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Original article

Effects of urbanisation on the parasitoid community of a leafminer

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ABSTRACT

Urbanisation may have detrimental effects on communities of parasitoids, affecting their species richness, abundance, and species dominance. Here we investigated the influence of the degree of urbanisation on parasitoid communities of *Liriomyza commelinae* (Frost) (Diptera: Agromyzidae), a leafminer of *Commelina erecta* L. (Commelinaceae), in the city of Córdoba, Argentina. To study changes in species richness, the specific composition of parasitic complexes and their degree of impact on the leafminer, 18 sampling sites from the centre to the outskirts of the city were selected and different variables indicative of urbanisation were quantified in each site. During January and February of 2005 and 2006, all mined leaves found in each plant patch were collected and the following variables were estimated: proportion of mined patches, abundance of the leafminer, total parasitoid species richness, total parasitism rates and parasitism due to the most abundant parasitoid species. The percentage of mined patches and leafminer abundance increased with urbanisation degree. Estimates of parasitoid species richness were not influenced by urbanisation degree but increased with species richness of mined plants. Changes in the specific composition of species along the urbanisation gradient were observed. Although parasitism by one of the species studied was higher in more urbanised sites of the city, the total parasitism rate of *L. commelinae* was not affected by urbanisation degree, species richness of mined plants or leafminer abundance. It appears that urbanisation benefits the herbivore species here studied but not through altering parasitoid activity. Changes in parasitoid community composition reflex dissimilar tolerance to environmental conditions displayed by different parasitoid species.

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1. Introduction

Urbanisation is a dominant demographic trend and an important component of global land transformation that is increasing day by day (Niemelä, 1999; Pickett et al., 2001).

Recent projections indicate that the world population living in urban areas will reach 60% by 2030 (United Nations, 2004), so a key challenge for conservation is to understand how urbanisation affects biodiversity (McKinney, 2002). The complex nature of urban habitats can have different effects on

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biotic communities (McKinney, 2008), with some species being favoured and others negatively affected (Niemiälä, 1999). On the one hand, urbanisation can promote an increase in biodiversity, usually by the incorporation of exotic species that replace native species faster than they are lost (McKinney, 2002, 2006) or by the high spatial heterogeneity resulting from different land uses, which offers a diversity of habitat types often supporting high species richness (Frankie and Ehler, 1978; Niemiälä, 1999). On the other hand, and as occurs in most cases, urbanisation causes a decrease in species richness, with the lowest richness to be found in the urban core (McKinney, 2002, 2008).

This decline in species richness can be attributed to several urban conditions mostly due to habitat fragmentation and environmental degradation (McDonnell and Pickett, 1990; Rebele, 1994; Pickett et al., 2001; McKinney, 2008). Reduced size of remnant patches and high habitat isolation caused by the presence of physical barriers and impervious surfaces (e.g. buildings and pavement) have shown to change community structure of arthropods (Gibb and Hochuli, 2002) and to reduce the number of insect species (Denys and Schmidt, 1998) respectively. Impervious surface cover also affects urban climate, generating the formation of urban “heat islands” (Botkin and Beveridge, 1997; Shochast et al., 2006), being the urban core the zone with higher temperatures (Prats et al., 2005). Although it is known that “heat island” effect extends the thermal window for arthropods (Backer et al., 2002), the actual impact of urban warming on insect populations is unknown (McIntyre, 2000). Urban habitat quality also is changed by the effects of high levels of air pollutants from industrial and traffic sources (McIntyre, 2000) which can adversely affect biota and ecosystems (Woodwell, 1970; Przybylski, 1979; Klump et al., 2000; Jana et al., 2006; Zvereva et al., 2008).

There is a great amount of evidence about the detrimental impact of urbanisation on abundance, species richness and community composition in different groups of insects (McGeoch and Chown, 1997; McIntyre and Hostetler, 2001; Ishitani et al., 2003; Weller and Ganzhorn, 2004; Yamaguchi, 2004; Rango, 2005). However the effects of urban characteristics on parasitoid community structure and host–parasitoid interactions have rarely been investigated (Frankie and Ehler, 1978; Denys and Schmidt, 1998). Here we examined how different levels of urbanisation influence the parasitoid community of *Liriomyza commelinae* (Frost) in the city of Córdoba, Argentina. This species is a specialist agromyzid fly, of neotropical origin, whose larvae mine the leaves of *Commelina erecta* L. (Commelinaceae), a native “ruderal” plant widely distributed in the north and centre of the country, especially in dry sandy soils (Sérsic et al., 2006) of urban and cultivated habitats. In central Argentina, *L. commelinae* has been registered associated to 19 native parasitoid species all belonging to hymenopteran families (Salvo and Valladares, 1998) and suffers parasitism levels around 50% (Salvo, 1996).

To date, we know of only two studies which have dealt with populations and communities of parasitoid insects in urban environments (Sawoniewicz, 1986; Denys and Schmidt, 1998), both reporting conflicting results. While one of these studies found that parasitoid species richness and parasitism rates decreased from the non-urbanised areas to the urbanised ones (Denys and Schmidt, 1998), the other registered

a higher number of parasitoid species in suburbs and parks in urban environments than in the natural habitat (Sawoniewicz, 1986). Results in the literature on air pollution in urban areas are also contradictory, showing positive (Stiling et al., 1999; Zvereva and Kozlov, 2000), negative (Zvereva and Kozlov, 2006), or nil (Koricheva, 1994) effects of pollution on parasitoid attack. In cases when air pollution negatively affects natural enemies, searching behaviour of insect parasitoids could be reduced (Gate et al., 1995), which may create an enemy-free space for herbivores leading to an increase in their population densities (Zvereva and Kozlov, 2006). Parasitoid communities can also be affected by habitat type (Marino and Landis, 1996; Salvo et al., 2005), and evidence indicates that they tend to be more sensitive to habitat fragmentation than their herbivorous hosts (Tscharntke and Brandl, 2004). Therefore, the study of the effects of urban development on parasitoid community structure is interesting not only when considering top-down control of herbivores but also for the potential use of this group as “indicators” of different environmental disturbances (La Salle and Gauld, 1993; Paoletti and Cantarino, 2000), including urbanisation (McIntyre, 2000).

In this study we compared parasitoid species richness, parasitoid community structure and parasitism rates observed for *L. commelinae* in sites with different urbanisation degree within the city of Córdoba, Argentina. For this purpose, we also examined the effect of urbanisation on leafminer density. We expected that: 1) being most of the parasitoid species of *L. commelinae* highly polyphagous (Salvo and Valladares, 1998, 2004), sites with lower level of urbanisation would have greater diversity of plants and potential leafminer hosts which ultimately may favour parasitoid species richness and parasitism levels; 2) the composition of specific parasitoid communities would be similar in sites with similar degree of urbanisation, as a consequence of the different tolerance to environmental conditions displayed by diverse parasitoid species; 3) parasitism rates would be low in zones with higher levels of urbanisation, given the higher sensitivity of parasitoids than herbivores to environmental disturbance (Tscharntke and Brandl, 2004); and 4) leafminer density would be higher in sites located in the urban core since *L. commelinae* is a specialist on a ruderal plant species which is very common in urban areas, and also because plants are expected to be more susceptible to herbivore attack in the core, where habitat conditions would be more stressful for them, increasing their susceptibility to insect colonisation (Connor et al., 2002). Also, as mentioned before, parasitoids will be less efficient in these areas given their high sensibility to disturbance.

2. Materials and methods

The study was carried out within Córdoba city, which is situated in the centre of Argentina (31° 20' S, 64° 10' W, elevation 440 m). With a population of 1,284,532 inhabitants (INDEC, 2001) Córdoba City covers an area of 562 km² (Dirección de Catastro, Córdoba, 2007). Within most urbanised areas, human activities have resulted in a landscape that is characterised by a highly developed urban core and a trend toward diminishing anthropogenic influences at the periphery

(Medley et al., 1995). We selected 18 sampling sites differing in their distance from the city centre in order to generate an urbanisation gradient (Fig. 1). Most studies on the influence of cities on invertebrates communities used the degree of urbanisation, classified in three classes (urban, suburban, and rural). However, measuring specific features of urban habitat can give a better quantification in gradient description and also improve the understanding of the main effects of urbanisation. Each site was represented by 9 blocks, in which different variables indicative of urbanisation were quantified in the first year of study (2005). The variables measured were:

- distance from the city core (km);
- vehicular traffic (mean number of vehicles/minute at maximum vehicular traffic time: 10:00 a.m.–12:00 p.m., from 4 replicates) as an indirect measure of air pollution (McGeoch and Chown, 1997). Emissions from motor vehicles have been mentioned as the main factor

contributing to air pollution in urban centres in general (Klump et al., 2000) and in Córdoba in particular (Olcese and Toselli, 1997; Pignata, 1998);

- total species richness of plants with mined leaves as an estimator of the number of alternative hosts for parasitoids (total number of mined plants registered in the central block);
- average surface (m²) of exterior house gardens (measured in 25 m at random in each side of the central block);
- vegetation coverage;
- ground temperature.

The latter two variables were obtained from Landsat 5 Thematic Mapper (TM) path/row 229/82 satellite image (acquired on January 4th, 2005). This image was provided and georeferenced by the National Commission of Space Activities (CONAE). Then, a subset of Córdoba City was generated using ENVI 4.2 software (2004). To quantify vegetation coverage,

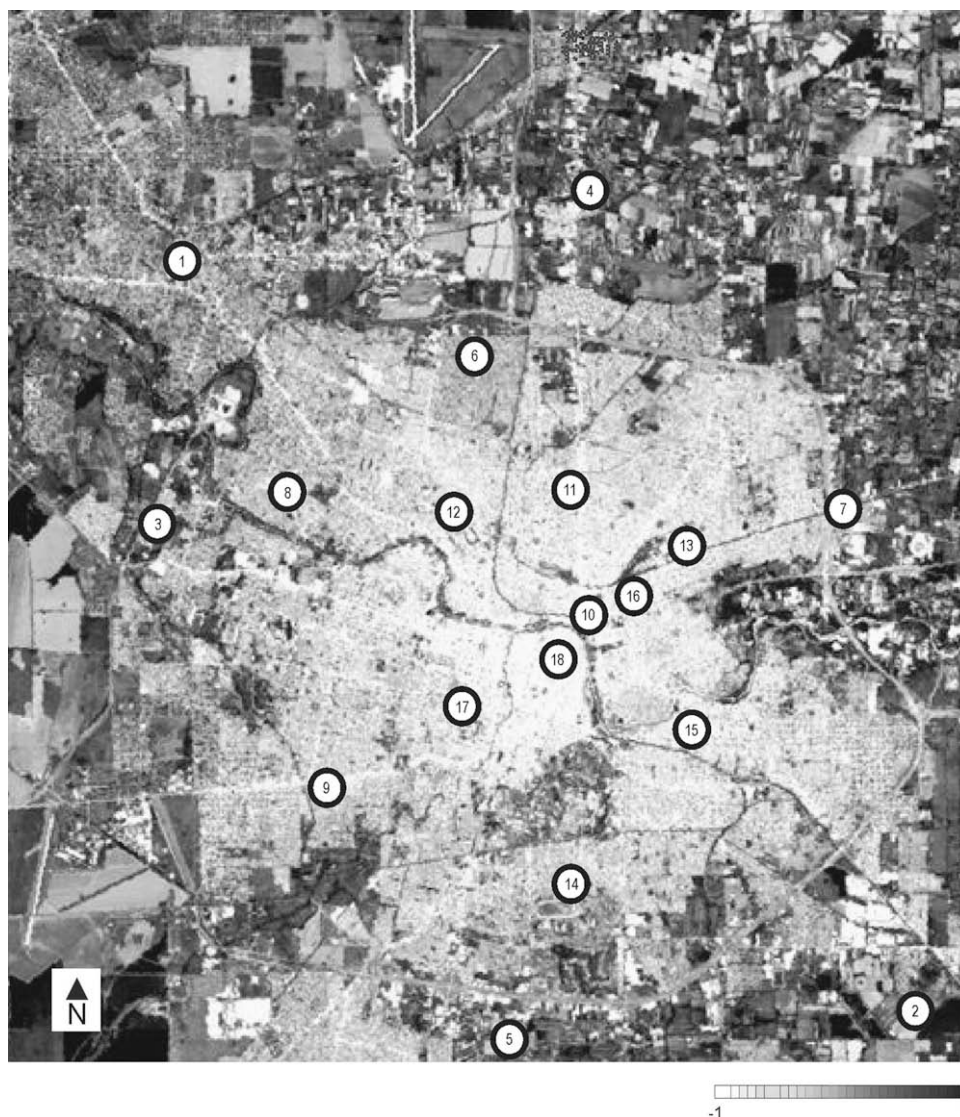


Fig. 1 – Location of sampling sites in Córdoba city (Argentina). The figure also shows vegetation cover measured through the Normalized Difference Vegetation Index (NDVI) from a subset of the Landsat 5 (January 4th 2005) image, path/row 229-82. NDVI ranges from -1 to 1 where negative values correspond to an absence of vegetation.

a Normalized Difference Vegetation Index (NDVI) was calculated from the image (Fig. 1) and regions of interest were created. Ground temperature was estimated through brightness temperature (Landsat images band 6) which gives an approximation of environmental temperature. In each sampling site we defined an area of 441 m² from the central block where the mean values of NDVI and brightness temperature were calculated. NDVI is an efficient index in differentiating vegetation and non-vegetation zones in urban areas (Myeong et al., 2001), considered as one of the basic indices to study the urban ecological environment (Yue and Tan, 2007) and found to be suitable for estimating the impervious surface composition (Matthias and Martin, 2004). NDVI is derived from the near-infrared band (band 4) and the red band (band 3) of the Landsat satellite image, and their values range from -1 to +1, where negative values correspond to an absence of vegetation (Jackson et al., 2004; Pettorelli et al., 2005; Estallo et al., 2008). Urban areas could be identified from the analyses of thermal infrared data acquired by a satellite, being the brightness and surface temperature measures frequently used to study the urban “heat island” effect (Gallo and Owen, 1999; Yue and Tan, 2007).

During January and February of 2005 and 2006, in each site, pavements of central blocks and adjacent ones (1350–3200 lineal meters per site) were checked in order to count the number of plants or plant groups of *C. erecta* separated by more than 2 m from each other (hereafter called “patches”). We restricted our sampling procedure only to pavements, given the logistical impossibility of sampling private gardens (Piel et al., 2005). All mined leaves in each plant patch were collected, placed in plastic bags, transported to the laboratory, and kept until flies and parasitoids emerged. Adult leafminers and parasitoids were stored in glass vials plugged with cotton-wool; once emergence had ceased they were counted and identified (Salvo and Valladares, 1998). Voucher specimens of parasitoid and leafminer species were deposited in the collection of the Entomology Department, Universidad Nacional de Córdoba, Argentina.

The following variables were estimated for each site: proportion of mined patches, abundance of the leafminer (estimated as total adults [flies plus parasitoids] bred), total parasitoid species richness, total parasitism rate of *L. commelinae* (sum of parasitoids over total number of adults bred) and parasitism rates caused by the most abundant parasitoid species (only for samples with $n > 100$). No significant

differences were found in the composition of parasitic species between the two years of sampling (data of log abundance of parasitoid species, MANOVA, Wilks' Lambda = 0.22, $F = 1.62$, $p = 0.23$), and so data from both years were pooled for statistical analysis. Only those sites occupied by the leafminer during both years of study ($n = 16$) were considered in the analyses.

The effect of urbanisation zone on the total number of *C. erecta* patches, proportion of occupied patches, leafminer abundance, parasitoid species richness, total parasitism rates and parasitism rates caused by the most abundant species were analyzed by simple and multiple regression analysis with stepwise selection. Before this, we assessed the degree of correlation between urbanisation variables which revealed that they were highly correlated (Table 1), except for the richness of mined plants. To reduce the number of variables, avoid multicollinearity problems (Philippi, 1993; Graham, 2003) and thus create an urbanisation index which takes into account most of the variation within the set of variables considered, we performed a Principal Component Analysis (PCA) on all variables except the richness of mined plants. The results of this analysis indicate that the first principal component (PC1) adequately represents the data (Eigenvalue > 1) and explains 74% of the total variation, with distance to urban core, NDVI and surface of gardens loading positively on this PC (all loadings ~0.45), and vehicular traffic and ground temperature loading negatively (both loadings ~-0.40). Therefore, high values of PC1 indicate low levels of urbanisation but to a better understanding we multiplied PC1 values to -1 and used them in posterior analysis. PC2 explained only 12% of the variability and had an eigenvalue of less than 1, indicating that it explained less of the variance than the original variables (Legendre and Legendre, 1998).

Simple linear regressions were performed between *C. erecta* and *L. commelinae* abundance and the scores of the PC1. Leafminer abundance was estimated by two variables: the percentage of mined patches and the abundance of leafminer larvae. Stepwise multiple linear regression analyses with forward selection were carried out to relate parasitoid species richness, total parasitism rates of *L. commelinae* and parasitism rates by most abundant species with PC1 and species richness of mined plants. Partial correlation was used to determine the importance of each variable. The effect of *L. commelinae* on parasitism was tested including leafminer abundance into the regression model. Possible sample size effects on the estimate of species richness were considered by constructing

Table 1 – Pearson correlation coefficients for urbanisation variables registered in different sites of Córdoba city ($n = 18$).

	Distance from city centre	Vehicular traffic (log)	Ground temperature	NDVI	Surface of external gradens	Species richness of mined plants (log)
Distance from city centre	–	–0.743***	–0.679**	0.675**	0.676**	0.168 ns
Vehicular traffic (log)		–	0.680**	–0.719***	–0.711***	–0.101 ns
Ground temperature			–	–0.722***	–0.709***	–0.230 ns
NDVI				–	0.797***	0.409 ns
Surface of external gradens					–	0.129 ns
Species richness of mined plants (log)						–

* $p \leq 0.05$; ** $p \leq 0.01$; *** $p \leq 0.001$; ns: not significant.

rarefaction curves for each site using the Ecosim program (Gotelli and Entsminger, 2001). This method enables the comparison of samples with different numbers of individuals (Magurran, 2004). The estimate of species richness in each site was calculated considering the least number of parasitoids obtained among all sites ($n = 53$). The species richness values obtained this way were then analyzed as previously was indicated for real values. When necessary to approach normality, leafminer abundance and species richness data were log-transformed, and parasitism rates were arc-sin square root transformed.

A cluster analysis based on Euclidean distances (furthest neighbour linkage method) was used on data of abundance of parasitoid species (log-transformed) to determine the degree of similarity between parasitic communities corresponding to the 16 sample sites. To discard a spatial effect on the parasitoid community composition (i.e. closer localities having a more similar parasitoid species composition irrespective of the urban zone), we performed a Mantel test, which examines the null hypothesis of non-concordance between two distance matrices (Legendre and Legendre, 1998). The strength of the relationship is measured by the standardized Mantel statistic r_M , which is identical to Pearson's correlation coefficient, ranging from -1 to 1 (Legendre and Legendre, 1998). We compared the Euclidean distances matrix based on parasitoid species abundance data (log $(x + 1)$ transformed) and a matrix based on geographical distance (km) between sampling sites. The Mantel's asymptotic approximation was the method used to detect significance in the pairwise relationships between matrices.

3. Results

From a total of 1,698 *C. erecta* patches registered over a 2-year sampling period, 1022 were occupied by *L. commelinae*. Although the total number of *C. erecta* patches was not related to PC1 ($F_{1,16} = 0.01$, $R^2 = 0.001$, $p = 0.91$), the proportion of mined patches ($F_{1,14} = 19.89$, $R^2 = 0.59$, $p = 0.0005$) and leafminer abundance ($F_{1,14} = 20.7$, $R^2 = 0.60$, $p = 0.0005$) were positively related to PC1 (sites with high values of PC1 have high urbanisation degree) (Fig. 2). The proportion of occupied patches was around 1.5 times higher in inner sites of the city, than in sites located in the periphery.

Of a total of 10,805 adults (flies and parasitoids) reared in laboratory, 51.92% were parasitoids belonging to four families of Hymenoptera Parasitica (Eulophidae, Pteromalidae, Figitidae and Braconidae). The 20 parasitic species detected caused different parasitism rates and showed a variable frequency of association with the leafminer, with *Chrysocharis flacilla* and *Diglyphus websteri* (both Eulophidae) being the most abundant species (Table 2). Total species richness associated to *L. commelinae* in each site varied between 7 and 17 species, being highest in one of the sites near from the city core (Site 16). The number of parasitoid species was significantly explained by the multiple regression model that included species richness of mined plants and PC1 scores ($F_{2,13} = 8.22$, $p = 0.005$, $R^2 = 0.49$). Parasitoid species richness increased with species richness of mined plants ($r = 0.69$, $p = 0.004$) and increased as urbanisation degree increased ($r = 0.62$, $p = 0.01$). However, after applying rarefaction, the term that only remained as significant was the

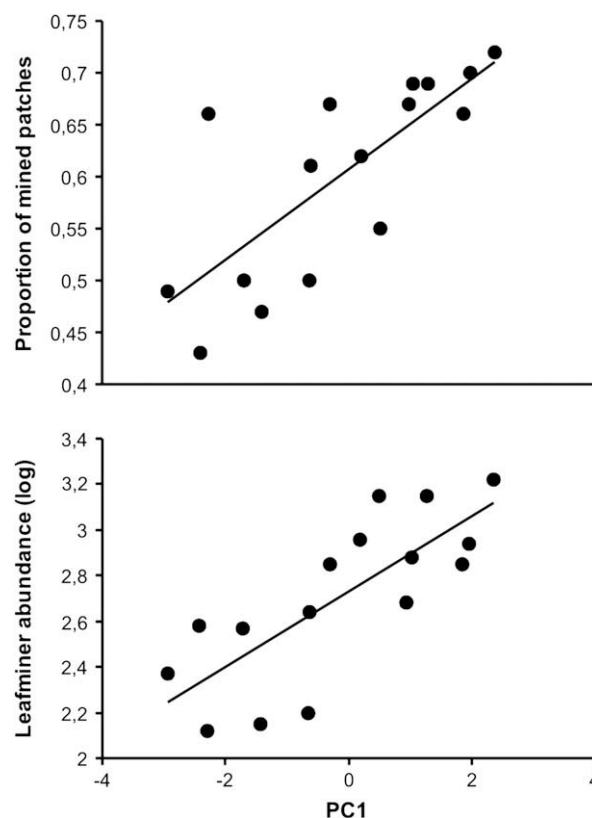


Fig. 2 – Relationship between the scores of the first principal component (PC1) as representation of urbanisation degree for Córdoba city (sites with high values of PC1 have high urbanisation degree) and (A) the proportion of mined patches of *C. erecta*, $y = 0.61 + 0.04x$ (B) the leafminer abundance, $y = 2.73 + 0.16x$.

species richness of mined plants ($F_{1,14} = 4.88$, $p = 0.04$, $R^2 = 0.26$) (Fig. 3).

Parasitism rates of *L. commelinae* varied between 38% and 71% throughout all sample sites and there were no significant effects of urbanisation degree (PC1), species richness of mined plants or leafminer abundance on them ($p > 0.05$). There was a positive relationship between parasitism rates due to *Chrysocharis caribea* and PC1 ($F_{1,14} = 8.97$, $p = 0.009$, $R^2 = 0.39$) (Fig. 4), whereas no significant relations were obtained for parasitism rates caused by the rest of the most abundant species ($p > 0.05$).

Parasitoid assemblages observed in sites less distant from the city centre were quite similar, differing from the species composition of parasitoid communities observed in sites with less urbanisation degree, more distant from the downtown area (Fig. 5). The Mantel test showed that parasitoid species composition did not depend on the geographical distances between sampling points ($r_M = 0.156$, $p = 0.26$).

4. Discussion

Our results showed that urbanisation had no detrimental effects on the species richness of parasitoid assemblages of

Table 2 – Parasitism rates by all species of parasitoids associated with *L. commelinae* at the sampling sites within the city of Córdoba (data from 2005 to 2006). Sites are listed according to their distance to the city core and species according to their abundance.

Site	S1	S2	S4	S5	S6	S7	S8	S9	S10	S11	S12	S13	S14	S15	S16	S17
Distance from city core (km)	10.4	8.26	7.05	6.69	6.53	6.49	6.02	4.46	3.96	3.78	3.77	3.59	3.48	2.89	2.25	1.65
<i>C. flacilla</i> (Walker)	5.57	2.07	14.77	27.27	7.88	7.14	2.55	13.73	14.43	13.45	5.43	11.22	9.23	18.34	8.19	9.66
<i>Diglyphus websteri</i> (Crawford)	0.53	5.06	2.11	8.33	3.53		15.92	5.32	7.07	7.46	11.64	7.85	8.81	9.79	4.39	3.15
<i>Chrysonotomyia</i> sp. B	0.27	21.15	5.49	5.30	1.36	2.86	3.82	8.64	3.00	11.58	5.00	12.76	4.40	2.05	3.97	13.87
<i>Chrysocharis vonones</i> (Walker)	5.04	2.53	5.06	4.55	6.52	10.71	5.73	6.31	7.79	6.79	4.43	5.47	4.40	6.72	4.76	3.57
<i>Chrysocharis</i> Forster sp. A	7.43	0.46	5.91	9.85	5.98	6.43	8.92	2.77	10.07	5.46	2.93	3.65	10.09	3.87	4.64	7.14
<i>Chrysocharis caribea</i> (Boucek)	1.86	1.38	2.53		1.63	7.14		3.43	7.57	2.13	5.21	7.29	1.85	10.71	5.48	7.77
<i>Agrostocynips enneatoma</i> (Diaz)	9.02	10.80	10.97		12.23	2.86	14.65	8.53	2.07	1.46	6.50	1.40	5.26	3.42	0.36	2.31
<i>Chrysonotomyia</i> Ashmead sp. A		1.61	2.11		0.54		1.27	1.11	1.36	2.13	3.79	2.38	1.42	0.11	2.05	4.83
<i>Thinodytes</i> Graham sp. A	1.06	1.15		0.76	0.27		3.82	0.78	1.71	0.80	2.00	0.70	4.40	1.03	0.54	1.26
<i>Elachertini</i> sp. A		2.99				0.71	11.46	0.33	1.86	0.53	0.79	0.28	1.28		0.06	2.10
<i>Halticoptera helioponi</i> (De Santis)	0.53	0.69	0.42	0.76	0.54				2.07	1.73	1.00	0.42			0.12	
<i>Phaedrotoma luteoclypealis</i> (Van Achterberg and Salvo)	1.06		1.69	2.27	0.82		0.64	0.55					3.13		0.12	
<i>Herbertia ca. brasiliensis</i> Ashmead	2.65		2.11	2.27	0.54											
<i>Elachertini</i> sp. B							1.27		0.43		0.84	0.28			0.12	
<i>Chrysonotomyia</i> sp. C									0.36			0.28			0.06	
<i>Eulophinae</i> sp.	0.27		0.42												0.12	
<i>Chrysonotomyia</i> sp. D	0.27											0.14			0.06	0.21
<i>Chrysocharis</i> sp. B												0.14				0.21
<i>Proacrias thysanoides</i> (De Santis)															0.06	
<i>Proacrias xenodice</i> (Walter)									0.07							

L. commelinae. The estimated species richness of parasitoids was almost as high in more urbanised sites of the city as in less ones. This disagrees with our predictions and differs from earlier results obtained for other insects in general (McGeoch and Chown, 1997; McIntyre and Hostetler, 2001; Weller and Ganzhorn, 2004; Ishitani et al., 2003; Yamaguchi, 2004) and for parasitoids in particular (Sawoniewicz, 1986; Denys and Schmidt, 1998). However, not all the organisms studied showed a negative response to urbanisation, as demonstrated previously for other arthropods and plants (Alarukka et al., 2002; Deichsel, 2006; Elek and Lövei, 2007; McKinney, 2008).

Other variables, independent of the level of urbanisation might be affecting the species richness of parasitoids of

L. commelinae. In this study, we found that the number of parasitoid species was positively related with the species richness of mined plants. This tendency is in agreement with that expected for generalist parasitoids, which are attracted to plants or habitats with larger numbers of potential host species (Askew, 1994). In urban areas, ornamental plants are common hosts of *Liriomyza huidobrensis* (Valladares and Salvo, 1999), a related species of *L. commelinae*, which can develop high densities in those environments (Salvo et al., 2005) and can constitute a potential source of parasitoids for *L. commelinae* because they share at least thirteen parasitoid species (Fenoglio and Salvo, unpublished).

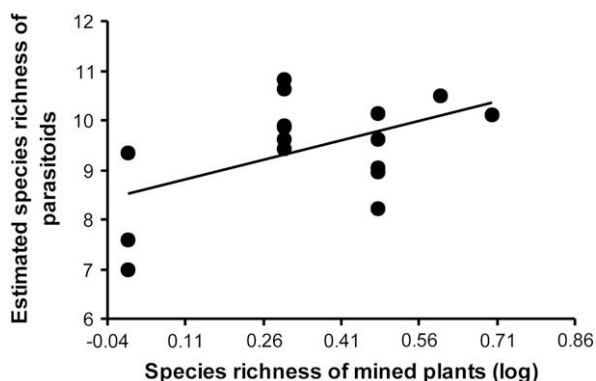


Fig. 3 – Relationship between parasitoid species richness estimated by rarefaction and the number of species of mined plants in different sites of Córdoba city, $y = 8.53 - 2.59x$.

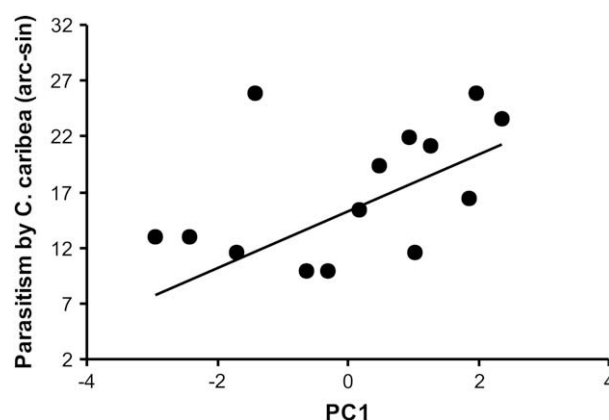


Fig. 4 – Relationship between parasitism rates by *Chrysocharis caribea* and scores of the first principal component (PC1) as representation of urbanisation degree for Córdoba city (sites with high values of PC1 have high urbanisation degree), $y = 10.75 + 2.09x$.

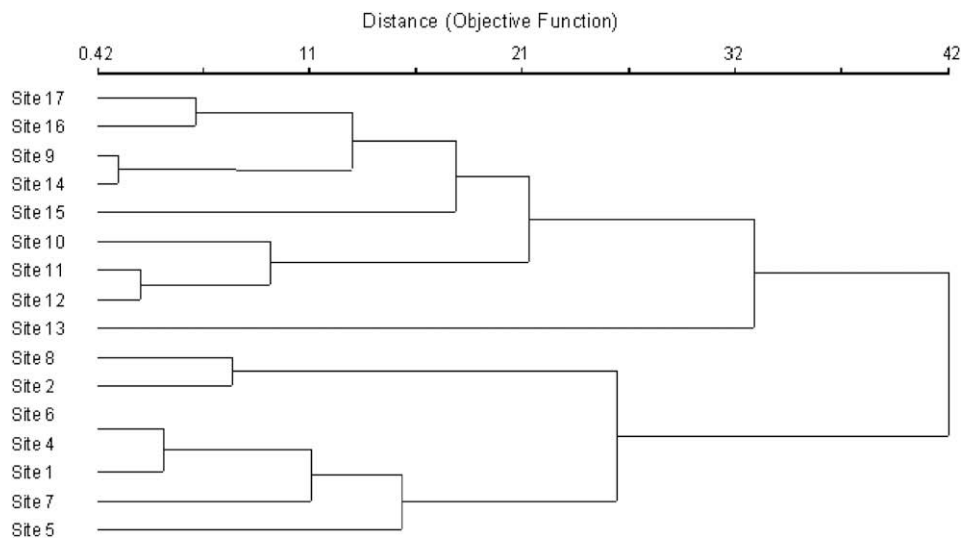


Fig. 5 – Cluster analysis of the parasitoid assemblages of *L. commelinae* corresponding to different sites of Córdoba city (distance measure: Euclidean distance, linkage method: Further neighbour). Site 1 – far distant from the city centre to Site 17 – near distant from the city centre.

Parasitoid species composition differed mainly between more urbanised sites and peripheral ones. One of the reasons for these differences might be the dominance of some species such as *D. websteri* and *C. caribea* in the parasitic complexes associated with the leafminer in sites more urbanised, while in peripheral areas of the city these species were less represented. *Herbertia brasiliensis*, the only specialist parasitoid attacking *L. commelinae*, was present in very low numbers and was absent in sites located in the urban core. It is possible that populations of this species undergo local extinctions due to a combination of extreme sensitivity to environmental disturbances and chronically low abundance (Rebele, 1994). The similarity of parasitoid communities associated with *L. commelinae* in different sites did not depend on the geographic distance between points, marking the importance of urbanisation for structuring parasitoid communities.

Even though urbanisation did not tend to reduce parasitoid diversity, relative abundance of species differed among zones, with certain species appearing to do well in disturbed and human dominated environments. *C. flacilla* was the only species associated with the leafminer in all sites. This species has been recorded as parasitizing agromyzid leafminers in several types of habitats, always with a high frequency of association (Salvo et al., 2005).

Contrary to our predictions, we did not find a detrimental effect of urbanisation on total parasitism rates of *L. commelinae*. This result agrees with that observed for parasitoids of *Eriocrania* miners in polluted areas in Finland (Koricheva, 1994) but differs from other studies (Denys and Schmidt, 1998; Zvereva and Kozlov, 2006). The lack of relation between total parasitism rates and urbanisation degree suggests that the parasitoid community altogether has the potential to impact leafminer populations irrespective of the city alterations. Evidence in the literature suggests that individual species may respond differently to urbanisation (Connor et al., 2002; Alar-uikka, 2003). In the system studied here, only one parasitoid

species, *C. caribea*, caused higher mortality rates to the leafminer in more urbanised areas, indicating that this species may be capable of tolerating high levels of anthropic disturbance. The rest of the species were not affected by the degree of urbanisation, a situation that is clearly reflected in total parasitism rates.

Pollution can also induce changes in host population densities and this could indirectly affect parasitoids (Koricheva, 1994). Although we detected a significant increase in patch occupancy and leafminer abundance in sites located near the urban core, parasitism rates were not related to urbanisation degree, indicating that neither urbanisation per se nor host density affected parasitism activity. One probable reason explaining the increase in *L. commelinae* abundance in more urbanised sites of the city is that plants located those areas would be more stressed by air pollution which would increase their susceptibility to the attack by the leafminer (McIntyre, 2000; Connor et al., 2002). Also, leafminer insects may be less sensitive to pollution than their plant hosts, as they are better protected against direct contamination by developing within leaf parenchyma (Zvereva and Kozlov, 2006). Nevertheless, we cannot support this hypothesis, because we have no measures of the chemical and nutritional state of the plants across the zones. Another possible explanation for the higher abundance of the leafminer in the urban core would be the higher abundance of the host plant in more polluted habitats, but our results also do not support this idea. Although we did not find a positive effect of urbanisation on plant patch abundance it seems that this ruderal species may be an “urban exploiter” because it can tolerate high levels of disturbance as it grows in and around pavements (McKinney, 2002).

In conclusion, the urbanisation level affected neither the species richness of parasitoid complexes of *L. commelinae* nor the parasitism rates they suffered. Urban pressure did affect parasitoid communities, with some species being more

tolerant to human disturbances than others. Changes in parasitoid communities along the urbanisation gradient were noticed in cluster analysis but they were not strong enough to be detected in the abundance/parasitism of particular species. In this study all the characteristics that were considered as representative of the degree of urbanisation, generally act at the site or landscape level. Possible local effects of features of plant patches, such as size or isolation, remain to be evaluated in order to obtain information of the impact of habitat fragmentation on parasitoid community and leafminer parasitism rates.

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