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Desertification and ecosystem services supply: The case of the Arid Chaco of South America

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ABSTRACT

New integrated perspectives are increasingly needed to bridge the gap between biophysical and ecosystem services' based assessments of desertification. For a vast area of the dry Chaco region we sought to: (1) assess the spatial extent of four syndromes of vegetation change, associated with human or climatic drivers and (2) estimate and compare the supply of ecosystem services among these syndromes. We used a remote sensing approach based on the growing season –October to March- normalized difference vegetation index from MODIS, and climatological datasets from 2003 to 2013 to estimate: i) precipitation use efficiency, ii) precipitation marginal response, iii) the temporal trends of the residuals from the normalized difference vegetation index - annual precipitation linear relationship, and iv) the ecosystem services provision index. We diagnosed vegetation syndromes based on the difference between actual and reference sites' precipitation use efficiency and precipitation marginal response. Negative residuals trends were interpreted as vegetation changes driven by inadequate human management. The ecosystem services provision index assumes that ecosystem services supply varies positively with primary production and a negatively with its seasonal variability. Our results showed that 9.1% of the observed area belonged to the vegetation improvement syndrome - positive Delta precipitation use efficiency and Delta precipitation marginal response - while 3.4% were classified as vegetation cover reduction -negative Delta precipitation use efficiency and Delta precipitation marginal response. In turn, 10.5% and 2% of the study area fell within the increment in herbaceous vegetation -negative Delta precipitation use efficiency and positive Delta precipitation marginal response - and woody encroachment syndromes - positive precipitation use efficiency and negative precipitation marginal response - respectively. Human management did not have a uniform impact as all 4 syndromes displayed positive and negative residuals trends. Contrary to our expectations, there was no apparent association between vegetation syndromes and the supply of ecosystem services as estimated by the ecosystem services provision index. This study serves as a prototype to remotely assess ecosystem properties indicative of different vegetation syndromes and the associated supply of ecosystem services in dryland regions.

1. Introduction

The study of desertification, land degradation in drylands, has been pervaded with debates concerning to its definition and measurements. In part, the ubiquitous lack of consensus in desertification studies is related to its historical context. The occurrence of extensive famines in sub-Saharan Africa during the late 1970s fostered conceptualizations focused mainly on biophysical aspects (Reed and Stringer, 2016). Emphasis was on variables characterizing soil or vegetation condition and change. Assessments were derived from different sources, ranging from informed opinion, field surveys and remote sensing. However, not all changes in the biophysical components have a direct and immediate effect on humans and neither of these can be straightforwardly interpreted as a loss or degradation (Reynolds et al., 2006).

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More recently, desertification has been approached from a different conceptual framework, one that strengthens the link between nature and human welfare (Rock, 2006). Thus this conceptualization of desertification focuses primary on the assessment of ecosystems services, local communities' perception and economic indicators (Reed and Stringer, 2016). However, difficulties involved in the measurement of ecosystem services have limited its use (Kremen, 2005; Norgaard, 2010). Thus, two main definitions coexist: "desertification as land degradation in arid, semiarid and sub humid areas resulting from various factors including climatic variations and human activities" (United Nations, 1994), and "desertification as a persistent reduction in the capacity of ecosystems to supply services … over extended periods of time" (Rock, 2006).

Despite this apparent disconnection, integrated approaches combining biophysical and socioeconomic perspectives are being increasingly used (e.g. Reynolds et al., 2007, 2011; Chapin et al., 2009; Scholes, 2009; Verstraete et al., 2009, Bestelmeyer et al., 2015). These integrated frameworks are essential for local or regional decision making and policy development because they provide tools that help organize ideas, identify key system properties –i.e. slow variables, variables that strongly influence ecosystems but remain relatively constant over years to decades despite interannual variation in weather, grazing, and other factors (Chapin et al., 2009)- at its relevant scale, and anticipate impacts of different policy options. However, combined perspectives have not necessary translated into an improved operational monitoring of desertification at a global scale because the key slow socioeconomic and biophysical variables to monitor are system dependent.

However, recent advances in the quantitative estimation of desertification and ecosystem services suggest a way to bridge the gap between biophysical and ecosystem services' based assessments of desertification. Verón et al. (2006) and Kaptué et al. (2015) schematized 4 frequent vegetation syndromes based on the spatial or temporal comparison of two desertification indicators: the Precipitation Use Efficiency (PUE, Le Houérou, 1984; Prince et al., 1998) and the Precipitation Marginal Response (PMR, Verón et al., 2005). PUE, the ratio between aboveground net primary production (ANPP, the rate of net carbon accumulation by vegetation in above-ground organs) and precipitation (Le Houérou, 1984), provides information on the ability of a system to convert precipitation water into vegetation growth. PMR, the slope of the linear relationship between annual aboveground net primary production and precipitation (Verón et al., 2005), describes the sensitivity of vegetation to interannual changes in precipitation. Thus, the first syndrome identified by these authors is compatible with a reduction in plant cover, which should translate into a decrease in PUE and PMR as more precipitation would be lost to evaporation and runoff. The second syndrome, compatible with an increase in the abundance of herbaceous vegetation, should decrease PUE but increase PMR as forbs and grasses tend to respond faster to annual precipitation as the relative growth rate (growth rate per mass unit) is in general higher for grasses than for shrubs (Lambers et al., 1998). The third syndrome, compatible with woody encroachment, is frequently observed at grazed areas where fire is absent or suppressed and generally translates into a decrease in PMR but not necessary in PUE. We use woody encroachment in opposition to shrub encroachment to differentiate when the expansion of woody vegetation occurs as sites where shrubs are an important component of vegetation communities –as it is in the dry Chaco-than when shrubs are marginal. In Patagonia, Verón and Paruelo (2010) showed that the replacement of grass steppes by shrublands decreased PUE and PMR. Finally, the fourth syndrome is compatible with an increase in vegetation cover and should increase PUE and PMR as the fraction of precipitation lost to drainage, runoff or bare soil evaporation should decrease. Also, increased vegetation cover would allow for larger responses to wet years as potentially more water can be transpired with higher leaf areas.

In addition to the potential utility of PUE and PMR to track functional and structural changes in ecosystems, a methodology to assess desertification would benefit from the attribution of these changes to human or climatic causes. Herrmann and Hutchinson (2005) and Wessels et al. (2012) proposed a methodology based on the residual trends (RESTREND) to discriminate human from climate induced desertification. RESTREND, is calculated from the temporal trend in the residuals of the annual ANPP-PPT linear relationship. Therefore, a negative RESTREND implies that the ANPP not explained by precipitation decreased during the studied period. In general, the causes of this decrease are assumed to be human.

Recently, Paruelo et al. (2016) proposed and tested an integrative indicator of regulating ecosystem services. The ESPI (ecosystem services provision index) relates the supply of regulation ecosystem services related to biodiversity (avian biodiversity), C (soil organic C) and water dynamics (ground water recharge and evapotranspiration) to the mean active photosynthetic radiation absorbed by vegetation and its intra-annual variability. The hypothesis underlying the use of the ESPI is that ecosystem services increase with primary production and decrease with seasonal variability. This hypothesis is based on solid conceptual and empirical evidence that i) primary production determines the energy available for the rest of the trophic levels (excluding autotrophs) (McNaughton et al., 1989), ii) energy availability at ecosystem level determines ecosystem services supply (Richmond et al., 2007), iii) increases in season variability in primary production is tightly linked to land use changes (Paruelo et al., 2001a; Guerschman et al., 2003) and iv) temporal mismatches between resource pulses and resource pulse consumers decreases the overall energy use and ecosystem processes (Schwinning and Sala, 2004). Tested over forests and grasslands regions in the southern South America using field and modelling data on different ecosystems services (groundwater level regulation, soil C sequestration and hydrological yield), Paruelo et al. (2016) concluded that ESPI could be used as an aggregate indicator of the status and/or trends of ecosystem services over large areas.

The Arid Chaco is the driest portion of the Gran Chaco region. It is located on alluvial and aeolian plains on central South America (Fig. 1) and represents the transition zone between dry forest and shrublands. In the Arid Chaco, as in other arid and semiarid regions of the world, the spatial degradation patterns are quite heterogeneous in the region due to the irregular distribution of watering points (Lange, 1969; Blanco et al., 2008). Areas close to water sources lost most of the herbaceous cover –i.e. piospheres-while areas more than 8 km away from watering points show minimum signs of degradation (Blanco et al., 2008). Pickup and Chewings (1994) were the first who defined the term 'grazing gradient' as "spatial patterns in soil or vegetation characteristics resulting from grazing activities and which are symptomatic of land degradation". Domestic animals (sheep, goats, cattle, and horses) prefer to graze in vicinity to a watering point. When food is depleted in this area they move away from the source of water but return regularly for drinking. Consequently, higher number of individuals is frequently concentrated around the watering points. Then, animal density decreases gradually with increasing distance from water (Pickup et al., 1993; Friedel, 1997). Degradation patterns in the Arid Chaco are, then, mainly associated with the distribution of watering points.

Our objectives were twofold. First we assessed the spatial extent of desertification in the dry Chaco region by means of the spatial changes in PUE, PMR and RESTRENDS and identified vegetation syndromes. Second we estimated the supply of ecosystem services using the ESPI and compared it among vegetation syndromes. We focused on the dry Chaco region, an area that presents a broad gradient of both precipitation and degradation in a relatively flat landscape that minimize water redistribution. To address our objectives we used freely available remote sensing time series, global climatological datasets and local knowledge.

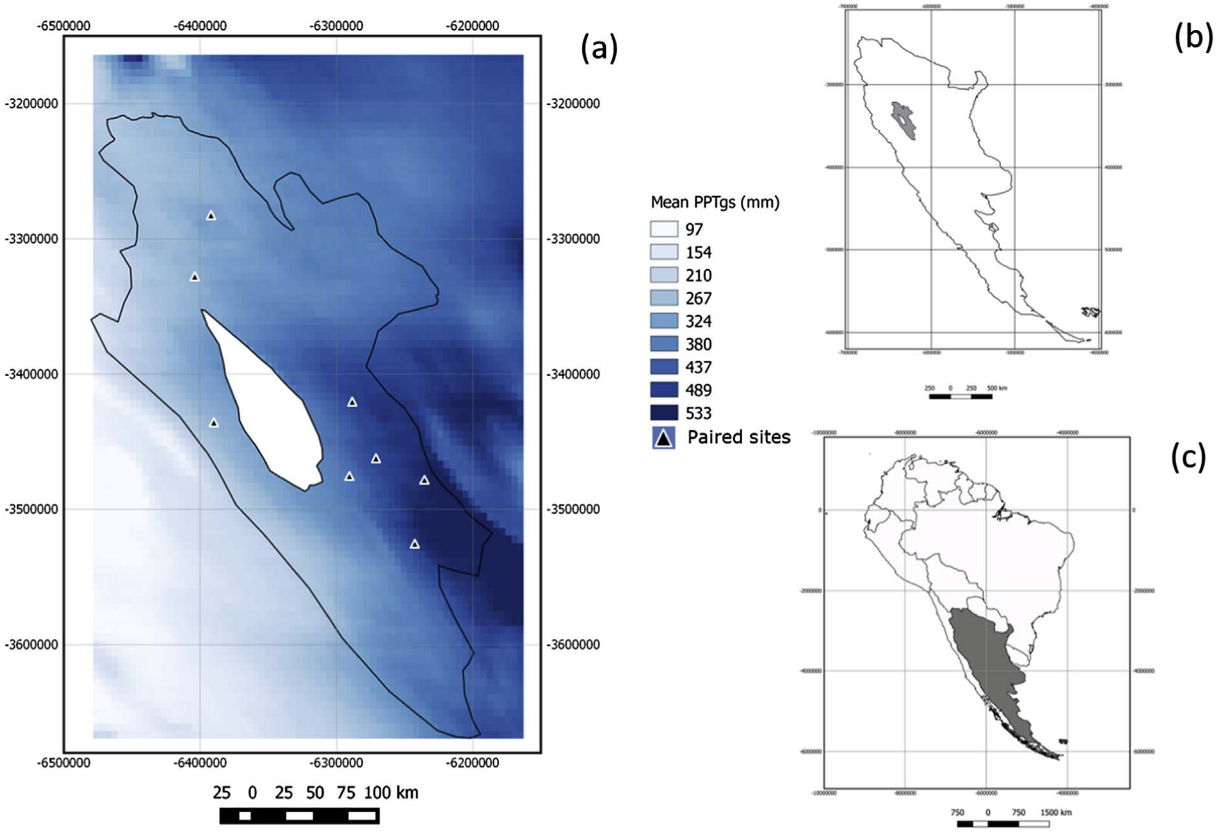

Fig. 1. Location of the paired sites studied within the Dry Chaco region –delimited by the irregular polygon-overlaying mean growing season precipitation (2003–2013) obtained from CHIRPS database (a); the Dry Chaco region location in Argentina (b); and the location of Argentina in South America (c). The white polygon in (a) corresponds to the Sierra de los Llanos area which was excluded from the analysis.

2. Materials and methods

2.1. Study area

The study was conducted in the Arid Chaco phytogeographic region, in northwestern Argentina (between 28° S and 33° S, and 64° W and 67° 0 W) (Fig. 1). This region covers $\sim$100 000 km and it is surrounded by mountains of $\sim$2000 m (Morello et al., 1985). Climate is subtropical with hot rainy summers and mild dry winters (Prohaska, 1959). Annual mean temperature varies between 17° and 20° C, with summers having 20–25 days with maximum temperatures > 40 °C and winters only 5–10 days with minimum temperatures below 0 °C (Morello et al., 1985). There is an east–west annual rainfall gradient ranging from 250 to 550 mm (Cabido et al., 1993; Blanco et al., 2008) with 80% of the annual rainfall occurring in the southern-hemisphere warm growing season, between November and March. Soils are coarse textured, with low organic matter content (< 1.5% of soil mass) and have a neutral to basic pH (Gómez et al., 1993). Topography is flat with the exception of the central Sierra de los Llanos which was excluded from the study area. The boundaries of the Arid Chaco were modified to exclude agricultural fields present at the north, east and southeast of the study area. Agricultural areas were identified by visual inspection of recent Google Earth high resolution images assuming that small to middle size paddocks displaying homogeneous non-woody cover were croplands or pasturelands. Both exclusions –Sierra de los Llanos and agricultural areas-assured that our results were not driven by significant water re-distribution, to which PUE and PMR could be very sensitive, or to human decision such as crop selection or management (mainly irrigation).

Typical vegetation is a subtropical xerophytic shrubland (dominated by *Larrea*, *Mimozyganthus*, *Senna* and *Capparis* genera), with scattered trees (mainly *Aspidosperma quebracho-blanco* and *Prosopis* spp.) and an herbaceous layer dominated by C_4 perennials grasses (*Trichloris*, *Pappophorum*, *Aristida* and *Setaria* genera) (Ragonese and Castiglioni, 1970; Morello et al., 1985). Overgrazing principally affects the herbaceous layer decreasing desired species and productivity (Biurrun et al., 2015). Along the precipitation gradient tree cover increases from 11% to 26%, herbaceous cover increases from 20% to 49%, and shrub cover remains almost constant, around 60% (Cabido et al., 1993).

Ranching (mainly cattle and goats) is the main economic activity in this area (Ferrando and Namur, 1984; Natenzon and Olivera, 1994). The area included approximately 10000 ranches, with 60% of them on fenced properties and the remaining 40% on communal grazing lands (Diez et al., 1991). Cattle productivity indices for the region are low (calf crop ~ 50%, beef productivity ~ 5,5 kg ha^{-1} y^{-1}). These low productivity indices are attributed to deteriorated rangeland conditions and to inadequate herd and grazing management practices (Diez et al., 1991; Orionte et al., 2001). Nowadays the Arid Chaco is characterized by a highly fragmented mosaic of isolated patches of forest and continuous shrublands (sometimes dense thorny scrub, locally called *fachinales*) with a reduced grass cover due to overgrazing (Zak et al., 2004; Kunst et al., 2006).

2.2. Satellite data

We based our analyses on the normalized difference vegetation index –NDVI-, an extensively used remotely sensed estimator of the interception of photosynthetic radiation by green tissues (Rouse et al., 1974; Pettorelli, 2013) and closely related to ecosystem carbon uptake (Bartlett et al., 1989; Ruimy and Saugier, 1994; Paruelo et al., 1997). We extracted MODIS Aqua and Terra platforms NDVI from the MYD and MOD13Q1 products from 2003 to 2014. Together, these products provide an 8-day temporal resolution at 250 m spatial resolution. Given that the studied area fall within two MODIS tiles (h12 v11 and h12 v12) we produced annual stacks of the growing season mosaicked and spatially subset tiles. Growing season was assumed to start at the beginning of October and end at the end of March from the following year (i.e. 22 images per annual stack) which agrees with previous studies describing the growing season length (Paruelo et al., 2001b). NDVI values were multiplied by 10000 for reporting purposes. Low quality data (i.e. MODIS quality other than "good data, use with confidence") was discarded. Missing or low quality data were linearly interpolated only when they were not located at the beginning or end of each growing season. Otherwise, the pixel was discarded for that year. Additionally, we accepted pixels with up to 5 years of missing data (i.e. a year where missing data occurred at the beginning or end of the growing season). We checked if a stricter treatment of missing data –pixels were discarded if there were two or more consecutive missing data or annual series were incomplete-would modify results. However, results were not qualitative different between both datasets. In the remainder of the study we present results from the inclusive treatment of missing data. Most of the discarded pixels corresponded to superficial salt deposits (Fig. 1).

2.3. Climatological datasets

We used the Climate Hazards Group InfraRed Precipitation with Station data (CHIRPS) version 2.0 dataset (Funk et al., 2014) to obtain spatially explicit estimates of precipitation. The spatial resolution from CHIRPS is 0.05° (~5,5 km at the study area) and the temporal resolution is daily but we integrated to 10 days and summed to the year. As suggested by Toté et al. (2015) we assessed the quality of this dataset by means of the Relative Mean Absolute Error (RMAE), Bias and Pearson correlation coefficient calculated between the observed precipitation at 12 meteorological stations with at least 5 years of data and located within or less than 10 km apart from the study area and the CHIRPS precipitation from the pixel where the meteorological station was found. The average RMAE, Bias and Pearson correlation thus calculated were 0.60, 1.94, and 0.42 (n = 12) and were lower than those calculated for the ERA-Interim from the European Center for Medium Range Weather Forecast dataset (Dee et al., 2011). To ease calculations we resampled precipitation data to 250 m to match the NDVI spatial resolution.

2.4. Ground truth

We selected 8 paired degraded-preserved sites well distributed throughout the regional gradient of mean annual precipitation, using grazing gradient concept. It is well documented that in the Arid Chaco, as in other arid and semi-arid regions, there is a reduction in the impact of grazing of domestic livestock on vegetation with distance to watering point (Lange, 1969; Pickup and Chewings, 1994). Thus, given the spatial homogeneity and that domestic grazing is the major disturbance in the area, we assumed that proximal and distal sites from watering points were representative of degraded and preserved vegetation conditions. Then, degraded sites were selected among piospheres located in large paddocks with unique watering points. We digitized the boundaries -average size: 40 has-of these piospheres by visual interpretation of very high resolution images from Google Earth. Preserved sites were selected based on previous studies (Blanco et al., 2008) and local knowledge. In general these sites were located further than 7 km away from a watering point where grazing effects were lowest (Blanco et al., 2008) and the average size was 82 has.

2.5. Analyses

To assess the spatial extent of desertification in the dry Chaco region by means of the changes in PUE, PMR and RESTREND (objective 1), we first performed a paired comparison of these indicators between the 8 proximal and distal sites to watering points. PUE (in units of NDVI mm^{-1}) was calculated as the ratio of the NDVI to the precipitation, in both cases averaged from the 2003 to 2014 growing seasons (n = 11). Growing season NDVI was quantified as the mean of the NDVI from October to March (i.e 22 8-day images). PMR (in units of NDVI mm^{-1}) was obtained from the slope of the linear regression adjusted between the growing season NDVI and growing season precipitation (PPT) from 2003 to 2014. RESTREND (in units of NDVI y^{-1}), in turn, was estimated from the linear slope of the residuals of the growing season NDVI-PPT relationship plotted against time (years). These indicators were calculated for each 250 m pixel that fell at least 75% within the boundaries of the proximal or distal sites. Then, we averaged the PUE, PMR and RESTREND from all the pixels belonging to each of the 8 proximal and 8 distal sites –irrespectively of the linear model p-values in the cases of PMR and RESTREND-. Finally we assessed the changes in PUE, PMR, and RESTREND between the proximal and distal sites using paired t-tests. To avoid pseudoreplication we used pixels as samples and sites as replicates (n = 8).

The identification of vegetation syndromes, as proposed by Verón et al. (2006), was based on the difference between the PUE and PMR from a given pixel to those of a reference situation (i.e. Delta PUE and Delta PMR), in this case the average of the pixels from the distal sites. In this analysis we only considered PMRs that were significantly different from zero ($p \leq 0.05$) for both, the given pixel and distal sites. Thus, we assumed that herbaceous vegetation increase where positive Delta PMR and negative PUE were registered (syndrome 1, S1); a decrease in plant cover occurred at sites where negative Delta PUE and negative Delta PMR took place (syndrome 2, S2); woody encroachment at sites with positive Delta PUE and negative Delta PMR (syndrome 3, S3), and. Sites with positive Delta PUE and Delta PMR were assumed to have gained vegetation and therefore process of vegetation improvement (syndrome 4, S4).

To assess the supply of ecosystem services among vegetation syndromes (objective 2), we first compared mean ESPI between the 8 proximal and distal. Mean ESPI was calculated as the average of the 11 seasonal ESPI values. Seasonal ESPI was obtained following Paruelo et al. (2016) as:

$$ESPI_{s,p} = NDVI_{gs,p} * (1 - CVNDVI_{gs,p})$$

where *s* stands for seasonal, *p* is for pixel, NDVI*gs* represents the average growing season NDVI and *CV* is the coefficient of variation expressed as a fraction. As calculated, ESPI varies between 0 and 10000 (i.e. assuming a constant and maximum NDVI of 10000 throughout the growing season and a resultant CV equal to 0). Similar to PUE, PMR and RESTREND, differences in ESPI between sites were assessed by means of paired t-tests. Second, we compared the supply of ecosystem services between vegetation syndromes by means of the temporal trend in ESPI. Given that ESPI was positively associated with MAP – slope = 5.6, $p < 0.0001$, $r^2 = 0.46$- we used the slope of the linear relationship between ESPI and year to compare the temporal dynamics of the supply of ecosystem services among sites differing in PPT inputs.

3. Results

3.1. Desertification status and syndromes

Mean PUE and PMR from proximal (degraded) and distal (preserved) sites were 8.72 and 10.32 NDVI.mm^{-1} and 1.71 and 2.6 NDVI.mm^{-1} respectively (Fig. 2). These magnitudes were significantly different with p-values lower than 0.01 (t = 4.87, df = 7, paired *t*-test for PUE) or lower than 0.05 (t = 2.5, df = 7, paired *t*-test for PMR). Thus, PUE was 19.5% and PMR 53% higher for distal than for proximal sites respectively. In turn, mean RESTREND was −11.2 NDVI y^{-1} and -7.76 NDVI y^{-1} at distal than at proximal sites with no significant differences between groups (t = −0.38, df = 7, p-value = 0.71, paired

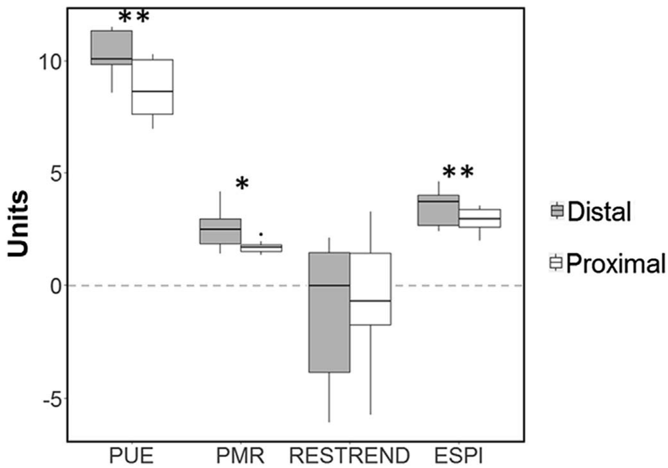

Fig. 2. Differences between distal (preserved) and proximal (degraded) sites in PUE, PMR, RESTREND and ESPI. RESTREND and ESPI values were scaled by dividing by 10 and 1000 respectively. The boxplot shows minimum, 25th, 50th, and 75th percentiles, and maximum-minimum values of the data. Asterisk indicates statistically significant differences at the p < 0.05 (*) or < 0.01 (**) confidence level from the paired *t*-test.

t-test). ESPI was on average 19.6% higher for distal than proximal sites (3461 vs 2893) and this difference was significant (t = 3.5, df = 7, p-value = 0.0094).

Delta PUE and Delta PMR displayed distinct spatial patterns (Fig. 3 a,b). Coarsely, Delta PUE was negative at the central-eastern portion of the study area while Delta PMR was mostly negative at the north-western part of the area. Of the 988184 pixels, 25% (n = 247868) displayed significant PMR, 245978 significantly positive and 1894 significantly negative. Delta RESTREND was mostly positive along the area with negative values sparse to the east and south of the Sierra de los Llanos (Fig. 3c). 125398 pixels (12.6%) had significant RESTREND, of which 28218 (2.8%) were positive and 97184 (9.8%) negative. Finally Delta ESPI was negative to the west and positive to the east of the study area (Fig. 3d).

From the 988184 250 m pixels encompassed in the study area, 247872 had significant PMR and thus could be classified into one of the 4 vegetation syndromes identified. 104014 and 34217 pixels (10.5% and 3.4% of the study area respectively) fell within the quadrant representing an increment in herbaceous vegetation (S1) and vegetation cover reduction respectively (S2, Fig. 4). In turn, 19786 pixels (2%) belonged to the woody encroachment syndrome (S3, Fig. 4). Finally, 89855 (9.1%) met the criteria for classification as vegetation improvement syndrome (S4) based on a positive Delta PUE and positive Delta PMR (Fig. 4 and Fig. S1). Moreover, from the subset of pixels with significant PMR, 25016 pixels (2.5%) had significant RESTREND, 1287 significantly positive and 23729 significantly negative.

3.2. Ecosystem services supply across vegetation syndromes

The temporal trend in ESPI between 2003 and 2013 varied from ca. $-270\ y^{-1}$ to $162\ y^{-1}$ over the dry Chaco region (Fig. 5) with the maximum values located at the southeast and center west areas (Fig. 3d). Similar to RESTRENDS, the minimum values in the temporal trend of ESPI were found in areas to the north and west of the study area limits (Fig. 3d). Moreover, there was no apparent difference by vegetation syndrome in the level of ecosystem services supply assessed by means of the ESPI temporal trend (Fig. 5). Overall mean ESPI trend was $-22\ y^{-1}$, while averaged by syndrome ESPI amounted to $-27\ y^{-1}$ (S1), $-16\ y^{-1}$ (S2), $-6\ y^{-1}$ (S3), and $-22\ y^{-1}$ (S4) (Fig. 5). Additionally, the range of variation of ESPI also overlapped among vegetation syndromes with a tendency towards higher minimum and maximum values for herbaceous increase (S1) and vegetation improvement syndromes (S4, Fig. 5).

4. Discussion

Our results showed that in the dry Chaco of Argentina heavily grazed sites had significantly lower PUE, PMR and ESPI than sites lightly grazed. Additionally, over the last decade, the area that had similar or better vegetation conditions compared to a reference situation tripled (9%–3%) the one that displayed decreased condition as evidenced by the combined use of Delta PUE and Delta PMR. The effects of human management on the ecosystem functioning were negative in 10% of the study area and positive in only 3%. In parallel, the average supply of ecosystem services between 2003 and 2013 was not consistently different among the vegetation syndromes described here.

Identification and quantification of the ecosystem properties that best correlate with dryland degradation are challenged by spatial and temporal variation in vegetation structure and phenology and by the difficulties to set a reference situation (Verón et al., 2006; Prince et al., 2007). Drylands are characterized by, among others, low and spatio-temporally variable precipitation (Noy Meier, 1973). This feature generally translates into water availability being the main driver of ecosystem functioning, patchiness in the phenological status of plants and marked interannual variations in vegetation productivity. The latter integrates differences in annual precipitation and in vegetation responses to precipitation interannual fluctuations. Extending previous work, we were interested in assessing to which extent two vegetation properties that describe the association between productivity and precipitation –i.e. PUE and PMR-can be used to diagnose changes in vegetation condition. By using variables that integrate precipitation we sought more robust indicators to spatiotemporal variations. Additionally, based on local knowledge on the impact of domestic herbivores on vegetation we identified reference and degraded areas to validate our approach.

Differences observed from paired comparisons between reference and degraded sites supported the use of PUE and PMR for land degradation assessment. The magnitude of the reduction in PUE (~20%) was lower than the one reported by Verón and Paruelo (2010) –50%- between grass and grass-shrub steppes and shrub steppes and semi-deserts of Patagonia. PMR reductions reported here (~53%) were in the range of those documented by the same authors in Patagonia (44%). One of the main differences in vegetation structure between the Dry Chaco and the Patagonian region is the presence of trees in the former. Adult trees are only indirectly affected by grazing through changes in the hydrological and biogeochemical cycles as they are seldom grazed by cattle. Indeed, Asner et al. (2003) found an increase in photosynthetic vegetation close to watering points in four major plant communities of the Monte Biome of Argentina and attributed its causes to increased germination of *Prosopis flexuosa* whose fruits were consumed and dispersed by cattle. Trees may, therefore, establish a bottom line that prevents further decreases in ecosystem functioning with land degradation. This would be particularly so close to watering points where cattle grazing notably diminishes herbaceous and sometimes shrub cover (Blanco et al., 2008) and adult trees provides shelter to cattle. Similarly, the relative increase in the abundance of woody –shrubs and trees-in relation to herbaceous vegetation at proximal sites could explain the marked decline in PMR. Verón and Paruelo (2010) and Williamson et al. (2012) showed that PMR was lower for woody than for herbaceous vegetation.

The combined use of spatial comparisons of PUE and PMR not only allowed the identification of changes in land condition but also permitted the diagnosis of 4 vegetation syndromes. These syndromes summarize widespread, although particular, effects upon vegetation structure and functioning, biogeochemistry and hydrology of drylands. Herbaceous vegetation increase was the most frequent syndrome reaching 10% of the area. This extension may reflect the increasingly common management practice of replacing natural vegetation by buffelgrass (*Cenchrus ciliaris* L.) roller chopped pastures (Blanco et al., 2005). This perennial C_4 species grows vigorously with warm

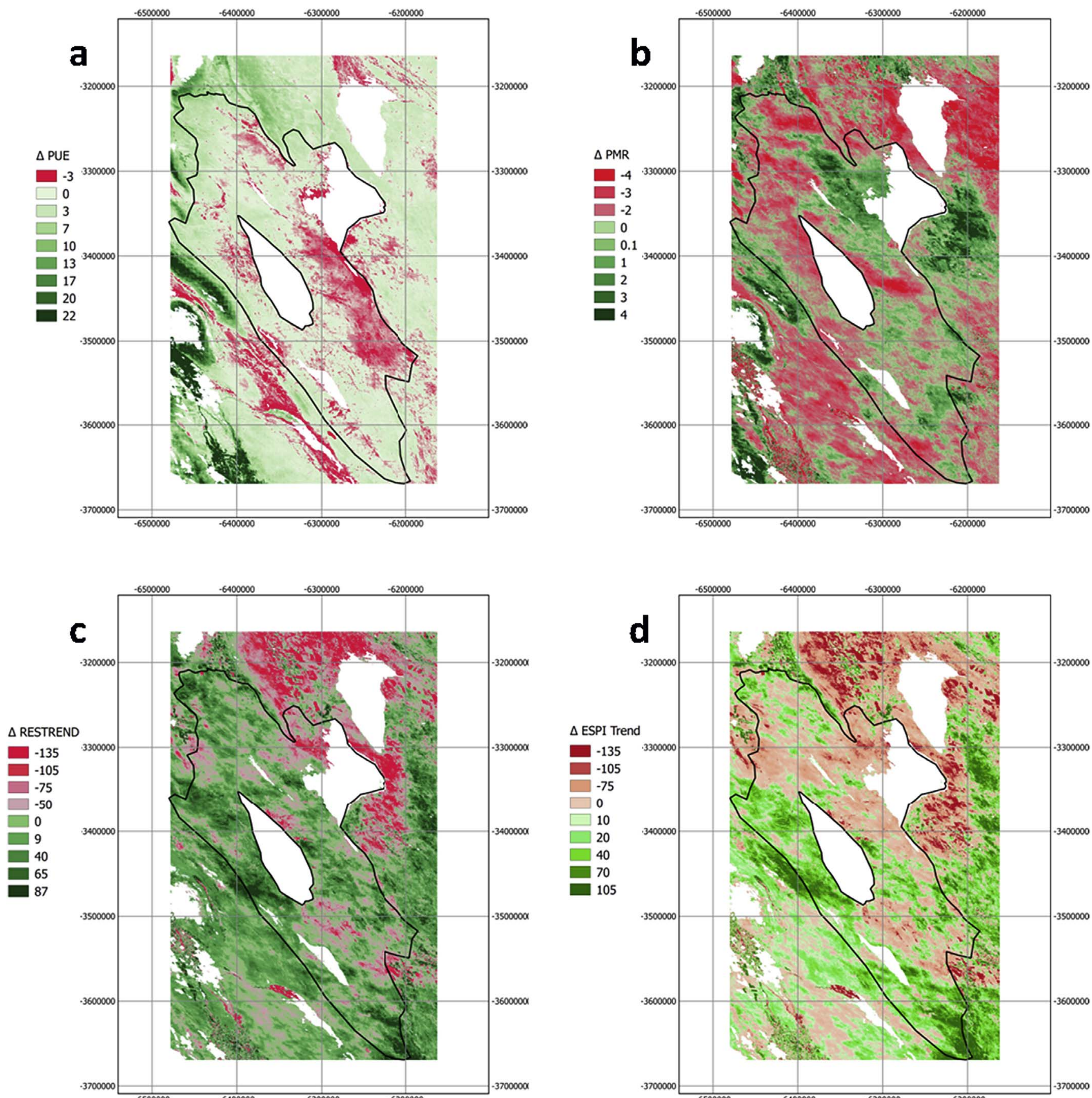

Fig. 3. Spatial distribution of Delta PUE (a) -in NDVI x 10000/mm MAP-, Delta PMR (b) -in NDVI x 10000/mm MAP-, Delta RESTREND (c) -in residuals NDVI x 10000/y-, and Delta ESPI trend (d) –ESPI/y-, throughout the study region.

temperatures and water availability and is completely senesced during winter. These features may explain the higher PMR and lower PUE than the reference –distal- sites particularly considering that we used a proxy for primary productivity (i.e. NDVI) that is associated to photosynthetic active radiation absorption (Sellers et al., 1992).

The reduction of vegetation cover, a syndrome found in 3.4% of the study area, is a possible outcome of inadequate grazing management (Cabido et al., 1994). Herein, excessive grazing pressure generally leads to increased bare soil surface area and decreased herbaceous cover (Asner et al., 2004). In turn, decreased plant abundance due to grazing leaves bare soils compacted preventing rainfall infiltration and promoting water redistribution, runoff and eventually wind and water erosion with nutrient losses (van de Koppel et al., 2002) which translates into lower PUE and PMR. Using the same rationale, improved grazing management –such as adequate grazing stocking rates and resting period-should result in an increment in vegetation cover in the dry Chaco (Quiroga et al., 2009; Blanco et al., 2009), a syndrome found to be taking place in 10% of the study area. Even though a detailed and quantitative description of grazing management in the dry Chaco during the las 15 years is unavailable, these results suggest that there was variability in this factor and thus emphasize the role of managers and extension agencies.

Another possible outcome of mismanaged grazing is the increase in woody vegetation and the decrease in the intercanopy herbaceous

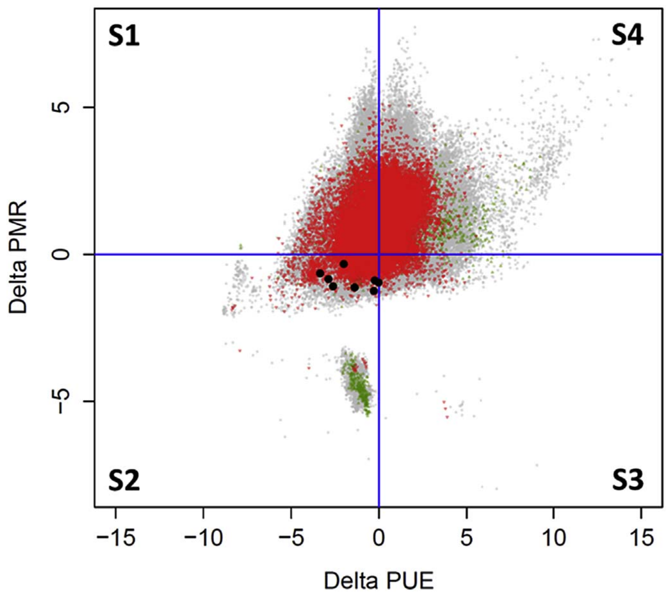

Fig. 4. Changes in vegetation condition of each pixel assessed from Delta PUE and Delta PMR. Only pixels where PMR was significantly different from zero were plotted (n = 247868). Also, when RESTREND was significantly different from zero grey dots were replaced by green –when positive- or red –when negative-triangles. Black dots represent the average Delta PUE and Delta PMR of pixels that fell at least 75% within each of the 8 polygons of the proximal –degraded- sites. Vertical and horizontal blue lines were plotted to help identify quadrants associated to vegetation syndromes. S1, S2, S3 and S4 stand for the following vegetation syndromes: herbaceous vegetation increase, vegetation cover reduction, woody encroachment and vegetation improvement. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

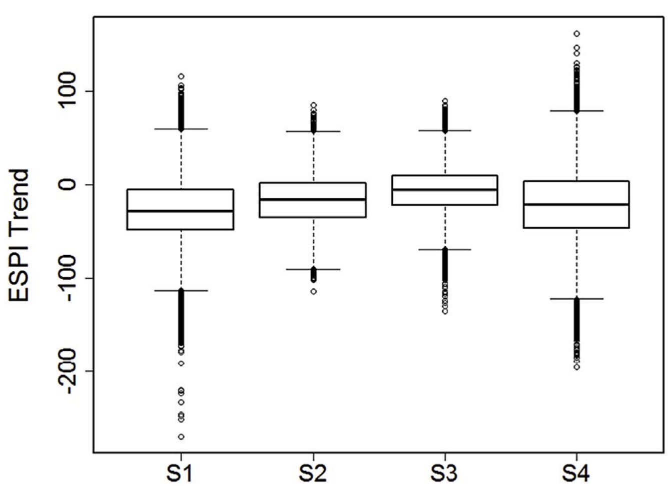

Fig. 5. ESPI temporal trend differences among vegetation syndromes. The boxplot shows minimum, 25th, 50th, and 75th percentiles, and maximum minimum values of the data. ESPI trend is expressed in ESPI units/y. S1, S2, S3 and S4 stand for the following vegetation syndromes: herbaceous vegetation increase, vegetation cover reduction, woody encroachment and vegetation improvement.

cover. From the perspective of cattle ranchers, woody encroachment is undesired as it represents a reduction in forage availability and thus is often considered as an indicator of land degradation. However, this view has been defied by recent evidences. Maestre et al. (2009) showed that the increase in shrubs abundance resulted in higher species richness, enhanced biomass of microorganism, greater soil fertility and N mineralization. Indeed, increases of up to 4× have been observed at sites were the dominant woody species was a nitrogen fixer (Hughes et al., 2006). Overall, the increase in PUE used here to diagnose woody encroachment is compatible with the idea that the increase in nitrogen fixer woody vegetation typical of the dry Chaco more than compensates for coincidental decreases in herbaceous vegetation in terms of average light interception. Similarly, the decrease in PMR assumed to characterize dry Chaco woody encroachment agrees with observations from other systems (Milchunas and Lauenroth, 1993; Paruelo and Lauenroth, 1995; Verón and Paruelo, 2010; Williamson et al., 2012).

Positive and negative RESTRENDs were found at all 4 syndromes implying that changes in vegetation condition did not stemmed only from differences in human management. RESTREND methodology removes -but not accounts for-the effect of changes in precipitation to assess the impacts of human activities upon vegetation (Wessels et al., 2004). Therefore, negative RESTRENDs occurring at sites showing improved vegetation condition (or viceversa) suggest that precipitation changes override the effects human actions. However, during the studied period CHIRPS and rain gauge precipitation showed an increasing trend during the first 5 years that was reverted afterwards with no overall trend (data not shown).

Contrary to our expectations, there was no apparent association between vegetation syndromes and ecosystem services supply (Fig. 5). We based our expectations on the lower ESPI found at proximal than at distal sites (Fig. 2). However, this difference became blurred when calculated as the temporal trend among all the pixels from the study area grouped in vegetation syndromes. Even the comparison between opposite situations yielded light differences towards less negative ESPI trends at vegetation increase syndrome than at the vegetation reduction syndrome. A possible explanation lies at the way each metric is calculated. PUE and PMR do not account primarily for the overall system output. Rather, they describe an input-output relationship and, in the context of a varying spatiotemporal precipitation, represent critical slow variables and more robust indicators of change (Chapin et al., 2009; Prince et al., 1998; Verón et al., 2006). On the contrary, ESPI is calculated from the annual mean NDVI and thus considers output magnitudes and its seasonal variability. Therefore, a high ESPI requires a high mean NDVI or a very low seasonal variability. In turn, high mean NDVI can be obtained at pixels receiving high precipitation –resulting in a low PUE- or low precipitation –high PUE-. Similar ambiguities can occur with PMR as high or low slopes can result from high or low mean NDVI and seasonal variation is neither considered directly. In combination, these issues indicate a tradeoff between the metrics indicative of land degradation in drylands used here and metrics that quantify the supply of ecosystem services therein.

Our results should be taken with caution. As any remote sensing study, ours suffers from uncertainties regarding the interpretation of the NDVI, the spatial and temporal resolution from MODIS and its geolocation error among others. More importantly, our study relies heavily on the identification of reference areas. This problem is pervasive in desertification studies (Verón et al., 2006; Prince et al., 2007) and has received considerable attention from restoration ecology practitioners. Briefly, there are three different approaches to solve the definition of a reference situation, none of which is free of difficulties. The spatial approach assumes that actual vegetation from certain sites is similar to what could be considered the pristine vegetation. This approach requires extensive knowledge of vegetation dynamics and environmental heterogeneity to define the total area that each reference site represents. The temporal approach relies on the comparison of vegetation conditions through time (e.g. Kaptué et al., 2015). However, the actual condition at the beginning of the period is unknown and thus, for example, a lack of temporal trends could be interpreted as an indication of a degraded or non-degraded condition alike. The third approach is based on modelling the reference situation for each site. This approach also requires extensive data and assumptions based on past performance that may not be adequate for present conditions. We here used a spatial approach where we pooled together the 8 reference situations and compared each pixel to that average reference. Thus our study suffers from errors stemming from comparisons of actual states to situations that may not be representative of its reference conditions.

However, the magnitude of this error must have been restricted by the relative homogeneity of the vegetation from the dry Chaco. Finally 11 years may be a short time period to characterize PMR and RESTRENDS as interannual variability may preclude the correct identification of the NDVI-PPT relationship. Had this been the case, our results are conservative given that we only considered significant PMR and RESTRENDS to diagnose vegetation syndromes and assess management impacts.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jaridenv.2017.11.001.

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