

Reply to Referee #1

We would like to thank referee #1 for the detailed review of our manuscript and the suggestions that will help to improve our manuscript. In the following, we will answer each of the referee's comments.

This short paper attempts to describe the impacts of land degradation in semiarid ecosystems on carbon fluxes on the basis of the differences observed between two eddy covariance flux sites in SE Spain. The authors clearly demonstrate that most of the expected meteorological controls over C flux are equivalent between sites, but the carbon fluxes are strikingly different, varying by a couple of orders of magnitude. As they highlight, this difference in observed net carbon flux is a result of contrasting fluxes of carbon from “subterranean ventilation”. As the authors have addressed in other publications, this large carbon efflux cannot be accounted for due to in-situ concurrent biological activity – and this greatly complicates interpretation of contrasting results between the sites, and thus the assessment of the impacts of land degradation.

Unfortunately, the authors do not address this challenge very effectively, and in its current form there is little support for any conclusion about the impacts of land degradation on carbon fluxes. It may be that the nature of the sites makes it impossible to carry out such a comparison convincingly, but addressing a number of areas is required before this can be determined.

General comments

1. First, the nature of the disturbance and extent to degradation needs to be described in more detail. The similarities between the sites are described in detail, but the crucial differences need more full description than Table 1, and more importantly, the biological implications of these differences (detailed hypotheses) need to be articulated.

We agree with the reviewer. Therefore, the following paragraph will be added to the revised manuscript in page 3 line 20:

“Some land degradation processes are evident when we compare the “natural” site with the “degraded” site. This land degradation processes can directly affect abiotic and/or biotic factors, which in turn influence the biological

and/or non-biological processes that compose the net ecosystem CO₂ exchange. Firstly, vegetation cover is almost 3 times higher in Balsa Blanca (BB), the “natural” site; this implies for the degraded site higher thermal and radiative stress in the soil, especially during the drought period (Rey et al., 2017). The overall hypothesized effects of this degradation driver on biological processes are a direct reduction in plant productivity and respiration, and an indirect decrease in heterotrophic respiration. Secondly, the higher cover of bare soil and outcrops in Amoladeras (AMO), the “degraded” site, may increase the soil-atmosphere interconnectivity, which indirectly can enhance advective CO₂ release through subterranean ventilation. Thirdly, the reduced soil fertility and depth may provoke changes in microbial communities (Evans and Wallenstein, 2014) due to stronger nutrient and water limitations. Consequently, a direct decrease in heterotrophic respiration and plant productivity and respiration is expected.”

2. Second, these hypotheses need to detail biological controls and the non-biological controls over C fluxes at these two sites, and the fluxes need to be interpreted in that light. In particular, it is differences in productivity that would be key to understanding this. Although it will be difficult given the atypical conditions of a large non-concurrent biological carbon efflux, NEE should be partitioned, and GPP between the sites compared. In addition, there should be a more detailed comparison of the ET fluxes, which in these ecosystems seem to be providing a more comparable indication of ecosystem function. And taken together, it would be interesting to assess inter-site differences in water use efficiency.

According to the referee’s suggestion, we have partitioned the net CO₂ fluxes (F_c) in order to assess the potential direct influence of land degradation on the gross primary production (GPP) and ecosystem respiration (R_{eco}) components. Given the extreme CO₂ release detected due to subterranean ventilation, two steps have been performed at each site.

Firstly, we have modelled the ventilative CO₂ efflux by adapting the approach proposed by (Pérez-Priego et al., 2013) with the results of previous studies performed in both sites (López-Ballesteros et al., 2016; 2017). Essentially, we aimed to isolate those moments when subterranean ventilation (V_n) dominates the F_c and biological fluxes are negligible. These moments correspond to daytime hours during the extremely dry periods. Data were selected using the following conditions:

- (i) Net radiation $> 10 \text{ W m}^{-2}$
- (ii) $8 < \text{Daily averaged bowen ratio} < 10$
- (iii) Daily soil water content (in bare soil) $< 10^{\text{th}}$ percentile (Amoladeras) and $< 20^{\text{th}}$ percentile (Balsa Blanca)

A less restrictive threshold was used in Balsa Blanca in order to get enough data to build the V_n model, since long-term data gaps occurred in Balsa Blanca during the summer seasons of 2012, 2014 and 2015. Afterwards, in order to build the linear model of V_n , these selected F_c data (maximum quality; QC flag=0) were related to the friction velocity (u_*). The model results for both sites are shown below:

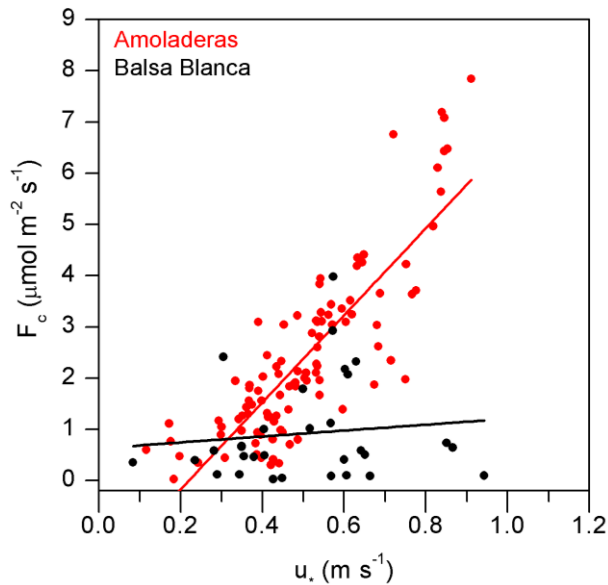


Figure S1: Half-hourly net CO_2 fluxes of maximum quality (QC flag=0) versus friction velocity (u_*) corresponding to daytime hours during the extremely dry periods when subterranean ventilation dominates the net CO_2 flux. Red and black dots represent Amoladeras and Balsa Blanca, respectively.

Model parameters	Amoladeras	Balsa Blanca
Intercept \pm error (p-value)	-1.876 ± 0.291 ($4e-09$)	0.628 ± 0.508 (0.226)
Slope \pm error (p-value)	8.500 ± 0.549 ($<2e-16$)	0.578 ± 0.944 (0.545)
R^2	0.706 ($<2.2e-16$)	0.013 (0.5451)
n	102	31

Table 3: Linear regression results between half-hourly net CO_2 fluxes of maximum quality (QC flag=0) and friction velocity (u_*) used to model subterranean ventilation.

As the table above shows, the V_n model is uniquely valid for Amoladeras.

Therefore, we only applied the V_n model to Amoladeras data, concretely, during those periods were ventilation occurs according to previous research (López-Ballesteros et al., 2017):

- (i) Net radiation $> 10 \text{ W m}^{-2}$
- (ii) Daily averaged bowen ratio > 4
- (iii) Daily soil water content (in bare soil) $< 0.01 \text{ m}^3 \text{ m}^{-3}$
- (iv) σ_{swc} (daily variance of soil water content in bare soil) $< 5e-6 (\text{m}^3 \text{ m}^{-3})^2$

We use those moments with very low σ_{swc} in order to discern R_{eco} increases caused by rain pulses (Birch effect) from V_n fluxes during the dry season.

Then, the modelled ventilative fluxes were subtracted from the measured F_c to obtain the F_c corresponding only to biological processes (i.e. biological F_c ; see Figure below).

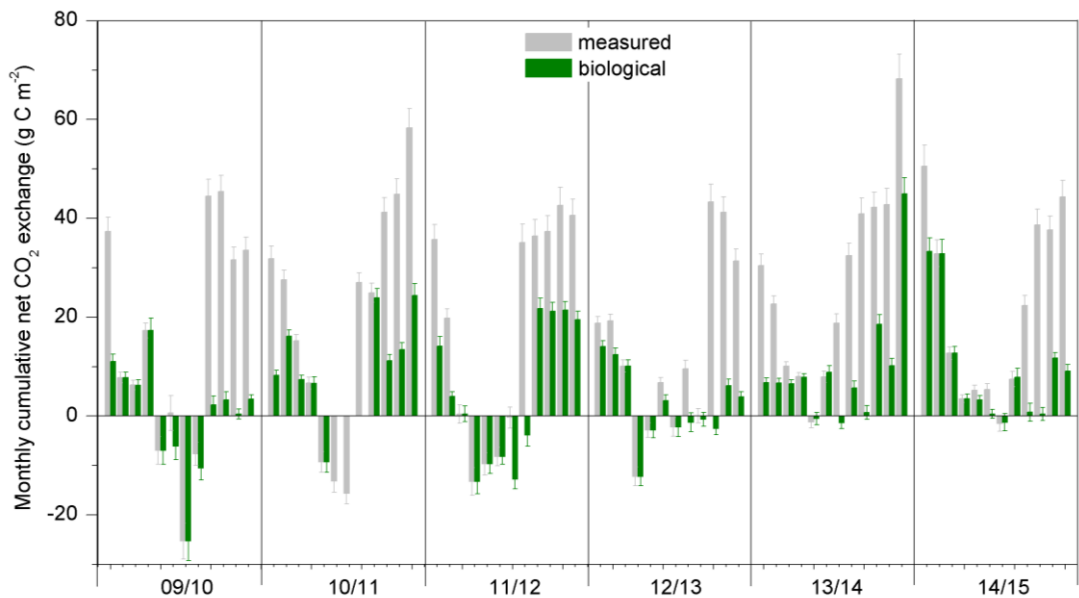


Figure S1: Cumulative measured and biological (after applying the ventilation model) net CO₂ exchange for every month of the study period (5 hydrological years; 2009-2015) in Amoladeras.

Secondly, the partitioning approach proposed by Lasslop et al. (2010) was applied to the biological F_c for both sites in order to obtain GPP and R_{eco} fluxes. We chose this approach given the determinant influence of hydric stress, in this case

atmospheric drought (assessed via VPD), on the physiology of *Machrocloa tenacissima*, the dominant plant species of the studied semiarid ecosystems (Pugnaire et al., 1996; López-Ballesteros et al. 2016).

This information would be added to a new section within the manuscript, concretely the subsection “2.3. Flux partitioning to estimate GPP and R_{eco} ” within the Material and Methods section.

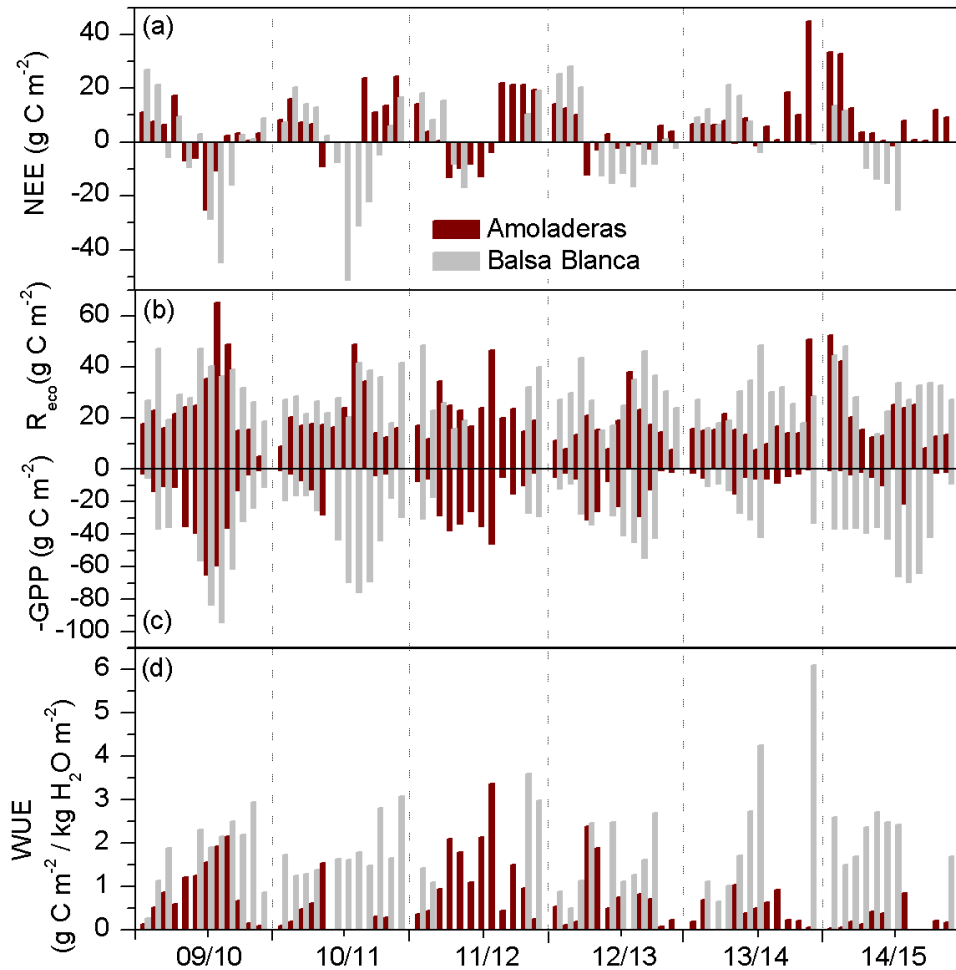


Figure 7: Monthly cumulative fluxes of (a) biological net ecosystem CO_2 exchange, (b) ecosystem respiration (R_{eco}), (c) negative gross primary production and (d) water use efficiency over the six hydrological years of study (2009-2015) for Amoladeras (dark red) and Balsa Blanca (grey). Lacking bars correspond to long-term data losses.

Finally, the results of the “biological” annual carbon balance are in accordance with the hypotheses, since annual C emission was always measured at the “degraded” site, whereas the “natural” site acted as a neutral and mild C sink. On average, Amoladeras emitted 32 g C m^{-2} more than Balsa Blanca.

Year	Amoladeras	Balsa Blanca
2009/2010	3+-7	-32+-10
2012/2013	28+-5	0+-8

We could not compare the annual C balance of 2010/2011 between sites due to a long-term data gap in the u^* time series in Amoladeras during the spring months (February-April).

During autumn, monthly biological net CO_2 fluxes were, on average, ~ 4 times higher at the “natural” site, excepting the last study year, when the net CO_2 emission at the “degraded” site was 21 times greater than at the “natural” site. However, during winter and spring months, net CO_2 uptake was generally higher at the “natural” site (Fig. 5a).

On average, during the six years of study, GPP, R_{eco} and WUE were nine, twice and ten times higher, respectively, at the “natural” site compared to the “degraded” site. Firstly, GPP was always higher at the “natural” site compared to the “degraded” site (Fig. 7c). Major differences occurred in autumn 2014/2015, when monthly cumulative GPP at the “natural” site was 32 times higher on average. Similarly, R_{eco} was generally higher, up to 786% (October 2014), at the “natural” site. However, respiratory fluxes were occasionally greater at the “degraded” site, from 2% to 31% higher, during spring and winter months of all studied years excepting 2013/2014 (Fig. 7b). Maximum inter-site differences in GPP and R_{eco} were found in winter and autumn 2014/2015, following the driest year, when monthly GPP was, on average, ~ 30 times higher at the “natural” site compared with the “degraded” site. Similarly, monthly R_{eco} was ~ 5 times greater at the “natural” site. Inter-site differences in partitioned fluxes could not be assessed during spring months due to the lack of data from the “natural site”. Secondly, Water Use Efficiency (WUE) was lower at “the “degraded” site showed during the whole study period, when maximum and minimum differences coincided with the highest and lowest differences in GPP between sites. On average, monthly WUE was 6 and 1.5 times higher in the “natural” site during winter and spring. Major inter-site differences were found in autumn and winter 2014/2015 (Fig. 7d).

This information would be added to a new section within the manuscript, the subsection “3.4. Biological Net Ecosystem Exchange, Gross Primary Production, Ecosystem Respiration and Water Use Efficiency” within the Results section. Accordingly, these results would be discussed and related to the study hypotheses in the Discussion section.

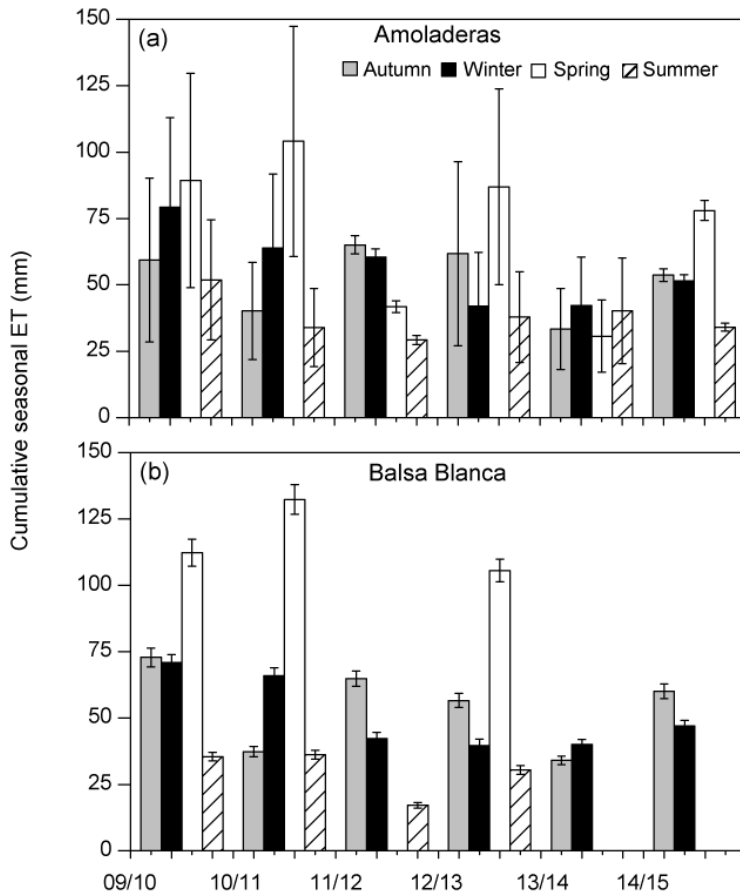


Figure 5: Cumulative seasonal evapotranspiration fluxes (ET) over the study period in both experimental sites. In case of Balsa Blanca, lacking bars correspond to long-term data losses (>50% data). Error bars denote uncertainty derived from the gap-filling procedure.

Apart from that, ET results showed ~30% higher ET at the “natural” site compared to the “degraded” site during spring. Major inter-site differences in autumn occurred in the first and last year of study, when ET was 23% and 12% higher at BB, respectively. In this regard, we think that the “natural” site shows more capacity to maintain water availability during the growing season, however, the

lack of data complicates the interpretation. The higher uncertainty of ET data in Amoladeras is due to a higher fraction of short-term data gaps compared to Balsa Blanca, on average annual fraction of data losses is 27% higher in Amoladeras.

This information would be added to the subsection 3.3., which would be renamed as “Seasonal and diurnal net CO₂ and water vapor exchanges” within the Results section. Accordingly, these results will be discussed in the Discussion section.

3. Third, the EVI time series as an indicator of productivity requires a closer examination. Given the differences in vegetation cover between the sites (Table 1), it is the similarity in EVI values, rather than the differences (except in the final year), between the two sites that seems most striking. This would suggest that productivity between the sites is not very different, and EVI based GPP estimates would be similar. Does observed tower GPP support this?

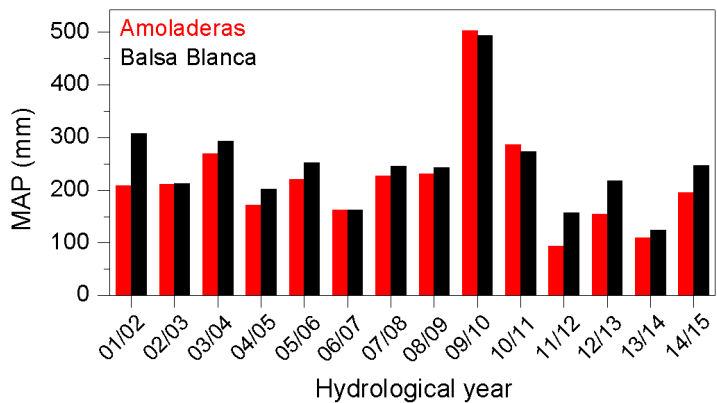
As stated before, we have found that GPP was always higher at the “natural” site compared to the “degraded” site. Thus, there is a discrepancy between GPP estimates and EVI values. We think that this is due to the different spatial scales defining every measurement. MODIS pixels have an area of ~6.25 ha while the eddy covariance footprint corresponds to a smaller area of ~1ha. Therefore, there is an EVI uncertainty that stems from the influence of other surface elements apart from vegetation, such as bare soil or outcrops within the pixel, which is our case. In fact, previous studies confirm the discrepancy between MODIS- and EC-derived GPP estimates, especially on sparse vegetation areas with low productivity (Gilabert et al., 2015).

This information would be added to the fifth paragraph of the Discussion section of the revised manuscript.

4. Fourth, the downward trend in maximum annual EVI is interesting, and could be investigated more, and potentially over a longer time period. Is it significantly related to a trend in precipitation, and a trend in productivity from the towers? The contrasting response between the sites in the final year of the record is striking, is it reflected in the tower flux record also – it seems the record is complete over the winter period at least?

The period we are studying is too short to assess trends. In addition, precipitation in this region is quite variable and as said before, the direct comparison between EVI and GPP estimates from EC measurements can lead to biased conclusions. Apart

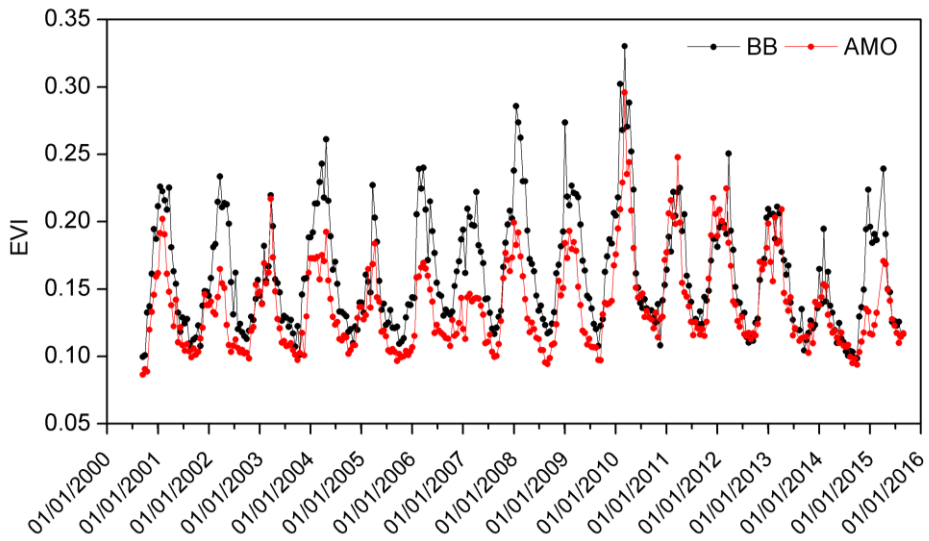
from that, if we look to a wider time window, by using longer time series of EVI and precipitation, we can realize that the precipitation in 2009/2010 was extremely high compared with the annual precipitation of the experimental sites, which equates to 220 mm (Table 1). Thus, we believe that instead of a decreasing trend what we see is a pulse response following the wettest year, as can be seen in the figures below. In fact, inter-site differences in EVI are greater before the study period, from 2000 to 2009, compared to 2009-2014. During the last year of the study period, this difference became similar to the pattern observed before the wettest year.



Notice that precipitation data shown in the figure above have not been measured in the experimental sites but quite near them since the EC stations were installed in 2006 (Balsa Blanca) and 2007 (Amoladeras). Concretely, the agro-climatic stations, “Almería” and “Níjar”, where precipitation was measured, are 13.55 km and 11.22 km from Amoladeras and Balsa Blanca, respectively.

Source:

<https://www.juntadeandalucia.es/agriculturaypesca/ifapa/ria/servlet/FrontController?action=Init>



5. Fifth, given that soil CO₂ concentration is measured at two depths, is it possible to estimate soil CO₂ flux? This could be used to partition the concurrent biological CO₂ signal, versus non-biological, and potentially the impacts of degradation on these two different processes.

The estimation of soil CO₂ efflux by using the gradient method (Sánchez-Cañete and Kowalski, 2014) assumes that the release of CO₂ from soil is performed by diffusion exclusively. However, as demonstrated in previous studies, advection (non-diffusive transport) can play an important role in the soil-atmosphere gaseous exchange (Kowalski et al., 2008; Sanchez-Cañete et al., 2011; Sánchez-Cañete et al., 2016; López-Ballesteros et al., 2017; Serrano-Ortiz et al., 2009; Subke et al., 2003; Risk et al., 2013; Roland et al., 2015). Further isotopic analyses are necessary to assess the role of biological vs non-biological CO₂ production processes in soil CO₂ efflux as well as to determine the transport processes driving the soil-atmosphere net CO₂ exchange. Unfortunately, we do not have these results, although we plan to work on it in the near future. Additionally, through the application of the ventilation model (previously explained), we have discriminated between the biological and non-biological net ecosystem CO₂ exchange in Amoladeras.

Overall, a considerable amount of additional analysis is required to separate out the signal from biological and non-biological controls over carbon fluxes from these two sites. It is only then when the flux can be interpreted in terms of vegetation

productivity that the impacts of degradation can be assessed in a way that provides insight into processes that are more broadly applicable across semiarid ecosystems.

We believe that the empirical ventilation model that we have added (see above) should satisfy the referee in this regard.

Specific comments

There are very few grammatical and spelling errors, a few very minor points:

P2 L19 – “concretely” is a strange word choice here and elsewhere – “definitively” is better in some cases, or it can just be removed.

We agree and we have removed “concretely” from the sentence.

P7 L31 - “punctual” is a strange word choice here – not sure what you are trying to convey.

We agree about “punctual” should be removed from the sentence. What we wanted to say is that EVI data is discrete as oppose to fluxes but this information is already explained in the material and methods section.

P7 L32 – Daily time series are hard to decipher in Figure 5. It is always a challenge to convey this information. Maybe using a solid black, and ensuring the graphic is a full-page width would help.

We believe that this picture carries much information that has to be shown and complements other figures (Fig. 4, 5 and 7) where flux data are aggregated. After trying several graphical options, we decided to use dashed lines because solid black lines (Balsa Blanca data) mask the red lines (Amoladeras data). We have increased the width of the figure and thickened the black lines.

P9 L3 – I believe it would be normal to correct pressure to sea-level equivalents before making comparisons such as these.

We agree so we have corrected pressure to sea level equivalents using the hypsometric equation and afterwards we have computed the Wilcoxon test for the different analysis periods. Results show a smaller difference between sites in corrected pressure compared to uncorrected values. These results (table below) would substitute previous ones (Tables 4, S1, S2 and S3) in the revised version of the manuscript.

<i>Period</i>	<i>Diff</i>	<i>Diff_{st}</i>	<i>p-value</i>	<i>n</i>
<i>All periods</i>	2.3226	0.3737	0	166336
<i>May-September</i>	2.2120	0.5828	0	71188
<i>May-September Daytime</i>	2.1101	0.5602	0	34280

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Reply to Referee #2

We would like to thank referee #2 for the detailed review of our manuscript and the suggestions that have helped to improve our manuscript. In the following, we will answer each of the referee's comments.

The authors compared NEE and biophysical factors between a “natural” grassland site and a “degraded” grassland site in the semiarid area of southeast Spain. They found that the “degraded” site showed less carbon uptake during the growing season but substantially more carbon release during the dry summer months. They attributed the inter-site differences in NEE to higher belowground CO₂ concentration at 1.5-m depth and stronger subterranean ventilation at the degraded site. The reported temporal patterns of NEE, ET and EVI at the two sites could promote a better understanding of the effects of land degradation on carbon sequestration in semiarid areas, and provide important information on ecosystem resilience and vulnerability under changing climate. However, I have some concerns regarding how the authors analyzed and interpreted their data.

General comments

1. The authors concluded that “subterranean ventilation of this vadose zone CO₂ largely drives the differences in C dynamics between them”. This conclusion is based on authors' analyses that compare many biophysical factors between the two sites. It turned out that belowground CO₂ concentration at 1.5-m depth differed the most between sites. In addition, they found a negative correlation between air pressure and subsoil CO₂. However, the reasoning behind this conclusion should be viewed with great caution. Large differences in subsoil CO₂ does not necessarily explain inter-site differences in NEE. There was no analysis showing a causal link between inter-site variations in subsoil CO₂ concentration and CO₂ fluxes. Although the authors examined many potential explanatory variables, there still could be other biophysical factors and processes that differ greatly between sites (e.g., soil microbial communities).

Although we did not test the direct link between subsoil CO₂ molar fraction at 1.5m and net CO₂ exchange fluxes, our results lead to this hypothesis, since inter-site differences in meteorological variables are minimal. On the other hand, although measurements of the metabolic activity of microbial communities would help to understand inter-site differences in heterotrophic respiration patterns, we think that

a potential difference in soil microbial communities should not totally explain the important carbon release observed in the “degraded” site during the dry season, mainly due to a lack of water in the soil at both sites. Apart from that, a previous study performed at Amoladeras relates this carbon release to atmospheric turbulence through linear regressions with friction velocity (u_*), and also shows an influence of net radiation and VPD in the CO_2 fluxes (López-Ballesteros et al., 2017). Overall, based on other investigations performed by this research group in several ecosystems located in the same province (Kowalski et al., 2008; Sanchez-Cañete et al., 2011; Serrano-Ortiz et al., 2009; Sánchez-Cañete et al., 2013; Pérez-Priego et al., 2013), we strongly believe that there is an advective transport of stored CO_2 -rich air from the vadose zone to the atmosphere, especially under high hydric stress and high turbulence conditions.

Apart from this, we have included in Table 4, the linear regression results for subsoil CO_2 at 1.5m and net CO_2 exchange fluxes, and as expected, a higher correlation was obtained in Amoladeras.

Table 4: Spearman correlation coefficients (r_s) for every paired simple correlation among maximum quality net CO_2 exchange fluxes ($\mu mol m^{-2} s^{-1}$), absolute and differential pressure (hPa) at 6, 12, 24 and 72hours time-step and absolute and differential CO_2 molar fraction measured at 1.50 m below ground (ppm) at the same time-steps. Bold values represent the highest correlation coefficients while shaded ones denotes non-significant relationships (p -values >0.05).

	Amoladeras						Balsa Blanca					
	Net CO_2 exchange	P	dP _{6h}	dP _{12h}	dP _{24h}	dP _{72h}	Net CO_2 exchange	P	dP _{6h}	dP _{12h}	dP _{24h}	dP _{72h}
$CO_{21.50m}$	0.30	-0.66	-0.33	-0.46	-0.56	-0.55	0.11	-0.33	-0.51	-0.53	-0.62	-0.45
$dCO_{21.50m_6h}$	0	-0.1	-0.84	-0.5	-0.26	-0.01	0.02	-0.07	-0.63	-0.19	-0.13	-0.04
$dCO_{21.50m_12h}$	0.06	-0.08	-0.57	-0.87	-0.55	-0.05	0.03	-0.03	-0.46	-0.50	-0.31	-0.03
$dCO_{21.50m_24h}$	0.03	-0.13	-0.47	-0.78	-0.85	-0.15	0.02	-0.04	-0.40	-0.59	-0.58	-0.04
$dCO_{21.50m_72h}$	0	-0.28	-0.28	-0.49	-0.64	-0.74	0.00	-0.13	-0.28	-0.43	-0.57	-0.48

2. An unanswered question related to the previous comment is why the degraded site showed such large subsoil CO_2 concentrations compared to the natural site. In addition, the degraded site (with much less vegetation cover) showed more carbon release than the natural site. Subterranean ventilation is only a transport process for

CO₂, but the question is who produced so much CO₂? Was it abiotic processes related to carbonate dissolution, or respiratory CO₂ production by plants and microbes? This question must be discussed in the paper.

As explained by López-Ballesteros et al. (2017), the potential origins of the released CO₂ could be geological degassing and/or subterranean translocation of CO₂ in both gaseous and aqueous phases. In this publication, there is a detailed argument of these two hypotheses and also it is recognized that future research is needed in order to understand how CO₂ transport and production processes interact and modulate drylands' terrestrial C balance.

3. The authors only examined NEE dynamics. I would encourage them to also partition NEE and check the two major components of NEE: GPP and Reco. These two components may respond differently to land degradation and interannual climatic variations. Separate analysis on GPP and Reco could provide more information on the differences between the two sites in terms of carbon dynamics.

In accordance to the comment of the referee #1 and your suggestion, we have performed the flux partitioning proposed by Lasslop et al.(2010) in order to estimate the magnitude of GPP and R_{eco} for both sites. However, in the case of Amoladeras, we first subtracted the flux magnitude corresponding to subterranean ventilation by applying a ventilation model (Pérez-Priego et al., 2013). The methodology used as well as the results are explained in the reply to referee #1, and would be included in the revised version of the manuscript.

4. Table 1 showed that the vegetation cover almost three times higher at the natural site than at the degraded site, while the EVI in Figure 5 does not show such a large difference, at least for most years. So I am wondering whether the pixels you used for extracting EVI well match the location of your ground measurements. Or are there any other reasons for this discrepancy?

We have verified that the chosen pixels match the location of our ground measurements. Since referee #1 made the same remark, please read the answer to the fourth general comment of the reply to referee #1.

Specific comments

1. Page 1, line 16-20. The background information is a bit too long.

We have modified these sentences as follows: "Currently, drylands occupy more than one third of the global terrestrial surface and are recognized as areas vulnerable to land degradation. The concept of land degradation stems from the loss

of an ecosystem's biological productivity, due to long-term loss of natural vegetation or depletion of soil nutrients."

2. Page 1, line 21-24. The sentence is too long and complex. Considering dividing it into shorter sentences.

We have rewritten this sentence as follows: "In the present study, we compare net carbon C and water vapor fluxes, together with meteorological and satellite data and vadose zone measurements (CO₂, water content and temperature) between two nearby (~23 km) experimental sites representing "natural" (i.e. site of reference) and "degraded" grazed semiarid grasslands. We utilized data acquired in two eddy covariance stations located in SE Spain during 6 years with highly variable precipitation magnitude and distribution."

3. Page 1, line 20. In "global C balance", symbols should be defined upon first mentioning.

We have defined the symbol at this sentence.

4. Page 1, line 21. Replace "needs further research" by "still need to be investigated".

We have followed your suggestion.

5. Page 1, line 25. Please specify what "_" stands for.

We did not find that character in the sentence.

6. Page 2, line 1-11. This paragraph is a bit too long. The importance of drylands has been well acknowledged and should only be mentioned very briefly here.

We have shortened this paragraph as follows: "The concept of land degradation stems from the loss of an ecosystem's biological productivity, which in turn relies on several degradation processes such as long-term loss of natural vegetation, deterioration of soil quality, depletion in biodiversity or water and wind erosion (UNCCD, 1994). Drylands (arid, semiarid and dry sub-humid areas), which occupy more than one third of Earth's land surface and are inhabited by more than 2 billion people (Niemeijer et al., 2005), have been recognized as areas vulnerable to land degradation processes. In fact, they have expanded globally for the last sixty years at an estimated annual rate of 5.8 million hectares in mid latitudes alone (Lal, 2001), and are projected to expand under future climate change scenarios (Feng and Fu, 2013; Cook et al., 2014), especially in the Mediterranean region, where major expansions of semiarid areas will occur (Gao and Giorgi, 2008; Feng and Fu, 2013)."

7. Page 2, line 14-22. These case studies are not directly related to this paper. The first sentence of this paragraph already well summarizes the subject of research. I would delete or reduce these case studies.

We have reduced the case studies as suggested. We want to mention regions or countries where desertification has been assessed previously in order to assess the spatial representativeness of land degradation research globally. However, we have followed your suggestion by deleting the last sentence of this paragraph which talks about the global studies using modelling approaches, since our study is local.

8. Page 3, line 12-14. I would delete this sentence since the EC technique is a widely used method, and is familiar to most researchers working on carbon exchange.

We agree and have deleted that sentence.

9. Page 3, line 16. What did you mean by “absorb fast changes”?

Here, we are talking about resilience, hence we have rewritten this sentence as follows: “Owing to the high temporal resolution of the EC method, we can assess the effect of land degradation as a slow change or disturbance legacy in the studied ecosystems and how, in turn, it influences ecosystem resilience to short-term disturbances, such as climate extremes (i.e. droughts, heat waves).”

10. Page 3, line 25-26. Please specify what kind of “short-term disturbances” you are talking about.

As written before, we meant climate extremes, such as droughts or heat waves.

11. Page 3, line 30. Replace “Experimental sites description” with “Site description”.

Done.

12. Page 4, line 20. The expression “different degradation stages” is not clear to me. More information on the history (degradation, recovery and succession) of the two sites should be provided. A basic question is what caused the degradation?

The stronger degradation effects observed in Amoladeras (“degraded” site) compared to Balsa Blanca (“natural” site) are probably due to its proximity to populated areas. The main factor provoking degradation in this Mediterranean area was the increase of rural population from the beginning of the 20th century until late 1950s (Grove and Rackham, 2001). At that time, timber extraction, the use of tussock fiber for textile manufacturing and extensive farming were common economic activities potentially causing a higher anthropic pressure on the

“degraded” site. Afterwards, rural exodus during the mid-century involved the abandonment of this agriculture and farming practices. However, although degradation drivers are not currently active, their effects are still observable in the area; this is a case of “relict” degradation (Puigdefábregas and Mendizábal, 2004).

We will add this information in the revised version of the manuscript.

13. Page 5, line 6. Please specify what “” stands for.

We did not find that character in the sentence. It seems that the referee has a problem with his/her PDF viewer, particularly regarding the symbol “±”.

14. Page 5, line 8. More details on estimating uncertainty should be provided.

We have rewritten this paragraph as follows: “Missing data were gap-filled by means of the marginal distribution approach proposed by Reichstein et al. (2005) and uncertainty derived from the gap-filling procedure by using the variance of the measured data, which was calculated by introducing artificial gaps and repeating the standard gap-filling procedure. Twice the standard deviation of sums of total data was taken as the uncertainty for the several aggregating time periods we used in the analysis.”

15. Page 5, line 15. Please clarify whether or not you took into account above- and below-ground storage terms of heat fluxes when calculating the slopes?

We have included the following sentence to clarify it: “The storage term in the soil heat fluxes was included in the estimates while in case of sensible and latent heat fluxes, this term was negligible given the short height of the vegetation (~50 cm).”

16. Page 5, line 21-22. I would delete this sentence. As you said, it is a widely used index, so there is no need to justify using it.

Done.

17. Page 6, line 5-9. Please reword the sentence.

We have reworded these sentences as follows: “This test was chosen because the variables used satisfied the independence and continuity assumptions but not all were normally distributed. The confidence level used was 95%. The effect size was evaluated using the median of the difference between the samples (Amoladeras minus Balsa Blanca), which was expressed as a standardized value (divided by its standard deviation; $Diff_{st}$; dimensionless) in order to be able to compare results among different variables.”

18. Page 6, line 18. Delete “over the study period”.

Done.

19. Page 6, line 20. The term “annual average precipitation” should be changed to “mean annual precipitation (MAP)”.

Done.

20. Page 7, line 5. Please delete “(C)” as you have defined it in Introduction.

Done.

21. Page 8, line 20-25. I would shorten or remove these sentences since interannual variability is not the focus of this study.

We have shortened this paragraph as follows: “On the other hand, differences in the inter-annual variability of EVI were found between years. Concretely, 2009/2010 and 2013/2014 were the years with maximum and minimum annual precipitation and EVI observations, respectively, for both sites. In 2009/2010, EVI observations were 28% and 20% higher than the six-year averaged values in BB and AMO, respectively. In case of the driest year, 2013/2014, the growing season (winter-spring) EVI was reduced 35% and 28% in BB and AMO, respectively.”

22. Page 9, line 3. Why did you use a threshold value of 1 for Diffst.

We chose that threshold just to comment the data, all results are shown in Tables 4 and 5, and therefore they can be interpreted by the readers.

23. Page 11, line 25. By saying “stable” did you mean “resilient”?

Yes. We have substituted “stable” with “resilient”.

24. Page 11, line 27. No need to give the definition of “ecosystem resilience”. It is a textbook concept that everyone knows.

We have removed the definition of “ecosystem resilience”.

25. Figure 1. It would be nice if you can add some photos of landscape or vegetation at the two sites.

We agree and we would include the picture below.



26. Table 1. The first part of the table (site characteristics) can be removed. These characteristics were well-described in the text and are therefore redundant here. What was the measurement depth for SOC?

We agree so we have deleted the first part of Table 1. Regarding the SOC measurement, given that these soils are shallow with maximum depth of 20 cm, we only took a composite sample of the profile.

27. Figure 3. Adding cumulative GPP and Reco may help understand inter-site differences in carbon dynamics.

Given the lack of flux data in both sites, instead of adding a figure of cumulative GPP and R_{eco} , we have added the following figure showing the monthly cumulative NEE (biological), GPP, R_{eco} and Water Use Efficiency (WUE) for both sites over the whole study period.

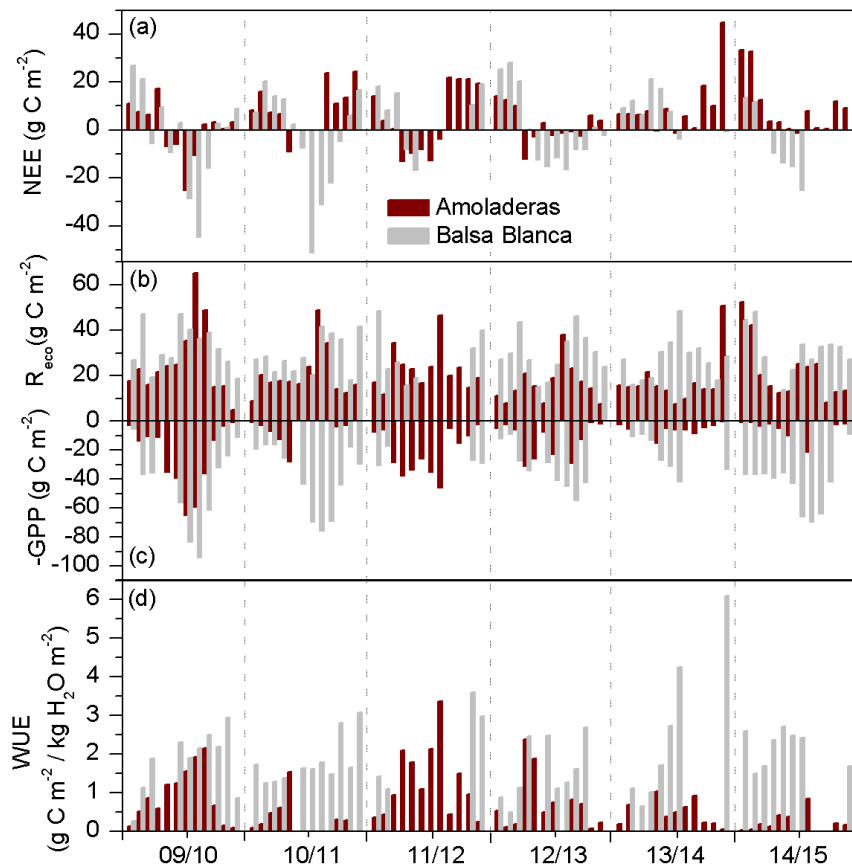


Figure 7: Monthly cumulative fluxes of (a) biological net ecosystem CO_2 exchange, (b) ecosystem respiration (R_{eco}), (c) negative gross primary production and (d) water use efficiency over the six hydrological years of study (2009-2015) for Amoladeras (dark red) and Balsa Blanca (grey). Lacking bars correspond to long-term data losses.

28. Table 2. I would put this table in Supplementary Online Materials.

We believe that this information is crucial to interpret and compare these results with other studies. Hence, we do not think that it should be placed in the Supplementary Online Materials.

29. Table 6. This table is confusing to me. Can you explain, for example, what does “ $dCO_2, 1.5m_6h$ ” mean?

$dCO_2, 1.5m_6h$ means the difference between the CO_2 molar fraction measured at a time “ t ” and the CO_2 molar fraction measured 6 hours before “ $t-6h$ ”. This analysis was performed to show how different is the influence of pressure variations in the subsoil CO_2 molar fraction between sites, as demonstrated at another study in the area (Sánchez-Cañete et al., 2013).

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Can land degradation drive differences in the C exchange of two similar semiarid ecosystems?

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Abstract. ~~Currently, drylands occupy more than one third of the global terrestrial surface and are recognized as areas vulnerable to land degradation. The concept of land degradation stems from the loss of an ecosystem's biological productivity, due to long-term loss of natural vegetation or depletion of soil nutrients. The concept of land degradation stems from the loss of an ecosystem's biological productivity, which in turn relies on several degradation processes, such as long-term loss of natural vegetation, depletion of soil nutrients, soil compaction or water and wind erosion, to which drylands are especially vulnerable. Currently, drylands occupy more than one third of the global terrestrial surface and will probably expand under future climate change scenarios.~~ Drylands' key role in the global ~~C~~-carbon (C) balance has been recently demonstrated, but the effects of land degradation on C sequestration by these ecosystems ~~still need to be investigated~~~~needs further research~~. In the present study, we compare net ~~carbon-C and water vapour exchange~~fluxes, together with satellite, ~~data and~~ meteorological, ~~ambient~~ and vadose zone (CO₂, water content and temperature) ~~variables~~measurements, between two nearby (~23 km) experimental sites representing "natural" (i.e. site of reference) and "degraded" grazed semiarid grasslands ~~located in SE Spain~~. We utilized ~~via data acquired from two~~ eddy covariance ~~stations located in SE Spain~~ measurements ~~overduring~~ 6 years, with highly variable precipitation magnitude and distribution. Results show a striking difference in the annual C balances with an average release of 196 ± 40 and -23 ± 20 g C m⁻² yr⁻¹ for the "degraded" and "natural" sites, respectively. At the seasonal scale, differing patterns in net CO₂ fluxes were detected over both growing and dry seasons. As expected, during the growing seasons, greater net C uptake over longer periods was observed ~~at~~ the "natural" site, however, much greater net C release, ~~probably derived from subterranean ventilation~~, was measured ~~in~~ at the "degraded" site during drought periods. ~~After subtracting the non-biological CO₂ flux from net CO₂ exchange, flux partitioning results point out that, during the six years of study, gross primary production, ecosystem respiration and water~~

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use efficiency were, on average, nine, twice and ten times higher, respectively, at the “natural” site versus the “degraded” site. We also tested differences in all monitored meteorological and soil variables and found it most relevant that CO₂ at 1.50 m belowground was around 1000 ppm higher in the “degraded” site. Thus, we believe that subterranean ventilation of this vadose zone CO₂, previously observed at both sites, largely drives the differences in C dynamics between them, especially during the dry season maybe due to enhanced subsoil-atmosphere interconnectivity in the “degraded” site. Overall, the 12 site-years of data allow direct exploration of the roles of climate and land degradation in the biological and non-biological processes that ultimately control the C sequestration capacity of semiarid ecosystems.

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1 Introduction

~~The concept of land degradation stems from the loss of an ecosystem’s biological productivity, which in turn relies on several degradation processes such as long term loss of natural vegetation, deterioration of soil quality, depletion in biodiversity or water and wind erosion (UNCCD, 1994). Arid, semiarid and dry sub-humid areas, commonly known as drylands, have been recognized as areas vulnerable to land degradation processes. Overall, drylands occupy more than one third of Earth’s land surface and are inhabited by more than 2 billion people (Niemeijer et al., 2005), and concretely, the semiarid ecoregion comprises a major fraction, 37% of global dryland area, where the dominant biome is grassland (Niemeijer et al., 2005). Drylands have expanded globally for the last sixty years at an estimated annual rate of 5.8 million hectares in mid-latitudes alone (Lal, 2001), and are projected to expand under future climate change scenarios (Feng and Fu, 2013; Cook et al., 2014). Furthermore, the Mediterranean region has been recognized as an especially vulnerable region (Gao and Giorgi, 2008), where major expansions of semiarid areas will occur (Feng and Fu, 2013).~~

The concept of land degradation stems from the loss of an ecosystem’s biological productivity, which in turn relies on several degradation processes such as long-term loss of natural vegetation, deterioration of soil quality, biodiversity depletion or water and wind erosion (UNCCD, 1994). Drylands (arid, semiarid and dry sub-humid areas), which occupy more than one third of Earth’s land surface and are inhabited by more than 2 billion people (Niemeijer et al., 2005), have been recognized as areas vulnerable to land degradation processes. In fact, they have expanded globally for the last sixty years at an estimated annual rate of 5.8 million hectares in mid latitudes alone (Lal, 2001), and are projected to expand under future climate change scenarios (Feng and Fu, 2013; Cook et al., 2014), especially in the Mediterranean region, where major expansions of semiarid areas will occur (Gao and Giorgi, 2008; Feng and Fu, 2013).

Over recent decades, most research focused on land degradation has been based on remote sensing and earth observation techniques. Much of these investigations have aimed to refine methodological issues in order to accurately track land degradation in vulnerable areas, reduce uncertainties and explain inconsistencies among studies. For instance, a wide array of satellite-derived data, such as vegetation indices, normalized surface reflectance, brightness temperature or biomass-net primary production derivatives (Mbow et al., 2015), has been utilized to appraise desertification effects in the Sahel (Mbow et al., 2015; Fensholt et al., 2013) and also in other African countries such as Kenya (Omuto, 2011), Somalia (Omuto et al.,

2010), South Africa (Thompson et al., 2009) or Zimbabwe (Prince et al., 2009). Likewise, desertification in the Mediterranean region has been studied through satellite imagery, ~~concretely,~~ in Greece (Bajocco et al., 2012), Israel (Shoshany and Karnibad, 2015) and the Iberian Peninsula (del Barrio et al., 2010). ~~Additionally, some global assessments based on modelling approaches have focused on the global loss of net primary production derived from land degradation (Zika and Erb, 2009) or the effect of desertification on climate (Wang et al., 2016).~~ However, although drylands' key role in the global carbon (C) balance has been demonstrated (Poulter et al., 2014; Ahlström et al., 2015), very few investigations have directly quantified how land degradation processes disturb the C sequestration capacity of drylands (Lal, 2001), ~~which despite being is~~ one of the most important ecosystem services (Watanabe and Ortega, 2011).

In this regard, the few C-related desertification studies conducted over last decade have centred on soil C dynamics. Concretely, soil organic carbon (SOC) inventories have been used to explore the effects of climate, human activities and grazing pressure in desertification-prone areas of China (Feng et al., 2006) and Brazil (Schulz et al., 2016). Similarly, other investigations have evaluated soil degradation processes by means of soil CO₂ effluxes together with other biometric measurements in drylands found in China (Hou et al., 2014; Wang et al., 2007), Chile (Bown et al., 2014) and southeast Spain (Rey et al., 2011; Rey et al., 2017). However, the degradation processes associated with desertification affect several subsystems as well as their interactions at multiple spatial and temporal scales. For instance, adverse effects on soil quality involve depletion of soil fertility, but also reduce soil-water storage (Mainguet and Da Silva, 1998), which, in turn, can constrain seed germination and vegetation reestablishment, modify climax vegetation, disrupt biogeochemical cycles, alter water and energy balances, and consequently lead to a loss of ecosystem resilience (Lal, 2001). This cascade of disturbances may result in a reduction of the C sequestration capacity of a given ecosystem, which is clearly a symptom of the loss of biological productivity, resulting in a positive feedback to global warming. Therefore, a quite suitable and holistic approach is to integrate all subsystems effects into a whole ecosystem-scale assessment when quantifying the C loss derived from land degradation. However, the use of this integrative method is mostly lacking in the available literature.

The present study is located in an area, the southeast of Spain, that has been recognized as a hotspot of land degradation owing to the synergistic interaction of sociological and climatic factors (Puigdefábregas and Mendizabal, 1998). ~~where, furthermore, rainfall has trended downward at ca. 3 mm yr⁻¹ since 1900 (Oñate Rubalcaba, 1993).~~ Our core aim is to evaluate how dryland degradation affects the dynamics of net ecosystem-atmosphere C exchange of two semiarid grasslands that represent differing degradation status ("natural" *versus* "degraded") by means of meteorological ~~and,~~ satellite and subsoil CO₂ measurements, subsoil CO₂ sensors and together with carbon and water fluxes acquired by the eddy covariance (EC) technique (Baldocchi et al., 1988). ~~This technique allows us to directly quantify the net exchange of energy, water and CO₂ flux densities (hereinafter fluxes) between an ecosystem (including all of its subsystems, i.e. soil, plants, microorganism, vadose zone...) and the atmosphere at a high frequency (from minutes to hours).~~ Owing to the high temporal resolution of the EC method, we can assess the effect of land degradation as a slow change or disturbance legacy in the studied ecosystems and how, in turn, it influences the capacity of these ecosystems' resilience to against absorb fast changes ~~or~~ short-term disturbances, such as droughts climate extremes (i.e. droughts, heat waves).

Accordingly, our main hypothesis is that the degree of land degradation affects net ecosystem C sequestration capacity through its impact on some or all of the processes that compose the overall ecosystem C balance: photosynthesis, ecosystem respiration (biological processes) and subterranean ventilation—a non-biological process that provokes the transfer of CO₂-rich air from subsoil to atmosphere under drought and high turbulence conditions. Some land degradation processes are evident when we compare the “natural” site with the “degraded” site. Accordingly, our main hypothesis is that land degradation processes can directly affect abiotic and/or biotic factors and, consequently, influence the biological and/or non-biological processes that compose the net ecosystem CO₂ exchange: gross primary production, ecosystem respiration (biological processes) and subterranean ventilation – a non-biological process that provokes the transfer of CO₂-rich air from subsoil to atmosphere under drought and high turbulence conditions. Firstly, the lower vegetation cover in the “degraded” site would entail a higher thermal and radiative stress at the surface, especially during the drought period (Rey et al., 2017). The hypothesized effects on biological processes are a direct reduction in plant productivity and respiration, and an indirect decrease in heterotrophic respiration. Secondly, the higher cover of bare soil and outcrops in the “degraded” site may increase the soil-atmosphere interconnectivity, which indirectly can enhance the presence of advective CO₂ release through subterranean ventilation, which has been previously measured at both experimental sites (Rey et al., 2012; López-Ballesteros et al., 2017). And thirdly, the reduced soil fertility and depth may provoke changes in microbial communities (Evans and Wallenstein, 2014) due to stronger nutrient and water limitations. Consequently, a direct decrease in heterotrophic respiration and plant productivity and respiration is expected. In this context, as subterranean ventilation has been measured in both experimental sites (Rey et al., 2012; López Ballesteros et al., 2017), we also hypothesize a significant role of this process in the monitored ecosystems.

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Hence, our specific objectives are: (1) to compare the C sequestration capacity of two semiarid ecosystems showing a with differing degradation status, (2) to study the underlying involved processes (biological versus non-biological) and influencing factors that can drive potential differences in the net C exchange of studied ecosystems, and (3) to evaluate whether degradation can modulate ecosystem responses against short-term disturbances. To do this, we analysed 12 site-years of EC data, Enhanced Vegetation Index (EVI) time series and monitored ambient variables registered over the same period (2009-2015) at both sites. Additionally, we used subsoil CO₂, moisture and temperature data obtained during 2014-2015.

2 Material and Methods

2.1 Experimental sites description

The study area is located in southeast Spain, the driest part of Europe. The two experimental sites, Amoladeras (N36°50'5''W2°15'1'') and Balsa Blanca (N36°56'26.0'' W2°01'58.8''), are found within the Cabo de Gata-Níjar Natural Park (Almería, Spain; Fig. 1) and are quite similar in terms of climate and ecosystem type. Both sites show a desert climate, according to Köppen classification (Bwh; Kottek et al., 2006), with a mean annual temperature of 18°C, and mean annual precipitation of approximately 220 mm.

The ecosystem type corresponds to *espartal*, a Mediterranean semiarid grassland where the dominant species is *Machrocloa tenacissima*. This ecosystem type is widely extended over the Western Mediterranean region ~~;~~ ~~in fact~~, in Cabo de Gata-Níjar Natural Park, a great fraction of agricultural areas that were abandoned over 1957-1994 resulted in *espartal* ecosystems (Alados et al., 2011; Alados et al., 2004). The functioning of both experimental sites can be divided into two main periods.

5 On one hand, the growing season usually extends from late autumn to early spring, when the temperature starts to rise and water resources have not yet become scarce (López-Ballesteros et al., 2016; Serrano-Ortiz et al., 2014). On the other hand, a long period of hydric stress, with high temperatures and scarce precipitation, results in a prolonged dry season that usually begins in May-June and ends in September-October, when the first autumn rainfall events occur. Additionally, water inputs derived from relevant dewfall episodes, which have been previously reported in the area (Uclés et al., 2014) ~~in the area~~, can rehydrate soil and plants during night and early morning hours.

10 Regarding the topographic, geologic and edaphic characteristics, both sites are located on an alluvial fan, where the main geological materials consist of quaternary conglomerates and Neogene-Quaternary sediments cemented by lime (caliche) (Rodríguez-Fernández et al., 2015) ~~and on~~ slopes of 2-6% (Rey et al., 2017) so no significant runoff occurs. Additionally, both sites present petrocalcic horizons. However, altitude and soil type differ. While Balsa Blanca (hereinafter BB) is located
15 at an altitude of 208 m and has MollicLithic Leptosols (Calcaric), Amoladeras (hereinafter AMO) is situated closer to sea level, at 65 m, and presents less developed soils Lithic Leptosol (Calcaric; Table 1).

Overall, as stated by Rey et al. (2011), these two experimental sites represent different degradation stages owing to their differing soil characteristics and surface fractions (Table 1). While BB has more deep and fertile soils and higher vegetation cover, AMO shows thinner and poorer soils and has half of Balsa Blanca's vegetation cover. Therefore, in accordance ~~to~~
20 with Rey et al. (2011; 2017), we considered that BB represents the “natural” site, being, currently, a representative ecosystem of the area, while AMO represents a “degraded” site with respect to BB. The stronger degradation effects observed in AMO (“degraded” site) compared to BB (“natural” site) are probably due to its proximity to populated areas. The main factor provoking degradation in this Mediterranean area was the increase of rural population from the beginning of the 20th century until late 1950s (Grove and Rackham, 2001). At that time, timber extraction, the use of tussock fiber for textile manufacturing and extensive farming were common economic activities likely increasing anthropic pressure on the “degraded” site. Afterwards, rural exodus during the mid-century involved the abandonment of these agriculture practices. However, although degradation drivers are not currently active, their effects are still observable in the area corresponding to a case of “relict” degradation (Puigdefábregas and Mendizábal, 2004).

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2.2 Meteorological and eddy covariance measurements

30 The net ecosystem-atmosphere exchange of water vapour, CO₂ and sensible and latent heat were measured in terms of fluxes via the eddy covariance (EC) technique. Thus, an EC station was installed at each experimental site, AMO and BB (with site codes “Es-Amo” and “Es-Agu” of the European Database Cluster <http://www.europe-fluxdata.eu>), where ambient and micrometeorological variables (detailed in Table 2) were monitored continuously since 2009. The EC footprint (i.e. actual

measured area) is well within the fetch (i.e. distance to a change in surface characteristics) at both sites. Regarding data processing, the half-hourly averaged fluxes were calculated from raw data collected at 10 Hz using EddyPro 5.1.1 software (Li-Cor, Inc., USA). Flux calculation, flux corrections and quality assessment were performed according to López-Ballesteros et al. (2016).

5 Additionally, flux measurements acquired under low-turbulence conditions were excluded from the analysis by using a friction velocity (u_*) threshold according to the approach proposed by Reichstein et al. (2005). The average u_* thresholds for the whole study period (i.e. 2009–2015) were 0.11 and 0.16 m s^{-1} , for AMO and BB, respectively. Furthermore, over the six years of measurements at both sites, data gaps due to low-turbulence conditions, instrument malfunction and theft were unavoidable and not randomly distributed, as noted by Ma et al. (2016). Therefore, the total annual fractions of missing half-
10 hourly net CO_2 fluxes accounted for $33 \pm 3 \%$ and $29 \pm 6 \%$ of night-time data and $8 \pm 6 \%$ and $14 \pm 5 \%$ of day-time data, for AMO and BB, respectively. Missing data were gap-filled by means of the marginal distribution approach proposed by Reichstein et al. (2005) and ~~errors-uncertainty~~ derived from the gap-filling procedure ~~were-was~~ calculated by using the variance of the measured data, which was calculated by introducing artificial gaps and repeating the standard gap-filling procedure. Twice the standard deviation of sums of total data was taken as the uncertainty for the several aggregating time periods we used in the analysis, from the variance of the gap-filled data, as explained by López-Ballesteros et al. (2016). The annual cumulative C balance was estimated, when possible, by integrating gap-filled half-hourly net CO_2 fluxes of good quality (0 and 1 quality flags, according to Mauder and Foken, 2004) over a hydrological year.
15 In order to test the validity of both EC stations, we assessed the energy balance closure (Moncrieff et al., 1997) by computing the linear regression of half-hourly turbulent energy fluxes, sensible and latent heat fluxes ($H+LE$; W m^{-2}) against available energy, net radiation less the soil heat flux (R_n-G ; W m^{-2}) with the whole six-years database. Storage term in the soil heat flux was included in the estimates while in case of sensible and latent heat fluxes, this term was negligible given the short height of the vegetation (~50 cm). The resulting slopes were 0.873 ± 0.002 ($R^2 = 0.907$) and 0.875 ± 0.001 ($R^2 = 0.920$) for AMO and BB, respectively.

2.3 Flux partitioning

25 In order to partition net CO_2 ecosystem exchange into Gross Primary Production (GPP) and ecosystem respiration (R_{eco}), we firstly modelled the ventilative CO_2 efflux by adapting the approach proposed by Pérez-Priego et al. (2013) with the results of previous studies of both sites (López-Ballesteros et al., 2016; 2017). Essentially, we aimed to isolate those moments when subterranean ventilation (V_v) dominates the net CO_2 fluxes (F_v) and biological fluxes are negligible. These moments correspond to daytime hours during the extremely dry periods. Accordingly, data were selected using the following
30 conditions: (i) net radiation $> 10 \text{ W m}^{-2}$, (ii) $8 < \text{daily averaged bowen ratio} < 10$, and (iii) daily soil water content (in bare soil) $< 10^{\text{th}}$ percentile (in AMO) and $< 20^{\text{th}}$ percentile (in BB). A less restrictive threshold was used in BB in order to get enough data to build the V_v model, since long-term data gaps occurred at this site during the summer seasons of 2012, 2014

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2.6-7 Statistical analysis

All meteorological and soil variables monitored at each site were compared through computation of the non-parametric two-sided Wilcoxon summed rank test in order to detect those factors/variables influencing potentially distinct ecosystem functioning between sites. ~~This test was chosen because variables used satisfied the independence and continuity assumptions but not all were normally distributed. The confidence level used was 95%. The effect size was evaluated using the median of the difference between the samples (AMO minus BB), which was expressed as a standardized value (divided by its standard deviation: $Diff_{st}$; dimensionless) in order to be able to compare results among different variables. This test was chosen because variables used satisfied the independence and continuity assumptions but not all were normally distributed. The confidence level used was 95% and the effect size was evaluated using an estimator based on confidence intervals that corresponded to the median of the difference between the samples (Amoladeras minus Balsa Blanca) and was expressed as an absolute value (Diff) and as a standardized value ($Diff_{st}$; dimensionless), which was obtained by using the sample values of every site divided by its standard deviation in order to be able to compare results among different variables.~~ This analysis was performed by using three different periods: the whole study period, the period from May to September and the period from May to September during only daytime. These periods were selected given their demonstrated coincidence with high relevance of non-biological processes. All calculations were performed using R software version 3.2.5.

Additionally, in order to include the relationship between pressure and subsoil CO₂ variations as a potential factor influencing net CO₂ exchange (Sánchez-Cañete et al., 2013), we firstly calculated, separately for each site, Spearman correlation coefficients to determine the time step (6, 12, 24 or 72 hours) with the highest correlation between the differential transformation of pressure and the subsoil CO₂ molar fraction at 1.50 m.

3 Results

3.1 Ambient conditions ~~over the study period~~

Over the study period, the wettest hydrological year was 2009/2010, with annual precipitation of ~500 mm (ca. twice the ~~mean~~ annual ~~average~~ precipitation for both sites over the study period, Fig. 2). On the contrary, the driest year was 2013/2014, with annual precipitation of ~100 mm for both sites, less than half the annual average precipitation registered at ~~Amoladeras (AMO) and Balsa Blanca (BB)~~. Generally, ~~the~~ months with precipitation ~~higher than~~ exceeding 20 mm occurred from the beginning of autumn until midwinter, however, in case of 2009/2010, 2010/2011, 2012/2013 and 2014/2015, relevant ~~precipitation-rain~~ events were registered during spring months. By contrast, in 2013/2014, precipitation was always below 20 mm with the exception of November and December, for both sites, and June, in the case of AMO (Fig. 2a). Commonly, while maximum precipitation usually occurred from November to February, there was a remarkable drought period over summer months (June-August) when it scarcely ever rained (Fig. 2).

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Regarding air temperature (T_{air}) patterns, monthly averaged T_{air} ranged from 9.6 and 8.1 °C to 27.6 and 27.9 °C in AMO and BB, respectively, over the entire study period. Based on half-hourly averaged data, minimum and maximum T_{air} values registered were 0.1 and 37.9 °C in AMO, and -1.3 and 39.9 °C, in BB, respectively. On one hand, those months with T_{air} above 15 °C usually corresponded to April-November, approximately. Additionally, August was the month with the highest average T_{air} at both sites, with T_{air} ranges of 25.2 - 27.6 °C at AMO and 24.9 - 27.9 °C at BB, respectively (Fig. 2), over the study period. On the other hand, the lowest monthly average T_{air} usually occurred in January but sometimes also in December and February, with 11.2 – 12.3 °C at AMO and 8.1 - 14.1 °C at BB.

3.2 Annual carbon balances

The comparison of the annual ~~carbon (C)C~~ balance among sites was only possible for three hydrological years, 2009/2010, 2010/2011 and 2012/2013, due to long-term data gaps existing in BB during other years. The annual cumulative net CO₂ exchange was always positive for AMO (i.e., net C release), whereas BB was neutral or even acted as a C sink over the three years (Fig. 3). For example, in 2009/2010, the net C uptake measured in BB equated to $32 \pm 10 \text{ g C m}^{-2}$ while in AMO, a total amount of $185 \pm 10 \text{ g C m}^{-2}$ was released to the atmosphere (Fig. 3a). The year with the largest difference between sites was 2010/2011, with annual C release of 240 ± 8 and $-38 \pm 10 \text{ g C m}^{-2}$ in AMO and BB, respectively (Fig. 3b). Likewise, ~~2011/2012~~2012/2013 was the year when the lowest CO₂ release was measured in AMO with $163 \pm 7 \text{ g C m}^{-2}$ while a neutral C balance was measured in BB with $0 \pm 8 \text{ g C m}^{-2}$ (Fig. 3c).

Overall, a positive and saturating trend was observed at both sites during autumn months until December-February when cumulative net CO₂ releases start to decline. The autumn net CO₂ release (i.e., positive values) was usually higher in AMO than in BB, excepting for 2012/2013, and the declining slope was always higher in BB, meaning greater net C uptake rates. Although the pattern of the cumulative net CO₂ exchange showed differences between sites over autumn, winter and spring months, stronger discrepancies were found during summer droughts. Concretely, from April-May until August, BB showed neutral behavior while a remarkable positive trend was observed in AMO, denoting a large net CO₂ release.

3.3 Seasonal and diurnal net CO₂ exchanges

Long-term data loss occurred in BB during the springs of 2011/2012, 2013/2014 and 2014/2015 and summers of 2013/2014 and 2014/2015, when annual C balances could not be estimated. However, by observing the available seasonal data, it is noticeable that, maximum and minimum seasonal net CO₂ exchanges were very different between sites (Fig. 4). On one hand, maximum seasonal net CO₂ uptake was measured during winter (December-February) in AMO and over spring (March-May) in BB, when peaking net CO₂ uptake fluxes equated to -31 g C m^{-2} (winter 2011/2012) and -105 g C m^{-2} (spring 2010/2011) in AMO and BB, respectively. Additionally, net CO₂ uptake was only observed during three winters in the case of AMO, whereas it was frequently measured during both winter and spring in BB. On the other hand, cumulative net CO₂ release to the atmosphere occurred over all ~~types of~~ seasons in AMO, but acutely in summer, when maximum

seasonal net CO₂ release was always observed ranging from 111 to 153 g C m⁻². In contrast, in BB, the highest CO₂ effluxes usually occurred in autumn ranging from 25 to 74 g C m⁻², although significant CO₂ release was also observed in winter 2013/2014 and the summers of 2009/2010-2011/2012. ~~Regarding seasonal evapotranspiration (ET) fluxes, results showed a ~30% higher ET at BB, compared to AMO, during spring. Major inter-site differences in autumn occurred in the first and last year of study, when ET was 23% and 12% higher at BB, respectively (Fig. 5).~~

Comparing daily-scale net CO₂ exchange and ~~evapotranspiration (ET)~~ET fluxes with ~~punctual~~ Enhanced Vegetation Index (EVI) data, we can notice some similarities in the general patterns of both sites (Fig. 56). Roughly, there was a common annual pattern ~~at both sites~~ in which the highest values of EVI coincided with maximum net CO₂ uptake rates (i.e. negative net CO₂ fluxes), which in turn, corresponded to peaking ET fluxes. Additionally, a decreasing trend in EVI over the 6 years of study was also noticeable for both sites. However, some inter-site and inter-annual differences were evident (Fig. 56).

On one hand, there were two main differences between sites. Firstly, extreme net CO₂ release was measured uniquely in AMO during summer months (June-August), when maximum net CO₂ fluxes ranging from 31 to 68 g C m⁻² were measured (Fig. 56b). Over the study period, the monthly net CO₂ exchange of AMO during dry seasons was up to one hundred times higher than in BB (in August 2013), since monthly net CO₂ fluxes measured in BB were much lower, from -8 to 16 g C m⁻² (Fig. 56b). Besides the striking differences in summer net CO₂ exchange between sites, minor discrepancies were also found in ET fluxes and EVI for the same drought periods. In this regard, monthly averaged ET over the dry season equated to 13 ± 4 and 10 ± 4 mm for AMO and BB, respectively, and EVI was on average 4% higher in BB than in AMO (Fig. 56a, c). The second inter-site difference was the greater net CO₂ uptake over longer periods measured in BB. Concretely, the period during which the ecosystems acted as C sinks lasted on average 38 days longer in BB than in AMO annually (Table 34). Accordingly, the annual amount of C fixation ranged from 6-59 g C m⁻² at AMO and 15-129 g C m⁻² at BB, respectively, with the annual averaged net C uptake in BB 162% higher than at AMO (Table 34). Consequently, peaking EVI values were usually observed during March-April for both sites, however, over winter and spring months (growing period), EVI measured at BB was 3 - 37% higher than AMO, with the largest inter-site differences in 2009/2010 and 2014/2015 (Fig. 56a). Likewise, monthly averaged ET fluxes measured at BB over winter and spring months (December-May) were from 3 to 24% larger than those measured at AMO. Additionally, the growing period of the driest year (2013/2014) corresponded to the lowest monthly ET fluxes and the least difference between sites.

On the other hand, differences in the inter-annual variability of EVI ~~were found between years. Concretely, carbon and water fluxes were also observed; however, due to long term data losses at BB, accurate and comparative inter-annual analysis could not be done with carbon and water fluxes. In this sense,~~ 2009/2010 and 2013/2014 were the years with maximum and minimum annual precipitation and EVI observations, respectively, for both sites. ~~Concretely, in~~ In 2009/2010, EVI observations were 28% and 20% higher than the six-year averaged values in BB and AMO, respectively. In case of the driest year, 2013/2014, growing season (winter-spring) EVI was reduced 35% and 28% in BB and AMO, respectively. Nevertheless, the largest difference between sites in winter-spring EVI observations was found in 2014/2015, following the

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driest year, when BB showed a pattern very similar to those registered over the years ~~previous-prior~~ to the dry spell, while AMO still presented EVI values 21% below the six-year average (Fig. ~~5a6a~~).

3.4 Biological net CO₂ exchange, gross primary production, ecosystem respiration and water use efficiency

~~The results of the “biological” annual C₂ balance, which was obtained, in case of AMO, by applying the ventilation model, are in accordance with our hypotheses. Annual C emission was always measured at AMO, whereas BB acted as a neutral and mild C sink. On average, AMO emitted 32 g C m⁻² more than BB. At a monthly time scale, net CO₂ fluxes during autumn were, on average, ~4 times higher at BB, excepting the last study year, when the net CO₂ emission at AMO was 21 times greater than at the “natural” site. However, during winter and spring months, net CO₂ uptake was generally higher at the BB (Fig. 7a).~~

~~On average, during the six years of study, GPP, R_{eco} and Water Use Efficiency (WUE) were nine, two and ten times higher, respectively, at BB compared to AMO. Firstly, GPP was always higher at BB compared to AMO (Fig. 7c). Major differences occurred in autumn 2014/2015, when monthly cumulative GPP at BB was 32 times higher on average. Similarly, R_{eco} was generally higher, up to ~8 times (October 2014), at BB. However, respiratory fluxes were occasionally greater at AMO, from 2% to 31% higher, during spring and winter months of all studied years excepting 2013/2014 (Fig. 7b). Maximum inter-site differences in GPP and R_{eco} were found in winter and autumn 2014/2015, following the driest year, when monthly GPP was ~30 times higher at BB compared to AMO. Similarly, monthly R_{eco} was ~5 times greater at BB. Inter-site differences in partitioned fluxes could not be assessed during spring months due to the lack of data from BB. Secondly, WUE was lower at AMO during the whole study period, when maximum and minimum differences coincided with the highest and lowest differences in GPP between sites. On average, monthly WUE was 6 and 1.5 times higher at BB during winter and spring seasons. Major inter-site differences were found in autumn and winter 2014/2015 (Fig. 7d).~~

3.4.5 Differences in meteorological and soil variables between sites

Results from the two-sided Wilcoxon summed rank test (Table 45) showed significant differences (p-value < 0.05) between sites in most of the monitored meteorological variables. The few exceptions were the friction velocity (u*), when using the whole study period, the maximum wind speed registered every half-hour (WS_{max}), when analyzing May-September data, and the wind speed (WS) and precipitation when assessing daytime May-September data (Table 45). The great amount of observations (n ranged from 21410 to 205751) produced highly significant results (Table 45). Hence, the standardized difference between the samples (Diff_{st}) allowed us to quantitatively explore the differences between sites. Relevant differences (Diff_{st}>1) were found only for ~~pressure when using all databases, due to the distinct altitude of monitored sites. Regarding soil variables, important differences (Diff_{st}>1) between sites were detected in the~~ subsoil CO₂ molar fraction measured at 1.50 m depth (CO_{2, 1.50m}) for all periods, and during May-September months even when using only daytime data (Table 56). Concretely, CO_{2, 1.50m} was always higher in AMO, from 889 to 1109 ppm (Table S1, S2 and S3). Additionally, volumetric water content at 0.05m depth (VWC_{0.05m}) was also higher in AMO compared to BB but only during summer

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months (Table 56), when absolute differences were very small, ranging from 0.028 to 0.037 m³ m⁻³ (Tables S1, S2 and S3). In contrast, subsoil CO₂ molar fraction measured at 0.05 m depth (CO_{2, 0.05m}) was from 89 to 150 ppm higher in BB when analyzing dry season (May-September) daytime data (Table 56, S1, S2 and S3).

The temporal dynamics of subsoil CO₂ molar fractions revealed similar annual patterns between sites; generally however, CO_{2, 0.05m} was higher in BB, from 6 to 88%, while CO_{2, 1.50m} was always greater in AMO, from 31 to 97% (Fig. 86). On one hand, the maximum monthly averaged values of CO_{2, 0.05m} were registered in autumn, concretely, in November and October with 642 and 1120 ppm in AMO and BB, respectively, whereas minimum values occurred in September and August with 373 and 400 ppm at each site (Fig. 6a8a). On the other hand, peak~~ing~~ monthly averaged values of CO_{2, 1.50m} occurred in July for both sites, with 2751 and 1602 ppm in AMO and BB, respectively, although relatively high CO_{2, 1.50m} was also measured during November in BB. On the contrary, minimum values were observed in December and February, with 1364 and 735 ppm in AMO and BB, respectively (Fig. 6b8b).

Finally, results of the Spearman correlation analysis between ~~net CO₂ exchange pressure~~ and belowground CO₂ at 1.50 m depth (CO_{2, 1.50m}) showed a positive relationship between both absolute variables, which was stronger in case of AMO compared to BB with Spearman correlation coefficients (r_s) of 0.30 and 0.11, respectively (Table 7). In contrast, a negative relationship was found between pressure and CO_{2, 1.50m} between both variables. Additionally, although most of the coefficients were significant (p value < 0.05), with higher r_s correlations were found in AMO compared to BB (Table 67). Additionally, r_s Spearman correlation coefficients (r_s) were maxima at 12 h intervals for CO_{2, 1.50m} and pressure increments (dP_{12h}) at AMO, and ~~and~~ at 6 h intervals for CO_{2, 1.50m} and pressure increments (dP_{6h}, respectively) at BB, with r_s equal to -0.87 and -0.63, respectively.

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4 Discussion

Our results verify that land-degradation affects the C sequestration capacity of semiarid ecosystems, since relevant differences between sites were observed during the growing season, when greater net C uptake over longer periods was observed in the “natural” site (BB). However, contrary to what we previously hypothesized, much greater net C release was measured at the “degraded” site (AMO) over drought periods due to the predominance of subterranean ventilation (López-Ballesteros et al., 2017). In fact, the great difference in annual C budgets between sites (Fig. 23) was largely related to this process resulting in an average release of 196 ± 40 and -23 ± 20 g C m⁻² yr⁻¹ for the “degraded” (AMO) and “natural” (BB) sites, respectively. Even when assessing only the biological net CO₂ exchange, by subtracting the non-biological CO₂ flux when feasible, the “degraded” site emitted 32 g C m⁻² more than the “natural” site. In this regard, the ecosystems’ functioning could be divided into three different phases. The first phase corresponded to the autumn months, when the first rainfall events after the dry summer (i.e. rain pulses) activated the soil microbiota triggering respiratory CO₂ emissions as previously measured at the same experimental sites (López-Ballesteros et al., 2016; Rey et al., 2017). During this phase, maximum net CO₂ release emission was observed at BB at the “natural” site, however higher CO₂ release was usually

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measured at AMO which exceeded the biological net CO₂ flux observed in the “degraded” site (Fig. 47a). In fact, as hypothesized, R_{eco} was generally higher at the “natural” site during autumn months (Fig. 7b); maybe probably due to the greater pool of organic carbon (Table 1) and the hydric stress experienced prior to rewetting, the potential differences in microbial communities (Rey et al., 2017), and the greater pool of soil inorganic carbon (Emmerich, 2003; Table 1) between sites. The second phase comprised the growing period, when plants photosynthesized and also respired along with microorganisms under milder temperatures and better hydric conditions. During this phase, larger net CO₂ uptake was measured in at BB the “natural” site, concretely 162% more than in AMO the “degraded” site (Table 34) due to the higher vegetation cover and more fertile soils (Table 1) of the “natural” site. Accordingly, GPP estimates were, on average, nine times higher at the “natural” site (Fig. 7c) and Moreover, these results were supported by the lower ET, WUE and EVI values obtained in AMO the “degraded” site during winter and spring months of over the whole study period (Fig. 5, 56 and 7d). The third phase consisted of the dormancy period when water scarcity and high temperatures constrained biological activity. During this period, as stated previously, a neutral C balance was observed in at BB the “natural” site while extreme CO₂ release was measured in at the “degraded” site AMO, as expected (Fig. 4, 6b), where ventilative fluxes were dominant.

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In order to detect potential factors driving the observed differences in the C balances, we checked whether soil and meteorological variables differed between sites. Our results demonstrated that some factors typically influencing gross primary production (GPP) and ecosystem respiration (R_{eco}), and hence net ecosystem CO₂ exchange, such as photosynthetic photon flux density (PPFD; Michaelis and Menten, 1913), precipitation (Berner et al., 2017; Jongen et al., 2011), vapor pressure deficit (VPD; Lasslop et al., 2010) and soil and air temperature (Lloyd and Taylor, 1994), did not differ between sites (Table 45, 56). Conversely, some differences were found for shallow soil volumetric water content (VWC_{0.05m}) during dry seasons (Table 56), when VWC_{0.05m} was two times higher in AMO than in BB, but absolute differences were slight, from 0.028 to 0.037 m³ m⁻³ (Tables S1,S2 and S3). Hence, although the important influence of soil moisture in both GPP and R_{eco} is known (Tang and Baldocchi, 2005), we believe that differences in VWC_{0.05m} are not relevant enough relevant to cause the differing ecosystems² functioning observed over the drought period. Additionally, we think that this inter-site difference in VWC_{0.05m} could be instrumental, or due to the spatial variability of VWC_{0.05m} derived from the heterogeneity of soil morphological characteristics, since we only used one sensor at each site. Similarly, important differences were not detected in several variables linked to subterranean ventilation, such as the friction velocity (u_{*}; Kowalski et al., 2008), wind speed (WS; Rey et al., 2012), half-hourly maximum wind speed (WS_{max}) and net radiation, which has been positively correlated to ventilative CO₂ fluxes (López-Ballesteros et al., 2017), when using the analysis periods when this process is supposed to be relevant, namely daytime hours during the dry seasons (Table 45). In contrast, inter site differences were found in net radiation (Table 4). However, although no turbulence and wind speed inter-site differences were found, interconnectivity of soil pores and fractures is probably higher at AMO the “degraded” site (Table 1) due to its higher gravel and rock fractions (Table 1), which could lead to an enhanced penetration of eddies within the vadose zone (Pérez-Priego et al., 2013).

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Apart from that, outstanding differences between sites were observed in subsoil CO₂ molar fractions measured at 0.05 and 1.50 m depths (CO_{2, 0.05m} and CO_{2, 1.50m}, respectively; Table 56). On one hand, CO_{2, 0.05m} was generally higher in ~~BB-the~~ “natural” site given its lower degradation level, which probably promotes a higher microbial activity supported by higher vegetation density and soil fertility (Table 1) especially during spring (Fig. 68), as pointed also by Oyonarte et al. (2012). On the other hand, CO_{2, 1.50m} values were acutely higher in ~~AMO~~the “degraded” site, by up to 1000 ppm compared to ~~BB-the~~ “natural” site (Tables S1, S2 and S3). Therefore, we suggest that CO_{2, 1.50m} is the main factor responsible for the inter-site differences in net CO₂ fluxes over the dry season. In this regard, previous research has suggested two potential origins of this vadose zone CO₂, geological degassing (Rey et al. 2012b) and/or subterranean translocation of CO₂ in both gaseous and aqueous phases (López-Ballesteros et al. 2017). –However, not only the amount of subsoil CO₂ matters but also how effective is its transport, since both determine the net CO₂ release from the vadose zone to the atmosphere. In this context, Oyonarte et al. (2012) found, in the same study area (Cabo de Gata-Níjar Natural Park), that soils with degradation symptoms, such as lower SOC, depleted biological activity, coarser texture and worse structure, showed higher soil CO₂ effluxes over the dry season. Additionally, soil CO₂ effluxes measured during summer months correlated positively with the fraction of rock outcrops, suggesting that deteriorated soil physical conditions actually enhanced vertical transfer of CO₂-rich air from subsoil to the atmosphere (Oyonarte et al., 2012). In fact, correlation analysis between CO_{2, 1.50m} and net CO₂ exchange/atmospheric pressure (Table 67) showed a stronger relationship between ~~both–these~~ variables at ~~AMO~~the “degraded” site. In this sense, ecosystem degradation could provoke a greater exposure of subsoil CO₂ to the pressure effect, as described by Sánchez-Cañete et al. (2013), probably due to a higher fraction of bare soil, coarser structure, differing porosity type and/or thinner soil depth (Table 1).

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Regarding EVI data, we found a discrepancy between GPP estimates and EVI values since, contrary to what is observed in EVI results, we observed that GPP was always higher at the “natural” site compared to the “degraded” site. We think that this is due to the different spatial scales defining each measurement. MODIS pixels have an area of ~6.25 ha while the eddy covariance footprint corresponds to a smaller area of ~1 ha. Therefore, there is an EVI uncertainty that stems from the influence of other surface elements apart from vegetation, such as bare soil or outcrops within the pixel, which is our case. In fact, previous studies confirm the discrepancy between MODIS- and EC-derived GPP estimates, especially on sparse vegetation areas with low productivity (Gilabert et al., 2015). However, ~~Regarding~~ EVI data, ~~these results~~ have allowed us to complement our findings based on CO₂ fluxes, especially when EC data losses occurred. For instance, the declining trend observed from 2009/2010 until the end of the study period, for both sites, was not noticeable from EC data alone (Fig. 56). This long-term decrease in EVI may be related to a gradual drying following the wettest year (2009/2010), when extraordinarily high precipitation (twice the mean –annual ~~average~~–precipitation for both sites over the study period) occurred. This EVI pattern also ~~denotes~~ suggests a pulse-like behaviour of ecosystem vegetation over the inter-annual time scale.

Moreover, in addition to demonstrating that degradation can influence the biological activity of ecosystems' vegetation, EVI results also showed that degradation level can modulate how an ecosystem responds to a short-term ~~change~~ disturbance. A clear example is the dry spell experienced in 2013/2014, when a reduction in EVI was measured during the growing season in both sites, i.e. 35% and 28% in ~~BB~~ the "natural" and "degraded" sites ~~AMO~~, respectively. However, a year later (2014/2015), EVI values below the six-year average were observed only at the "degraded" site ~~AMO~~ (21% lower; Fig. 5a6a) and major inter-site differences were found for GPP, R_{eco} and WUE during autumn and winter months (Fig. 7b, 7c and 7d). Accordingly, the "natural" site (~~BB~~) seemed to be more stable-resilient than the "degraded" site (~~AMO~~) against the short-term disturbance, since the effect of drought persisted in AMO even during the following year, while BB recovered to a pre-perturbation state within the same period (Fig. 5a6a). As a result, ecosystem resilience ~~defined by~~ (Holling, (1973) ~~as the amount of disturbance that a system can withstand without changing state~~, was lessened by long-term disturbances such as land degradation, making degraded ecosystems more vulnerable to climate extremes (Reichstein et al., 2013). In this sense, mitigation policies to confront land degradation should be focused on prevention programs since ecosystem restoration does not recover complete ecosystem functionality (Lal, 2001; Moreno-Mateos et al., 2017). Moreover, even after several decades, relict degradation legacies can remain (Alados et al., 2011).

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15 5 Conclusions

The present study can be seen as a step forward to better understanding the effect of land degradation on the intricate network of multi-scale processes, factors and structures that define ecosystems' biological productivity and ultimately control their C balances. Despite some limitations, such as long-term data gaps, this research demonstrates that continuous ecosystem-scale EC observations remain crucial to comprehend how climate and land use change can modify the C sequestration capacity of ecosystems. In fact, annual average release of 196 ± 40 and -23 ± 20 g C m⁻² yr⁻¹ for the "degraded" and "natural" (i.e. site of reference) sites were measured, respectively. Additionally, larger net CO₂ uptake over longer periods was observed at the "natural" site, concretely an amount of C 162% higher compared to the "degraded" site, whereas much greater net CO₂ release was measured at the "degraded" site during drought periods. Accordingly, the estimates of gross primary production, ecosystem respiration and water use efficiency were, on average, nine, two and ten times higher in the "natural" site, respectively. Future research should be based on the continuity of long-term monitoring stations, such as eddy covariance stations, in order to calibrate and validate satellite data, reduce uncertainties in the relationships between ecosystem productivity, land degradation and climate change and finally, to improve the predictive ability of current terrestrial C models.

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Data availability

The eddy covariance data are available in the European Database Cluster (<http://www.europe-fluxdata.eu>) where experimental sites have the codes “Es-Amo” and “Es-Agu”. Other data can be obtained by contacting the corresponding author.

5 Author contribution

FD, CO, AK and PSO designed the experiment. ALB, PSO, EPSC and MRM calibrated the sensors, collected the data and maintained the field instrumentation. EPSC designed subsoil data acquisition system and MRM processed subsoil data. ALB processed the eddy covariance data, made the figures and tables and wrote the manuscript. All authors reviewed the manuscript.

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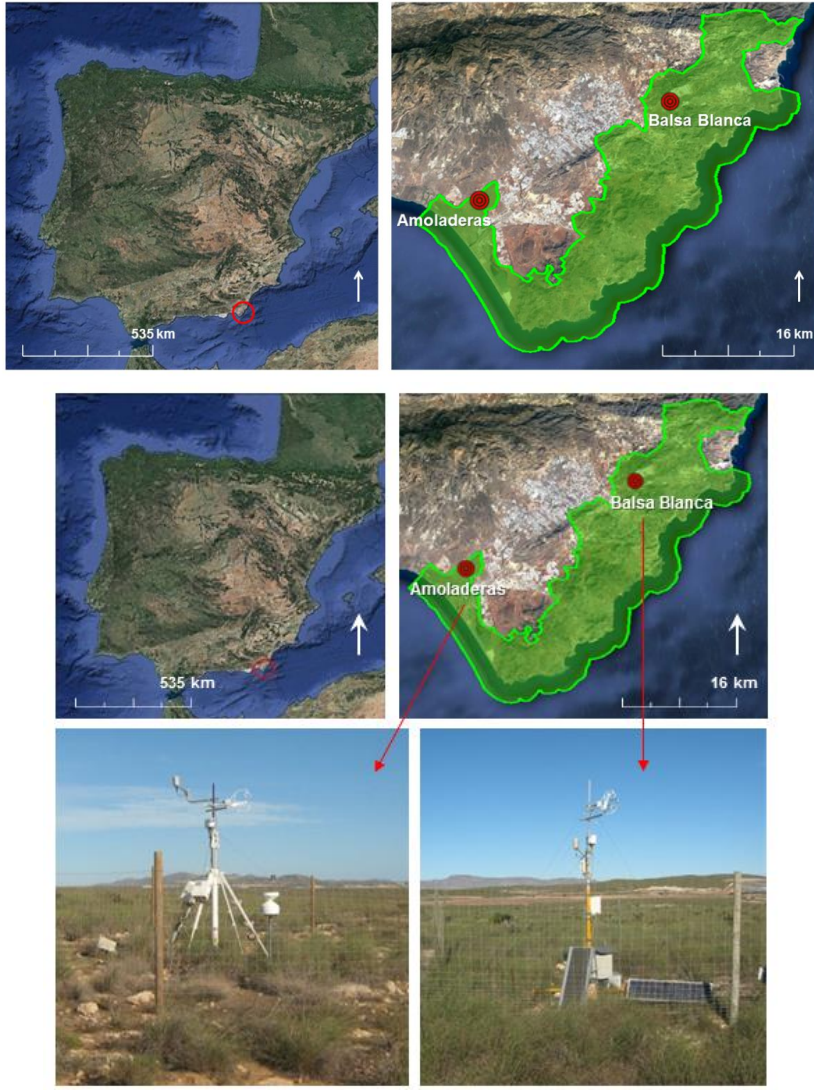
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5 | Figure 1: Location (above) and photographs (below) of the experimental sites. Green area represents the Cabo de Gata-Níjar Natural Park (Almeria, Spain).

Table 1: Site characteristics, surface fractions and soil properties of both experimental sites studied. Asterisks denote significant differences (p-value<0.05). Adapted from Rey et al., (2011).

	Amoladeras	Balsa Blanca
<i>Site characteristics</i>		
Location	Almeria N36°50'52" W2°15'12"	Almeria N36°56' 02" W2°1'58"
Altitude (m)	65	208
Climate	Subtropical, dry, semiarid	Subtropical, dry, semiarid
Annual T	18	18
Annual P	220	220
Predominant sp.	<i>Machrocloa tenacissima</i>	<i>Machrocloa tenacissima</i>
<i>Surface fractions</i>		
Vegetation cover (%)	23.1 ± 2.4*	63.2 ± 5.2*
Litter (%)	10.5 ± 2.0	8.1 ± 1.9
Biological crust (%)	23.1 ± 2.8	18.2 ± 3.8
Bare soil (%)	8.1 ± 0.1*	0.3 ± 0.3*
Gravel (%)	21.1 ± 0.1*	8.6 ± 2.5*
Rock (%)	14.0 ± 1.2*	1.5 ± 0.5*
<i>Soil properties</i>		
Soil type	Lithic Leptosol (Calcaric)	Mollic Lithic Leptosol (Calcaric)
Maximum soil depth (cm)	10	20
Soil texture class	Sandy loam	Sandy loam
Clay (%)	14.6	16.1
Silt (%)	27.0	22.8
Sand (%)	58.4	61.1
Bulk density (g cm ⁻³)	1.11 ± 0.04	1.25 ± 0.09
SOC (kg m ⁻²)	1.24	4.64
Carbonates (%)	14	2

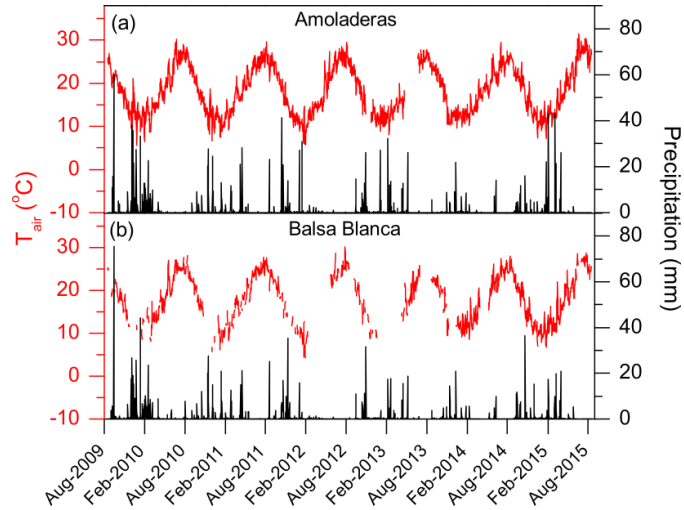
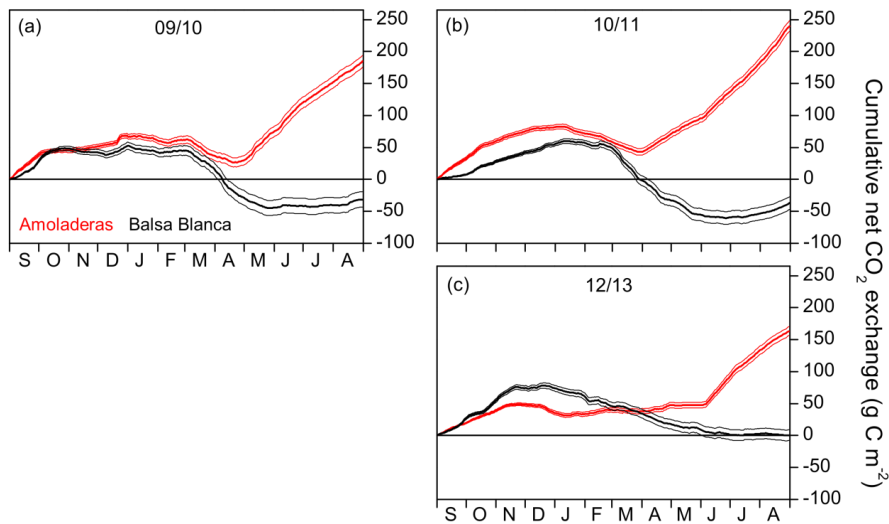


Figure 2: Daily averages of air temperature (T_{air}) and precipitation in (a) Amoladeras and (b) Balsa Blanca.



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Figure 3: Cumulative annual net CO_2 exchange over the three hydrological years without long-periods of missing data in both experimental sites, Amoladeras (red lines) and Balsa Blanca (black lines). Negative values denote net carbon uptake while positive values denote net carbon release. Thin lines indicate uncertainty derived from the gap-filling procedure.

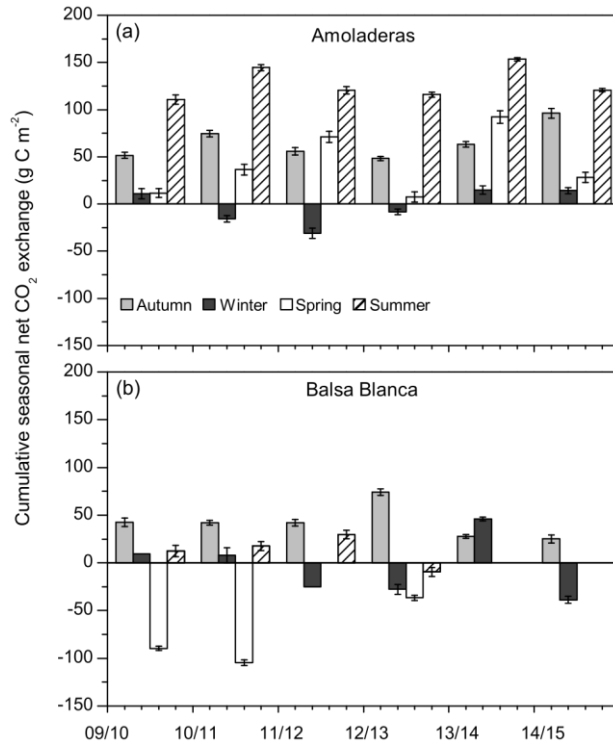
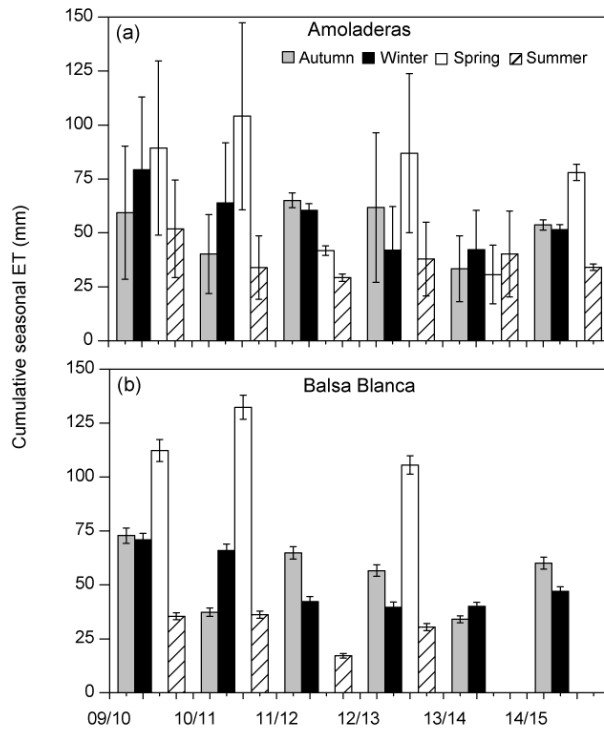


Figure 4: Cumulative seasonal net CO₂ exchange over the study period in both experimental sites. Negative values denote net carbon uptake while positive values denote net carbon release. In case of Balsa Blanca, lacking bars correspond to long-term data losses (>50% data).



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Figure 5: Cumulative seasonal evapotranspiration fluxes (ET) over the study period in both experimental sites. In case of Balsa Blanca, lacking bars correspond to long-term data losses (>50% data). Error bars denote uncertainty derived from the gap-filling procedure.

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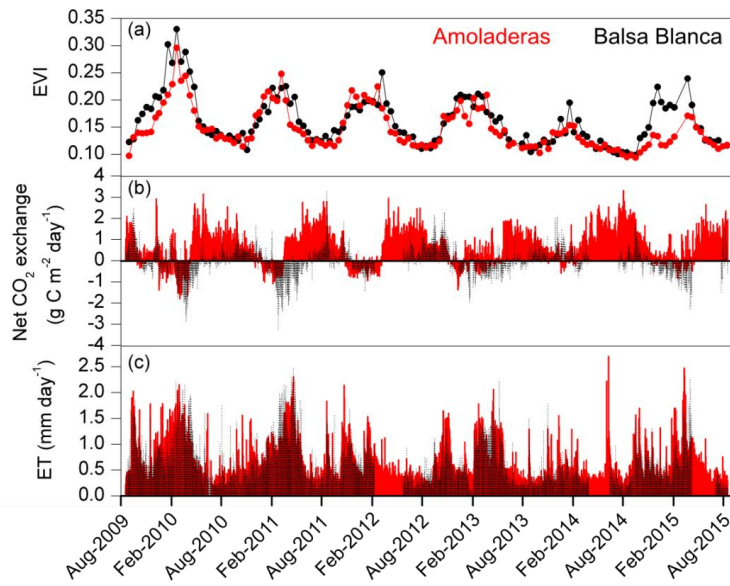
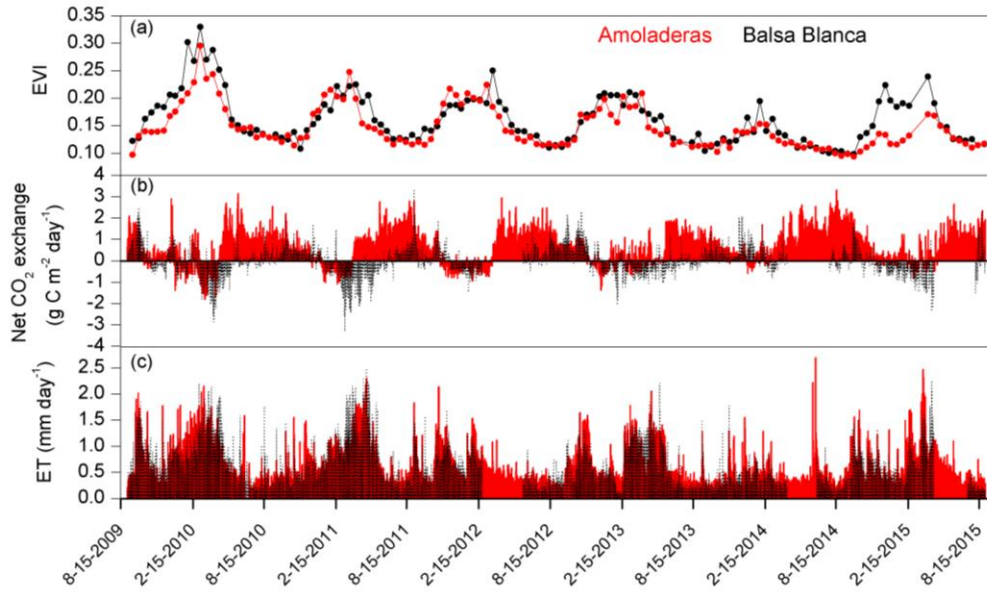


Figure 65: Time series of (a) Enhanced Vegetation Index (EVI), (b) daily net CO₂ exchange and (c) daily evapotranspiration fluxes measured in Amoladeras (red lines and dots) and Balsa Blanca (black lines and dots) over six hydrological years (2009-2015). Long-term data losses correspond to periods of several months when ET and CO₂ fluxes are absent.

5

Table 34: Number of days with daily net CO₂ uptake and the related total C absorbed for every hydrological year and every field site of the study. Asterisks denote those years with abundant data losses (~30% data).

Site	Amoladeras		Balsa Blanca	
	N. days of net CO ₂ uptake	Total net CO ₂ uptake (g C m ⁻²)	N. days of net CO ₂ uptake	Total net CO ₂ uptake (g C m ⁻²)
09/10	58	-59 ± 7	196	-125 ± 12
10/11	86	-41 ± 4	160	-129 ± 10
11/12	114	-43 ± 5	104*	-40 ± 6*
12/13	103	-31 ± 4	212	-96 ± 8
13/14	31	-6 ± 12	64*	-15 ± 5*
14/15	59	-14 ± 3	172*	-103 ± 9*

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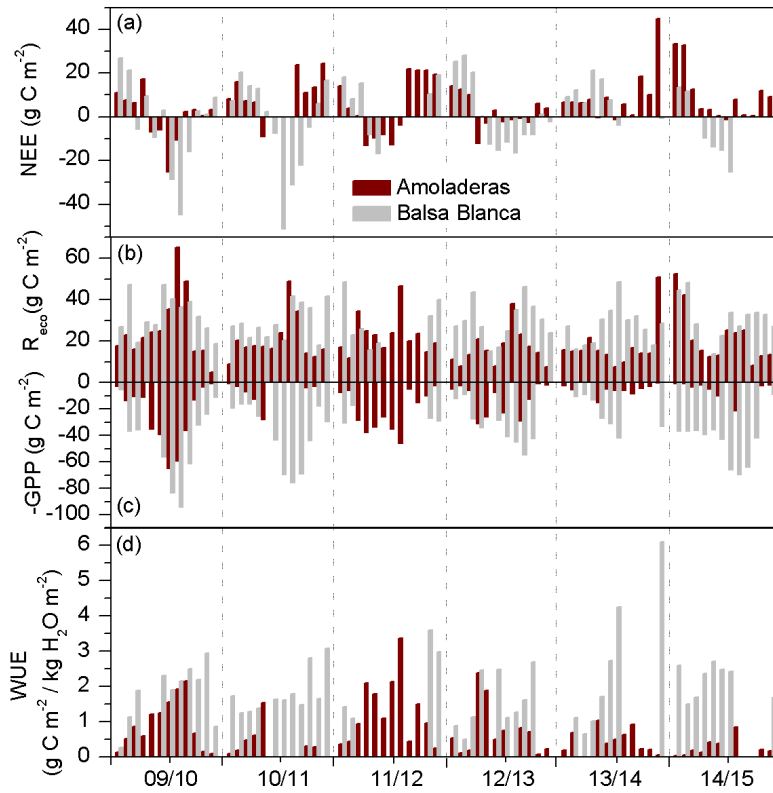


Figure 7: Monthly cumulative fluxes of (a) biological net ecosystem CO₂ exchange, (b) ecosystem respiration (R_{eco}), (c) negative gross primary production and (d) water use efficiency over the six hydrological years of study (2009-2015) for Amoladeras (dark red) and Balsa Blanca (grey). Lacking bars correspond to long-term data losses.

Table 5: Results of the two-sided Wilcoxon summed rank test used to assess differences among meteorological variables measured at each experimental site over all periods, from May to September and from May to September during daytime, separately. Medians of the difference between the samples (Amoladeras minus Balsa Blanca) in standardized terms ($Diff_{st}$) and number of observations are detailed. Significant results (p -value<0.05) are denoted with asterisks, and bold values represent those variables with $Diff_{st}$ between sites above 1.

Variables	All periods		May - September		May - September Daytime	
	$Diff_{st}$	n	$Diff_{st}$	n	$Diff_{st}$	n
PPFD ($\mu\text{mol m}^{-2} \text{s}^{-1}$)	0.0009*	205751	0.0009*	84491	0.1378*	38963
Net radiation (W m^{-2})	-0.0457*	197924	-0.0476*	81019	-0.1205*	38963
T_{air} ($^{\circ}\text{C}$)	0.0310*	182240	0.1935*	77866	0.0502*	37480
VPD (hPa)	0.0783*	166918	0.1370*	71474	-0.0938*	34430
RH (%)	-0.1636*	197649	-0.1031*	80950	0.1784*	38935
u_* (m s^{-1})	-0.0054	166346	-0.0563*	71194	-0.1340*	34284
WS (m s^{-1})	0.1628*	166359	0.0793*	71195	0.0165	34285
WS_{max} (m s^{-1})	0.1001*	165458	0.0124	70635	-0.0796*	33994
Pressure (hPa)	1.6999*0.37	16635916	1.8696*0.58	711957	1.8633*0.56	342853
	37*	6336	28*	1188	02*	4280
Precipitation (mm)	-1.95E-05*	204892	-4.84E-05*	83860	5.32E-05	38963

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Table 56: Results of the two-sided Wilcoxon summed rank test used to assess differences among soil variables measured at each experimental site over all periods, from May to September and from May to September during daytime, separately. Medians of the difference between the samples (Amoladeras minus Balsa Blanca) in standardized terms ($Diff_{st}$) and number of observations are detailed. Significant results (p -value<0.05) are denoted with asterisks, and bold values represent those variables with $Diff_{st}$ between sites above 1.

Variables	All periods		May - September		May - September Daytime	
	$Diff_{st}$	n	$Diff_{st}$	n	$Diff_{st}$	n
$\text{CO}_2, 0.05\text{m}$ (ppm)	-0.4027*	46340	-0.6578*	21413	-1.1396*	9816
$\text{CO}_2, 1.50\text{m}$ (ppm)	1.1196*	50133	1.3517*	24347	1.3062*	11385
$T_{0.05\text{m}}$ ($^{\circ}\text{C}$)	0.0927*	46337	-0.1160*	21410	-0.2119*	9813
$T_{1.50\text{m}}$ ($^{\circ}\text{C}$)	0.1476*	50137	-0.0591*	24350	-0.0834*	11385
$\text{VWC}_{0.05\text{m}}$ ($\text{m}^3 \text{m}^{-3}$)	0.8265*	52353	1.2724*	25231	1.2839*	11303
$\text{VWC}_{1.50\text{m}}$ ($\text{m}^3 \text{m}^{-3}$)	-0.8385*	53865	0.0674*	24570	0.0547*	11462

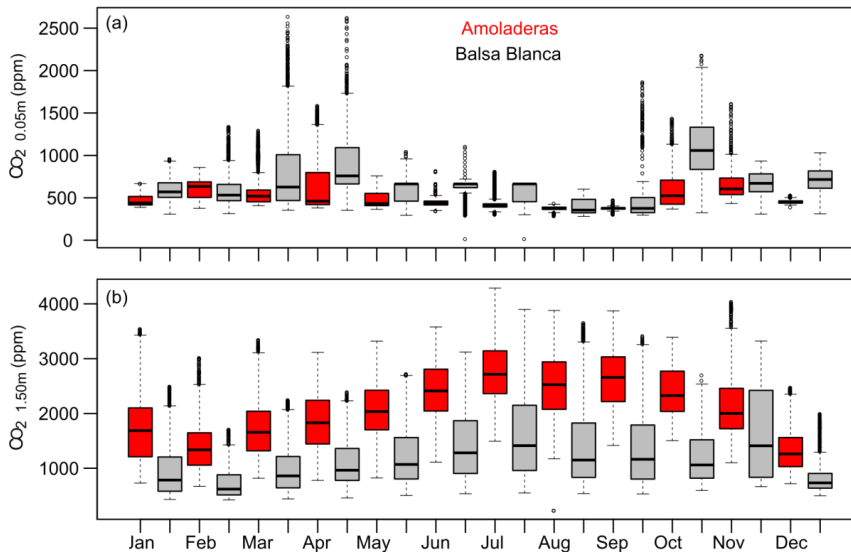


Figure 86: Box-and-whisker plots of CO₂ molar fractions measured at (a) 0.05 m and (b) 1.50 m belowground in Amoladeras (red boxes) and Balsa Blanca (grey boxes) from January 2014 to August 2015. The box extends from the first (Q1) to the third quartiles (Q3) and the central line represents the median (50% percentile). Dots represent outliers; upper whisker is located at the smaller of the maximum value and Q3 + 1.5 IQR (Interquartile Range), and lower whisker is located at the larger of the minimum value and Q1 - 1.5 IQR.

Table 7: Spearman correlation coefficients (r_s) for every paired simple correlation among maximum quality net CO₂ exchange fluxes ($\mu\text{mol m}^{-2} \text{s}^{-1}$), absolute and differential pressure (hPa) at 6, 12, 24 and 72 hours time-step and absolute and differential CO₂ molar fraction measured at 1.50 m below ground (ppm) at the same time-steps. Bold values represent the highest correlation coefficients while shaded ones denotes non-significant relationships (p -values > 0.05).

	Amoladeras						Balsa Blanca					
	Net CO ₂ exchange	P	dP _{6h}	dP _{12h}	dP _{24h}	dP _{72h}	Net CO ₂ exchange	P	dP _{6h}	dP _{12h}	dP _{24h}	dP _{72h}
CO ₂ _{1.50m}	0.30	-0.66	-0.33	-0.46	-0.56	-0.55	0.11	-0.33	-0.51	-0.53	-0.62	-0.45
dCO ₂ _{1.50m 6h}	0	-0.1	-0.84	-0.5	-0.26	-0.01	0.02	-0.07	-0.63	-0.19	-0.13	-0.04
dCO ₂ _{1.50m 12h}	0.06	-0.08	-0.57	-0.87	-0.55	-0.05	0.03	-0.03	-0.46	-0.50	-0.31	-0.03
dCO ₂ _{1.50m 24h}	0.03	-0.13	-0.47	-0.78	-0.85	-0.15	0.02	-0.04	-0.40	-0.59	-0.58	-0.04
dCO ₂ _{1.50m 72h}	0	-0.28	-0.28	-0.49	-0.64	-0.74	0.00	-0.13	-0.28	-0.43	-0.57	-0.48

Comentario [T1]: Add this new info (NEE) to results and discussion sections.

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Table 6: Spearman correlation coefficients (r_s) for every paired simple correlation among maximum quality net CO_2 exchange fluxes ($\mu\text{mol m}^{-2} \text{s}^{-1}$), absolute and differential pressure (hPa) at 6, 12, 24 and 72 hours time step and absolute and differential CO_2 molar fraction measured at 1.50 m below ground (ppm) at the same time steps. Bold values represent the highest correlation coefficients while shaded ones denotes non-significant relationships ($p\text{-value} > 0.05$).

	Amoladeras					Balsa Blanca				
	P	dP _{6h}	dP _{12h}	dP _{24h}	dP _{72h}	P	dP _{6h}	dP _{12h}	dP _{24h}	dP _{72h}
$\text{CO}_{2,1.50\text{m}}$	-0.66	-0.33	-0.46	-0.56	-0.55	-0.33	-0.51	-0.53	-0.62	-0.45
d $\text{CO}_{2,1.50\text{m}_6\text{h}}$	-0.1	-0.84	-0.5	-0.26	-0.01	-0.07	-0.63	-0.19	-0.13	-0.04
d $\text{CO}_{2,1.50\text{m}_12\text{h}}$	-0.08	-0.57	-0.87	-0.55	-0.05	-0.03	-0.46	-0.50	-0.31	-0.03
d $\text{CO}_{2,1.50\text{m}_24\text{h}}$	-0.13	-0.47	-0.78	-0.85	-0.15	-0.04	-0.40	-0.59	-0.58	-0.04
d $\text{CO}_{2,1.50\text{m}_72\text{h}}$	-0.28	-0.28	-0.49	-0.64	-0.74	-0.13	-0.28	-0.43	-0.57	-0.48

Supplementary Material

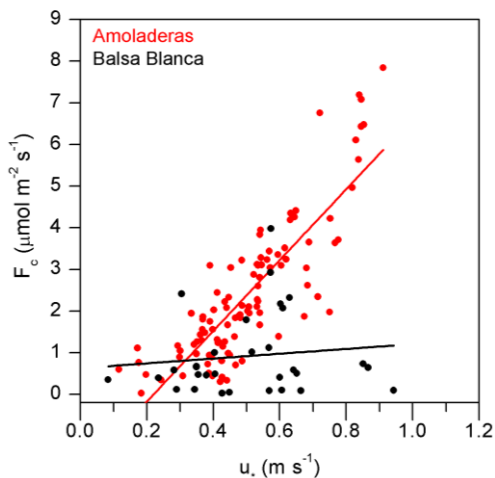


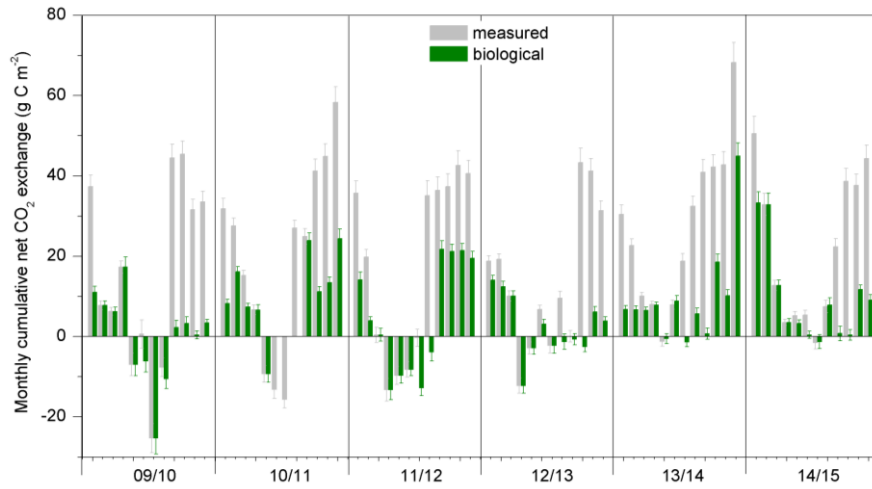
Figure S1: Half-hourly net CO_2 fluxes of maximum quality (QC flag=0) versus friction velocity (u_*) corresponding to daytime hours during the extremely dry periods when subterranean ventilation dominates the net CO_2 flux. Red and black dots represent Amoladeras and Balsa Blanca, respectively.

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Figure S2: Cumulative measured and biological (after applying the ventilation model) net CO₂ exchange for every month of the study period (5 hydrological years: 2009-2015) in Amoladeras.

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Table S1: Results of the two-sided Wilcoxon summed rank test used to assess differences between meteorological variables measured at each experimental site over the all periods. Medians of the absolute (Diff) and standardized differences (Diff_{st}) between the samples (Amoladeras minus Balsa Blanca), p-values and number of observations (n) are detailed.

Variables	All periods			
	Diff	Diff _{st}	p-value	n
PPFD ($\mu\text{mol m}^{-2} \text{s}^{-1}$)	0.4701	0.0009	0	205751
Net radiation (W m^{-2})	-8.8620	-0.0457	0	197924
T _{air} (°C)	0.1928	0.0310	6.42E-10	182240
VPD (hPa)	0.5358	0.0783	4.38E-104	166918
RH (%)	-3.1947	-0.1636	6.39E-244	197649
u _s (m s^{-1})	-0.0016	-0.0054	0.1214	166346
WS (m s^{-1})	0.3621	0.1628	0	166359
WS _{max} (m s^{-1})	0.4327	0.1001	5.53E-132	165458
Pressure (hPa)	19.00612. <u>3226</u>	1.69990.3 <u>737</u>	0	166359 <u>166336</u>
Precipitation (mm)	-0.0001	-1.95E-05	0.0003	204892
CO _{2, 0.05m} (ppm)	-93.8644	-0.4027	0	46340
CO _{2, 1.50m} (ppm)	888.9239	1.1196	0	50133
T _{0.05m} (°C)	0.9215	0.0927	5.51E-21	46337
T _{1.50m} (°C)	0.5780	0.1476	1.43E-51	50137
VWC _{0.05m} ($\text{m}^3 \text{m}^{-3}$)	0.0372	0.8265	0	52353
VWC _{1.50m} ($\text{m}^3 \text{m}^{-3}$)	-0.0123	-0.8385	0	53865

Table S2: Results of the two-sided Wilcoxon summed rank test used to assess differences between meteorological variables measured at each experimental site from May to September. Medians of the absolute (Diff) and standardized differences (Diff_{st}) between the samples (Amoladeras minus Balsa Blanca), p-values and number of observations (n) are detailed.

5

Variables	May - September			
	Diff	Diff _{st}	p-value	n
PPFD ($\mu\text{mol m}^{-2} \text{s}^{-1}$)	0.5230	0.0009	8.11E-147	84491
Net radiation (W m^{-2})	-10.9546	-0.0476	4.78E-167	81019
T _{air} (°C)	0.8378	0.1935	1.53E-145	77866
VPD (hPa)	1.1136	0.1370	4.50E-90	71474
RH (%)	-2.0229	-0.1031	1.21E-38	80950
u _s (m s^{-1})	-0.0186	-0.0563	8.47E-31	71194
WS (m s^{-1})	0.1720	0.0793	1.84E-31	71195
WS _{max} (m s^{-1})	0.0498	0.0124	0.0749	70635
Pressure (hPa)	18.67072 2120	1.86960.5 828	0	71195 71188
Precipitation (mm)	0.0000	-4.84E-05	4.29E-05	83860
CO _{2, 0.05m} (ppm)	-89.2404	-0.6578	9.99E-207	21413
CO _{2, 1.50m} (ppm)	1109.1440	1.3517	0	24347
T _{0.05m} (°C)	-0.8057	-0.1160	3.04E-16	21410
T _{1.50m} (°C)	-0.1570	-0.0591	3.13E-07	24350
VWC _{0.05m} ($\text{m}^3 \text{m}^{-3}$)	0.0286	1.2724	0	25231
VWC _{1.50m} ($\text{m}^3 \text{m}^{-3}$)	0.0011	0.0674	3.51E-18	24570

Table S3: Results of the two-sided Wilcoxon summed rank test used to assess differences between meteorological variables measured at each experimental site from May to September during daytime. Medians of the absolute (Diff) and standardized differences (Diff and Diff_{st}) between the samples (Amoladeras minus Balsa Blanca), p-values and number of observations (n) are detailed.

5

Variables	May – September Daytime			
	Diff	Diff _{st}	p-value	n
PPFD ($\mu\text{mol m}^{-2} \text{s}^{-1}$)	64.3400	0.1378	5.62E-42	38963
Net radiation (W m^{-2})	-21.9460	-0.1205	4.80E-34	38963
T _{air} (°C)	0.1939	0.0502	1.39E-06	37480
VPD (hPa)	-0.7579	-0.0938	3.87E-20	34430
RH (%)	2.9921	0.1784	1.45E-65	38935
u* (m s^{-1})	-0.0430	-0.1340	3.76E-87	34284
WS (m s^{-1})	0.0356	0.0165	0.1341	34285
WS _{max} (m s^{-1})	-0.2920	-0.0796	5.83E-14	33994
Pressure (hPa)	18.40852 <u>1101</u>	1.86330.5 <u>602</u>	0	3428534 <u>280</u>
Precipitation (mm)	0.0000	5.32E-05	0.9875	38963
CO _{2, 0.05m} (ppm)	-149.513	-1.1396	0	9816
CO _{2, 1.50m} (ppm)	1046.358	1.3062	0	11385
T _{0.05m} (°C)	-1.473	-0.2119	2.70E-21	9813
T _{1.50m} (°C)	-0.220	-0.0834	4.75E-07	11385
VWC _{0.05m} ($\text{m}^3 \text{m}^{-3}$)	0.028	1.2839	0	11303
VWC _{1.50m} ($\text{m}^3 \text{m}^{-3}$)	0.001	0.0547	1.05E-09	11462

Can land degradation drive differences in the C exchange of two similar semiarid ecosystems?

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Abstract. ~~Currently, drylands occupy more than one third of the global terrestrial surface and are recognized as areas vulnerable to land degradation. The concept of land degradation stems from the loss of an ecosystem's biological productivity, due to long-term loss of natural vegetation or depletion of soil nutrients. The concept of land degradation stems from the loss of an ecosystem's biological productivity, which in turn relies on several degradation processes, such as long-term loss of natural vegetation, depletion of soil nutrients, soil compaction or water and wind erosion, to which drylands are especially vulnerable. Currently, drylands occupy more than one third of the global terrestrial surface and will probably expand under future climate change scenarios.~~ Drylands' key role in the global C-carbon (C) balance has been recently demonstrated, but the effects of land degradation on C sequestration by these ecosystems ~~still need to be investigated~~ ~~needs further research~~. In the present study, we compare net ~~carbon-C and water vapour exchange~~ fluxes, together with satellite, ~~data and~~ meteorological, ~~ambient~~ and vadose zone (CO₂, water content and temperature) ~~variables~~ measurements, between two nearby (~23 km) experimental sites representing "natural" (i.e. site of reference) and "degraded" grazed semiarid grasslands ~~located in SE Spain~~. ~~We utilized via data acquired from two~~ eddy covariance ~~stations located in SE Spain~~ measurements ~~over~~ during 6 years, with highly variable precipitation magnitude and distribution. Results show a striking difference in the annual C balances with an average release of 196 ± 40 and -23 ± 20 g C m⁻² yr⁻¹ for the "degraded" and "natural" sites, respectively. At the seasonal scale, differing patterns in net CO₂ fluxes were detected over both growing and dry seasons. As expected, during the growing seasons, greater net C uptake over longer periods was observed ~~at~~ the "natural" site, however, much greater net C release, ~~probably derived from subterranean ventilation~~, was measured ~~in~~ ~~at~~ the "degraded" site during drought periods. ~~After subtracting the non-biological CO₂ flux from net CO₂ exchange, flux partitioning results point out that, during the six years of study, gross primary production, ecosystem respiration and water~~

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use efficiency were, on average, nine, twice and ten times higher, respectively, at the “natural” site versus the “degraded” site. We also tested differences in all monitored meteorological and soil variables and found it most relevant that CO₂ at 1.50 m belowground was around 1000 ppm higher in the “degraded” site. Thus, we believe that subterranean ventilation of this vadose zone CO₂, previously observed at both sites, largely drives the differences in C dynamics between them, especially during the dry season maybe due to enhanced subsoil-atmosphere interconnectivity in the “degraded” site. Overall, the 12 site-years of data allow direct exploration of the roles of climate and land degradation in the biological and non-biological processes that ultimately control the C sequestration capacity of semiarid ecosystems.

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1 Introduction

~~The concept of land degradation stems from the loss of an ecosystem’s biological productivity, which in turn relies on several degradation processes such as long term loss of natural vegetation, deterioration of soil quality, depletion in biodiversity or water and wind erosion (UNCCD, 1994). Arid, semiarid and dry sub-humid areas, commonly known as drylands, have been recognized as areas vulnerable to land degradation processes. Overall, drylands occupy more than one third of Earth’s land surface and are inhabited by more than 2 billion people (Niemeijer et al., 2005), and concretely, the semiarid ecoregion comprises a major fraction, 37% of global dryland area, where the dominant biome is grassland (Niemeijer et al., 2005). Drylands have expanded globally for the last sixty years at an estimated annual rate of 5.8 million hectares in mid-latitudes alone (Lal, 2001), and are projected to expand under future climate change scenarios (Feng and Fu, 2013; Cook et al., 2014). Furthermore, the Mediterranean region has been recognized as an especially vulnerable region (Gao and Giorgi, 2008), where major expansions of semiarid areas will occur (Feng and Fu, 2013).~~

The concept of land degradation stems from the loss of an ecosystem’s biological productivity, which in turn relies on several degradation processes such as long-term loss of natural vegetation, deterioration of soil quality, biodiversity depletion or water and wind erosion (UNCCD, 1994). Drylands (arid, semiarid and dry sub-humid areas), which occupy more than one third of Earth’s land surface and are inhabited by more than 2 billion people (Niemeijer et al., 2005), have been recognized as areas vulnerable to land degradation processes. In fact, they have expanded globally for the last sixty years at an estimated annual rate of 5.8 million hectares in mid latitudes alone (Lal, 2001), and are projected to expand under future climate change scenarios (Feng and Fu, 2013; Cook et al., 2014), especially in the Mediterranean region, where major expansions of semiarid areas will occur (Gao and Giorgi, 2008; Feng and Fu, 2013).

Over recent decades, most research focused on land degradation has been based on remote sensing and earth observation techniques. Much of these investigations have aimed to refine methodological issues in order to accurately track land degradation in vulnerable areas, reduce uncertainties and explain inconsistencies among studies. For instance, a wide array of satellite-derived data, such as vegetation indices, normalized surface reflectance, brightness temperature or biomass-net primary production derivatives (Mbow et al., 2015), has been utilized to appraise desertification effects in the Sahel (Mbow et al., 2015; Fensholt et al., 2013) and also in other African countries such as Kenya (Omuto, 2011), Somalia (Omuto et al.,

2010), South Africa (Thompson et al., 2009) or Zimbabwe (Prince et al., 2009). Likewise, desertification in the Mediterranean region has been studied through satellite imagery, ~~concretely,~~ in Greece (Bajocco et al., 2012), Israel (Shoshany and Karnibad, 2015) and the Iberian Peninsula (del Barrio et al., 2010). ~~Additionally, some global assessments based on modelling approaches have focused on the global loss of net primary production derived from land degradation (Zika and Erb, 2009) or the effect of desertification on climate (Wang et al., 2016).~~ However, although drylands' key role in the global carbon (C) balance has been demonstrated (Poulter et al., 2014; Ahlström et al., 2015), very few investigations have directly quantified how land degradation processes disturb the C sequestration capacity of drylands (Lal, 2001), ~~which despite being is~~ one of the most important ecosystem services (Watanabe and Ortega, 2011).

In this regard, the few C-related desertification studies conducted over last decade have centred on soil C dynamics. Concretely, soil organic carbon (SOC) inventories have been used to explore the effects of climate, human activities and grazing pressure in desertification-prone areas of China (Feng et al., 2006) and Brazil (Schulz et al., 2016). Similarly, other investigations have evaluated soil degradation processes by means of soil CO₂ effluxes together with other biometric measurements in drylands found in China (Hou et al., 2014; Wang et al., 2007), Chile (Bown et al., 2014) and southeast Spain (Rey et al., 2011; Rey et al., 2017). However, the degradation processes associated with desertification affect several subsystems as well as their interactions at multiple spatial and temporal scales. For instance, adverse effects on soil quality involve depletion of soil fertility, but also reduce soil-water storage (Mainguet and Da Silva, 1998), which, in turn, can constrain seed germination and vegetation reestablishment, modify climax vegetation, disrupt biogeochemical cycles, alter water and energy balances, and consequently lead to a loss of ecosystem resilience (Lal, 2001). This cascade of disturbances may result in a reduction of the C sequestration capacity of a given ecosystem, which is clearly a symptom of the loss of biological productivity, resulting in a positive feedback to global warming. Therefore, a quite suitable and holistic approach is to integrate all subsystems effects into a whole ecosystem-scale assessment when quantifying the C loss derived from land degradation. However, the use of this integrative method is mostly lacking in the available literature.

The present study is located in an area, the southeast of Spain, that has been recognized as a hotspot of land degradation owing to the synergistic interaction of sociological and climatic factors (Puigdefábregas and Mendizabal, 1998). ~~where, furthermore, rainfall has trended downward at ca. 3 mm yr⁻¹ since 1900 (Oñate Rubalcaba, 1993).~~ Our core aim is to evaluate how dryland degradation affects the dynamics of net ecosystem-atmosphere C exchange of two semiarid grasslands that represent differing degradation status ("natural" *versus* "degraded") by means of meteorological ~~and,~~ satellite and subsoil CO₂ measurements, subsoil CO₂ sensors and together with carbon and water fluxes acquired by the eddy covariance (EC) technique (Baldocchi et al., 1988). ~~This technique allows us to directly quantify the net exchange of energy, water and CO₂ flux densities (hereinafter fluxes) between an ecosystem (including all of its subsystems, i.e. soil, plants, microorganism, vadose zone...) and the atmosphere at a high frequency (from minutes to hours).~~ Owing to the high temporal resolution of the EC method, we can assess the effect of land degradation as a slow change or disturbance legacy in the studied ecosystems and how, in turn, it influences the capacity of these ecosystems' resilience to against absorb fast changes ~~or~~ short-term disturbances, such as droughts climate extremes (i.e. droughts, heat waves).

Accordingly, our main hypothesis is that the degree of land degradation affects net ecosystem C sequestration capacity through its impact on some or all of the processes that compose the overall ecosystem C balance: photosynthesis, ecosystem respiration (biological processes) and subterranean ventilation—a non-biological process that provokes the transfer of CO₂-rich air from subsoil to atmosphere under drought and high turbulence conditions. Some land degradation processes are evident when we compare the “natural” site with the “degraded” site. Accordingly, our main hypothesis is that land degradation processes can directly affect abiotic and/or biotic factors and, consequently, influence the biological and/or non-biological processes that compose the net ecosystem CO₂ exchange: gross primary production, ecosystem respiration (biological processes) and subterranean ventilation – a non-biological process that provokes the transfer of CO₂-rich air from subsoil to atmosphere under drought and high turbulence conditions. Firstly, the lower vegetation cover in the “degraded” site would entail a higher thermal and radiative stress at the surface, especially during the drought period (Rey et al., 2017). The hypothesized effects on biological processes are a direct reduction in plant productivity and respiration, and an indirect decrease in heterotrophic respiration. Secondly, the higher cover of bare soil and outcrops in the “degraded” site may increase the soil-atmosphere interconnectivity, which indirectly can enhance the presence of advective CO₂ release through subterranean ventilation, which has been previously measured at both experimental sites (Rey et al., 2012; López-Ballesteros et al., 2017). And thirdly, the reduced soil fertility and depth may provoke changes in microbial communities (Evans and Wallenstein, 2014) due to stronger nutrient and water limitations. Consequently, a direct decrease in heterotrophic respiration and plant productivity and respiration is expected. In this context, as subterranean ventilation has been measured in both experimental sites (Rey et al., 2012; López Ballesteros et al., 2017), we also hypothesize a significant role of this process in the monitored ecosystems.

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Hence, our specific objectives are: (1) to compare the C sequestration capacity of two semiarid ecosystems showing a with differing degradation status, (2) to study the underlying involved processes (biological versus non-biological) and influencing factors that can drive potential differences in the net C exchange of studied ecosystems, and (3) to evaluate whether degradation can modulate ecosystem responses against short-term disturbances. To do this, we analysed 12 site-years of EC data, Enhanced Vegetation Index (EVI) time series and monitored ambient variables registered over the same period (2009-2015) at both sites. Additionally, we used subsoil CO₂, moisture and temperature data obtained during 2014-2015.

2 Material and Methods

2.1 Experimental sites description

The study area is located in southeast Spain, the driest part of Europe. The two experimental sites, Amoladeras (N36°50'5''W2°15'1'') and Balsa Blanca (N36°56'26.0'' W2°01'58.8''), are found within the Cabo de Gata-Níjar Natural Park (Almería, Spain; Fig. 1) and are quite similar in terms of climate and ecosystem type. Both sites show a desert climate, according to Köppen classification (Bwh; Kottek et al., 2006), with a mean annual temperature of 18°C, and mean annual precipitation of approximately 220 mm.

The ecosystem type corresponds to *espartal*, a Mediterranean semiarid grassland where the dominant species is *Machrocloa tenacissima*. This ecosystem type is widely extended over the Western Mediterranean region ~~;~~ ~~in fact~~, in Cabo de Gata-Níjar Natural Park, a great fraction of agricultural areas that were abandoned over 1957-1994 resulted in *espartal* ecosystems (Alados et al., 2011; Alados et al., 2004). The functioning of both experimental sites can be divided into two main periods.

5 On one hand, the growing season usually extends from late autumn to early spring, when the temperature starts to rise and water resources have not yet become scarce (López-Ballesteros et al., 2016; Serrano-Ortiz et al., 2014). On the other hand, a long period of hydric stress, with high temperatures and scarce precipitation, results in a prolonged dry season that usually begins in May-June and ends in September-October, when the first autumn rainfall events occur. Additionally, water inputs derived from relevant dewfall episodes, which have been previously reported in the area (Uclés et al., 2014) ~~in the area~~, can rehydrate soil and plants during night and early morning hours.

10 Regarding the topographic, geologic and edaphic characteristics, both sites are located on an alluvial fan, where the main geological materials consist of quaternary conglomerates and Neogene-Quaternary sediments cemented by lime (caliche) (Rodríguez-Fernández et al., 2015) ~~and on~~ slopes of 2-6% (Rey et al., 2017) so no significant runoff occurs. Additionally, both sites present petrocalcic horizons. However, altitude and soil type differ. While Balsa Blanca (hereinafter BB) is located
15 at an altitude of 208 m and has MollicLithic Leptosols (Calcaric), Amoladeras (hereinafter AMO) is situated closer to sea level, at 65 m, and presents less developed soils Lithic Leptosol (Calcaric; Table 1).

Overall, as stated by Rey et al. (2011), these two experimental sites represent different degradation stages owing to their differing soil characteristics and surface fractions (Table 1). While BB has more deep and fertile soils and higher vegetation cover, AMO shows thinner and poorer soils and has half of Balsa Blanca's vegetation cover. Therefore, in accordance ~~to~~
20 with Rey et al. (2011; 2017), we considered that BB represents the “natural” site, being, currently, a representative ecosystem of the area, while AMO represents a “degraded” site with respect to BB. The stronger degradation effects observed in AMO (“degraded” site) compared to BB (“natural” site) are probably due to its proximity to populated areas. The main factor provoking degradation in this Mediterranean area was the increase of rural population from the beginning of the 20th century until late 1950s (Grove and Rackham, 2001). At that time, timber extraction, the use of tussock fiber for textile manufacturing and extensive farming were common economic activities likely increasing anthropic pressure on the “degraded” site. Afterwards, rural exodus during the mid-century involved the abandonment of these agriculture practices. However, although degradation drivers are not currently active, their effects are still observable in the area corresponding to a case of “relict” degradation (Puigdefábregas and Mendizábal, 2004).

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2.2 Meteorological and eddy covariance measurements

30 The net ecosystem-atmosphere exchange of water vapour, CO₂ and sensible and latent heat were measured in terms of fluxes via the eddy covariance (EC) technique. Thus, an EC station was installed at each experimental site, AMO and BB (with site codes “Es-Amo” and “Es-Agu” of the European Database Cluster <http://www.europe-fluxdata.eu>), where ambient and micrometeorological variables (detailed in Table 2) were monitored continuously since 2009. The EC footprint (i.e. actual

measured area) is well within the fetch (i.e. distance to a change in surface characteristics) at both sites. Regarding data processing, the half-hourly averaged fluxes were calculated from raw data collected at 10 Hz using EddyPro 5.1.1 software (Li-Cor, Inc., USA). Flux calculation, flux corrections and quality assessment were performed according to López-Ballesteros et al. (2016).

5 Additionally, flux measurements acquired under low-turbulence conditions were excluded from the analysis by using a friction velocity (u_*) threshold according to the approach proposed by Reichstein et al. (2005). The average u_* thresholds for the whole study period (i.e. 2009–2015) were 0.11 and 0.16 m s^{-1} , for AMO and BB, respectively. Furthermore, over the six years of measurements at both sites, data gaps due to low-turbulence conditions, instrument malfunction and theft were unavoidable and not randomly distributed, as noted by Ma et al. (2016). Therefore, the total annual fractions of missing half-
10 hourly net CO_2 fluxes accounted for $33 \pm 3 \%$ and $29 \pm 6 \%$ of night-time data and $8 \pm 6 \%$ and $14 \pm 5 \%$ of day-time data, for AMO and BB, respectively. Missing data were gap-filled by means of the marginal distribution approach proposed by Reichstein et al. (2005) and ~~errors-uncertainty~~ derived from the gap-filling procedure ~~were-was~~ calculated by using the variance of the measured data, which was calculated by introducing artificial gaps and repeating the standard gap-filling procedure. Twice the standard deviation of sums of total data was taken as the uncertainty for the several aggregating time periods we used in the analysis, from the variance of the gap-filled data, as explained by López-Ballesteros et al. (2016). The annual cumulative C balance was estimated, when possible, by integrating gap-filled half-hourly net CO_2 fluxes of good quality (0 and 1 quality flags, according to Mauder and Foken, 2004) over a hydrological year. In order to test the validity of both EC stations, we assessed the energy balance closure (Moncrieff et al., 1997) by computing the linear regression of half-hourly turbulent energy fluxes, sensible and latent heat fluxes ($H+LE$; W m^{-2}) against
15 available energy, net radiation less the soil heat flux (R_n-G ; W m^{-2}) with the whole six-years database. Storage term in the soil heat flux was included in the estimates while in case of sensible and latent heat fluxes, this term was negligible given the short height of the vegetation (~50 cm). The resulting slopes were 0.873 ± 0.002 ($R^2 = 0.907$) and 0.875 ± 0.001 ($R^2 = 0.920$) for AMO and BB, respectively.

2.3 Flux partitioning

25 In order to partition net CO_2 ecosystem exchange into Gross Primary Production (GPP) and ecosystem respiration (R_{eco}), we firstly modelled the ventilative CO_2 efflux by adapting the approach proposed by Pérez-Priego et al. (2013) with the results of previous studies of both sites (López-Ballesteros et al., 2016; 2017). Essentially, we aimed to isolate those moments when subterranean ventilation (V_v) dominates the net CO_2 fluxes (F_v) and biological fluxes are negligible. These moments correspond to daytime hours during the extremely dry periods. Accordingly, data were selected using the following
30 conditions: (i) net radiation $> 10 \text{ W m}^{-2}$, (ii) $8 < \text{daily averaged bowen ratio} < 10$, and (iii) daily soil water content (in bare soil) $< 10^{\text{th}}$ percentile (in AMO) and $< 20^{\text{th}}$ percentile (in BB). A less restrictive threshold was used in BB in order to get enough data to build the V_v model, since long-term data gaps occurred at this site during the summer seasons of 2012, 2014

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and 2015. Afterwards, in order to build the linear model of V_n , these selected F_e data (maximum quality; QC flag=0) were related to the friction velocity (u_e).

As the results show (Table 3; Fig. S1), the V_n model is uniquely valid for AMO. Therefore, we only applied the V_n model to AMO data, concretely, during those periods when ventilation (but not exclusively) occurs according to previous research (López-Ballesteros et al., 2017). Hence, the model was applied when: (i) net radiation $> 10 \text{ W m}^{-2}$, (ii) daily averaged bowen ratio > 4 , (iii) daily soil water content (in bare soil) $< 0.01 \text{ m}^3 \text{ m}^{-3}$, and (iv) σ_{swc} (daily variance of soil water content in bare soil) $< 5 \cdot 10^{-6} (\text{m}^3 \text{ m}^{-3})^2$. We use those moments with very low σ_{swc} in order to discern R_{eco} increases caused by rain pulses (Birch effect) from V_n fluxes during the dry season. Then, the modelled ventilative fluxes were subtracted from the measured net CO_2 exchange to obtain the CO_2 flux corresponding only to biological processes (i.e. biological F_e ; see Fig. S2).

Finally, the partitioning approach proposed by Lasslop et al. (2010) was applied to the biological F_e for both sites in order to obtain GPP and R_{eco} fluxes. We chose this approach given the determinant influence of hydric stress, in this case atmospheric drought (assessed via VPD), on the physiology of *Machrochloa tenacissima*, the dominant plant species of the studied semiarid ecosystems (Pugnaire et al., 1996; López-Ballesteros et al. 2016).

2.3-4 Enhanced Vegetation Index data series

We used Enhanced Vegetation Index (EVI) data acquired by the Moderate Resolution Imaging Spectroradiometer (MODIS), which is on board the Earth Observing System-Terra platform, in order to track vegetation dynamics at both experimental sites. The nominal resolution of EVI products (code "MOD13Q1") is 250 m at nadir and temporal resolution corresponds to 16-day compositing periods. The spatial coordinates used for AMO and BB were N36.8340°, E-2.2526° and N36.9394°, E-2.0341°, respectively. ~~The EVI is a three band vegetation index encompassing blue, red and NIR spectral reflectance information and has been widely used to track temporal variation of vegetation structure (Gao et al., 2000).~~

2.4-5 Vadose zone measurements

Subsoil CO_2 molar fraction, temperature and volumetric water content were measured at 0.05 m and 1.50 m below the surface (Table 2) from January 2014 to August 2015 at both experimental sites. In the case of the shallower CO_2 sensor, it was installed vertically with an in-soil adapter (211921GM, Vaisala, Inc., Finland) to avoid water entrance. Subsoil CO_2 molar fractions were sampled every 30 s and 5 min averages were stored in a data logger (CR3000 and CR1000, CSI; for AMO and BB, respectively). The deeper CO_2 sensor was equipped with a soil adapter for horizontal positioning (215519, Vaisala, Inc., Finland), consisting of a PTFE filter to protect to the CO_2 sensor from water. It was buried in the summer of 2013 and the measurements were made every 30 s and stored as 5 min averages in a datalogger (CR1000 and CR23X Campbell Sci., Logan, UT, USA, for AMO and BB, respectively). All CO_2 molar fraction records were corrected for variations in soil temperature and atmospheric pressure.

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2.6-7 Statistical analysis

All meteorological and soil variables monitored at each site were compared through computation of the non-parametric two-sided Wilcoxon summed rank test in order to detect those factors/variables influencing potentially distinct ecosystem functioning between sites. ~~This test was chosen because variables used satisfied the independence and continuity assumptions but not all were normally distributed. The confidence level used was 95%. The effect size was evaluated using the median of the difference between the samples (AMO minus BB), which was expressed as a standardized value (divided by its standard deviation: $Diff_{st}$; dimensionless) in order to be able to compare results among different variables. This test was chosen because variables used satisfied the independence and continuity assumptions but not all were normally distributed. The confidence level used was 95% and the effect size was evaluated using an estimator based on confidence intervals that corresponded to the median of the difference between the samples (Amoladeras minus Balsa Blanca) and was expressed as an absolute value (Diff) and as a standardized value ($Diff_{st}$; dimensionless), which was obtained by using the sample values of every site divided by its standard deviation in order to be able to compare results among different variables.~~ This analysis was performed by using three different periods: the whole study period, the period from May to September and the period from May to September during only daytime. These periods were selected given their demonstrated coincidence with high relevance of non-biological processes. All calculations were performed using R software version 3.2.5.

Additionally, in order to include the relationship between pressure and subsoil CO₂ variations as a potential factor influencing net CO₂ exchange (Sánchez-Cañete et al., 2013), we firstly calculated, separately for each site, Spearman correlation coefficients to determine the time step (6, 12, 24 or 72 hours) with the highest correlation between the differential transformation of pressure and the subsoil CO₂ molar fraction at 1.50 m.

3 Results

3.1 Ambient conditions ~~over the study period~~

Over the study period, the wettest hydrological year was 2009/2010, with annual precipitation of ~500 mm (ca. twice the ~~mean~~ annual ~~average~~ precipitation for both sites over the study period, Fig. 2). On the contrary, the driest year was 2013/2014, with annual precipitation of ~100 mm for both sites, less than half the annual average precipitation registered at ~~Amoladeras (AMO) and Balsa Blanca (BB)~~. Generally, ~~the~~ months with precipitation ~~higher than~~ exceeding 20 mm occurred from the beginning of autumn until midwinter, however, in case of 2009/2010, 2010/2011, 2012/2013 and 2014/2015, relevant ~~precipitation-rain~~ events were registered during spring months. By contrast, in 2013/2014, precipitation was always below 20 mm with the exception of November and December, for both sites, and June, in the case of AMO (Fig. 2a). Commonly, while maximum precipitation usually occurred from November to February, there was a remarkable drought period over summer months (June-August) when it scarcely ever rained (Fig. 2).

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Regarding air temperature (T_{air}) patterns, monthly averaged T_{air} ranged from 9.6 and 8.1 °C to 27.6 and 27.9 °C in AMO and BB, respectively, over the entire study period. Based on half-hourly averaged data, minimum and maximum T_{air} values registered were 0.1 and 37.9 °C in AMO, and -1.3 and 39.9 °C, in BB, respectively. On one hand, those months with T_{air} above 15 °C usually corresponded to April-November, approximately. Additionally, August was the month with the highest average T_{air} at both sites, with T_{air} ranges of 25.2 - 27.6 °C at AMO and 24.9 - 27.9 °C at BB, respectively (Fig. 2), over the study period. On the other hand, the lowest monthly average T_{air} usually occurred in January but sometimes also in December and February, with 11.2 – 12.3 °C at AMO and 8.1 - 14.1 °C at BB.

3.2 Annual carbon balances

The comparison of the annual ~~carbon (C)C~~ balance among sites was only possible for three hydrological years, 2009/2010, 2010/2011 and 2012/2013, due to long-term data gaps existing in BB during other years. The annual cumulative net CO₂ exchange was always positive for AMO (i.e., net C release), whereas BB was neutral or even acted as a C sink over the three years (Fig. 3). For example, in 2009/2010, the net C uptake measured in BB equated to $32 \pm 10 \text{ g C m}^{-2}$ while in AMO, a total amount of $185 \pm 10 \text{ g C m}^{-2}$ was released to the atmosphere (Fig. 3a). The year with the largest difference between sites was 2010/2011, with annual C release of 240 ± 8 and $-38 \pm 10 \text{ g C m}^{-2}$ in AMO and BB, respectively (Fig. 3b). Likewise, ~~2011/2012~~2012/2013 was the year when the lowest CO₂ release was measured in AMO with $163 \pm 7 \text{ g C m}^{-2}$ while a neutral C balance was measured in BB with $0 \pm 8 \text{ g C m}^{-2}$ (Fig. 3c).

Overall, a positive and saturating trend was observed at both sites during autumn months until December-February when cumulative net CO₂ releases start to decline. The autumn net CO₂ release (i.e., positive values) was usually higher in AMO than in BB, excepting for 2012/2013, and the declining slope was always higher in BB, meaning greater net C uptake rates. Although the pattern of the cumulative net CO₂ exchange showed differences between sites over autumn, winter and spring months, stronger discrepancies were found during summer droughts. Concretely, from April-May until August, BB showed neutral behavior while a remarkable positive trend was observed in AMO, denoting a large net CO₂ release.

3.3 Seasonal and diurnal net CO₂ exchanges

Long-term data loss occurred in BB during the springs of 2011/2012, 2013/2014 and 2014/2015 and summers of 2013/2014 and 2014/2015, when annual C balances could not be estimated. However, by observing the available seasonal data, it is noticeable that, maximum and minimum seasonal net CO₂ exchanges were very different between sites (Fig. 4). On one hand, maximum seasonal net CO₂ uptake was measured during winter (December-February) in AMO and over spring (March-May) in BB, when peaking net CO₂ uptake fluxes equated to -31 g C m^{-2} (winter 2011/2012) and -105 g C m^{-2} (spring 2010/2011) in AMO and BB, respectively. Additionally, net CO₂ uptake was only observed during three winters in the case of AMO, whereas it was frequently measured during both winter and spring in BB. On the other hand, cumulative net CO₂ release to the atmosphere occurred over all ~~types of~~ seasons in AMO, but acutely in summer, when maximum

seasonal net CO₂ release was always observed ranging from 111 to 153 g C m⁻². In contrast, in BB, the highest CO₂ effluxes usually occurred in autumn ranging from 25 to 74 g C m⁻², although significant CO₂ release was also observed in winter 2013/2014 and the summers of 2009/2010-2011/2012. ~~Regarding seasonal evapotranspiration (ET) fluxes, results showed a ~30% higher ET at BB, compared to AMO, during spring. Major inter-site differences in autumn occurred in the first and last year of study, when ET was 23% and 12% higher at BB, respectively (Fig. 5).~~

Comparing daily-scale net CO₂ exchange and ~~evapotranspiration (ET)~~ET fluxes with ~~punctual~~ Enhanced Vegetation Index (EVI) data, we can notice some similarities in the general patterns of both sites (Fig. 56). Roughly, there was a common annual pattern ~~at both sites~~ in which the highest values of EVI coincided with maximum net CO₂ uptake rates (i.e. negative net CO₂ fluxes), which in turn, corresponded to peaking ET fluxes. Additionally, a decreasing trend in EVI over the 6 years of study was also noticeable for both sites. However, some inter-site and inter-annual differences were evident (Fig. 56).

On one hand, there were two main differences between sites. Firstly, extreme net CO₂ release was measured uniquely in AMO during summer months (June-August), when maximum net CO₂ fluxes ranging from 31 to 68 g C m⁻² were measured (Fig. 56b). Over the study period, the monthly net CO₂ exchange of AMO during dry seasons was up to one hundred times higher than in BB (in August 2013), since monthly net CO₂ fluxes measured in BB were much lower, from -8 to 16 g C m⁻² (Fig. 56b). Besides the striking differences in summer net CO₂ exchange between sites, minor discrepancies were also found in ET fluxes and EVI for the same drought periods. In this regard, monthly averaged ET over the dry season equated to 13 ± 4 and 10 ± 4 mm for AMO and BB, respectively, and EVI was on average 4% higher in BB than in AMO (Fig. 56a, c). The second inter-site difference was the greater net CO₂ uptake over longer periods measured in BB. Concretely, the period during which the ecosystems acted as C sinks lasted on average 38 days longer in BB than in AMO annually (Table 34). Accordingly, the annual amount of C fixation ranged from 6-59 g C m⁻² at AMO and 15-129 g C m⁻² at BB, respectively, with the annual averaged net C uptake in BB 162% higher than at AMO (Table 34). Consequently, peaking EVI values were usually observed during March-April for both sites, however, over winter and spring months (growing period), EVI measured at BB was 3 - 37% higher than AMO, with the largest inter-site differences in 2009/2010 and 2014/2015 (Fig. 56a). Likewise, monthly averaged ET fluxes measured at BB over winter and spring months (December-May) were from 3 to 24% larger than those measured at AMO. Additionally, the growing period of the driest year (2013/2014) corresponded to the lowest monthly ET fluxes and the least difference between sites.

On the other hand, differences in the inter-annual variability of EVI ~~were found between years. Concretely, carbon and water fluxes were also observed; however, due to long term data losses at BB, accurate and comparative inter-annual analysis could not be done with carbon and water fluxes. In this sense,~~ 2009/2010 and 2013/2014 were the years with maximum and minimum annual precipitation and EVI observations, respectively, for both sites. ~~Concretely, in~~ In 2009/2010, EVI observations were 28% and 20% higher than the six-year averaged values in BB and AMO, respectively. In case of the driest year, 2013/2014, growing season (winter-spring) EVI was reduced 35% and 28% in BB and AMO, respectively. Nevertheless, the largest difference between sites in winter-spring EVI observations was found in 2014/2015, following the

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driest year, when BB showed a pattern very similar to those registered over the years ~~previous-prior~~ to the dry spell, while AMO still presented EVI values 21% below the six-year average (Fig. ~~5a6a~~).

3.4 Biological net CO₂ exchange, gross primary production, ecosystem respiration and water use efficiency

~~The results of the “biological” annual C₂ balance, which was obtained, in case of AMO, by applying the ventilation model, are in accordance with our hypotheses. Annual C emission was always measured at AMO, whereas BB acted as a neutral and mild C sink. On average, AMO emitted 32 g C m⁻² more than BB. At a monthly time scale, net CO₂ fluxes during autumn were, on average, ~4 times higher at BB, excepting the last study year, when the net CO₂ emission at AMO was 21 times greater than at the “natural” site. However, during winter and spring months, net CO₂ uptake was generally higher at the BB (Fig. 7a).~~

~~On average, during the six years of study, GPP, R_{eco} and Water Use Efficiency (WUE) were nine, two and ten times higher, respectively, at BB compared to AMO. Firstly, GPP was always higher at BB compared to AMO (Fig. 7c). Major differences occurred in autumn 2014/2015, when monthly cumulative GPP at BB was 32 times higher on average. Similarly, R_{eco} was generally higher, up to ~8 times (October 2014), at BB. However, respiratory fluxes were occasionally greater at AMO, from 2% to 31% higher, during spring and winter months of all studied years excepting 2013/2014 (Fig. 7b). Maximum inter-site differences in GPP and R_{eco} were found in winter and autumn 2014/2015, following the driest year, when monthly GPP was ~30 times higher at BB compared to AMO. Similarly, monthly R_{eco} was ~5 times greater at BB. Inter-site differences in partitioned fluxes could not be assessed during spring months due to the lack of data from BB. Secondly, WUE was lower at AMO during the whole study period, when maximum and minimum differences coincided with the highest and lowest differences in GPP between sites. On average, monthly WUE was 6 and 1.5 times higher at BB during winter and spring seasons. Major inter-site differences were found in autumn and winter 2014/2015 (Fig. 7d).~~

3.4.5 Differences in meteorological and soil variables between sites

Results from the two-sided Wilcoxon summed rank test (Table 45) showed significant differences (p-value < 0.05) between sites in most of the monitored meteorological variables. The few exceptions were the friction velocity (u*), when using the whole study period, the maximum wind speed registered every half-hour (WS_{max}), when analyzing May-September data, and the wind speed (WS) and precipitation when assessing daytime May-September data (Table 45). The great amount of observations (n ranged from 21410 to 205751) produced highly significant results (Table 45). Hence, the standardized difference between the samples (Diff_{st}) allowed us to quantitatively explore the differences between sites. Relevant differences (Diff_{st}>1) were found only for ~~pressure when using all databases, due to the distinct altitude of monitored sites. Regarding soil variables, important differences (Diff_{st}>1) between sites were detected in the~~ subsoil CO₂ molar fraction measured at 1.50 m depth (CO_{2, 1.50m}) for all periods, and during May-September months even when using only daytime data (Table 56). Concretely, CO_{2, 1.50m} was always higher in AMO, from 889 to 1109 ppm (Table S1, S2 and S3). Additionally, volumetric water content at 0.05m depth (VWC_{0.05m}) was also higher in AMO compared to BB but only during summer

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months (Table 56), when absolute differences were very small, ranging from 0.028 to 0.037 m³ m⁻³ (Tables S1, S2 and S3). In contrast, subsoil CO₂ molar fraction measured at 0.05 m depth (CO_{2, 0.05m}) was from 89 to 150 ppm higher in BB when analyzing dry season (May-September) daytime data (Table 56, S1, S2 and S3).

The temporal dynamics of subsoil CO₂ molar fractions revealed similar annual patterns between sites; generally however, CO_{2, 0.05m} was higher in BB, from 6 to 88%, while CO_{2, 1.50m} was always greater in AMO, from 31 to 97% (Fig. 86). On one hand, the maximum monthly averaged values of CO_{2, 0.05m} were registered in autumn, concretely, in November and October with 642 and 1120 ppm in AMO and BB, respectively, whereas minimum values occurred in September and August with 373 and 400 ppm at each site (Fig. 6a8a). On the other hand, peak~~ing~~ monthly averaged values of CO_{2, 1.50m} occurred in July for both sites, with 2751 and 1602 ppm in AMO and BB, respectively, although relatively high CO_{2, 1.50m} was also measured during November in BB. On the contrary, minimum values were observed in December and February, with 1364 and 735 ppm in AMO and BB, respectively (Fig. 6b8b).

Finally, results of the Spearman correlation analysis between ~~net CO₂ exchange pressure~~ and belowground CO₂ at 1.50 m depth (CO_{2, 1.50m}) showed a positive relationship between both absolute variables, which was stronger in case of AMO compared to BB with Spearman correlation coefficients (r_s) of 0.30 and 0.11, respectively (Table 7). In contrast, a negative relationship was found between pressure and CO_{2, 1.50m} between both variables. Additionally, although most of the coefficients were significant (p value < 0.05), with higher r_s correlations were found in AMO compared to BB (Table 67). Additionally, r_s Spearman correlation coefficients (r_s) were maxima at 12 h intervals for CO_{2, 1.50m} and pressure increments (dP_{12h}) at AMO, and ~~and~~ at 6 h intervals for CO_{2, 1.50m} and pressure increments (dP_{6h}, respectively) at BB, with r_s equal to -0.87 and -0.63, respectively.

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4 Discussion

Our results verify that land-degradation affects the C sequestration capacity of semiarid ecosystems, since relevant differences between sites were observed during the growing season, when greater net C uptake over longer periods was observed in the “natural” site (BB). However, contrary to what we previously hypothesized, much greater net C release was measured at the “degraded” site (AMO) over drought periods due to the predominance of subterranean ventilation (López-Ballesteros et al., 2017). In fact, the great difference in annual C budgets between sites (Fig. 23) was largely related to this process resulting in an average release of 196 ± 40 and -23 ± 20 g C m⁻² yr⁻¹ for the “degraded” (AMO) and “natural” (BB) sites, respectively. Even when assessing only the biological net CO₂ exchange, by subtracting the non-biological CO₂ flux when feasible, the “degraded” site emitted 32 g C m⁻² more than the “natural” site. In this regard, the ecosystems’ functioning could be divided into three different phases. The first phase corresponded to the autumn months, when the first rainfall events after the dry summer (i.e. rain pulses) activated the soil microbiota triggering respiratory CO₂ emissions as previously measured at the same experimental sites (López-Ballesteros et al., 2016; Rey et al., 2017). During this phase, maximum net CO₂ release emission was observed at BB at the “natural” site, however higher CO₂ release was usually

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measured at AMO which exceeded the biological net CO₂ flux observed in the “degraded” site (Fig. 47a). In fact, as hypothesized, R_{eco} was generally higher at the “natural” site during autumn months (Fig. 7b); maybe probably due to the greater pool of organic carbon (Table 1) and the hydric stress experienced prior to rewetting, the potential differences in microbial communities (Rey et al., 2017), and the greater pool of soil inorganic carbon (Emmerich, 2003; Table 1) between sites. The second phase comprised the growing period, when plants photosynthesized and also respired along with microorganisms under milder temperatures and better hydric conditions. During this phase, larger net CO₂ uptake was measured in at BB the “natural” site, concretely 162% more than in AMO the “degraded” site (Table 34) due to the higher vegetation cover and more fertile soils (Table 1) of the “natural” site. Accordingly, GPP estimates were, on average, nine times higher at the “natural” site (Fig. 7c) and Moreover, this these results was were supported by the lower ET, WUE and EVI values obtained in AMO the “degraded” site during winter and spring months of over the whole study period (Fig. 5, 56 and 7d). The third phase consisted of the dormancy period when water scarcity and high temperatures constrained biological activity. During this period, as stated previously, a neutral C balance was observed in at BB the “natural” site while extreme CO₂ release was measured in at the “degraded” site AMO, as expected (Fig. 4, 6b), where ventilative fluxes were dominant.

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In order to detect potential factors driving the observed differences in the C balances, we checked whether soil and meteorological variables differed between sites. Our results demonstrated that some factors typically influencing gross primary production (GPP) and ecosystem respiration (R_{eco}), and hence net ecosystem CO₂ exchange, such as photosynthetic photon flux density (PPFD; Michaelis and Menten, 1913), precipitation (Berner et al., 2017; Jongen et al., 2011), vapor pressure deficit (VPD; Lasslop et al., 2010) and soil and air temperature (Lloyd and Taylor, 1994), did not differ between sites (Table 45, 56). Conversely, some differences were found for shallow soil volumetric water content (VWC_{0.05m}) during dry seasons (Table 56), when VWC_{0.05m} was two times higher in AMO than in BB, but absolute differences were slight, from 0.028 to 0.037 m³ m⁻³ (Tables S1,S2 and S3). Hence, although the important influence of soil moisture in both GPP and R_{eco} is known (Tang and Baldocchi, 2005), we believe that differences in VWC_{0.05m} are not relevant enough relevant to cause the differing ecosystems² functioning observed over the drought period. Additionally, we think that this inter-site difference in VWC_{0.05m} could be instrumental, or due to the spatial variability of VWC_{0.05m} derived from the heterogeneity of soil morphological characteristics, since we only used one sensor at each site. Similarly, important differences were not detected in several variables linked to subterranean ventilation, such as the friction velocity (u_{*}; Kowalski et al., 2008), wind speed (WS; Rey et al., 2012), half-hourly maximum wind speed (WS_{max}) and net radiation, which has been positively correlated to ventilative CO₂ fluxes (López-Ballesteros et al., 2017), when using the analysis periods when this process is supposed to be relevant, namely daytime hours during the dry seasons (Table 45). In contrast, inter site differences were found in net radiation (Table 4). However, although no turbulence and wind speed inter-site differences were found, interconnectivity of soil pores and fractures is probably higher at AMO the “degraded” site (Table 1) due to its higher gravel and rock fractions (Table 1), which could lead to an enhanced penetration of eddies within the vadose zone (Pérez-Priego et al., 2013).

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Apart from that, outstanding differences between sites were observed in subsoil CO₂ molar fractions measured at 0.05 and 1.50 m depths (CO_{2, 0.05m} and CO_{2, 1.50m}, respectively; Table 56). On one hand, CO_{2, 0.05m} was generally higher in ~~BB-the~~ “natural” site given its lower degradation level, which probably promotes a higher microbial activity supported by higher vegetation density and soil fertility (Table 1) especially during spring (Fig. 68), as pointed also by Oyonarte et al. (2012). On the other hand, CO_{2, 1.50m} values were acutely higher in ~~AMO~~the “degraded” site, by up to 1000 ppm compared to ~~BB-the~~ “natural” site (Tables S1, S2 and S3). Therefore, we suggest that CO_{2, 1.50m} is the main factor responsible for the inter-site differences in net CO₂ fluxes over the dry season. In this regard, previous research has suggested two potential origins of this vadose zone CO₂, geological degassing (Rey et al. 2012b) and/or subterranean translocation of CO₂ in both gaseous and aqueous phases (López-Ballesteros et al. 2017). –However, not only the amount of subsoil CO₂ matters but also how effective is its transport, since both determine the net CO₂ release from the vadose zone to the atmosphere. In this context, Oyonarte et al. (2012) found, in the same study area (Cabo de Gata-Níjar Natural Park), that soils with degradation symptoms, such as lower SOC, depleted biological activity, coarser texture and worse structure, showed higher soil CO₂ effluxes over the dry season. Additionally, soil CO₂ effluxes measured during summer months correlated positively with the fraction of rock outcrops, suggesting that deteriorated soil physical conditions actually enhanced vertical transfer of CO₂-rich air from subsoil to the atmosphere (Oyonarte et al., 2012). In fact, correlation analysis between CO_{2, 1.50m} and net CO₂ exchange/atmospheric pressure (Table 67) showed a stronger relationship between ~~both–these~~ variables at ~~AMO~~the “degraded” site. In this sense, ecosystem degradation could provoke a greater exposure of subsoil CO₂ to the pressure effect, as described by Sánchez-Cañete et al. (2013), probably due to a higher fraction of bare soil, coarser structure, differing porosity type and/or thinner soil depth (Table 1).

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Regarding EVI data, we found a discrepancy between GPP estimates and EVI values since, contrary to what is observed in EVI results, we observed that GPP was always higher at the “natural” site compared to the “degraded” site. We think that this is due to the different spatial scales defining each measurement. MODIS pixels have an area of ~6.25 ha while the eddy covariance footprint corresponds to a smaller area of ~1 ha. Therefore, there is an EVI uncertainty that stems from the influence of other surface elements apart from vegetation, such as bare soil or outcrops within the pixel, which is our case. In fact, previous studies confirm the discrepancy between MODIS- and EC-derived GPP estimates, especially on sparse vegetation areas with low productivity (Gilabert et al., 2015). However, ~~Regarding~~ EVI data, ~~these results~~ have allowed us to complement our findings based on CO₂ fluxes, especially when EC data losses occurred. For instance, the declining trend observed from 2009/2010 until the end of the study period, for both sites, was not noticeable from EC data alone (Fig. 56). This long-term decrease in EVI may be related to a gradual drying following the wettest year (2009/2010), when extraordinarily high precipitation (twice the mean –annual ~~average~~–precipitation for both sites over the study period) occurred. This EVI pattern also ~~denotes~~ suggests a pulse-like behaviour of ecosystem vegetation over the inter-annual time scale.

Moreover, in addition to demonstrating that degradation can influence the biological activity of ecosystems' vegetation, EVI results also showed that degradation level can modulate how an ecosystem responds to a short-term ~~change~~ disturbance. A clear example is the dry spell experienced in 2013/2014, when a reduction in EVI was measured during the growing season in both sites, i.e. 35% and 28% in ~~BB~~ the "natural" and "degraded" sites ~~AMO~~, respectively. However, a year later (2014/2015), EVI values below the six-year average were observed only at the "degraded" site ~~AMO~~ (21% lower; Fig. 5a6a) and major inter-site differences were found for GPP, R_{eco} and WUE during autumn and winter months (Fig. 7b, 7c and 7d). Accordingly, the "natural" site (~~BB~~) seemed to be more stable-resilient than the "degraded" site (~~AMO~~) against the short-term disturbance, since the effect of drought persisted in AMO even during the following year, while BB recovered to a pre-perturbation state within the same period (Fig. 5a6a). As a result, ecosystem resilience ~~defined by~~ (Holling, (1973) ~~as the amount of disturbance that a system can withstand without changing state,~~ was lessened by long-term disturbances such as land degradation, making degraded ecosystems more vulnerable to climate extremes (Reichstein et al., 2013). In this sense, mitigation policies to confront land degradation should be focused on prevention programs since ecosystem restoration does not recover complete ecosystem functionality (Lal, 2001; Moreno-Mateos et al., 2017). Moreover, even after several decades, relict degradation legacies can remain (Alados et al., 2011).

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15 5 Conclusions

The present study can be seen as a step forward to better understanding the effect of land degradation on the intricate network of multi-scale processes, factors and structures that define ecosystems' biological productivity and ultimately control their C balances. Despite some limitations, such as long-term data gaps, this research demonstrates that continuous ecosystem-scale EC observations remain crucial to comprehend how climate and land use change can modify the C sequestration capacity of ecosystems. In fact, annual average release of 196 ± 40 and -23 ± 20 g C m⁻² yr⁻¹ for the "degraded" and "natural" (i.e. site of reference) sites were measured, respectively. Additionally, larger net CO₂ uptake over longer periods was observed at the "natural" site, concretely an amount of C 162% higher compared to the "degraded" site, whereas much greater net CO₂ release was measured at the "degraded" site during drought periods. Accordingly, the estimates of gross primary production, ecosystem respiration and water use efficiency were, on average, nine, two and ten times higher in the "natural" site, respectively. Future research should be based on the continuity of long-term monitoring stations, such as eddy covariance stations, in order to calibrate and validate satellite data, reduce uncertainties in the relationships between ecosystem productivity, land degradation and climate change and finally, to improve the predictive ability of current terrestrial C models.

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Data availability

The eddy covariance data are available in the European Database Cluster (<http://www.europe-fluxdata.eu>) where experimental sites have the codes “Es-Amo” and “Es-Agu”. Other data can be obtained by contacting the corresponding author.

5 Author contribution

FD, CO, AK and PSO designed the experiment. ALB, PSO, EPSC and MRM calibrated the sensors, collected the data and maintained the field instrumentation. EPSC designed subsoil data acquisition system and MRM processed subsoil data. ALB processed the eddy covariance data, made the figures and tables and wrote the manuscript. All authors reviewed the manuscript.

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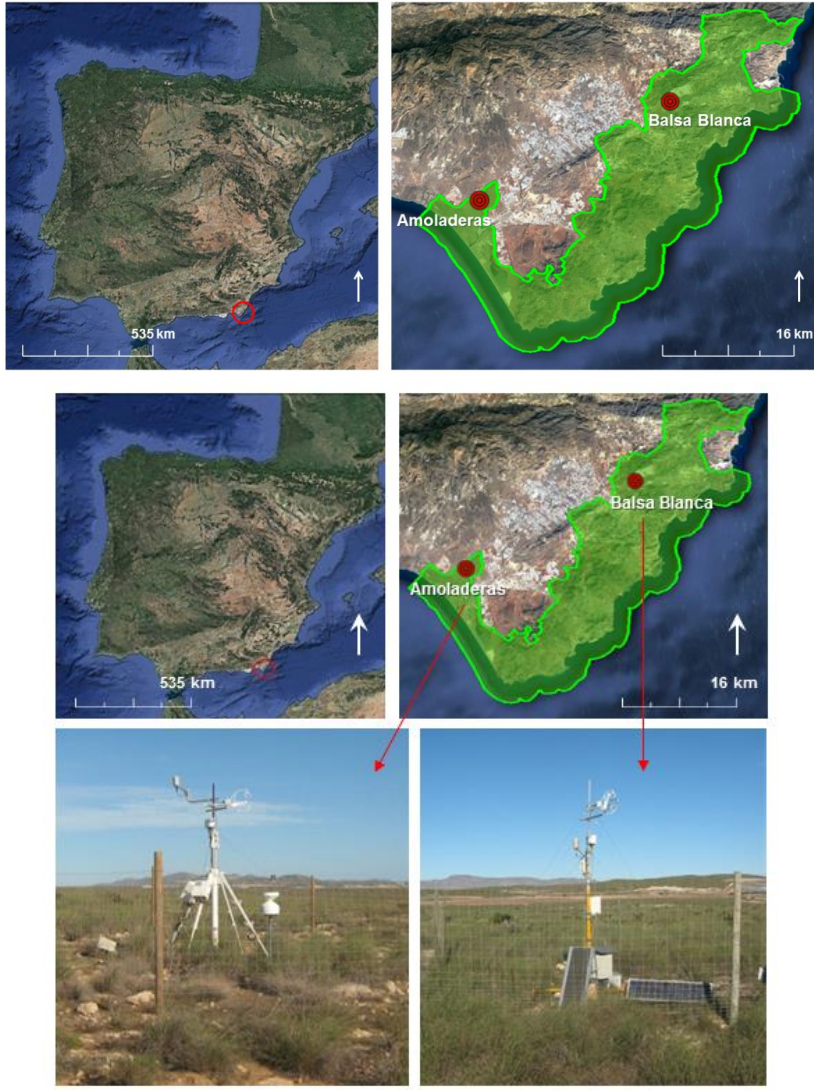
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5 | Figure 1: Location (above) and photographs (below) of the experimental sites. Green area represents the Cabo de Gata-Níjar Natural Park (Almeria, Spain).

Table 1: Site characteristics, surface fractions and soil properties of both experimental sites studied. Asterisks denote significant differences (p-value<0.05). Adapted from Rey et al., (2011).

	Amoladeras	Balsa Blanca
<i>Site characteristics</i>		
Location	Almeria N36°50'52" W2°15'12"	Almeria N36°56'02" W2°1'58"
Altitude (m)	65	208
Climate	Subtropical, dry, semiarid	Subtropical, dry, semiarid
Annual T	18	18
Annual P	220	220
Predominant sp.	<i>Machrocloa tenacissima</i>	<i>Machrocloa tenacissima</i>
<i>Surface fractions</i>		
Vegetation cover (%)	23.1 ± 2.4*	63.2 ± 5.2*
Litter (%)	10.5 ± 2.0	8.1 ± 1.9
Biological crust (%)	23.1 ± 2.8	18.2 ± 3.8
Bare soil (%)	8.1 ± 0.1*	0.3 ± 0.3*
Gravel (%)	21.1 ± 0.1*	8.6 ± 2.5*
Rock (%)	14.0 ± 1.2*	1.5 ± 0.5*
<i>Soil properties</i>		
Soil type	Lithic Leptosol (Calcaric)	Mollic Lithic Leptosol (Calcaric)
Maximum soil depth (cm)	10	20
Soil texture class	Sandy loam	Sandy loam
Clay (%)	14.6	16.1
Silt (%)	27.0	22.8
Sand (%)	58.4	61.1
Bulk density (g cm ⁻³)	1.11 ± 0.04	1.25 ± 0.09
SOC (kg m ⁻²)	1.24	4.64
Carbonates (%)	14	2

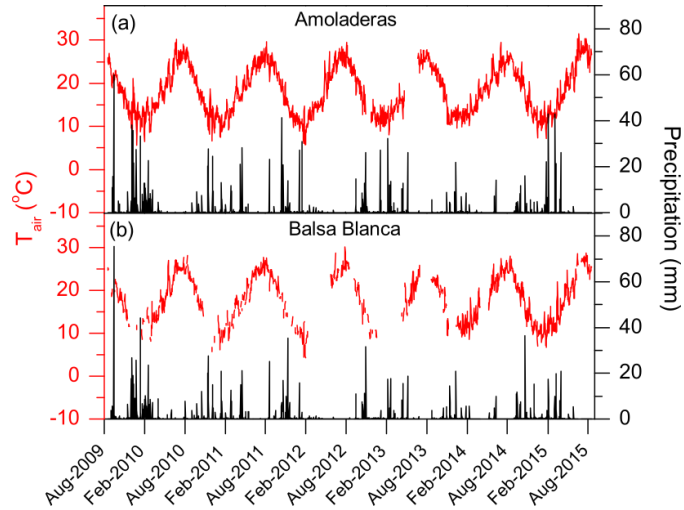


Figure 2: Daily averages of air temperature (T_{air}) and precipitation in (a) Amoladeras and (b) Balsa Blanca.

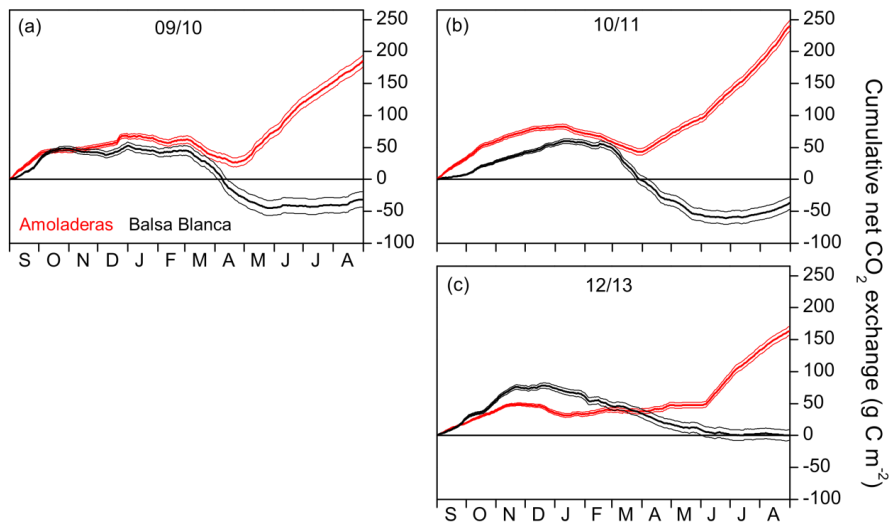


Figure 3: Cumulative annual net CO_2 exchange over the three hydrological years without long-periods of missing data in both experimental sites, Amoladeras (red lines) and Balsa Blanca (black lines). Negative values denote net carbon uptake while positive values denote net carbon release. Thin lines indicate uncertainty derived from the gap-filling procedure.

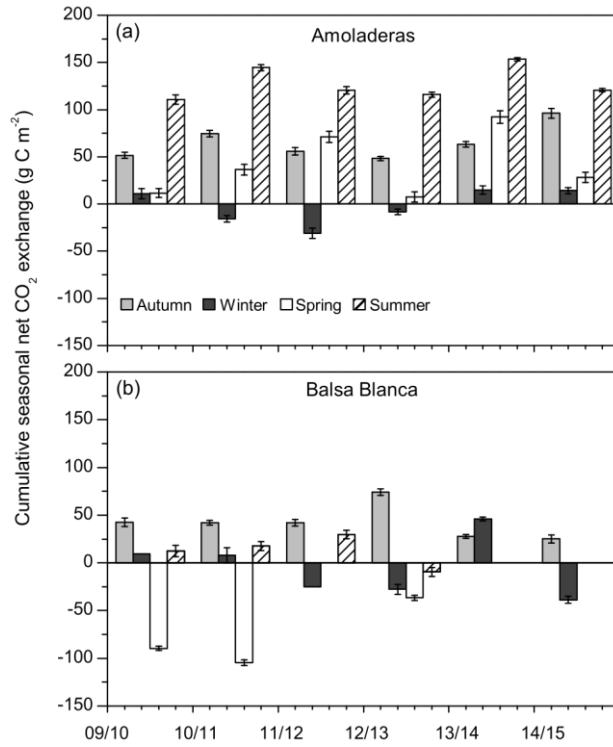
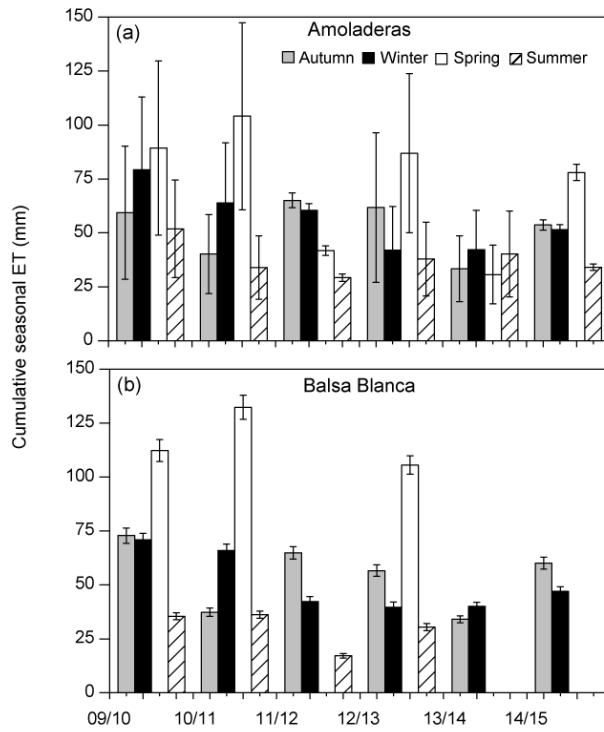


Figure 4: Cumulative seasonal net CO₂ exchange over the study period in both experimental sites. Negative values denote net carbon uptake while positive values denote net carbon release. In case of Balsa Blanca, lacking bars correspond to long-term data losses (>50% data).



Con formato: Centrado

Figure 5: Cumulative seasonal evapotranspiration fluxes (ET) over the study period in both experimental sites. In case of Balsa Blanca, lacking bars correspond to long-term data losses (>50% data). Error bars denote uncertainty derived from the gap-filling procedure.

Con formato: Fuente: Negrita, Sin Cursiva

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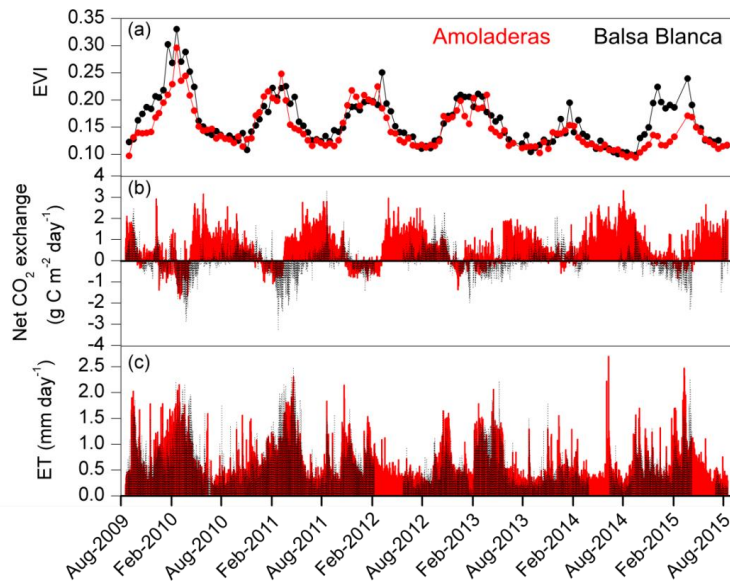
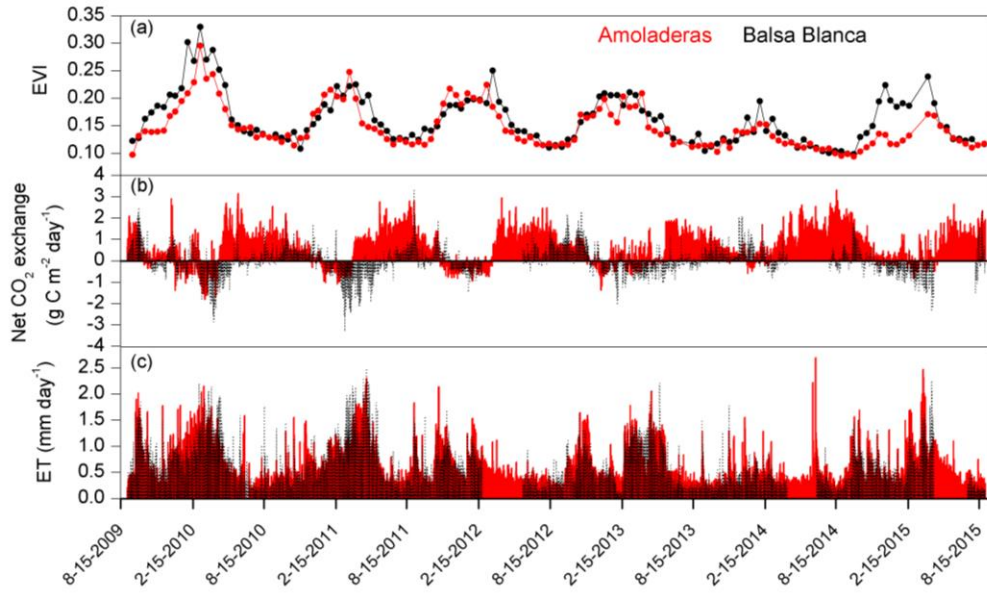


Figure 65: Time series of (a) Enhanced Vegetation Index (EVI), (b) daily net CO₂ exchange and (c) daily evapotranspiration fluxes measured in Amoladeras (red lines and dots) and Balsa Blanca (black lines and dots) over six hydrological years (2009-2015). Long-term data losses correspond to periods of several months when ET and CO₂ fluxes are absent.

5

Table 34: Number of days with daily net CO₂ uptake and the related total C absorbed for every hydrological year and every field site of the study. Asterisks denote those years with abundant data losses (~30% data).

Site	Amoladeras		Balsa Blanca	
	N. days of net CO ₂ uptake	Total net CO ₂ uptake (g C m ⁻²)	N. days of net CO ₂ uptake	Total net CO ₂ uptake (g C m ⁻²)
09/10	58	-59 ± 7	196	-125 ± 12
10/11	86	-41 ± 4	160	-129 ± 10
11/12	114	-43 ± 5	104*	-40 ± 6*
12/13	103	-31 ± 4	212	-96 ± 8
13/14	31	-6 ± 12	64*	-15 ± 5*
14/15	59	-14 ± 3	172*	-103 ± 9*

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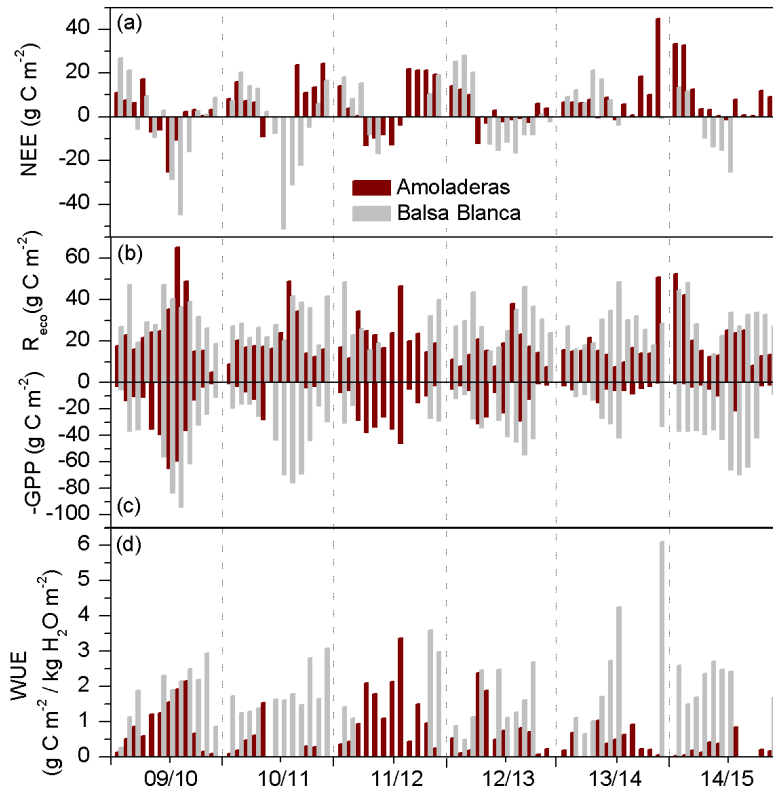


Figure 7: Monthly cumulative fluxes of (a) biological net ecosystem CO₂ exchange, (b) ecosystem respiration (R_{eco}), (c) negative gross primary production and (d) water use efficiency over the six hydrological years of study (2009-2015) for Amoladeras (dark red) and Balsa Blanca (grey). Lacking bars correspond to long-term data losses.

Table 5: Results of the two-sided Wilcoxon summed rank test used to assess differences among meteorological variables measured at each experimental site over all periods, from May to September and from May to September during daytime, separately. Medians of the difference between the samples (Amoladeras minus Balsa Blanca) in standardized terms ($Diff_{st}$) and number of observations are detailed. Significant results (p -value<0.05) are denoted with asterisks, and bold values represent those variables with $Diff_{st}$ between sites above 1.

Variables	All periods		May - September		May - September Daytime	
	$Diff_{st}$	n	$Diff_{st}$	n	$Diff_{st}$	n
PPFD ($\mu\text{mol m}^{-2} \text{s}^{-1}$)	0.0009*	205751	0.0009*	84491	0.1378*	38963
Net radiation (W m^{-2})	-0.0457*	197924	-0.0476*	81019	-0.1205*	38963
T_{air} ($^{\circ}\text{C}$)	0.0310*	182240	0.1935*	77866	0.0502*	37480
VPD (hPa)	0.0783*	166918	0.1370*	71474	-0.0938*	34430
RH (%)	-0.1636*	197649	-0.1031*	80950	0.1784*	38935
u_* (m s^{-1})	-0.0054	166346	-0.0563*	71194	-0.1340*	34284
WS (m s^{-1})	0.1628*	166359	0.0793*	71195	0.0165	34285
WS_{max} (m s^{-1})	0.1001*	165458	0.0124	70635	-0.0796*	33994
Pressure (hPa)	1.6999*0.37	16635916	1.8696*0.58	711957	1.8633*0.56	342853
	37*	6336	28*	1188	02*	4280
Precipitation (mm)	-1.95E-05*	204892	-4.84E-05*	83860	5.32E-05	38963

Con formato: Interlineado: sencillo

Con formato: Fuente: Sin Negrita

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Table 56: Results of the two-sided Wilcoxon summed rank test used to assess differences among soil variables measured at each experimental site over all periods, from May to September and from May to September during daytime, separately. Medians of the difference between the samples (Amoladeras minus Balsa Blanca) in standardized terms ($Diff_{st}$) and number of observations are detailed. Significant results (p -value<0.05) are denoted with asterisks, and bold values represent those variables with $Diff_{st}$ between sites above 1.

Variables	All periods		May - September		May - September Daytime	
	$Diff_{st}$	n	$Diff_{st}$	n	$Diff_{st}$	n
$\text{CO}_2, 0.05\text{m}$ (ppm)	-0.4027*	46340	-0.6578*	21413	-1.1396*	9816
$\text{CO}_2, 1.50\text{m}$ (ppm)	1.1196*	50133	1.3517*	24347	1.3062*	11385
$T_{0.05\text{m}}$ ($^{\circ}\text{C}$)	0.0927*	46337	-0.1160*	21410	-0.2119*	9813
$T_{1.50\text{m}}$ ($^{\circ}\text{C}$)	0.1476*	50137	-0.0591*	24350	-0.0834*	11385
$\text{VWC}_{0.05\text{m}}$ ($\text{m}^3 \text{m}^{-3}$)	0.8265*	52353	1.2724*	25231	1.2839*	11303
$\text{VWC}_{1.50\text{m}}$ ($\text{m}^3 \text{m}^{-3}$)	-0.8385*	53865	0.0674*	24570	0.0547*	11462

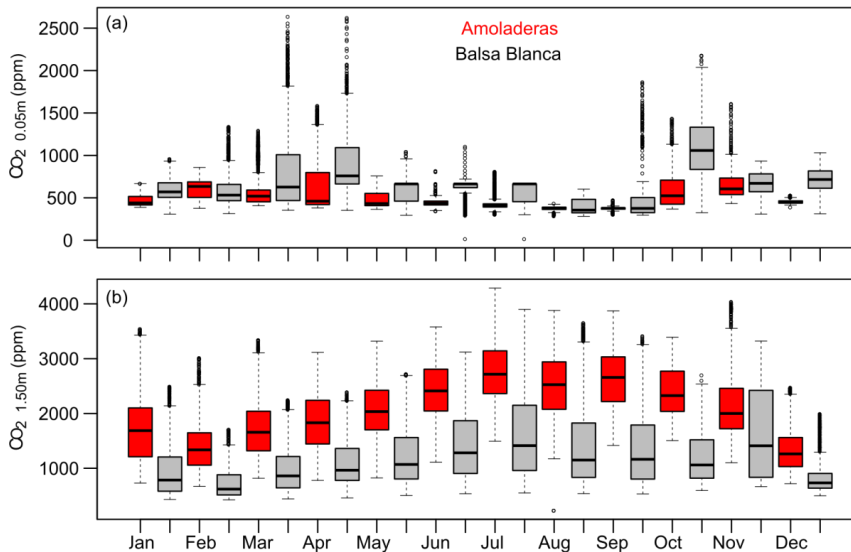


Figure 86: Box-and-whisker plots of CO₂ molar fractions measured at (a) 0.05 m and (b) 1.50 m belowground in Amoladeras (red boxes) and Balsa Blanca (grey boxes) from January 2014 to August 2015. The box extends from the first (Q1) to the third quartiles (Q3) and the central line represents the median (50% percentile). Dots represent outliers; upper whisker is located at the smaller of the maximum value and Q3 + 1.5 IQR (Interquartile Range), and lower whisker is located at the larger of the minimum value and Q1 - 1.5 IQR.

Table 7: Spearman correlation coefficients (r_s) for every paired simple correlation among maximum quality net CO₂ exchange fluxes ($\mu\text{mol m}^{-2} \text{s}^{-1}$), absolute and differential pressure (hPa) at 6, 12, 24 and 72hours time-step and absolute and differential CO₂ molar fraction measured at 1.50 m below ground (ppm) at the same time-steps. Bold values represent the highest correlation coefficients while shaded ones denotes non-significant relationships ($p\text{-values} > 0.05$).

	Amoladeras						Balsa Blanca					
	Net CO ₂ exchange	P	dP _{6h}	dP _{12h}	dP _{24h}	dP _{72h}	Net CO ₂ exchange	P	dP _{6h}	dP _{12h}	dP _{24h}	dP _{72h}
CO ₂ _{1.50m}	0.30	-0.66	-0.33	-0.46	-0.56	-0.55	0.11	-0.33	-0.51	-0.53	-0.62	-0.45
dCO ₂ _{1.50m 6h}	0	-0.1	-0.84	-0.5	-0.26	-0.01	0.02	-0.07	-0.63	-0.19	-0.13	-0.04
dCO ₂ _{1.50m 12h}	0.06	-0.08	-0.57	-0.87	-0.55	-0.05	0.03	-0.03	-0.46	-0.50	-0.31	-0.03
dCO ₂ _{1.50m 24h}	0.03	-0.13	-0.47	-0.78	-0.85	-0.15	0.02	-0.04	-0.40	-0.59	-0.58	-0.04
dCO ₂ _{1.50m 72h}	0	-0.28	-0.28	-0.49	-0.64	-0.74	0.00	-0.13	-0.28	-0.43	-0.57	-0.48

Comentario [T1]: Add this new info (NEE) to results and discussion sections.

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Table 6: Spearman correlation coefficients (r_s) for every paired simple correlation among maximum quality net CO_2 exchange fluxes ($\mu\text{mol m}^{-2} \text{s}^{-1}$), absolute and differential pressure (hPa) at 6, 12, 24 and 72 hours time step and absolute and differential CO_2 molar fraction measured at 1.50 m below ground (ppm) at the same time steps. Bold values represent the highest correlation coefficients while shaded ones denotes non-significant relationships ($p\text{-value} > 0.05$).

	Amoladeras					Balsa Blanca				
	P	dP _{6h}	dP _{12h}	dP _{24h}	dP _{72h}	P	dP _{6h}	dP _{12h}	dP _{24h}	dP _{72h}
$\text{CO}_{2,1.50\text{m}}$	-0.66	-0.33	-0.46	-0.56	-0.55	-0.33	-0.51	-0.53	-0.62	-0.45
d $\text{CO}_{2,1.50\text{m}_6\text{h}}$	-0.1	-0.84	-0.5	-0.26	-0.01	-0.07	-0.63	-0.19	-0.13	-0.04
d $\text{CO}_{2,1.50\text{m}_12\text{h}}$	-0.08	-0.57	-0.87	-0.55	-0.05	-0.03	-0.46	-0.50	-0.31	-0.03
d $\text{CO}_{2,1.50\text{m}_24\text{h}}$	-0.13	-0.47	-0.78	-0.85	-0.15	-0.04	-0.40	-0.59	-0.58	-0.04
d $\text{CO}_{2,1.50\text{m}_72\text{h}}$	-0.28	-0.28	-0.49	-0.64	-0.74	-0.13	-0.28	-0.43	-0.57	-0.48

Supplementary Material

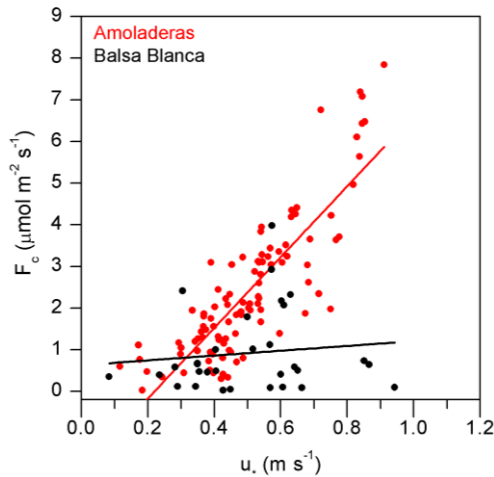


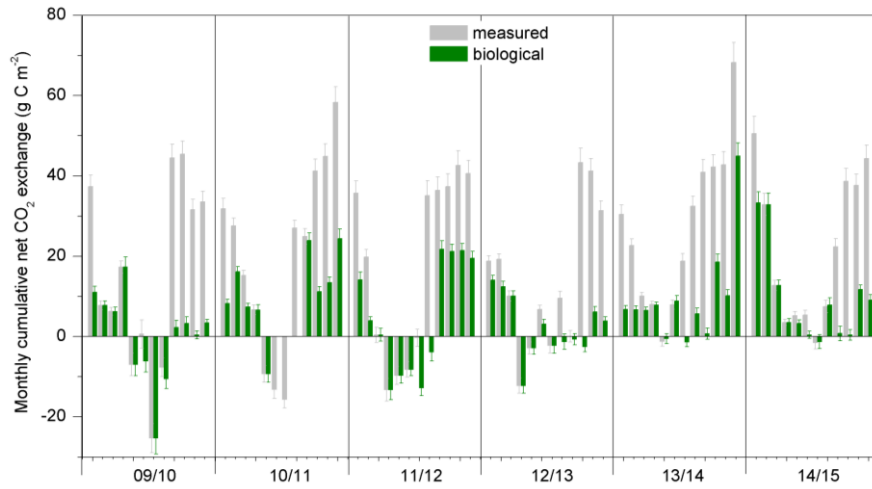
Figure S1: Half-hourly net CO_2 fluxes of maximum quality (QC flag=0) versus friction velocity (u_*) corresponding to daytime hours during the extremely dry periods when subterranean ventilation dominates the net CO_2 flux. Red and black dots represent Amoladeras and Balsa Blanca, respectively.

Con formato: Centrado

Con formato: Fuente: Negrita, Sin Cursiva

Con formato: Interlineado: sencillo

Con formato: Fuente: Negrita, Sin Cursiva



Con formato: Centrado

Figure S2: Cumulative measured and biological (after applying the ventilation model) net CO₂ exchange for every month of the study period (5 hydrological years: 2009-2015) in Amoladeras.

Con formato: Fuente: Negrita, Sin Cursiva

Con formato: Interlineado: sencillo

Con formato: Centrado

5

Table S1: Results of the two-sided Wilcoxon summed rank test used to assess differences between meteorological variables measured at each experimental site over the all periods. Medians of the absolute (Diff) and standardized differences (Diff_{st}) between the samples (Amoladeras minus Balsa Blanca), p-values and number of observations (n) are detailed.

Variables	All periods			
	Diff	Diff _{st}	p-value	n
PPFD ($\mu\text{mol m}^{-2} \text{s}^{-1}$)	0.4701	0.0009	0	205751
Net radiation (W m^{-2})	-8.8620	-0.0457	0	197924
T _{air} (°C)	0.1928	0.0310	6.42E-10	182240
VPD (hPa)	0.5358	0.0783	4.38E-104	166918
RH (%)	-3.1947	-0.1636	6.39E-244	197649
u _s (m s^{-1})	-0.0016	-0.0054	0.1214	166346
WS (m s^{-1})	0.3621	0.1628	0	166359
WS _{max} (m s^{-1})	0.4327	0.1001	5.53E-132	165458
Pressure (hPa)	19.00612 3226	1.69990.3 737	0	166359 166336
Precipitation (mm)	-0.0001	-1.95E-05	0.0003	204892
CO _{2, 0.05m} (ppm)	-93.8644	-0.4027	0	46340
CO _{2, 1.50m} (ppm)	888.9239	1.1196	0	50133
T _{0.05m} (°C)	0.9215	0.0927	5.51E-21	46337
T _{1.50m} (°C)	0.5780	0.1476	1.43E-51	50137
VWC _{0.05m} ($\text{m}^3 \text{m}^{-3}$)	0.0372	0.8265	0	52353
VWC _{1.50m} ($\text{m}^3 \text{m}^{-3}$)	-0.0123	-0.8385	0	53865

Table S2: Results of the two-sided Wilcoxon summed rank test used to assess differences between meteorological variables measured at each experimental site from May to September. Medians of the absolute (Diff) and standardized differences (Diff_{st}) between the samples (Amoladeras minus Balsa Blanca), p-values and number of observations (n) are detailed.

5

Variables	May - September			
	Diff	Diff _{st}	p-value	n
PPFD ($\mu\text{mol m}^{-2} \text{s}^{-1}$)	0.5230	0.0009	8.11E-147	84491
Net radiation (W m^{-2})	-10.9546	-0.0476	4.78E-167	81019
T _{air} (°C)	0.8378	0.1935	1.53E-145	77866
VPD (hPa)	1.1136	0.1370	4.50E-90	71474
RH (%)	-2.0229	-0.1031	1.21E-38	80950
u _s (m s^{-1})	-0.0186	-0.0563	8.47E-31	71194
WS (m s^{-1})	0.1720	0.0793	1.84E-31	71195
WS _{max} (m s^{-1})	0.0498	0.0124	0.0749	70635
Pressure (hPa)	18.67072 2120	1.86960.5 828	0	71195 71188
Precipitation (mm)	0.0000	-4.84E-05	4.29E-05	83860
CO _{2, 0.05m} (ppm)	-89.2404	-0.6578	9.99E-207	21413
CO _{2, 1.50m} (ppm)	1109.1440	1.3517	0	24347
T _{0.05m} (°C)	-0.8057	-0.1160	3.04E-16	21410
T _{1.50m} (°C)	-0.1570	-0.0591	3.13E-07	24350
VWC _{0.05m} ($\text{m}^3 \text{m}^{-3}$)	0.0286	1.2724	0	25231
VWC _{1.50m} ($\text{m}^3 \text{m}^{-3}$)	0.0011	0.0674	3.51E-18	24570

Table S3: Results of the two-sided Wilcoxon summed rank test used to assess differences between meteorological variables measured at each experimental site from May to September during daytime. Medians of the absolute (Diff) and standardized differences (Diff and Diff_{st}) between the samples (Amoladeras minus Balsa Blanca), p-values and number of observations (n) are detailed.

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Variables	May – September Daytime			
	Diff	Diff _{st}	p-value	n
PPFD ($\mu\text{mol m}^{-2} \text{s}^{-1}$)	64.3400	0.1378	5.62E-42	38963
Net radiation (W m^{-2})	-21.9460	-0.1205	4.80E-34	38963
T _{air} (°C)	0.1939	0.0502	1.39E-06	37480
VPD (hPa)	-0.7579	-0.0938	3.87E-20	34430
RH (%)	2.9921	0.1784	1.45E-65	38935
u* (m s^{-1})	-0.0430	-0.1340	3.76E-87	34284
WS (m s^{-1})	0.0356	0.0165	0.1341	34285
WS _{max} (m s^{-1})	-0.2920	-0.0796	5.83E-14	33994
Pressure (hPa)	18.40852 <u>1101</u>	1.86330.5 <u>602</u>	0	3428534 <u>280</u>
Precipitation (mm)	0.0000	5.32E-05	0.9875	38963
CO _{2, 0.05m} (ppm)	-149.513	-1.1396	0	9816
CO _{2, 1.50m} (ppm)	1046.358	1.3062	0	11385
T _{0.05m} (°C)	-1.473	-0.2119	2.70E-21	9813
T _{1.50m} (°C)	-0.220	-0.0834	4.75E-07	11385
VWC _{0.05m} ($\text{m}^3 \text{m}^{-3}$)	0.028	1.2839	0	11303
VWC _{1.50m} ($\text{m}^3 \text{m}^{-3}$)	0.001	0.0547	1.05E-09	11462