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Exploring the linkage between spontaneous grass cover biodiversity and soil degradation in two olive orchard microcatchments with contrasting environmental and management conditions

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Abstract. Spontaneous grass covers are an inexpensive soil erosion control measure in olive orchards. Olive farmers allow grass to grow on sloping terrain to comply with the basic environmental standards derived from the Common Agricultural Policy (CAP, European Commission). However, to date there are few studies assessing the environmental quality considering such covers. In this study, we measured biodiversity indices for spontaneous grass cover in two olive orchards with contrasting site conditions and management regimes in order to evaluate the potential for biodiversity metrics to serve as an indicator of soil degradation. In addition, the differences and temporal variability of biodiversity indicators and their relationships with environmental factors such as soil type and properties, precipitation, topography and soil management were analysed.

Different grass cover biodiversity indices were evaluated in two olive orchard catchments under conventional tillage and no tillage with grass cover, during 3 hydrological years (2011–2013). Seasonal samples of vegetal material and photographs in a permanent grid (4 samples ha^{-1}) were taken to characterize the temporal variations of the number of species, frequency of life forms, diversity and modified Shannon and Pielou indices.

Sorensen's index showed strong differences in species composition for the grass covers in the two olive orchard catchments, which are probably linked to the different site conditions. The catchment (CN) with the best site conditions (deeper soil and higher precipitation) and most intense management presented the highest biodiversity indices as well as the highest soil losses (over $10\,\text{t}\,\text{ha}^{-1}$). In absolute terms, the diversity indices of vegetation were reasonably high for agricultural systems in both catchments, despite the fact that management activities usually severely limit the landscape and the variety of species. Finally, a significantly higher content of organic matter in the first 10 cm of soil was found in the catchment with worse site conditions in terms of water deficit, average annual soil losses of $2\,\text{t}\,\text{ha}^{-1}$ and the least intense management. Therefore, the biodiversity indices considered in this study to evaluate spontaneous grass cover were not found to be suitable for describing the soil degradation in the study catchments.

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1 Introduction

Soil degradation is defined as the deterioration and loss of soil functions, involving processes such as soil erosion, sedimentation problems in flood plains and reservoirs, climate change, watershed functions and changes in natural habitats leading to loss of genetic stock and biodiversity (Chen et al., 2002). The agricultural intensification of 20th century Europe has led in general terms to a widespread decline in farmland biodiversity across many taxa (Benton et al., 2003). The new 2020 Biodiversity Strategy (European Commission, 2011; 2011/2307 INI) aims to improve the contribution of fisheries and agricultural and forestry sectors to biodiversity. In addition, the Multiannual Financial Framework for 2014–2020 offers significant opportunities to improve synergies not only in soil biodiversity but also with respect to other degradation processes such as soil loss (Cross-compliance, Agriculture and Rural Development; European Commission, 2014).

An area of over 2.5 Mha is dedicated to olive cultivation in Spain (MAGRAMA, 2013), which represents about 41 % of the world's olive production. Olive harvesting and its associated agri-food industries are especially important in rural areas from a socioeconomic viewpoint. Over 60 % of the area dedicated to olives is located in Andalusia, the southernmost region of the country. A high risk of soil degradation has been described by multiple authors, such as Goméz-Limón et al. (2009) and Gómez et al. (2014a), as the result of the interaction of climatological and topographical factors and/or inappropriate soil management. Olive trees have traditionally been cropped under rainfed conditions and on sloping areas where other crops are difficult to grow; they usually provide very low yields or require large investments in order to exploit them properly. The characteristics of the Mediterranean type of climate, where long dry periods alternate with intense rainfall events, in conjunction with soil management systems that pursue bare soils to minimize water competition by weeds entail a high susceptibility to severe water erosion of the soil (Gómez et al., 2014a). Therefore, the use of cover crops has been promoted for soil protection, given their proven effectiveness in controlling water erosion (Gómez et al., 2004, 2009a, b; Márquez-García et al., 2013; Taguas et al., 2013, among others). In fact, growing crops in between the olive tree rows is currently a compulsory requirement if the mean slope of the plot is over 15 %, according to cross-compliance rules (European Commission, 2014). Spontaneous covers are usually irregular and develop slowly but tend to achieve significant growth during spring which may result in greater competition for water and nutrients during the most critical periods of the olive growing cycle. However, due to its zero cost, it is a common alternative in low production olive farms (e.g. Taguas et al., 2013). Furthermore, additional advantages of spontaneous covers in terms of biodiversity, carbon sequestration and the aesthetic improvement of the landscape might make it worth to study their potential contribution.

The study of spontaneous grass cover and its interactions with soil have been traditionally associated with the improvement in crop yield (e.g. Graziani et al., 2012; Kamoshita et al., 2014;) or habitat and species conservation (e.g. Albrecth, 2003; Hyvönen and Huusela-Veistola, 2008; Aavik and Liira, 2009) in agronomical and ecological terms, respectively. However, their importance as indicators of soil degradation has scarcely been explored.

The bioindicators of soil quality are commonly associated with the biological activity of their microorganisms; however, spontaneous grass cover biodiversity may be a simpler way to indicate the risk of soil degradation, given that richer and more complex ecological niches might produce more vegetal biomass, efficient cover and, eventually, soil protection, as well as habitat and food opportunities for other elements of the trophic chain, such as birds or reptiles. In addition, one key drawback for the proper implementation of environmental protection policies is the lack of a well-defined quantitative measure or indicator of biodiversity which was suitable to describe, compare or measure possible changes (Büchs, 2003; Spangenberg, 2007; Moonen and Bàrberi, 2008). The use of biological indices – in this case associated to grass spontaneous cover – might be helpful because they are more sensitive to changes than chemical and physical soil indicators and because they could give a broader picture of soil quality (Bastida et al, 2008).

The main hypothesis of this study was that richer ecological niches mean lower risks of soil degradation in terms of indicators such as organic matter decline, bulk density and runoff coefficients and soil losses. This would be associated with an optimum space occupation derived from the presence of distinct species. In addition, we postulate that the interactions of soil and management explain better the diversity of spontaneous grass covers than the environmental site conditions (annual/seasonal patterns) due to minor soil disturbances which might produce conditions which bring it closer to natural systems.

The specific objectives of this work were (1) to describe and compare the biodiversity indices for spontaneous grass covers in two olive orchards with contrasting management intensities, environmental conditions and yields; (2) to analyse the temporal patterns of these indices, relative to meteorological conditions and soil management; and (3) to evaluate the relevance of biodiversity indices as indicators for soil quality, in terms of soil degradation.

2 Materials and methods

2.1 Study sites

The study catchments are located in the province of Córdoba (Fig. 1, Table 1), in southern Spain. Both were described in detail by Gómez et al. (2014b) and Taguas et al. (2013) to

Table 1. Summary of the main environmental features in the study catchments.

Name	La Conchuela	Arroyo Blanco
Location	Córdoba	Puente Genil (Córdoba)
Drainage area (ha)	8.0	6.1
Mean elevation (m)	142	239
Mean slope (%)	9	15
Mean annual precipitation (mm)	642	400
Max. and min. daily average temperatures	27.8° July/8.1° January	26.5° July/ 8.4° January
Soil type (FAO; see details in Table 3)	Vertisol	Cambisol
Texture	Clay loam	Sandy loam
OM content (%, topsoil)	1.1	1.4
Mean olive yield (kg ha^{-1})	8000	1300
Management (see details in Table 2)	Spontaneous grass cover controlled with a combination of mowing, and occasional herbicide application	Extensive, non-tillage with a spontaneous grass cover

evaluate the erosive patterns for the periods 2006–2011 and 2005–2011, respectively. The results of those studies were considered an accurate representation of the soil degradation state.

The Conchuela catchment (CN; 37.6° N, 5.0° W, Spain) is situated in a fertile area along the old terraces of the River Guadalquivir (Gómez et al., 2014b). The drainage area of the catchment is 8.0 ha and it presents an average elevation of 142 m and a mean slope equal to 9 %. The climate is classified as Mediterranean with an average annual precipitation of 642 mm, which mainly occurs between October and March (about 76 % of the precipitation). The average annual temperature is 17.5 °C. The maximum daily mean temperature is usually recorded in July (27.8 °C) while the minimum is generally observed in January (8.1 °C). The soil is a Vertisol, according to the FAO classification (FAO, 2006). It is a deep soil, very plastic when wet but, when dry, the presence of cracks induces high infiltration rates. The predominant soil texture is clay loam (Table 1). The olive trees were planted in 1993 with 6 × 7 m tree spacing. The mean olive yield in the catchment is 8000 kg ha^{-1}. During the study period, the farmer allowed for the growth of spontaneous grass cover in the lanes, from the end of winter until April. Herbicide (glyphosate and oxyfluorfen) treatments were applied to control their growth in the tree line from March to September (Table 2). Occasionally, surface tillage was made at selected locations within the catchment to cover rills and small gullies obstructing machinery traffic within the orchard. Mowing in the tree lane was performed in areas of excessive grass cover from late winter to early spring. Harvesting is semi-mechanized using tree vibrators from late autumn to mid-winter, depending on weather conditions and when the fruit ripens (Gómez et al., 2014b; Table 2).

The Puente Genil catchment (PG; 37.4° N, 4.8° W) represents a marginal olive orchard with a very low production. Management operations are kept to a minimum in order to reduce costs. It is located in an area with a long tradition of olive cropping in the upper reaches of the Guadalquivir Valley (Taguas et al., 2013). The catchment has a drainage area of 6.1 ha and the mean elevation is 239 m. The average slope is equal to 15 %. As for the climate type, the catchment is located in a Mediterranean area with a mean annual precipitation of 400 mm. The average temperature in the hottest month (July) is 26.5 °C, while in the coldest month (January) it is 8.4 °C. The main soil category of the catchment is Cambisol (FAO classification; FAO, 2006) with sandy loam texture (Tables 1 and 3). Calcic parental material is located at different points of the catchment with a very shallow soil, mainly on the western hillslope (Fig. 1b). In contrast, on the eastern hillslope, soil depth is more than 3 m. The areas closer to the catchment outlet are old terraces with abundant coarse calcarean material. The mean olive yield is 1300 kg ha^{-1}. The olive trees' age is 17 years. They were planted on a 7 m × 7 m grid. No-tillage with spontaneous grass cover growing from winter to spring was the management type corresponding with the first few years. Spontaneous grass is removed once (only in spring) or twice a year (September or October and March, April or May), mechanically or using phytosanitary products under the canopies (or combining both; see also Taguas et al., 2013). The details of the management applied during the study period are summarized in Table 2.

2.2 Spontaneous grass cover sampling

Four spontaneous grass cover surveys were performed per year (one per season) during 2011, 2012 and 2013. Survey dates were based on the preceding meteorological conditions that determined the germination periods, as well as the development of the spontaneous grass cover. A grid was established in each catchment (Fig. 1) with a sampling density between 4 and 6 points ha^{-1}. In each georeferenced grid

Table 2. Management operations applied during the study periods in both catchments (PG: Puente Genil; CN: Conchuela).

Catchment	Month	2011	2012	2013
CN	January		Harvesting: mechanical vibrators combined with a buggy with an umbrella to collect the olives	Harvesting: mechanical vibrators combined with a buggy with an umbrella to collect the olives
	February			
	March	Herbicide treatments around trees (glyphosate and oxyfluorfen in infested areas)		Herbicide treatments around trees (glyphosate and oxyfluorfen in infested areas) Mowing of lane areas
	April	Mowing of lane areas	Herbicide treatments around trees (glyphosate and oxyfluorfen in infested areas) Mowing of lane areas	
	May	Drip irrigation	Drip irrigation	Drip irrigation
	June	Drip irrigation	Drip irrigation	Drip irrigation
	July	Drip irrigation Herbicide treatments around trees (glyphosate and oxyfluorfen in infested areas)	Drip irrigation	Drip irrigation Herbicide treatments around trees (glyphosate and oxyfluorfen in infested areas)
	August	Drip irrigation	Drip irrigation Herbicide treatments around trees (glyphosate and oxyfluorfen in infested areas)	Drip irrigation
	September	Drip irrigation	Drip irrigation	Drip irrigation
	October			
	November			
	December	Harvesting: mechanical vibrators combined with a buggy with an umbrella to collect the olives	Harvesting: mechanical vibrators combined with a buggy with an umbrella to collect the olives	
PG	January			
	February			
	March			
	April	Four tractor passes to mechanically clear the spontaneous grass cover		
	May	Foliar fertilization (N, Mg and Fe)	Four tractor passes to mechanically clear the spontaneous grass cover Herbicide treatments around trees (glyphosate)	
	June			
	July			
	August			
	September			Four tractor passes to mechanically clear the spontaneous grass cover Herbicide treatments around trees (glyphosate)
	October			
	November	Harvesting: mechanical vibrators combined with a buggy with an umbrella to collect the olives	Harvesting: mechanical vibrators combined with a buggy with an umbrella to collect the olives	Harvesting: mechanical vibrators combined with a buggy with an umbrella to collect the olives
	December			

Table 3. Soil properties in two profiles of the catchments (PG: Puente Genil; CN: Conchuela; OM: organic matter content).

Catchment	Horizon	Width (cm)	Coarse elements (%)	Sand (%)	Silt (%)	Clay (%)	Texture class	pH	OM (%)
PG	A	10	22.7	59.5	35.2	5.3	Sandy-loam	8.8	1.59
	C	40	24.4	60.8	34.3	4.9	Sandy-loam	8.8	1.59
CN	A	0–56	0.36	5.9	45.1	49.0	Clay	8.6	0.96
	B	56–110	0.00	5.9	46.4	47.7	Clay	8.7	0.53
	BC	110–138	0.00	–	–	–	Clay-loam	–	–
	C	> 138	0.00	–	–	–	Clay-loam	–	–

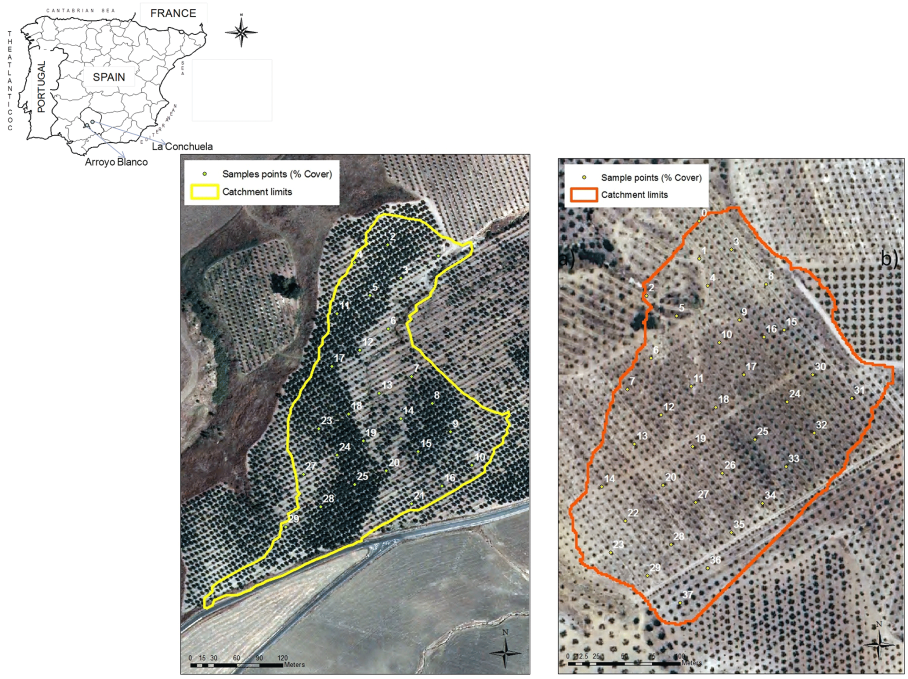

Figure 1. Locations of the study catchments and sample grids: **(a)** La Conchuela (CN) and **(b)** Arroyo Blanco in Puente Genil (PG).

point, a 0.5×0.5 m frame was used to delimit the survey area (Fig. 2). These sampling points were always placed in the lanes between the lines of trees, away from the olive canopy and the areas of drip irrigation and herbicide application. Plant samples were taken in order to identify the species present at each grid point. In addition, photographs of each point were taken (Reflex Olympus E-420, ED 14–42 mm; height 1.4–1.7 m; Fig. 2) to observe the annual and seasonal differences of the spontaneous grass cover.

2.3 Data analyses: biodiversity indices, meteorological variables and soil quality indicators

2.3.1 Biodiversity indices

The indices considered to evaluate the biodiversity associated with the spontaneous grass cover were richness (R), Sorensen's index (I_s), transformed Shannon (H_{mod}) and Pielou indices (J_{mod}), absolute frequency of occurrence and biological spectrum. R was determined for the total number of grasses and forbs found per catchment per season and per point. Firstly, in each sample point of the grid (Figs. 1, 2a, b), the species present were identified with pictures and vegetal material and then the total number species in each catchment (on a seasonal and annual scale) were calculated.

I_s indicates the degree of similarity between two samples (study sites) in regard to the species composition (Eq. 1). It ranges from 0 to 1, where 0 means that the two samples are completely different and 1 completely equal.

$$I_s = \frac{2 \cdot C}{A + B}, \quad (1)$$

where A is the number of species identified in PG, B the number of species identified in CN, and C is the number of species common to both study sites.

Shannon's index, H, (Eq. 2; Shannon and Weaver, 1949) represents the uncertainty associated with the prediction of species identity of an individual taken from a sample. It usually produces values of between 1.5 and 4.5. Minimum val-

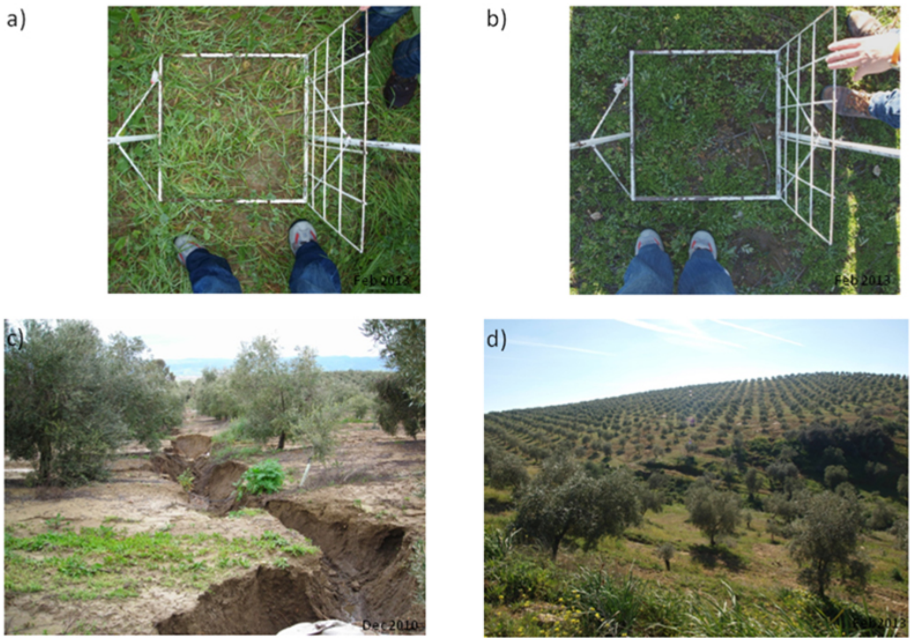

Figure 2. Examples of spontaneous grass cover sample plots and view of the catchments: **(a)** a plot in La Conchuela; **(b)** a plot in Puente Genil; **(c)** gully with cover crop in CN; **(d)** view of a hillslope in PG.

ues are obtained when most of the individuals belong to the same species or to a small group of (less diverse) species, while the highest values are produced in communities where all the species have the same number of individuals. If there is only one group of species, Shannon's index is equal to 0.

$$H = \sum_{i:1...n} \left(p_i Ln(p_i)\right), \tag{2}$$

where $p_i = n_i/N$, n_i is the number of individuals corresponding to the species i, and N is the total number of individuals. In this case, a modification of Shannon's index, H_{mod}, was used to simplify the analysis, based on the evaluation of pictures that presented each grid point of those considered in the catchment sample (see Figs. 1 and 2).

Therefore, n_i was substituted by the number of grid points where a species was present and N, the total number of grid points considered. The suitability of the transformations associated with H_{mod} was verified with the samples taken in spring 2013 in both catchments.

Pielou's equity index (Eq. 3; Pielou, 1969) measures the ratio of the observed diversity and the maximum expected diversity. It varies between 0 and 1, with 1 describing systems where all species are equally abundant.

$$J = \frac{H}{Ln(S)}, \tag{3}$$

where H is Shannon's index and S is the number of species. If H (Eq. 3) is substituted for H_{mod}, then J_{mod} is obtained.

Finally, the biological spectrum or life form (Raunkiaer, 1934) was identified for each species according to its behaviour during the unfavourable season (June–September): epiphytes, phanerophytes, chamaephytes, hemicryptophytes, therophytes, and cryptophytes.

2.3.2 Meteorological variables to describe temporal variability of biodiversity indicators

The cumulative precipitation (P), cumulative reference evapotranspiration (ETP) and average minimum daily temperatures (T_{m}) were considered in order to evaluate their influence on the biodiversity indices. The daily precipitation was recorded in the gauging stations of the catchments, while the daily values of ETP and T_{m} were collected from La Reina and Santaella CSIC (Institute of Sustainable Agriculture) meteorological stations for CN and PG, respectively (CSIC, 2014).

2.3.3 Soil degradation indicators: soil loss, runoff, organic matter and bulk density

The relationships between the mean values of soil losses, runoff coefficients and organic matter content (0–10 cm) in the catchments with R, J_{mod} and H_{mod} were explored to discuss the role of biodiversity indices as a proxy of soil quality indicators. Soil loss (SL) and runoff coefficient (R_{c}) were measured in the catchments over 5 years (Taguas et al., 2013; Gómez et al., 2014b).

The samples for organic matter (OM) analysis were taken between 0 and 10 cm combining the inter-row and the area under the tree canopies obtained on regular grids with a density of 6–10 samples ha^{-1}. The number of samples was 90 and 65 in CN and PG, respectively. The Walkley–Black procedure (Nelson and Sommers, 1982) with samples (2 mm sieve) was followed to determine the organic matter content. Bulk density (BD) was measured on the same grid using undisturbed soil cores of approximately 250 cm^3. The differences in grid and number of samples are due to the tree spacing in the catchments.

2.3.4 Statistical analyses

Basic statistics (mean, standard deviation and coefficient of variation) were evaluated for the annual values of R, J_{mod}, H_{mod}, I_s, T_m, ETP and P. In the case of I_s, the average seasonal values were calculated to observe the possible differences in the study sites over the year. The histograms of the biological spectrum measured in the catchments for the study period were also compared.

In addition, in order to evaluate the influence of the meteorological variables on the biodiversity indices H_{mod}, J_{mod} and R, a correlation analysis was carried out with meteorological features: P, ETP and T_m. The analysis was carried out with the mean values of the variables P, ETP and T_m corresponding to the 5, 15, 30, 60, and 365 days previous to the sample date. As for soil properties OM and BD, box and whisker plots and t tests for independent samples were used to determine whether there were significant differences between the study sites. For SL and R_c, only box and whisker plots were represented because the number of samples was five. These properties were compared with the biodiversity indices to qualitatively describe the correlation degree.

3 Results and discussion

3.1 Variability of the biodiversity indicators

The mean values of R, H_{mod} and J_{mod} were higher in CN than in PG, which probably shows that site-specific conditions have greater importance than long-term management effects (Table 4). A lower diversity was identified in PG, which was probably associated with worse environmental conditions in terms of water deficit, as compared to CN (Table 4), coupled with coarser soil texture and lower soil water holding capacity (Table 3). Precipitation was on average 25 % lower in PG while ETP was slightly higher, with respect to CN (Table 4). The soils at PG were also shallower than at CN and of coarser texture (Table 3), leading to a smaller water storage capacity which might limit the development of vegetation in PG.

With the exception of J_{mod}, the highest coefficients of variation were also observed in PG (Table 4). Despite the extremely simplified landscapes of both catchments, H_{mod} values were notably high for agricultural systems, particularly in the driest year (2011) with values near to 2.2 and 1.9 in CN and PG, respectively (Table 4). As references, Guzmán and Forester (2007) observed for olive orchards with leguminous cover crops H values close to 1.2, whereas in natural systems of Mediterranean semi-arid areas, H values were approximately equal to 1 (Kawada et al., 2012). Under conventional cereal crops, Armengot et al. (2013) quantified a mean H value of 1.5 for 11 fields in Catalonia (Spain) while for pine afforestation, in a semi-arid catchment in Southwestern Spain, Bonet et al. (2004) came up with H values of 2.8.

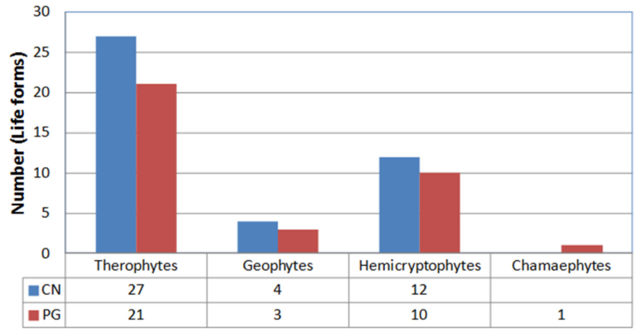

Figure 3. Number of species by life forms (biological spectrum) in the study catchments (CN: La Conchuela; PG: Puente Genil).

On the other hand, J_{mod} values closed to 1, indicated that there were no dominant species in either of the catchments. The lack of a dominant species is frequent in Mediterranean agricultural areas, where a high inter-annual and intra-annual variability of precipitation and temperature produce a wide range of colonizing species awaiting their optimal development conditions. In spite of the selective herbicide treatments (Table 2), differences in J_{mod} between both catchments were small.

Sorensen's index numerically illustrated the notable differences of species existing in the catchments (Tables 4, 5: Fig. 3). It is worth noting that winter was the period when the floristic composition was the most similar ($I_s = 0.378$) and the spring the most different ($I_s = 0.139$). Although similar distributions of life forms were found (Fig. 3), a different floristic catalogue of species was observed in both catchments, where the lack of monocotyledons in PG is remarkable (Table 5). From the soil protection point of view, the current spectrum is not appropriate because most of the species are not permanently present for a long period of the year. However, most of the species constitute the nutritional base for insects and birds. Enrichment of the biological spectrum with hemicryptophytes and chamaephytes is suggested in locations where e.g. hedges are compatible with agricultural operations (Guzmán and Foraster, 2007).

The coefficients of correlation between weather variables (T_m, ETP and P) and seasonal biodiversity indicators (H_{mod}, J_{mod} and R) were in general low (Table 6). Significant cor-

Table 4. Annual values of biodiversity indices: Richness (R), modified Shannon (H_{mod}) and Pielou indices (J_{mod}) and seasonal Sorensen index (I_s). Meteorological attributes: average minimum temperature (T_m), annual evapotranspiration (ETP) and precipitation (P) for both catchments. (CV: coefficient of variation).

Index	Catchment/season	2011	2012	2013	Mean	SD	CV(%)
R	CN	23	26	28	25.7	2.5	9.7
	PG	24	14	24	20.7	5.8	28.0
H_{mod}	CN	2.194	1.947	1.826	1.989	0.187	9.4
	PG	1.880	1.213	1.751	1.614	0.354	21.9
J_{mod}	CN	0.897	0.839	0.850	0.862	0.031	3.6
	PG	0.840	0.834	0.817	0.830	0.012	1.4
I_s	Winter	0.231	0.571	0.333	0.378	0.174	46.0
	Spring	0.231	0.100	0.087	0.139	0.080	57.6
	Summer	0.320	0.000	0.363	0.228	0.198	86.8
	Autumn	0.166	0.333	0.000	0.166	0.167	100.6
T_m (°C)	CN	11.7	11.6	11.1	11.5	0.3	2.6
	PG	12.4	11.6	11.7	11.9	0.4	3.4
ETP (mm)	CN	1270.5	1310.2	1230.4	1270.4	39.9	3.1
	PG	1383.7	1359.8	1355.1	1366.2	15.3	1.1
P (mm)	CN	401	610	621.1	544	124	22.8
	PG	376.8	434.4	423.8	411.7	30.7	7.5

relations were only found for PG as a result of the shallow sandy soil with short-term water availability controlling vegetation. In contrast, the deeper clay soil at CN (Tables 1, 3) enhanced long-term water availability and weakened the correlations between weather variables and biodiversity indicators. Significant negative correlations for *ETP15*, *ETP60* (and *Tm60*) are related to water stress, whereas the positive correlations for short-term indicators such as *Tm15* and *ETP5* might indicate optimal conditions for the seed germination and the growth of grass.

3.2 Relationships between biodiversity indices and indicators of soil quality

In addition to R, J_{mod} and H_{mod}, the mean annual values of SL and R_c, measurements of OM and BD, are also shown in Table 7 and Fig. 4. R, J_{mod} and H_{mod} were not correlated with soil indicators. The highest values of soil losses and the lowest values of organic matter were found in CN. The differences in OM between the catchments were significant as is shown in Table 7 and Fig. 4a (average OM-CN = 1.249 g cm^{-3}; average OM-PG = 1.479 g cm^{-3}). A large quantity of coarse elements was found in PG, which must be taken into account when understanding the differences in BD (Table 7), although they were not significant (Table 7, Fig. 4b; BD-CN = 1.57 g cm^{-3} and BD-PG = 1.50 g cm^{-3}). Substantially higher mean soil loss in CN (16.1 t ha^{-1}) was found with respect to PG (1.8 t ha^{-1}; Fig. 4c). Likewise, the mean R_c in CN (15.3 %) tripled the value of PG (5.1 %; Fig. 4d).

3.3 General discussion

Indicators of spontaneous grass cover biodiversity were not correlated with soil losses and organic matter. The role of cover crops in soil erosion is associated with the dissipation of energy from rainfall and runoff and with the increase of infiltration, which reduces the sediment transport. It was expected that a wider ecological niche would allow for a more efficient occupation of space with probably more biomass, as well as a higher efficiency in the runoff control on the hillslopes. However, in CN, other factors such as precipitation, soil hydrologic characteristics and the possible dominance of concentrated flow (gullies and rills; Gómez et al., 2014b) accounted for higher soil losses and runoff coefficient (much higher than PG values). Lewis et al. (2013) highlighted the potential for soil erosion to disseminate the spontaneous grass cover seedbank and to improve the biodiversity indicators in agro-ecosystems of northern Europe. In natural Mediterranean systems, Cerdá and García-Fayos (2002) and García-Fayos et al. (2010) described the susceptibility to seed removal by water erosion according to seed and landscape features. In this context, an annual sediment delivery ratio of 4 % was found in PG using the SEDD model (Sediment Delivery Distributed; Taguas el al., 2011), while in Conchuela the value was over 90 % indicating an efficient rate of transport, as calculated by Burguet (2015). The different values of soil losses and the annual sediment delivery ratios might illustrate the very different sediment dynamics which contribute towards explaining the greater biodiversity in CN.

Table 5. Species identified in the study catchments present in Puente Genil (PG), Conchuela (CN) or both catchments (Both) for the study period.

Species scientific name	Biological spectrum	Location
	Dicotyledonous	
	APIACEAE (UMBELLIFERAE)	
Daucus carota L.	Hemicryptophites	CN
	ASTERACEAE (COMPOSITAE)	
Anacyclus clavatus (Desf.) Pers.	Therophytes	Both
Anthemis arvensis L.	Therophytes	Both
Calendula arvensis L.	Therophytes	CN
Centaurea melitensis L.	Therophytes	Both
Cirsium arvense (L.) Scop.	Geophytes	Both
Cichorium intybus L.	Hemicryptophites	CN
Conyza sumatrensis (Retz) E. Walker	Therophytes	PG
Chrysanthemum segetum L.	Therophytes	Both
Picris echioides L.	Hemicryptophites, Therophytes	Both
Senecio vulgaris L.	Therophytes	Both
Silybum marianum (L.) Gaertn	Hemicryptophites	CN
Sonchus asper (L.) Hill	Hemicryptophites, Therophytes	Both
Sonchus oleraceus L.	Hemicryptophites, Therophytes	Both
Taraxacum officinale Weber ex F.H. Wigg	Hemicryptophites	Both
Taraxacum obovatum (Willd.) DC.	Hemicryptophites	PG
Pulicaria paludosa Link	Hemicryptophites, Therophytes	Both
	BORAGINACEAE	
Anchusa azurea Mill.	Hemicryptophites	PG
Echium plantagineum L.	Hemicryptophites, Therophytes	Both
Heliotropium europaeum L.	Therophytes	Both
	BRASSICACEAE (CRUCIFERAE)	
Diplotaxis virgata (Cav.) DC.	Therophytes	PG
Raphanus raphanistrum L.	Geophytes, Therophytes	Both
Rapistrum rugosum(L.) Bergeret	Therophytes	Both
Sinapis arvensis L.	Therophytes	CN
	CARYOPHYLLACEAE	
Spergula arvensis L.	Therophytes	PG
Stellaria media (L.) Vill.	Therophytes	Both
	CISTACEAE	
Fumana ericoides (cav) Gand. in Magnier	Chamaephytes	PG
	CONVOLVULACEAE	
Convolvulus arvensis L.	Geophytes, Hemicryptophites	CN
	CRASSULACEAE	
Umbilicus rupestris (Salisb.) Dandy	Hemicryptophites	PG
	CUCURBITACEAE	
Ecballium elaterium	Hemicryptophites	CN

Table 5. Continued.

Species scientific name	Biological spectrum	Location
	FABACEAE (LEGUMINOSAE)	
Ononis pubescens L.	Therophytes	PG
Trifolium repens L.	Hemicryptophites	CN
Trifolium campestre Schreb.	Therophytes	CN
	GERANIACEAE	
Erodium cicutarium (L.) L'Hér	Therophytes	Both
Erodium moschatum (L.) L'Hér	Therophytes	CN
Erodium malacoides (L.) L'Hér	Therophytes, Hemicryptophites	PG
Geranium molle L.	Therophytes	CN
	LAMIACEAE	
Lamium amplexicaule L.	Therophytes	Both
	MALVACEAE	
Malva sylvestris L.	Hemicryptophites	Both
	PAPAVERACEAE	
Fumaria officinalis L.	Therophytes	CN
	POLYGONACEAE	
Polygonum aviculare L.	Therophytes	PG
	PRIMULACEAE	
Anagallis arvensis L.	Therophytes	Both
	RANUNCULACEAE	
Ranunculus arvensis L.	Therophytes	Both
	RUBIACEAE	
Galium aparine L.	Therophytes	Both
	SCROPHULARIACEAE	
Veronica arvensis L.	Therophytes	PG
Veronica hederifolia L.	Therophytes	PG
	URTICACEAE	
Urtica urens L.	Therophytes	PG
	Monocotyledonous	
	LILIACEAE	
Muscari comosum (L.) Miller	Geophytes	PG
	POACEAE	
Bromus hordaceus L.	Therophytes	CN
Bromus madritensis L.	Therophytes	CN
Bromus squarrosus L.	Therophytes	CN
Hordeum murinum L.	Therophytes	CN
Hordeum leporinum (Link)	Therophytes	CN
Lolium rigidum Gaudin	Therophytes	CN
Poa annua L.	Therophytes	CN

Table 6. Matrix of correlation between diversity indices (seasonal values) and climatological features. H_{mod}: Shannon's modified index; J_{mod}: Pielou's modified index; R: richness; P: cumulative precipitation; T_m: average of minimum daily temperatures; ETP: cumulative evapotranspiration. Numbers indicate the interval of previous days (5, 15, 30 and 60).

		$P5$	$P15$	$P30$	$P60$	T_m5	T_m15	T_m30	T_m60	ETP5	ETP15	ETP30	ETP60
CN	H_{mod}	0.12	0.33	0.40	0.39	−0.28	−0.26	−0.25	−0.31	−0.35	−0.36	−0.42	−0.43
	J_{mod}	−0.19	−0.25	−0.20	−0.10	0.55	0.52	0.41	0.17	0.29	0.54	0.55	0.44
	R	0.35	0.52	0.49	0.45	−0.16	−0.17	−0.20	−0.29	−0.25	−0.32	−0.36	−0.37
PG	H_{mod}	0.23	0.29	0.11	0.39	−0.12	−0.05	−0.42	**−0.64**	−0.27	**−0.58**	−0.39	**−0.58**
	J_{mod}	−0.19	−0.29	−0.42	−0.18	0.40	**0.60**	0.29	−0.01	**0.61**	0.26	0.51	0.36
	R	0.29	0.38	0.16	0.36	−0.22	−0.09	−0.42	**−0.61**	−0.35	**−0.62**	−0.46	**−0.61**

$N = 12$; bold indicates correlations are significant at $p < 0.05$.

Table 7. Means and standard deviations of the annual biodiversity indicators and parameters of soil quality. H_{mod}: Shannon's modified index; J_{mod}: Pielou's modified index; R: richness; OM: organic matter content in upper horizon (0–10 cm); BD: bulk density of upper horizon (0–10 cm); SL: annual soil loss; R_c: runoff coefficient (ratio of the annual values of precipitation and runoff).

Catchment	Statistics	R	J_{mod}	H_{mod}	OM[a] (%)	BD[b] ($g\,cm^{-3}$)	SL[c] ($t\,ha^{-1}$)	R_c^c (%)
CN	Mean	25.7	0.86	1.99	1.25	1.57	16.1	15.3
	SD	2.5	0.03	0.19	0.37	0.19	20.8	12.7
PG	Mean	20.7	0.83	1.61	1.48	1.50	1.8	5.1
	SD	5.8	0.01	0.35	0.53	0.25	2.3	4.2

[a] t test showed $p = 0.00054$, CN ($n = 95$), and PG ($n = 65$) (see also Fig. 4a). [b] t test showed $p = 0.07764$, CN ($n = 95$), and PG ($n = 65$) (see also Fig. 4b) [c] See Fig. 3c–d; t test was not carried out because the number of samples was very low, CN ($n = 5$ years), and PG ($n = 6$ years).

As for the values of organic matter content, these might be explained by the management systems. No tillage operations were applied in PG in 2005 and machinery traffic was usually minimal (Table 2), which implies less mechanical soil disturbance than in CN, where productive farm management was carried out. In two sites with a silt loam texture in the Ebro Valley in Spain, Fernández-Ugalde et al. (2009) also described an increase in soil organic carbon content associated with non-tillage practices.

It is important not to confuse non-tillage allowing spontaneous grass cover vegetation, as used in PG, with non-tillage management with herbicide to maintain bare soil in olive orchards. The later led to larger soil losses, runoff coefficients and soil compaction as compared to conventional tillage and cover crops, as was described by Gómez et al. (2004); however, larger carbon and organic matter contents were found in the topsoil, particularly under the canopy (Gómez et al., 1999). As for surface tillage operations in CN, Márquez-García (2013) also found lower values of organic carbon in the topsoil of olive orchards under conventional tillage as compared to cover crops (spontaneous and sown). Near the study catchments, in other agricultural land uses under conservation agriculture, smaller amounts of crop residues, lower soil water contents and larger CO_2 emissions were observed in managements where tillage operations were applied (Cid, 2013).

Despite the annual and seasonal variations of meteorological conditions, overall, a larger availability of water was observed in CN, as a result of the higher annual precipitation and the notably deeper soil. More extensive management did not lead to greater spontaneous grass cover biodiversity in PG compared to CN. Benton et al. (2003) highlighted the importance of differential seed or edaphic factors contributing distinctly to plant growth and to patchiness in the presence of insects. Similarly, Albrecht and Mattheis (1998) found that a management change from conventional to integrated farming in dicotyledonous crops in Germany did not lead to a substantial increment of the number of rare species in spontaneous grass cover. Hyvönen and Huusela-Veistola (2008) described that differences in spontaneous grass cover species numbers between organically and conventionally cropped fields in Finland were small. Similar results were highlighted under Mediterranean conditions by Graziani et al. (2012) for a sequence of six rotations in Italy. They found that the number of spontaneous grass cover species was only slightly higher in organic systems as compared to low-input conventional systems.

Although single steps, such as the application of fertilizers or certain herbicides, may lead to the dominance of some species such as in the case of monocotyledonous in CN (Table 5), no clear sensitivity to the management was found, as described by Albrecht (2003) in Germany or Pysek et

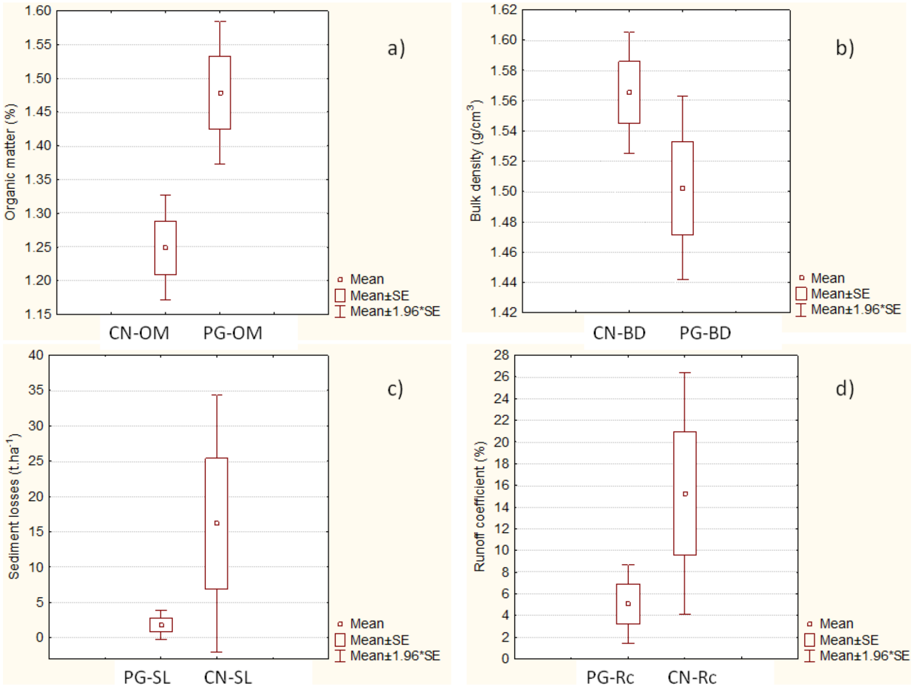

Figure 4. Box and whisker plots of the measurements of soil degradation indicators: **(a)** organic matter content in the upper horizon, **(b)** bulk density in the upper horizon, **(c)** annual soil losses in the catchment outlets, and **(d)** annual runoff coefficients (SE: standard error). For **(a)** and **(b)**, the sample size was 65 in PG and 95 in CN. For **(c)** and **(d)** the sample size was 6 in PG and 5 in CN. The data of **(c)** and **(d)** were described in Taguas et al. (2013) and Gómez et al. (2014b).

al. (2005) in central Europe for different crops. This is likely to be a result of the site conditions in CN being substantially better for vegetation growth, which becomes evident given the olive yields at both catchments (CN, 5000–8000 kg ha^{-1}, and PG < 2000 kg ha^{-1}). In fact, crop yield was also used with other soil properties (such as bulk density, water retention, pH, electrical conductivity, plant-available nutrients, organic matter, microbial biomass, and soil enzymes) by Masto et al. (2007) to define a soil quality index in an agricultural area with a rotation of maize, pearl millet, wheat and cowpea in India. In fact, the yield is a common agronomical factor of soil quality for farmers, which may be well correlated with biodiversity indices of spontaneous grass cover. On the other hand, the traditional metrics used in this study to measure biodiversity – widely used in ecological studies since they are simple to calculate and understand and have been used for a long time (Lamb et al., 2009) – have been criticized because they provide a limited part of the information (Magurran, 2004) and may be unsuitable for monitoring biodiversity intactness (Lamb et al., 2009). These traditional indices, for example, cannot indicate the presence of non-native species or rare plants. In addition to the yield, R, H_{mod}, J_{mod} and I_{s}, the group of species shown in Table 5 support short-term environmental advantages of the vegetation growth found in CN, which is likely to be linked to greater water availability despite a more intense management.

4 Conclusions

Sorensen's index for two olive orchard catchments in the province of Córdoba (Spain) showed notable differences in composition, which were probably associated with the different site conditions. Although CN had a more intense management, its better site conditions (higher precipitation, deeper soils and less steep slopes) can explain the higher values in richness, Pielou's index and Shannon's index. Water stress is a limiting factor for the development of vegetation in the Mediterranean area, so the notable differences in annual precipitation (400 mm in PG versus 600 mm in CN) account for the differences observed. In addition, a more active sediment transport dynamic might contribute to seed dispersal and to increasing the biodiversity indices.

Shannon's index and Pielou's index were relatively high in both catchments, in spite of the major simplifications derived from the agricultural systems. This can be related to the typical Mediterranean dynamics where temporal variability allows different individual species to be incorporated each year according to certain climatological features. The impact of land use and management in both catchments explains the dominance of short-cycle therophytes, hemicryptophytes and cryptophytes, which are extremely resistant to mechanical/chemical treatments since their buds are kept underground. On the other hand, therophytes and hemicryptophytes do not provide efficient soil protection, since their aerial parts are not present during the autumn and winter seasons. However, these species are ecologically important for feeding numerous insects and local birds such as partridge (*Alectoris rufa* L.).

Higher contents of organic matter were determined in PG, the catchment with the worst site conditions in terms of water availability and the least intense management. Additionally, low soil losses have been measured in this catchment. Therefore, biodiversity indicators associated with spontaneous grass cover were not appropriate to describe the soil degradation state in the study areas. Additional efforts should focus on increasing the number of study sites to evaluate if, under more similar environmental conditions, the weight of the management in the olive orchards might determine the biodiversity indices of spontaneous grass cover.

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