



Landscape and land-use effects on weed flora in Mediterranean cereal fields

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ABSTRACT

Land-use intensity and landscape complexity could potentially affect weed flora of agroecosystems, by means of changes in species richness and composition. However, previous results indicate diverging patterns. This paper evaluates the relative importance of both factors on weed species richness and weed community composition within cereal fields in Catalonia (NE Spain). The percentage of arable land plus that occupied by human settlements within a circular sector of 1 km radius was used as a surrogate for landscape complexity and the amount of nitrogen inputs for land-use intensity. The seedbank, which could reflect the long-term effects of the agricultural intensification, and the emerged vegetation with and without weed control were surveyed to obtain comparable assessments of the weed flora, taking into account differences caused by weed control effectiveness. Intensive management induced changes in the emerged weed flora and caused a decline in species richness. Moreover, in the long-term (seedbank), it has also led to a certain level of weed loss. Conversely, landscape complexity had little effect on the seedbanks and emerged flora of cereal fields. Consequently, weed conservation policies in Mediterranean cereal fields should focus on farming practices and need not take account of the surrounding landscape.

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1. Introduction

Most of the literature on biodiversity in agroecosystems states that land-use intensity and landscape complexity could potentially affect arable weed diversity. However, research on the effect of these facets of agricultural intensification has produced opposing results. Fields managed less intensively, mainly under organic farming guidelines, usually harbour more species richness than more intensively managed fields, e.g. under conventional farming (Bengtsson et al., 2005; Hole et al., 2005). Nevertheless, some authors have pointed out that organic farming does not always enhance weed species richness (Kleijn et al., 2001; Weibull et al., 2003). On the other hand, complex landscapes have been reported to benefit weed diversity in arable fields compared to simple ones, since the former support a higher number of habitats, which may act as a source of propagules for colonizing the fields (Gabriel et al., 2005; Roschewitz et al., 2005). However, the high pressure exerted by farming practices, mainly weed control, either chemical or mechanical, and the short dispersal range of most weeds (Benvenuti, 2007) could cause the lack of effect of the surrounding landscape on weed diversity reported in some studies (Marshall, 2009; Gaba et al., 2010). Furthermore, although it is less well-studied, diverging results have also been obtained when assessing the role of land-use and landscape complexity on weed assem-

blages (Weibull and Östman, 2003; Fried et al., 2008; Marshall, 2009; José-María et al., 2010).

Studies on the effect of agricultural intensification on weed communities have mainly concerned the emerged flora. However, standing vegetation depends to a large extent on the specific conditions and the management practices of the growing season and therefore could mask more general tendencies. Overall, weed control, and specifically the application of herbicides, is the agricultural practice that exerts the largest negative influence on weed species richness and abundance (Doucet et al., 1999; Hyvönen and Salonen, 2002) and it is one of the main filters shaping weed community assemblages (Booth and Swanton, 2002). Conversely, weed harrowing in the organic systems is a weaker and less selective filter than herbicides (Booth and Swanton, 2002; Smith and Gross, 2007). The effects of agricultural intensification processes (both land-use intensity and landscape complexity) over the years can be evaluated more accurately by suppressing these practices in a given year. In doing so, differences due to weed control effectiveness between fields during the sampling year are eliminated. On the other hand, the assessment of the seedbank should be also useful to ascertain the effects of agricultural intensification on weed flora, as seedbanks are the result of processes that occurred in the past and, therefore, they can be regarded as the memory of weed communities (Cavers, 1995; Geertsema and Sprangers, 2002). Consequently, the seedbank could reflect the long-term effects of agricultural intensification. In this sense, previous studies have reported changes in weed species richness and composition of the seedbank due to different land-use intensity and landscape complexity (Roschewitz et al., 2005; Hawes et al., 2010).

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The aim of this paper is to evaluate the effect of land-use intensity and landscape complexity on weed flora in dryland cereal fields in a Mediterranean environment. Most previous research addressing this issue has been conducted in central and northern Europe (e.g. Weibull et al., 2003; Gabriel et al., 2005). However, the present area of interest is characterized by an uneven topography and the scarcity of broad plains where intense cultivation is concentrated (Folch et al., 1997), high diversity of non-cultivated habitats interspersed among fields in both simple and complex landscapes (Lavorel, 1999) and low rainfall, which modifies competitive relations among plants (Liancourt et al., 2005). These particularities should allow instructive comparisons with results from more temperate areas. Weed species richness and composition of the seedbank and of the emerged vegetation, with and without weed control practices, were analysed in organic and conventional winter cereal field pairs located on a gradient of landscape complexity. The questions aimed to answer were: (1) what is the relative importance of landscape complexity and land-use intensity on weed species richness and composition in winter cereal fields? and (2) does the relative importance of these factors differ according to the source of data: seedbank vs. emerged vegetation; with vs. without weed control?

2. Materials and methods

2.1. Study site

The study was conducted between 2006 and 2007 in Central Catalonia (north-eastern Spain), at 11 localities differing in agricultural landscape complexity. The area of interest covered approximately 100 km × 50 km, extending from 1° 05' to 2° 05' E and from 41° 24' to 42° 05' N and included simple landscapes with a high percentage of arable land around the selected fields (ca 99% in a sector of 1 km radius) and complex landscapes with a lower percentage of arable land (ca 25%). Simple landscapes were dominated by arable fields, with winter barley and wheat as the main crops, while woodlands, shrublands and perennial natural grasslands appeared intermingled between fields in complex landscapes. The mean (±SE) altitude was 537 ± 55 m a.s.l. (min = 250 m, max = 800 m). In the study area complex landscapes are mainly found in relief land, whereas simple ones are found on the plains. The geological substrate is constituted mainly by marls and calcareous sandstones, leading to basic soils with a loamy-clayish texture (% clay content of the selected fields: 21.0 ± 1.1%; min = 12.8%, max = 28.6%). The climate is Mediterranean, with mean precipitation from September 2006 to August 2007 of 513 ± 15 mm (min = 350 mm, max = 650 mm), which is within the average annual rainfall range (500–600 mm), and mean annual temperatures of 12.7 ± 0.2 °C (min = 11.5 °C, max = 13.5 °C) (Ninyerola et al., 2005; Servei Meteorològic de Catalunya, 2008).

Apart from the gradient in landscape complexity, the main criterion for selecting a locality was the presence of farms which had been managed organically at least since 2001 (median = 1996). In each locality, two nearby farms, one organic and one conventional, were selected, and from each farm, one winter wheat or barley field (22 fields were selected, 11 organic and 11 conventional fields). Fields were representative of the study area. They shared similar area (conventional: 1.68 ± 0.22 ha, mean ± SE; organic: 1.39 ± 0.20 ha; Wilcoxon's paired tests within locality $P=0.31$), perimeter (conventional: 558.2 ± 24.82 m; organic: 512.16 ± 37.32 m; $P=0.51$) and shape (area/perimeter: conventional: 2.94 ± 0.25 ha m⁻¹; organic: 2.64 ± 0.24 ha m⁻¹; $P=0.34$), and these properties were not correlated with the landscape complexity, assessed as the percentage of natural habitats (area: Spearman's correlation coefficient $\rho=-0.24$, $P=0.28$; perime-

ter: Spearman's $\rho=0.07$, $P=0.76$; shape: Spearman's $\rho=-0.07$, $P=0.70$). The most frequent habitats in the boundaries were dry perennial grasslands, lowland ruderal communities, bramble dominated thickets and Mediterranean grasslands (pseudo-steppe with grasses and annuals forbs interspersed between dwarf shrubs). The height of the vegetation was relatively low (<1 m), mainly for the grasslands and ruderal communities. However, some thickets and sparse trees measured more than 2 m. All farmers sowed their crop between September and October 2006.

2.2. Landscape complexity analysis

Landscape complexity was measured around each studied field in a circular sector of 1 km radius using the Catalan Habitats Cartography (Carreras and Diego, 2004), produced at a scale of 1:25,000. The proportion of arable land has been widely used as a surrogate of landscape complexity (e.g. Gabriel et al., 2005). Nevertheless, in the Mediterranean landscape studied, arable land is usually interspersed with minor crops (vineyards and almond groves) and human settlements. Therefore, they were added to arable land to obtain an estimator of the landscape simplification, hereafter PIL (percentage cover of intensive land-use; José-María et al., 2010). PIL is complementary to the proportion of natural habitats, which could act as a source for seed dispersal into the fields; previous analyses have shown that PIL is negatively correlated with habitat diversity and positively correlated with habitat fragmentation (Romero et al., 2008a). Moreover, as expected, PIL positively correlated with the flatness of the sites (A. Romero, unpublished results). However, PIL did not correlate with mean annual rainfall (Spearman's correlation coefficient $\rho=-0.24$, $P=0.48$), mean annual temperatures (Spearman's $\rho=0.19$, $P=0.57$) and mean altitude (Spearman's $\rho=-0.28$, $P=0.40$). Hence, it could be stated that the potential effects of landscape complexity are not a consequence of these environmental factors, which could also affect weed flora and blur the interpretation of the results.

2.3. Land-use intensity analysis

Each farmer was interviewed to characterise farming practices in the selected fields for the last 5 years. Questions were used to score the following items: weed control (use of herbicides or harrowing with long-flex spring tines), crop diversity (number of different plant botanical families in the rotational scheme), cereal ratio (proportion of cereal crops in the last 5 years) and mean annual inputs of exogenous nitrogen, estimated from the type and the amount of fertilisers using local tables of nitrogen content (Arco and Romanyà, 2010). The main source of nitrogen in conventional fields was inorganic fertilisers, but manures and pig slurry were also applied. Conversely, composted manures were used in organic fields. Preliminary analysis showed high variability in management within organic and conventional fields, but especially among the organic ones (Table 1), which led us to use a more adequate proxy for land-use intensity than the classical organic-conventional dichotomy (Armengot, 2010). Therefore, the correlations among the management variables and their relationship with PIL were analysed. Finally, the mean annual inputs of exogenous nitrogen (hereafter N) were selected as an indicator of land-use intensity, which has been previously used in other studies (Firbank et al., 2008; Kleijn et al., 2009; José-María et al., 2010). N was significantly higher in conventional fields than in organic ones (Student's t -test, $t=-4.69$, $P<0.001$) and it correlated significantly with cereal ratio (Spearman's correlation coefficient $\rho=0.69$, $P<0.001$) and crop diversity ($\rho=-0.62$, $P=0.003$). Moreover, it was independent of PIL when both conventional and organic fields were analysed

Table 1

Mean \pm SE, minima and maxima for the data on agricultural practices of 11 organic and 11 conventional winter cereal fields. Data were obtained through questionnaires from the farmers and summarize the management practices for the last five years before the survey. Nitrogen inputs: mean annual inputs of exogenous nitrogen (kg ha^{-1}); cereal ratio: proportion of cereal crops in a 5-year rotational scheme, 1 accounts for cereal monoculture; crop diversity: number of different plant families introduced in the rotation; weed control: proportion of farmers that control weeds (organic farmers controlled weeds by harrowing with long-flex spring tines once a year and conventional farmers applied herbicide once a year as a maximum).

	Organic			Conventional		
	Mean \pm SE	Min	Max	Mean \pm SE	Min	Max
Nitrogen inputs (kg ha^{-1})	43 \pm 11	0	108	148 \pm 18	58	228
Cereal ratio	0.42 \pm 0.10	0.20	0.79	0.90 \pm 0.05	0.67	1
Crop diversity	3.45 \pm 0.40	2	5	1.26 \pm 0.19	1	3
Weed control	5/11	–	–	11/11	–	–

together ($\rho = 0.11$, $P = 0.63$) or separately (conventional: $\rho = -0.05$, $P = 0.88$; organic: $\rho = 0.21$, $P = 0.56$).

2.4. Experimental design

After cereal sowing, four $7 \text{ m} \times 7 \text{ m}$ blocks were randomly placed in the centre of each field (at least 10 m away from the field edge). In each block, three $2 \text{ m} \times 2 \text{ m}$ plots 3 m apart were delimited and two of them were assigned to weed control treatment. In one plot, weeds were controlled (hereafter, weeded) by means of herbicides in the conventional fields and harrowing with long-flex spring tines in the organic ones, while weeds were not controlled in the other plot (hereafter, non-weeded). In conventional fields, these plots were protected with a plastic sheet during herbicide application, which was removed immediately after the treatment. Similarly, non-weeded plots in the organic fields were delimited with stakes to avoid weed harrowing. Weed control (both herbicide spraying and weed harrowing) was performed once, between January and February. The third plot of each block was used for the seedbank sampling (see below).

2.5. Weed sampling

Aboveground weed sampling was conducted in May 2007 by means of weed species scouting. The presence of every weed species in each $2 \text{ m} \times 2 \text{ m}$ plot (4 weeded and 4 non-weeded plots per field) was recorded; nomenclature follows de Bolòs et al. (2005). Weed species richness was selected as a proxy for weed diversity because of its simplicity and sensitivity to agricultural intensification (Guerrero et al., 2010). The weed seedbank was sampled in autumn 2006, immediately after cereal sowing. Twenty-four soil samples (3 cm diameter \times 10 cm depth) were randomly taken from one $2 \text{ m} \times 2 \text{ m}$ plot in each block and were mixed to obtain a composite soil sample per block. They were crumbled and spread evenly on three $23 \text{ cm} \times 15 \text{ cm} \times 4 \text{ cm}$ aluminium trays mixed with 0.5 dm^3 of inert substrate (perlite and vermiculite) to ease drainage. Trays were placed on a bench in a non-heated greenhouse where they were kept well watered and under natural light. The bench was covered with a mosquito net to prevent contamination with seeds from the surrounding area. Positions of the trays were randomized every 1–2 weeks. Emergence was monitored for 14 months and samples were mixed monthly to favour the germination of seeds and prevent the growth of mosses. Initially seedlings were recorded weekly, but the census interval was increased after six months. At each census, new seedlings were counted and removed if they were identified or marked for later identification.

2.6. Weed species richness analysis

Response variables were analysed using linear mixed-effect models, which can account for random effects and nested sampling designs (Pinheiro and Bates, 2000). The effects of landscape complexity (PIL), land-use intensity (N) and their interaction ($N \times \text{PIL}$)

on aboveground weed species richness in weeded and non-weeded plots and on the seedbank were tested as fixed factors. Locality and field were introduced as random factors. The same analysis was performed for rare weed species, considering those species that only appeared in one or two localities. For statistical reasons, due to the low occurrence of these weeds, data of the four blocks per field were considered to obtain a single value of rare weed species richness per field, and thus only locality was introduced as a random factor. As absolute values of PIL and N differed to a great extent, they were standardized to have a mean of zero and a standard deviation of one, which enabled comparison of their effects based on regression coefficients (Hendrickx et al., 2007). Species richness was log-transformed to achieve normality of the residuals. Adequacy of the models was tested through normality and unbiasedness of residuals, and through the predictive power of the model.

Statistical analyses were performed following the multimodel inference method (Burnham and Anderson, 2002). This method allows inference to be made on the basis of all the models from an *a priori* set, rather than based only on the best estimated model. The set of candidate models (5 models) comprised all the possible combinations of the variables, including the null model. For each analysis, individual models were compared by Akaike's information criterion corrected for small samples (AICc) comparing the loss of information of each model in relation to the best model, which has the minimum value of AICc ($\Delta_i = \text{AICc}_i - \text{AICc}_{\min}$). Afterwards, the Akaike weight (w_i) of each model was calculated, which can be interpreted as the "weight of evidence" in favour of a given model as being the actual best model in the set of candidate models (Burnham and Anderson, 2004). Model-averaged parameter estimates and their unconditional standard errors were calculated from the entire set of models. The 95% confidence intervals were estimated to assess the magnitude of the effect of each variable. Statistical analyses were carried out under R 2.7.1 (R Development Core Team, 2008) with package "lme4" (Bates et al., 2008) for mixed models.

2.7. Weed composition analysis

Species composition of each survey (emerged flora in weeded and non-weeded plots and seedbank) was analysed using multivariate analysis based on presence/absence data. Species present in only one locality were removed. A permutational multivariate analysis of variance was performed using distance matrices (the Jaccard dissimilarity index was applied) to analyse how N and PIL affect emerged and seedbank species composition. This analysis allows partitioning a distance matrix among sources of variation and fitting a linear model to it. The significance of each explanatory variable was obtained by means of *F*-tests based on sequential sums of squares from permutations of the raw data, restricting permutations within each locality to take into account the hierarchical sampling design.

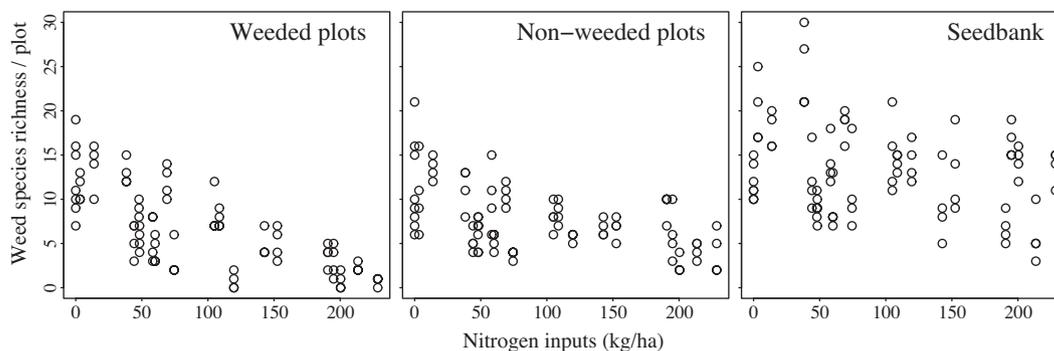


Fig. 1. Weed species richness of the emerged flora (weeded and non-weeded) and of the seedbank in relation to land-use intensity (total amount of nitrogen inputs).

In addition, a Non-metric Multidimensional Scaling (NMDS) analysis was performed, which is deemed the most robust unconstrained ordination method in community ecology (Minchin, 1987). The number of dimensions was restricted to two, as it facilitates the graphic representation of the results. For each factor (N or PIL) with a significant effect on weed species composition (according to the permutational multivariate analysis of variance), a smooth surface (using thinplate splines) was fitted onto this ordination to represent graphically the relationship between the ordination and the given factor. Compositional analyses were carried out using the “vegan” package for R (Oksanen et al., 2009).

3. Results

3.1. Weed species richness

In total, 122 weed species were recorded: 81 in the aboveground surveys (79 in non-weeded plots and 70 in weeded plots) and 92 in the seedbank. Mean (\pm SE) weed species number per field was 11.9 ± 1.5 (min=2, max=26) in the emerged vegetation of weeded plots, 13.2 ± 1.1 (min=4, max=28) in the emerged vegetation of non-weeded plots and 23.6 ± 1.4 (min=11, max=40) in the seedbank. Thirty-three species only appeared in the aboveground vegetation and 44 only in the seedbank. Many weeds were detected in only one or two localities (e.g. *Centaurea scabiosa*

L., *Asperugo procumbens* L., *Matricaria recutita* L.) and were considered rare weeds. Thirty-six rare species were found in the emerged vegetation without weed control (2.27 ± 0.57 , mean \pm SE; min=0; max=9), 43 in the emerged vegetation of weeded plots (2.59 ± 0.62 , mean \pm SE; min=0; max=10) and 51 in the seedbank (3.13 ± 0.57 , mean \pm SE; min=0; max=10). By contrast, only ca 17% of weed species appeared in more than 25% of the selected fields (e.g. *Lolium rigidum* Gaudin, *Papaver rhoeas* L., *Polygonum aviculare* L.). These results indicate a high variability in local occurrence.

Emerged and seedbank weed species richness were influenced by land-use intensity (Fig. 1), expressed in terms of N input. In all the set of surveys, models including only N had the highest Akaike weight (w_i), i.e. the “weight of evidence” of being the best model in the set of candidate models (Table 2). Moreover, the effect of land-use intensity was strongly supported by the high estimated magnitude of N and its confidence intervals, which do not include 0 (Table 3). The major effect of land-use intensity on weed species richness was found in weeded plots, whereas it was weaker in the seedbank, as shown by the regression coefficients. Conversely, any effects of landscape complexity and its interaction with land-use intensity were not supported either by their confidence intervals or by their estimated coefficients. Similar results were obtained when analysing rare weed species (Tables 2 and 3). Again, for these species land-use intensity was the only factor supported by the

Table 2
Candidate models for all and rare weed species richness (log-transformed) of the emerged flora (weeded and non-weeded) and the seedbank. The table compiles the variables included in the models: N, nitrogen inputs; PIL, percentage of cover of intensive land-use; AICc, Akaike’s information criterion corrected for small samples; Δ_i , differences between the AICc for a given model and the best fitting model; w_i , Akaike weights (more details in text).

Models	All weeds			Rare species		
	AICc	Δ_i	w_i	AICc	Δ_i	w_i
(a) Weeded						
N	98.37	0.00	0.51	53.11	0.00	0.77
PIL	117.85	19.48	0.00	61.84	8.73	0.00
N + PIL	99.43	1.06	0.30	56.50	3.40	0.14
N + PIL + N \times PIL	100.39	2.02	0.19	59.35	6.25	0.03
Null model	115.63	17.26	0.00	58.94	5.84	0.04
(b) Non weeded						
N	52.35	0.00	0.64	46.76	0.00	0.77
PIL	62.61	10.26	0.00	55.62	8.86	0.00
N + PIL	54.51	2.16	0.22	49.95	3.18	0.16
N + PIL + N \times PIL	55.47	6.11	0.13	53.80	7.03	0.02
Null model	60.37	8.02	0.01	52.60	5.84	0.04
(c) Seedbank						
N	34.80	0.00	0.49	54.83	0.00	0.45
PIL	37.83	3.03	0.11	58.06	3.22	0.09
N + PIL	36.71	1.91	0.19	58.17	3.33	0.08
N + PIL + N \times PIL	38.97	4.17	0.06	61.40	6.57	0.02
Null model	37.08	2.28	0.16	55.27	0.44	0.35

Table 3

Model-averaged estimate, unconditional standard error (UnSE) and 95% confidence interval (CI) for predictor variables and their interactions for all and rare weed species richness (log-transformed) of the emerged flora (weeded and non-weeded) and the seedbank. N, nitrogen inputs; PIL, percentage of cover of intensive land-use.

	All weeds			Rare species		
	Estimate	UnSE	CI	Estimate	UnSE	CI
(a) Weeded						
Intercept	1.79	0.09	1.97, 1.60	0.94	0.14	0.70, 1.24
N	-0.54	0.09	-0.35, -0.73	-0.46	0.14	-0.73, -0.19
PIL	0.12	0.10	0.30, -0.07	0.01	0.14	-0.27, 0.29
N × PIL	0.12	0.10	0.31, -0.07	0.12	0.10	-0.14, 0.44
(b) Non-weeded						
Intercept	2.06	0.06	2.19, 1.94	0.94	0.12	1.17, 0.70
N	-0.23	0.06	-0.11, -0.36	-0.40	0.12	-0.64, -0.16
PIL	0.03	0.07	0.16, -0.10	0.05	0.12	-0.19, 0.30
N × PIL	0.10	0.06	0.23, -0.02	0.00	0.13	-0.26, 0.26
(c) Seedbank						
Intercept	2.59	0.08	2.76, 2.43	1.19	0.14	0.90, 1.48
N	-0.11	0.05	-0.01, -0.21	-0.28	0.14	-0.56, -0.01
PIL	-0.06	0.08	0.09, -0.22	-0.06	0.15	-0.36, 0.24
N × PIL	-0.02	0.06	0.10, -0.14	0.15	0.02	-0.14, 0.18

confidence intervals and the estimated coefficients, and its effect was stronger on the aboveground vegetation than on the seedbank.

3.2. Weed species composition

The effect of land-use intensity on weed species composition was only evident in weeded plots (Table 4), although it only explained about 10% of their variation. The surface fitted onto the NMDS analysis, which represents graphically land-use intensity, showed that some species were related to intense land-use (high values of N; i.e. *Galium aparine* L., *Polygonum aviculare*, *Lolium rigidum*), while others appeared preferentially in fields where land-use intensity is low (low values of N; i.e. *Convolvulus arvensis* L., *Diplotaxis erucoides* (L.) DC., *Galium tricorntum* Dandy); test of smooth terms: $F=7.06$, $P<0.001$ (Fig. 2). As with the findings on weed species richness, no significant effect of landscape complexity or of its interaction with land-use intensity was found on weed composition in any of the surveys analysed.

4. Discussion

4.1. Land-use intensity

The positive effects of reduced-intensity farming on weed species richness reported in this paper are in accordance with previous studies (Bengtsson et al., 2005; Hole et al., 2005; Kleijn et al., 2009), which highlight the influence of agricultural practices on weed diversity. Reduced intensive-farming favoured the occurrence of all weeds within the fields, but also of rare weeds, which include the species with more conservation value. Interestingly, the magnitude of the effect of land-use intensity differed depending on the type of data (seedbank vs. emerged flora; with vs. without weed control). Intensive land-use had more detrimental effects on weed species richness in weeded plots than in the seedbank or in the emerged vegetation without weed control, which is clearly related to the direct negative effect of weed control practices. Non-weeded plots revealed that intensive management exerts long-lasting negative effects (Ryan et al., 2010), since a certain level of weed species loss due to land-use intensification was found even when weed control practices were suppressed. This pattern of species richness depletion was also confirmed by the seedbank analysis. However, the inherent variability of the seedbank (Davis et al., 2005), coupled with the presence of some weeds that are seldom found in the emerged flora of dryland cereal fields (ca 11% of the total species in the seedbank; e.g. *Portulaca oleracea* L., *Juncus bufonius* L., *Scirpus*

holoschoenus L.), could explain the weaker relationship between land-use intensity and the species richness found.

In contrast to the pattern found for species richness, weed species composition was only affected by land-use intensity in weeded plots, although it explained a small part of the total variability (ca 10%). These results could be a consequence of the low occurrence of weeds, since most of them only thrived in few localities (ca 50% of species found were considered rare species).

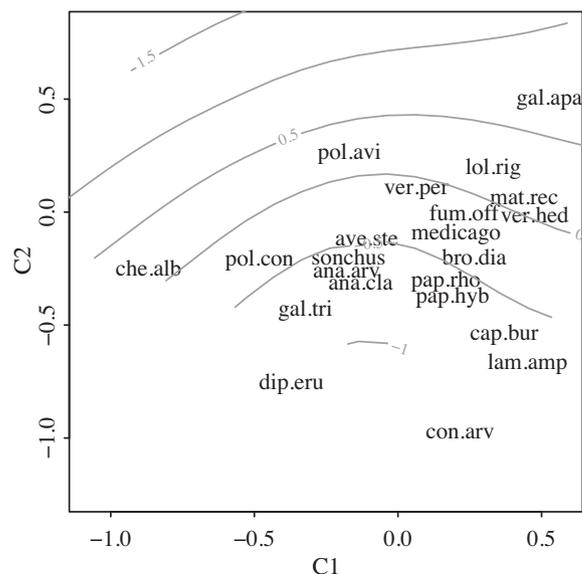


Fig. 2. Non-Metric Multidimensional Scaling (NMDS) analysis ($k=2$, non-metric fit: $r^2=0.956$) based on floristic similarities of weed species of weeded plots. A smooth surface (contours) representing land-use intensity (N inputs) is included to test the relationship between the ordination of weed species obtained in the analysis and land-use intensity. C1 and C2 are the arbitrary axes extracted by the NMDS method from the Jaccard distance matrix between fields/samples. Species codes: ana.cla=*Anacyclus clavatus* (Desf.) Pers.; ana.arv=*Anagallis arvensis* L.; ave.ste=*Avena sterilis* L.; bro.dia=*Bromus diandrus* Roth; cap.bur=*Capsella bursa-pastoris* (L.) Medik.; che.alb=*Chenopodium album* L.; con.arv=*Convolvulus arvensis* L.; dip.eru=*Diplotaxis erucoides* (L.) DC.; fum.off=*Fumaria officinalis* L.; gal.apa=*Galium aparine* L.; gal.tri=*Galium tricorntum* Dandy; lam.amp=*Lamium amplexicaule* L.; lol.rig=*Lolium rigidum* Gaudin; mat.rec=*Matricaria recutita* L.; medicago=*Medicago* sp.; pap.hyb=*Papaver hybridum* L.; pap.rho=*Papaver rhoeas* L.; pol.avi=*Polygonum aviculare* L.; pol.con=*Polygonum convolvulus* L.; sonchus=*Sonchus* sp.; ver.hed=*Veronica hederifolia* L.; ver.per=*Veronica persica* Poiret in Lam. Only species present in more than a 15% of the fields are shown. Nomenclature follows de Bolòs et al. (2005).

Table 4
Permutational multivariate analysis of variance in weed species composition conducted for the emerged flora (weeded and non-weeded) and the seedbank. SS, sum of squares, r^2 , partial R -squared, N, nitrogen inputs; PIL, percentage of cover of intensive land-use.

	Weeded		Non-weeded		Seedbank	
	SS	r^2	SS	r^2	SS	r^2
N	0.624	0.094***	0.292	0.049	0.170	0.035
PIL	0.405	0.061	0.479	0.081	0.499	0.103
N × PIL	0.278	0.042	0.349	0.059	0.180	0.037
Total	6.621		5.918		4.828	

*** $P < 0.001$.

Additionally, the fact that a significant relationship between land-use intensity and weed species composition was only consistent in weeded plots reflects that the emerged weed flora responds strongly to yearly management events (Hawes et al., 2010), and especially to weed control practices, indicating their importance in determining actual weed assemblages. Weed assemblages of fields with herbicide applications (also corresponding to those managed more intensively), differ from those subject to harrowing because, as has been previously reported, herbicides are one of the most important factors in determining weed assemblages of the emerged flora (Booth and Swanton, 2002; Légère et al., 2005). Conversely, the lack of such differences in weed seedbank composition highlights their ability to buffer weed population fluctuations due to agricultural practices (Légère et al., 2005; Harbuck et al., 2009).

4.2. Landscape complexity

Data from cereal fields does not support the role of landscape complexity on promoting weed species richness (for both total and rare species) nor on causing changes in community composition. Therefore, they do not support the idea that complex landscapes compensate for the negative effects of farming intensification on weed flora (Compensation for landscape complexity hypothesis, Tschardt et al., 2005). The lack of influence of the surrounding landscape in this study may be attributed to several causes. Firstly, the low range of seed dispersal of many weeds (Bischoff and Mahn, 2000; Geertsema and Sprangers, 2002; Devlaeminck et al., 2005; Benvenuti, 2007) could limit their spread into the fields. Secondly, although seeds from the surrounding habitats could reach arable fields, agricultural practices could hinder the establishment of viable populations. Weed control practices, and in general all farming operations (i.e. soil tillage, cereal sowing, fertilization), are more efficiently performed within fields than at the edges (Wagner and Edwards, 2001; Romero et al., 2008b). Hence, in inner fields, due to the higher agricultural pressure and the lower probability of seed arrival because of the distance to field margins (Marshall, 1989), it should be more difficult to detect any effect of the landscape context. In this sense, José-María et al. (2010) reported that landscape complexity affects weed flora differently depending on the location within the fields, and that the surrounding landscape influences emerged weed flora in boundaries and field edges but not within the field. Moreover, most studies reporting some effect of the landscape context on weeds have been performed on field boundaries or merging data from the inner field and crop edges (Weibull et al., 2003; Roschewitz et al., 2005; Boutin et al., 2008). Thus, these results agree with other studies performed in inner fields, which have not found any significant effect of the surrounding landscape (Marshall, 2009; Gaba et al., 2010).

The experimental layout assessing the emerged vegetation with and without weed control and also the seedbank has provided a better understanding of the role of landscape complexity under different levels of management intensity. In this sense, the similar pattern of species richness in non-weeded and weeded plots in relation to landscape complexity reveals that weed control prac-

tices do not blur the effect of landscape complexity. Moreover, the analyses of the seedbank, which reflects processes that occurred in the past, did not show any change in weed species richness or composition due to different landscape complexity. Thus, the similar pattern found across data sets, confirms that landscape complexity has a negligible effect on weed flora within cereal fields in this study area.

5. Conclusions

The results of this study point out that intensive management has negative effects on weed flora. This becomes evident by changes in weed species composition and by a decline in weed species richness which also affects rare species, whose conservation is an issue of growing concern. Moreover, in the long-term, these effects have led to a certain level of weed diversity erosion. Conversely, these findings reveal that landscape complexity has a limited role in affecting weed flora of inner fields. Accordingly, strategies for weed flora conservation within arable fields in a Mediterranean context should focus on promoting low-intensity agricultural practices rather than on the surrounding landscape. Further research should assess the effects of land-use intensity on crop-weed competition in Mediterranean areas in order to develop suitable practices that balance productivity and biodiversity conservation.

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