Science of the Total Environment 651 (2019) 713–724

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Short-term effects of thinning on soil CO_2, N_2O and CH_4 fluxes in Mediterranean forest ecosystems

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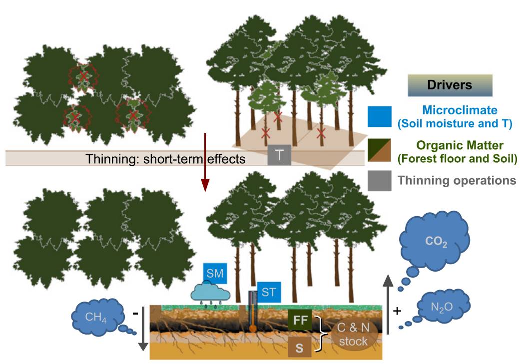
HIGHLIGHTS

- We analysed site-related GHG fluxes in two Mediterranean pine forests after thinning.
- Soil moisture content and organic matter availability affected CO_2 emission patterns.
- Short-term CO_2 emissions increased after disturbance induced by logging operations.
- Small gaps created with selective thinning increased CH_4 uptake.
- Low N_2O emissions were found in both sites and were not affected by thinning.

GRAPHICAL ABSTRACT



ARTICLE INFO

Article history:
Received 2 July 2018
Received in revised form 17 September 2018
Accepted 18 September 2018
Available online 20 September 2018

Editor: Elena PAOLETTI

Keywords:
Forest floor
Global warming potential
Green-house gas fluxes
Pine plantations restoration
Soil temperature
Soil moisture

ABSTRACT

In Mediterranean ecosystems an increasing demand for in situ trace gas exchange data is emerging to enhance the adaptation and mitigation strategies under forest degradation. Field-chamber green-house gas fluxes and site characteristics were analysed in two Mediterranean peri-urban pine forests showing degradation symptoms. We examined the effect of different thinning interventions on soil CO_2, CH_4 and N_2O fluxes, addressing the relationships with the environmental variables and C and N contents along forest floor-soil layers.

Soil temperature resulted as the main driving variable for CO_2 efflux and CH_4 uptake. Soil moisture content and organic matter availability affected CO_2 emission patterns in the two sites. N_2O fluxes showed a positive correlation with soil moisture under wetter climatic conditions only. GHG fluxes showed significant correlations with C and N content of both forest floor and mineral soil, especially in the deepest layers, suggesting that it should be considered, together with environmental variables when accounting GHG fluxes in degraded forests.

Short-term effects of thinning on CO_2 emissions were dependent on disturbance induced by logging operations and organic matter inputs. After thinning CH_4 uptake increased significantly under selective treatment, independently from specific site-induced effects. N_2O fluxes were characterized by low emissions in both sites and were not affected by treatments. Soil CO_2 efflux was the largest component of global warming potential (GWP) from both sites (11,553 kg ha^{-1} y^{-1} on average). Although it has a large global warming potential, N_2O contribution to GWP was about 131 kg CO_2eq ha^{-1} y^{-1}. The contribution of CH_4-CO_2 equivalent to total GWP showed a

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https://doi.org/10.1016/j.scitotenv.2018.09.241

clear and significant CH_4 sink behaviour under selective treatment (36 kg ha^{-1} y^{-1} on average). However, in the short-term both thinning approaches produced a weak effect on total GWP.

1. Introduction

Pine species have been largely used for land restoration in the Mediterranean basin since the late nineteenth century. The traditional strategy for reforesting degraded lands was to plant fast-growing pioneer species, usually pines (Gil and Prada, 1993), to facilitate the introduction (either artificial or natural) of late-successional broadleaves able to reach a higher biodiversity and more resilient and resistant ecosystems (Barbéro et al., 1998; Pausas et al., 2004).

Nowadays, the most of these pine stands are concluding their rule of pioneers' species and the lack of timely silvicultural interventions negatively affected stands adaptability to natural processes (i.e., secondary successions). Moreover, these stands often present mechanical stability problems (Cantiani and Chiavetta, 2015; Marchi et al., 2017). From the National Forest Inventory, 31% of Italian pine forests in the Mediterranean zone show degradation symptoms with many dead, fallen and/or damaged trees for a total of 462,568 ha.

The lack of silvicultural interventions can affect the annual C sequestration rate due to a decrease in canopy cover and regeneration (Flint and Richards, 1994), reducing the potential sink capacity of these forests. In most cases these pine stands, densely overgrown with large deadwood, have become a potential source of greenhouse gases (GHGs) through decomposition of remaining plant material and soil carbon and altering the balance between gross photosynthesis and respiration (Law et al., 2004; Misson et al., 2005).

A major source of carbon and organic matter available for decomposition is represented by plant litter, with leaf litter accounting for 22% to 81% of total litter annual production and contributing up to >70% of the annual N input via litter fall (Bauer et al., 2000). Litter decomposition is an important source of CO_2 to the atmosphere; mineralization of the annual litter fall contributes to approximately half of the CO_2 output from the soil, and the average global annual CO_2 flux from the soil is estimated to be about 98 Pg C (Bond-Lamberty and Thomson, 2010), an amount comparable to that of global gross primary production.

Soil is a predominant component of C cycle, storing two-three times more C than the atmosphere. The significant role of soil in the global C balance depends on its potential to store C in pools with a slow turnover. Only small fractions of new C inputs into soils will become long-term SOC, whereas the largest fraction will be respired back to the atmosphere. Furthermore, soils have been reported to be a key factor for the overall response of the terrestrial biosphere to global change.

How thinning affects soil–atmospheric fluxes of GHGs is poorly understood. Contrasting results have been reported on soil CO_2 effluxes under different forest management practices (Zerva and Mencuccini, 2005). Determining how restoration thinning changes soil C fluxes is necessary to understand potential feedbacks between forest management activities and climate warming (Fang et al., 2016a, 2016b). Besides CO_2, thinning can have a significant effect also on methane (CH_4) and nitrous oxide (N_2O) fluxes by altering the environmental factors related to these fluxes, such as soil temperature, soil water content, decomposition of organic matter (Hendrickson et al., 1989), availability of substrate (Sahrawat and Keeney, 1986; Skiba and Smith, 2000), soil N dynamics (Smolander et al., 1998). Most studies assessing the impact of environmental variables and disturbances on pools and GHG fluxes from forest soil have been conducted in boreal or temperate forests and have mainly focused on CO_2, so few data are available for CH_4 and N_2O. Although the absolute quantities of CH_4 and N_2O emitted are small compared with those of CO_2, their importance depend on their larger global warming potential (GWP) that is respectively 34 and 298 times greater than CO_2 over a 100 year period (Myhre et al., 2013).

Enhanced N deposition due to increased human activity can directly promote N_2O emission by increasing inorganic N availability for microbial processes (Meng et al., 2011). Regions with elevated atmospheric N-deposition due to anthropogenic activity showed increased N_2O emissions (Butterbach-Bahl et al., 2002). However, inorganic N availability may not increase in response to increasing N deposition due to plant uptake and microbial immobilization (Edith et al., 2014). Atmospheric N deposition can be particularly important in peri-urban areas that are receiving contributions of N compounds from both urban and agricultural activities (García-Gomez et al., 2016). It is also hypothesized that increased N deposition will result in increased N_2O emissions in combination with reduced soil CH_4 oxidation and CO_2 emissions (Butterbach-Bahl et al., 2002; Ambus and Robertson, 2006).

Forest soils have also been identified as a significant sink for atmospheric CH_4, and it is estimated that CH_4 uptake of soils activities represent 3–9% of the global atmospheric CH_4 sinks (Prather et al., 1995). Well aerated forest soils seem to play a major role in this context (Papen et al., 2001). Uncertainty is associated, however, with global soil CH_4 consumption because of the few data available from Mediterranean-type ecosystems (Castaldi and Fierro, 2005), increasing the need for in situ trace gas exchange data (Butterbach-Bahl and Kiese, 2005; Rosenkranz et al., 2006). In this region warm and dry condition in summer and moderate precipitation in winter, might create favourable conditions for CH_4 oxidation throughout the year, making this area suitable for studying CH_4 fluxes dependence on meteorological parameters (Savi et al., 2016). To our knowledge, no studies in the Mediterranean basin have examined the effect of thinning practices on GHGs fluxes and their relationships with environmental factors and organic matter pools in litter and soil.

The main objectives of this study were: i) to quantify the short-term effect of different thinning treatments on soil CO_2, CH_4 and N_2O fluxes and ii) to investigate how soil temperature, soil moisture, C and N content in forest floor and soil influenced the GHGs production during and after the different thinning treatments applied. We hypothesized that thinning disturbance would significantly influence soil GHG fluxes due to modification of environmental factors, and that the response of CO_2, CH_4 and N_2O to the disturbance may differ, driven by site-specific induced effects.

2. Materials and methods

2.1. Study sites

The study sites are located in the peri-urban forests of Monte Morello (43° 51′ N–11° 51′ E, Italy) and Xanthi (41° 09′ N–24° 54′ E, Greece). Their main characteristics are shown in Table 1.

Monte Morello forest has been planted during 1910–1980 years on degraded soils affected by overgrazing, with the aim to restore the forest cover. The stands are characterized by a dominant crop layer of *P. nigra* and *P. brutia* and minor presence of *Cupressus sempervirens* and *Quercus cerris*. The understory layer is mainly occupied by *Cupressus arizonica* (with many decaying trees) and agamic regeneration of Mediterranean shade-tolerant broadleaves species such as *Fraxinus ornus* and *Acer campetris*. These stands show clear degradation symptoms with many dead, fallen and/or damaged trees, due to the absence of proper silvicultural practices. Deadwood is very widespread, reaching about 75.1 m^3 ha^{-1} divided in 80% of lying deadwood, 18% of standing dead

Table 1
Site characteristics. Climate: annual average precipitation and temperatures, Palmer drought severity index (PDSI) and Standardised Precipitation–Evapotranspiration Index (SPEI) for the period considered, gridded on a 0.5 × 0.5° network. Vegetation type is "Plantations of site-native species", mainly conifers (see text for species details).

	Monte Morello (IT)	Xanthi (GR)
Climate (1980–2014; CRU dataset)		
Min – mean – max T (°C)	9.3–13.3–17.9	8.5–13.3–18.0
Precipitation (mm)	876	526
PDSI; SPEI	−0.53; −0.04	−0.73; −0.12
Stand characteristics (pre-thinning)		
Vegetation type	EEA-EFTs code: 6.14.1	
Stand age (years)	55–65	50–60
Mean tree density (tree ha^{-1})	980	2626
Basal area (m^2 ha^{-1})	62.9	38.8
Mean height (m)	17.1	10.3
Soil characterization (pre-thinning)		
Soil type	Calcaric cambisols and cambic calcisols	Cambisols
pH – carbonates (%)	8.1–8.2	5.6–6.2
Sand (%) – clay (%)	38.4–27.1	60.0–17.5
Thinning rate		
Traditional (% of biomass)	23.9	31.5
Selective (% of biomass)	36.4	49.3

trees and 2% of stumps (De Meo et al., 2017). The soils present a loam or clay loam texture, they are rich in carbonates and show a moderately alkaline pH. The climate is typically Mediterranean, with a dry summer in which July and August are the driest months.

Xanthi forest is a part of Xanhti–Gerakas–Kimerion public forest. The elevation ranges from 100 up to 630 m. The planting activities began in 1936 and took place periodically up to 2007, even though most of them were made till 1973. In the reforestations *P. brutia* was mainly planted. *P. maritima*, *P. pinea*, *Cupressus* spp. and *P. nigra*, as well as some broadleaves like *Robinia pseudoacakia* have also been used. In many areas there is an understory of broadleaves (*Quercus* spp., *Carpinus orientalis* etc.). The amount of deadwood is about eight times less than that measured in the Monte Morello peri-urban forest (9.21 m^3 ha^{-1}) especially because the lying deadwood is regularly collected by households for domestic use or by the Forest Service to prevent forest fires (De Meo et al., 2017). The soils present a sandy clay and sandy clay loam texture and showed a moderately acidic pH. The climate is typically Mediterranean, with a dry summer period longer than Monte Morello site and in which July and August are the driest months.

2.2. Thinning treatments

Thinning interventions took place in September 2016 and were based on three silvicultural options (in triplicate): traditional thinning, selective thinning and absence of intervention (control).

In Monte Morello forest the traditional silvicultural treatment was a medium-heavy intensity 'thinning from below', which removed most of the dominated trees including also some co-dominant one. With the selective thinning, the best 100 trees per ha were selected according to vigour and stability ("positive selection") and their growth and development were actively promoted by removing competitors in the dominant layer (Marchi et al., 2017). According to our main purpose, the selective thinning promotes the growth and development of trees (groups of 2–3 trees in some cases) characterized by the best H/D (high/diameter ratio) and a large and symmetric crown which can guarantee the highest stand stability and C accumulation rates in the medium- to long-term run. Native broadleaves trees (*Quercus spp.*) with these characteristics were favoured. Moreover, considering that species planted in our stand are very light demanding, all the suppressed and sub-dominant trees were removed with the aim to avoid a consequent deadwood increasing, although these trees don't affect the growth of the selected ones and usually may not be removed in the conventional selective thinning. During the selective thinning the understory trees where coppiced with the release of one shoot per stool. This operation aimed at increasing the carbon sequestration of understory layer, avoiding a strong re-shooting at the same time.

In Xanthi forest, the traditional silvicultural treatment was a medium-heavy intensity thinning which removed mainly dead, bad formed, damaged and intermediate - suppressed trees of the overstory. Thinning of broadleaves was also made in order to have a uniform distribution of broadleaves in the understory. In the selective silvicultural treatment, both intense cuttings of overstory pines for the release of broadleaved trees and innovative thinning of the understory, were performed. In the overstorey, bad formed trees were primarily removed increasing cutting intensity over dense areas with broadleaves, while the best formed trees were retained. The intense thinning of the understory resembles "positive selection" thinning, where competitors of the best broadleaved trees were removed.

At the IT site the whole felled trees were brought to the landing site by skidding and logging operations lasted 3 months. At the GR site, the cut trees were separated in trunks and branches and only the trunks were moved to the nearest forest roads. Moreover, logging operations lasted only 1 month.

In both study sites, the selective thinning approach resulted to be more intense compared to the traditional one above all over denser areas with a broadleaves understory (Table 1).

2.3. Sampling design

In each site, nine demonstration areas each of about 1.5 ha representing three replicates for each silvicultural option, have been randomly selected. Within each demonstration area two monitoring plots (circular fixed-area of 531 m^2) has been established, for a total of 36 sampling points in the two sites.

One collar (30 cm in diameter) was randomly positioned and inserted in the soil in each plot, at least 5 cm depth, for a total of 18 collars per site. The exact height above ground was recorded for following calculation of exact chambers volume. After thinning operations, collars were replaced on the ground and left on site till the end of the study. The position of the collars before thinning was approximately found according to wooden stakes fixed and georeferenced at the centre of the plots also because only the small logging residues (e.g. twigs) were left on the soil surface where the collars were previously positioned.

Gas sampling was performed biweekly or once per month, depending on weather and personnel availability, and adjusting sampling dates and frequency to include specific events such as thinning operations. All measurements were taken between 9.30 and 14.30 to minimise changes in soil CO_2 effluxes associated with diurnal cycles (Davidson et al., 1998). Moreover, considering the time both for gas sampling events and for moving from one plot to another, the experimental design based on 18 sampling points was considered reasonable and satisfactory, since in this way the three different treatments were measured simultaneously and then repeated for the three replicates.

Two litter sampling were randomly performed within each plot near to the collar for gas sampling (for a total of 36 sampling points for each site). From the same position, soil samples were collected at depth of 0–10 and 10–30 cm within each plot. The post-thinning soil sampling was carried out 1 year after the previous one (corresponding to the pre-thinning sampling) to avoid a strong seasonal variability in microclimatic conditions (i.e. summer vs. autumn-winter) influencing mineralization processes and rates. The 1 year observation period between pre and post thinning soil sampling followed the contemporary forest floor sampling, which needs a whole growing season for a complete quantification of the fresh L horizon.

2.4. Measurements of soil gas fluxes

Gas sampling was performed in all treatment replicates within two 13 m radius plots for each replicate (for a total of 18 plots), biweekly or once per month.

The closed chamber method described in Adviento-Borbe et al. (2013) was used for measuring soil CO_2, CH_4 and N_2O fluxes, having i) the advantage to measure simultaneously the three most important GHGs (CO_2, N_2O and CH_4) and ii) a high precision and detection limits, compared for example to eddy covariance measurements taken near the boundary between thinned and unthinned forest sectors with relatively small size (Wilkinson et al., 2016).

During each gas sampling event, chambers were closed for 30 min with four gas sampling (at 0, 10, 20 and 30 min). Headspace gas samples was collected with air-tight 30 mL propylene syringes and was immediately pressurized into pre-evacuated 12 mL glass Exetainer® vials (Labco Ltd., Buckinghamshire, UK). The gas samples were analysed within 4 weeks of collection.

GHG concentrations were analysed using a GC-2014 gas chromatograph (Shimadzu Scientific) with a thermal conductivity detector (TCD) for CO_2, 63Ni electron capture detector (ECD) for N_2O and flame ionization detector (FID) for CH_4. Chamber gas concentrations were converted to mass per volume units using the Ideal Gas Law and measured chamber air temperatures and volumes. Fluxes of CO_2, N_2O and CH_4 were calculated using the slope of linear regression of gas concentration versus chamber closure time and the enclosed soil surface area. Fluxes were set to zero if the change in gas concentration during chamber enclosure fell below the minimum detection limit determined for the GC, and flux values will be rejected (i.e. treated as missing data) if they passed the detection test but had a $R^2 < 0.75$.

Estimates of cumulative CO_2, CH_4 and N_2O emissions and global warming potential (GWP) for each field replicate in the 6 months after thinning was obtained by summing daily fluxes over time. For predicting GHG fluxes on non-measurement days, an integration over time by linear interpolation was used, considering the difference between two measurement events divided by the number of days between the two (Adviento-Borbe et al., 2010). Further, CH_4 and N_2O fluxes were expressed in g C-CO_2 eq m^{-2}, using the climate warming factor on 100-year horizon equal to 34 and 298 for CH_4 and N_2O, respectively (Myhre et al., 2013).

2.5. Measurement of environmental factors

Simultaneously with each GHG flux measurement, soil temperature and volumetric soil water content were measured, every time a measurement of soil trace gas efflux was performed.

Soil temperature was measured with a probe (HI 9043 Hand Held Thermometer) at a depth of 5 cm. In an adjacent position, soil sampling was performed to measure soil moisture. Soil moisture of field samples was determined afterwards in laboratory by measuring fresh and dry weight after incubation at 105 °C for 24 h, and the water loss expressed as a per cent of oven-dry weight. Soil samples were collected next to each collar.

2.6. Measurements of C and N pools in forest floor and soil

Samples of organic layer were obtained by pressing a 625 cm^2 steel sheet sampling frame (10 cm deep) into the forest floor and collecting all organic material above the mineral soil (Kavvadias et al., 2001). The samples were transported to the laboratory, dried and then fractionated by hand sorting and sieving (<2 mm). In each of these samples the horizons L, F and H will be separated and carefully placed in plastic bags. The L horizon is composed of fresh or slightly discoloured, with no or weak breaking up, material. The F horizon is composed of medium to strongly fragmented material with many mycelia and thin roots and the H horizon is a humified amorphous material, containing highly decomposed (i.e., unrecognizable) plants parts (Kavvadias et al., 2001; Hoosbeek and Scarascia-Mugnozza, 2009). Mineral soil will be removed after successive sieving (10–5 and 2 mm mesh stainless steel sieves).

The subsamples of each fraction were then ground in a Willey mill to pass a 500 μm mesh stainless steel sieve before being analysed.

Nitrogen and carbon contents of homogenized samples from each forest floor fraction were measured by dry combustion on a Thermo Flash 2000 NC soil analyzer (Fisher Scientific, Waltham, MA, USA). To this aim, 10 to 20 mg samples were weighed into Sn capsules and % of N and C was measured by thermal conductivity detector.

The soil samples were air dried and sieved through a 2-mm mesh. For C and N quantitative analysis and C chemical fractionation, soil sub-samples were grinded and homogenized to 0.5 mm. Total organic C (TOC) and total N (TN) contents in the bulk soil were measured by dry combustion on a Thermo Flash 2000 CN soil analyzer. To this aim, 20 to 40 mg soil were weighed into Ag-foil capsules and pre-treated with 10% HCl until complete removal of carbonates.

Further, undisturbed soil samples were collected for soil bulk density (BD) measurement, to calculate soil organic C stock for each depth. To this aim, undisturbed 100 cm^3 soil cores were collected from each pit with a hammer-driven liner sampler (Eijkelkamp, The Netherlands). The samples were dried at 105 °C until constant weight and the BD calculated by the ratio between the dry weight and the soil core volume (Blake and Hartge, 1986).

2.7. Statistical analysis

The effect of thinning on GHG fluxes was evaluated during and 3–-6 months after thinning operations, with the aim to investigate the short-term effect. Differences in gas fluxes among treatments were analysed using a one-way analysis of variance (ANOVA). The Bonferroni method was used for multiple comparison correction of the significance levels. Data were tested for normal distribution by applying the Kolmogorov–Smirnov test and Barlett test for the homogeneity of variance. When the normality test failed, the non-parametric Kruskal–Wallis test for analysis of variance by ranks and the Wilcoxon-Mann-Whitney (W) non-parametric test for independent samples were performed. We also used a one-way 'repeated measurements' ANOVA to determine differences in gas fluxes at each measurement date, considering the treatment applied as source of variation and using the individual measurements as replicates.

Multiple linear regression model (MLRM) assuming the residuals were independent and normally distributed $(0, \sigma^2)$, model analysis of variance and linear correlations were performed to evaluate the effects of environmental variables as well as C and N pools in forest floor and soil on GHG emissions following thinning. Variance inflation factor (VIF) analysis was applied to explored the co-linearity among explanatory variables.

Since CH_4 fluxes were always found negative, for correlation and regression analyses we transformed all data as absolute values (positive), considering CH_4 uptake as variable.

Data analysis was carried out using the free software environment R (R Development R Core Team, 2014. http://cran.r-project.org/).

3. Results

3.1. Effect of thinning on C and N pools

3.1.1. Forest floor

Before thinning, principal component analysis of the forest floor horizon patterns [using total mass, C and N stock (kg m^{-2}) as variables] featured a clear and distinct grouping among L, F and H horizons in both sites (Fig. 1), with 97.8 and 95.8% of the common variance explained by the two principal components at the Italian (IT) and Greek (GR) sites, respectively.

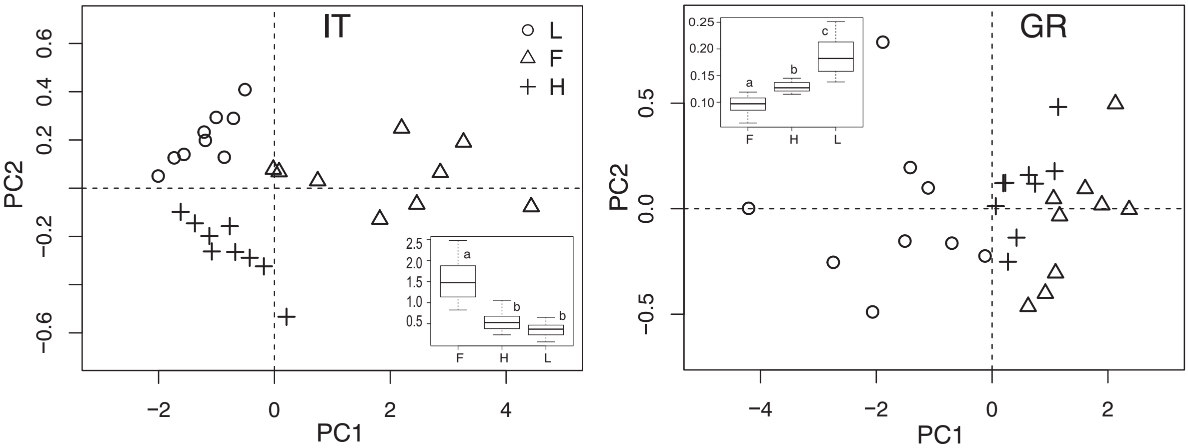

Fig. 1. Scatter plots of weighting coefficients for PC1 and PC2 of the forest floor horizons (L, F and H) calculated considering total mass, C and N stock (kg m^{-2}). Different letters in the box plots show significant differences in total mass (kg m^{-2}) among horizons, according to the pairwise comparisons between group levels with corrections for multiple testing.

At the IT site F was the mostly thick horizon, with the greatest amount of total mass, C and N stock (Table 2). The multiple comparison showed significant differences between F vs. L and F vs. H ($P < 0.001$), while no significant difference was found between L vs. H (Fig. 1). The deeper horizons F and H were significantly different only in the total mass amount ($P < 0.05$). At the GR site all the three horizons resulted significantly different (Fig. 1).

Comparing the two sites, the total mass accumulated in the forest floor as well as C and N stock (kg m^{-2}) was greater in IT than in GR, especially in the deeper horizons F and H ($P > 0.001$).

Overall, thinning produced a slightly significant increase in C and N stocks according to the site-specific forest floor patterns ($P < 0.05$), in the F and H horizons at the IT site and in the L horizon at the GR site (Table 2).

Quantifying the effect of thinning based on parameters normalized with respect to control, a common increase of C stock was found especially in the upper L (GR) and F (IT) horizons (Fig. 2). Comparing the two treatments, at IT site there was a reduction in C stock under selective thinning especially in the L horizon. On the other hand, at GR site selective thinning produced an increase in C stock especially in the F horizon. Similar trend was found for N stock, except for H horizon that was characterized by the same thinning effect at both sites (Fig. 2).

3.1.2. Mineral soil

As in the forest floor, C stock in the mineral soil layer was greater ($P < 0.001$) in IT than GR at both 0–10 (6.4 ± 1.4 vs. 3.3 ± 0.6 kg m^{-2}) and 10–30 (5.6 ± 1.4 vs. 1.8 ± 0.2 kg m^{-2}) cm depths (Table 3). N stock was significantly greater ($P < 0.001$) in the Italian site only in the deep layer, while very similar amounts were recorded at 0–10 cm depth (~0.44 kg m^{-2}).

The C stock variability from upper to deep layer showed a similar pattern in both sites, although its decrease was significant only at the GR site. On the other hand, N stock increased at the IT site and decreased at GR ($P < 0.001$).

The effect of thinning based on parameters normalized with respect to control appeared clearer under selective treatment, inducing a general increase of C and N stock (kg m^{-2}) in the deeper layer at both sites, although C stock was greater in GR and N in IT (Fig. 3). The selective thinning increased soil C and N stock also in comparison with the traditional one, although at the IT site it was not significant. On the other hand, at the GR site thinning produced a significant increase of C

Table 2
Pre- and post-thinning total mass, C and N content (mean ± standard deviation, g m^{-2}) of the forest floor at the control (C) and thinning plots (T – traditional, S – selective). Significant differences among horizons (3 replicates for each treatment) were indicated by different letters in brackets ordered in sequence for each parameter (Total mass, C stock, N stock) according to the pairwise comparisons between group levels (Tukey HSD test with Bonferroni correction for multiple comparisons, $P < 0.05$). Similarly, significant differences after thinning among treatments were indicated by different letters.

Site	FF horizons	Treat	Total mass		C stock		N stock	
			Before	After	Before	After	Before	After
Italy (IT)	L (b,b,b)	C	319 ± 200	322 ± 208	148 ± 93	135 ± 86	2.5 ± 1.8	2 ± 0.8
		T	395 ± 298	610 ± 356	179 ± 136	258 ± 150	3.2 ± 2.2	4.1 ± 0.9
		S	398 ± 147	431 ± 220	184 ± 23	175 ± 93	3.0 ± 0.5	2.7 ± 1.1
	F (a,a,a)	C	1587 ± 667	$1422^{(b)} \pm 372$	569 ± 240	$544^{(b)} \pm 169$	15.4 ± 6.5	12.0 ± 4.4
		T	1500 ± 787	$2190^{(a)} \pm 694$	543 ± 316	$862^{(a)} \pm 334$	15.4 ± 9.7	16.0 ± 7.3
		S	1606 ± 237	$1927^{(ab)} \pm 300$	641 ± 92	$729^{(ab)} \pm 92$	17.9 ± 1.6	13.3 ± 2.4
	H (b,b,b)	C	657 ± 114	$958^{(b)} \pm 375$	168 ± 17	246 ± 100	7.4 ± 0.9	10.8 ± 4.1
		T	494 ± 97	$1285^{(a)} \pm 388$	134 ± 36	338 ± 81	6.2 ± 1.5	14.0 ± 4.8
		S	519 ± 168	$1126^{(ab)} \pm 97$	156 ± 75	329 ± 58	7.3 ± 2.8	12.5 ± 0.9
Greece (GR)	L (a,a,a)	C	186 ± 25	$1.6^{(b)} \pm 0.1$	61 ± 8	$65^{(b)} \pm 11$	2.4 ± 0.4	$2.0^{(b)} \pm 0.2$
		T	212 ± 50	$2.2^{(a)} \pm 0.2$	79 ± 23	$114^{(a)} \pm 13$	2.7 ± 0.7	$2.7^{(a)} \pm 0.3$
		S	165 ± 31	$2.2^{(a)} \pm 0.3$	48 ± 11	$117^{(a)} \pm 25$	2.1 ± 0.5	$2.9^{(a)} \pm 0.4$
	F (b,b,b)	C	100 ± 18	84 ± 30	18 ± 6	25 ± 8	1.1 ± 0.4	0.8 ± 0.3
		T	97 ± 19	98 ± 24	16 ± 5	29 ± 10	0.8 ± 0.6	0.9 ± 0.2
		S	90 ± 25	99 ± 18	17 ± 9	33 ± 10	1.0 ± 0.3	1.1 ± 0.2
	H (c,b,b)	C	132 ± 15	139 ± 6	31 ± 4	33 ± 2	1.1 ± 0.4	1.3 ± 0.1
		T	125 ± 9	143 ± 5	26 ± 3	30 ± 2	1.2 ± 0.3	1.5 ± 0.1
		S	129 ± 9	133 ± 6	30 ± 3	31 ± 3	1.2 ± 0.1	1.6 ± 0.1

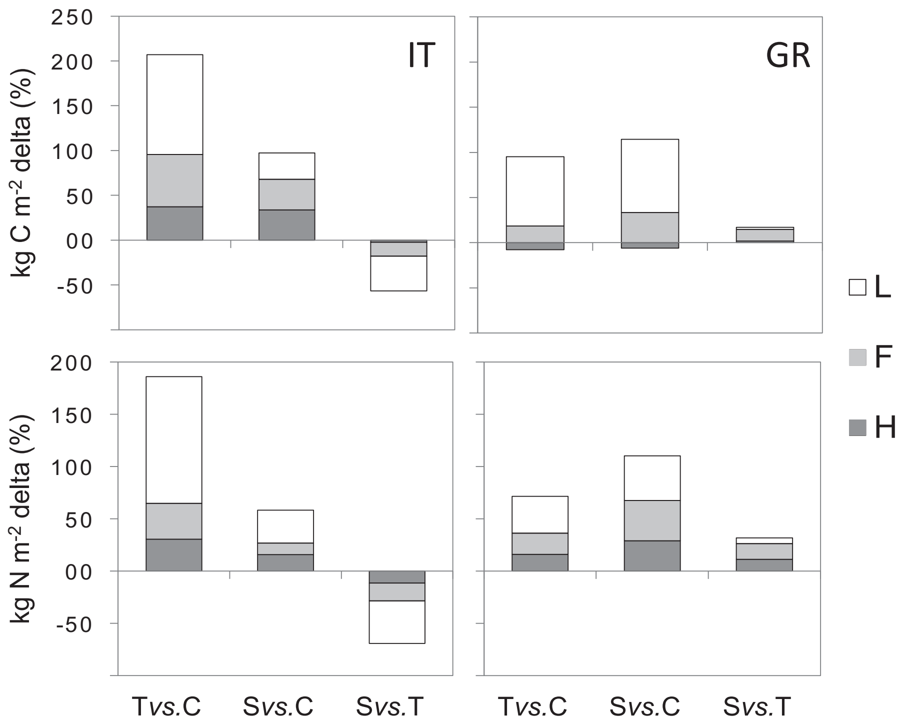

Fig. 2. Percentage of carbon and nitrogen stock (kg C and N m^{-2}) in the forest floor horizons (L, F and H) under thinning (traditional – T and selective – S) normalized with respect to control (C).

stock in the upper soil layer both under selective and traditional treatments ($P < 0.01$) and in the deeper layer under the selective one ($P < 0.05$).

3.2. Effect of thinning on soil gas fluxes

3.2.1. CO_2 effluxes

The effect of thinning in the short-term did not produced significant differences between the two sites, although the absolute values of CO_2 effluxes were higher in IT. Indeed, considering CO_2 effluxes normalized with respect to control, the different values found between the two sites were not significant both 3 and 6 months after thinning. On the other hand, comparison based on seasonal variability showed a contrasting pattern between sites with higher CO_2 effluxes at the IT site during autumn 2016 (only at control plots: 27237.3 vs. 11,968.6 g CO_2-C ha^{-1} d^{-1}, $P < 0.001$) and at the GR site during spring 2017 (26,134.6 vs. 36,588.3 g CO_2-C ha^{-1} d^{-1}, $P < 0.001$).

At the IT site, during thinning operations significant peaks ($P < 0.05$) were recorded for both traditional and selective treatments (Fig. 4). After thinning, similar CO_2 effluxes were found for all treatments especially during winter (3 months after). During spring (6 months after) an increasing trend following temperature increase appeared, with slightly higher CO_2 effluxes under both thinning treatments and control.

On the other hand, at the GR site the effect of thinning was not significant (Fig. 4) and CO_2 effluxes showed the same increasing trend related to temperature increase during spring.

3.2.2. CH_4 and N_2O fluxes

Both sites showed CH_4 uptake, highlighting a similar pattern with CH_4 fluxes ranging from −0.2 to −15 g CH_4-C ha^{-1} d^{-1} (Fig. 5). Overall, both during and after thinning no significant difference in CH_4 uptake between sites was found, although at the IT site the lowest peaks were observed. In both sites, during thinning intervention no significant difference was found among treatments, while after thinning the most significant short-term effect ($P < 0.05$) appeared in the first 3 months with increased uptake under selective treatment in both sites (1.8 and 2.4 time higher than control in IT and GR, respectively). In the second 3 months after thinning the CH_4 uptake ratio between selective and

Table 3
Pre- and post-thinning C and N content (mean ± standard deviation, ton ha^{-1}) of the mineral soil at the control (C) and thinning plots (T – traditional, S – selective).

			C stock		N stock	
Site	Soil depth (cm)	Treat	Before	After	Before	After
Italy (IT)	0–10	C	55.0 ± 4.2	81.7 ± 22.4	3.9 ± 0.4	5.6 ± 0.8
		T	71.0 ± 9.6	78.8 ± 13.4	4.6 ± 0.1	4.6 ± 0.1
		S	64.9 ± 21.9	82.1 ± 22.1	4.6 ± 1.4	4.8 ± 1.1
	10–30	C	54.5 ± 6.0	87.0 ± 19.9	5.6 ± 0.3	6.8 ± 1.4
		T	48.5 ± 4.1	72.7 ± 9.0	5.6 ± 0.6	6.6 ± 0.3
		S	64.9 ± 23.5	93.3 ± 19.3	6.2 ± 1.7	8.0 ± 1.7
Greece (GR)	0–10	C	33.0 ± 4.3	32.4 ± 2.8	4.8 ± 0.5	5.6 ± 0.8
		T	36.4 ± 8.1	47.7 ± 2.0	4.2 ± 0.7	4.9 ± 0.2
		S	29.3 ± 5.1	46.5 ± 2.8	4.1 ± 0.3	6.3 ± 0.4
	10–30	C	18.2 ± 3.2	20.4 ± 1.6	2.6 ± 0.5	3.1 ± 0.2
		T	17.6 ± 3.3	22.9 ± 2.3	2.7 ± 0.3	3.3 ± 0.3
		S	17.9 ± 2.3	28.4 ± 2.7	2.5 ± 0.5	3.3 ± 0.3

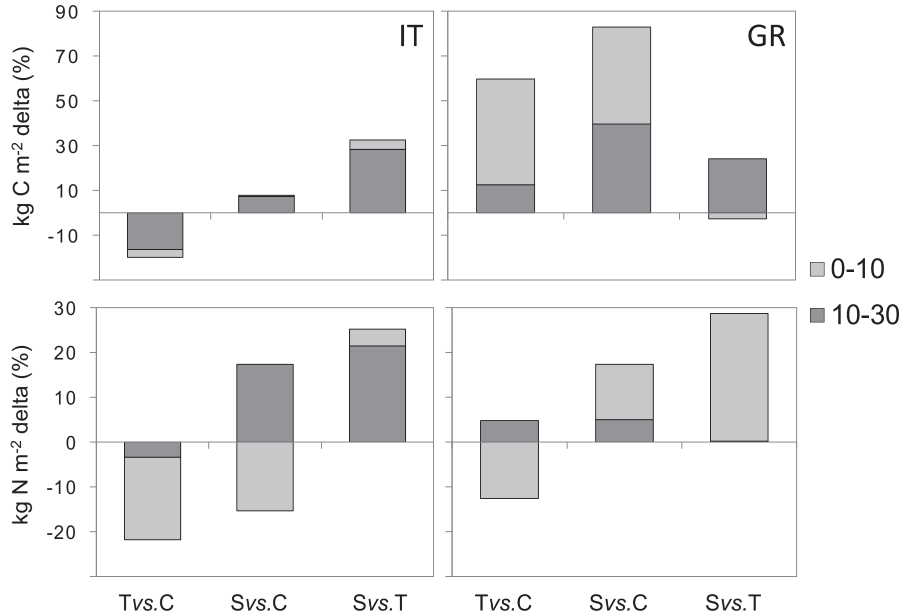

Fig. 3. Percentage of carbon and nitrogen stock (kg C and N m^{-2}) in the mineral soil layers (0–10 and 10–30 cm depths) under thinning (traditional – T and selective – S) normalized with respect to control (C).

control decreased to 1.3 and 1.6 time in IT and GR, respectively. Independently of the site effect, CH_4 uptake was significantly higher under selective treatment compared to that measured both under traditional and control, both 3 and 6 months after thinning ($P < 0.05$).

N_2O fluxes were characterized by very low emissions in both sites and under the different treatment applied (Fig. 6). Overall, in the Italian site N_2O fluxes were higher than those measured at the GR site ($P < 0.001$) independently of the treatment applied. At the GR site N_2O fluxes trend was very flat and close to zero, while at the IT site N_2O peaks were found during the period of thinning interventions, although both in thinned and control plots.

3.3. Dependence of gas fluxes on soil temperature and moisture

Comparing similar seasonal periods, no significant differences were observed on 5 cm depth soil temperature between sites (4.1 ± 1.8 and 3.7 ± 1.4 °C during winter and 11.2 ± 3.1 and 11.3 ± 3.0 °C during spring in It and GR, respectively). On the other hand soil moisture (%) showed significant differences ($P < 0.001$), with higher values at the IT site especially in the winter season (43.9 ± 12.5 and 12.5 ± 7.8 during winter and 26.6 ± 7.2 and 13.5 ± 4.2 during spring in It and GR, respectively).

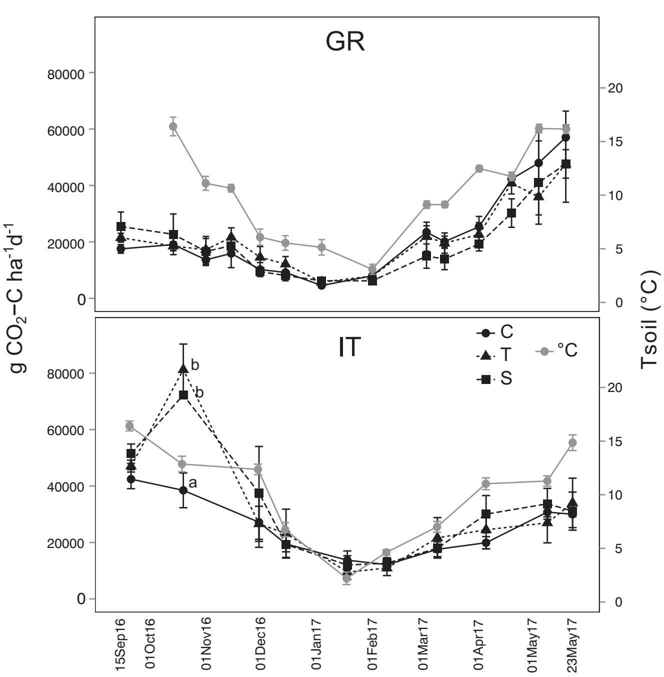

Fig. 4. Soil carbon dioxide (CO_2) efflux and soil temperature from the control (C) and thinned plots (traditional – T and selective – S). The vertical bars indicate the standard error of the mean.

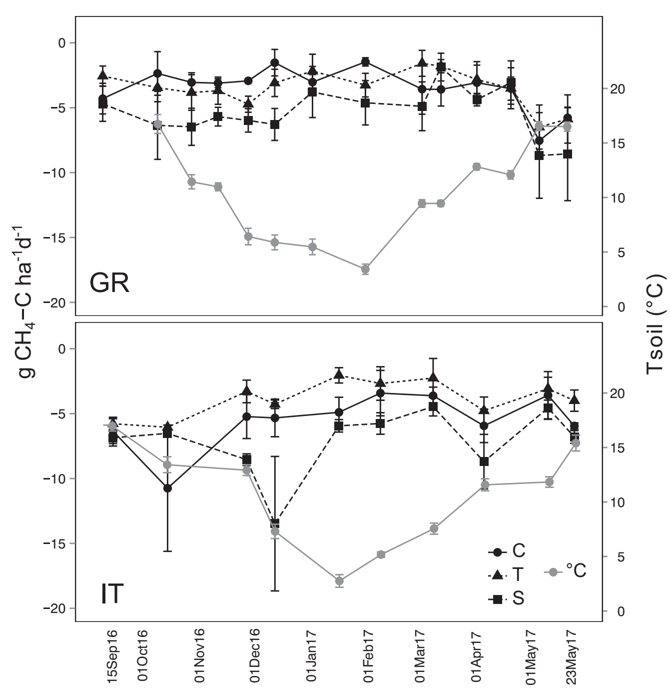

Fig. 5. Soil methane (CH_4) flux and soil temperature from the control (C) and thinned plots (traditional – T and selective – S). Symbols as in Fig. 4.

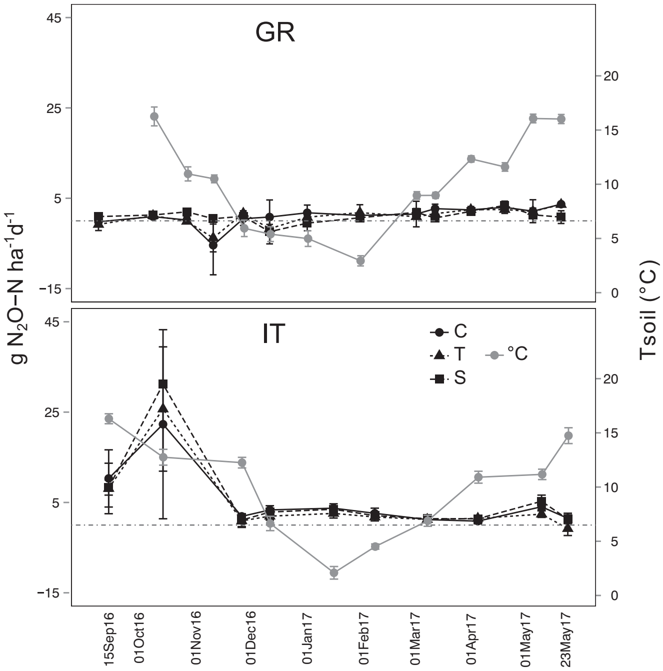

Fig. 6. Soil nitrous oxide (N_2O) flux and soil temperature from the control (C) and thinned plots (traditional – T and selective – S). Symbols as in Fig. 4.

The short-term effect of thinning was not significant neither on soil temperature nor on soil moisture, indeed the values recorded within the thinned plots were very similar in comparison with those of control plot in both sites.

Soil temperature resulted the main driving variables for CO_2 efflux, explaining 58% and 72% of the variability for GR and IT, respectively (Fig. 7). After applying the analysis of covariance (ANCOVA), the *t*-test *P*-value < 0.001 suggested that the population regression lines for sites IT and GR have unequal intercepts. So there was a significant difference between the CO_2 emissions of the two sites, after adjusting for the effect of the variable soil temperature. Moreover, the test for the interaction (Tsoil:Site) has a *P*-value = 0.0005, which indicates that different slope is reasonable. Including soil moisture with soil temperature in a multiple linear model did not improve the goodness-of-fit and nor produced a significant interaction between the two variables on CO_2 efflux.

During winter CO_2 efflux was positive correlated with soil moisture ($P < 0.001$) and the multiple linear model produced a slight but not significant interaction between the two variables on CO_2 efflux ($P = 0.06$) that it became significant in spring ($P = 0.03$).

CH_4 uptake was positively correlated with soil temperature ($P < 0.001$) and negatively with soil moisture ($P < 0.05$). On the other hand, N_2O fluxes were not significant correlated to soil temperature but showed a weak linear correlation with soil moisture ($P < 0.05$). No significant influence of the interaction between the two environmental variables on CH_4 uptake and N_2O fluxes was found.

Based on site specific effects of environmental variables on GHG fluxes, soil temperature showed positive correlations with CO_2 effluxes and CH_4 uptake at both sites, as expected. Another common pattern was the negative influence of soil moisture on CH_4 uptake ($P < 0.01$). On the other hand, N_2O fluxes under wetter climatic conditions (IT) showed a positive correlation with soil moisture ($P < 0.05$). Under drier climatic conditions (GR) soil moisture did not have any significant influence on N_2O fluxes while soil temperature produced a weak positive correlation ($P < 0.05$), that increased during spring ($P < 0.05$).

3.4. Gas fluxes and C-N pools relationships

GHG fluxes showed significant correlations with C and N stock (kg m^{-2}) of both forest floor and mineral soil (Fig. 8). Regarding the forest floor, all GHG fluxes were positively correlated with total mass, C and N stock of the deepest H horizon. The intermediate F horizon showed significant correlations between CO_2 effluxes and total mass and C stock and between N_2O fluxes and N stock (Fig. 8).

Regarding mineral soil, all GHG fluxes showed significant correlations with C stock at both soil depths (Fig. 8), while N stock was positively correlated with CO_2, CH_4 and N_2O only at the deepest soil layer (10–30 cm).

Applying a multilinear model after testing each parameter separately due to the collinearity, CO_2 effluxes were likewise related with total mass, C and N stock (~$R^2 = 0.43$, $P < 0.001$) while for mineral soil the highest relation was found with 10–30 cm depth ($R^2 = 0.42$, $P < 0.001$). CH_4 uptake showed a similar trend but with lower relationships than CO_2 (~$R^2 = 0.30$ and 0.33 with $P < 0.01$ for forest floor and mineral soil, respectively). N_2O fluxes showed the highest relationships with N content of forest floor ($R^2 = 0.50$, $P < 0.001$) and with both soil depths (~$R^2 = 0.31$, $P < 0.01$).

3.5. Contribution of trace gas emissions to global warming potential

The short-term effect of thinning based on comparing the same seasonal periods produced in winter a larger GWP at the IT than GR site (5591.8 vs. 3441.5 kg CO_2eq ha^{-1} period^{-1}; $P < 0.05$, Wilcoxon rank

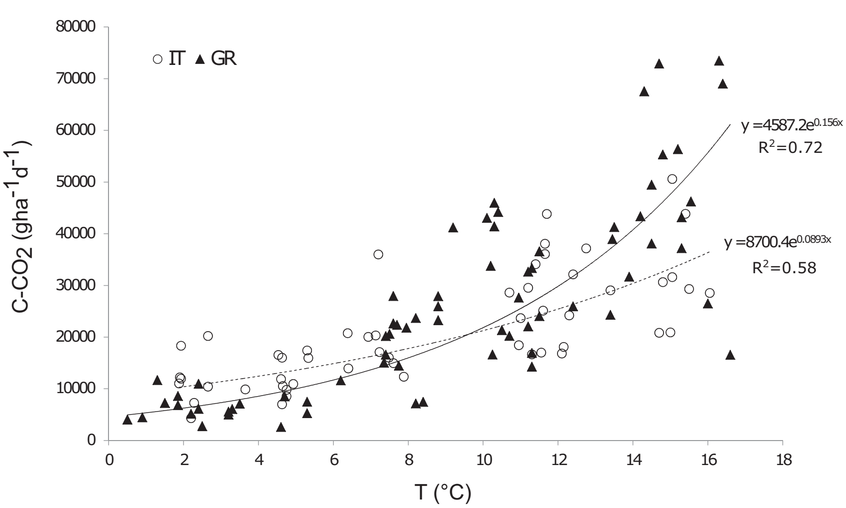

Fig. 7. The exponential relationships between soil CO_2 efflux and soil temperature at both sites.

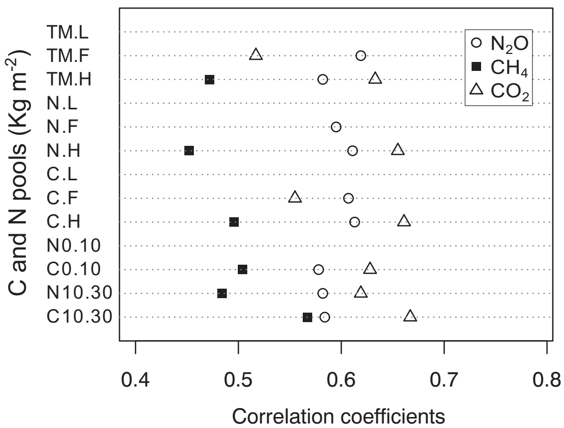

Fig. 8. Correlation coefficients between soil GHG fluxes and total mass, carbon and nitrogen pools (kg TM, C and N m^{-2}) of forest floor horizons (L, F and H) and mineral soil layers (0–10 and 10–30 cm depths). Only significant values are shown.

non-parametric test). On the contrary, in spring GR showed slightly larger GWP values that become significant considering only May (946,496.3 vs. 1,419,927.1 kg CO_2eq ha^{-1} $period^{-1}$ with $P < 0.05$ at IT and GR site, respectively).

Soil CO_2 efflux was the largest flux to the atmosphere from both sites, accounting during the first 6 months after thinning for almost all of the GWP with 14,022 and 9083 kg ha^{-1} y^{-1} at IT and GR sites, respectively (Table 4). Although its large global warming potential (298 times greater than CO_2), N_2O contribution to GWP was about 187 and 74 kg ha^{-1} y^{-1} at IT and GR sites, respectively.

Both sites resulted as CH_4 sink, despite that the reduction of GWP by CH_4 uptake was just about 42 and 29 kg ha^{-1} y^{-1} at IT and GR sites, respectively.

In the short-term GWP was larger under traditional treatment in both sites, although no significant differences were found. For the first 3 months after thinning, there was an increase of about 7% and 15% compared to the control at IT and GR sites, respectively. After 6 months, this increase still remained and showed an opposite trend (12% and 4% at IT and GR sites, respectively).

Overall, in the short-term thinning produced a weak effect on total GWP with a slightly increase under traditional treatment (8%) and a negligible reduction (4%) under selective one.

4. Discussion

4.1. GHG emissions spatial variability and main drivers

In our study, climatic conditions and organic inputs featured distinct GHG fluxes trends between sites, discriminating for geographical location, mainly driven by a higher soil moisture content and forest floor accumulation at the IT site. The amount of C and N available in an ecosystem may determine soil GHG fluxes (Janssens et al., 2001; Selmants et al., 2008). We found significant correlations between GHG fluxes and C and N stock (kg m^{-2}) of both forest floor (F and H fractions) and mineral soil. In the upper L horizon of forest floor no significant correlations were found, according to the slow decomposition rate in Mediterranean conifer ecosystems where net mineralization appeared to be confined to the lower part of the forest floor (Escudero et al., 1987).

Accumulation of litter and nutrients in Mediterranean environments could increase in moisture-limited forests due to a much slower litter decomposition rate (Kavvadias et al., 2001). Consequently, the higher forest floor accumulation at the IT site, characterized by a higher soil moisture content, was likely more related to the stands structure characteristics and the higher degree of forest degradation processes increasing the input of plant material above the soil surface, than to a limitation of decomposition rates. The higher CO_2 effluxes observed during autumn-winter period at the IT site was thus related to both larger forest floor and soil moisture, likely increasing the heterotrophic component of soil respiration (Rey et al., 2002; Sullivan et al., 2008). Differently, under drier conditions (GR), the increase of CO_2 effluxes was limited to rainy events during late-spring, which possibly stimulated decomposition, mineralization and CO_2 emission, as is common in regions with Mediterranean climate especially after summer drought periods because of lack of precipitation (Jarvis et al., 2007).

Soil moisture content is found to be positively related to soil CO_2 flux (Fang et al., 2016a, 2016b), until reaching a soil water-filled pore space (WFPS) of about 60% (Xu and Qi, 2001; Rey et al., 2002). Both sites were below this threshold, showing prevalent oxidative conditions throughout the year, making both sites net CH_4 sinks, which is usually found in Mediterranean ecosystems (Nicolini et al., 2013; Savi et al., 2016).

Differently, N_2O fluxes considerably varied with forest typology and they were characterized by a strong temporal variability and were generally directed upward (Nicolini et al., 2013). At our study sites N_2O fluxes were characterized by very low emissions independently from the treatment applied. Overall, at the IT site N_2O fluxes were higher than those found in GR, especially for the peaks measured during autumn-early winter period when soil moisture was significantly higher than at the GR site. In addition, the significantly larger N stock in forest floor and soil triggered N_2O emissions at the IT site. There is evidence that autotrophic nitrification may proceed under short-term O_2 limitation (Bollmann and Conrad, 1998; Wrage et al., 2001) supplying NO_3 for denitrification in the process of nitrifier-denitrification, which was facilitated also by a faster diffusion in the gas-filled pore spaces of the coarser soil (McNicol and Silver, 2014; Mazza et al., 2017) and the presence of sufficient organic matter (Umarov, 1990). Indeed, N_2O was highly correlated with N stock in the F horizon, which stored the largest amount. Studies on litter decomposition with litterbags (Berg and Ekbolm, 1983; McClaugherty et al., 1985) have shown that fresh litter is a net N immobilizer for 1 ± 3 years, whereas F horizon has been reported to be a N source (Federer, 1983; Boone, 1992).

4.2. Short-term effects of thinning

Previous studies on soil-derived CO_2 efflux in forest ecosystems have shown contrasting responses to thinning, reporting both increases and decreases in soil respiration (Selig and Seiler, 2004; Tang et al., 2005; Selmants et al., 2008; Sullivan et al., 2008). Logging-induced soil compaction can substantially modify the set of gases released and their rates of exchange with the atmosphere (Cambi et al., 2015). Compaction of the soil by heavy equipment decreases the soil macroporosity and causes reduction in air diffusion and water infiltration rates (Cambi et al., 2015), increasing the soil water content. Nevertheless, the compaction level can produce a highly variable response in the CO_2 effluxes as found by Hartmann et al. (2014) in a loamy soil covered by a European beech and Norway spruce forest. They found that unlike with severe compaction, moderate compaction increased CO_2 emissions, possibly because of enhanced microbial mineralization of freshly exposed organic matter with a still sufficient oxygen supply.

In our study, at the IT site CO_2 emissions peaks were observed only during thinning interventions. Felled trees were brought to the landing

Table 4
Global warming potential (mean ± standard deviation) expressed as CO_2 eq (kg ha^{-1}) for the first 6 months after thinning.

Site	Treatment	CO_2	N_2O-CO_2 eq	CH_4-CO_2 eq	Total
IT	Control	13,267 ± 1175	208.8 ± 11.7	−38.6 ± 9.5	13,437 ± 1165
	Traditional	14,910 ± 4172	140.7 ± 20.0	−27.3 ± 4.5	15,023 ± 4184
	Selective	13,890 ± 2439	212.5 ± 29.7	−59.3 ± 8.3	14,043 ± 2414
GR	Control	9343 ± 1381	58.7 ± 71.4	−20.6 ± 2.3	9381 ± 1450
	Traditional	9693 ± 1868	107.8 ± 24.1	−24.6 ± 4.2	9776 ± 1844
	Selective	8214 ± 1022	55.7 ± 10.4	−41.1 ± 9.0	8229 ± 1019

site by skidding without using machineries within measurements plots due to the steep slopes. This operation most likely caused a light soil compaction and an increasing mineralization rate of freshly exposed organic matter and detrital material as litter and dying tree roots, which are easily decomposed. On the other hand, at the GR site thinning intervention lasted about only 1 month, so its short-term effect on soil-derived CO_2 efflux was not able to induce significant emission peaks as at the IT site. The lack of changes of CH_4 and N_2O during thinning suggested that critical levels of soil permeability were never reached, even at IT site where logging operations lasted longer than at GR site. Indeed, soil temperature and moisture were not significantly affected, confirming the light impact of treatments applied.

After thinning operations, no significant changes on soil CO_2 effluxes in the short term were observed at both sites. However, an increase of forest floor mass and C inputs was found between thinned and unthinned plots, indicating larger organic C substrates available for heterotrophic respiration. At the same time, removing some individual trees, thinning reduced the photosynthetic leaf area and living roots (Sampson et al., 2007), thus reducing the autotrophic contribution to soil respiration. These changes, together with the lack of significant effects of thinning on total CO_2 efflux, suggested a new balance between heterotrophic and autotrophic respiration. We hypothesized that a reduction in autotrophic respiration balanced out an increase of heterotrophic component in the short term, as also observed by Sullivan et al. (2008) and Olajuyigbe et al. (2012).

Therefore, we hypothesized an altered balance between autotrophic and heterotrophic respiration components (Olajuyigbe et al., 2012; Wilkinson et al., 2016), influencing GHG exchange with the atmosphere (Castro et al., 2000; Zerva and Mencuccini, 2005; Sullivan et al., 2008; Mojeremane et al., 2012; Sundqvist et al., 2014).

Moreover, even slightly soil temperature increases affected CH_4 uptake after selective thinning. In fact, this treatment removed about 40.4% of basal area (10.2% more than the traditional one) and opened small gaps (of about 100 m^2 around the selected trees, while under traditional thinning they were negligible) that increased soil C and N stock, as also found in other studies (Muscolo et al., 2007; Settineri et al., 2018). The coupled effect of enhancing microbial mineralization of freshly exposed organic matter and gap size effect on microclimate and soil properties might have increased the CH_4 sink.

The effects of forest management practices on soil N_2O fluxes are still less known and few studies on this topic are available. Increased nitrification after clear-felling was found in a Sitka spruce forest of Northern England (Dutch and Ineson, 1990) and in a Norway spruce stand of Finland (Smolander et al., 1998). In another Sitka spruce plantation on peaty gley soil, in N.E. England the short-term effect of clearfelling on soil N_2O was not so clear since the two stands prior to clearfelling exhibited different patterns in soil N_2O fluxes (Zerva and Mencuccini, 2005). An increase of N_2O fluxes was found after understory removal in forest plantations in China (Li et al., 2010). In a 100-year-old Norway spruce stand in Germany clear-cut and selective cutting were compared. After reaching the highest N_2O emission in the second year after clear-cutting, N_2O emissions gradually declined to pre-harvest levels in the sixth year (Gundersen et al., 2012). In two Atlantic temperate forest sites located in Nova Scotia, Canada, clear-cutting was inconsistent for N_2O during the sampling period (Lavoie et al., 2013). Soil compaction for forest harvesting caused a considerable increase of N_2O emissions up to 40 times the uncompacted ones (Teepe et al., 2004).

Even if N_2O fluxes magnitude is usually low in unfertilized forest ecosystems, there are several factors that warrant investigation of changes: i) the almost 300-times higher GWP of N_2O than CO_2, ii) an increase of N deposition across Europe (Galloway et al., 2008; De Vries et al., 2011), iii) the lack of data and the highly uncertain national estimates (Butterbach-Bahl et al., 2013), iv) the absence of clear and effective mitigation strategies, v) the vicinity of urban areas, including airport, in our specific case.

Our study showed no significant short-term effect of thinning on N_2O fluxes, which were close to 0 and seemed affected more on seasonal changes related to N uptake. The light impact of interventions was therefore lower a threshold able to change soil physical characteristics, which may have a stronger impact on nitrification/denitrification processes.

4.3. Global warming potential

The largest contribution to GWP was ascribable to CO_2 emissions at both sites as also found in other studies (Zerva and Mencuccini, 2005). Significant changes in soil-derived GWP related to CH_4 and N_2O fluxes contribution were found only after site preparation practices as N fertilization and drainage (Mojeremane et al., 2012).

The short-term impact of thinning on GWP was negligible both 3 and 6 months after, so measurements over longer time periods are required to have more consistent results on this trend, taking into account also the annual seasonal variability. On the other hand, the contribution of CH_4-CO_2 equivalent to total GWP showed a clear and significant CH_4 sink behaviour under selective treatment, with a CH_4 uptake higher of about 43 and 35% after 3 and 6 months, respectively, compared to no thinning.

5. Conclusions and perspectives

Soil moisture content and organic matter inputs led to specific site-level patterns in GHG fluxes. The significant correlations found between GHG fluxes and C and N pools in both forest floor and mineral soil confirm a primary role of physical and chemical processes occurring at the soil surface level. This suggests that C and N pools should be considered, together with environmental variables as soil temperature and moisture, for accounting GHG fluxes in degraded forests.

Overall, the short-term increase of CO_2 emissions was related to the timing of logging operations and an increase of C and N inputs. After thinning, the effect on CO_2 disappeared shortly, whereas the CH_4 sink behaviour of both sites was increased, especially under selective treatment.

Analysing the short-term effects of thinning allows to investigate how rapid changes in site conditions can influence GHG fluxes, but measurements over several years are required to take into account the annual seasonal variability and the long-term responses of gas fluxes to thinning-induced variability in environmental variables and C and N pools. Evaluating the coupled effect of short and long-term impact of management practices on GHG balance may be useful to calibrate specific and appropriate management options of forest ecosystem growing in climate sensitive regions and under degradation processes. Moreover, the GHG balance of thinning interventions can be completed after accounting the increase in net primary productivity (NPP) due to the photosynthetic uptake by trees typically stimulated by thinning practices, especially the selective treatment.

Acknowledgements

This research was funded by LIFE project FoResMit “Recovery of degraded coniferous Forests for environmental sustainability Restoration and climate change Mitigation” (LIFE14 CCM/IT/000905).

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