FUNDAMENTAL RESEARCH

For. Sci. XX(XX):1–14
doi: 10.1093/forsci/fxy069

forest ecology

Annual Greenhouse-Gas Emissions from Forest Soil of a Peri-Urban Conifer Forest in Greece under Different Thinning Intensities and Their Climate-Change Mitigation Potential

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Knowledge of the response of thinning implementation on forest soil–atmospheric greenhouse gas (GHG) (CO_2, CH_4, N_2O) fluxes exchange system in Mediterranean region is limited because of the high heterogeneity of both soil properties and forest biomass. The novelty of this study is grounded predominantly in evaluating for the first time the response of annual GHG fluxes to thinning in a coniferous peri-urban forest soil in Greece, thus contributing significantly to the enrichment of the GHG fluxes database from the Mediterranean forest ecosystem. Results suggest that CH_4 uptake increased with increasing thinning intensity. The reduction in CO_2 emissions in both thinning treatments was possibly related to an indirect effect of soil heterotrophic and autotrophic respiration. Coniferous peri-urban forests in Greece can act temporally as sinks of atmospheric N_2O in the coldest months and a weak source of N_2O fluxes in summer. The GHG variation depended largely on soil environmental factors with soil temperature representing the dominant factor for CO_2 and CH_4, whereas soil moisture correlated, albeit weakly, with N_2O variability. Reduction in global warming potential was observed in both thinning treatments, markedly in selective treatment, giving an initial indication that high-intensity thinning in coniferous peri-urban forests in Greece presents a high potential for global change mitigation.

Keywords: greenhouse gases, thinning treatments, global-warming potential, climate-change mitigation

The role of greenhouse gases (GHGs) in the atmosphere is based on the absorption of an amount of radiation emitted by the Earth's surface and the re-emitting of it back to Earth, increasing the surface temperature (Thomas et al. 2016). CO_2, CH_4, and N_2O, which amount to 80 percent of total GHGs (Ciais et al. 2013), are produced by natural processes, such as respiration and other biological processes, and also by anthropogenic activities (Thomas et al. 2016).

Although, in 2014, GHG emissions in EU were 24 percent lower than 1990 levels, additional policies will need to be implemented to achieve the 2030 target that was adopted in October 2014, aimed at reducing GHG emissions by 40 percent by 2030 compared to 1990 levels (EC 2016). One of the targets of the Paris Agreement of United Nations Framework Convention on Climate Change (UNFCCC) in December 2015, the first climate conference attended by 195 countries, was the reduction in emissions, as part of a broader climate change platform (UNFCCC 2015).

An emission metric that was introduced in the IPCC First Assessment Report was the global warming potential (GWP) (IPCC 2014). This metric is used to express different GHG and other climate forcing agents with different components and natural properties in a common unit of CO_2-equivalent emissions. GWP has been integrated over 20, 100, or 500-year time horizons, and the GWP 100-year horizon (GWP100) has recently been adopted by the UNFCCC and Kyoto Protocol as a default metric (Myhre et al. 2013).

Manuscript received June 4, 2018; accepted January 4, 2019.

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Acknowledgments: *This research was funded by the LIFE14 CCM/IT/000905 project entitled "Recovery of degraded coniferous Forests for environmental sustainability, Restoration and climate change Mitigation" (FoResMit). We thank the National Observatory of Athens, Institute of Meteorology and Atmospheric Physics, Athens, Greece, for the supply of meteorological data.*

In forest ecosystems, degradation effects caused by abandonment, overharvesting, unsustainable timber production, overgrazing (De Fries et al. 2006), and fire (Langenfelds et al. 2002) affect GHG emissions and reduce the C sink potential (De Fries et al. 2007). In the Mediterranean region, millions of hectares of forested land are in pine plantations. The main objective of land restoration with conifers was the colonization of a fast-growing pioneer species in these degraded areas in the short term, assuming that these species would facilitate the introduction (either artificial or natural) of late-successional hardwoods, providing high biodiversity, landscape heterogeneity, resistant, and resilient ecosystems (Barbéro et al. 1998, Pausas et al. 2004). Now that the initial strategy of reforestation (i.e., introduction of fast-growing species) in the Mediterranean region has been completed (Pausas et al. 2004), there is an interest in the establishment of natural forests with high biodiversity, characterized by native broadleaved species (Gómez-Aparicio et al. 2009). To date, this second objective has not been completed because of silvicultural costs and the failure of traditional silvicultural methods to address the current disturbance regimes (Pausas et al. 2004).

In Greece, peri-urban forests have a mean age of approximately 60–70 years and are mainly represented by conifer plantations, such as *Pinus brutia*, *Pinus halepensis*, and *Cupressus sempervirens*. Today, a significant percentage (67 percent) of peri-urban forests consist of coniferous species that were established through reforestation or afforestation, with the remainder covered by natural mixed forests (Christopoulou et al. 2007). However, most of the peri-urban forests in Greece have been affected by forest and soil degradation, mainly the result of wild fire and lack of proper management practices (Christopoulou et al. 2007), as well as through overgrazing that occurs in most of Europe's forests (Spiecker 2003).

Ultimately, the creation of mixed-wood forests is recommended, combining the dominant features of both conifer plantations (i.e., fast growth) and native broadleaved (i.e., fire-resistant) species (Vallejo et al. 2012). In addition, implementation of proper silvicultural and sustainable management practices would ensure productivity and C storage capacity, as well as biodiversity (Scarascia-Mugnozza et al. 2000, Torras et al. 2008). Stand thinning is among the range of management practices employed in this region and can affect GHG exchange with the atmosphere through the alteration of soil characteristics and functioning. Thinning results in a lower stem density and a more open canopy (Boczon et al. 2016) and alters the water and energy balance of the ecosystem (Gathany and Burke 2014). These canopy gaps result in environmental changes such as an increase in light availability reaching the forest floor (Naaf et al. 2007), which, in turn, increases seedling establishment in the months following gap creation (Boudreau et al. 2005, Kukkonen et al. 2008, Ren et al. 2015). In addition, it influences the site productivity and homogeneity of understory vegetation (Navarro et al. 2013), which causes seasonal and annual alteration of nutrients, organic debris, initial litterfall decomposition rate, and seasonal and annual litterfall (Caldentey et al. 2001). Thinning also has contrasting effects on soil respiration. Many studies have observed increases (Selig and Seiler 2004), decreases (Tang et al. 2005), and no changes in soil respiration in response to thinning (Wilkinson et al. 2016). The alteration of these biotic and abiotic factors can influence the rate of exchange of GHG emissions that are released by forest soil to the atmosphere, mainly through the change in plant-derived processes (e.g., photosynthesis), and microbial-mediated processes (Gathany and Burke 2014).

Knowledge of thinning impacts on forest soil–atmospheric GHG flux exchange in forested ecosystem in the Mediterranean region remains limited. In order to understand how thinning affects net GHG emissions from soil, it is essential to understand the impacts of thinning on the soil environment (Gathany and Burke 2014). The aim of this study was (1) to quantify the spatial and temporal effect of different thinning treatments on GHG fluxes from forest soil one year after thinning operations, (2) to investigate the effect of soil environmental drivers on GHG fluxes, and (3) to estimate the GWP of each thinning treatment and therefore their contribution to climate-change mitigation. A reduction in CO_2 emissions is expected after thinning treatments, resulting from heterotrophic component decrease, as well as an increase in CH_4 uptake related to changes in soil moisture conditions. N_2O changes are more difficult to predict because of the scarcity of information. However, we hypothesized a decrease in N_2O fluxes in spring caused by an increase in N uptake by the new vegetation, which in turn lowered the amount of N available for microbial processes (e.g., nitrification and denitrification). GWP is expected to decrease, driven mostly by CO_2 reduction.

Materials and Methods

Study Site

The study site is located in the peri-urban forest of Xanthi (41°09′27.3″N, 24°54′09.8″E, Greece) that covers an area of approximately 2,400 hectares and is a part of the Xanthi–Gerakas–Kimerion public forest (Figure 1). The forest landscape is characterized by its steepness with different geographic orientations and slopes ranging from 5 to 80 percent.

Planting began in 1936 and continued periodically until 2007, with the majority of the plantations established prior to 1973. Today the main forested area in the study site consists of even-aged conifer plantations dominated by Calabrian pine (*Pinus brutia Ten*.). Intermediate species include *Pinus pinaster*, *Pinus pinea*, and *Pinus nigra*, with a broadleaved understory (Table 1). The stands are approximately 55 years old with a mean total height of approximately 20 m. The stand tree characteristics of the study site are listed in Table 1 (De Meo et al. 2017).

The soil type of the study site is campisols with a moderately acidic pH (5.6) and a low carbonate content (6.2 percent). Soils had a sandy clay to sandy clay loam texture, with mean of sand and clay values

Management and Policy Implications

Reducing GHG emissions is a fundamental aspect of climate change mitigation strategies. In forest ecosystems, climate change mitigation actions are strictly associated with policies related to implementation of forest-management practices such as thinning, aimed at reducing GHG emissions and their climate change mitigation potential. Monitoring of GHG fluxes from forest soil that is responsible for 70 percent of total GHG emissions, under different thinning treatments, is fundamental in order to meet EU targets. Therefore, the implementation of proper management practices in forest ecosystems contributes significantly to climate change mitigation, beyond the forest sector.

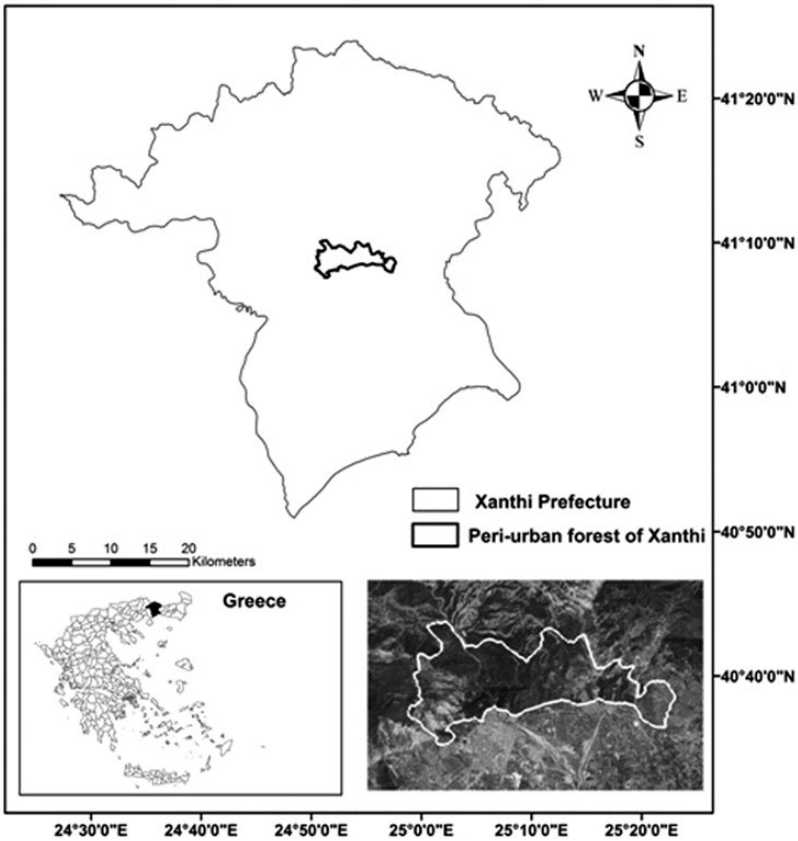

Figure 1. Peri-urban forest of Xanthi, northeastern Greece. The peri-urban forest is marked by a white solid line (Xanthi Forest Directorate).

Table 1. Study-site characteristics.

Vegetation				
Dominant species	*Pinus brutia*			
Intermediate species	*Pinus pinaster*, *Pinus pinea*, *Pinus nigra*, *Cupressus* spp.			
Broad leaves as undergrowth	*Robinia pseudoacakia*, *Acer campstre*, *Acer sempervirens*, *Carpinus orientalis*, *Cornus mas*, *Fraxinus ornus*, *Quercus coccifera*, *Quercus dalechampii*, *Quercus frainetto*, *Ulmus campestre*			
Stand tree characteristics				
Mean tree density (trees per hectare)	2,626			
Basal area (m^2 ha^{-1})	38.81			
Summary statistics of individual tree variables prior to thinning (De Meo et al. 2017)				
Variables	Mean	SD	Minimum	Maximum
Total height (m)	19.74	4.58	2.80	31.40
dbh (cm)	32.19	8.37	10.00	57.40
Form height (fh)	6.39	3.21	0.26	17.73
Form factor	0.31	0.13	0.01	0.78
Mean stand volume, V (m^3)	0.74	0.61	0.01	3.79

of 60 percent and 17.5 percent respectively. The soil organic matter content at the study site was as expected, for campisols, with a clear vertical distribution of total N (TN) and total organic C (TOC) (Soil Survey Staff 2010). The climate of the area is semiwet Mediterranean with a tendency to dry, and it is characterized by mild, rainy winters and relatively warm, dry summers. Meteorological time-series data were used for the period 1960–2002 to describe mean temperature and precipitation. The xerothermic period begins in the middle of June until the end of September. The mean temperature was comparable, but the precipitation during the study period was lower than the normal amounts received in the region (Figure 2). A significant decrease in rainfall was observed in the last decade, estimated at around 25 percent (Theodoridis 2016), whereas in the study period the rainfall decreased about 50 percent.

Experimental Design and Thinning Treatments

Within the study site, nine circular plots (radius 13 m and area 531 m^2) were randomly established in three replicates for each thinning treatment (traditional, selective, and control). The thinning operations took place in September 2016, based on the following silvicultural guidelines.

The traditional thinning was a medium-heavy intensity thinning (20.7 percent BA removal) that removed primarily dead, poorly formed, damaged, and intermediate-suppressed trees. Broadleaves were also thinned to create a uniform distribution in the understory.

In selective thinning, overstory pines were removed to release the broadleaved trees. Selective thinning largely focused on the removal of poorly formed stems, leaving the higher-quality stems in the residual stand. The understory thinning resembles "positive selection" thinning, where competitors of the most vigorous and stable broadleaved trees are removed, in order to release them and, thereby, increase C sequestration at this layer. The selective thinning was more intense (almost double the BA removal) than traditional thinning, reducing the basal area by 39.2 percent. The three unthinned plots were used as the control.

Measurements of Soil GHG Fluxes

GHG exchanges between forest soil and atmosphere were measured using the static closed chamber technique that is most appropriate for comparing the effects of different treatments in replicated field experiments (Chadwick et al. 2014). This method has been widely used for measuring GHG emissions from soil in different forest ecosystems (Von Arnold et al. 2005, Fang et al. 2016, Leitner et al. 2016, Savi et al. 2016).

This method is based on the gas-exchange ability between the soil below the chamber and the chamber headspace that is installed in a particular area of the soil in the experimental area (Pihlatie et al. 2013). Gas sampling was carried out throughout the year after thinning operations in each of the nine plots twice per month. Two chamber collars (30 cm in diameter) were installed to a depth of at least 5 cm into the soil in each plot, in September 2016. The location of the collars, installed on a flat or slightly slope soil, was chosen to cover as much of the different topography and soil conditions of each plot as possible. At each treatment, the collars were installed under different canopy densities (clos ed canopy in control, slightly open in traditional, widely open in selective thinning), representing the canopy conditions of each treatment after thinning (Figure 3). GHG fluxes were measured from October 2016 to September 2017.

During each gas-sampling event, both chambers in each plot were closed for 30 minutes, and gas samples were collected four times at intervals of 0, 10, 20, and 30 minutes after closing the chambers. The gas was collected with a plastic syringe 30 mL with a 22G 0.7 × 25.4 mm hypothermic needle inserted through the septum. After pushing 5 mL of gas out of the syringe, 25 mL was transferred immediately in a pre-evacuated 12 mL glass exetainer® vial (Labco Limited, Lampeter, UK), and the air temperature within the chamber was recorded. At each sampling time, eight gas samples were collected from each plot, totaling 72 samples. All samples were stored in a refrigerator before analysis.

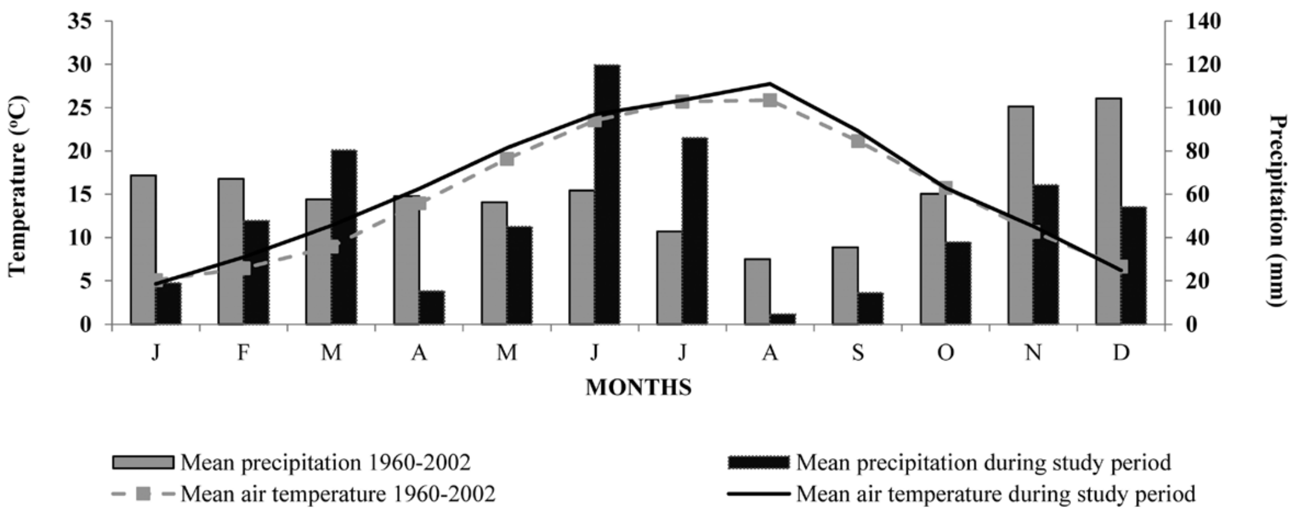

Figure 2. Meteorological data for the period 1960–2002 source (Meteorological Kapnikon Station, Xidas, 1983, Theodoridis 2016) compared to the meteorological data for the study period (October 2016 to September 2017) from the local meteorological station in Xanthi, which is located very close to Xanthi city named Biological Cleaning of Xanthi, 115 m in altitude. The height of the temperature and humidity sensors is 2 m, and the height of the anemometer is 5 m (Lagouvardos et al. 2017).

Figure 3. Localization of collars in (MSA) under different canopy densities: (a) control (b) traditional, and (c) selective thinning.

GHG concentrations were analyzed within 4 weeks of collection using a GC-2014 (Shimadzu Inc., Kyoto, Japan) gas chromatograph (Yuesi et al. 2003), with a thermal conductivity detector for CO_2, a ^{63}Ni electron capture detector for N_2O, and a flame ionization detector for CH_4). The GC was calibrated using standard N_2O and CH_4 with accuracies certified at 95 percent (Airgas Inc., Radnor, PA). Gas concentrations were converted to mass per unit volume using the ideal gas law, and chamber air temperatures and volumes were measured during each sampling.

Fluxes were estimated using the linear slope of gas concentration against chamber closure time and enclosed soil surface area. Fluxes were set to zero if the change in gas concentration during chamber enclosure was below the minimum detection limit determined from the GC, and values rejected if they passed the detection test but had an R^2 of <0.75 and treated as missing data (Adviento-Borbe et al. 2013, Mazza et al. 2019).

In order to estimate the cumulative CO_2, CH_4, and N_2O fluxes and GWP for each replicate, we summed the daily fluxes over time. All measurements were taken between 9:30 am and 2:30 pm to minimize changes in soil CO_2 effluxes associated with diurnal cycles (Davidson et al. 1998). For days with no measurements, GHG fluxes were estimated using linear interpolation between two measurement days divided by the number of days between them (Adviento-Borbe et al. 2010). The GWP of N_2O and CH_4 was calculated in mass of CO_2 equivalents (kg CO_2 eq ha^{-1}) using the climate warming factor on a 100-year horizon. Radiative forcing potentials relative to CO_2, equal to 34 and 298, were used for CH_4 and N_2O, respectively (Myhre et al. 2013, Mazza et al. 2018). Assuming that the root contribution (autotrophic respiration) to the total soil respiration system (root-autotrophic and microbial-heterotrophic respiration) in forest ecosystems is approximately 50 percent throughout the year (Hanson et al. 2000), we divided CO_2 fluxes in two to estimate only the contribution of microbial components to the GHG exchange system.

In this study, the temperature response of soil respiration was measured for each thinning treatment and season with an exponential model. Soil respiration was calculated according to the equation $R_T = a \exp^{bT}$ where R_T = soil respiration at a given temperature, a and b are fitted parameters, and T is temperature (Meyer et al. 2019). The Q_{10} function provides an indication of temperature sensitivity of soil respiration by a 10°C increase in temperature (Kirschbaum 1995). Q_{10} values were calculated using the function $Q_{10*} = \frac{R_{(T+10)}}{R_T}$ where R_T = soil respiration at given temperature, in order to study the effect of Tsoil on soil respiration (Kirschbaum 2006).

Measurements of Soil Temperature and Soil Moisture

Each sampling date, soil temperature, and volumetric soil water content was measured simultaneously with soil GHG flux measurements. The soil temperature (TSoil) was recorded from each plot using a thermometer soil sensor (Hanna thermometer, Part Code HI7661). The sensor was inserted into the soil to a depth of 10 cm near each collar at each sampling period and remained in place for 10 minutes before measuring.

For measuring the volumetric soil water content (MSoil) of each plot, three samples (approximately 50 g) of mineral soil near each collar were taken from 0 to 10 cm depth. The soil moisture of each sample was determined at the laboratory, after sampling, using the PMB Moisture Analyzer (P.N. 3076610544, Adam Equipment). According to its measured principles, the initial weight of each soil sample was recorded, and then a halogen radiator was used to dry the sample at 150 °C, while an integrated balance continually recorded the sample weight. The total loss in weight was interpreted as the moisture content (percent). The mean of the soil moisture of each three soil samples was used to determine the soil moisture of each plot. Soil moisture measurements (Msoil) started in January 2017 and ended in September 2017.

Statistical Analysis

All statistical analyses were performed using SPSS statistical software based on the mean GHG flux data of the two replicated chambers per site. A repeated-measures analysis of variance (ANOVA), with a Greenhouse-Geisser correction, was used to test the significant differences in fluxes on each measurement date among thinning treatments and among months. Post hoc tests using the Bonferroni correction were used to present the mean differences among treatments and among months. The effect size of these factors on GHG mean differentiation was estimated by univariate ANOVA, calculating the partial eta squared. Autocorrelation was checked using the Durbin–Watson test, by testing the assumptions for the least-squares method.

A Pearson correlation analysis was used to examine the relation of Tsoil and Msoil with GHG fluxes. A stepwise multiple linear regression (MLR) analysis was conducted, for the period January 2017 to September 2017, in order to determine the effect of the combination of measured soil environmental factors (soil temperature and soil moisture) on GHG fluxes.

Results

Soil Environmental Factors

The seasonal pattern of soil temperature was similar among treatments throughout the study year (Figure 4a). Soil temperature was slightly higher than 15 °C at the start of the sampling period (October 2016), steadily declined through to February 2017 to a low of 2 °C, and then increased uniformly until August 2017, peaking at 25 °C. No significant differences (P = .99) in Tsoil among thinning treatments were detected.

The pattern of Msoil was similar among treatments (Figure 4b), without any significant differences (P > .05). Soil moisture was close to 15 percent at the start of the period of soil moisture measurements (January 2017) until May 2017, with a peak observed in the middle of January 2017 at 25 percent and less than 5 percent in February 2017, and then steadily declining until September 2017 to 5 percent. There were clear differences in mean Msoil among months (F [8,144] = 46.55, P = .00).

GHG Flux Variation

CO_2

CO_2 fluxes followed a similar pattern in all treatments with a wide range from 6.1 to 57 kg C-CO_2 ha^{-1} d^{-1}. The overall mean cumulative fluxes were 7,813.11 ± 345.38 kg C-CO_2 ha^{-1} y^{-1} with mean monthly rate of 17.91 kg C-CO_2 ha^{-1} (Table 2). The annual CO_2 emissions kg C-CO_2 ha^{-1} y^{-1} ranked in the order control > traditional > selective, although the differences were not significant (P = .249).

Significant differences in CO_2 fluxes occurred among months (F [2.52, 20.19] = 21.94, P = .000, η^2 = 73.30 percent). The lowest mean emissions were measured in January (9.35 ± 2.28 kg CO_2-C ha^{-1} d^{-1}) and February (7.35 ± 3.23 kg CO_2-C ha^{-1} d^{-1}) and the highest in May (46.20 ± 16.84 kg CO_2-C ha^{-1} d^{-1}) following a typical pattern driven by Tsoil until June. During the summer, a reduction in CO_2 fluxes was observed with the exception of August as opposed to Tsoil, which predominantly increased (Figure 5a). No clear differences in mean CO_2 fluxes occurred among treatments within months apart from June where mean CO_2 fluxes in the control plots (40.09 ± 13.51 kg CO_2-C ha^{-1} d^{-1}) were significantly (P = .040) higher than selective (28.49 ± 8.36 kg CO_2-C ha^{-1} d^{-1}), but were not significantly different from the traditional (36.74 ± 14.58 kg CO_2-C ha^{-1} d^{-1}) (Figure 6a).

In order to calculate the Q_{10} value, we divided our data, because of their high seasonal and spatial heterogeneity, into a small data set in order to increase the predictability. Taking into consideration the high interaction of soil moisture with the temperature sensitivity of soil respiration (Meyer et al. 2018), we investigated the Q_{10} variability of soil respiration at different soil moisture levels. Therefore, we evaluated the temperature sensitivity for all seasons and also throughout the study year, apart from the summer. Thus,

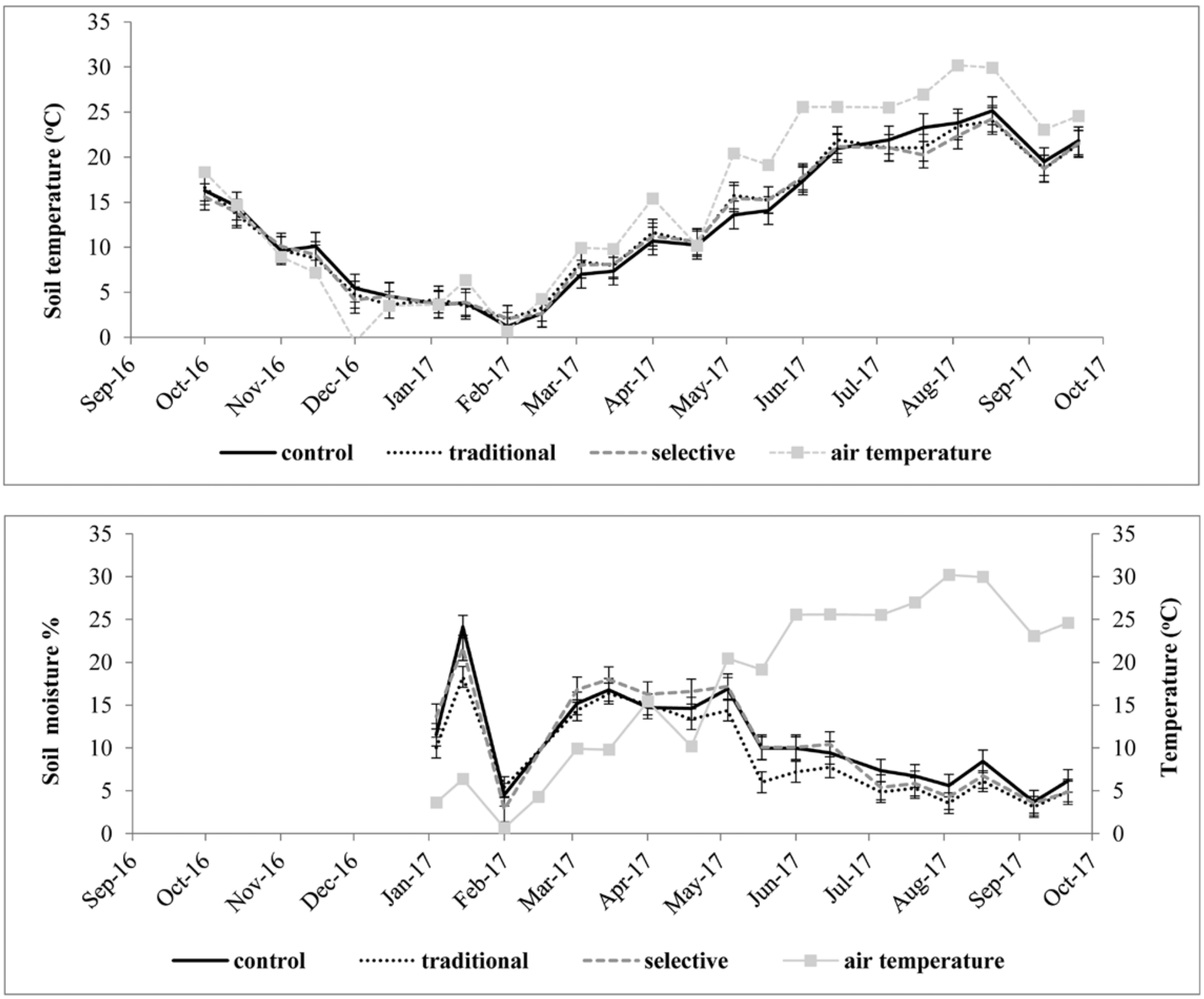

Figure 4. (a) Mean soil temperature of three replicates for three treatments over the study period, (b) mean soil moisture of three replicates for the three treatments from January 2017 to September 2017. Each data point for each treatment represents the mean of the three replicates for each sampling date. Each data point for air temperature represents the mean of the three treatments (control, traditional, and selective) for each sampling date. Control is marked by a black solid line, traditional is marked by black dots, selective is marked by a gray dashed line, and air temperature is marked by filled white squares. Error bars indicate the standard error.

Table 2. Mean annual cumulative greenhouse gas from three replicates and global warming potential ± SD in three thinning treatments.

Thinning treatments	CO_2 (kg C-CO_2 ha^{-1} y^{-1})	CH_4 (kg C-CH_4 ha^{-1} y^{-1})	N_2O (kg N-N_2O ha^{-1} y^{-1})	GWP (kg CO_2 eq ha^{-1} y^{-1})
Control	8243.94 ± 343.55	−1.46 ± 0.19	0.36 ± 0.18	30,312 ± 1313
Traditional	7983.96 ± 764.78	−1.58 ± 0.07	0.51 ± 0.09	29,422 ± 2805
Selective	7211.44 ± 842.19	−2.03 ± 0.02	0.32 ± 0.07	26,482 ± 3077
Total mean	7813.11 ± 345.38	−1.69 ± 0.18	0.40 ± 0.07	28,738 ± 1269

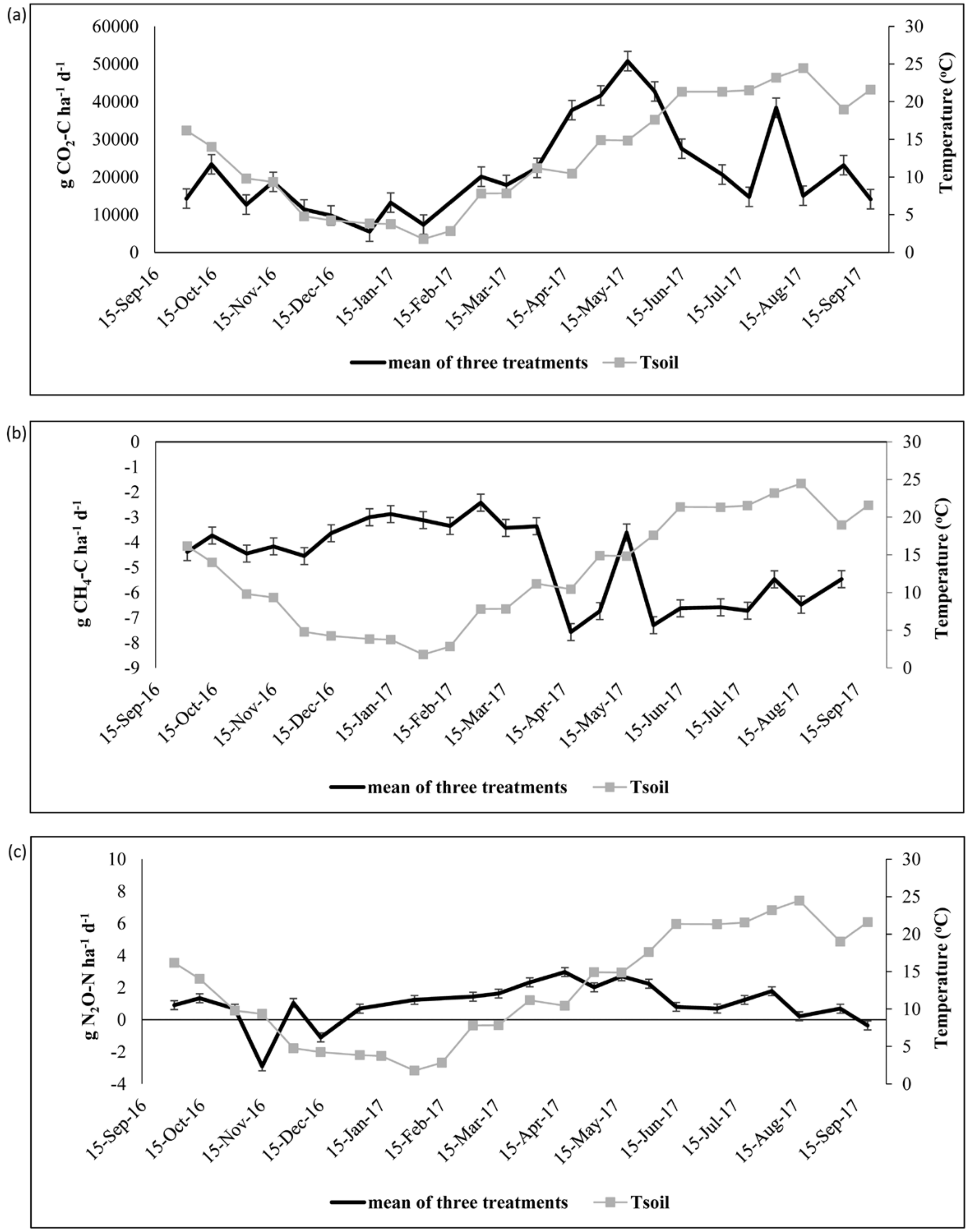

Figure 5. Temporal pattern of soil GHG of (a) CO_2, (b) CH_4, and (c) N_2O over the study period. Each data point of each GHG represents the mean of the three treatments (control, traditional, selective) of three replicates. Each data point of soil temperature (Tsoil) represents the mean of three replicates. Treatments are marked by a black solid line, and Tsoil is marked by filled white squares. Error bars indicate the standard error.

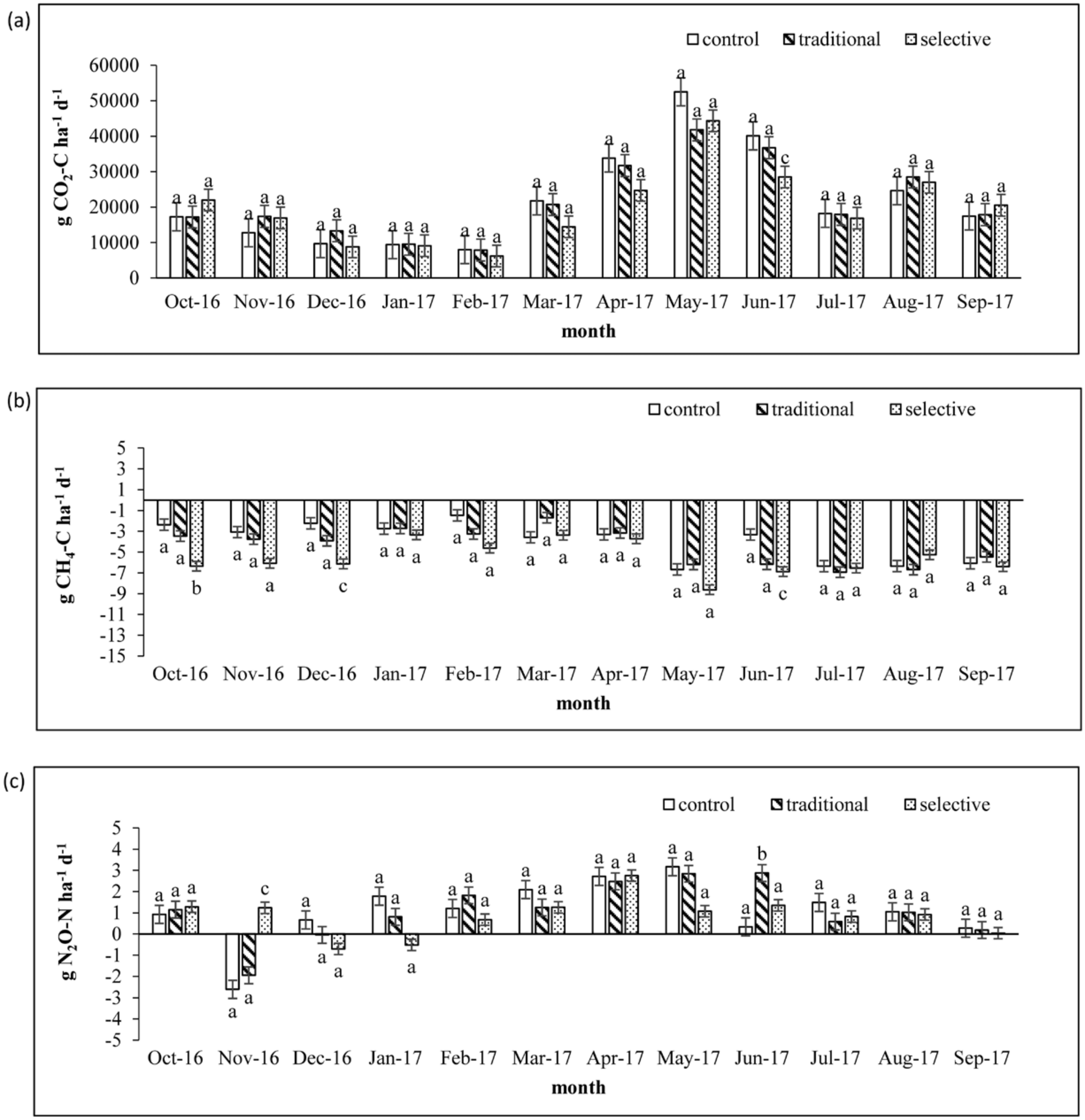

Figure 6. Monthly mean GHG of (a) CO_2, (b) CH_4, and (c) N_2O in three different thinning treatments over the study period. Each data bar represents the monthly mean data of three replicates of two sampling dates. Error bars indicate the standard error of the monthly mean GHG for each treatment. Different lower-case letters indicate significant differences (P < .05) among treatments within a month.

we divided our data into three thinning treatments and five periods (Table 3).

The Q_{10} values among thinning treatments decreased in the order control > selective > traditional, with the highest $R^2 = 0.27$ being observed in selective thinning (Figure 7). A significant effect of Tsoil on CO_2 fluxes was observed throughout the study year apart from summer, with an exponential model explaining about 40 percent of the increase in CO_2 fluxes with Tsoil (CO_2 fluxes $= 6582.60e^{0.099T}$, $R^2 = 0.40$) (Table 3, Figure 8a). In contrast, during summer with high Tsoil and low Msoil, no significant prediction of soil respiration was observed ($R^2 = 0.08$) (Figure 8b). With respect to seasonal variation, the Q_{10} value decreased in the order winter > spring > autumn > summer (Table 3). Tsoil was significantly correlated with CO_2 fluxes, but not Msoil. Further, the combination of the two, through the multiple stepwise regression, explained approximately 25 percent of CO_2 flux variation (Table 4).

CH_4

CH_4 uptake was observed in all three treatments, with forest soil acting as a net CH_4 sink. The monthly trend of CH_4 uptake was similar among treatments, ranging from –8.68 g to –1.47 g CH_4-C ha^{-1}d^{-1} (Figure 5b). The overall mean cumulative uptake for all treatments was –1.69 ± 0.18 kg CH_4-C ha^{-1}y^{-1} with a mean monthly rate of –4.26 g CH_4-C ha^{-1}d^{-1}, with the highest cumulative uptake observed to selective thinning ($P = .003$) (Table 5).

A significant effect of monthly variation on CH_4 uptake was observed, explaining 36.20 percent of CH_4 uptake variation ($F[3.16, 18.98] = 3.41$, $\eta^2 = 30.20$ percent, $P = .037$). The largest mean uptake for all treatments was observed in May (–7.156 ± 0.74 CH_4-C ha^{-1} d^{-1}) and the lowest in March (–2.893 ± 0.59 CH_4-C ha^{-1} d^{-1}) (Figure 5b). Significant ($P < .05$) differences in mean CH_4 uptake were observed among treatments in June, October, and December (Figure 6b). Thinning was able to significantly affect CH_4 uptake ($F[1.8, 117.55] = 5.88$, $\eta^2 = 8.4$ percent, $P = .005$), although the effect was weak. A post hoc test using Bonferroni correction showed that CH_4 uptake was significantly ($P = .001$) higher at selective –5.579 ± 0.358 g CH_4-C ha^{-1} d^{-1} than control –3.953 ± 0.369 g CH_4-C ha^{-1} d^{-1}, but not traditional –4.345 ± 0.365 g CH_4-C ha^{-1} d^{-1} ($P = .089$). The CH_4 uptake was negatively correlated with Msoil and positively with Tsoil (Table 4).

N_2O

In all three treatments, N_2O followed a similar pattern with observed values very close to zero ranging from –2.60 to 3.17 g N_2O-N ha^{-1} d^{-1} with a mean monthly rate of 1.01 g N_2O-N ha^{-1} d^{-1} (Figure 5c). The mean annual N_2O fluxes ranked among treatments in the order control (1.091 ± 0.42 g N_2O-N ha^{-1} y^{-1}) > traditional (1.086 ± 0.39 g N_2O-N ha^{-1} y^{-1}) > selective (0.850 ± 0.26 g N_2O-N ha^{-1} y^{-1}), although these rates were not significantly different from each other. Thus, selective thinning was able to save 44 g N_2O-N ha^{-1} y^{-1} with respect to control and 189 g N_2O-N ha^{-1} y^{-1} with respect to traditional. The overall mean cumulative N_2O flux, combining all treatments, was 0.40 ± 0.07 kg N_2O-N ha^{-1} y^{-1} (Table 2).

No significant differences were observed in the N_2O flux among months ($P = .212$). The lowest mean N_2O flux was observed as uptake in November (–1.11 ± 1.18 g N_2O-N ha^{-1} d^{-1}) and the highest emission in April (2.65 ± 1.28 g N_2O-N ha^{-1} d^{-1}). No significant differences in N_2O flux occurred among treatments within any given month, expect for November, where selective (1.14 ± 1.09 g N_2O-N ha^{-1} d^{-1}) was significantly ($P = .002$) higher than control (–1.30 ± 5.57 g N_2O-N ha^{-1} d^{-1}) but not significantly higher than the traditional thinning treatment (–0.69 ± 3.19 g N_2O-N ha^{-1} d^{-1}) (Figure 6c).

Table 3. Q_{10} prediction for three thinning treatments, control, traditional, and selective, and for five different periods.

Thinning treatments	*a*	*k*	Q_{10}	R^2
Control	9387.70	0.049	1.63	0.21
Traditional	11853.00	0.033	1.39	0.14
Selective	9356.70	0.044	1.55	0.27
Periods	***a***	***k***	**Q_{10}**	**R^2**
Autumn	10369.00	0.022	1.24	0.07
Winter	4683.60	0.163	5.12	0.39
Spring	10299.00	0.088	2.41	0.35
Summer	71229.00	–0.056	0.57	0.08
October-June	6582.60	0.099	2.68	0.40

Note: *a*, *k* = predictive parameters which were calculated by exponential model for each treatment and period separately, R^2 = coefficient of determination.

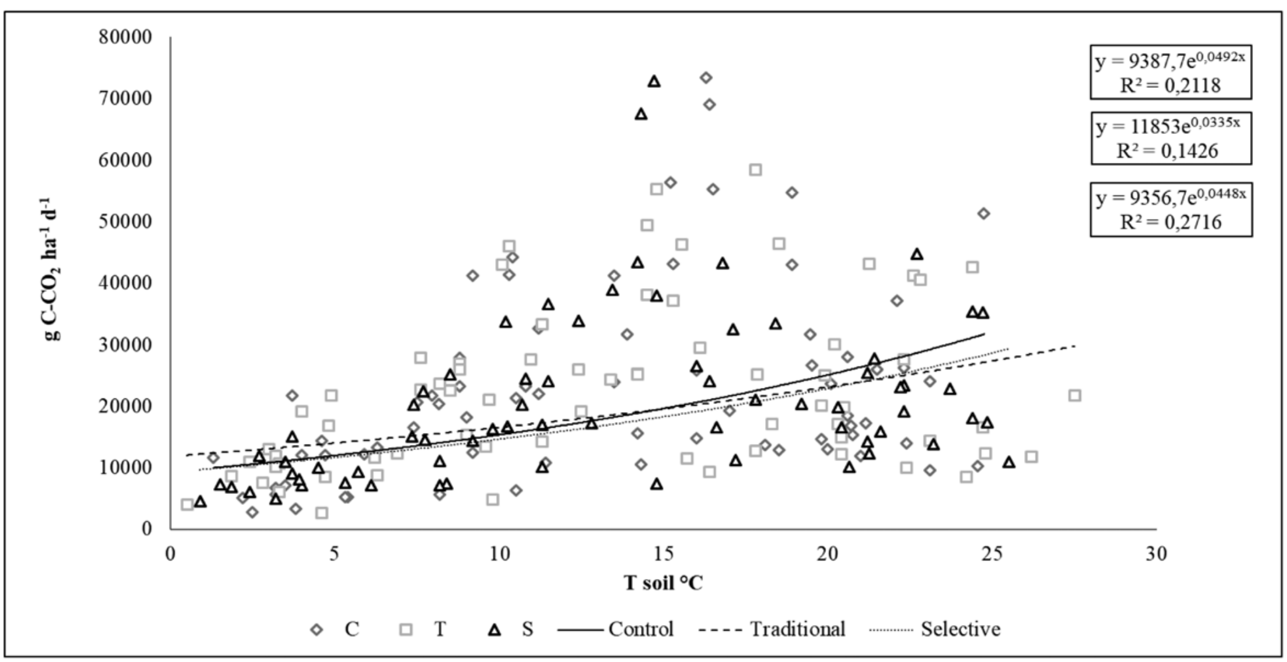

Figure 7. Exponential regression of mean CO_2 fluxes and Tsoil of three thinning treatments, C (control), T (traditional), and S (selective), of the three replicates throughout the study year. Each data point represents a mean flux of three replicates.

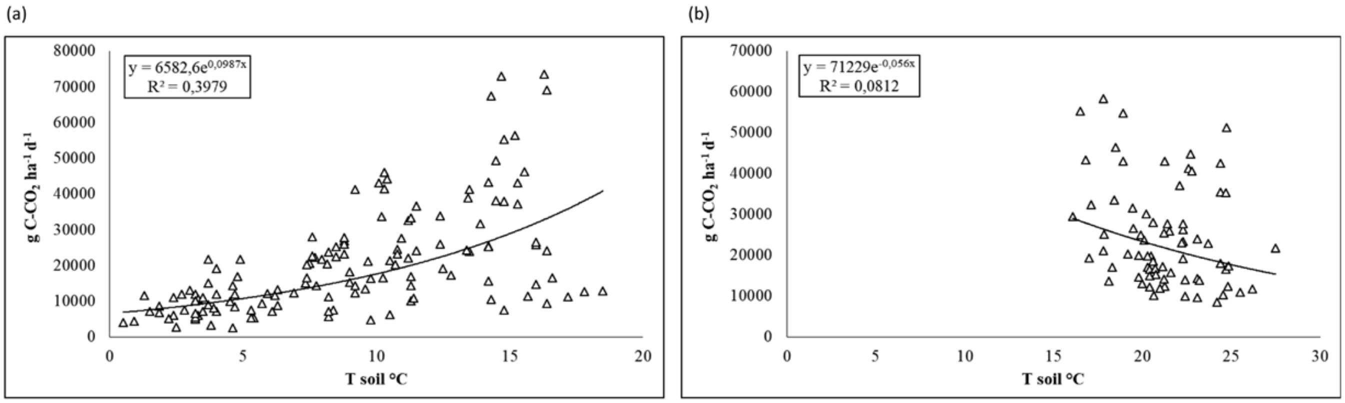

Figure 8. (a) Exponential model of mean soil respiration and Tsoil during the study period without summer (October to June). Each data point represents a mean flux of three replicates. (b) Exponential model of mean soil respiration and Tsoil during the summer. Each data point represents a mean flux of three replicates.

Table 4. Pearson correlation analysis among greenhouse gas fluxes and environmental factors, where *r* = correlation coefficient, and *P* value = significant level, and stepwise multiple linear regression between greenhouse gas fluxes and the combination of environmental factors, where *R* = correlation coefficient, R^2 = coefficient of determination, and *P* value = significant level.

	Pearson correlation analysis		Multiple linear regression
	Tsoil	Msoil	Tsoil + Msoil
CO_2	$r = 0.348^{**}$, $P = .000$	$r = 0.045$, $P = .578$	$R^2 = 0.227$, $P = .000$
CH_4 uptake	$r = 0.337^{**}$, $P = .000$	$r = -0.177^{*}$, $P = .033$	
N_2O	$r = 0.051$, $P = .483$	$r = 0.208^{*}$, $P = .014$	$R^2 = 0.094$, $P = .001$

Note: Msoil, soil moisture; Tsoil, soil temperature.
**Correlation is significant at the .01 level (one-tailed).
*Correlation is significant at the .05 level (one-tailed).

Tsoil did not, in itself, affect N_2O variation, but Msoil was significant correlated with N_2O (Table 4). A stepwise MLR did show a combined weak effect of Tsoil and Msoil on N_2O flux (F[2,136] = 7.04, $R^2 = 0.09$, $P = .01$), but only explained 9 percent of the variation in N_2O flux.

GWP and Annual GHG Fluxes

Soil CO_2 emissions (28,628 ± 1265 kg CO_2 ha^{-1} y^{-1}) contributed significantly to GWP at 99.2 percent. This can be explained by the fact that, despite the large warming potential of CH_4 and N_2O (equivalents are 34 and 298 times higher than CO_2, respectively) (Myhre et al. 2013), CH_4-CO_2 equivalent uptake was able to offset N_2O-CO_2 equivalent emissions by about 68 percent, resulting in a net total of only 0.05 percent, accounting for the largest contribution to CO_2 on GWP (Table 5).

One year after thinning, the mean GWP from all three GHGs (CO_2 + CH_4 + N_2O) was 28,738 ± 1,269 kg CO_2 eq ha^{-1} y^{-1}, ranking among treatments in the order control > traditional > selective (Table 5). A comparison of GWP among thinning treatments showed that selective thinning was able to reduce CO_2 emissions by 2,940 kg CO_2 eq ha^{-1} (10 percent) relative to the traditional thinning treatment and 3,829 kg CO_2 eq ha^{-1} (14.5 percent) compared to control. The CH_4–CO_2 equivalent uptake in the selective thinning was about 28 percent higher than control and 29 percent higher than the traditional thinning treatment, whereas N_2O-CO_2 equivalent emissions were about 36 percent lower in selective than traditional and 37 percent lower than control.

Discussion

Thinning Effect on CO_2 Fluxes

Several previous studies have evaluated the effect of thinning on CO_2 efflux and soil respiration. Most have indicated a decrease in soil respiration after thinning such as in pine plantations in the United States (Tang et al. 2005), fir plantations in China (Da-Lun et al. 2009), and sitka spruce plantations in Ireland (Saunders et al. 2012). A slight increase was observed in a mixed conifer forest in Sierra Nevada, United States (Ma et al. 2004) and in pine plantations in Virginia, United States (Selig and Seiler 2004). In our study, there were no observed significant impacts of thinning on soil CO_2 fluxes one year after thinning treatments. This is in agreement with other studies conducted in a coniferous plantations (Sitka spruce) in Ireland (Saunders et al. 2012) and in loblolly pine (*Pinus taeda*) in the southern United States (Selig and Seiler, 2004) showing that thinning did not directly affect (statistically significantly) soil CO_2 fluxes. Gathany and Burke (2014) also indicated that, although there was a significant effect of sampling date on CO_2 fluxes from ponderosa pine forest soil in Colorado, there was no significant effect of thinning on them.

Although not significant, we documented a 12.9 percent reduction in CO_2 emissions in the more intensive selective thinning treatment compared to a 2.8 percent reduction in the traditional thinning treatment. We hypothesized that the reduction in autotrophic respiration (root respiration) resulting from tree cutting thereby decreased soil respiration (Tang et al. 2005, Da-Lun et al. 2009) and thus CO_2 emissions. However, this reduction may be only temporary, as one would expect a subsequent increase in

Table 5. Mean annual cumulative greenhouse gas CO_2 equivalent and GWP ± (SE) of three thinning treatments of three replicates.

Thinning treatments	CO_2 (kg CO_2 ha^{-1} y^{-1})	CH_4 (kg CO_2 eq ha^{-1} y^{-1})	N_2O (kg CO_2 eq ha^{-1} y^{-1})	Global warming potential (kg CO_2 eq ha^{-1} y^{-1})
Control	30,206.99 ± 1,258	–66.51 ± 8.85	171.06 ± 87.87	30,312 ± 1,313
Traditional	29,254.36 ± 2,802	–71.79 ± 3.27	239.16 ± 42.63	29,422 ± 2,805
Selective	26,423.73 ± 3,085	–92.30 ± 1.24	150.70 ± 33.71	26,482 ± 3,077
Total mean	28,628.36 ± 1,265	–76.87 ± 8.22	186.97 ± 32.72	28,738 ± 1,269

autotrophic respiration because of increases associated with understory plant growth (Sullivan et al. 2008). Root respiration and rhizosphere C decomposition have a significantly higher temperature sensitivity of soil respiration Q_{10} than soil microbial respiration (Boone et al. 1998). In our study, Q_{10} changed in response to thinning. A reduction in Q_{10} was observed in both thinning treatments compared to the control. Thus, we assumed that the reduction in root biomass resulting from thinning caused a decrease in Q_{10} in the thinned plots because of the smaller contribution of autotrophic respiration to the total respiration system (Tang et al. 2005).

Seasonal Variation and Effect of Soil Edaphic Factors on CO_2 Fluxes

Season significantly affected CO_2 flux variation, explaining more than 50 percent. CO_2 flux varied significantly between seasons, with a peak occurring in the spring period, during the growing season. This was possibly caused by a direct effect of soil temperature on microbial activity increasing heterotrophic respiration. In addition, the growth of trees would also have triggered autotrophic respiration. We assume that the combined increase in microbial and root respiration at the start of the growing season might coincide with high values of total soil respiration (Tang et al. 2005, Meacham 2013).

The mean annual CO_2 fluxes (653.84 g C-CO_2 m^{-2} y^{-1}) in our study were lower than those measured from a thinned deciduous forest (801 g C-CO_2 m^{-2} y^{-1}) in UK after thinning (Yamulki and Morison 2017), but our results in summer (97 mg C m^{-2} h^{-1}) were similar to those measured by Sullivan et al. (2008) during the summer months in a ponderosa pine forest in the southwestern United States (109 mg C m^{-2} h^{-1}).

In the current study, soil environmental factors (soil temperature and soil moisture) contributed significantly to the temporal variation in CO_2 flux. Tsoil was the main driving factor for CO_2 emissions independent of thinning treatments. The strongest relation between CO_2 and Tsoil was observed over the period October to June. The effect of Tsoil on CO_2 fluxes in summer was negligible. Indeed, in summer a moisture limitation occurred at about 55 percent, which offset the expected increase because of temperature. Soil moisture alone did not affect CO_2, but combining Tsoil and Msoil, approximately 25 percent of CO_2 variation was explained, more than with Tsoil alone.

Although there was a combined influence of Tsoil and Msoil on the temporal variation in the CO_2 flux, there were no indications of differential effects in thinned compared with unthinned plots, as reported in other studies (Tang et al. 2005, Sullivan et al. 2008). In Europe, a study in deciduous forests in the south-east of England also showed that CO_2 fluxes responded to soil temperature and moisture similarly in both thinned and unthinned sites (Yamulki and Morison 2017). Similar results has also been observed in a *Picea abies* forest in Norway, where the alteration of soil respiration in thinned plots was likely caused by factors other than soil temperature and moisture, which followed similar patterns among treatments (Nilsen et al. 2008).

CH_4

Thinning Effect on CH_4 Oxidation

In this study, CH_4 oxidation was observed in all treatments, with the pine forest acting as a CH_4 sink. The high percentage (60 percent) of sand content in the soil, as expected for Caprisols, at the study site, contributed to a good soil aeration that led to a significant CH_4 sink (Rosenkranz et al. 2006). This is consistent with studies arguing that Mediterranean ecosystems can act as a sink of atmospheric CH_4, such as in Portugal (Shvaleva et al. 2014), Italy (Rosenkranz et al. 2006, Savi et al. 2016), and Spain (Merino et al. 2004), as well as in other European forests such as conifer (Steinkamp et al. 2001) and beech forests (Butterbach-Bahl and Papen 2002) in Germany.

Our results indicated that thinning was able to increase CH_4 oxidation significantly. Other studies conducted in coniferous forests in the United States reported that thinning did not affect CH_4 uptake significantly (Sullivan et al. 2008, Gathany and Burke 2014), whereas in a poplar plantation in China, a higher thinning intensity did shift the forest soil from CH_4 sink to CH_4 source (Fang et al. 2016).

In our study, the mean CH_4 uptake increased by 38.9 percent in the selective thinning treatment compared to the control. By contrast, the traditional thinning treatment increased CH_4 uptake by only 10.7 percent compared to the control. The creation of small gaps has been shown to increase soil organic matter, microbial biomass, and C/N, and alter microbiological processes in the upper soil horizons of Mediterranean conifer forests (Butterbach-Bahl and Papen 2002, Muscolo et al. 2017). In addition, CH_4 uptake increased under well-aerated soils with a high sand fraction (Rosenkranz et al. 2006). Thus, the creation of small gap sizes because of thinning combined with the high content of sand in the soil texture at the study site might explain the increase in CH_4 uptake after thinning.

Seasonal Variation and Effect of Soil Edaphic Factors on CH_4 Fluxes

CH_4 uptake was significantly and positively related to soil temperature. The highest values of CH_4 uptake were measured in summer with high soil temperature and low soil moisture, which were almost double that in winter. These results confirmed what has been reported in other studies. For example, Rosenkranz et al. (2006) showed increases in CH_4 uptake in seasons with high soil temperatures in Mediterranean pine forests. Similarly, Castaldi and Fierro (2005) recorded the highest CH_4 oxidation rates during dry and warm periods. Fang et al. (2016) also observed a significant positive correlation of CH_4 uptake with soil temperature in poplar plantations in China.

Our results also showed a weak negative correlation between CH_4 uptake and soil moisture one year after thinning implementation. This result is also consistent with other studies that have illustrated significant negative correlations between CH_4 uptake and soil moisture (Rosenkranz et al. 2006, Fang et al. 2016). Sullivan et al. (2008) also reported that CH_4 oxidation was low at high soil water contents because of the limitations of CH_4 diffusion into soil.

N_2O

Thinning Effect on N_2O Fluxes

The annual fluxes in our study (38.77 mg N_2O-N m^{-2} y^{-1}), ranging from –95.25 to 115.70 mg N_2O-N m^{-2} y^{-1}, are higher than those reported by Yamulki and Morison (2017) in conifer forests in the UK (mean 22 mg N_2O-N m^{-2} y^{-1}) and by Bernal et al. (2003) in broadleaved forests of Spain (0–25 mg N_2O-N m^{-2} y^{-1}).

Our temporal results would suggest that this peri-urban conifer forest in Greece can act temporally as a sink of atmospheric N_2O in the coldest months and a weak source of N_2O fluxes in warmer months with high soil temperatures and low soil moisture content. This is consistent with other studies indicating that forest soil in the Mediterranean area can both emit and consume N_2O, even resulting in a net sink of N_2O (Bernal et al. 2003, Butterbach-Bahl and Kiese 2005). However, Rosenkranz et al. (2006) indicated that a coniferous forest soil in Italy was able to act as an N_2O sink in both periods, with a higher uptake observed in winter with mild, wet weather than in spring with warm, dry weather.

There were no significant differences in N_2O fluxes among thinning treatments, suggesting that changes in environmental variables caused by thinning were not so strong to alter the balance between N immobilization and mineralization. Moreover, critical levels of Msoil were never reached, even during thinning intervention.

Seasonal Variation and Effect of Soil Edaphic Factors on N_2O Fluxes

Our results indicated a significant seasonal effect on N_2O fluxes, and any change throughout the year might be explained by the effect of altered soil environmental factors on microbial processes leading to N_2O production. Indeed, the combination of Msoil and Tsoil explained the monthly variation of N_2O fluxes, confirming the role of soil moisture as the prevailing environmental factor influencing N_2O exchange with the atmosphere (Rosenkranz et al. 2006, Butterbach-Bahl et al. 2013). Moreover, a high soil moisture content might explain the N_2O uptake observed during the winter months with high rain precipitation. In fact, as demonstrated by Butterbach-Bahl et al. (2013), a complete process can occur under high water-content conditions, leading to N_2O consumption and N_2 production in the final step of denitrification. Thus, we assumed that the temporal N_2O uptake during winter months with high rain precipitation can be a consequence of an optimum soil moisture content, resulting in soil conditions stimulating complete denitrification. However, the few data available and the complexity of mechanisms involved highlight the need for further research to examine in situ the soil N_2O production processes and their interpretation with soil micro-organism components.

GWP

Soil CO_2 emissions contributed significantly (about 99 percent) to total GWP independent of thinning. Overall, about 187 kg CO_2 eq ha^{-1} y^{-1} was emitted as N_2O, and 60 percent of it was offset by CH_4 uptake of 77 kg CO_2 eq ha^{-1} y^{-1}. Thus, CO_2 remained the dominant gas contributing to total GWP. This result is consistent with other studies (Zerva and Mencuccini 2005, Yamulki and Morison 2017).

Our results indicated that one year after thinning, both CO_2 emissions and N_2O-CO_2 equivalents were reduced, whereas CH_4-CO_2 equivalent uptake increased, resulting in a 15 percent reduction in GWP with selective thinning but only negligible (about 3 percent reduction) in traditional thinning when compared to the unthinned control.

Conclusion

This study investigated the effects of three thinning interventions on GHG emissions and their relation with soil environmental factors in a peri-urban conifer forest in Greece one year after thinning. The GHG variation in this study was largely driven by changes in soil environmental factors. Soil temperature was the dominant factor controlling CO_2 and CH_4 efflux, whereas soil moisture was related, albeit weakly, with N_2O variability. A significant effect of Tsoil on CO_2 fluxes has been observed throughout the study year apart from summer, explaining about 40 percent of the increase of CO_2 fluxes during this period with mean $Q_{10} = 2.68$. However, a combination of soil temperature and moisture did improve the relation with CO_2 and N_2O variability. Although not significant, the reduction in CO_2 emissions was possibly related to an indirect effect of both soil heterotrophic and autotrophic respiration and soil environmental factors alteration caused by thinning. Increased CH_4 uptake occurred in both thinning treatments, with the greatest increase being detected in the more intensive thinning treatment.

Our results suggested that this peri-urban conifer forest in Greece can act temporally as a sink of atmospheric N_2O during the coldest months, but a weak source of N_2O fluxes in the warmer, drier months. Regarding GWP, CO_2 was the dominant GHG in all treatments. Reduction in GWP was observed with both thinning treatments, markedly in selective thinning, giving some initial evidence that high-intensity thinning in this conifer forest in Greece is a potential management option for global change mitigation.

This study was grounded predominantly in evaluating for the first time the response of annual GHG fluxes to thinning in a conifer forest soil in Greece, contributing significantly to the creation of a GHG fluxes database from a conifer ecosystem in Greece. Hence, because of limited knowledge of the response of thinning implementation on the forest soil–atmospheric GHG (CO_2, CH_4 and N_2O) exchange system in the Mediterranean region, this study also contributes to the enrichment of the GHG fluxes database from the Mediterranean forest ecosystem. However, long-term studies in other conifer forest ecosystems in Greece and in the Mediterranean Basin are essential to extend the responses of GHG fluxes to thinning and thus to extract definite conclusions for management options in global change-mitigation targets.

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