Tree cover influences the soil C balance in Mediterranean cork oak-based silvopastoral systems

Original
Tree cover influences the soil C balance in Mediterranean cork oak-based silvopastoral systems / Pulina, A.; Campus, S.; Cappai, C.; Roggero, P. P.; Salis, L.; Seddaiu, G.. - In: SOIL & TILLAGE RESEARCH. - ISSN 0167-1987. - 215:(2022), p. 105234. [10.1016/j.still.2021.105234]

Tree cover influences the soil C balance in Mediterranean cork oak-based silvopastoral systems / Pulina, A.; Campus, S.; Cappai, C.; Roggero, P. P.; Salis, L.; Seddaiu, G.. - In: SOIL & TILLAGE RESEARCH. - ISSN 0167-1987. - 215:(2022), p. 105234. [10.1016/j.still.2021.105234]

Tree cover influences the soil C balance in Mediterranean cork oak-based silvopastoral systems

Antonio Pulina[a*], Sergio Campus[a], Chiara Cappai[a], Pier Paolo Roggero[a], Lorenzo Salis[b], Giovanna Seddaiu[a]

[a] Department of Agricultural Sciences and Desertification Research Centre (NRD), University of Sassari, Sassari, Italy

[b] AGRIS Sardegna - Servizio Ricerca per la Zootecnia, Sassari, Italy

*Corresponding Author:
Antonio Pulina
Viale Italia 39A 07100 Sassari (IT)
Email: anpulina@uniss.it; ORCID 0000-0002-2220-0922

Abstract

Mediterranean silvopastoral systems provide a wide range of ecosystem services, among which soil C sequestration is one of the most relevant. Although the role of trees in enhancing soil C content of silvopastoral systems is widely recognized, the impacts of the tree cover on the overall soil C balance and hence the soil C sequestration rates are not clear enough to identify feasible options and effective management pathways. This study aimed to assess the effect of trees on soil C balance in a Mediterranean cork-oak silvopastoral system under contrasting cover classes. A three-year study was conducted in a Long-Term Observatory in Sardinia (Italy). A hierarchical design was used to test the tree effect (below and outside the tree crown projection) and the tree cover (high tree cover – from 39% to 42% – and regular tree distribution; low tree cover – from 15% to 36% – and irregular tree distribution) on the soil C balance. The C balance was calculated as the difference between the C input from the herbaceous and tree components of the ecosystem and the C output from soil heterotrophic respiration. The position with respect to the tree stem influenced the C input, the C output, and the overall C balance. The tree class influenced only the C output and C balance. The annual C balance was higher (more gains) in low-covered areas below the tree crown projection (8.30±0.72 Mg ha^{-1} of C) than in open pasture and high-covered areas. At the field level, the overall weighted C balance was significantly influenced by tree cover; according to this last evidence, the maximum C balance was reached at about 24% tree cover. The results highlight the importance of management practices oriented to increase herbage production, tree cover, and regulating tree distribution, which in turn can provide ecosystem services such as SOC sequestration.

Keywords

Agroforestry systems; C sequestration; Soil respiration; Grassland productivity; Litterfall

1. Introduction

Mediterranean silvopastoral activities on wooded grasslands play a crucial role in ecosystem services provision, such as forage provisioning and C cycling regulation (Moreno et al., 2018) and hence climate change mitigation potential (Lal, 2020). Mediterranean *Quercus*-based silvopastoral systems are recognized as a priority by the “Habitats” Directive 92/43/EEC (type 6310 ‘‘Dehesas with evergreen Quercus spp”). These agroforestry systems have been shaped by centuries of traditional agro-silvopastoral activities (Eichhorn et al., 2006), as the typical scattered-trees vegetation structure is generated by the combination of extensive livestock farming and forestry activities (Bagella et al., 2013).

The role of trees in enhancing the soil C storage is recognized under a wide range of environmental conditions (Haile et al., 2010; Takimoto et al., 2009), which also includes Mediterranean silvopastoral systems (Cappai et al., 2017; Howlett et al., 2011; Seddaiu et al., 2018). Trees positively influence the soil organic carbon (SOC) content under their crown projection due to continuous provisioning to the soil of leaves, inflorescences, infructescences (e.g. acorns), and woody material such as branches, usually referred to as litterfall, which contributes to the formation of organic matter with different levels of quality and persistence (Cappai et al., 2017). The tree root systems influence the SOC underneath the tree crown because of the fine roots rhizodeposition and turnover (Casals et al., 2010; Moreno et al., 2005), which also influence soil fertility and the cycling of N and P (Nair et al., 2019). The trees also contribute to establishing a microclimate under the crown projection in terms of radiation, water balance, and temperature, resulting in differences in terms of plant communities and biomass production (López-Sánchez et al., 2016; Seddaiu et al., 2018). Furthermore, the interaction between biotic and abiotic factors results in differences in the microbiological activities and hence in SOC losses as soil respiration (Lai et al., 2014; Uribe et al., 2015). In fact, it is widely recognized that in these agroecosystems the main drivers of soil respiration are the soil water content and temperature (Rey et al., 2002; Tang and Baldocchi, 2005). The soil water content constrains the microbiological activity, and hence the soil respiration, under summer drought periods, thus resulting the most important factor affecting SOC variations under Mediterranean climate (Oyonarte et al., 2012; Pulina et al., 2018a). Also, the open areas within wooded grasslands significantly contribute to the overall SOC content (Lal, 2004; Seddaiu et al., 2013). In these areas, the C input to soil is largely affected by grassland above- and belowground primary productivity (Pausch and Kuzyakov, 2018). Soil management can also be an important factor affecting the SOC content, e.g. through grazing management, tillage (Uribe et al., 2015), fertilization (Fornara et al., 2016), oversowing with legume species (Hernández-Esteban et al., 2018), and land use (Seddaiu et al., 2013).

The Mediterranean silvopastoral systems are threatened because of both trends of intensification, e.g. causing a lack of tree regeneration (Rossetti and Bagella, 2014), and abandonment, e.g. leading to shrubs encroachment, and increasing firewood risk (Moreno et al., 2018). The provision of silvopastoral-associated ecosystem services, as SOC sequestration (Seddaiu et al., 2018), is guaranteed over time by the anthropic presence and influence (Bugalho et al., 2011), without which such habitats would tend to their natural potential vegetation (Bagella and Caria, 2011).

A lack of knowledge emerges from the literature on the effect of the tree cover in influencing SOC sequestration and hence the climate change mitigation potential of these silvopastoral systems. As these agroecosystems result from the close interaction between anthropic and environmental factors, it becomes basic to understand the impact of management practices through the assessment of the role of tree cover, as a proxy of the intensity of human activities, especially in terms of SOC balance.

This study aimed to assess the impacts of the position with respect to the tree crown projection within different classes of tree cover on (i) seasonal dynamics of soil CO_2 emissions and their relationships between environmental drivers, (ii) pasture and tree biomass production and contribution to soil C input, and (iii) SOC balance, during a three-year experiment in a Mediterranean silvopastoral system. The hypothesis was that in a Mediterranean silvopastoral agroforestry system the drivers regulating the SOC balance are influenced not only by the position with respect to the tree, but also by the area covered by trees within homogeneous farming units.

2. Materials and Methods

2.1. Study site

The study site was located in a subset of private farms inside the Long-Term Observatory of Berchidda-Monti (NE Sardinia, Italy) (40° 49' N / 9° 17' - 9° 19' E; 287-325 m a.s.l.). The Observatory is representative of the Sardinian wooded grassland-based semi-extensive livestock systems and, overall, of the silvopastoral farming systems within the Mediterranean basin (Caballero et al., 2009). Human activities have shaped the landscape for centuries as a mosaic of land uses such as cork oak forests, grasslands, vineyards, and cork oak wooded grassland, which represents the largest land use covering about the 30% of the Observatory (Seddaiu et al., 2018). The mean annual rainfall in the Observatory is 632 mm, of which about 70% falls between October and May. The mean annual temperature is 14.2 °C, and the aridity index (mean annual precipitation divided by mean annual reference evapotranspiration) is 0.53. The dominant soil type is a Typic Dystroxerept (USDA, 2010), derived from a granitic substratum (Carmignani et al., 2012) and characterized by a sandy loam texture in the upper soil horizon. The SOC content at 0-40 cm depth ranges from 13.3 g kg^{-1} in areas outside the tree crown projection to 31.8 g kg^{-1} in areas below the tree crown. The tree role in SOC variation was already reported and discussed by Seddaiu et al. (2018). In this study, the SOC, the total N content, and the C/N ratio measured in May 2014 are reported in Table 1 according to the study layout below described. The potential vegetation is referable as *Violo dehnhardtii-Quercetum suberis* association, mainly represented by *Quercus suber* L. forests (Bagella and Caria, 2011). During the study period (2012-2015), the average temperature was 15.8 °C, the average maximum air temperature occurred in August (32.9 °C), while the average minimum temperature occurred in February (2.7 °C). The average yearly rainfall from 2012 to 2015 was 558 mm, and on average the

70% of total rainfall occurred from October to March. The average ET_0 (Allen et al., 1998) was 1022 mm, resulting in an average aridity index of 0.56.

2.2. Study layout

The study was carried out from autumn 2012 to spring 2015 at eight fields within silvopastoral farms. Each field was a homogeneous and fenced wooded grassland. The fields during the observational period had almost similar grazing management in terms of breed (dairy sheep, occasionally beef cattle) and stocking rates, both according to farmer’s management choices (Seddaiu et al., 2018).

The study layout was a two-factor hierarchical design, which was set as follows:

- Tree cover as the main factor, which was distinguished in two levels: i) high-covered fields (High), with tree cover ranging from 42% to 39% of the field area, and ii) low-covered fields (Low), with a tree cover ranging from 36% to 15%. The tree-covered area within each field was measured by analyzing the available online orthophotos of the Sardinia Region (http://webgis.regione.sardegna.it/geoserver/ows?). The distinction between High and Low was carried out also taking into account the spatial distribution of trees. The spatial distribution allowed to differentiate between them fields having similar cover but different pattern of tree aggregation, i.e. identifying or not identifying large open grazing areas. The spatial distribution pattern was defined according to the Uniform Angle Index (W) (Zhao et al., 2014), which allows identifying regular, random, and cluster patterns of tree spatial distribution. Each level was replicated in four different fields.
- Position with respect to the tree crown projection, which was distinguished in two levels: i) areas below the crown projection of randomly selected trees (BT), and ii) areas outside the tree crown projection (OT), far enough to trees to avoid any influence e.g. shading effects.

Field cover, W index values and patterns, and related classification are in detail reported in Figure 1.

All the variables were measured, unless otherwise specified as reported below in the text, in six sampling points per field: three points BT, and three points OT. A list of the acronyms here and below reported is provided in the Appendix (Table A.1).

2.3. *Measurements*

2.3.1. *Soil Respiration, Water Content, and Temperature*

The measurements of soil CO_2 fluxes were carried out from 1st August 2013 to 5th August 2014. The total Soil Respiration (SR) and its heterotrophic fraction (Rh) were measured *in situ* at BT and OT positions, using a portable, closed chamber (Smith et al., 2010), soil respiration system (EGM-4 with SRC-1, PP-Systems, Hitchin, UK, http://ppsystems.com). Measurements of SR and Rh were carried out between 8:30 and 13:00 (solar time) to collect data representative of daily means (Almagro et al., 2009 and citations therein). Measurements of Rh were made in trenched plots (Hanson et al., 2000) adjacent to SR sampling points, in which soil was isolated with a PVC cylinder (0.40 m diameter and 0.25 m height) open at both ends (Lai et al., 2012). The cylinder was inserted into the soil profile till 0.20 m depth, assuming this depth as sufficient to exclude the large part of active tree root biomass, according to Unger et al. (2009, and citations therein). The soil within the cylinder was weeded using 10 mL L^{-1} of glyphosate (ROUNDUP® POWER 2.0). The measurements started about six months after trenching to avoid any influence of weeding on respiration (Lane et al., 2012) and disturbance due to decomposition of decaying roots (Unger et al., 2009). The new seedlings emerging within the cylinders over the observational period were immediately and carefully cut to ensure the absence of any root activity.

At the same time and adjacent at the sampling point, soil temperature (SoilT, °C) (by using a thermometer HD2101.2, Delta Ohm, Padua, Italy) and water content (SWC, %) (using a FieldScout TDR 300 Soil Moisture Meter, Spectrum Technologies, Inc., Bridgend, UK) were measured at -10 cm and -7 cm depth, respectively.

Daily SWC and SoilT from February 2014 to May 2015 were measured at both BT and OT both at one sampling point per plot using a WatchDog 1000 Series Micro Station (Spectrum Technologies, Inc., IL, USA, http://www.specmeters.com), equipped with a WaterScout SM 100 Soil Moisture Sensor and Soil Temperature Sensor for SoilT and SWC, respectively. The SoilT and SWC sensors were placed in the soil at 10 cm and 7 cm depth, respectively. To calculate the soil C balance along the whole period, from September 2012 to January 2014 daily SWC and SoilT at OT plots were estimated using the PaSim - Pasture Simulation Model (Riedo et al., 1998), properly calibrated, and validated for the study area (Pulina et al., 2018a). The monthly Rh at BT was estimated from monthly OT/BT ratios observed from February 2014 to May 2015.

2.3.2. Pasture production

Pasture dry matter (DM) production and residues were measured at about monthly sampling intervals between October 2012 to May 2015 using 1m x 1 m movable grazing exclusion cages (Frame, 1981). Cages were placed onto the ground in the sampling areas. Under BT, cages were placed fully below the crown projection about 1.5 m away to the tree trunk in the North-Est orientation. The North-East orientation (mainly unshaded) was taken because the dominant wind in the study area is from the South-West direction and the trees have typically flag-shape crowns. The pasture DM production between two sampling dates (from day1 to day2) was calculated making the difference between the DM biomass measured inside the cage at day2 and the DM biomass outside the cage at day1. The intake was calculated as the production from day1 to day2 plus the difference of DM outside the cage between day 1 and 2. Methodology, sampling intervals, and animal intake computation are in detail described by Seddaiu et al. (2018).

2.3.3. Stoking rates and animal faeces

The stocking rates (LSU ha^{-1}) at each field were monitored over the whole period by weekly recording the stocking size, the animal typology and condition, and the actual grazing hours. Monthly average values of stoking rates were hence used to calculate the faeces amount produced per stock unit (see below).

2.3.4. Tree standing biomass

The diameters of *Q. suber* L. trees were measured within fields in sixteen circular sampling plots – three of which around the BT sampling plots above described –with variable size (radius ranging from 9.8 m to 29.0 m) according to the local stand density and neighborhood competition. Each plot was centered at one of the scattered cork oak trees inside the plot. If trees were close to each other, the plot center was established to be coincident with their barycentre. The above- and belowground stands biomass was determined using the allometric equations developed by the National Institute of Agricultural Research and Technology and Food of Spain (Montero et al., 2005), whose parameters are reported in the Appendix (Table A.2). The field-scale standing biomass was estimated through linear regression models between the within-plot cover and both above- and belowground biomass.

2.3.5. Tree litterfall

Tree litterfall was sampled from September 2014 to September 2015 in three areas per field identified by trees. Four traps (0.7 m x 0.7 m) were placed in each area 1.0 m upper the soil level. The trap was a plastic net with meshes of 2 mm x 2 mm wide, aiming to guarantee water drainage without make the litterfall pass. Three of the traps were placed starting from the trunk following the cardinal NE direction according to spatial criteria: the first trap was placed within the space from the trunk to half of the crown’s radius length; the second trap was placed within the space of the second half of crown’s radius; the third one was placed in the orthogonal direction with respect to the first three ones, in the homologous position of the first trap The average amount of litterfall collected within these first three traps represented the amount of litterfall BT per sampling date. The fourth trap was placed outside the crown in the same direction of the first and second trap, and keeping almost the same distances between the first and the second trap. The distance from the edge of the crown of this last trap allowed to avoid any influence of adjacent trees, especially in high-covered fields. . The litterfall collected in the fourth trap represented the amount of litterfall OT per sampling date. The collected samples were dried in a ventilated oven at 65 °C until achieving constant weight, after their separation in wood, leaves, flowers, and acorns components.

2.4. Soil C balance

The soil C balance was calculated in a three-year timeframe (September 2012 to May 2015) as the differences between the soil C losses as cumulated Rh in the considered period, and the soil C input in the same timeframe, by following the framework proposed by Lai et al. (2018) and Pulina et al. (2018b).

2.4.1. Soil C output

At each plot, soil C output was calculated as the result of the cumulative Rh (Mg ha^{-1} of C-CO_2), obtained as the sum of the average cumulative daily values of Rh (kg ha^{-1} of C-CO_2). To estimate daily cumulative Rh values within plots, a multiple regression model (Tang and Baldocchi, 2005) was fitted as a function of both SWC and SoilT, as follows:

$$F = \beta_0 e^{\beta_1 T} e^{\beta_2 \theta + \beta_3 \theta^2} \qquad (1)$$

where F (μmol m^{-2} s^{-1} of CO_2) was the measured Rh, T (°C) and θ (m^3 m^{-3}) were the contemporary measured SoilT and SWC, respectively, and β_0, β_1, β_2, and β_3 are the model parameters. The equation was log-transformed to conduct the linear regression to estimate the parameters (Table 2), which were hence used to estimate daily cumulative Rh by fitting the model with the daily average values of measured SoilT and SWC.

2.4.2. Soil C input

The soil C input at the end of the yearly observation period (September 30th) was represented by the C input from the herbaceous component of the pasture (Cpas), the C input from the grazing animal faeces (Cfae), and the C input from trees (Ctre).

The Cpas was calculated as follows:

$$Cpas = Cres + Croo + Crhi$$

where Cres was the C input from pasture aboveground residues, Croo was the C input from the herbaceous decomposed root, and Crhi was the C input from herbaceous species rhizodeposition. The Cres was yearly estimated from pasture DM residues. The OT Croo was calculated starting from the Root biomass (Rb), which was estimated by multiplying the yearly aboveground DM biomass production with a root:shoot ratio of 2.43, resulting from the average value between those proposed by Mokany et al. (2006) for temperate grassland and savanna ecosystems. The Rb was then multiplied by a fine roots turnover rate of 0.80 (Casals et al., 2010). The BT Croo was calculated by multiplying the OT value by 1.45, resulting in the average ratio between BT and OT root:shoot derived from what was observed by Nair et al. (2019). The yearly Crhi was estimated by multiplying the yearly Rb by 0.65 (Lai et al., 2017).

The Cfae was calculated starting from the amount of estimated monthly faeces from grazing animals. The monthly amount of faeces was estimated according to Van Soest (1994) by following the equation

$$F_a = I - (D * I)$$

where Fa was daily faeces produced per animal, I was daily DM intake per animal, and D the DM digestibility rate estimated at 0.7 (Van Soest, 1994).

The Ctre was calculated as a sum between the C input from the tree root system (Ctrs), and the C input from tree litterfall (Clit).

The Ctrs was calculated starting from the estimated tree belowground biomass as follows:

$$Ctrs = Cfin + Crhf$$

where Cfin was the yearly C input from tree fine roots turnover and Crhf was the yearly C input from tree fine roots rhizodeposition. The belowground biomass C content was calculated by multiplying by 0.50 (Merino et al., 2003; Montero et al., 2005) the estimated tree belowground biomass at field scale.

The Cfin was estimated multiplying by 0.20 (Millikin and Bledsoe, 1999) the belowground stored C, which was calculated dividing that estimated according to the allometric equations by 0.80, as only the coarse roots (80% of total root biomass) were taken into account by the model, and then dividing the obtained value by a fine roots turnover coefficient of 0.3 yr^{-1} (López et al., 2003). The Crhf was hence estimated by multiplying the fine roots C by 0.50 (Pausch and Kuzyakov, 2018).

The Clit was calculated by first multiplying each measured litterfall component by their C concentration, and then multiplying the litter C content by a yearly litterfall decomposition rate of 58% (Arosa et al., 2017). The C content in litterfall was measured twice (at autumn-winter and summer-spring period, respectively) by an elemental analyzer (CHN628, LECO Corporation, MI, USA).

2.5. Data Analysis

The effect of study factors on the relationships between both SoilT-SWC and CO_2 fluxes was assessed. The significance of the linearized Flux = a $e^{b\ SoilT}$ (Rey et al., 2002) was assessed to test the effect of SoilT on both SR and Rh. The model was assessed above the SWC threshold of 15.9%, corresponding to a matric potential of -100 kPa (Saxton and Rawls, 2006), below which the limiting water conditions overlap the temperature effects, according to Lado-Monserrat et al. (2014). Below the same threshold, the effect of SoilT was tested through a simple linear model (lm). The effect of SWC was tested by a second-degree polynomial model after SWC normalization with respect to a fixed value of soil temperature (25 °C), according to Tang and Baldocchi (2005). The fitted models were compared considering both SoilT and SWC as covariates. Contrasts between the levels of the factors were performed testing the fitted ANCOVA models by multiple comparisons under general linear hypothesis (glht).

The effect of the interaction between tree class and position and monitoring dates on SR, Rh, SWC, SoilT, and tree litterfall was tested by generalized least squares (gls) models, through which models with different correlation structures were tested (Onofri et al., 2016). The best model was identified making an “*a posteriori”* selection based on the Akaike Information Criterion (AIC). The effect of factors on pasture production and residues and the C balance items, as well as the field-scale C balance, was tested by fitting a linear mixed model (lme) considering the random effect of the year where necessary. Only the effect of the tree class was tested for Cfae, Cfin, Crhf, and Ctrs. If the analysis of variance (anova) of the fitted model was significant, the least-square means (eemeans) were computed to compare means between levels. The parameters of the polynomial regression in (Eq. 1) were estimated fitting a linearized model at each field for both BT and OT areas. The ANOVA was computed on fitted models to assess their significance.

The C balance and their components (Mg ha^{-1} of C) were also field-scaled by making the sum of the weighted C values at both BT and OT according to the tree-covered (Tcov) and non-covered area of each field, as follows:

$$Weighted\ C = C_{BT}\ Tcov + \ C_{OT}\ (1 - Tcov)$$

Linear and polynomial regressions were performed to assess the effects of the tree cover (%) on weighted Cpas, Ctre, Cinput, Coutput and Cbalance; the best models between linear and polynomial are reported in the results. The maximum value of the second-order regression between Cbalance and cover was calculated through the first derivative of the regression function. The significance of the Pearson’s correlation coefficient (cor.test) was tested to assess the relationship between weighted SOC and C balance items. The significance of statistics was evaluated at $P<0.05$ unless otherwise stated. The lm, anova and cor.test, glht (Hothorn et al., 2008), gls and lme (Pinheiro et al., 2018), emmeans (Lenth, 2018) computations were performed using the RStudio application of R software (version 3.5.1) (R Core Team, 2021).

3. Results

3.1. Soil Respiration, Water Content, and Temperature

The position significantly influenced the relationships between both SR and Rh and SoilT above the SWC threshold of 15.9%, while no significant effects of SoilT on both SR and Rh were observed above the same SWC critical value. Overall, under High-BT both SR and Rh showed a higher sensitivity to SoilT increase than the other levels of position, as reported in Figure 2. The position influenced the relationships between both normalized (with respect to a constant SoilT of 25 °C) SR and Rh and SWC. Under High-BT the response of both SR and Rh to SWC variation was different than OT positions, while differences between High-BT and High-OT were observed only for Rh, as reported in Figure 3.

The SR and Rh were both significantly influenced by the interaction between the monitoring dates and the position ($P<0.0001$). Overall, both SR and Rh dynamics showed two peaks (autumn and spring) profiles. Maximum fluxes of soil CO_2 were observed on May 20th in High-BT, when SR and Rh were 12.03±2.02 $\mu mol\ m^{-2}\ s^{-1}$ and 8.06±0.58 $\mu mol\ m^{-2}\ s^{-1}$, respectively. Minimum soil CO_2 emissions occurred on September 27th in High-OT areas, where SR and Rh were 0.73±0.12 $\mu mol\ m^{-2}\ s^{-1}$ and 0.55±0.15 $\mu mol\ m^{-2}\ s^{-1}$, respectively (Figure 4a,b). The SR was higher in High-BT than OT in late spring and autumn (Aug 27th, Sep 11th, Oct 22nd, and Nov 8th), and at the beginning of summer (June 24th), while no differences between treatments within dates were observed in winter and early spring 2014 (Figure 4a). The Rh was higher in High-OT than the other treatments at the beginning of autumn 2013 and from May 20th, 2014 to June 24th, 2014 (Figure 4b). No differences between areas in both SR and Rh were observed in winter.

The SWC was significantly influenced by the interaction between the monitoring dates and the position ($P <0.0001$). Overall, the SWC showed an increasing dynamic from summer to spring, to then decrease until the end of the experiment (Figure 4c). The maximum SWC was observed on March 27th under Low-BT (0.32±0.02 $m^3\ m^{-3}$), which was higher than High-OT (0.27±0.02 $m^3\ m^{-3}$), while no differences were observed with the other treatments (0.29±0.01 $m^3\ m^{-3}$ on average in High-BT and Low-OT, respectively). Minimum SWC was observed in High-OT at the beginning of the monitoring period (0.02±0.00 m^3 m^{-3}), which was lower than Low-BT and Low-OT (0.05±0.0 and 0.06±0.01 m^3 m^{-3}).

The SoilT dynamics were significantly influenced by the interaction between the date and the position ($P<0.0001$). The SoilT overall showed an opposite dynamic to SWC (Figure 4d). The maximum values were observed on August 1st, 2013 in OT areas (31.5±1.2 °C and 31.2±0.6 °C in High-OT and Low-OT, respectively), as well as the minimum values, observed at November 29th, 2013 (5.0±0.7 °C and 6.7±0.3 °C in High-OT and Low-OT, respectively). Overall, the OT SoilT was higher and lower than the other treatments first during summer and spring and then in late autumn and winter, respectively, as reported in Figure 4d.

3.2. Pasture production and residues

The pasture biomass production (Mg ha^{-1} yr^{-1} of DM) averaged for the three-year experiment was significantly influenced ($P<0.0001$) by both tree class and position. The biomass production on average was higher under Low-OT (3.8±0.3 Mg ha^{-1} of DM) than Low-BT and High-OT (2.6±0.3 Mg ha^{-1} and 2.5±0.3 Mg ha^{-1} of DM, respectively), in which the biomass production was in turn higher than High-BT (1.4±0.3 Mg ha^{-1}). The average biomass residues at the end of the year (Mg ha^{-1} yr^{-1} of DM) were significantly influenced ($P<0.01$) by the tree class, while no significant effects of the position were observed. The residues were on average higher under Low(1.4±0.2 Mg ha^{-1}) than High (0.9±0.2 Mg ha^{-1}). The average pasture biomass production and residues along the experiment are reported in Figure 5.

3.3. Tree and Litterfall Biomass

The tree above- and belowground biomass was higher ($P<0.05$) in High (on average 53.7±1.5 Mg ha^{-1} and 11.0±0.3 Mg ha^{-1} above and below, respectively) than the Low class (28.6±8.2 Mg ha^{-1} and 6.0±1.6 Mg ha^{-1} above and below, respectively).

The tree litterfall biomass (Mg ha^{-1} of DM) was influenced by the interaction between the sampling date (month) and the position (P<0.0001). The tree litterfall dynamic had a bimodal profile, whose two maximum values were observed in May and in October, respectively. In May 2014 the litterfall was higher in High-BT (2.7±0.4 Mg ha^{-1}) than Low-OT (0.8±0.4 Mg ha^{-1}), while in October 2013 the litterfall was higher in High-BT (0.8±0.1 Mg ha^{-1}) than the other levels. The litterfall dynamics and the differences between position/cover within the month are reported in Figure 6.

3.4. Soil C Balance

The position significantly influenced the Clit, the Ctre, the Cinput, the Coutput (P<0.0001), and the Cbalance (P<0.001), while the Cres (P<0.01), Croo, Crhi, Cpas (P<0.0001) were significantly influenced by the tree class. The Cpas and all the components determining it (Cres, Croo, Crhi) were always higher in Low than High. The Clit, the Ctree, and the Coutput were higher in High-BT than Low-BT, in which were higher than High-OT and Low-OT. The Cinput was higher in High-BT and High-OT than Low-OT, in which in turn the Cinput was higher than High-OT. The overall Cbalance was higher in Low-BT than High-BT, in which was in turn higher than High-OT and Low-OT. No significant effects of factors were observed for Cfin, Crhf, Ctrs, and Cfae. Results are in detail reported in Table 3.Table 3.

Considering the C balance items at field scale weighted according to the tree cover, the the tree cover significantly influenced Cpas, Ctre, Coutput, and Cbalance at field scale, while a slight significant effect on Cinput (P=0.023) was observed (Figure 7). The overall C balance at field scale was higher under Low (4.36±0.34 Mg ha^{-1} of C) than High (2.43±0.34 Mg ha^{-1} of C) fields, as reported in Figure 8. A significant Person's correlation coefficient (0.41, P<0.05) between SOC and Cpas was observed, while no significant correlations were observed between SOC and Ctre, Cinput, Coutput, and Cbalance.

4. Discussion

4.1. The effect of trees on SoilT, SWC, and CO_2 emissions

The SoilT and SWC showed inverse dynamics over the monitoring period, as widely reported for Mediterranean areas (e.g. González-Ubierna and Lai, 2019; Pulina et al., 2018a). Consequently, the highest temperatures are associated with the driest periods, during which water scarcity strongly affects the SOC cycle and hence SR and Rh (Almagro et al., 2009). The observed relationship between SWC and normalized CO_2 fluxes confirmed this as the response of both SR and Rh to SWC variation was directly proportional for SWC values below the critical threshold (15.9%). Above the threshold, when water availability was not limiting for microbial and roots respiration, both SR and Rh were mostly driven by SoilT variation. At the same time, under constant SoilT, the response of CO_2 fluxes to SWC above the critical threshold tends to be inversely proportional, probably due to a reduction of oxygen availability for respiration processes (Lado-Monserrat et al., 2014; Tang and Baldocchi, 2005).

The higher sensitivity of both SR and Rh to SoilT and SWC variation observed in High-BT than OT areas was associated with the higher accumulation of labile SOC and nutrients beneath trees than outside the tree canopy projection (Cappai et al., 2017; García-Angulo et al., 2020; Seddaiu et al., 2018; Uribe et al., 2015) which, in turns, was mainly related to trees litterfall (Cappai et al., 2017; Rossetti et al., 2015b). Furthermore, the higher response of SR-Rh to increasing SoilT and SWC at High-BT positions could be also led to an overall improvement of soil fertility and nutrients under the tree canopy (Moreno and Obrador, 2007; Nair et al., 2019), that promotes physiological activities of heterotrophic communities (Lai et al., 2014; Rossetti et al., 2015a) and that it is particularly marked in high tree-covered areas (Bagella et al., 2020). Nevertheless, the absence of differences between the responses of SR under High-BT and Low-BT to SWC variations could be explained as a result of increased autotrophic activity due to the observed higher pasture biomass production beneath the tree canopy under low-covered areas, which is below discussed.

4.2. The effect of trees on biomass production of the herbaceous layer and litterfall

On average, the three-year DM production of the herbaceous layer irrespectively of the cover class was about two-fold higher outside the tree crown projection than beneath the tree. The detrimental role of the trees on the biomass production of grasslands was already reported for Mediterranean sites by Seddaiu et al. (2018) in the same study site and by other authors in the Spanish dehesa (López-Carrasco et al., 2015), in the Portuguese montado (Serrano et al., 2018), and silvopastoral systems of Northern Greece (Klossas et al., 2012). However, other authors highlighted a negligible impact of the tree on pasture production (e.g. Carranca et al., 2015) or even a beneficial effect (e.g. Moreno, 2008) depending mainly on soil water conditions that, in turn, were related to the annual rainfall amount. In fact, under wetter conditions a much greater herbage yield of the pasture component underneath the tree canopy was reported, while in drier ones differences were smoother or null or even the opposite (Moreno, 2008; Seddaiu et al., 2018). In the Spanish dehesas, López-Carrasco et al. (2015) observed a lower yield of the herbaceous layer under the tree than in open areas when annual rainfall was lower than 600 mm. This water limiting effect was also reported by Lozano-Parra et al. (2018), who in Spanish dehesas observed a more intense reduction of pasture growth below the tree canopies than the open grassland during the driest years, as a result of more intense water deficit beneath the trees. Other factors that may have influenced biomass production under the tree are the light and nutrients competition between tree and herbaceous components (e.g. Orefice et al., 2019; Serrano et al., 2018) and the different grazing behavior of the livestock. Grazing animals exploit less efficiently the available pasture biomass beneath the tree than outside the tree canopy (Seddaiu et al., 2018) and this can constrain pasture growth that can be promoted by grazing pressure if not too intense. Under Mediterranean climate in sub-clover and ryegrass-based grasslands in Australia, Witschi and Michalk (1979) observed a detrimental effect of both treading and high grazing pressure on late winter production and pasture recovery. Areas beneath the tree canopies are submitted to intense treading since animals rest below the tree crowns after grazing to restore during the hottest hours in spring and summer (e.g. Bojkovski et al., 2014), and this could contribute to the lower forage production underneath the trees (Seddaiu et al., 2018). Furthermore, the lower pasture growth below the tree canopies could also be associated to the reduced PAR penetration to the ground (Marañón et al., 2009) for the high amount of litterfall, up to nearly 90% higher than in the open areas, which can affect seed germination and seedling establishment, especially of the more light demanding plant species (Rossetti et al., 2015).

The strong influence of season in affecting the pattern of cork oak litterfall observed in this study agrees with what was widely described under Mediterranean conditions (e.g. Andivia et al., 2018; Andivia et al., 2015; Caritat et al., 2006; Zribi et al., 2015). The higher litterfall amount under High-BT observed during almost the whole observation period was mainly interpreted as the result of higher tree cover and higher tree vicinity. Under tropical forests with different tree species, Uriarte et al. (2015) associated the observed litterfall increase to the effect of trees neighborhood, and in the meanwhile of tree cover, in establishing favorable edaphic conditions in terms of nutrient (N, P, and C:N) cycles. Therefore, the tree distribution patterns could play a positive role in nutrient cycling and then can explain the increasing tree productivity and litterfall beneath trees in high tree-covered areas.

4.3. The effect of trees on the annual soil C balance

The higher Cpas under Low fields reflects what was observed and discussed above about pasture productivity and residues, which led also to a higher contribution of root systems to the C input. The contribution of trees to the C input was higher under High-BT due to a higher litterfall amount beneath the trees, though the cover class did not influence the Ctrs, as a result of even-aged tree stands within the study area (Rossetti and Bagella, 2014). The contribution of trees to the total Cinput decreased from about 81% in High-BT to 64% in Low-BT, while at the same conditions the pasture contribution increased from 18% to 35%. Overall, the contrasting effect of cover classes on Cpas and Ctre resulted in a slight significance of the effect of tree cover on field-weighted C input.

As above discussed, the higher Coutput observed in High-BT could be attributable to both different SOC contents and responses to environmental drivers of Rh, particularly SWC, as already observed under Mediterranean conditions. This evidence confirms that drought periods are the most important factor regulating SOC fluxes under semi-arid conditions (e.g. Reichstein et al., 2002), as highlighted in these experimental conditions by the lack of significance of the effect of increasing SoilT above the critical SWC threshold (about 16%). Water availability plays a crucial role in affecting the C balance items and hence SOC stocks in the soil. In fact, despite faster soil drying in open areas under increasing temperatures, resulting, as observed in 2014, in lower SWC in OT areas in spring, under OT conditions the pasture vegetation could be advantaged from the absence of competition with trees for water resources (Moreno et al., 2007). Moreover, since Ctre under BT conditions is mostly represented by quickly mineralizable organic matter (Cappai et al., 2017), the higher water availability for microbial activities may have promoted a stronger response to both SoilT and SWC variation and then higher C-CO_2 losses in these conditions. However, it is necessary to point out the uncertainty associated to the methodology of Rh measurement under BT. In fact, tranching may not have completely excluded the deeper tree active roots, thus leading to an overestimation of the C balance in that microhabitat.

In this study, areas beneath trees under low cover seemed the most favorable conditions to stock soil C in wooded grasslands on a yearly timeslot. These findings highlight that trees are not the most important factor which allows considering Mediterranean wooded grasslands as “winning” agroecosystems in terms of C sequestration capacity as suggested by other authors (Howlett et al., 2011; Lagomarsino et al., 2011; Mosquera-Losada et al., 2011; Seddaiu et al., 2018; Seddaiu et al., 2013) but that a relevant role is played by pasture primary productivity. Despite a higher average SOC in woodland than wooded grassland topsoils, in a study conducted in the same area, Bagella et al. (2020) reported no significant differences between soil C within the same position with respect to the trees along a land-use gradient from open grassland to woodland and no differences between SOC under open areas in woodland and open grassland. Furthermore, Bagella et al. (2020) reported also no differences in SOC between wooded grasslands and “patchy” scenarios combining woodland, wooded and open grassland with the same overall tree cover. These findings support our results since the relationship between tree cover and C balance identified a maximum value of C balance corresponding to about 24% tree cover when tree distribution pattern become to be identified as cluster. Beyond the tree cover value of 24%, the C balance showed a decreasing pattern until 42% of tree cover, beyond which the balance was considered in a state of equilibrium. The results of this study should however be interpreted with caution, as this evidence emerged from an apparent C balance, whose absolute values can be misleading. These results and the lack of significant correlation between the SOC content and the C balance suggested that the C balance did not take into account the plausible levels of SOC saturation under these specific conditions, and the considered C balance items do not explain the different recalcitrance levels of fractions composing the organic matter input to soil (Cappai et al., 2017), as well as the rates of litterfall incorporation (e.g. Ma et al., 2014) and rhizodeposition (e.g. Rees et al., 2005). As reported by Cappai et al. (2017), beneath the trees SOC is near to saturation or oversaturated, and this could bias the correlation between the field-scaled C balance and the SOC content. Therefore, the Cinput and the resulting Cbalance over years should not be interpreted as an absolute rate of SOC sequestration or storage (Chenu et al., 2019), for whose estimation a temporal upscaling is needed in analyzing both SOC contents and Cbalance dynamics. Finally, the resulting C balance may have been affected by the propagation of the errors from every single items, due to adopted “budget-like” approach, which could limit the interpretation of the observed differences. Nevertheless, our results put on evidence the high potential SOC sequestration ability of the studied agro-silvopastoral systems. The socio-economic contexts that cause the abandonment and intensification trends threatening Mediterranean wooded grassland (Camilli et al., 2017; Moreno et al., 2018), generate the need for policies supporting the maintenance of such systems (Mosquera-Losada et al., 2018) also considering their high potential in climate change mitigation.

5. Conclusion

The results emerging from this study highlight that in Mediterranean cork oak-based agro-silvopastoral systems the pasture productivity plays a strong role in determining the C balance and thus promoting SOC sequestration. Although an important role of trees in affecting C inputs and outputs was confirmed, a good pasture production both outside and beneath the tree canopy was reached under lower tree cover and/or with a tree distribution from random to tending to cluster. Moreover, the environmental role of water deficit seemed to play a crucial role in regulating all the complex interactions between factors influencing the soil C balance in Mediterranean agro-silvopastoral systems.

The results highlight the decisive role of agronomic management in insuring SOC sequestration in wooded grassland under Mediterranean conditions, as both grassland and livestock management good practices could benefit pasture production and quality. The maintenance of such agroforestry systems could be considered as a win-win strategy for enhancing ecosystem services provision, but also to maintain the habitat *sensu* 92/43/EEC Directive, and their conservation is strongly dependent on the human intervention on trees and management of open areas.

Acknowledgments

The study was carried out within the PASCUUM and GASPAM projects (L.R. 7/8/07 n. 7, Regione Autonoma della Sardegna). The authors are grateful to Mr. Agostino Piredda and Mr. Tore Pala from the University of Sassari, and Mr. Gianfranco Becugna and Mr. Andrea Pintore from AGRIS for their field technical support. The authors are also very grateful to farmers that hosted field experiments within their farms: Dr. Sara Casu, Mr. Giovanni Casu, Mr. Sebastiano Mu, and Mr. Domenico Sanna.

Appendix

Table A.1 List of main abbreviations, details on reported data, and related references

Abbreviation	Meaning	Details	References
High	High covered areas		
Low	Low covered areas		
BT	Below the Tree crown projection		
OT	Outside the Tree crown projection		
SR	Total Soil Respiration	Measured from August 2014 to August 2015	
Rh	Heterotrophic soil respiration	Estimated from September 2012 to July 2014. Measured from August 2014 to August 2015	Tang and Baldocchi, 2005
SoilT	Soil Temperature at -10 cm depth	Measured from August 2014 to August 2015. Estimated through PaSim model from September 2012 to July 2014.	
SWC	Soil Water Content at -7 cm depth	Measured from August 2014 to August 2015. Estimated through PaSim model from September 2012 to July 2014.	
Cres	C input from pasture residues	Calculated from the DM residues yearly measured at the end of grazing seasons from 2013 to 2015 and multiplied by 0.45 (C content in DM herbaceous biomass)	Seddaiu et al., 2018
Rb	Root biomass of pasture	Estimated by multiplying the yearly aboveground DM biomass production with a root:shoot ratio of 2.4, and by 0.45 to estimate the C content	Mokany et al., 2006
Croo	C input from pasture roots decomposition	Under OT: estimated by multiplying Rb * 0.80 Under BT: Rb * 0.80 * 1.45	Casals et al., 2010; Nair et al., 2019
Crhi	C input from pasture rhizodeposition	Estimated multiplying Rb by 0.65	Lai et al., 2017
Cpas	C input from pasture	Cres + Croo + Crhi	
Clit	C input from tree litterfall	Measured from September 2014 to September 2015 and then multiplied by a decomposition rate of 58%	Arosa et al., 2017
Cfin	C input from tree fine roots decomposition	Estimated multiplying by 0.20 the belowground stored C, and then by 1/0.3 turnover coefficient	Millikin and Bledsoe, 1999; López et al., 2003
Crhf	C input from tree rhizodeposition	Estimated by multiplying the C in fine roots C by 0.50	Pausch and Kuzyakov, 2018
Ctrs	C input from tree root system	Cfin + Crhf	
Ctre	C input from trees	Ctrs + Clit	
Cfae	C input from animal faeces	Estimated from the average daily DM intake per animal	Van Soest, 1994
Cinput	Total C input	Cpas + Ctre +Cfae	
Coutput	Total C output	Cumulative Rh	
Cbalance	C balance	Cinput – Coutput	

Table A.2 Values of the parameters a and b for the function $Y = e^{SEE^2/2} \cdot e^{a} \cdot d^{b}$, the adjusted coefficient of determination (R^2_{adj}), and the standard error of the estimation (SEE) for *Quercus suber* L. and each fraction of the biomass.

Y	**a**	**b**	**R^2_{adj}**	**SEE**
BT	-3.36627	2.60685	0.914	0.416653
BF	-3.01942	2.25213	0.893	0.405357
BR7	-6.43076	3.21136	0.87	0.574419
BR2-7	-3.3924	1.99526	0.796	0.525534
BR2	-5.33638	2.10315	0.676	0.754127
BH	-6.05826	2.14483	0.648	0.818511
Br	-2.81593	2.07774	0.924	0.356549

d: diameter at breast height under cork (cm); BT: total tree aboveground biomass; BF: biomass of the trunk; BR7: biomass of branches with a diameter greater than 7 cm; BR2–7: biomass of the branches with diameter between 2 and 7 cm; BR2: biomass of the branches of diameter less than 2 cm; BH: leaf biomass and Br: root biomass (source: Montero et al., 2005).

References

Allen, R., Pereira, L., Raes, D., Smith, M., 1998. Crop evapotranspiration – Guidelines for computing crop water requirements – FAO Irrigation and Drainage Paper 56. FAO.

Almagro, M., López, J., Querejeta, J., Martínez-Mena, M., 2009. Temperature dependence of soil CO_2 efflux is strongly modulated by seasonal patterns of moisture availability in a Mediterranean ecosystem. Soil Biol. Biochem. 41, 594-605.

Andivia, E., Bou, J., Fernandez, M., Caritat, A., Alejano, R., Vilar, L., Vazquez-Pique, J., 2018. Assessing the relative role of climate on litterfall in Mediterranean cork oak forests. iForest 11, 786-793.

Andivia, E., Fernández, M., Alejano, R., Vázquez-Piqué, J., 2015. Tree patch distribution drives spatial heterogeneity of soil traits in cork oak woodlands. Ann. For. Sci. 72, 549-559.

Arosa, M.L., Costa, S.R., Freitas, H., 2017. Leaf decomposition of cork oak under three different land uses within a montado of southern Portugal. Soil Research 55.

Bagella, S., Caria, M.C., 2011. Vegetation series: a tool for the assessment of grassland ecosystem services in Mediterranean large-scale grazing systems. Fitosociologia 48, 47-54.

Bagella, S., Caria, M.C., Seddaiu, G., Leites, L., Roggero, P.P., 2020. Patchy landscapes support more plant diversity and ecosystem services than wood grasslands in Mediterranean silvopastoral agroforestry systems. Agr. Syst. 185, 102945.

Bagella, S., Salis, L., Marrosu, G.M., Rossetti, I., Fanni, S., Caria, M.C., Roggero, P.P., 2013. Effects of long-term management practices on grassland plant assemblages in Mediterranean cork oak silvo-pastoral systems. Plant Ecol. 214, 621-631.

Bojkovski, D., Štuhec, I., Kompan, D., Zupan, M., 2014. The behavior of sheep and goats co-grazing on pasture with different types of vegetation in the karst region1. J. Anim. Sci. 92, 2752-2758.

Bugalho, M.N., Caldeira, M.C., Pereira, J.S., Aronson, J., Pausas, J.G., 2011. Mediterranean cork oak savannas require human use to sustain biodiversity and ecosystem services. Front. Ecol. Environ. 9, 278-286.

Camilli, F., Pisanelli, A., Seddaiu, G., Franca, A., Bondesan, V., Rosati, A., Moreno, G.M., Pantera, A., Hermansen, J.E., Burgess, P.J., 2017. How local stakeholders perceive agroforestry systems: an Italian perspective. Agroforest. Syst.

Cappai, C., Kemanian, A.R., Lagomarsino, A., Roggero, P.P., Lai, R., Agnelli, A.E., Seddaiu, G., 2017. Small-scale spatial variation of soil organic matter pools generated by cork oak trees in Mediterranean agro-silvo-pastoral systems. Geoderma 304, 59-67.

Caritat, A., García-Berthou, E., Lapeña, R., Vilar, L., 2006. Litter production in a Quercus suberforest of Montseny (NE Spain) and its relationship to meteorological conditions. Ann. For. Sci. 63, 791-800.

Carmignani, L., Oggiano, G., Funedda, A., Conti, P., Pasci, S., Barca, S., 2012. Carta Geologica della Sardegna. Scala 1:250.000. LAC, Firenze, IT.

Carranca, C., Castro, I.V., Figueiredo, N., Redondo, R., Rodrigues, A.R., Saraiva, I., Maricato, R., Madeira, M.A., 2015. Influence of tree canopy on N_2 fixation by pasture legumes and soil rhizobial abundance in Mediterranean oak woodlands. Sci. Total Environ. 506-507, 86-94.

Casals, P., Garcia-Pausas, J., Montané, F., Romanyà, J., Rovira, P., 2010. Root decomposition in grazed and abandoned dry Mediterranean dehesa and mesic mountain grasslands estimated by standard labelled roots. Agric. Ecosyst. Environ. 139, 759-765.

Chenu, C., Angers, D.A., Barré, P., Derrien, D., Arrouays, D., Balesdent, J., 2019. Increasing organic stocks in agricultural soils: Knowledge gaps and potential innovations. Soil Till. Res. 188, 41-52.

Eichhorn, M.P., Paris, P., Herzog, F., Incoll, L.D., Liagre, F., Mantzanas, K., Mayus, M., Moreno, G., Papanastasis, V.P., Pilbeam, D.J., Pisanelli, A., Dupraz, C., 2006. Silvoarable Systems in Europe – Past, Present and Future Prospects. Agroforest. Syst. 67, 29-50.

Fornara, D.A., Wasson, E.A., Christie, P., Watson, C.J., 2016. Long-term nutrient fertilization and the carbon balance of permanent grassland: any evidence for sustainable intensification? Biogeosciences 13, 4975-4984.

Frame, J., 1981. Herbage mass, In: Hodgson, J., Baker, R., Davies, A., Laidlaw, A. (Eds.), Sward measurements handbook. The British Grassland Society, Maidenhead, UK, pp. 39-67.

García-Angulo, D., Hereş, A.M., Fernández-López, M., Flores, O., Sanz, M.J., Rey, A., Valladares, F., Curiel Yuste, J., 2020. Holm oak decline and mortality exacerbates drought effects on soil biogeochemical cycling and soil microbial communities across a climatic gradient. Soil Biol. Biochem. 149, 107921.

González-Ubierna, S., Lai, R., 2019. Modelling the effects of climate factors on soil respiration across Mediterranean ecosystems. J. Arid Environ. 165, 46-54.

Haile, S.G., Nair, V.D., Nair, P.K.R., 2010. Contribution of trees to carbon storage in soils of silvopastoral systems in Florida, USA. Global Change Biol. 16, 427-438.

Hanson, P., Edwards, N., Garten, C., Andrews, J., 2000. Separating root and soil microbial contributions to soil respiration: a review of methods and observations. Biogeochemistry 48, 115-146.

Hernández-Esteban, A., López-Díaz, M.L., Cáceres, Y., Moreno, G., 2018. Are sown legume-rich pastures effective allies for the profitability and sustainability of Mediterranean dehesas? Agroforest. Syst.

Hothorn, T., Bretz, F., Westfall, P., 2008. Simultaneous Inference in General Parametric Models. Biometrical J. 50, 346-363.

Howlett, D.S., Moreno, G., Mosquera-Losada, M.R., Nair, P.K.R., Nair, V.D., 2011. Soil carbon storage as influenced by tree cover in the Dehesa cork oak silvopasture of central-western Spain. J. Environ. Monit. 13, 1897-1904.

Hu, Y., Hui, G., Wang, H., Li, Y., Zhao, Z., Liu, W., 2014. Uniform angle index (W) confidence interval of the random distribution and its application. For. Res. 27, 302-308.

Klossas, G., Kyriazopoulos, A.P., Koukoura, Z., 2012. Post-planting treatments and shading effects in a Fraxinus angustifolia Vahl. silvopastoral system. Ann. For. Res. 56, 179-186.

Lado-Monserrat, L., Lull, C., Bautista, I., Lidón, A., Herrera, R., 2014. Soil moisture increment as a controlling variable of the “Birch effect”. Interactions with the pre-wetting soil moisture and litter addition. Plant Soil 379, 21-34.

Lagomarsino, A., Benedetti, A., Marinari, S., Pompili, L., Moscatelli, M.C., Roggero, P.P., Lai, R., Ledda, L., Grego, S., 2011. Soil organic C variability and microbial functions in a Mediterranean agro-forest ecosystem. Biol. Fertility Soils 47, 283-291.

Lai, L., Kumar, S., Folle, S.M., Owens, V.N., 2018. Predicting soils and environmental impacts associated with switchgrass for bioenergy production: a DAYCENT modeling approach. GCB Bioenergy 10, 287-302.

Lai, R., Arca, P., Lagomarsino, A., Cappai, C., Seddaiu, G., Demurtas, C.E., Roggero, P.P., 2017. Manure fertilization increases soil respiration and creates a negative carbon budget in a Mediterranean maize (Zea mays L.)-based cropping system. CATENA 151, 202-212.

Lai, R., Lagomarsino, A., Ledda, L., Roggero, P.P., 2014. Variation in soil C and microbial functions across tree canopy projection and open grassland microenvironments. Turk. J. Agric. For. 38, 62-69.

Lai, R., Seddaiu, G., Gennaro, L., Roggero, P.P., 2012. Effects of nitrogen fertilizer sources and temperature on soil CO_2 efflux in Italian ryegrass crop under Mediterranean conditions. Italian Journal of Agronomy 7, e27.

Lal, R., 2004. Soil carbon sequestration to mitigate climate change. Geoderma 123, 1-22.

Lal, R., 2020. Managing soils for negative feedback to climate change and positive impact on food and nutritional security. Soil Sci. Plant Nutr. 66, 1-9.

Lane, M., Lorenz, N., Saxena, J., Ramsier, C., Dick, R.P., 2012. The effect of glyphosate on soil microbial activity, microbial community structure, and soil potassium. Pedobiologia 55, 335-342.

Lenth, R., 2018. emmeans: Estimated Marginal Means, aka Least-Squares Means.

López-Carrasco, C., López-Sánchez, A., San Miguel, A., Roig, S., 2015. The effect of tree cover on the biomass and diversity of the herbaceous layer in a Mediterranean dehesa. Grass Forage Sci. 70, 639-650.

López-Sánchez, A., San Miguel, A., López-Carrasco, C., Huntsinger, L., Roig, S., 2016. The important role of scattered trees on the herbaceous diversity of a grazed Mediterranean dehesa. Acta Oecol. 76, 31-38.

López, B.C., Sabate, S., Gracia, C.A., 2003. Thinning effects on carbon allocation to fine roots in a Quercus ilex forest. Tree Physiol. 23, 1217-1224.

Lozano-Parra, J., Schnabel, S., Pulido, M., Gómez-Gutiérrez, Á., Lavado-Contador, F., 2018. Effects of soil moisture and vegetation cover on biomass growth in water-limited environments. Land Degrad. Dev. 29, 4405-4414.

Ma, Y., Filley, T.R., Szlavecz, K., McCormick, M.K., 2014. Controls on wood and leaf litter incorporation into soil fractions in forests at different successional stages. Soil Biol. Biochem. 69, 212-222.

Marañón, T., Pugnaire, F.I., Callaway, R.M., 2009. Mediterranean-climate oak savannas: the interplay between abiotic environment and species interactions. Web Ecol. 9, 30-43.

Merino, A., Rey, C., Brañas, J., Rodríguez-Soalleiro, R., 2003. Biomasa arbórea y acumulación de nutrientes en plantaciones de Pinus radiata D. Don en Galicia. Invest. Agrar.: Sist. Recur. For. 12, 85-98.

Millikin, C.S., Bledsoe, C.S., 1999. Biomass and distribution of fine and coarse roots from blue oak (Quercus douglasii) trees in the northern Sierra Nevada foothills of California. Plant Soil 214, 27-38.

Mokany, K., Raison, R.J., Prokushkin, A.S., 2006. Critical analysis of root:shoot ratios in terrestrial biomes. Global Change Biol. 12, 84-96.

Montero, G., Ruiz-Peinado, R., Munoz, M., 2005. Producción de biomasa y fijación de CO_2 por los bosques españoles. INIA-Instituto Nacional de Investigación y Tecnología Agraria y Alimentaria.

Moreno, G., 2008. Response of understorey forage to multiple tree effects in Iberian dehesas. Agric. Ecosyst. Environ. 123, 239-244.

Moreno, G., Aviron, S., Berg, S., Crous-Duran, J., Franca, A., de Jalón, S.G., Hartel, T., Mirck, J., Pantera, A., Palma, J.H.N., Paulo, J.A., Re, G.A., Sanna, F., Thenail, C., Varga, A., Viaud, V., Burgess, P.J., 2018. Agroforestry systems of high nature and cultural value in Europe: provision of commercial goods and other ecosystem services. Agroforest. Syst. 92, 877-891.

Moreno, G., Obrador, J.J., 2007. Effects of trees and understorey management on soil fertility and nutritional status of holm oaks in Spanish dehesas. Nutr. Cycl. Agroecosys. 78, 253-264.

Moreno, G., Obrador, J.J., Cubera, E., Dupraz, C., 2005. Fine Root Distribution in Dehesas of Central-Western Spain. Plant Soil 277, 153-162.

Moreno, G., Obrador, J.J., García, E., Cubera, E., Montero, M.J., Pulido, F., Dupraz, C., 2007. Driving competitive and facilitative interactions in oak dehesas through management practices. Agroforest. Syst. 70, 25-40.

Mosquera-Losada, M.R., Freese, D., Rigueiro-Rodríguez, A., 2011. Carbon sequestration in European agroforestry systems, Carbon sequestration potential of agroforestry systems. Springer, pp. 43-59.

Mosquera-Losada, M.R., Santiago-Freijanes, J.J., Pisanelli, A., Rois-Díaz, M., Smith, J., den Herder, M., Moreno, G., Ferreiro-Domínguez, N., Malignier, N., Lamersdorf, N., Balaguer, F., Pantera, A., Rigueiro-Rodríguez, A., Aldrey, J.A., González-Hernández, M.P., Fernández-Lorenzo, J.L., Romero-Franco, R., Burgess, P.J., 2018. Agroforestry in the European common agricultural policy. Agroforest. Syst. 92, 1117-1127.

Nair, R.K.F., Morris, K.A., Hertel, M., Luo, Y., Moreno, G., Reichstein, M., Schrumpf, M., Migliavacca, M., 2019. N : P stoichiometry and habitat effects on Mediterranean savanna seasonal root dynamics. Biogeosciences 16, 1883-1901.

Onofri, A., Seddaiu, G., Piepho, H.-P., 2016. Long-Term Experiments with cropping systems: Case studies on data analysis. Eur. J. Agron. 77, 223-235.

Orefice, J., Smith, R.G., Carroll, J., Asbjornsen, H., Howard, T., 2019. Forage productivity and profitability in newly-established open pasture, silvopasture, and thinned forest production systems. Agroforest. Syst. 93, 51-65.

Oyonarte, C., Rey, A., Raimundo, J., Miralles, I., Escribano, P., 2012. The use of soil respiration as an ecological indicator in arid ecosystems of the SE of Spain: spatial variability and controlling factors. Ecol. Indicators 14, 40-49.

Pausch, J., Kuzyakov, Y., 2018. Carbon input by roots into the soil: Quantification of rhizodeposition from root to ecosystem scale. Global Change Biol. 24, 1-12.

Pinheiro, J., Bates, D., DebRoy, S., Sarkar, D., RCoreTeam, 2018. nlme: Linear and Nonlinear Mixed Effects Models.

Pulina, A., Lai, R., Salis, L., Seddaiu, G., Roggero, P.P., Bellocchi, G., 2018a. Modelling pasture production and soil temperature, water and carbon fluxes in Mediterranean grassland systems with the Pasture Simulation model. Grass Forage Sci. 73, 272-283.

Pulina, A., Lai, R., Seddaiu, G., Bertora, C., Rizzu, M., Grignani, C., Roggero, P.P., 2018b. Global warming potential of a Mediterranean irrigated forage system: Implications for designing the fertilization strategy. Eur. J. Agron. 98, 25-36.

R Core Team, 2021. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, AT.

Rees, R.M., Bingham, I.J., Baddeley, J.A., Watson, C.A., 2005. The role of plants and land management in sequestering soil carbon in temperate arable and grassland ecosystems. Geoderma 128, 130-154.

Reichstein, M., Tenhunen, J., Roupsard, O., Ourcival, J.M., Rambal, S., Dore, S., Valentini, R., 2002. Ecosystem respiration in two Mediterranean evergreen Holm Oak forests: drought effects and decomposition dynamics. Funct. Ecol. 16, 27-39.

Rey, A., Pegoraro, E., Tedeschi, V., De Parri, I., Jarvis, P.G., Valentini, R., 2002. Annual variation in soil respiration and its components in a coppice oak forest in Central Italy. Global Change Biol. 8, 851-866.

Riedo, M., Grub, A., Rosset, M., Fuhrer, J., 1998. A pasture simulation model for dry matter production, and fluxes of carbon, nitrogen, water and energy. Ecol. Model. 105, 141-183.

Rossetti, I., Bagella, S., 2014. Mediterranean Quercus suber wooded grasslands risk disappearance: New evidences from Sardinia (Italy). For. Ecol. Manage. 329, 148-157.

Rossetti, I., Bagella, S., Cappai, C., Caria, M., Lai, R., Roggero, P., Martins da Silva, P., Sousa, J., Querner, P., Seddaiu, G., 2015a. Influence of isolated cork oak trees on soil features, plant and collembolan assemblages in a Mediterranean wooded grassland. Agric. Ecosyst. Environ. 202, 203-216.

Rossetti, I., Bagella, S., Cappai, C., Caria, M.C., Lai, R., Roggero, P.P., Martins da Silva, P., Sousa, J.P., Querner, P., Seddaiu, G., 2015b. Isolated cork oak trees affect soil properties and biodiversity in a Mediterranean wooded grassland. Agric. Ecosyst. Environ. 202, 203-216.

Saxton, K., Rawls, W., 2006. Soil water characteristic estimates by texture and organic matter for hydrologic solutions. Soil Sci. Soc. Am. J. 70, 1569-1578.

Seddaiu, G., Bagella, S., Pulina, A., Cappai, C., Salis, L., Rossetti, I., Lai, R., Roggero, P.P., 2018. Mediterranean cork oak wooded grasslands: synergies and trade-offs between plant diversity, pasture production and soil carbon. Agroforest. Syst. 92, 893-908.

Seddaiu, G., Porcu, G., Ledda, L., Roggero, P.P., Agnelli, A., Corti, G., 2013. Soil organic matter content and composition as influenced by soil management in a semi-arid Mediterranean agro-silvo-pastoral system. Agric. Ecosyst. Environ. 167, 1-11.

Serrano, J., Shahidian, S., Marques Da Silva, J., Sales-Baptista, E., Ferraz De Oliveira, I., Lopes De Castro, J., Pereira, A., Cancela De Abreu, M., Machado, E., Carvalho, M.d., 2018. Tree influence on soil and pasture: contribution of proximal sensing to pasture productivity and quality estimation in montado ecosystems. Int. J. Remote Sens. 39, 4801-4829.

Smith, P., Lanigan, G., Kutsch, W.L., Buchmann, N., Eugster, W., Aubinet, M., Ceschia, E., Béziat, P., Yeluripati, J.B., Osborne, B., Moors, E.J., Brut, A., Wattenbach, M., Saunders, M., Jones, M., 2010. Measurements necessary for assessing the net ecosystem carbon budget of croplands. Agric. Ecosyst. Environ. 139, 302-315.

Takimoto, A., Nair, V.D., Nair, P.K.R., 2009. Contribution of trees to soil carbon sequestration under agroforestry systems in the West African Sahel. Agroforest. Syst. 76, 11-25.

Tang, J., Baldocchi, D.D., 2005. Spatial–temporal variation in soil respiration in an oak–grass savanna ecosystem in California and its partitioning into autotrophic and heterotrophic components. Biogeochemistry 73, 183-207.

Unger, S., Máguas, C., Pereira, J.S., Aires, L.M., David, T.S., Werner, C., 2009. Partitioning carbon fluxes in a Mediterranean oak forest to disentangle changes in ecosystem sink strength during drought. Agr. Forest Meteorol. 149, 949-961.

Uriarte, M., Turner, B.L., Thompson, J., Zimmerman, J.K., 2015. Linking spatial patterns of leaf litterfall and soil nutrients in a tropical forest: a neighborhood approach. Ecol. Appl. 25, 2022-2034.

Uribe, C., Inclán, R., Hernando, L., Román, M., Clavero, M.A., Roig, S., Van Miegroet, H., 2015. Grazing, tilling and canopy effects on carbon dioxide fluxes in a Spanish dehesa. Agroforest. Syst. 89, 305-318.

USDA, 2010. Keys to Soil Taxonomy. 11th Edition. Soil Survey Staff, United States Department of Agriculture, Natural Resources Conservation Service, Washington, DC, USA.

Van Soest, P.J., 1994. Nutritional ecology of the ruminant. Cornell University Press.

Witschi, P.A., Michalk, D.L., 1979. The effects of sheep treading and grazing on pasture and soil characteristics of irrigated annual pastures. Aust. J. Agric. Res. 30, 741-750.

Zhao, Z., Hui, G., Hu, Y., Wang, H., Zhang, G., von Gadow, K., 2014. Testing the significance of different tree spatial distribution patterns based on the uniform angle index. Can. J. For. Res. 44, 1419-1425.

Zribi, L., Mouillot, F., Gharbi, F., Ourcival, J.-M., Hanchi, B., 2015. Warm and fertile sub-humid conditions enhance litterfall to sustain high soil respiration fluxes in a Mediterranean Cork Oak Forest. Forests 6, 2918-2940.

Tables

Table 1. Soil Organic Carbon (SOC, Mg ha^{-1}), total N (N, Mg ha^{-1}), and C/N ratio at 0.40 m soil depth (mean value ± standard error of the mean).

	SOC (Mg ha^{-1})	**N (Mg ha^{-1})**	**C/N**
High	125.5±12.9	9.3±1.1	13.8±0.6
Low	141.7±14.0	10.8±1.2	13.2±0.3
High-BT	152.0±8.2 a	10.9±1.1	14.2±0.8
High-OT	99.0±15.5 b	7.6±1.6	13.5±0.9
Low-BT	162.2±11.6 a	12.0±1.0	13.6±0.2
Low-OT	121.2±22.5 ab	9.6±2.1	12.8±0.5

Means followed by the same letters indicate no significant differences between levels of the factors according to the least-square means test (P<0.05). High= high cover; Low= low cover; OT: outside tree canopy

Table 2. Parameters of the polynomial regression between Rh and both SoilT and SWC for each level used to estimate the daily Rh, F statistic (model variance / residual variance), degrees of freedom of residual variance (df residual), and P-value.

	β_0	**β_1**	**β_2**	**β_3**	**F value**	**df residual**	**P value**
High-BT	0.18634	0.1067	0.1537	-0.0034	11.65	56	<0.0001
High-OT	0.07279	0.0590	0.2651	-0.0075	12.39	55	<0.0001
Low-BT	1.28364	0.0041	0.1120	-0.0046	6.97	56	<0.001
Low-OT	0.21601	0.0382	0.1674	-0.0046	7.97	56	<0.001

High= high cover; Low= low cover; BT: below tree canopy; OT: outside tree canopy

Table 3. Annual soil C balance items and overall balance (Mg ha^{-1} yr^{-1} of C, mean values ± standard errors of means) and P values associated with the study factors (ANOVA). Means followed by the same letters indicate no significant differences between levels according to the least-square means test ($P<0.05$). Values within brackets represent the range of the observations (min-max)

		Class			**Position**				
#	**Item**	**P**	**High**	**Low**	**P**	**High-BT**	**High-OT**	**Low-BT**	**Low-OT**
1	Cres	<0.01	0.41 ± 0.06 b	0.65 ± 0.06 a	NS	0.34 ± 0.04 (0.11-0.64)	0.47 ± 0.12 (0-1.31)	0.60 ± 0.04 (0.32-0.77)	0.69 ± 0.13 (0-1.38)
2	Croo	<0.0001	1.88 ± 0.19 b	3.27 ± 0.18 a	NS	1.71 ± 0.30 (0.56-3.28)	2.06 ± 0.25 (0.59-2.92)	3.25 ± 0.30 (1.58-4.62)	3.30 ± 0.20 (2.03-3.82)
3	Crhi	<0.0001	1.53 ± 0.16 b	2.66 ± 0.14 a	NS	1.39 ± 0.24 (0.69-4.03)	1.67 ± 0.20 (0.72-3.59)	2.64 ± 0.24 (1.95-5.69)	2.68 ± 0.16 (2.5-4.71)
4=1+2+3	Cpas	<0.0001	3.82 ± 0.40 b	6.58 ± 0.35 a	NS	3.43 ± 0.57 (1.44-7.74)	4.21 ± 0.56 (1.32-7.82)	6.49 ± 0.56 (3.85-10.88)	6.67 ± 0.45 (4.59-9.73)
5	Clit	<0.05	2.27 ± 0.55 a	1.57 ± 0.24 b	<0.0001	3.59 ± 0.46 a (2.88-4.93)	0.95 ± 0.25 c (0.39-1.600)	2.16 ± 0.17 b (1.86-2.47)	0.99 ± 0.11 c (0.74-1.28)
6	Cfin*	NS	11.45 ± 0.07 (11.28-11.56)	9.14 ± 1.11 (6.19-11)	-	-	-	-	-
7	Crhf*	NS	1.72 ± 0.01 (1.69-1.73)	1.37 ± 0.17 (0.93-1.65)	-	-	-	-	-
8=6+7	Ctrs*	NS	13.17 ± 0.08 (12.98-13.3)	10.51 ± 1.27 (7.11-12.65)	-	-	-	-	-
9=5+8	Ctre	<0.05	8.85 ± 3.00 a	6.83 ± 2.28 b	<0.0001	16.76 ± 0.51 a (15.86-18.23)	0.95 ± 0.25 c (0.39-1.60)	12.66 ± 1.23 b (9.54-14.73)	0.99 ± 0.11 c (0.74-1.28)
10	Cfae	NS	0.13 ± 0.02 (0.06-0.29)	0.18 ± 0.01 (0.1-0.24)	-	-	-	-	-
11=4+9+10	Cinput	NS	12.81 ± 1.62	13.58 ± 1.27	<0.0001	20.32 ± 0.5 a (18.22-24.53)	5.29 ± 0.63 c (2.38-9.5)	19.33 ± 0.72 a (15.78-22.92)	7.84 ± 0.42 b (5.97-10.7)
12	Coutput	<0.01	9.83 ± 1.06 a	7.65 ± 0.70 b	<0.0001	14.24 ± 0.90 a (9.81-21.45)	5.41 ± 0.55 c (3.14-9.49)	10.12 ± 0.94 b (4.3-13.46)	5.19 ± 0.21 c (4.09-6.56)
13=11-12	Cbalance	<0.001	2.98 ± 0.96 b	5.93 ± 0.78 a	<0.0001	6.08 ± 1.13 b (-1.72-12.79)	-0.12 ± 0.91 c (-4.91-4.3)	9.21 ± 0.73 a (4.98-12.39)	2.65 ± 0.29 c (0.99-4.33)

*Cfin, Crhf, Ctrs refer to average values within BT areas

High: high cover; Low: low cover; BT: below tree canopy; OT: outside tree canopy; Cres: C input from pasture residues; Croo: C input from pasture roots decomposition; Crhi: C input from pasture rhizodeposition; Cpas: C input from pasture: Clit: C input from tree litterfall; Cfin: C input from tree fine roots decomposition: Crhf: C input from tree rhizodeposition; Ctrs: C input from tree root system; Ctre: C input from trees; Cfae: C input from animal faeces

Figures

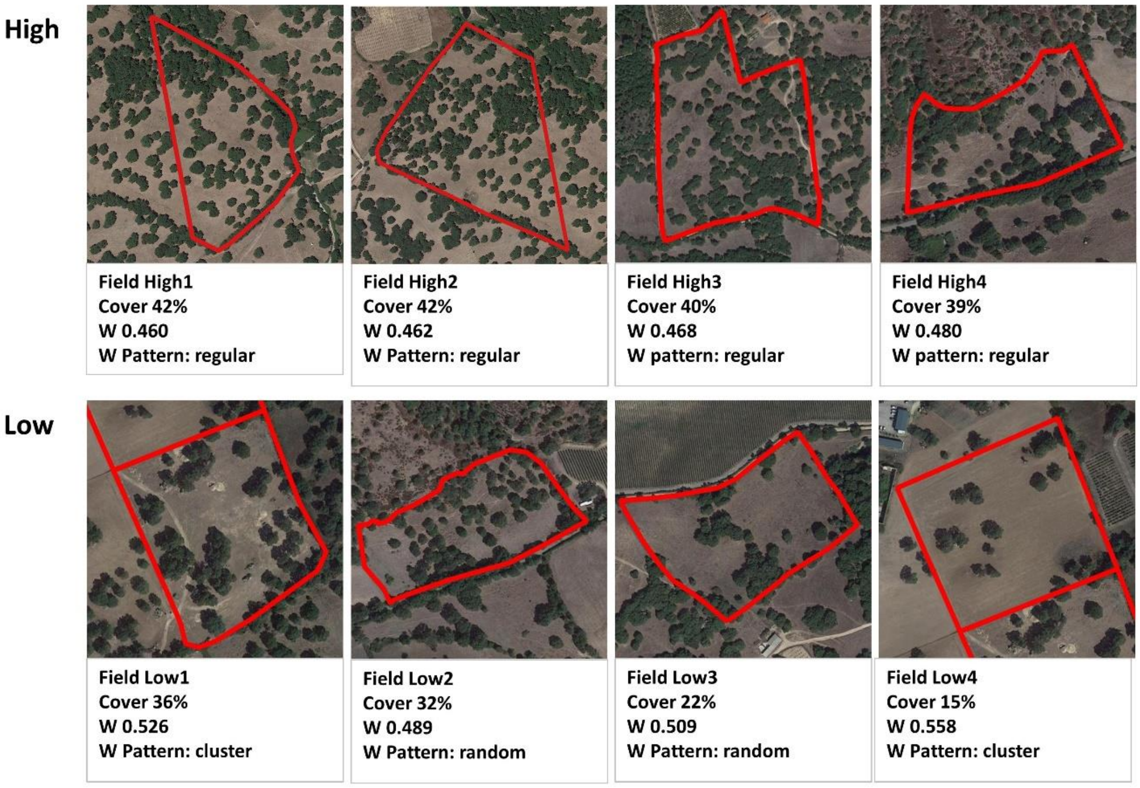


Figure 1. Study fields classification. The W pattern is defined, according to Hu et al. (2014), as random when W is within the 95% confidence interval of a reference mean of 0.500 (0.485÷0.515), regular when W is lower than the lower limit, and cluster when W is higher than the upper limit. High = high cover; Low = low cover

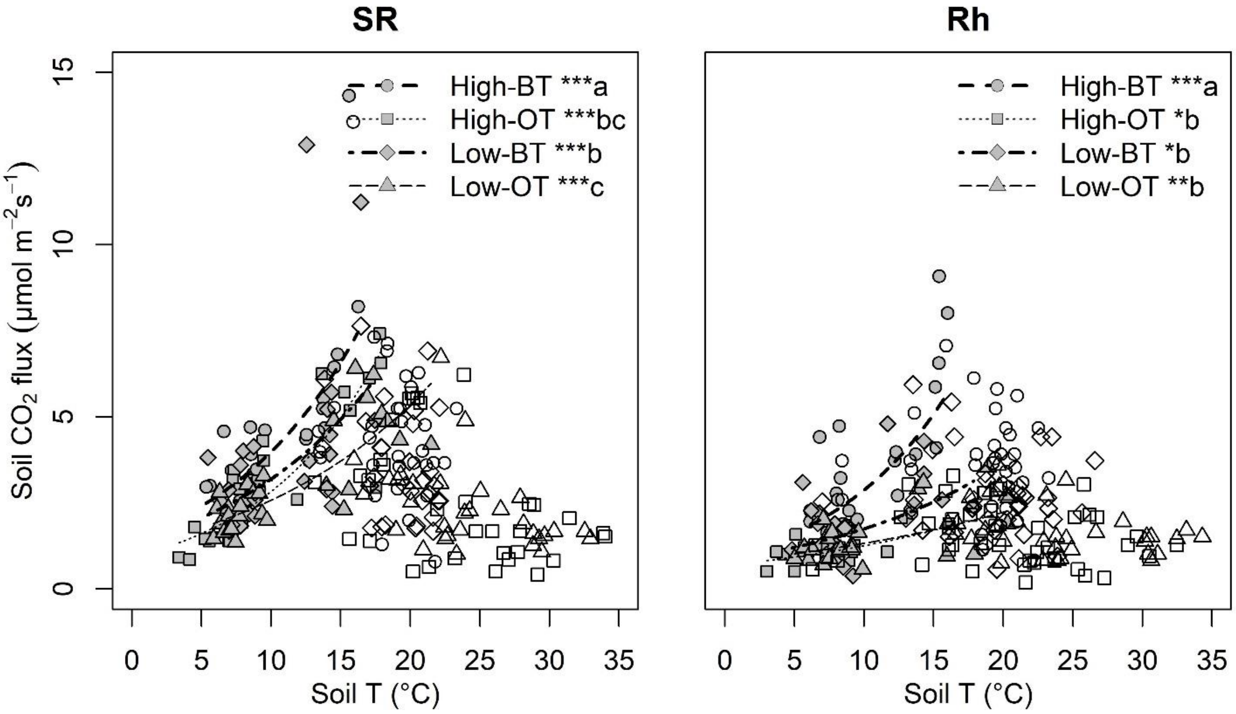


Figure 2. Relationships between total Soil Respiration (SR, µmol m^{-2} s^{-1}), heterotrophic Respiration (Rh, µmol m^{-2} s^{-1}), and Soil Temperature (°C) above (full bullets) and below (blank bullets) the soil water content critical threshold. Different letters indicate different models according to ANCOVA multiple comparisons. High = high cover; Low = low cover; BT: below tree canopy; OT: outside tree canopy. (***: P<0.0001; **: P<0.001; *: P<0.01)

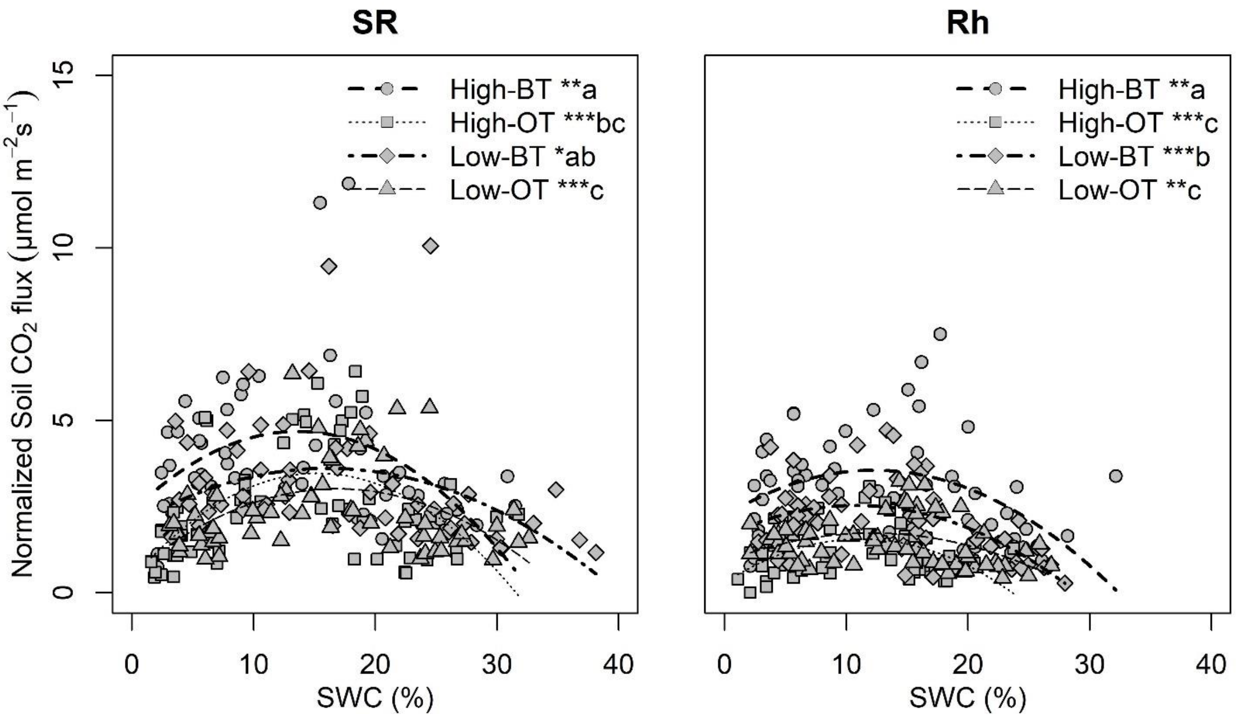


Figure 3. Relationships between normalized total Soil Respiration (SR, µmol m^{-2} s^{-1}), normalized heterotrophic Respiration (Rh, µmol m^{-2} s^{-1}), and Soil Water Content (SWC, %). Different letters indicate different models according to ANCOVA multiple comparisons. High = high cover; Low = BT: below tree canopy; OT: outside tree canopy. (***: $P<0.0001$; **: $P<0.001$; *: $P<0.01$)

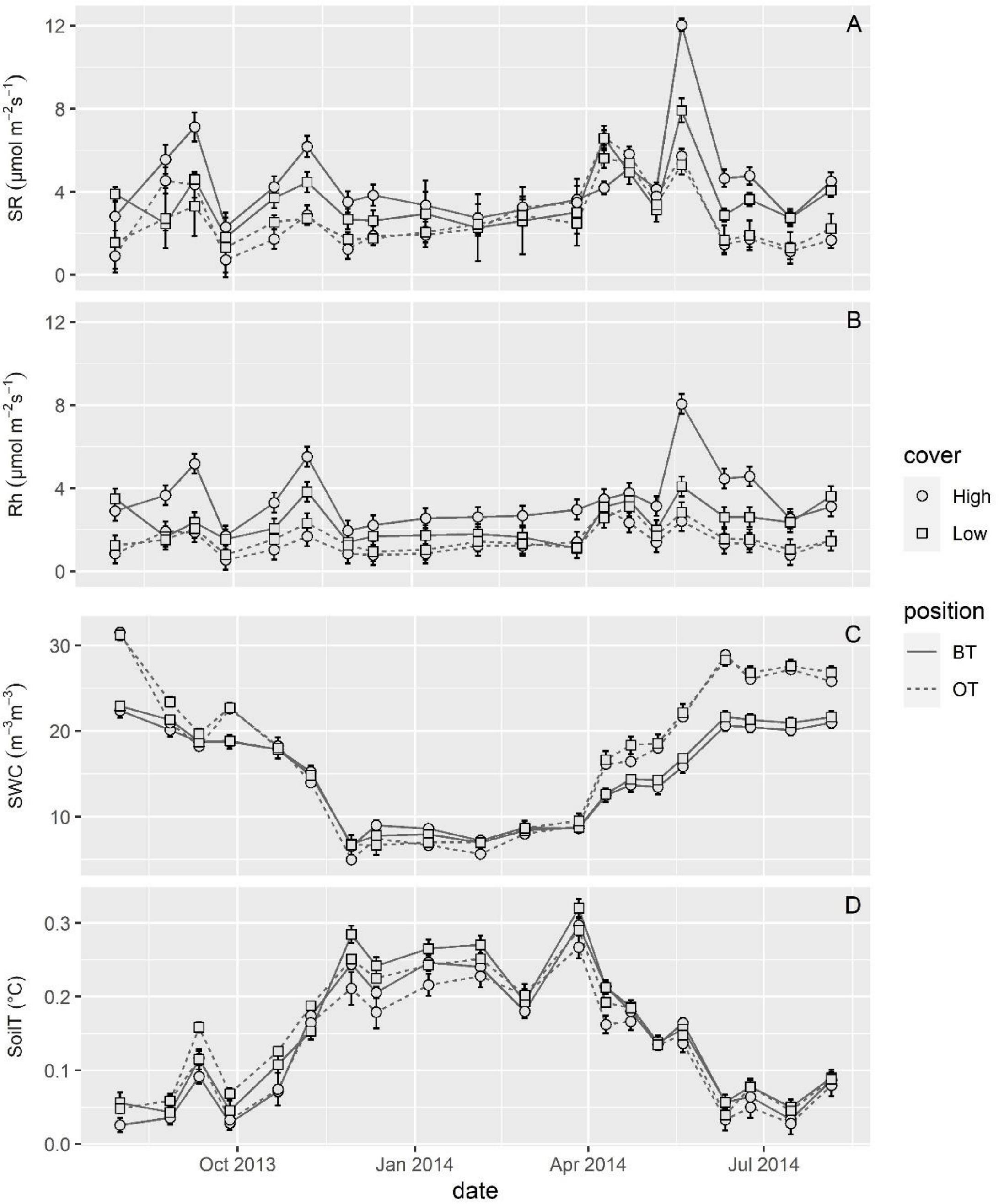


Figure 4. Dynamics of total Soil Respiration (SR, A), its heterotrophic fraction (Rh, B), Soil Water Content at -7 cm depth (SWC, C), and Soil Temperature at -10 cm depth (T – 10 cm, D) from August 2013 to August 2014. Vertical bars represent the within-date standard error of the least square means (n=4). High = high cover; Low = low cover; BT = below the tree crown; OT = outside the tree crown projection

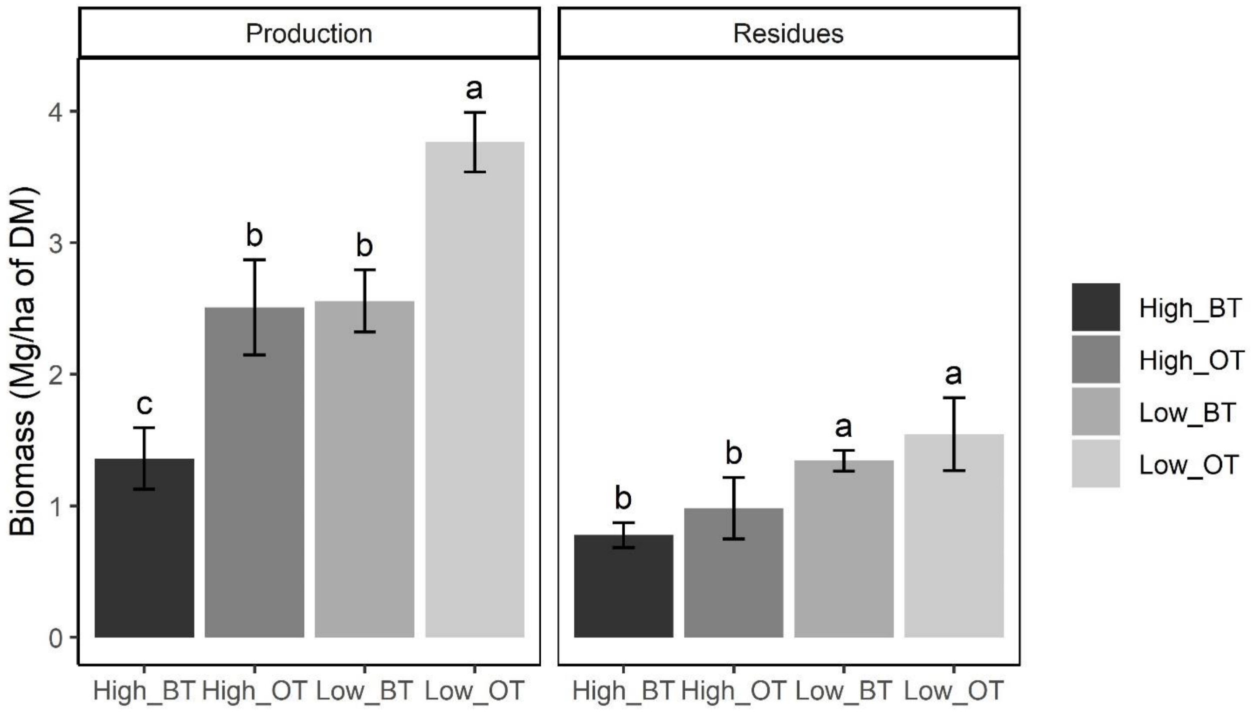


Figure 5. Mean values (from 2012-13 to 2014-15) of Pasture Production (left box) and Pasture Residues (right box) (Mg ha-1 of DM). Bars overtopped by the same letter represent means not significantly different according to the least square means test (P<0.05). Bars represent the standard errors of means (n=4). High = high cover; Low = low cover; BT = below the tree crown; OT = outside the tree crown projection

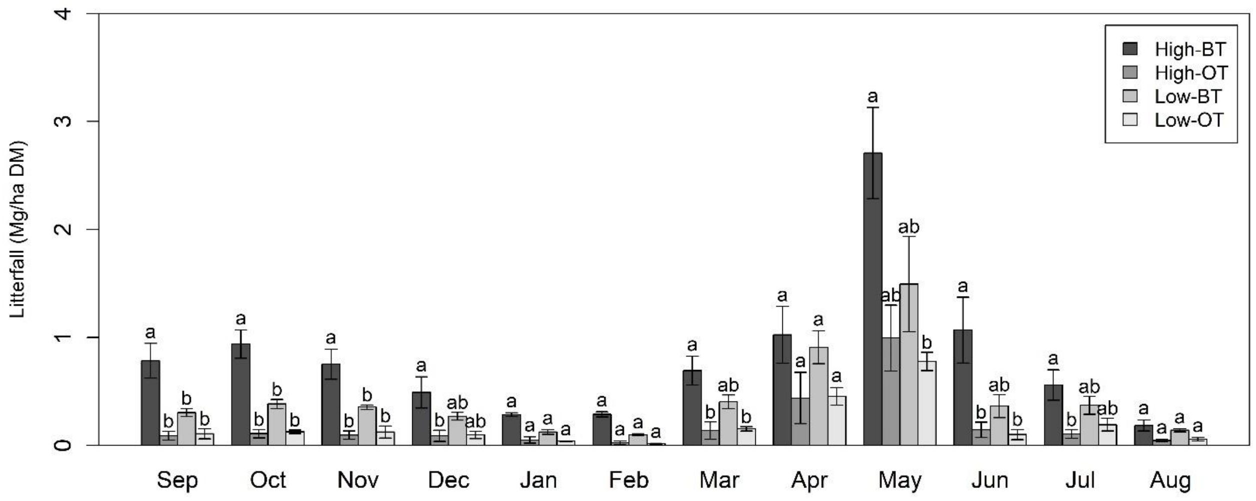


Figure 6. Tree litterfall (Mg ha-1 of DM) monthly dynamics (September 2014 – August 2015). Bars overtopped by the same letter represent means not significantly different according to the least-square means test (P<0.05). Bars represent the standard errors of means (n=4). High = high cover; Low = low cover; BT = below the tree crown; OT = outside the tree crown projection

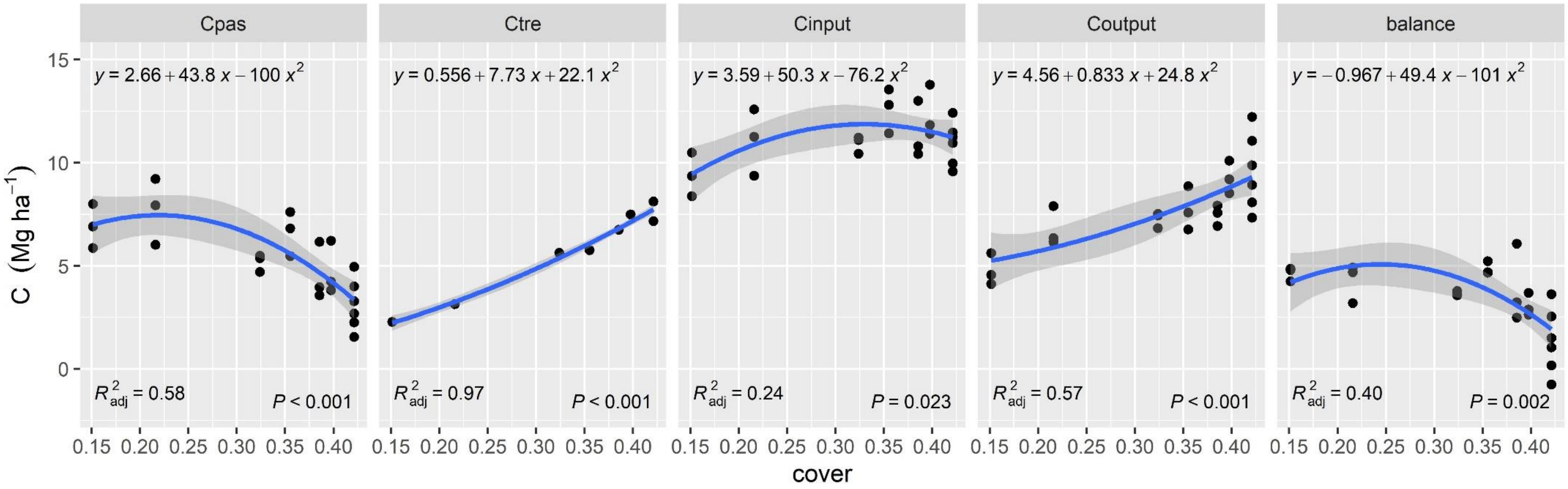


Figure 7. Relationships between the field-scale weighted C input from pasture (Cpas), C input from trees (Ctre), total C input (Cinput), C output (Coutput), and C balance (balance), and the tree cover (cover)

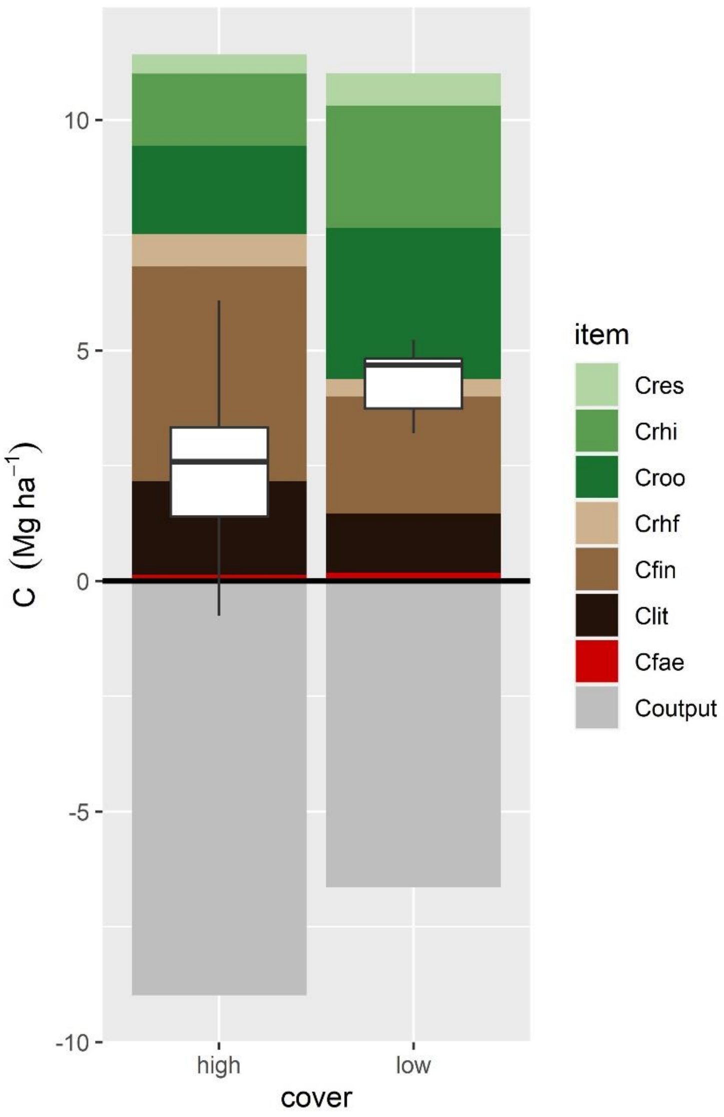
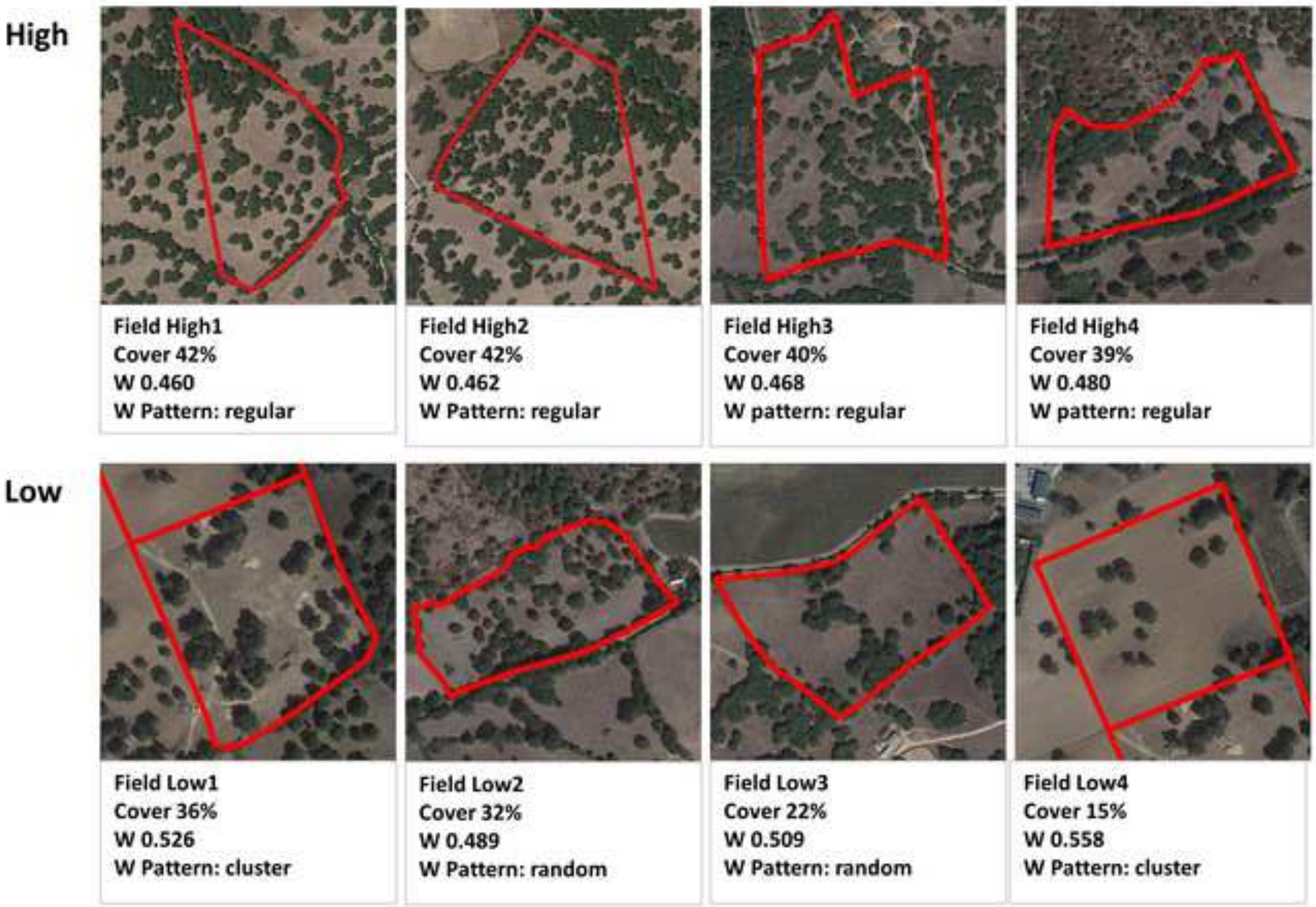
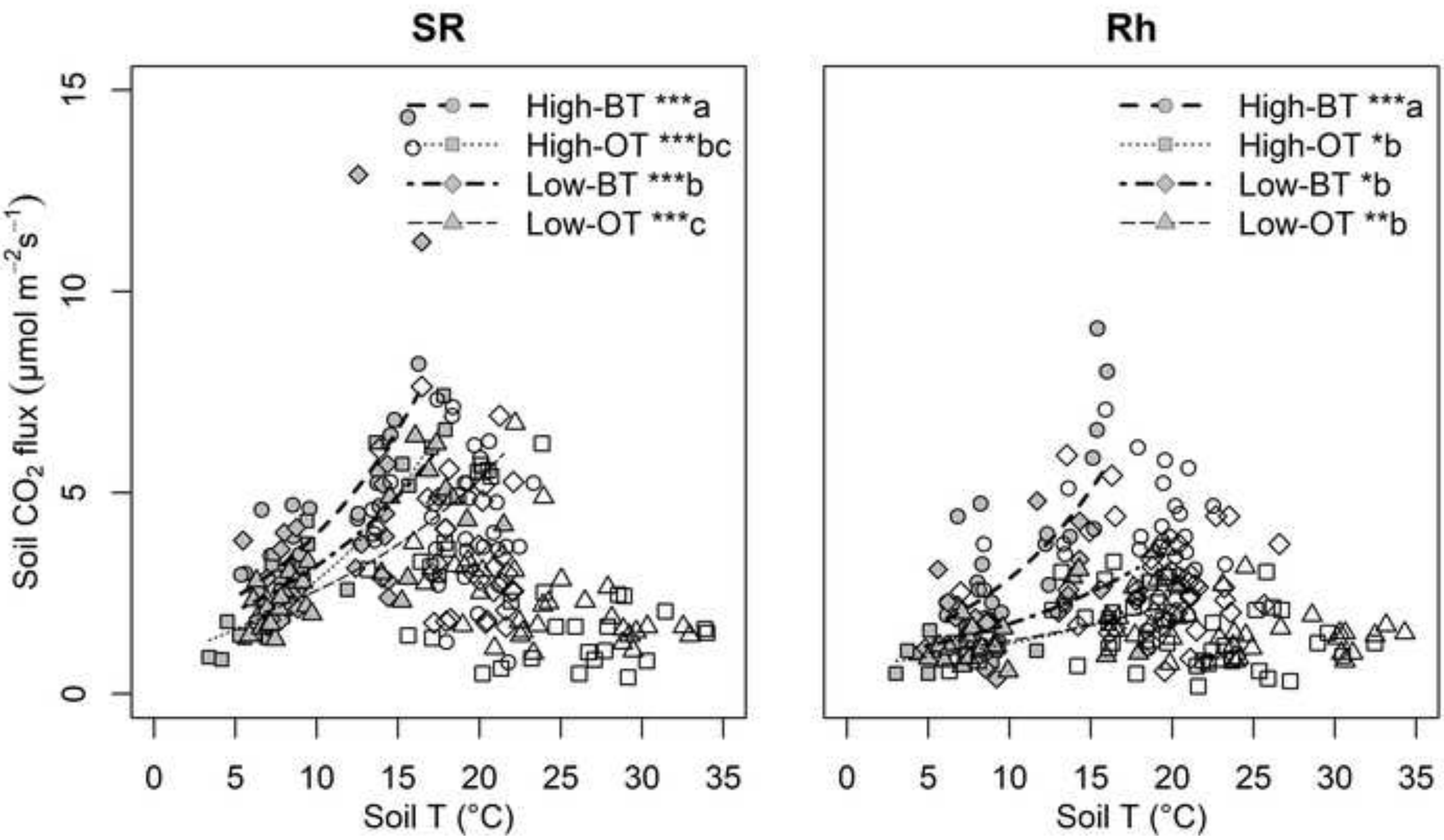
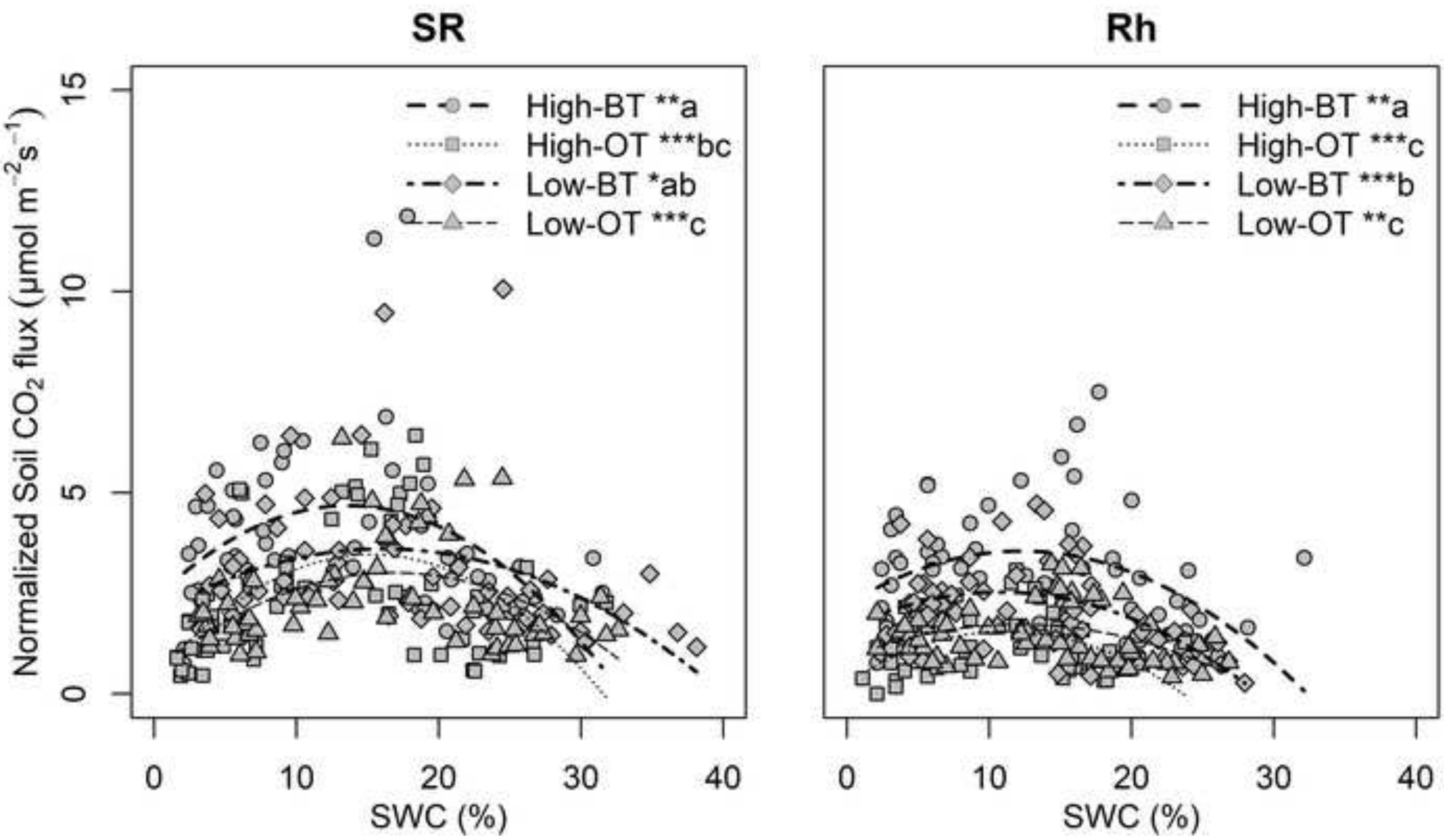


Figure 8. Field-weighted C balance items (Mg ha^{-1} of C) under High and Low tree cover. Green-scale bars represent the balance item contributing to the C input from pasture (Cpas), brown-scale bars represent the C input from trees (Ctre), red bar represents the C input from faeces (Cfae), grey bar represents the C output. The boxplot within bars represent the field-weighted C balance (box's edges represent the interquartile range, vertical lines are the 95% confidence interval, lines within boxes represent are median values)

Figure 1


High
Field High1
Cover 42%
W 0.460
W Pattern: regular
Field High2
Cover 42%
W 0.462
W Pattern: regular
Field High3
Cover 40%
W 0.468
W pattern: regular
Field High4
Cover 39%
W 0.480
W pattern: regular
Low
Field Low1
Cover 36%
W 0.526
W Pattern: cluster
Field Low2
Cover 32%
W 0.489
W Pattern: random
Field Low3
Cover 22%
W 0.509
W Pattern: random
Field Low4
Cover 15%
W 0.558
W Pattern: cluster


SR
Rh
Soil CO2 flux (μmol m⁻²s⁻¹)
0
5
10
15
High-BT ***a
High-OT ***bc
Low-BT ***b
Low-OT ***c
High-BT ***a
High-OT *b
Low-BT *b
Low-OT **b
0
5
10
15
20
25
30
35
Soil T (°C)
Soil T (°C)


SR
Rh
Normalized Soil CO_2 flux ($\mu mol\ m^{-2} s^{-1}$)
0
5
10
15
SWC (%)
0
10
20
30
40
High-BT **a
High-OT ***bc
Low-BT *ab
Low-OT ***c
High-BT **a
High-OT ***c
Low-BT ***b
Low-OT **c

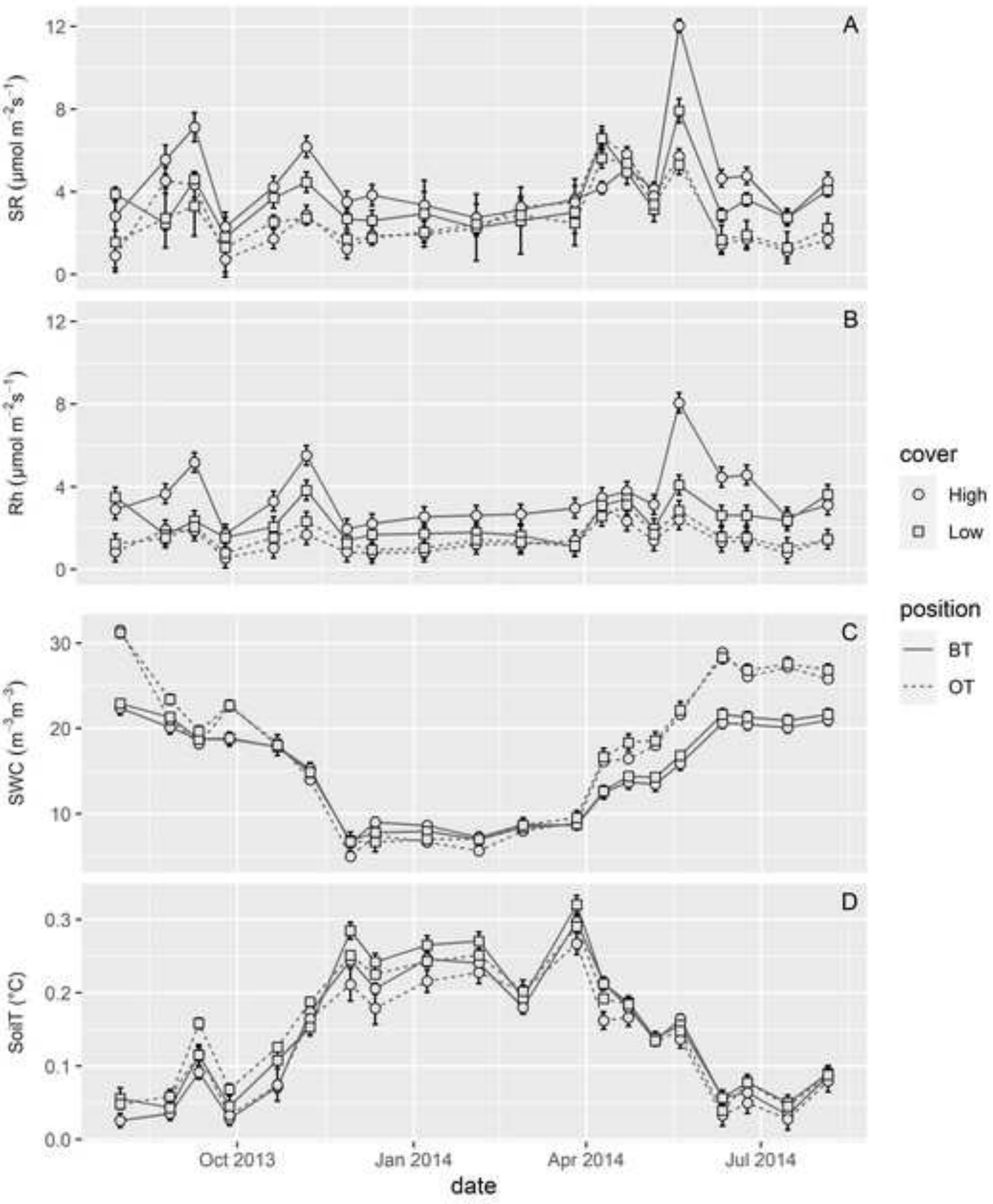

A
SR (μmol m^{-2}s^{-1})
0
4
8
12
B
Rh (μmol m^{-2}s^{-1})
0
4
8
12
C
SWC (m^{-3}m^{-3})
10
20
30
D
SoilT (°C)
0.0
0.1
0.2
0.3
Oct 2013
Jan 2014
Apr 2014
Jul 2014
date
cover
High
Low
position
BT
OT

Figure 5

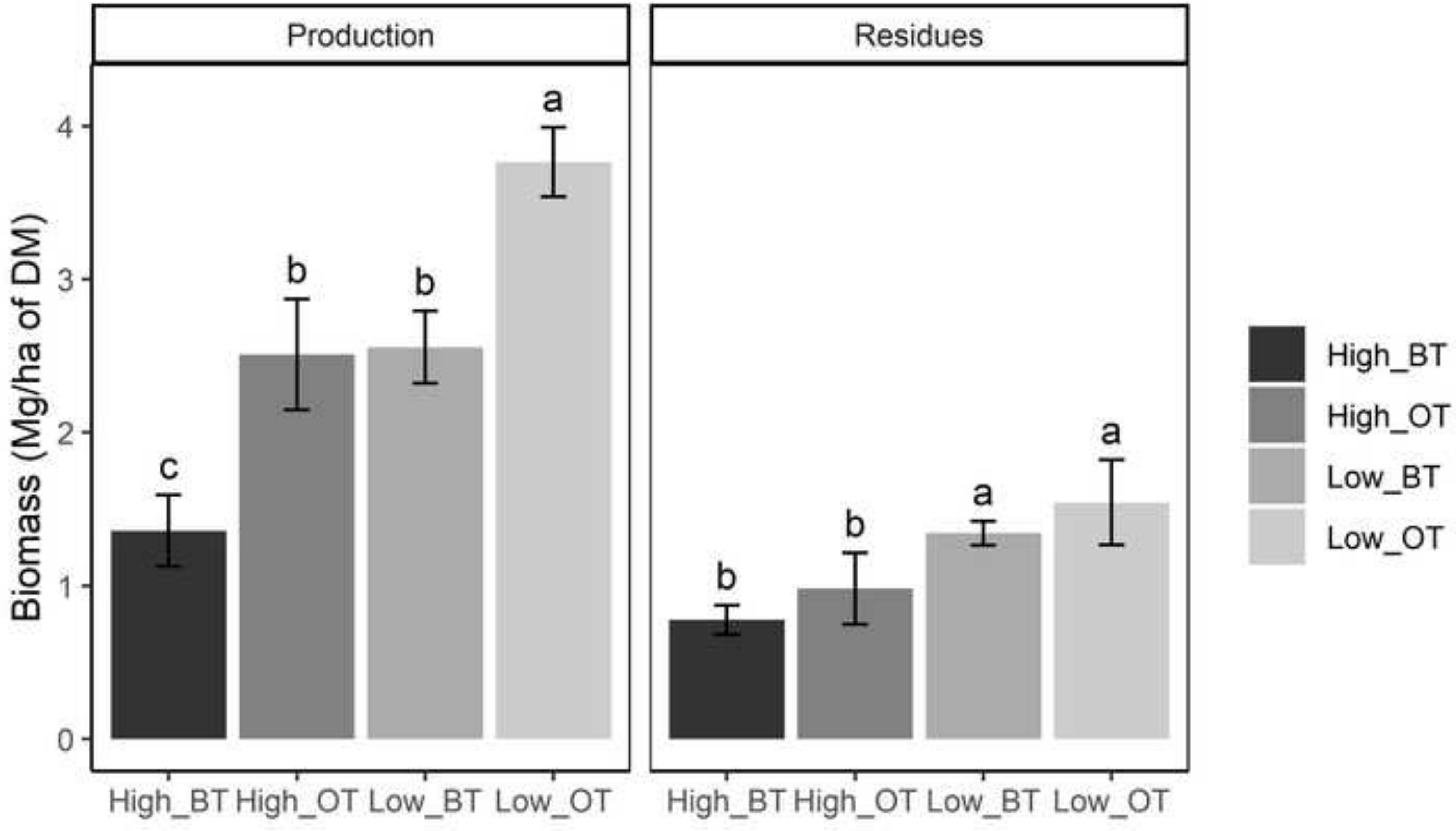

Production
Residues
Biomass (Mg/ha of DM)
0
1
2
3
4
c
b
b
a
b
b
a
a
High_BT
High_OT
Low_BT
Low_OT
High_BT
High_OT
Low_BT
Low_OT
High_BT
High_OT
Low_BT
Low_OT

Figure 6

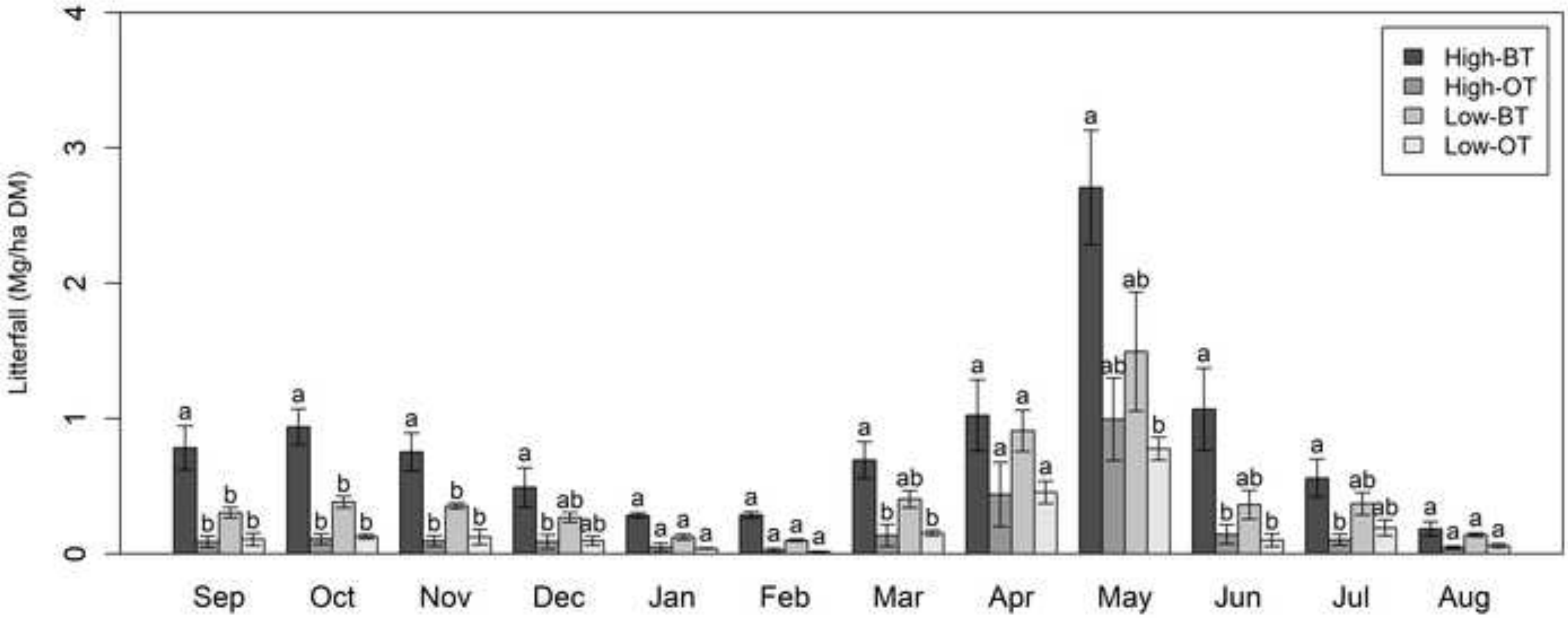
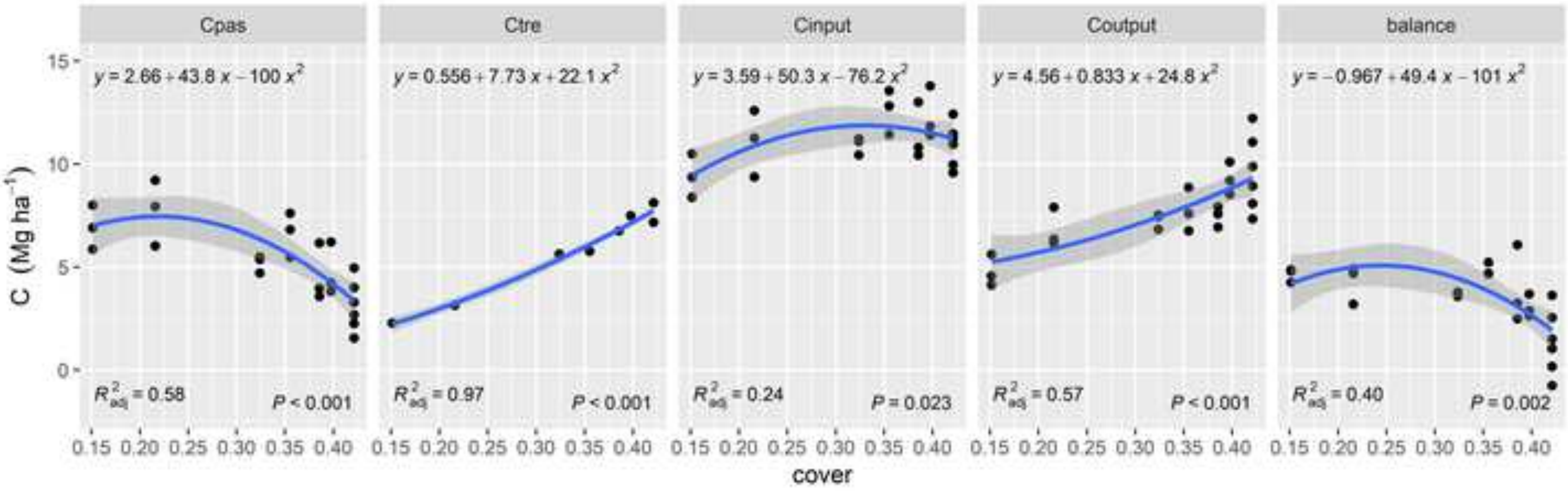

Litterfall (Mg/ha DM)
0
1
2
3
4
High-BT
High-OT
Low-BT
Low-OT
a b b b
a b b b
a b b b
a b ab ab
a a a a
a a a a
a b ab b
a a a a
a ab ab b
a b ab b
a b ab ab
a a a a
Sep
Oct
Nov
Dec
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug

Figure 7


Cpas
Ctre
Cinput
Coutput
balance
y = 2.66 + 43.8 x − 100 x²
y = 0.556 + 7.73 x + 22.1 x²
y = 3.59 + 50.3 x − 76.2 x²
y = 4.56 + 0.833 x + 24.8 x²
y = −0.967 + 49.4 x − 101 x²
R²adj = 0.58
P < 0.001
R²adj = 0.97
P < 0.001
R²adj = 0.24
P = 0.023
R²adj = 0.57
P < 0.001
R²adj = 0.40
P = 0.002
C (Mg ha⁻¹)
15
10
5
0
0.15
0.20
0.25
0.30
0.35
0.40
cover

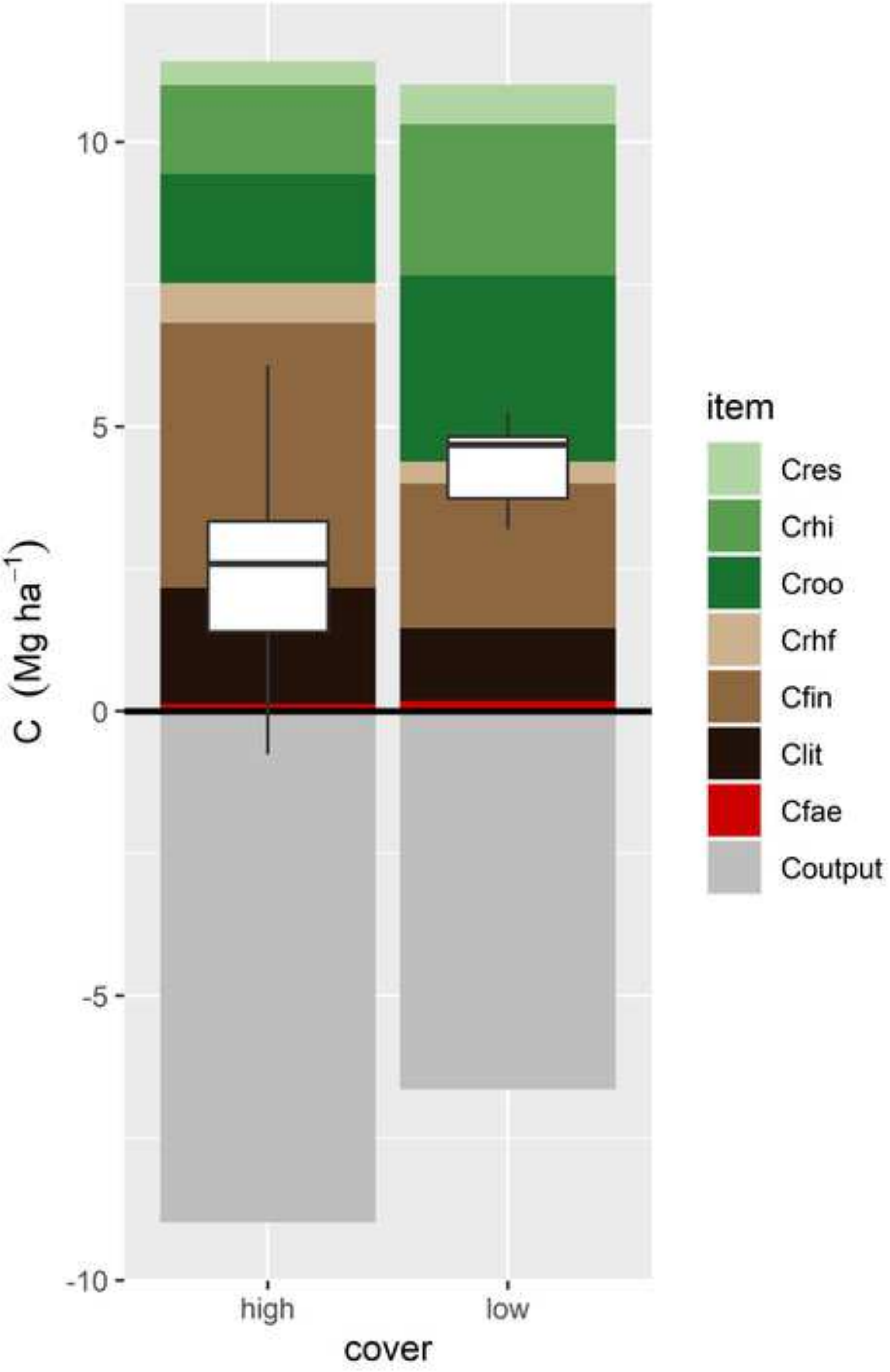

C (Mg ha^{-1})
10
5
0
-5
-10
high
low
cover
item
Cres
Crhi
Croo
Crhf
Cfin
Clit
Cfae
Coutput

Table 1. Soil Organic Carbon (SOC, Mg ha^{-1}), total N (N, Mg ha^{-1}), and C/N ratio at 0.40 m soil depth (mean value ± standard error of the mean).

	SOC (Mg ha^{-1})	**N (Mg ha^{-1})**	**C/N**
High	125.5±12.9	9.3±1.1	13.8±0.6
Low	141.7±14.0	10.8±1.2	13.2±0.3
High-BT	152.0±8.2 a	10.9±1.1	14.2±0.8
High-OT	99.0±15.5 b	7.6±1.6	13.5±0.9
Low-BT	162.2±11.6 a	12.0±1.0	13.6±0.2
Low-OT	121.2±22.5 ab	9.6±2.1	12.8±0.5

Means followed by the same letters indicate no significant differences between levels of the factors according to the least-square means test (P<0.05). High= high cover; Low= low cover; OT: outside tree canopy

Table 1. Parameters of the polynomial regression between Rh and both SoilT and SWC for each level used to estimate the daily Rh, F statistic (model variance / residual variance), degrees of freedom of residual variance (df residual), and P-value.

	β_0	**β_1**	**β_2**	**β_3**	**F value**	**df residual**	**P value**
High-BT	0.18634	0.1067	0.1537	-0.0034	11.65	56	<0.0001
High-OT	0.07279	0.0590	0.2651	-0.0075	12.39	55	<0.0001
Low-BT	1.28364	0.0041	0.1120	-0.0046	6.97	56	<0.001
Low-OT	0.21601	0.0382	0.1674	-0.0046	7.97	56	<0.001

High= high cover; Low= low cover; BT: below tree canopy; OT: outside tree canopy

Table 1. Annual soil C balance items and overall balance (Mg ha^{-1} yr^{-1} of C, mean values ± standard errors of means) and P values associated with the study factors (ANOVA). Means followed by the same letters indicate no significant differences between levels according to the least-square means test (P<0.05). Values within brackets represent the range of the observations (min-max)

		Class			**Position**				
#	**Item**	**P**	**High**	**Low**	**P**	**High-BT**	**High-OT**	**Low-BT**	**Low-OT**
1	Cres	<0.01	0.41 ± 0.06 b	0.65 ± 0.06 a	NS	0.34 ± 0.04 (0.11-0.64)	0.47 ± 0.12 (0-1.31)	0.60 ± 0.04 (0.32-0.77)	0.69 ± 0.13 (0-1.38)
2	Croo	<0.0001	1.88 ± 0.19 b	3.27 ± 0.18 a	NS	1.71 ± 0.30 (0.56-3.28)	2.06 ± 0.25 (0.59-2.92)	3.25 ± 0.30 (1.58-4.62)	3.30 ± 0.20 (2.03-3.82)
3	Crhi	<0.0001	1.53 ± 0.16 b	2.66 ± 0.14 a	NS	1.39 ± 0.24 (0.69-4.03)	1.67 ± 0.20 (0.72-3.59)	2.64 ± 0.24 (1.95-5.69)	2.68 ± 0.16 (2.5-4.71)
4=1+2+3	Cpas	<0.0001	3.82 ± 0.40 b	6.58 ± 0.35 a	NS	3.43 ± 0.57 (1.44-7.74)	4.21 ± 0.56 (1.32-7.82)	6.49 ± 0.56 (3.85-10.88)	6.67 ± 0.45 (4.59-9.73)
5	Clit	<0.05	2.27 ± 0.55 a	1.57 ± 0.24 b	<0.0001	3.59 ± 0.46 a (2.88-4.93)	0.95 ± 0.25 c (0.39-1.600)	2.16 ± 0.17 b (1.86-2.47)	0.99 ± 0.11 c (0.74-1.28)
6	Cfin*	NS	11.45 ± 0.07 (11.28-11.56)	9.14 ± 1.11 (6.19-11)	-	-	-	-	-
7	Crhf*	NS	1.72 ± 0.01 (1.69-1.73)	1.37 ± 0.17 (0.93-1.65)	-	-	-	-	-
8=6+7	Ctrs*	NS	13.17 ± 0.08 (12.98-13.3)	10.51 ± 1.27 (7.11-12.65)	-	-	-	-	-
9=5+8	Ctre	<0.05	8.85 ± 3.00 a	6.83 ± 2.28 b	<0.0001	16.76 ± 0.51 a (15.86-18.23)	0.95 ± 0.25 c (0.39-1.60)	12.66 ± 1.23 b (9.54-14.73)	0.99 ± 0.11 c (0.74-1.28)
10	Cfae	NS	0.13 ± 0.02 (0.06-0.29)	0.18 ± 0.01 (0.1-0.24)	-	-	-	-	-
11=4+9+10	Cinput	NS	12.81 ± 1.62	13.58 ± 1.27	<0.0001	20.32 ± 0.5 a (18.22-24.53)	5.29 ± 0.63 c (2.38-9.5)	19.33 ± 0.72 a (15.78-22.92)	7.84 ± 0.42 b (5.97-10.7)
12	Coutput	<0.01	9.83 ± 1.06 a	7.65 ± 0.70 b	<0.0001	14.24 ± 0.90 a (9.81-21.45)	5.41 ± 0.55 c (3.14-9.49)	10.12 ± 0.94 b (4.3-13.46)	5.19 ± 0.21 c (4.09-6.56)
13=11-12	Cbalance	<0.001	2.98 ± 0.96 b	5.93 ± 0.78 a	<0.0001	6.08 ± 1.13 b (-1.72-12.79)	-0.12 ± 0.91 c (-4.91-4.3)	9.21 ± 0.73 a (4.98-12.39)	2.65 ± 0.29 c (0.99-4.33)

*Cfin, Crhf, Ctrs refer to average values within BT areas

High: high cover; Low: low cover; BT: below tree canopy; OT: outside tree canopy; Cres: C input from pasture residues; Croo: C input from pasture roots decomposition; Crhi: C input from pasture rhizodeposition; Cpas: C input from pasture: Clit: C input from tree litterfall; Cfin: C input from tree fine roots decomposition: Crhf: C input from tree rhizodeposition; Ctrs: C input from tree root system; Ctre: C input from trees; Cfae: C input from animal faeces

Declaration of interests

☒The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

☐The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: