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Salvage logging management reduces C sequestration

Title:
Post-fire salvage logging reduces carbon sequestration in Mediterranean coniferous forest

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Abstract

Post-fire salvage logging is a common silvicultural practice around the world, with the potential to alter the regenerative capacity of an ecosystem and thus its role as a source or a sink of carbon. However, there is no information on the effect of burnt wood management on the net ecosystem carbon balance. Here, we examine for the first time the effect of post-fire burnt wood management on the net ecosystem carbon balance by comparing the carbon exchange of two treatments in a burnt Mediterranean coniferous forest treated by salvage logging (SL, felling and removing the logs and masticating the woody debris) and non intervention (NI, all trees left standing) using eddy covariance measurements. Using different partitioning approaches, we analyze the evolution of photosynthesis and respiration processes together with measurements of vegetation cover and soil respiration and humidity to interpret the differences in the measured fluxes and underlying processes. Results show that SL enhanced CO_2 emissions of this burnt pine forest by more than 120 g C m^{-2} compared to the NI treatment for the period June-December 2009. Although soil respiration was around 30% higher in NI during growing season, this was more than offset by photosynthesis, as corroborated by increases in vegetation cover and Evapotranspiration. Since SL is counterproductive to climate-change and Kyoto protocol objectives of optimal C sequestration by terrestrial ecosystems, less aggressive burnt wood management policies should be considered.

Keywords

Burnt wood management, eddy covariance, forest carbon balance, photosynthesis, respiration, wildfire

1. Introduction

Wildfire is a frequent perturbation in Mediterranean-type ecosystems (Moreno *et al.*, 1998) inducing changes in land use/cover types (Lloret *et al.*, 2002; Quintana *et al.*, 2004; Viedma *et al.*, 2006) and thereby altering the balances of water, energy and carbon (Amiro *et al.*, 1999; Beringer *et al.*, 2003; Santos *et al.*, 2003; Amiro *et al.*, 2006). Although CO_2 emission immediately after fire can be reasonably estimated (Conard and Ivanova, 1997; Harden *et al.*, 2000; Page *et al.*, 2002; Van der Werf *et al.*, 2003), long-term effects on the carbon balance during ecosystem regeneration are less certain and influenced by several factors. Enhanced rates of soil CO_2 effluxes as well as large changes in the rate of ecosystem photosynthetic carbon uptake may also occur during several months after wildfire (Santos *et al.*, 2003). However, other studies suggest a reduction of the soil CO_2 efflux in regenerating ecosystems (Dore *et al.*, 2010; Irvine *et al.*, 2007), which could be attributed to the positive relation between aboveground productivity and respiration (Irvine *et al.*, 2007; Janssens *et al.*, 2001). Finally, some studies reveal decreased in evapotranspiration (*ET*) and a conversion from carbon sink to source with magnitudes differing over the years following wildfire (Amiro, 2001; Amiro *et al.*, 2003; Amiro *et al.*, 2006; Mkhabela *et al.*, 2009).

Post-fire management may affect the fluxes of carbon and hence the role of the ecosystem as a carbon source or sink. The capacity for carbon sequestration after a wildfire will depend on the regenerative capacity of the vegetation that determines net primary production. For example, reforestation soon after a stand-replacing disturbance accelerates the conversion from carbon source to sink (Magnani *et al.*, 2007) although natural regeneration may similarly increase carbon sequestration (Amiro, 2001). In addition, forest fires leave large amounts of partially burnt wood that may be handled in several ways according to ecological or management requirements, increasing productivity (Donato *et al.*, 2006; Castro *et al.*, 2010a) and simultaneously enhancing C emissions due to decomposition (Jomura *et al.*, 2008; Marañón-Jiménez *et al.*, 2011). Therefore, the net carbon balance after a wildfire may differ as a consequence of forest management (Stark *et al.*, 2006), whether by a direct effect on vegetation cover and development or as mediated by the presence of burnt wood.

One of the first and most important post-fire management decisions regards the fate of the burnt wood. After a fire, forest managers frequently apply salvage logging, removing the burnt tree trunks, and often eliminating the remaining woody debris by chopping, mastication, fire, etc. (McIver and Starr, 2000; Bautista *et al.*, 2004; Lindenmayer *et al.*, 2008). Post-fire salvage logging has been routinely practiced by forest managers worldwide, motivated by factors economic, silvicultural, or even aesthetic (McIver and Starr, 2000; Lindenmayer and Noss, 2006; Castro *et al.*, 2010b). However, there is increasing evidence that salvage logging degrades ecosystem function and structure in terms of vegetation regeneration, animal and plant diversity, watershed runoff and erosion, or nutrient cycling (Donato *et al.*, 2006; Lindenmayer *et al.*, 2008; Castro *et al.*, 2010a, 2010b). In the same way, post-fire burnt wood management can potentially alter the ecosystem carbon balance. On one hand, large amounts of carbon stored in the burnt wood can decompose and be emitted as CO_2 to the atmosphere. On the other hand, the presence of burnt wood can enhance the regeneration capacity both by incorporating nutrients into the soil as it decomposes, and also by improving microclimatic conditions that benefit net primary productivity (Donato *et al.*, 2006; Lindenmayer *et al.*, 2008; Castro *et al.*, 2010a). Post-fire burnt wood management could therefore affect the net ecosystem carbon balance even during several years after the wildfire. To date however, there are no studies on the effects of burnt wood management on net carbon exchange after a wildfire.

The aim of this paper is to examine the effect of the post-fire salvage logging on the net ecosystem carbon balance. We compare the CO_2 exchange, measured during the fourth year following wildfire, of two treatments with different post-fire management treatments: “Salvage Logging” (SL) and “Non Intervention” (NI). We used the eddy covariance (EC) technique to directly measure net carbon, water vapour and energy exchanges between the atmosphere and the biosphere (Wofsy *et al.*, 1993; Baldocchi, 2003). In addition, soil CO_2 effluxes, vegetation cover and meteorological variables were measured to interpret the patterns of carbon fluxes and underlying processes. We hypothesized that post-fire burnt wood management would influence the magnitude of carbon exchange between the ecosystem and the atmosphere, as the presence of the burnt wood may alter both respiration rates and gross primary production. These measurements are critical to understand ecosystem carbon exchange at a global scale given the large areas of forest burned every year, and are a necessary step to ascertain the effect of management practices on the ecosystem carbon balance.

2. Materials and Methods

2. 1 *Study area and experimental design*

The study site is located in the Sierra Nevada National Park (SE Spain). In September 2005, a wildfire burned ca. 1300 ha of reforested pine between 35 and 45 years age. The area selected for this study is located at 2320 m a.s.l. (36°58'3.68"N; 3°28'37.04"W). The climate is Mediterranean-type, with precipitation falling mostly during autumn and winter, and by a dry summer with a mean annual temperature of 7.8±0.7ºC (period 2008–10) and annual precipitation of 470±50mm (period 1988–2008; climatic data from a nearby meteorological station at 1500 m a.s.l). Snow falls during winter, usually persisting from November to March, and the growing season usually starts in the second half of May. The slope is between 15-20%. The dominant pine species present before the wildfire was *Pinus sylvestris* with a density of 1060±50 ha^{-1}, 13.4±0.3cm d.b.h. and 6.63±0.17m height. Burnt wood biomass was estimated at 46.9 Mg ha^{-1} (70% above and 30% belowground), according to alometric equations based on pine density and tree size (Castro et al., 2010a). This supposes a C stock in wood of 23.6 Mg ha^{-1} (C concentration was measured in the sawdust of 50 burnt logs with a Leco TruSpec autoanalyzer, St. Joseph, MI, USA). The fire was of high intensity and no trees survived inside the study area. Current vegetation is mainly composed by grass and forbs typical of disturbed areas in the Oromediterranean belt (Molero-Mesa *et al.*, 1996) the most common perennial species being *Genista versicolor*, *Festuca spp*. and *Sessamoides prostata*.

Nine months after the fire, two post-fire management treatments were applied to the burnt trees of two 35ha stands: (1) “Non-Intervention” (NI): all burnt trees were left standing and fell naturally and progressively over the years, with around 25% still standing at the beginning of this study; and (2) “Salvage Logging” (SL): trees were cut and the trunks cleaned of branches by chainsaw and piled manually in groups of 10-12, with woody debris chopped by machine and trunks removed from the site with a log forwarder. The two treatments were contiguous (Figure 1) and showed similar characteristics in terms of tree size and density, slope, bedrock (michaschists) and soil type (Humic cambisols).

2.2 *Meteorological and eddy covariance measurements*

An eddy covariance tower - with additional instrumentation for environmental and soil measurements - was installed in each treatment. Fluxes of CO_2, water vapour (latent heat) and sensible heat were estimated from fast-response (10 Hz) instruments mounted atop towers of 10 m (NI) and 2.5 m (SL). Densities of CO_2 and H_2O were measured by open-path infrared gas analysers (Li-7500, Lincoln, NE, USA) and calibrated periodically using an N_2 standard for zero and a 479.5 μmol (CO_2) mol^{-1} gas standard as a span for both treatments. Winds and sonic temperature were measured by three-axis sonic anemometers (for NI: Model 81000, R. M. Young, Traverse City, MI, USA; for SL: CSAT-3, Campbell Scientific, Logan, UT, USA). Published comparison analyses between fluxes measured using both anemometers have shown good agreement (Loescher et al., 2005; Tanny et al., 2010). Measurements were made in 2009 (the 4th year after the fire), year-round in NI, and from early June to late December in SL.

Air temperature and humidity were measured by thermohygrometers (HMP 45C, CSI, USA) at 7m (NI) and 2m (SL) above the surface. Soil water content (SWC) was measured by two water content reflectometers (CS616, CSI) at 4 cm depth for the NI treatment. Over a representative ground surface, incident and reflected photosynthetic photon flux densities were measured by quantum sensors (Li-190, Lincoln, NE, USA) for both treatments. In the NI treatment, a net radiometer (NR Lite, Kipp & Zonen, Delft, Netherlands) located 8 m above the surface and four heat flux plates (HFP01SC, Hukseflux, Delft, Netherlands) at 8 cm depth and two pairs of soil temperature probes (TCAV, Campbell Scientific, Logan, UT, USA) at 2 and 6 cm depth, were installed parallel to the surface to examine the energy balance (Wilson *et al.*, 2002). For both treatments, data loggers (CR3000, CSI) managed the measurements and recorded the data. Eddy covariance data were saved at 10 Hz by the logger. Means, variances and covariances on half-hour bases following Reynolds' rules, eddy flux corrections for density perturbations (Webb *et al.*, 1980) and coordinate rotation (McMillen, 1988) were applied, as well as quality control checks following Reverter *et al.* (2010) using an in-house program (PECADO) based on MATLAB routines.

2.3 *Data quality control, gap filling for long term integration of fluxes, and partitioning*

Half-hour statistics were computed when data eliminated by quality control did not exceed 25% of the total. Night-time data during periods with low turbulence (friction velocity, $u_* < 0.35$ m s^{-1} for the NI treatment; $u_* < 0.25$ m s^{-1} for the SL treatment) were rejected (Goulden *et al.*, 1996), as were three nights in February with unrealistic values. The Flux-Source Area footprint model (Schmid, 1994, 1997, 2002) was applied to verify that fluxes originated from well within the fetch (Figure 1). Even during periods of relative static stability (0.2 m s^{-1} < u_* < 0.4 m s^{-1}; sensible heat fluxes (H) < 0), the estimated maximum source location was 101 m for NI and 36 m for SL; the maximum distance of the 50% source area isopleths (Figure 1) was 228 m (NI) and 68 m (SL). In addition, the energy balance closure (ratio of the sum of sensible and latent turbulent fluxes, $H + LE$, to the difference between net radiation and the soil heat flux, $R_n - G$) was 90% ($R^2 = 0.67$; n = 755) for the NI treatment. This value is in the range reported by most FLUXNET sites (Wilson *et al.*, 2002) and provides additional information regarding turbulent flux quality (Moncrieff *et al.*, 1997).

Data rejected due to environmental conditions or instrument malfunction amounted to 29% and 23% of the total measured period for the NI and SL treatments respectively. In addition, night-time low turbulence conditions rejected 18% and 13% of the data, resulting in 47% total data missing for NI and 36% for SL, requiring gap filling in order to estimate the annual CO_2 and water vapour exchanges. Gaps were filled using the ‘‘Marginal Distribution Sampling’’ (MDS) technique (Falge *et al.*, 2001; Reichstein *et al.*, 2005), replacing missing values using a time window of several adjacent days. The length of the time window depends on environmental conditions and meteorological data availability. In a parallel way and only for CO_2 fluxes, a semi-empirical gap-filling method based on the response to temperature and photosynthetic photon flux density for respiration and photosynthesis respectively (Falge *et al.*, 2001; Lasslop *et al.*, 2010) was also applied. Results from this alternative gap-filling method are mentioned only when significant differences with the MDS method were detected (P<0.05). Random uncertainty and errors in net ecosystem carbon and water vapour exchanges introduced by the gap-filling processes were calculated using Monte Carlo simulations (Richardson and Hollinger, 2007); see Reverter *et al.*, (2010) for more information. Positive values of net ecosystem carbon denote a net CO_2 release to the atmosphere while negative values denote a net CO_2 uptake.

Half hourly net CO_2 fluxes were broken into gross primary production (GPP) and ecosystem respiration (R_{eco}) components using two different techniques: the "night-time data-based estimate" (NB; (Reichstein *et al.*, 2005)) and the "daytime data-based estimate" (DB; (Lasslop *et al.*, 2010)) flux partitioning algorithms. The NB algorithm assumes that GPP is zero at night and models R_{eco} as a function of temperature using night-time data; this relationship is extrapolated to daytime, for which the difference between the modelled R_{eco} and measured CO_2 fluxes yields the estimated GPP (see Reichstein *et al.*, 2005 for more information). For the DB algorithm, the daytime measured CO_2 fluxes are modelled using a hyperbolic light–response curve (Falge *et al.*, 2001) for GPP and a respiration model depending on temperature for R_{eco} (Equation 1); where F_C is the measured CO_2 flux, α (µmol C J^{-1}) the canopy light utilization efficiency representing the initial slope of the light–response curve, β (µmol C m^{-2} s^{-1};) the maximum CO_2 uptake rate of the canopy at light saturation adjusted for vapour pressure deficit limitations, R_g the global radiation (W m^{-2}) that can be easily estimated using the measured photosynthetic photon flux density (Ceulemans *et al.*, 2003), R_{15} (µmol C m^{-2} s^{-1}) the base respiration at 15ºC, E_0 (ºC) the temperature sensitivity and T_a (ºC) the air temperature (see Lasslop *et al.*, 2010 for more details).

$$F_C = \frac{\alpha\beta R_g}{\alpha R_g + \beta} + R_{15}\exp\left(E_0\left|\frac{1}{15-46.02} - \frac{1}{T_a + 46.02}\right|\right) \quad (1)$$

To track the respiratory and photosynthetic capacity of both treatments, mean monthly values of R_{15} and α estimated every two days from the DB partitioning algorithm were selected.

2.4 P*lant cover and soil respiration and moisture measurements*

In order to determine possible causes of the differences in measured F_C between treatments, plant cover and soil CO_2 fluxes and humidity were measured. Plant cover was sampled with a point-linear method one and two years after the fire (June 2006 and 2007, respectively) as a surrogate for regenerative capacity and primary production. In June 2006, measurements were done in 12 randomly established linear transects of 25x2 m along the maximum slope of the terrain for each treatment. The number of individuals of perennial plants was counted within each transect. For June 2007 the methodology was changed due to the high plant cover that impeded the monitoring of all individuals. In that case, three points (central and transversal sides) at each 50 cm along the transect (n=150 points per transect) were sampled, observing the nature of contact (soil or vegetation). Plant height (if present) was measured at every central point of the transect. Differences between treatments were analyzed with one-way ANOVAs for each year.

Soil respiration and water content were measured six times throughout the spring of 2009 at three-week intervals from March to June. Twenty PVC collars per treatment were installed in the soil to ca. 5 cm depth, randomly distributed over an area of ca. 1 ha and separated by at least 10 m. Soil respiration measurements were performed on the collars from ca. 9am to 3pm using two CO_2 analyzer systems: the manual EGM-4/SRC-1 (PP-Systems, Hitchin, UK); and an automated LI-8100 (Lincoln, NE, USA). The two instruments were used in both treatments. A previous instrument intercomparison (Marañón-Jiménez *et al.*, 2011) allowed correction of the EGM-4/SRC-1 data to match the LI-8100. During these campaigns, soil water content was also measured at 10, 20, 30 and 40 cm depth at 15 points per treatment, using the PR-2 profile probe (Delta T, Services, Cambridge, UK). Soil CO_2 effluxes and their variation over sampling dates (time) were analyzed with a repeated-measure analysis of variance (rmANOVA), with sampling dates defined as the within-factor and treatment as the between-factor. Soil water content was similarly analyzed with rmANOVA. Throughout the paper mean values are followed by ±1SE.

3. Results

3.1 *Meteorological conditions*

Meteorological conditions showed a strongly asynchronous pattern of rainfall and temperature throughout the year (Figure 2). During summer (June, July, August), the mean daily air temperature (T_a) was 17.1ºC, while precipitation was almost negligible with only one rain event exceeding 5 mm. In winter (January, February, and December) mean daily T_a was 1ºC and the greatest precipitation fell, mostly as snow which persisted from December to March. During spring and fall, rain and T_a showed intermediate values compared with the other two seasons, with a mean daily T_a of 7.8ºC, and accumulated rainfall of 170 mm. Annual values of mean T_a and total rainfall in 2009 were 8.4ºC and 678 mm respectively.

3.2 *Monthly net carbon exchange and evapotranspiration*

Overall, the Non Intervention (NI) treatment acted as a monthly net carbon sink during nearly the whole year 2009, whereas the Salvage Logging (SL) treatment acted consistently as a source following the June installation of the eddy system (Figure 3). The most productive period for NI was the end of spring and beginning of summer, reaching the maximum value of carbon uptake in May (around 30 g C m^{-2}). Then, from August to October, NI emitted *ca* 2 g C m^{-2} per month. In November (end of autumn, with fair weather) the ecosystem absorbed more than 10 g C m^{-2}. During winter, NI was very nearly carbon neutral. However, December and January are interpreted as carbon source months if gaps were filled using the semi-empirical approach, emitting 13 and 9 g C m^{-2} respectively. By contrast, SL consistently emitted carbon, with maximum emissions in July (more than 20 g C m^{-2}) and decreasing from then until the year's end. The semi-empirical approach could not be applied in SL due to the inability to correlate measured CO_2 fluxes with temperature or light (Lasslop *et al.*, 2010)

During the measured period in both treatments, NI presented usually higher monthly evapotranspiration values (*ET*), with the exception of December (Figure 3). Monthly *ET* for NI reached maximum values at the end of spring (May and June; *ca* 60 mm) and minima at the beginning and end of the year (<25 mm). In early autumn (October), *ET* was similar to that of early spring (*ca* 40 mm). In SL, during the measured period (June-December), maximum *ET* values where reached in October when the soil was moist and the temperature mild (Figures 2 and 3). Nonetheless, monthly *ET* remained very low and stable over the measured period and never exceeded 40 mm. The monthly Bowen ratio (ratio of sensible to latent heat flux) increased from February to August for NI treatment and decreased afterwards (Table 1). For SL treatment the monthly Bowen ratio was higher than NI. Both treatments presented higher values in July and August (Table 1).

3.3 *Diurnal trends of CO_2 fluxes across treatment*

Diurnal trends of CO_2 fluxes were explored in three representative months for simplicity (Figure 4). In general, during daytime NI acted as a consistent net CO_2 sink while SL acted as a source. During night-time both treatments acted as sources of CO_2. However, while SL presented values lower than 0.6 µmol m^{-2} s^{-1}, NI reached values exceeding 1 µmol m^{-2} s^{-1} in June. Concretely, in June, daytime CO_2 uptake in NI was often near 3 µmol m^{-2} s^{-1} while SL acted as daytime CO_2 source (*ca* 1.5 µmol m^{-2} s^{-1}). In July, SL presented similar behaviour to June whereas NI reduced its CO_2 assimilation by more than a half. In November, early daytime CO_2 uptake was measured in SL (F_C=-0.5 µmol m^{-2} s^{-1}). For the NI treatment, autumn values of daytime CO_2 uptake reached 2 µmol m^{-2} s^{-1} and nighttime CO_2 release was considerably lower than June and July.

3.4 *Accumulated carbon exchange*

Figure 5 shows the accumulated carbon exchange estimated for NI and SL over the period when simultaneous measurements in both treatments are available (June-December of 2009). For the SL treatment, the accumulated carbon exchange showed a near constant slope (*a*) [*a*=0.6; R^2=0.995] from the start of the measurements (June) until October. During this period, this treatment acted as a daily constant carbon source, emitting between 80 and 110 g C m^{-2} and thereafter, it acted as near neutral C sink until the end of the year. The NI treatment acted as a net carbon sink during spring, absorbing 60 g C m^{-2} (from April to June; data not shown). After this productive period, the net carbon uptake capacity was reduced and the ecosystem absorbed 30 g C m^{-2} from June to July. From this point, the NI treatment behaved as near neutral C sink until the middle of November, when this treatment recovered its sink activity until the end of the year.

Over the course of 2009, the NI treatment absorbed 77±11 g C m^{-2}. Such a confident value cannot be given for SL treatment, due to the absence of carbon exchange measurements from January to May 2009. However, a crude annual estimation can be given assuming a range of possible behaviours of SL during the non measured period. During winter, we can consider similar behaviour for SL and NI, acting as a neutral net carbon sink due to the existence of snow cover (Harding *et al.*, 2001). For April and May, the accumulated carbon exchange could be considered as delimited by two extreme situations: (1) a neutral net carbon exchange, given the lack of net carbon assimilation throughout the measurement period (Figure 3) and (2) a scenario of maximum carbon emission. For the estimations under this assumption we used the "daytime data-based estimate" (DB) respiration model (Lasslop *et al.*, 2010). The model was applied using maximum values of base respiration at 15ºC (R_{15}) and temperature sensitivity (E_0) estimated during the measured period (1.25 µmol $m^{-2}s^{-1}$ and 335ºC respectively). Thus, in any case considered under these preliminary assumptions, the SL treatment would act as a net annual carbon source, emitting between 90 and 120 g C m^{-2} in 2009.

3.5 P*lant cover and soil respiration and moisture measurements*

Plant cover in June 2006 was higher in NI (11.1±1.6 individuals m^{-2}) than in SL (7.5±1.1 individuals m^{-2}; F=3.24, d.f.=1, 22; P=0.086). Plant cover similarly differed between treatments in June 2007 (F=18.17, d.f.=1, 22; P<0.001), being higher in NI (61.2±1.7%) than in SL (46±4%; see also Figure 6 for photos of the study areas in 2009). Plant height also differed between treatments (F=4.69; d.f.=1, 453; P=0.031; log-transformed data), being likewise higher in NI (22.9±1.4 cm), than in SL (19.5±1.4 cm).

Soil respiration was consistently higher in NI than SL (Figure 7a, Table 2). Soil water content decreased throughout the growing season and was constantly higher in SL (Figure 7b; Table 3).

3.6 *Photosynthesis and respiration partitioning*

Mean estimated values of base respiration at 15ºC (R_{15}) and canopy light utilization efficiency (α;) from the DB partitioning algorithm (Figure 8) were used to track the respiratory and photosynthetic capacities of NI. Monthly trends of R_{15} and α were very similar, showing peaks at the end of spring (May) and in the fall, with lower values during the dry summer. However, R_{15} lagged α by about one month in reaching its fall maximum (in October, R_{15}=0.86 µmol m^{-2} s^{-1}) by which time α had dropped back to low values (*ca.* 0.008 µmol C J^{-1}). Relatively high values of R_{15} were also estimated in

December, but were accompanied by only a slight increase in α. For SL, no dependence of GPP on light, nor of R_{eco} on temperature, was detected and thus, the DB partitioning algorithm could not been applied, except from mid-October to December, where early daytime CO_2 uptake was measured in SL (see November 2009 in Figure 4) and R_{15} reached values near 0.54 μmol m^{-2} s^{-1}. The estimated α was generally null, except for 22-24 October (0.0012 μmol C m^{-2} s^{-1}) and 5-7 November (0.0853 μmol C m^{-2} s^{-1}).

Thus, due to the lack of measured CO_2 fluxes dependencies on light or temperature for SL, estimated values of gross primary production and ecosystem respiration are given only for NI. Using both algorithms, higher values of GPP were obtained in May and June, while lower values corresponded to cold winter months (January-March; Figure 9). During end of summer, fall and beginning of winter the estimated GPP remained nearly constant according both algorithms. By contrast, modelled R_{eco} showed significant differences depending on the algorithm used. For "DB" algorithm, R_{eco} presented higher estimated values during the end of summer and early fall, and maximum in September (only for DB algorithm). A peak in R_{eco} was also estimated in May. The beginning and end of the year (January and December) also presented high values similar to June and October respectively. Using the NB algorithm higher values of R_{eco} were estimated in May and June, and minimum values during winter.

4. Discussion

During the fourth year after a fire, SL management hindered the recovery of C sequestration in the Mediterranean coniferous forest compared to the NI treatment. Photosynthesis and respiration processes also presented different patterns between post-fire treatments. Carbon loss was mostly constant in SL and not related to temperature at short time scales (30 min), with very small oscillations throughout the whole measurement period at both daily and seasonal scales, evidencing very low biological activity in the soil and vegetation. By contrast, the NI treatment showed more biological activity, with higher soil respiration rates and vegetation productivity, yielding higher daily and seasonal ranges of carbon exchange. In fact, the results of this study underline higher vegetation cover and performance for the NI treatment, explaining the higher *ET* and lower Bowen ratio compared to the SL treatment, with a consequent decrease in soil water content. Furthermore, while opposing processes in the carbon cycle (plant uptake and respiration) were both enhanced in NI, the additional contributions of CO_2 released by the wood decomposition (Gough *et al.*, 2004) was overwhelmed by photosynthesis such that annual carbon emissions were reduced considerably compared to the SL treatment. Thus, despite the limited temporal extent of data coverage, the strong impact of SL management on ecosystem CO_2 fluxes has been clearly demonstrated even at the initial stages of natural regeneration.

Several reasons may contribute to the marked differences in the net CO_2 fluxes between SL and NI treatments. First, burnt trees and coarse woody debris left after the wildfire represent a large pool of nutrients (Wei *et al.*, 1997; Johnson *et al.*, 2005; Kappes *et al.*, 2007; Merino *et al.*, 2007), that will be progressively incorporated to the soil as the trees fall and wood decomposes (Harmon *et al.*, 1986; Grove, 2003; Coleman *et al.*, 2004), improving soil fertility. Second, burnt trees and branches (even after falling) act as nurse structures that improve microclimatic conditions for plant regeneration (Harmon *et al.*, 1986; Lindenmayer *et al.*, 2008; Smaill *et al.*, 2008; Stoddard *et al.*, 2008; Castro *et al.*, 2010a; Castro *et al.*, 2010b). Third, salvage logging may damage the banks of seedlings and shoots that regenerate soon after the fire (Martinez-Sanchez *et al.*, 1999; McIver and Starr, 2000; Lindenmayer *et al.*, 2008), reducing plant density. In addition, the presence of burnt logs and branches creates habitat complexity that may reduce herbivore damage to the vegetation (Ripple and Larsen, 2001; see also Relva *et al.*, (2009) for similar effect in non-burnt woody debris), and soil erosion (Wondzell, 2001; Robichaud, 2005; Kim *et al*., 2008; Lindenmayer *et al.*, 2008; Robichaud *et al.*, 2008;), and attract seed-dispersing birds (Rost *et al.*, 2009; Castro *et al.*, 2010b; Rost *et al.*, 2010). All this may translate to a higher capacity in NI for vegetation and hence carbon sequestration, while SL retards vegetation recovery and carbon uptake capacity. Differences could be more accentuated in the long term, as wood decomposes and progressively releases its nutrients (Irvine *et al.*, 2007).

These results are likely extensible to many other burnt coniferous forest ecosystems subjected to post-fire salvage logging. Coarse woody debris has been widely reported to contribute to soil fertility and soil microclimate improvement in different ecosystem types (Pérez-Batallón *et al.*, 1998; Hafner and Groffman, 2005; Smaill *et al.*, 2008;

Stoddard *et al.*, 2008; Castro *et al.*, 2011), and consequently to enhance primary productivity (Burton *et al.*, 2000; Stark *et al.*, 2006; Irvine *et al.*, 2007; Stoddard *et al.*, 2008). Since partially burnt woody debris (with charring limited to the bark and the superficial layers) has similar nutrient concentrations to unburnt wood (Wei *et al.*, 1997), the effects of burnt wood on soil fertility enrichment will be comparable to those reported for unburned coarse woody debris. In addition, reductions in plant cover and regeneration capacity after salvage logging have been also reported in different forest types across the world (Lindenmayer *et al.*, 2004; Donato *et al.*, 2006; Greene *et al.*, 2006; Lindenmayer and Noss, 2006; Stark *et al.*, 2006; Beghin *et al.*, 2010; Castro *et al.*, 2010a; Svoboda *et al.*, 2010), thus with the potential to reduce carbon uptake. Finally, the general increase of erosion risk after a wildfire (Thomas *et al.*, 1999; Yang *et al.*, 2003; Spanos *et al.*, 2005; Lindenmayer *et al.*, 2008) leads to a negative synergic effect through soil impoverishment, reinforcing the impact of salvage logging on carbon emissions. Thus, in general salvage logging applied after a wildfire in coniferous forests has the potential to alter soil properties, retarding vegetation recovery and thus the carbon uptake capacity.

4.1 Management implications

Fires destroy large areas of forest every year in many areas of the world (FAO, 2007). A key management decision after a forest fire is to determine the fate of the burnt wood, and an intense debate surrounds the practice of salvage logging as it has ecological, economical and silvicultural implications (Beschta *et al.*, 2004; DellaSala *et al.*, 2006; Donato *et al.*, 2006; Lindenmayer *et al.*, 2008). Our study demonstrates, for the first time, that the removal of burnt wood retards the capacity of such ecosystem to restore its carbon sink capacity in Mediterranean climates. Thus, in terms of policies for optimization of carbon sequestration in the context of the climate change, salvage logging should be discouraged. Potential implications at the global scale are aggravated by the predicted increase in wildfire incidence for climate change scenarios in Mediterranean and other semi-arid climates of the world (IPCC, 2007). Applying alternative management strategies for burnt wood following wildfire could therefore suppose a notable variation in carbon release to the atmosphere at a global scale, even without considering CO_2 emissions by the heavy machinery used in salvage logging operations (Stephens *et al.*, 2009).

5. Acknowledgements

Thanks to Francisco Domingo, Alfredo Durán and Ángela Sánchez–Miranda Moreno for their help in the field. We are especially grateful to the Environmental Council (Junta de Andalucía) and to the Natural and National Reserves of Sierra Nevada for support in establishing the treatments. This work was financed by INIA Project SUM2006-00010-00-00, by the Autonomous Organism of National Parks (MMA) Project 10/2005 and in part by the Spanish national CO_2 flux tower network (Carbored-II; CGL2010-22193-C04-02), Consolider-Ingenio MONTES CSD2008-00040 and the European Community 7th 9 Framework Programme project GHG-Europe (FP7/2007-2013; grant agreement 244122).

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Figure captions

Figure 1: Eddy tower locations as bull's eyes in Non Intervention (white) and Salvage Logged (black) treatments. For each tower, according to the Flux-Source Area model of Schmid (1994) during periods of relative static stability (periods where measured fluxes are generated most distant from the eddy tower) defined in terms of the friction velocity (0.2 m s^{-1} < u_* < 0.4 m s^{-1}) and sensible heat flux ($H < 0$), the maximum source location is denoted by the circle of grey dots. Similarly, the white circles denote the near- and far-limits of the 50% source area isopleths. According to Schmid (1997) a flux source point located on or outside the 50% source area boundaries would have to be 5 to 10 times stronger than the point of maximum source weight, in order to achieve a similar response on the eddy covariance sensors. Consequently, maximum sources and near- and far-limits for other atmospheric conditions are inside the respective circles. Frequency (%) of each wind direction, over the measured period, is represented by number inside each octant.

Figure 2: Mean daily values of air temperature (T_a; ºC; grey dots), soil water content (SWC; % vol.; black line) and total precipitation (Rain; mm; black bars) during 2009. Shaded bars denote periods of snow cover [ratio of mean daytime reflected photon flux density to mean daytime incident photon flux density higher than 0.2]

Figure 3: Monthly totals of exchanged carbon (g C m^{-2}) and evapotranspiration (mm) by forest treatments during 2009 for Non Intervention (NI, grey bars) and since June 2009 for Salvage Logged (SL, dark bars) treatments, using MDS gap-filling technique. Ecosystem CO_2 uptake is depicted as negative while ecosystem CO_2 release is positive.

Figure 4: Diurnal trends in CO_2 flux (F_c, μmol m^{-2} s^{-1}) for the monthly means (+/- standard error) of June, July and November 2009 for treatments treated by (A) Non Intervention and (B) Salvage Logging.

Figure 5: Cumulative carbon exchange (g C m^{-2}) from June-December 2009 by forest treated with Non Intervention (NI, grey line) and Salvage Logging (SL, black line).

Figure 6: Appearance of forest treatments treated by Non Intervention (NI) and Salvage Logging (SL) for the 27th and 28th of May 2009 respectively.

Figure 7: Mean values (±SE) of (A) soil CO_2 effluxes of 20 PVC collars and (B) Soil Water Content from 10 to 40cm depth for NI and SL, for six campaigns in spring 2009.

Figure 8: Mean monthly values (±SE) of respiratory and photosynthetic parameters used to estimate both processes in the treatment treated with Non Intervention (NI). Estimated values outside the range defined as "mean monthly value ±SD" were rejected.

Figure 9: Estimated monthly gross primary production (GPP; negative exchanges) and ecosystem respiration (R_{eco}; positive exchanges) using the "daytime data-based estimate" (DB; lined bars) and the "night-time data-bases estimate" (NB; white bars) flux partitioning algorithms for the Non Intervention treated treatment.

Figure 1
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Figure 2

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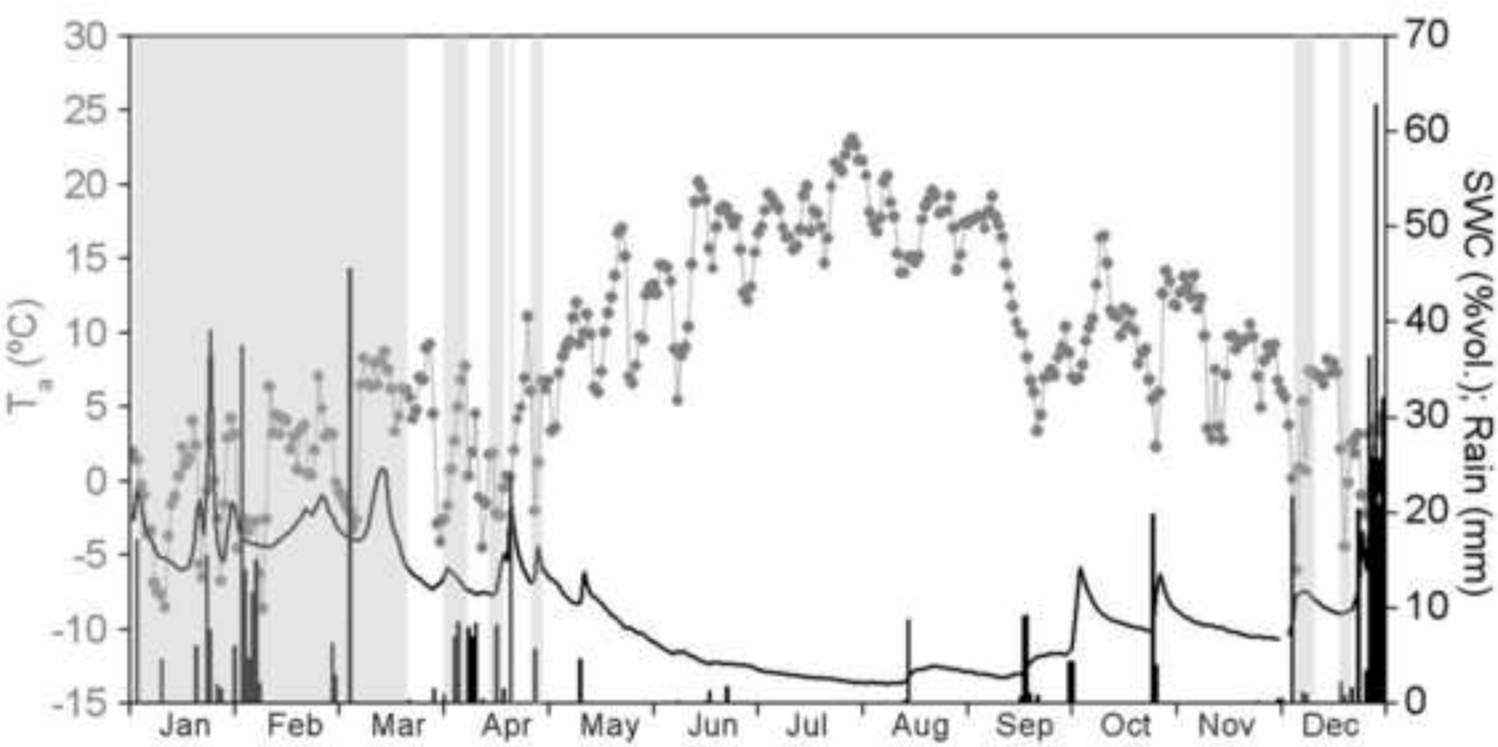

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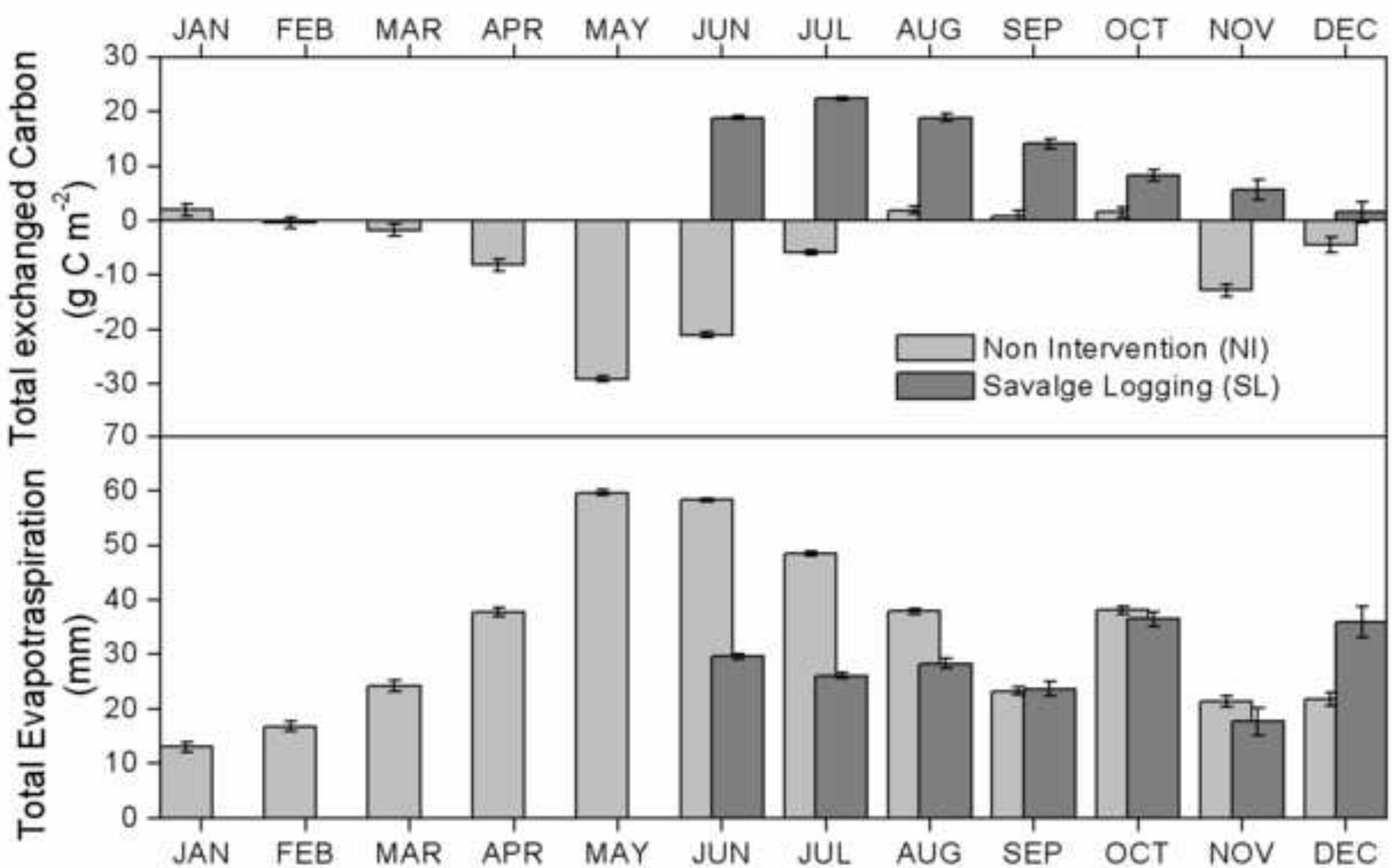

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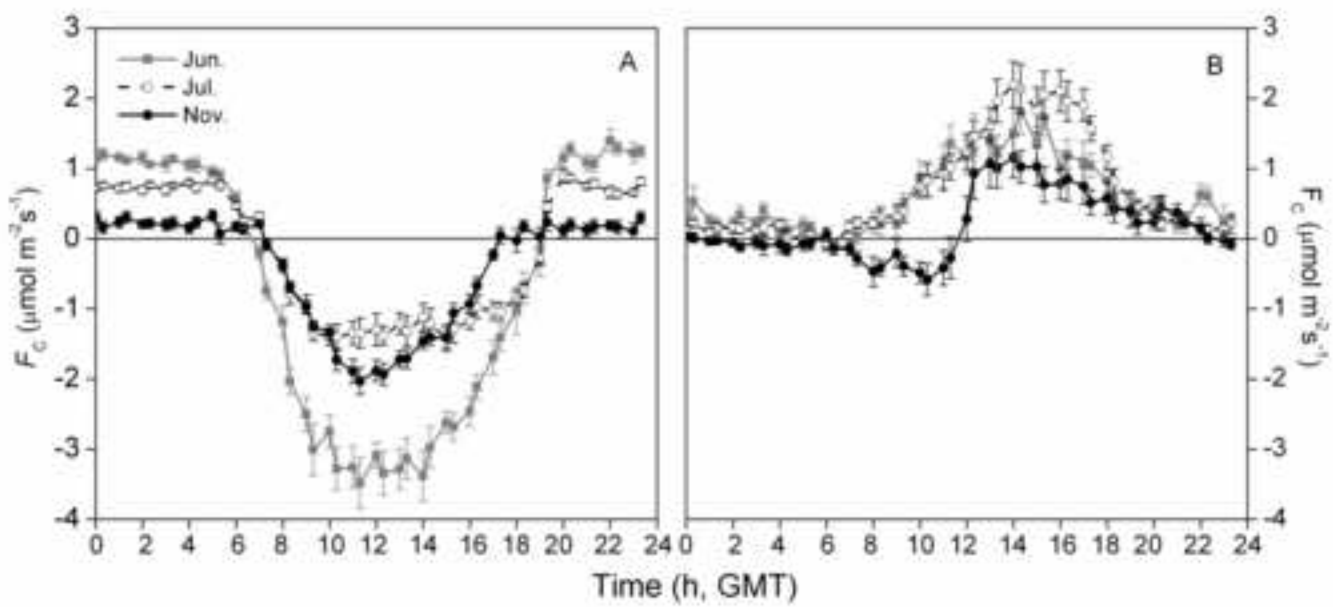

Figure 5 rev
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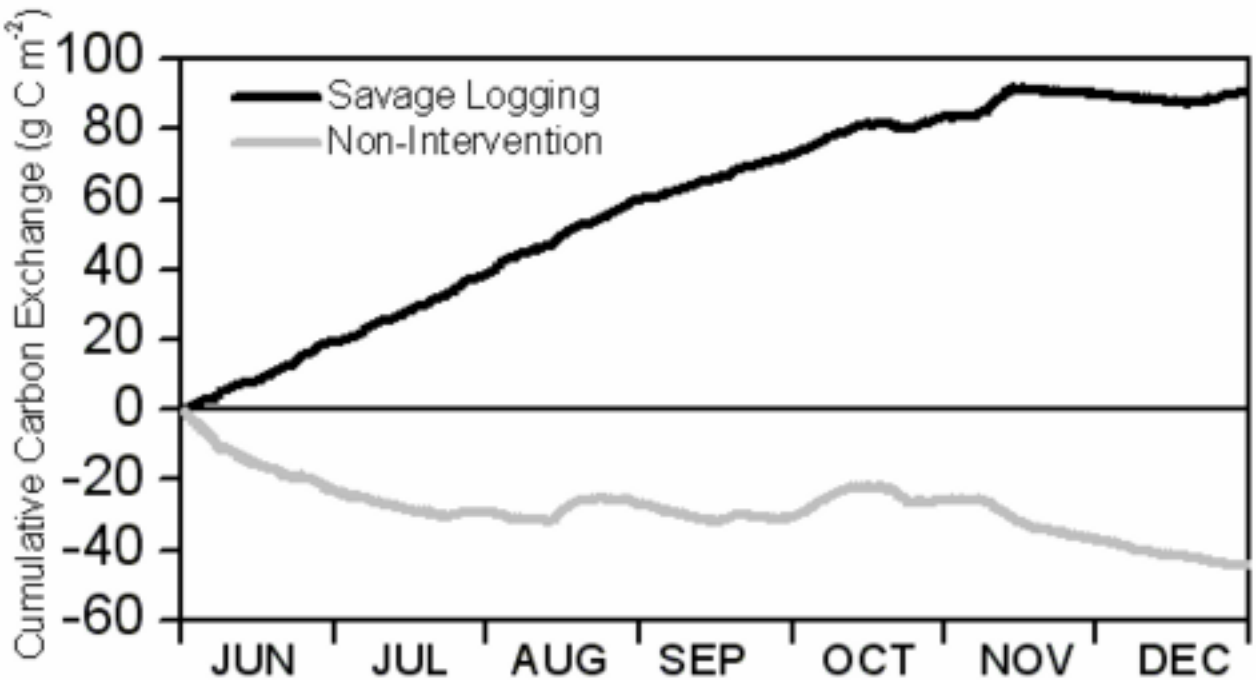

Figure 6
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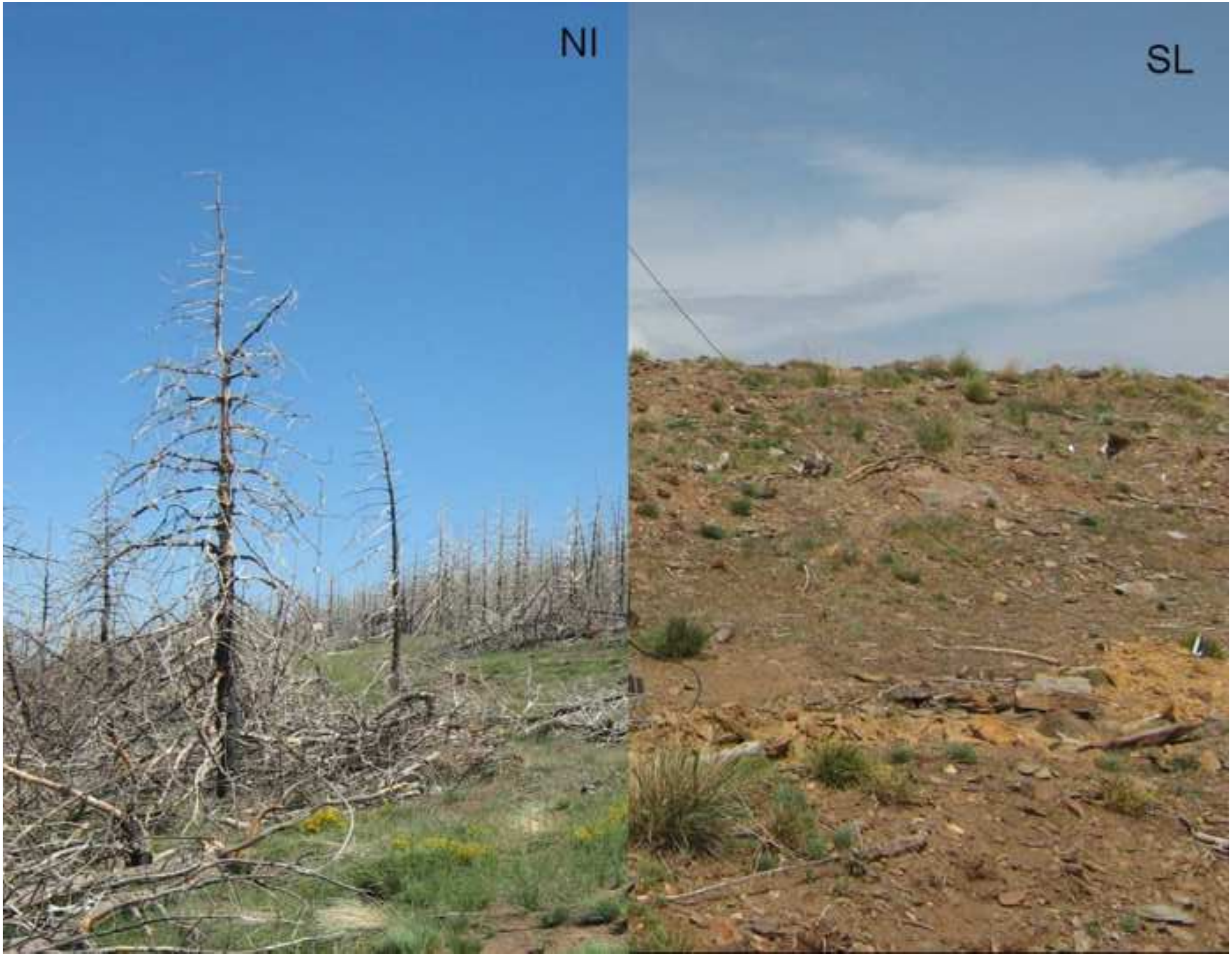

Figure 7

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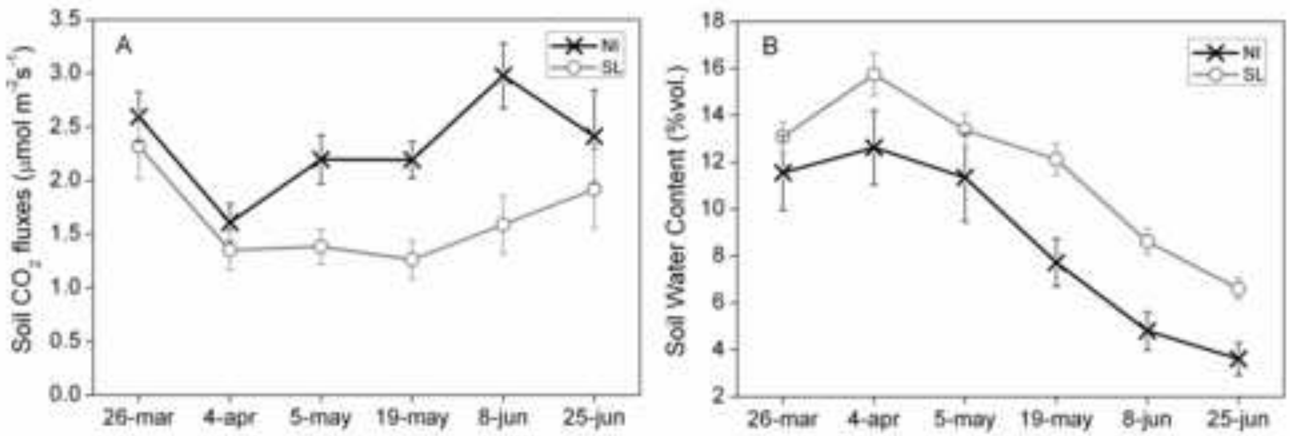

Figure 8
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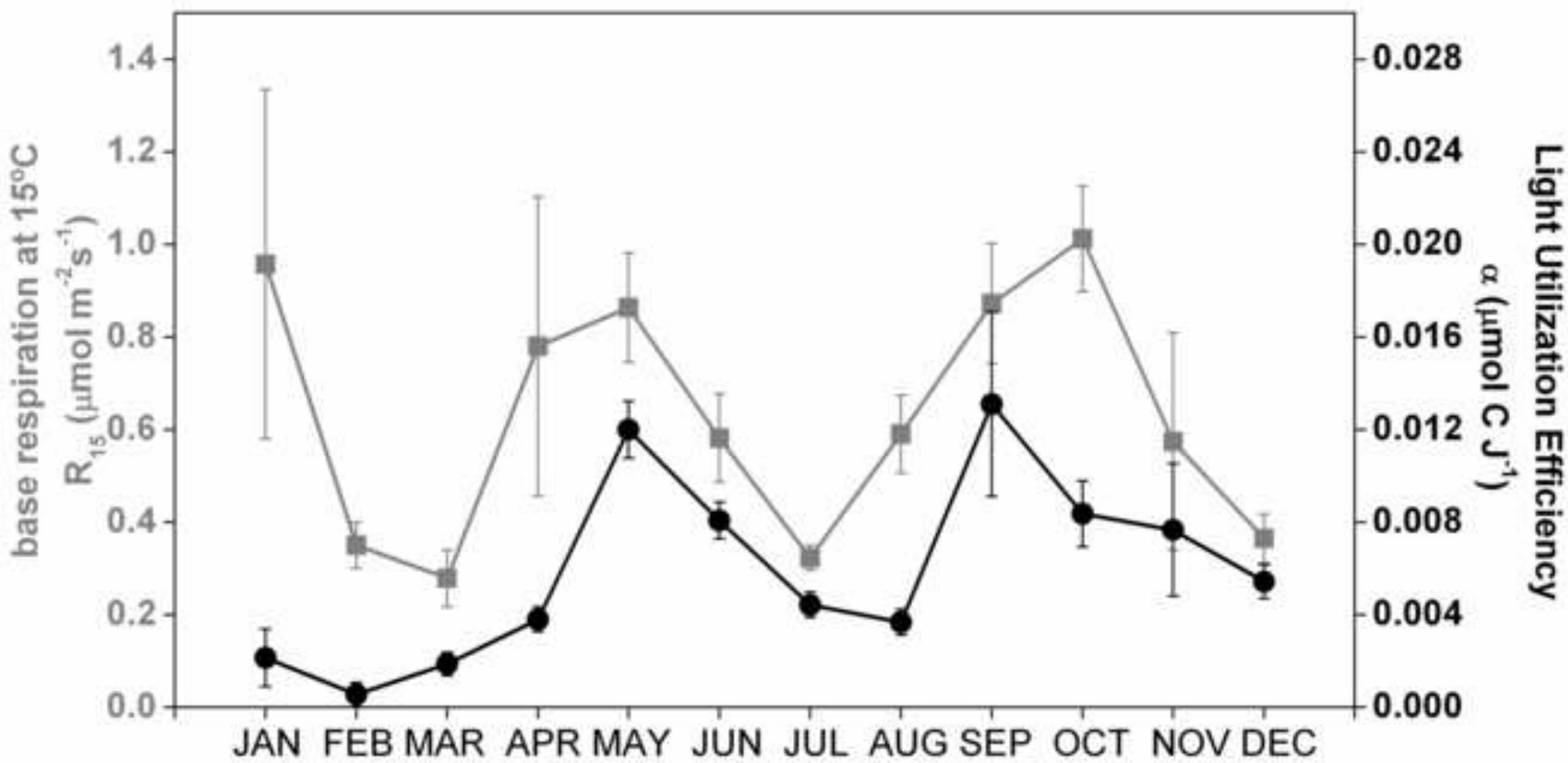

Figure 9 rev
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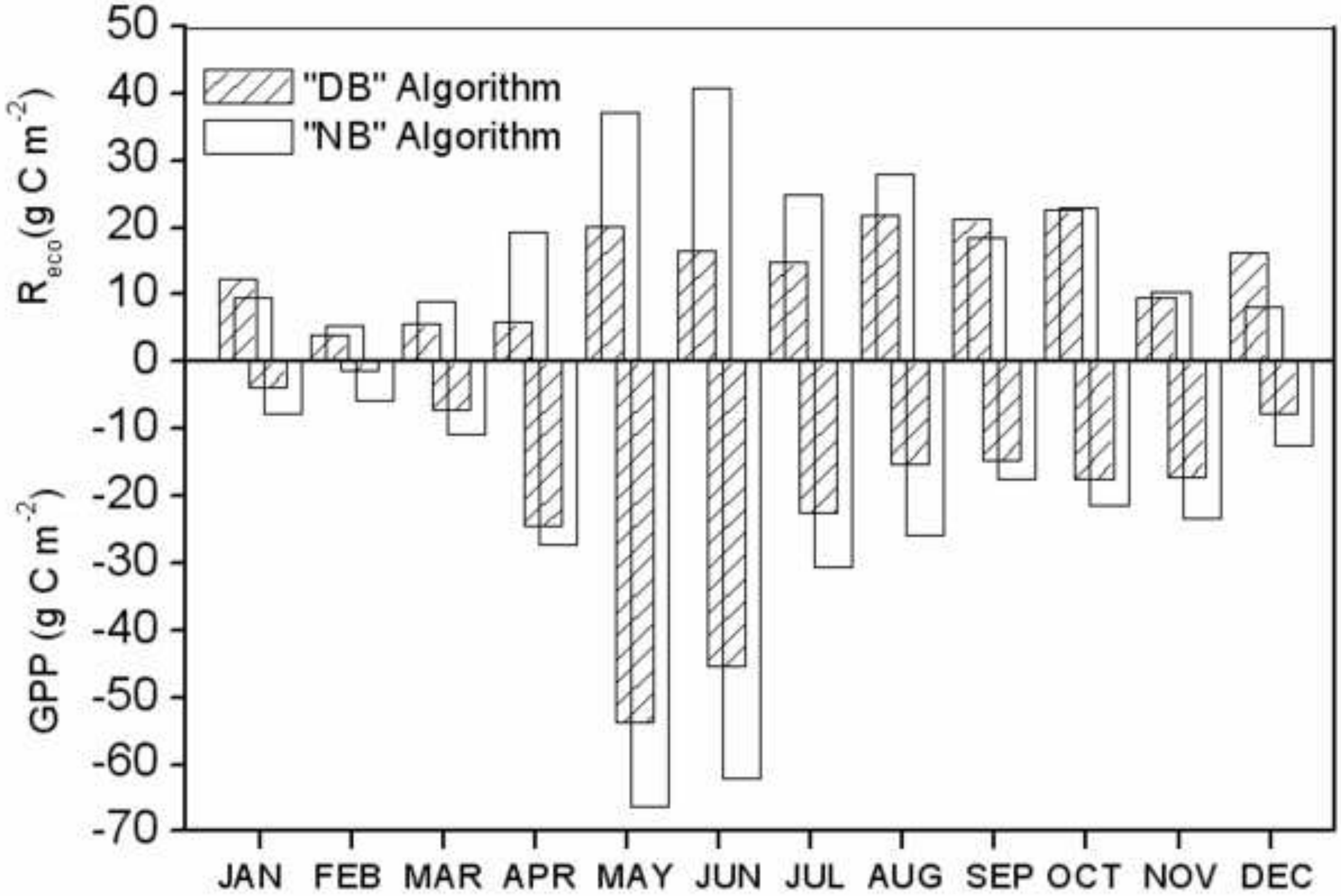

Table 1

Bowen Ratio												
	Jan.	Feb.	Mar.	Apr.	May.	Jun.	Jul.	Aug.	Sep.	Oct.	Nov.	Dec.
NI	-	0.4(0.2)	1.4(0.5)	2.2(0.4)	1.9(0.2)	2.1(0.2)	3.6(0.3)	3.1(0.3)	-	1.6(0.2)	1.2(0.3)	-
SL	-	-	-	-	-	3.4(0.5)	4.2(0.5)	3.5(0.6)	1.8(0.4)	1.8(0.3)	2.0(1.1)	-

Monthly values of the Bowen ratio for NI and SL treatments along 2009. The error (in parentheses) is calculated based on the standard errors of H and LE.

Table 2

Source	df	F	P
Between-subject			
Treatment	1, 28	7.34	0.0114
Within -subject			
Time	5, 24	4.76	0.0037
Time*Treatment	5, 24	2.07	0.1048
Error	28		

Summary of Repeated Measures Analysis of Variance (rmANOVA) for soil CO_2 fluxes measured throughout the spring 2009. df= degrees of freedom of the numerator and denominator respectively. F= Value of the F statistic. P=Critical probability of the analysis.

Table 3.

Source	df	F	P
Between-subject			
Treatment	13.61	1, 54	0.0005
Depth	0.41	3, 54	0.7464
Treatment*Depth	0.18	3, 54	0.9101
Within -subject			
Time	5, 50	156.61	<0.0001
Time*Treatment	5, 50	9.23	<0.0001
Time*Depth	15, 138.43	3.94	<0.0001
Time*Treatment*Depth	15, 138.43	1.10	0.3572
Error	54		

Summary of Repeated Measures Analysis of Variance (rmANOVA) for the soil water content measured throughout the spring of 2009. df= degrees of freedom of the numerator and denominator respectively. F= Value of the F statistic. Approximate value of F adjusted for the Time*Depth and Time*Treatment*Depth interactions (Wilk`s-Lambda multivariate test). P=Critical probability of the analysis.