exchange for the 4 day sampling of the 3 sampling periods were shown in Fig. 2. All the sampling sites showed negative CH_4 emissions rates during the three sampling events, or uptake atmospheric CH_4. At the burned sites, mean CH_4 uptake was significant increased by 64 % three months after the prescribed burning ($p < 0.001$), while during the third sampling period, CH_4 uptake had similar CH_4 uptake rate as that before the burning ($p = 0.843$). At the unburned sites, CH_4 uptake was relative stable during the dates of each sampling period and also showed less variation in uptake rate between August and November 2014. The significant difference in mean uptake rate in August 2014 ($p < 0.001$) but similar in November 2014 ($p = 0.921$) also confirmed that the CH_4 uptake increased at the first three months but was recovered to pre-burning level about six months after the burning.

Mean CO_2 emission from all sampling sites was significantly decreased by 41 % in August 2014 ($p < 0.001$). In November 2014, CO_2 efflux rates had exceeded that

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before the burning by 28 % but the difference was not significant ($p = 0.392$). Similar CO_2 emission rates between the burned and unburned sites during the sampling dates in August 2014 ($p = 0.549$) and in November 2014 ($p = 0.218$) were also observed.

As for N_2O, lower 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 sites were not solely performed as source of atmospheric N_2O, on 27 August 2013, 6 August 2014 and most days on November 2014, but the sites also took up N_2O from the atmosphere. No observed significant difference in N_2O emission between the burned and unburned sites in both August and November 2014.

3.2 Soil basic properties and their relationship with gas exchange rates

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

When relating these soil parameters to greenhouse gas emissions from soil surface, soil temperature showed a positive correlation with 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 CH_4 ($R = -0.595$, $P = 0.006$) and CO_2 ($R = -0.591$, $p = 0.006$) emission. NH_4-N was negatively correlated with N_2O emission ($R = -0.533$, $p = 0.015$).

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3.3 Soil C and N dynamics before and after burning

There was no significant difference in WSOC at burned sites between August 2013 and August 2014, only slightly decreased WSOC was observed in August 2014. However, WSOC was significant higher in November 2014 ($p = 0.034$). Comparing to the unburned sites, WSOC in most burned sites (3 out of 4) was lower in August 2014 ($p = 0.387$) while higher in November 2014 ($p = 0.237$). No significant difference was found between any sampling periods for WSTN, despite higher WSTN at some burned sites than those before burning and unburned sites. 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 burning ($p = 0.929$). The difference in HWEOC between burned and unburned sites were also significant in August 2014 ($p = 0.0361$) but insignificant in November 2014. The situation was similar for HWETN.

Mean MBC at burned sites 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 $mg\,kg^{-1}$). Burned sites also showed lower MBC values when compared to the unburned sites both in August ($p = 0.121$) and November ($p = 0.516$) 2014. MBN had the same dynamics as MBC.

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 has 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 ($R = 0.565$, $p = 0.009$).

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4 Discussions

4.1 Burning impacts on soil properties

The prescribed burning has resulted in a slightly increase in surface soil temperature, which is in agreement with most existing literature results (Burke et al., 1997; Certini, 2005). The burning of the understory vegetative cover, together with the resulted consumption of fuels, removal of litter layer and increased charred materials on the soil surface would all moderate 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 insignificant level. Meanwhile, the 4 selected sites in the study did not show consistent fluctuations in surface soil moisture before and after burning, which generally 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 restricts the movement of water into soil profile (Burke et al., 1997; Certini, 2005). This might attribute to the representativeness of the measurements and one measurement for each sampling period might not suffice to represent the physical state of water in the soil (Weber, 1990).

An increase in pH values was found at the burned areas in August 2014 and it was returned to a comparative level in November 2014. Although no significant difference was found between burned and the reference unburned sites in 2014, values for the burned sites were still higher than those at the unburned sites. The increased pH after the burning would be probably due to the release of extractable basic cations from the deposited ashes during the burning. Several studies also find increased pH after fire (Guinto et al., 1999; Certini, 2005; Kim et al., 2011; Xue et al., 2014) and the increased pH would either recover to unburned 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.

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NH_4-N was significantly increased after the burning, but no significant changes were observed for NO_3-N. Since NH_4-N is a direct product of combustion and NO_3^- is 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 phenomenon found by Nardoto and Bustamante (2003) in savannas of Central Brazil and Covington and Sackett (1992) in a ponderosa pine forest in USA.

The burning has significantly reduced MBC in the surface soil and it showed no apparent sign of recovery six months later after the 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 significantly increased in August 2014 and returned to pre-burning level. While the low intensity of the 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 burning and six months after burning from burned sites and all fluxes from unburned sites fall in the range of CH_4 fluxes obtained by Kiese et al. (2003) (varies from 0.84–1.63 $mg\,m^{-2}\,day^{-1}$) and a recent study by Rowlings et al. (2012) which were conducted in a similar forest ecosystem in Australia. While

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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 pre-burning level within six months after the burning. We obtain this conclusion from the similar CH_4 uptake rates in November 2014 when compared to the CH_4 uptake at unburned sites and the rates before burning at the burned sites. The low fire intensity of the prescribed burning in this study may cause less impact on the system and therefore shorten the required time to recover to pre-burning conditions for the studied forest. 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 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, probably due to the low intensity burning, did not affect the soil water conditions in the soil horizons relevant to CH_4 oxidation, or the soil moisture was partly recovered to pre-burning level and was also lower than the range of strong moisture control. 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

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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 may have increased CH_4 diffusion into soil profile. Removal of the C rich O horizon caused by the burning eliminated a barrier of 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 plants regrowth. This effect of 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 characteristic of CH_4 uptake is the physical changes in surface soil. Although not measured in current study, literature has shown that low to moderate fires would increase soil structure stability due to the formation of the dydrophobic 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 create a channel ideal for diffusion of atmosphere CH_4 into soil profile and thereby increase CH_4 oxidation rates.

4.2.2 CO_2 effluxes

The studied sites acted as a persistent source of atmospheric CO_2 before and after the burning, while the CO_2 emission rates, either before or after burning, were similar to the results obtained by Carlyle and Than (1988) in a native forest with low soil moisture (about 5520 $mg\,m^{-2}\,day^{-1}$) and by Rowlings et al. (2012) in an Australian subtropical rainforest (around 3600 $mg\,m^{-2}\,day^{-1}$). However, the CO_2 emission values were much lower than the reported high soil respirations (over 20 000 $mg\,m^{-2}\,day^{-1}$) in various Australian forest ecosystems with high soil moisture and temperature (Car-

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lyle and Than, 1988; Fest et al., 2009; Rowlings et al., 2012). A number of existing studies have reported reduced CO_2 emissions after 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 year (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 burned and the unburned sites. 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 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 at the burned sites compared to the unburned sites 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, the microbial biomass was likely to be limited by the amount 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 recovering

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CO_2 flux six months after the burning, combined with recovered MBC to 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 current study were very small (-0.21 to $0.54\,mg\,m^{-2}\,day^{-1}$ before burning and -0.18–$0.11\,mg\,m^{-2}\,day^{-1}$ after 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–$8.19\,mg\,m^{-2}\,day^{-1}$ recorded by Kiese et al. (2003) in a tropical rainforest and the range of 0.62–$1.57\,mg\,m^{-2}\,day^{-1}$ by Rowlings et al. (2012) in a subtropical rainforest. No significant effect of 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 during the chamber employment were quite low, and these changes were below the detection limit of the GC system. Even though the dry and well aerated soil of the sampled sites makes it prone to nitrification rather than denitrification, the observed small inorganic N pool (NH_4^+ < than $10\,mg\,N\,kg^{-1}$ while NO_3^- < $0.1\,mg\,N\,kg^{-1}$) 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 enhanced situation of low soil water availability and dry conditions after burning, due to the removal of understory plants and litter layers and increased evapotranspiration rates, limited the activities of soil nitrifiers (Livesley et al., 2011). Also the burning induced charcoal at the soil surface would also supress N_2O exchange rates which were reported in a recent controlled experiment in Japan (Kim et al., 2011). However, the accumulated N substrate, either NH_4^+ or NO_3^-,

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might cause further high N_2O emissions with 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, forest soil in Toohey Forest still acts 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 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 layer, surface soil organic C and also the consumption of fossil fuels to manipulate fires. Data on C burned, fuel consumed and continuous measurement of soil gas exchanges are required to quantify the burning caused greenhouse effect in future studies.

5 Conclusions

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 soil profiles. The removal of litter layer and surface soil organic materials and the altered soil physical structural caused by the burning were the major factors contributing to the increased CH_4 diffusion. The CO_2 emission was largely decreased but it was a combination of burning introduced variation and natural seasonal variations. Changes in root respiration and soil microbial community were the two controlling factors related to burning effect on CO_2 emission. Due to the controlled condition of the prescribed burning, both CH_4 uptake and CO_2

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emission started to recover about three months after the burning and it appears that the gas exchange rates were recovered to pre-burning level about six months after burning. This quick recovery was closely related to the limited effect of burning on soil and no dramatic damages in the mineral soils. 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 at the studied sites and showed no obvious impacts from the burning. Finally, a continuous monitoring of soil properties and soil greenhouse gas exchanges and even ecosystem gas exchange rates before and after burning is important to reveal the key mechanisms and quantify the complex impacts of burning on forest ecosystem and regional climate.

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Table 1. 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 replicates of each sampling period. Significant differences between measurements before and after the burning presented in lowercase letters. Significant differences between burned and unburned sites 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 $mg\,m^{-2}\,day^{-1}$	CO_2 $mg\,m^{-2}\,day^{-1}$	N_2O $mg\,m^{-2}\,day^{-1}$
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

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Table 2. Surface soil properties in Toohey Forest before and after the prescribed burning. Values in parentheses indicate standard errors for the 4 replicates 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	4.33	1.78	0.02	88.83	7.10	875.44	67.80	522.45	56.37
		(4.4)	(0.10)	(0.55)	(0.03)	(13.54)	(0.83)	(180.32)	(10.38)	(76.18)	(14.2)
	Aug 2014	10.3	5.76	6.76	0.09	80.00	6.81	2809.99	183.75	378.94	35.77
		(2.7)	(0.17)	(2.30)	(0.05)	(20.20)	(2.16)	(479.18)	(39.10)	(103.58)	(10.00)
	Nov 2014	10.3	4.88	10.63	0.10	148.09	7.38	893.47	54.08	444.68	48.02
		(3.0)	(0.24)	(4.18)	(0.12)	(38.25)	(2.97)	(310.29)	(19.32)	(45.27)	(7.15)
Unburned	Aug 2014	10.1	5.44	3.79	0.03	97.27	7.22	3638.29	203.91	493.62	45.26
		(2.0)	(0.24)	(1.67)	(0.04)	(28.36)	(1.19)	(571.01)	(17.70)	(73.81)	(8.19)
	Nov 2014	9.8	4.67	6.44	0.04	114.35	5.14	942.56	51.94	406.07	42.76
		(2.3)	(0.16)	(1.83)	(0.07)	(28.30)	(1.29)	(254.30)	(13.02)	(249.90)	(14.56)

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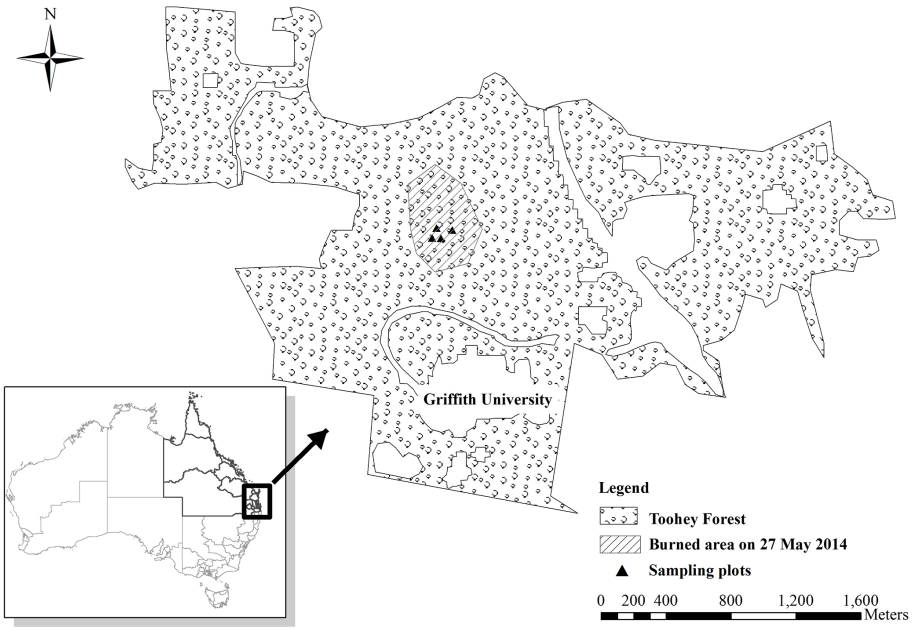

Figure 1. Map of the study site in Toohey Forest located in south-eastern Queensland, Australia.

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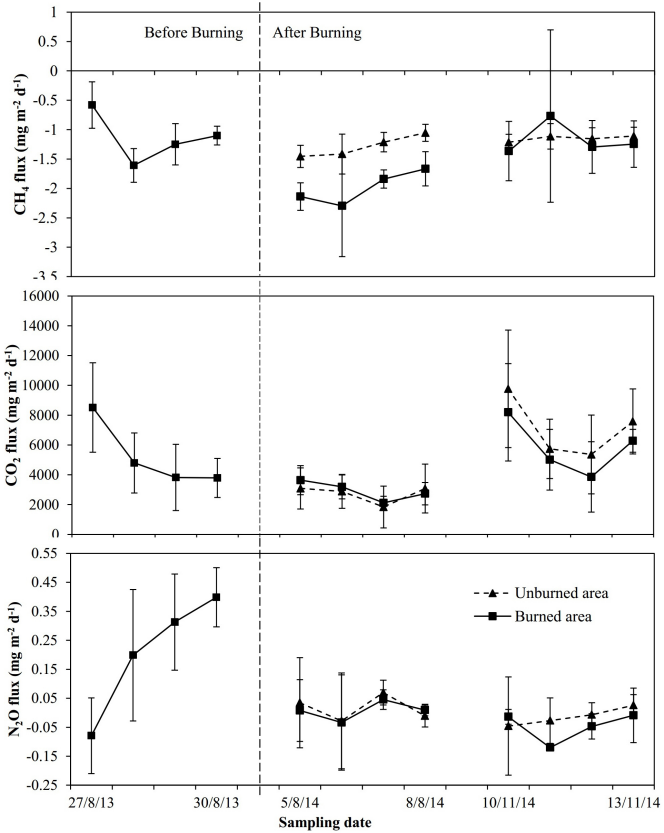

Figure 2. CH_4, CO_2 and N_2O exchange rates before and after the prescribed burning. The dash line indicated the date of burning conducted on 27 May 2014. Each sampling period lasted for 4 days and the values were averaged from the 4 selected sites each day. The vertical bars indicated the standard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

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