



Effects of afforestation on soil respiration in an arid shrubland in Chile

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ABSTRACT

We studied the effects of afforestation on soil respiration (R_s) in an arid shrubland in northern Chile. We sampled R_s monthly for three years in two adjacent plots, one left as control in natural condition, and one afforested with *Acacia saligna*. Sampling occurred 3–6 years after afforestation. Soil temperature (T_s) was a good predictor of R_s on certain periods of the year, although we could not relate this to specific levels of soil moisture (θ). Seasonal R_s variation related more to θ than to T_s , which could be explained by extremely low annual precipitation (138 mm), concentrated during winter months. The correlation between T_s and R_s was higher when measured under plants compared to bare ground. The afforested site had in average 17% lower annual R_s ($p = 0.06$) than the control site. During some months of the third year of measurements, the afforested site showed greater R_s than the control, suggesting that in the near future, R_s might be higher for the afforested site than for the natural condition, depending on the effect that trees may have on soil carbon stability, temperature and hydrological conditions.

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1. Introduction

Total carbon dioxide (CO_2) global emission from soils is recognized as one of the largest fluxes in the global carbon (C) cycle (Li et al., 2008; Schlesinger and Andrews, 2000), estimated to be 50–78 Pg C yr⁻¹ and accounting for about 25% of global CO_2 exchange (IPCC, 2001). Soils contain the largest C reservoir in the biosphere (~1500 Pg C), about twice the atmospheric CO_2 -C pool (IPCC, 2001; Jia et al., 2006). Faster oxidation of soil organic matter due to global warming can therefore significantly increase atmospheric CO_2 concentration (Raich and Potter, 1995). Hence, soil respiration (R_s) is critical to global C cycling and climate change (Jia et al., 2006).

Predicting R_s and understanding the drivers that underlie its seasonal and spatial variation is fundamental to predict ecosystem responses to climate change (Li et al., 2008; Raich and Schlesinger, 1992). R_s has been reported to differ across temporal and spatial

scales (e.g. Li et al., 2008; Jia et al., 2006) as a result of changes in soil temperature (Lloyd and Taylor, 1994), soil moisture (Gaumont-Guay et al., 2006), vegetation (Buchmann, 2000), topography (Kang et al., 2003), soil texture (Dilustro et al., 2005) and C uptake, which is tightly linked to C supply for respiratory activities (Bahn et al., 2008), among others. Accurate estimates of R_s require understanding its biotic and abiotic controls. Similarly, it is important to measure R_s across all biomes for extended time periods and at large spatial scales.

Arid and semi-arid environments cover approximately one-third of the world's land area and are among the world's most fragile ecosystems, a condition that is becoming accentuated due to periodic droughts and mounting overexploitation of scarce resources (Malagnoux et al., 2007). Global warming is predicted to affect arid lands through temperature increase and rainfall decrease all over the world with only a few exceptions (Dai, 2010; IPCC, 2007), leading to irremediable natural vegetation loss and a decrease of agriculture, rangeland and forest productivity, accompanied by loss of biodiversity, soil organic matter and soil fertility (Malagnoux et al., 2007). Arid and semiarid regions account for 41% of Chile's continental territory, covering around 31 million ha (Benites et al., 1994). Within this area, natural shrublands of the Coquimbo Region cover 25% of its area (INE, 1998) and are largely degraded by over-grazing due to intensive goat husbandry and

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firewood extraction. Whether these shrublands act as a C source or sink for atmospheric CO₂ depends largely on land use, grazing intensity and climate (change), among other factors (Frank, 2002).

Afforestation in arid and semi-arid lands may play a key role in land stabilization, desertification control, watershed protection and other ecosystem services such as fuelwood production and fodder for domestic animals (Malagnoux et al., 2007). *Acacia saligna* (Labill.) H. Wendl. is a small tree that fixes nitrogen (N), native to the southern coast of Western Australia. Even though it commonly grows on alkaline and infertile sandy soils, it shows a better development in deep sands and loams along the water courses throughout its distribution (Maslin, 1974). Since the 1960s, *A. saligna* has been planted in Chile, mainly in the Coquimbo region (currently covers ~ 16,000 ha), where the species has shown to be well adapted, providing fuelwood and fodder (INFOR, 2009; Meneses and Flores, 1999).

This study was set out to compare monthly measures of R_s in a native arid shrubland with and without planting of *Acacia saligna*. In a previous study on this same site, Perez-Quezada et al. (2011) found that total ecosystem C was 36% lower in the afforested shrubland site, compared to the natural condition, two years after plantation. In both conditions, the soil represented the largest C pool, with 19.8 and 32.5 Mg C ha⁻¹ in the afforested and natural sites, which were 94% and 84% of total C, respectively. The fact that soil C represented a higher percentage of total C in the afforested site suggests that C loss in the afforested site was mainly due to the loss of woody biomass during soil preparation.

The addition of a fast-growing N-fixing legume should significantly increase the ecosystem C fixation and R_s over the base level in the long term, as a result of an increase in (i) transpiration:evapotranspiration ratio, (ii) turnover of soil organic matter, and (iii) N availability from N-fixing bacteria. However, we hypothesized that in the short term, the disturbance produced by ripping, mounding and planting of *A. saligna* significantly decreases R_s and that soil moisture (θ) will vastly prevail over soil temperature (T_s) as a driver of R_s .

To the best of our knowledge, no previous study has reported R_s measurements from arid shrublands in Chile. We monitored R_s and micrometeorological and soil variables from 2008 to 2010, 3 to 6 years after afforestation with *A. saligna*. The objectives of our study were: 1) to quantify the effect of soil temperature on R_s daily variation, 2) to model the seasonal variability of R_s based on soil moisture and temperature and, 3) to estimate annual R_s in both natural and afforested sites.

2. Materials and methods

2.1. Study sites and measurement periods

The study area is located in Las Cardas Experimental Station (University of Chile), in the Coquimbo region of Chile (30°15' S; 71°17' W), 45 km south of La Serena, in the central depression between the Coastal Ranges and the Andes mountain. The climate is arid Mediterranean, with temperate conditions due to high marine influence (less than 20 km from the Pacific Ocean). A data series from a weather station located at the experimental station (years 1977–2010), showed an average annual precipitation of 138 mm, concentrated between June and September, with average temperatures between 26 °C in January (summer) and 5 °C in July (winter). Rainfall was 32 mm in 2007, increasing during the study years to 182 mm in 2008, 112 mm in 2009 and 150 mm in 2010.

The dominant vegetation is an open low-statured (1.2 m) shrubland, part of the ecotone of two large floristic units of 'pichanilla' (*Gutierrezia resinosa* S.F. Blacke) and 'inciense' (*Flourensia thurifera* [Mol.] DC.), with some sparse *Acacia caven* (espino)

trees, *Litharea caustica* (litre) and succulents, and a temporally active grass stratum (winter and beginning of spring) dominated by therophytes (Lailhacar and Aylwin, 1988). For our study, we used two 16 ha plots of native shrubland, 500 m apart. One of the sites was left as a control 'natural' condition (NAT), while the other was afforested (AFF) with *Acacia saligna* (Labill.) H.L. Wendl between 2005 and 2006 (1200 trees ha⁻¹).

The soil type at both study sites is Aridisol derived from granitic and andesitic alluvium, showing sandy loam texture in the surface horizons and loamy sand at lower depths. Drainage is moderate with low sodium content and no carbonates (Casanova and Luzio, 2008). Soil depth is variable from 50 to 100 cm over a duripan. We measured volumetric soil water content (θ) at field capacity to be approximately 0.29 m³ m⁻³. Soil preparation for *A. saligna* plantation involved land clearing and building 40-cm mounds and infiltration ditches to increase soil depth using a bulldozer with a ripper. Mounds were separated 5 m from each other. Land clearing removed about half of *G. resinosa* and *F. thurifera* shrubs, without affecting *A. caven* and *L. caustica* trees. Moderate grazing (0.12 goat animal units per hectare) occurred in both enclosures from 2006 to 2009, which is representative of the traditional range management in the region, and no grazing occurred after that.

Field campaigns were carried out approximately once a month between February 1st 2008 and January 31st 2011. A total of 29 field campaigns were completed during the 3-year period (in 2008 months 01, 02, 04, 05, 09, 10; in 2009 all twelve months; in 2010 all months except 10 and 11; in 2011 month 01).

2.2. Soil CO₂ fluxes and micrometeorological data

On each field campaign, R_s was measured four times per day, for two days. Each day, two of the measurements were carried out at minimum and maximum T_s (6:00–7:30 and 13:00–15:00, respectively), while the remaining two measurements were distributed in equivalent time intervals between these two extremes, including dark periods. The cumulative flux for the two days was calculated as the integral of the eight measurements, and the daily flux for each field campaign was calculated as the average of the two days.

R_s was measured with an Automated Soil CO₂ Flux System (LI-8100, LI-COR, Lincoln, NB, USA) connected to a 20-cm-diameter accessory chamber (LI-8100-103). This system works as a dynamic closed chamber, which was manually controlled using a LifeDrive PDA (Palm Inc., Sunnyvale, CA, USA). Measurements were made as short as possible in order to keep the inside of the chamber as similar to environmental conditions as possible: 3 min in periods of high vegetation activity during spring, and up to 10 min in periods of low soil temperature and low soil moisture in autumn, when R_s values were very low. Air temperature inside the chamber (T_a), was registered using a thermistor incorporated to the soil chamber. A soil temperature probe Type E (LI-COR) allowed estimating T_s at 5 cm depth on each measurement of R_s . Soil moisture (θ) was estimated with a soil moisture probe (model EC-5, ECH₂O, Decagon Devices Inc., Pullman, WA, USA) at 5 cm depth, twice on each sampling campaign, at times of the day that showed the highest and lowest values of T_s .

Five PVC collars (12 cm height, inner diameter 21.34 cm) were installed on each site at 7 cm deep, 48 hours prior to the first measurement in order to stabilize soil conditions, and were kept for the rest of the study period. Plants growing inside the soil respiration collar were carefully removed by hand prior to each field campaign so that measurements represented below-ground CO₂ efflux. Collars were placed in a 30 m radius area around a tower where ecosystem-level fluxes were being measured at each site. For the NAT site, three collars were installed on bare ground (BG), one under *F. thurifera* (FT) and one under *G. resinosa* (GR); in the afforested site, two collars

were installed on bare ground over the berm (BG-B), two on bare ground over flat soil (BG) and one under GR. Collars were apportioned to each cover type (e.g., bare ground, under shrub cover) according to estimations made by Perez-Quezada et al. (2011) at the same sites. This implied that to obtain the daily flux, the cover types were multiplied by the percentage they represented (i.e., $R_s\text{-NAT} = \text{BG} \cdot 0.69 + \text{GR} \cdot 0.12 + \text{FT} \cdot 0.19$ and $R_s\text{-AFF} = \text{BG-B} \cdot 0.44 + \text{BG} \cdot 0.44 + \text{GR} \cdot 0.12$).

Continuous measurements (averaged every 20 min) of microclimatic and soil variables were collected with sensors mounted on a tower at the NAT site. Photosynthetically active radiation (Q) was measured with a quantum sensor (model LI190SB, Li-COR), and wind speed was monitored with a cup-anemometer (model WS03101 Campbell Scientific Inc. (CSI), Logan, UT, USA), at 2.5 and 3 m height, respectively. Precipitation was recorded using a tipping bucket rain gauge (Model TE525, CSI), installed at 1.5 m height. Soil moisture (θ , $\text{m}^3 \text{m}^{-3}$) and soil temperature (T_s) were monitored at 5 cm depth using a water content reflectometer (model CS615, CSI) and soil temperature probes (model TCAV, CSI). These parameters were used as referential for describing the seasonal variation of R_s and to model three R_s values for 2008 (described in section 2.3).

Quality assurance of R_s data was obtained during the measurements by looking at the coefficient of variation (CV) of the flux estimation. Measurements were repeated if CV was higher than 5%. The software LI-8100 (M) PC Client v2.0.0 (LI-COR Inc. 2004–2006) was used to calculate soil respiration flux using a linear and a logarithmic model. The flux that showed the lowest CV value was selected.

Seasonal (cumulative) R_s was obtained by integrating the area under the curve, using the trapezoidal rule:

$$R_s = \sum_{t=1}^{n-1} \frac{(R_{st} + R_{st+1})}{2} \cdot (D_{t+1} - D_t), \quad (1)$$

where D_t and D_{t+1} represent the days of the year (DOY) of two successive measurements of R_s and n is the total number of field campaigns.

2.3. Modelling soil CO_2 fluxes

We used the exponential equation of Lloyd and Taylor (1994) to model R_s as applied by Reichstein et al. (2005):

$$R_s = R_{10} \cdot e^{[E_0 \cdot (1/(T_{\text{ref}} - T_0) - 1/(T_s - T_0))]}, \quad (2)$$

where R_{10} is the R_s at 10°C ($\mu\text{mol CO}_2 \text{m}^{-2} \text{s}^{-1}$); E_0 is the temperature sensitivity parameter (K^{-1}), analogous to activation energy; T_{ref} is the reference temperature = 283.15 K (10°C); T_0 is a constant at 227.13 K (-46.02°C); and T_s is the temperature (K) of the soil.

The nonlinear model (Eq. (2)) was fit to the data in each shrubland condition of the shrubland (NAT and AFF), for each cover type (BG, FT and GR at the NAT site; BG, BG-B and GR at the AFF site), for each field campaign. A total of 174 models were fit, with 4–24 observations each. The fitness of the nonlinear models was evaluated with the determination coefficient (R^2), which was calculated as the ratio of the sum of squares regression (SSR) and the total sum of squares (SST):

$$R^2 = \frac{\text{SSR}}{\text{SST}}. \quad (3)$$

Because of equipment failure, R_s was modelled for three campaigns in 2008 (July, August and September) using micrometeorological and soil data from the tower at the NAT site (for DOY 186, 217 and 245). A stepwise-forward regression analysis was performed for each condition, which resulted in a model based on

T_s and θ for the NAT site ($R_s\text{-NAT} [\text{g CO}_2 \text{m}^{-2} \text{day}^{-1}] = -1.405 + 0.039 T_s + 40.053 \theta$, $R^2 = 0.89$, $N = 22$) and based on Q and θ for the AFF site ($R_s\text{-AFF} [\text{g CO}_2 \text{m}^{-2} \text{day}^{-1}] = -1.230 + 0.001 Q + 37.972 \theta$, $R^2 = 0.82$, $N = 16$).

Comparisons of means of R_s and soil variables between conditions were obtained using paired t -test and analysis of variance (ANOVA). When the ANOVA showed significant differences, we used a multiple comparison test (Tukey's honestly significant difference (HSD), $\alpha = 0.05$) to separate the means. The relation between T_s and θ with R_s was analysed using the correlation coefficient (r). Statistical analyses were performed using the open source software R (R Foundation for Statistical Computing, Vienna, Austria).

3. Results

3.1. Seasonal variation of R_s and micrometeorological variables

Minimum R_s was 0.20, 0.21 and $0.53 \text{ g CO}_2 \text{m}^{-2} \text{day}^{-1}$ in the NAT condition for 2008, 2009 and 2010, respectively, whereas for AFF these values were 0.13, 0.11 and $0.28 \text{ g CO}_2 \text{m}^{-2} \text{day}^{-1}$ (Fig. 1a). The minimum R_s occurred in different months each year (May 2008, June 2009 and March 2010), right before the first rainfalls occurred (Fig. 1c). R_s values increased drastically after the occurrence of precipitation, reaching maximum daily values of 1.96, 7.27 and $4.46 \text{ g CO}_2 \text{m}^{-2} \text{day}^{-1}$ for 2008, 2009 and 2010 at the NAT condition, while the maximum values in AFF were considerably lower, 1.68, 3.26 and $2.46 \text{ g CO}_2 \text{m}^{-2} \text{day}^{-1}$ for 2008, 2009 and 2010, respectively. The ratio between the maximum and minimum R_s was relatively similar between conditions for each year, but varied widely among years (around 11 in 2008, 32 in 2009 and 8 in 2010).

The maximum R_s values recorded in 2008 did not match the precipitation level registered that year (Fig. 1c). This suggests that the R_s peak was missed during that year, when equipment failure occurred during the winter. Mean θ were not significantly different between shrubland conditions ($p = 0.29$, paired t -test, $N = 29$) (Fig. 1c). Levels of Q reached maximum daily averages of around $60 \text{ mol photons m}^{-2} \text{day}^{-1}$, which occurred during December each year (Fig. 1d). Wind speed followed the same seasonal pattern, with maximum daily averages around 2.5 m s^{-1} in November (spring), and minimum values during the winter months (Fig. 1e). T_s was always higher than T_a , except for the September 2009 campaign (Fig. 1f). There were no statistical differences in T_s and T_a between shrubland conditions (paired t -test, $N = 29$, $p = 0.79$ and 0.96 , respectively).

R_s fluxes varied widely depending on the cover type (Fig. 2). The highest values of daily R_s were observed in collars located under shrubs of *F. thurifera* and *G. resinosa*, reaching a maximum in August 2009, at around $5 \text{ g CO}_2 \text{m}^{-2} \text{day}^{-1}$. The largest differences among cover types were observed when the soil was wet, after the first rains occurred (May, June and March of 2008, 2009 and 2010, respectively), and from then until the soil dried up, around November each year. In the AFF site, R_s on the BG-B was commonly lower compared to BG, although they were not significantly different ($p = 0.08$). The BG in NAT was not different than the BG in AFF (paired t -test, $p = 0.72$, $N = 29$), but was significantly higher than the BG-B in AFF (paired t -test, $p = 0.02$, $N = 29$). *G. resinosa* in NAT showed low levels of R_s compared to the other cover types under shrubs, being significantly lower than GR in AFF ($p < 0.001$) and equal to BG in AFF ($p = 0.97$). *G. resinosa* in AFF and *F. thurifera* in NAT showed the highest values of R_s , being similar between them ($p = 0.29$).

3.2. Effect of T_s on daily variation of R_s

One model of the exponential response of R_s to T_s (Eq. (2)) was fit for each condition-cover-campaign combination. Table 1

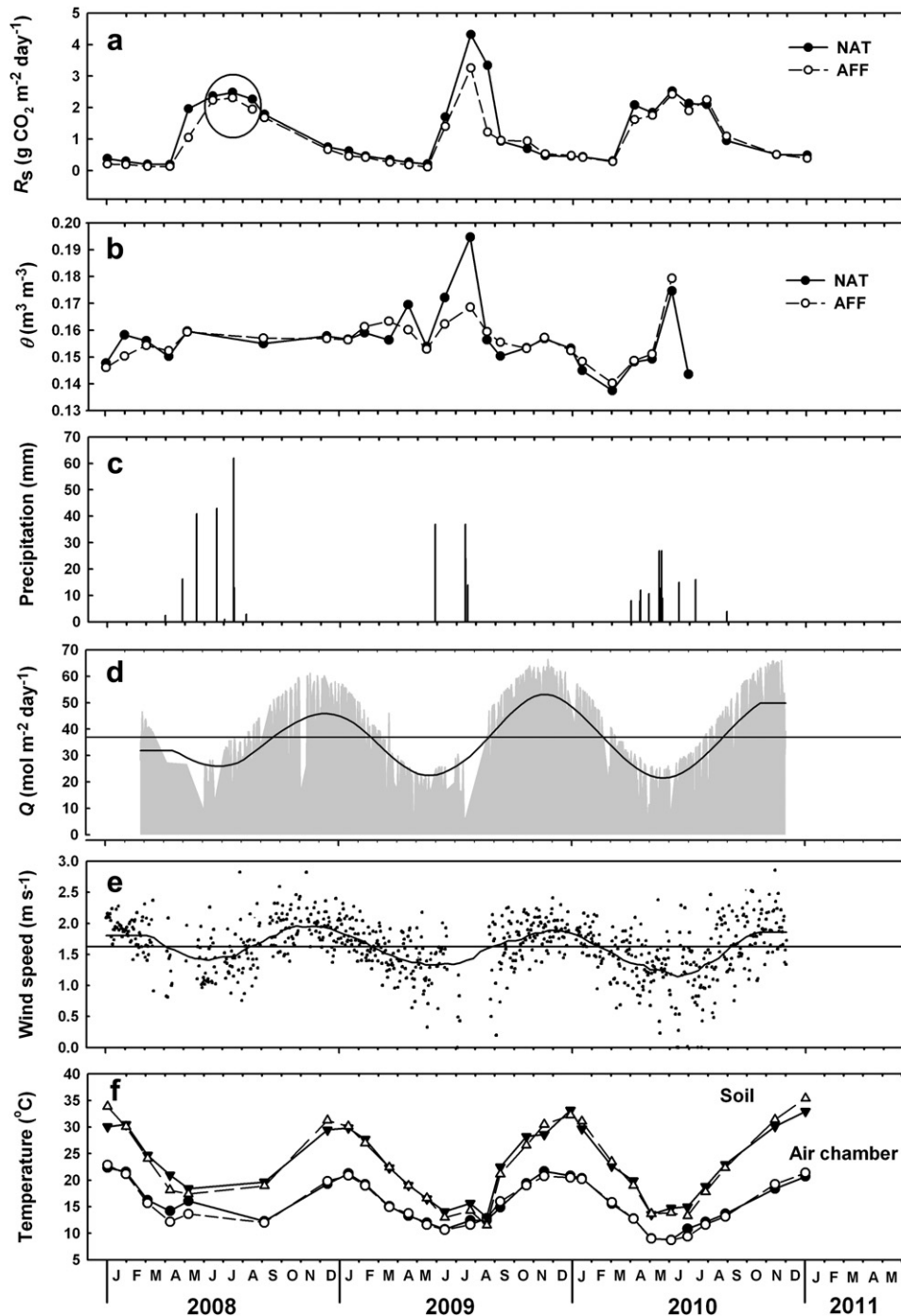


Fig. 1. Fluxes and micrometeorological variables in an arid shrubland in both natural (NAT) and afforested (AFF) conditions; a) soil respiration (R_s), circle shows modelled data; b) volumetric water content of the soil (θ); c) precipitation; d) daily sum of photosynthetically active radiation (Q), solid line is 10 day average and; e) wind speed, solid line is 10 day average and; f) soil temperature (T_s) and air temperature inside the soil chamber (T_a).

summarizes the results of the models for each month. Soil respiration at 10°C (R_{10}) followed a similar trend than R_s (Fig. 1a); minimum values remained low (<0.06) until June, and maximum values started increasing in April. The overall maximum R_{10} was reached in August, when θ is usually high and temperatures start to increase after the minimum in July. The higher daily R_s in NAT compared to AFF shown in Fig. 1a is not reflected in higher maximum R_{10} values, being these values larger for AFF. This was influenced by large R_s values observed in the collar under *G. resinosa*. The activation energy parameter (E_0) showed maximum

values during the months when the first precipitation events occurred (April–June).

In general, T_s was a good predictor of R_s . Maximum R^2 values were >0.73 for all months in both conditions. Low R^2 were more common during months when soil was dryer, between October and April, which were associated to low R_{10} and negative E_0 values (Table 1) (Fig. 1b).

Fig. 3 shows examples of data fitness of nonlinear models from the August 2009 and December 2010 campaigns, as examples of what occurred during winter and summer periods, respectively. R_{10}

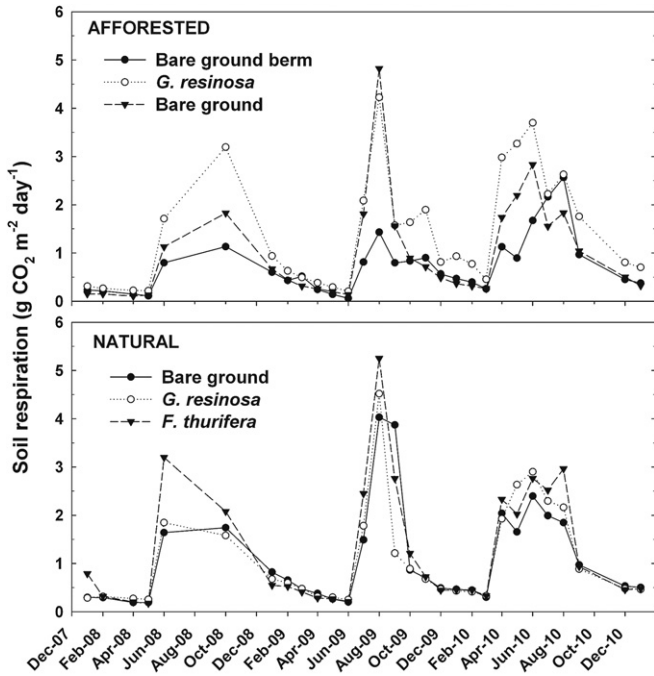


Fig. 2. Daily R_s ($\text{g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$) for different cover types in the arid shrubland sites for the three-year period.

values were much greater in August 2009 ($0.28\text{--}1.05 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$), when fluxes were larger (Fig. 1), compared to December 2010 ($0.11\text{--}0.20 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) when fluxes were smaller. The same pattern was observed for E_0 , with much greater values for August 2009 ($210\text{--}274 \text{ K}^{-1}$) than for December 2010 ($-14\text{--}67 \text{ K}^{-1}$). Coefficient of determination (R^2) showed higher values in winter (August 2009, $0.51\text{--}0.78$), than in summer (December 2010, $0.0\text{--}0.54$), being the highest values in both periods observed for GR-AFF. In general, models showed better fitness during months when the soil was wet.

3.3. Effects of T_s and θ on the seasonal variation of R_s

Seasonal variation of R_s was similar for both conditions, being NAT higher than AFF except for a few months, mostly during the third year (Fig. 4). Correlation models showed that θ was positively related to R_s in NAT ($r = 0.53$) and AFF ($r = 0.51$) ($N = 25$, $p < 0.01$). On the contrary, T_s showed negative effects in NAT ($r = -0.67$) and

AFF ($r = -0.61$) ($N = 29$, $p < 0.01$). This was caused by the fact that the maximum T_s values were reached during the summer months (December–January), while the maximum R_s values were observed during the winter months (around June) (Fig. 1).

As shown in Fig. 4, the relation between T_s and R_s varied depending on the location of the collars; those located under a plant showed higher coefficients of determination than those located on bare ground.

3.4. Estimation of annual R_s

Annual R_s ($\text{g C m}^{-2} \text{ year}^{-1}$) for NAT was 129 in 2008, 113 in 2009 and 117 in 2010 (Table 2). For AFF, the corresponding fluxes were 110, 87 and $112 \text{ g C m}^{-2} \text{ year}^{-1}$. The mean ($\pm \text{S.E.}$) for the three years in NAT ($120 \pm 5 \text{ g C m}^{-2} \text{ year}^{-1}$) was higher than for AFF ($103 \pm 8 \text{ g C m}^{-2} \text{ year}^{-1}$) and this difference was marginally significant (paired t -test, $p = 0.06$, $N = 3$).

Annual R_s separated by cover type is reported in Table 2. The highest average for the three-year period was observed under *G. resinosa* in AFF, with $153 \text{ g C m}^{-2} \text{ year}^{-1}$, followed by *F. thurifera* in NAT ($135 \text{ g C m}^{-2} \text{ year}^{-1}$), *G. resinosa* and BG in NAT ($111 \text{ g C m}^{-2} \text{ year}^{-1}$), BG in AFF ($103 \text{ g C m}^{-2} \text{ year}^{-1}$), and BGB in AFF ($75 \text{ g C m}^{-2} \text{ year}^{-1}$). The ANOVA did not show a significant effect of the year ($p = 0.21$) but did show a significant effect of cover type ($p < 0.01$).

4. Discussion

4.1. Daily R_s

Daily values of R_s in this study showed great variability within the year in both conditions. The maximum daily R_s in NAT ($7.3 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$) and AFF ($3.3 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$) were low compared to the values observed by Grunzweig et al. (2003) for an Aleppo pine plantation at the Negev desert ($16.4 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$). Even higher daily R_s values were reported by Li et al. (2008) ($57 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$) from three shrubland sites in a mountainous area in inland China. Jia et al. (2006) monitored R_s in a steppe in inner Mongolia and found maximum R_s of $13.5 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ for an ungrazed pasture and $4.7 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ for the grazed condition, which are closer to the values observed in this study. Our maximum daily R_s values were also similar to those reported by McLain et al. (McLain et al., 2008), who reported maximum fluxes of $9.4 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ in collars located under *Prosopis velutina* (Woot) trees, $7.0 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ in open spaces and $5.7 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ in

Table 1

Parameters R_{10} ($\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) and E_0 (K^{-1}) and coefficient of determination (R^2) for the exponential models of R_s based on T_s by month during the three years in the natural (NAT) and afforested (AFF) conditions.

Month	N	NAT						AFF					
		R_{10}		E_0		R^2		R_{10}		E_0		R^2	
		Min	Max	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max
January	24	0.04	0.17	83.0	199.9	0.21	0.97	0.03	0.21	-38.1	237.0	0.01	0.89
February	18	0.04	0.13	39.9	204.8	0.08	0.81	0.02	0.17	72.5	267.0	0.15	0.75
March	12	0.06	0.10	115.3	243.7	0.45	0.88	0.06	0.13	56.7	143.3	0.17	0.94
April	24	0.03	0.57	-27.4	311.5	0.02	0.81	0.01	0.89	-193.4	318.3	0.00	0.87
May	12	0.03	0.97	170.7	874.1	0.48	0.97	0.02	0.92	184.4	542.1	0.26	0.73
June	18	0.04	0.84	146.4	249.9	0.25	0.96	0.02	1.05	136.7	394.1	0.28	0.92
July	12	0.41	0.66	70.1	201.7	0.34	0.94	0.20	0.61	66.0	274.4	0.31	0.66
August	12	0.47	1.23	72.4	262.8	0.27	0.90	0.38	1.28	80.7	186.2	0.39	0.83
September	12	0.22	0.86	65.5	219.6	0.06	0.86	0.19	0.44	26.2	160.0	0.10	0.78
October	12	0.20	0.52	57.5	126.6	0.21	0.82	0.20	0.72	17.4	101.9	0.06	0.82
November	6	0.14	0.16	36.3	126.7	0.44	0.85	0.20	0.43	-9.7	76.0	0.02	0.77
December	12	0.09	0.15	-22.5	154.5	0.03	0.76	0.11	0.29	30.0	88.6	0.04	0.76
Average		0.15	0.53	67.2	264.6	0.24	0.88	0.12	0.60	35.8	232.4	0.15	0.81

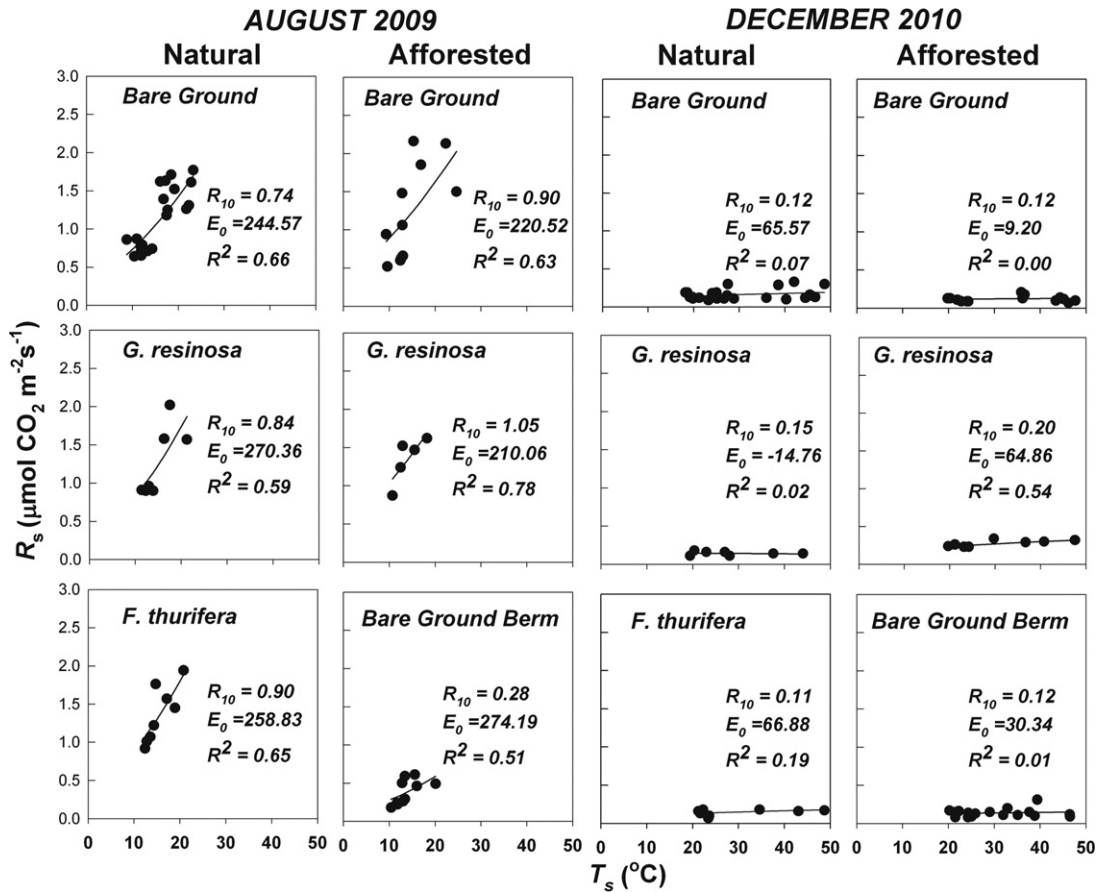


Fig. 3. Nonlinear models of R_s based on T_s for campaigns August 2009 and December 2010. Parameters correspond to those of Eq. (2).

collars located under dead mesquite, in a semiarid grassland in Arizona.

4.2. Effect of T_s on the daily variation of R_s

Average R_{10} values for NAT (0.15–0.53 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) and AFF (0.12–0.60 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) (Table 1) were low compared to the values obtained by Perez-Quezada et al. (2010) (1.08–3.64 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$), and by Li et al. (2008), who reported values of R_{10} of 1.27–2.9 $\mu\text{mol CO}_2$

$\text{m}^{-2} \text{ s}^{-1}$ for three shrubland sites in a mountain area in China. Fitted values of E_0 (K^{-1}) showed broad ranges in NAT (67.2–264.6 K^{-1}) and AFF (35.8–232.4 K^{-1}) (Table 1) which are similar to the range obtained by Perez-Quezada et al. (2010) (58.2–208.9 K^{-1}). Such important fluctuations in the sensitivity of R_s to temperature have been reported by Janssens and Pilegaard (2003) who studied Q_{10} (ratio between the respiratory flux at one soil temperature and the flux at a temperature 10 °C lower) of the R_s at a Danish beech forest soil. We tried fixing the value of E_0 to 308.6 as suggested by other studies (Buchmann, 2000; Li et al., 2008), but the R^2 values

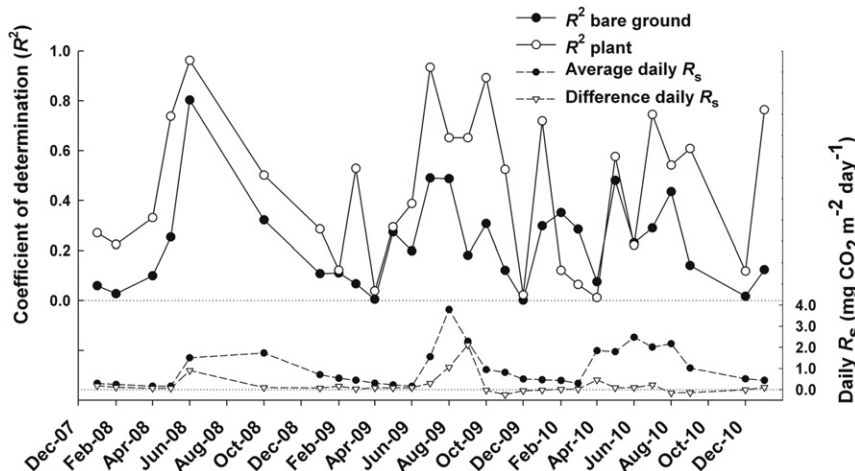


Fig. 4. Coefficient of determination of the linear regression R_s-T_s for collars located under plant cover or in bare ground (upper panel), and the average and difference (NAT–AFF) daily R_s between shrubland conditions (lower panel).

Table 2
Annual R_s ($\text{g CO}_2 \text{ m}^{-2} \text{ year}^{-1}$) for different cover types during years 2008–2010.

Year	GR-AFF ^a	FT-NAT	GR-NAT	BG-NAT	BG-AFF	BGB-AFF
2008	157	148	108	110	95	68
2009	130	125	101	112	102	61
2010	172	131	125	112	112	96
Average ^b	152.9 ± 12 ^A	134.8 ± 7 ^{AB}	111.3 ± 7 ^{BC}	111.2 ± 1 ^{BC}	102.9 ± 5 ^{BC}	74.9 ± 11 ^C

^a GR, *G. resinosa*; FT, *F. thurifera*; BG, bare ground; BGB, bare ground berm; AFF, afforested; NAT, natural.

^b Cover types are sorted from higher to lower average annual R_s values. Averages that do not have the same superscript letter are significantly different ($\alpha = 0.05$).

decreased (data not shown), given that the variability had to be represented solely by the R_{10} parameter. Some extremely low values obtained for R_{10} and E_0 , and some low R^2 values in our study suggest that the Lloyd and Taylor model (Eq. (2)), does not always work well for the dry conditions observed during the summer-fall months.

The highest R^2 values observed for the NAT site, compared to AFF, are likely due to the higher vegetation cover, since R_s measured in collars located below vegetation showed higher dependence on T_s . This was supported by the fact that the coefficient of determination obtained for plant cover was higher than for bare ground for most of the field campaigns (Fig. 4). Moreover, the average R^2 for NAT (0.44) was significantly higher than that for AFF (0.23) (paired t -test $p < 0.01$, $N = 29$). These results are also in line with the higher sensitivity shown for autotrophic R_s than for heterotrophic R_s to T_s , as documented by Boone et al. (1998).

4.3. Environmental controls of the seasonal variability of R_s

As we expected, daily R_s was positively correlated with θ . We believe the higher absolute value of the correlation coefficients between R_s and T_s is not causal since it showed a negative effect. It rather seems an artifact generated by increased water content in the ecosystem when temperature was lower (during the winter) (Fig. 1).

Working in a semiarid grassland for two monsoon seasons, McLain et al. (2008) found a significant effect of accumulated precipitation (3 days before flux measurement) and T_s over R_s . Zhang et al. (2011) found that T_s and fine root biomass accounted for 78–85% of R_s variation in stands of poplar plantations in China.

The aridity of our study sites (annual precipitation = 138 mm) place them at the very bottom of the rainfall conditions reported in the literature for studies of R_s . This explains why soil moisture and not T_s , was the main driver of soil C efflux in our study. Precipitation in the Coquimbo region occurs only during winter months, generating an increase in θ , which in turn triggers the increase in autotrophic and heterotrophic activity in the soil. An opposite example is given by Peichl et al. (2010) who found a positive effect of T_s but not of θ on R_s in a chronosequence of pine plantation in Canada, likely due to the fact that precipitation at their study site was >1000 mm.

Maximum daily R_s in our study seemed to be related to the precipitation of the previous year (data not shown). A possible explanation for this is that higher precipitation should generate larger C assimilation, root growth and C reserve storage in permanent tissues during the previous year. This memory or carryover effect has been shown to be greater in shrubs than in grass species (Jobbagy and Sala, 2000). This would create a higher potential for root and rhizo-microbial activity the current year due to a higher C substrate availability in roots (Hogberg et al., 2001), which would stimulate high pulses of R_s when precipitation occurs during the beginning of winter. Annual R_s apparently did not depend on these large peaks, but on the persistence of higher levels of θ , which relates to the total precipitation during the current year.

4.4. Total annual R_s

Total annual R_s levels in our study (113–129 $\text{g C m}^{-2} \text{ year}^{-1}$ for NAT and 87–112 $\text{g C m}^{-2} \text{ year}^{-1}$ for AFF) were low compared to the range observed by Perez-Quezada et al. (2010) in four different land-use types in the Kazakh steppe (288–431 $\text{g C m}^{-2} \text{ year}^{-1}$). Even higher annual R_s were found in a Sitka spruce chronosequence (564–991 $\text{g C m}^{-2} \text{ year}^{-1}$), being the highest value from the youngest stand (10 years old) and the lowest from the oldest (47 years old) (Saiz et al., 2006). Kellman et al. (2007) found higher R_s on grassland sites (425–533 $\text{g C m}^{-2} \text{ year}^{-1}$), compared to adjacent forest sites (219–281 $\text{g C m}^{-2} \text{ year}^{-1}$).

Annual R_s in NAT was higher compared to AFF for the three years, although the difference between the two conditions varied among years, being 17.1% and 29.6% higher in NAT for 2008 and 2009, but only 4.3% during 2010, when some daily R_s values in AFF were higher (Fig. 1a). This difference seemed to be related to two independent processes: the pattern of precipitation (not necessarily annual precipitation) and the progressive increase of native shrub and tree (*A. saligna*) cover. In 2010 it rained 150 mm at the study area, but precipitation events were less intense than in 2009, when it rained only 112 mm but concentrated in fewer days (Fig. 1c). More intense precipitation events allow deeper water infiltration into the soil. This water can be used more easily by deep-rooted shrubs and trees. Higher cover of woody plants in the NAT site could generate higher root activity in years when precipitation reaches deeper soil layers, increasing the difference in annual R_s between NAT and AFF. Similar results were shown by McLain et al. (2008), who reported fluxes from a semiarid grassland in Arizona, where *P. velutina* trees were proliferating. They found that in one year R_s was higher under live mesquite trees, followed by open space areas and areas under dead trees. As in 2009 for our study, the study by McLain et al. (2008) also showed lower total precipitation but higher difference in θ between cover types (Fig. 1b). The previous year however, they found no difference between sites, suggesting that gas flux alterations were moderated by inter-annual variation in monsoon precipitation patterns and the associated difference in soil moisture (Fig. 1b).

4.5. Effect of afforestation on R_s patterns

We measured R_s in two adjacent plots covered by an arid shrubland in either natural condition or afforested with *A. saligna* for three years, starting when the planting was three years old. The AFF condition site showed lower annual R_s (average 17%) than the NAT site. The lower R_s level of the AFF condition was likely due to the loss of soil C and woody species cover at the time of soil preparation prior to planting (Perez-Quezada et al., 2011). This was supported by the lower R_s registered in collars installed on bare ground on the planting berms, compared to those installed on bare ground in undisturbed soil at the AFF site (Fig. 2).

As *A. saligna* trees progressively increase in size, shrub cover will also recover, likely increasing soil organic C and generating higher root growth, particularly of large roots. Don et al. (2009) found that,

after the first year of afforestation in a grassland site, the gross primary productivity recovered, increasing the C sink at the afforested site. At the same time, an increase in R_s was observed in the afforested site, being higher than in the grassland, although they sampled only during three years. The opposite trend was reported by Zhang et al. (2011), who verified an increase of $0.13 \text{ Mg C ha}^{-1} \text{ year}^{-1}$ in a poplar plantation chronosequence of 2, 8 and 15 years, but a decreasing trend in R_s explained mainly by higher T_s and fine-root growth in the younger plantation.

An alternative scenario is that the AFF site will never recover the ecosystem C stock level of the NAT site. Liao et al. (2010) support this view, reporting that plantations decreased net primary production, litterfall and R_s by 11, 34 and 32%, respectively. Their results held true even when plantations were separated by age (<25 years old vs. ≥ 25 years old), stand types (broadleaved vs. coniferous and deciduous vs. evergreen), origin of tree species (native vs. exotic), land-use history (afforestation vs. reforestation), site preparation for plantations (unburnt vs. burnt), and study regions (tropic vs. temperate). Saiz et al. (2006) sampled R_s in a longer chronosequence of spruce plantation, and did not register an increase in soil C, but observed a decrease of R_s between 10 and 31 year-old stands and a stabilization in the older one (47 years after plantation). Two other studies compared grasslands with adjacent forest ecosystems (Kellman et al., 2007; Smith and Johnson, 2004), and even though no differences were seen in C content in the soils, grassland sites showed around 40% higher annual R_s .

The results from the studies reviewed in this section suggest that R_s could either increase or decrease in the next few years at the AFF site. However, in most cases reviewed, afforestation was carried out over grassland or plowed soils, which is not exactly the same situation as in our study, where woody vegetation was already present at the NAT site. In other words, the question remains whether woody vegetation cover at the AFF site will grow denser than at the NAT site in the future, and whether this will increase soil C content and R_s . The fact that daily R_s values were higher in AFF for a few campaigns during the third year of measurements (Fig. 1) could be indicative of a trend of equilibrating R_s between conditions, and a potential higher level for the AFF condition in the future.

5. Conclusions

Understanding the effects of afforestation on carbon cycle components is important because of its implications for the global C cycle. *Acacia saligna* is the most widely planted non-timber species, with around 600,000 ha established worldwide, and projected to grow (Griffin et al., 2011).

We found that the afforestation of an arid shrubland resulted in a 17% reduction of annual R_s , 3–6 years after plantation. Soil temperature was a good predictor of R_s on certain periods of the year, although we could not relate this to specific levels of soil moisture. The extremely arid conditions of our site determined that the seasonal variation of R_s was more related to soil moisture than to soil temperature. The larger shrub cover in the natural condition made this site more sensitive to intense precipitation events, which cause water to be available for shrubs at deeper levels, due to increased water infiltration. This implied that the difference between the two conditions was greater in years in which precipitation events were more intense, rather than being related to total annual precipitation.

The fact that soil respiration was greater in the afforested site for a few months during the third year generates an uncertain scenario for the near future. The potentially greater R_s in the afforested site will depend on the stability of the soil C that may increase in this

condition, and how a larger tree cover may affect soil temperature and hydrological conditions.

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