Beneficial insects in urban environments: Parasitism rates increase in large and less isolated plant patches via enhanced parasitoid species richness

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A B S T R A C T

In urban habitats, plant resources for herbivorous insects and their parasitoids are spatially subdivided and embedded in a matrix of built environment. These conditions could affect insect colonization and persistence, leading to altered trophic interactions. Here we tested whether spatial attributes (size and isolation) of patches of an urban spontaneous plant (*Commelina erecta*) affect the parasitism rates on a specialist leaf miner (*Liriomyza commelinae*) from its diverse parasitoid assemblage. To achieve this goal we performed an observational study in Córdoba city (Argentina), by sampling 893 naturally established *C. erecta* patches on 18 neighborhoods. Overall parasitism rates of the leaf miner augmented at increasing patch size and decreasing patch isolation, and those effects were mediated by changes in parasitoid richness. Leaf miners in larger or less isolated plant patches supported more parasitoid species which, independently of host abundance, led to higher parasitism rates. Individual parasitoid species showed different responses to patch attributes, with some species being more efficient in large or less isolated plant patches whereas parasitism rates of other species were independent of patch attributes. Our results highlight the importance of spatial availability of urban plant resources for parasitoid diversity and for the key process of biological control in which they are involved. Understanding how natural enemies respond to the patchiness of herbivore resources in the urban context could contribute to the development of strategies to preserve beneficial insects in these highly modified landscapes.

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1. Introduction

Urban areas are modified and complex landscapes, characterized by high levels of disturbance and fragmentation (Grimm et al., 2008; Pickett et al., 2001). Concrete buildings, asphalt paving and vegetation reduced to patches scattered in the landscape are distinctive features of urban areas (Niemelä, 1999; Rebele, 1994). Although the urbanization process usually has negative effects on the native biota (Czech et al., 2000; McKinney, 2008), green areas like gardens and wastelands may sustain a significant diversity of beneficial arthropods allowing for the maintenance of ecosystems services such as pollination, seed dispersal and insect pest regulation (McDonald and Marcotullio, 2011; Robinson and Lundholm, 2012).

Parasitoid insects are important biological control agents of herbivorous populations and have been found sensitive to urbanization (reviewed in Fenoglio and Salvo (2010)) at different spatial scales (Bennett and Gratton, 2012). Previous works have explored the influence of urban patches of vegetation on parasitoid communities (Christie and Hochuli, 2009; Christie et al., 2010; Gibb and Hochuli, 2006) applying a landscape perspective. For specialist herbivores and their parasitoids, the local habitat is essentially represented by the host plants, which in urban areas could resemble green islands within an inhospitable matrix of building environment. The spatial subdivision of the plant resource becomes then crucial to insect colonization and persistence (Hanski, 1999; Niemelä, 1999), with patch attributes such as size and isolation being particularly relevant. Although there is evidence of spatial attributes of local patches affecting host–parasitoid interactions in natural or agricultural environments (Cronin, 2003, 2004; Doak, 2000; Elzinga et al., 2005; Kruess and Tscharntke, 2000; Woodcock and Vanbergen, 2008), it remains unclear how those variables act in an urban context (Denys and Schmidt, 1998).

On urban plant patches, populations of insect herbivores and their parasitoid assemblages might function like metacommunities (Leibold et al., 2004; Swan et al., 2011) because insect dispersal, the key process linking local communities, can be low due to the

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presence of physical barriers but still large enough for long-term persistence. In this context, colonization, extinction and population abundance are expected to be affected by the area and isolation of patches (Hanski, 1999). Moreover, the differential susceptibility of herbivores and parasitoids to habitat patchiness could lead to altered trophic interactions (Holt et al., 1999; Kruess and Tscharntke, 1994).

In addition to metacommunity processes, spatial characteristics of plant patches can influence parasitoids through changes in the abundance or density of herbivore insects (Bach, 1988; Pareja et al., 2008). Direct effects of patch attributes on parasitism rates can also occur independently of the actual amount of host resources, for instance via differences in individual ability to find and/or exploit hosts. According to Sheehan and Shelton (1989), who proposed an extension of the resource concentration hypothesis (Root, 1973) to the third trophic level, parasitism rates would increase with increasing plant patch size since parasitoids may be more likely to find or less likely to leave large plant patches (see Esch et al., 2005; Kruess and Tscharntke, 2000; Pareja et al., 2008). In fact, larger green patches in a concrete urban matrix could constitute an easier target to detect for visually searching parasitoids (Chittka, 1996). At the same time, plant volatiles and herbivore-derived chemicals may play an important role in parasitoid foraging (Turlings and Wäckers, 2004). Olfactory and visual signals probably increase with increasing patch size, thus attracting more parasitoids.

On the other hand, parasitoids may be unable to track hosts in isolated patches because of restricted movement within the urban matrix (Denys and Schmidt, 1998; Raupp et al., 2010). Species susceptibility to habitat isolation not only depends on dispersal ability (Tscharntke and Brandl, 2004) or distance between plant patches, but also on the nature of the surrounding matrix (Ricketts, 2001). The urban matrix is typically hostile to less mobile species and the effects of isolation might therefore be higher than in simpler landscapes (Gilbert, 1989). Negative relationships between patch isolation and parasitism rates have been previously reported for other environments (Faeth and Simberloff, 1981; Krewenka et al., 2011; Kruess and Tscharntke, 1994, 2000) as well as for urban habitats (Denys and Schmidt, 1998; Peralta et al., 2011).

The majority of studies examining spatial effects of plant patches on host–parasitoid interactions have focused on pair-wise species systems (Esch et al., 2005; Pareja et al., 2008; Vanbergen et al., 2007). For herbivore insects supporting complex parasitoid assemblages, the effects of plant patch features on parasitism rates could be mediated by changes in parasitoid diversity, which could ultimately drive overall parasitism rates (Fenoglio et al., 2012; Tylianakis et al., 2006).

This study aims to understand parasitoid community responses to plant patchiness in urban environments. Particularly, we investigated the role of plant spatial availability on parasitism rates caused by a parasitoid assemblage on a specialist leaf miner, by performing an observational study in a populated city of Central Argentina. We previously found that neither parasitoid species richness nor parasitism rates were influenced by urbanization degree, leaf miner abundance or host diversity at landscape scale (Fenoglio et al., 2009). However at local scale, plant patch size positively affected the number of parasitoid species associated with the leaf miner (Fenoglio et al., 2010).

Here we specifically ask (i) whether size and isolation of urban plant patches affect overall parasitism rates of the leaf miner and, if so, (ii) whether the effects are mediated by the changes in parasitoid richness previously reported or just by differences in host abundance. We also look at parasitism rates caused by individual species, in an attempt to further understand the effects of spatial attributes of plant patches on this valuable ecosystem service.

2. Material and methods

2.1. Study system

The study system comprised the leaf miner *Liriomyza commelinae* (Frost, 1931) (Diptera: Agromyzidae), its parasitoid community and its host plant *Commelina erecta* L. (Commelinaceae). The neotropical *L. commelinae* has been recorded feeding on the genera *Commelina* and *Tradescantia* (Smith, 1987; Stegmaier, 1966), but appears to be specialized on *C. erecta* in central Argentina (Valladares, 1984) where mines are abundant in the summer, especially during February and March. The larvae burrow into the leaf parenchyma forming distinct serpentine mines (Fig. 1A) visible on the upper surface of the leaves, within which they pupate (Smith, 1987).

Parasitoids are an important source of mortality in this species, with 25 native Hymenopteran species (Fig. 1B) causing on average 50% mortality, being recorded in central Argentina (Fenoglio and Salvo, 2009). Most of these parasitoids are solitary and polyphagous, feeding on various leaf-miner species in the study region (Fenoglio and Salvo, 2009; Salvo and Valladares, 1999). To our knowledge, there are no records of hyperparasitoids associated with *L. commelinae*.

The host plant *C. erecta* is an herbaceous plant, common and widely distributed in the north and center of Argentina, frequently appearing as spontaneous urban vegetation. Leaves and flowers are available from November to April, but rhizomes are present all year (Sérsic et al., 2006). In urban areas, *C. erecta* typically shows a highly patchy distribution (Fig. 1C), growing in plant groups of variable size and distance from other patches (Fenoglio et al., 2010).

2.2. Survey

The research was conducted within Córdoba city, which is located in central Argentina (31°20′S, 64°10′W, elevation 440 m) and has a population of 1,329,604 inhabitants (Indec, 2010). The climate is subhumid, with an average annual precipitation of 790 mm, concentrated principally in summer (October–March). The city has an irregular topography and it is crossed by the Suquía River from west to east. Córdoba covers an area of 576 km² from which 39.75% correspond to urban development, 28.48% to rural area, 21.3% to industrial area and 10.5% to other uses (Observatorio Urbano Córdoba, 2007). Eighteen sites located in different neighborhoods spread across the city (Fig. 1D) were sampled during January and February 2006, corresponding to the peak period of *L. commelinae* activity. At each site, the pavements of one central and 8 adjacent urban blocks (about 2100 lineal meters per site, standard Córdoba blocks being 100 m × 100 m) were checked in order to find *C. erecta* plant patches. The sampling procedure was restricted only to pavements, as is commonly done in this type of studies (Piel et al., 2005), given the logistical problems of sampling private gardens and also due to pavements being potential corridors for *L. commelinae* and its parasitoids (Peralta et al., 2011).

The size of plant patches was calculated as covered area (cm²) from their length–width measurements. Only mined plant patches were considered in the survey. Isolation, the distance (m) to the closest patch, was measured at field using a measuring tape since most of patches (95%) were located nearby (<50 m). 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 leaf miners and parasitoids were stored in glass vials plugged with cotton wool; once emergence had ceased, they were counted and identified (Salvo and Valladares, 1999). Voucher specimens of parasitoid and leaf miner species were deposited in the collection of the Entomology Department, Universidad Nacional de Córdoba, Argentina. The following variables were
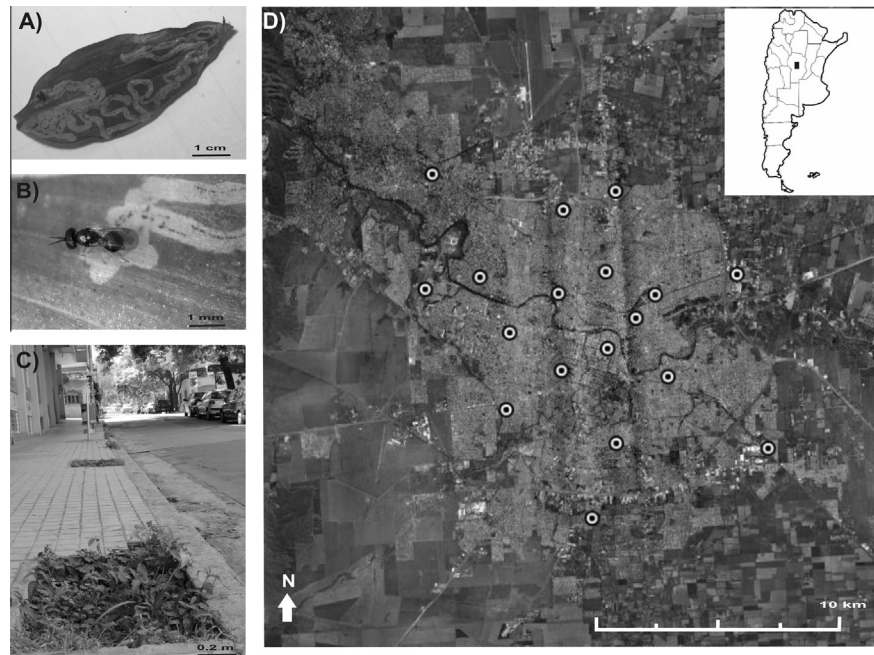
Fig. 1. (A) Leaf of *Commelina erecta* mined *by Liriomyza commelinae*, (B) *Chrysocharis flacilla* (Hymenoptera: Chalcidoidea), one of the parasitoid species of *L. commelinae*, (C) *Commelina erecta* patch in the urban habitat and (D) Location of sampling sites (*n* = 18) in the city of Córdoba (Argentina).

estimated for each plant patch: abundance of leaf miner larvae [estimated from total adults (flies plus parasitoids) reared] and density (leaf miner abundance divided by total number of leaves), parasitoid species richness, overall parasitism rates (the total number of parasitoids divided by leaf miner abundance) and parasitism rates caused by individual species (the total number of parasitoids of species *i* divided by leaf miner abundance).

2.3. Statistical analysis

At increasing levels of urbanization, patches of vegetation could become more isolated by a higher proportion of human-made barriers (Denys and Schmidt, 1998). Thus to discard potential confounding effects between these variables we performed a correlation between urbanization degree (see Fenoglio et al., 2009) and plant patch attributes (mean size or isolation value per site). We also checked the correlation between patch size and isolation.

2.3.1. Overall effects of patch size and isolation

Data from the observational survey were analyzed using generalized linear mixed models (GLMMs) (Bolker et al., 2009) with patch size, isolation [both transformed to natural logarithm, $\log(x)$] and the interaction between them as fixed effects, whereas site was included as random factor. The inclusion of site as random factor allowed contemplating the likely spatial correlation between patches (Faraway, 2006).

The dependent variable (proportion of total parasitized leaf miners in a patch) was modeled assuming binomial probability distribution and log link function. The data set included all mined patches where ⩾5 adult individuals (flies + parasitoids) were obtained (*n* = 370). Patches with fewer individuals were excluded in order to reduce errors attributable to small samples in parasitism estimation. The same analysis was repeated for the parasitism rates caused by individual parasitoid species, which was restricted to those species that were present in at least five patches.

The GLMMs were fitted using lme4 package (Bates and Maechler, 2009) from R 2.11.1 software (R Development Core Team, 2010). The Lapplace method was applied and in all cases over dispersion was checked. The significance of predictor variables was determined by likelihood ratio tests (LRTs) in GLMMs (Bolker et al., 2009). All interaction terms between patch size and isolation were non-significant thus they were removed from the final models. Plots of partial effects corresponding to GLMM for total parasitism rates were built using the function plotLMER.fnc from language R package.

2.3.2. Direct and indirect effects of patch size and isolation

To assess whether plant patch effects on parasitism were direct or mediated through changes in leaf miner abundance/density or parasitoid species richness we used a structural equation model (SEM) (Shipley, 2000). The initial model included direct effects of patch size and isolation (log transformed) on parasitism, as well as indirect effects through leaf miner abundance (log transformed) and the species richness of parasitoids (Fig. 2A). As the number of parasitoid species could depend on host abundance (Fenoglio et al., 2010) we included a path between these two variables. Patch structure effects on leaf miner density were assessed via changes in host abundance to avoid redundant paths. The goodness-of-fit of the model was evaluated using several statistics such as the χ^2 test, Goodness-of-Fit Index (GFI), the Tucker–Lewis Non-Normed Fit Index (NNFI) and the Comparative Fit Index (CFI) (Arbuckle, 2003). Values of indexes close to one indicate a very good fit. The significance of each path was evaluated with a *z* test (Arbuckle, 2003; Zar, 1999). The model was simplified by removing non-significant paths and the significance of the difference in fit was tested with a simple χ^2 test. Patches with no reared parasitoids were excluded from the analysis since those data could drive by default a positive relationship between parasitism and parasitoid richness (*n* = 330). Site was not included in the model since results of GLMM for overall parasitism rates indicated lack of effects for the random factor (see Results). The Browne's Asymptotically distribution free criterion was used as estimation procedure since
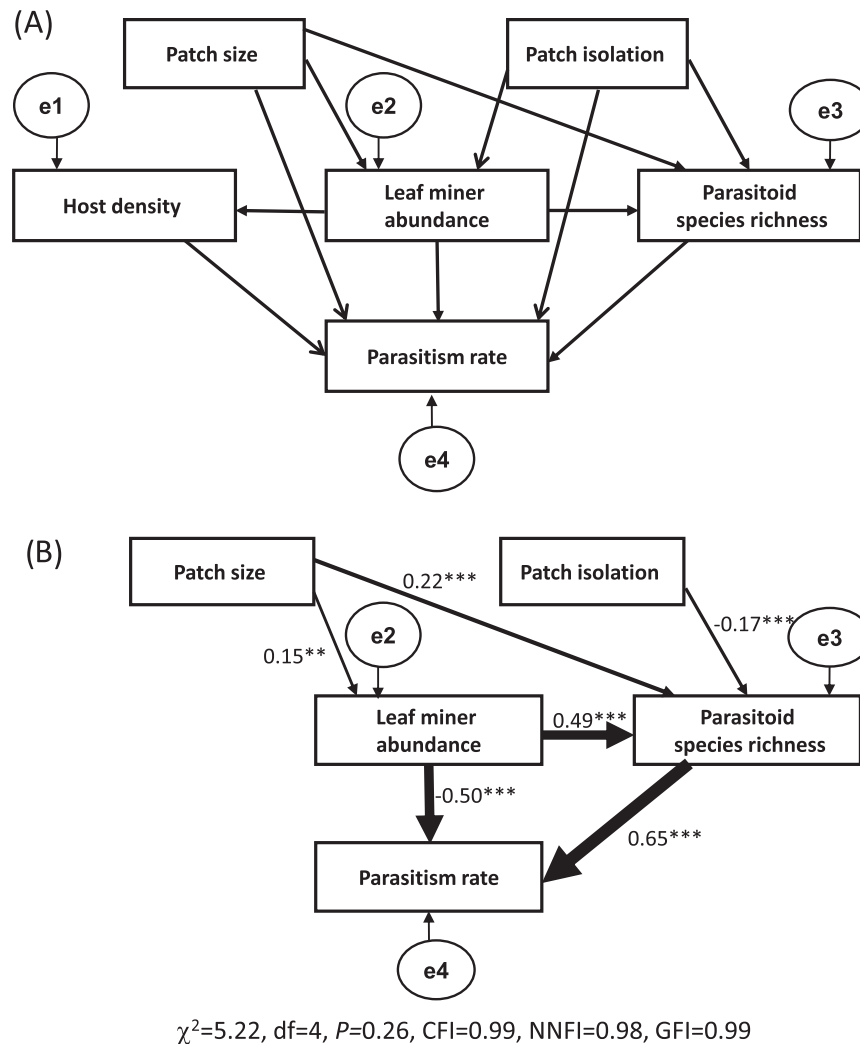
Fig. 2. (A) Hypothesized and (B) final structural equation model examining the direct and indirect effects of urban *C. erecta* patch size and isolation (log transformed) on overall parasitism rates of *L. commelinae*. Standardized path coefficients are given next to path arrows with significances indicated by **p < 0.05, ***p < 0.001. Arrow widths are proportional to path coefficients. Residual error variation associated with the dependent variables is indicated by "e". The statistics provided correspond to the final, most parsimonious model.

normality of data was not achieved (Browne, 1984). SEM analyses were carried out using AMOS 5.0 software.

In addition, to fully disentangle effects of parasitoid species richness from host abundance on parasitism rates, we performed a similar GLMM, but using the species richness estimated by rarefaction as predictor variable. This method enables the comparison of samples with different numbers of individuals (Magurran, 2004). The estimate of species richness in each patch was calculated from the lowest number of leaf miners among all patches ($n = 5$) by using the rarefy function of vegan package from R software.

3. Results

In total, 893 *C. erecta* patches were surveyed in the city, from which 76% were occupied by the leaf miner *L. commelinae*. Parasitoids were recorded in approximately 72% of these mined patches. Out of 7041 adults (flies and parasitoids) reared from those patches, 51.51% were parasitoids representing 20 species of Hymenoptera. The identity of these species has been previously reported (Fenoglio et al., 2010). The size of occupied patches varied from 0.0025 to 14 m^2 and their isolation from 0.5 to 120 m. These

variables were uncorrelated between them ($p = 0.60$) and were also independent of urbanization degree ($p = 0.27$ for patch size and $p = 0.76$ for patch isolation).

3.1. Overall effects of patch size and isolation

Overall parasitism rates per plant patch (mean = 52.86, SE = 1.61, $n = 370$) increased with increasing patch size ($\chi^2 = 113.9$, df = 1,366; $p < 0.0001$, $b = 0.21$) and decreasing patch isolation ($\chi^2 = 42.20$, df = 1,366; $p < 0.0001$, $b = -0.19$) (Fig. 3). Included as a random factor, site accounted for near zero variance (0.28 ± 0.53) thus having negligible effect on overall parasitism rates.

Parasitism rates by individual parasitoid species showed different relations to plant patch spatial features in the urban environment (Table 1). About half of the species responded to either size or isolation of plant patches or to both traits. Three out of the four most abundant parasitoid species performed better at increasing patch size, whereas three other species showed the opposite pattern. On the other hand, four of seven species responding to patch isolation tended to show higher parasitism rates on less isolated patches.
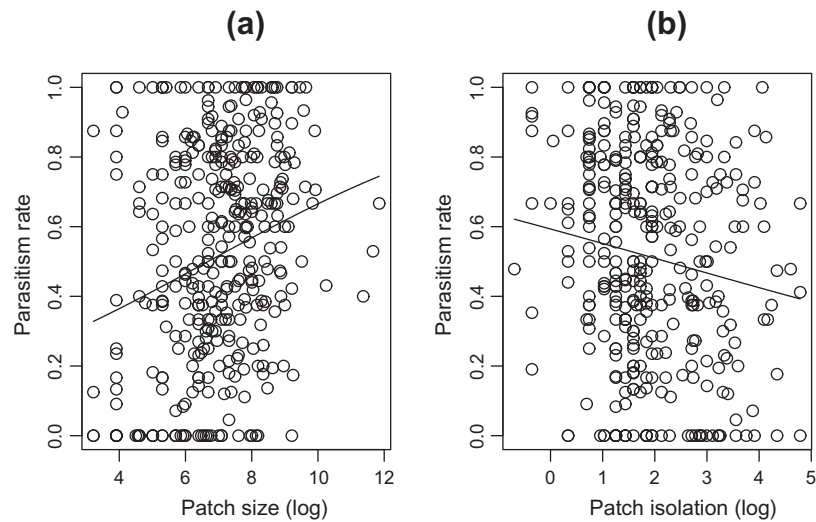
Fig. 3. The relationship between overall parasitism rates of *L. commelinae* and (a) plant patch size (χ^2 = 113.9, df = 1,366; p < 0.0001, b = 0.21), (b) plant patch isolation (χ^2 = 42.20, df = 1,366; p < 0.0001, b = −0.19). Fitted lines corresponded to partial effects of the GLMM where a binomial error distribution with logit link function was used.

Table 1
Parasitism rates of *L. commelinae* by individual parasitoid species as a function of plant patch attributes (n = 328 patches). The table shows the mean parasitism per patch and the results of generalized linear mixed models (modeled with binomial error and logit link function) evaluated through likelihood ratio test. Estimated parameters (b) are on the logit scale. Values in bold show significant relationships (p < 0.05).

Parasitoid species	Parasitism (mean ± se)	Random effect (variance ± sd)	Patch size (log$_e$)			Patch isolation (log$_e$)		
			b	χ^2	p	b	χ^2	p
Chrysocharis flacilla	10.41 ± 0.91	0.64 ± 0.80	0.06	3.72	**0.05**	0.08	4.01	**0.04**
Chrysonotomyia sp. B	7.88 ± 0.79	0.98 ± 0.99	–	0.90	0.34	−0.12	5.28	**0.02**
Chrysocharis vonones	7.82 ± 0.75	0.13 ± 0.35	0.25	37.62	**<0.0001**	–	0.32	0.57
Chrysocharis sp. A	6.87 ± 0.64	0.43 ± 0.66	0.15	13.72	**0.0002**	−0.12	6.11	**0.01**
Diglyphus websteri	6.22 ± 0.79	0.70 ± 0.83	–	0.56	0.45	–	1.32	0.25
Agrostocynips enneatoma	5.58 ± 0.78	0.73 ± 0.86	−0.14	6.39	**0.01**	0.11	2.95	0.08
Chrysocharis caribea	3.96 ± 0.53	0.95 ± 0.97	−0.09	4.09	0.04	−0.46	61.23	**<0.0001**
Chrysonotomyia sp. A	1.70 ± 0.28	0.27 ± 0.52	–	0.55	0.45	–	0.12	0.72
Halticoptera helioponi	0.73 ± 0.35	6.71 ± 2.59	−0.23	4.20	**0.04**	0.36	5.92	0.01
Elachertini sp. B	0.72 ± 0.35	6.62 ± 2.57	–	1.94	0.16	−0.57	4.06	0.04
Elachertini sp. A	0.04 ± 0.01	3.44 ± 1.85	–	0.75	0.38	–	1.97	0.15

3.2. Direct and indirect effects of patch size and isolation

The best model resulting from our SEM analyses indicated that the effects of both patch size and isolation on parasitism were indirect, mediated by changes in host abundance and parasitoid species richness (Fig. 2B). On one hand, patch size had an indirect effect via increased number of leaf miners which negatively affected parasitism rates. On the other hand, larger or less isolated plant patches had an indirect positive effect on parasitism rates through increments in parasitoid richness, which also depends on host availability (Fig. 2B).

Host abundance had a direct negative effect on parasitism but this was partially offset by an indirect positive effect of host abundance on parasitoid richness which in turn strongly increased parasitism rates. Overall effects of patch size on parasitism rates were positive highlighting the importance of parasitoid richness as indirect pathway. Indeed, we found that parasitoid richness estimated by rarefaction was significantly correlated to parasitism rates (χ^2 = 69.57; df = 1,323; p < 0.0001; b = 0.73; Fig. 4), indicating that the number of parasitoid species, independently of host abundance, influenced the proportion of parasitized larvae.

4. Discussion

Our results showed that spatial attributes of plant patches in urban environments indirectly affected overall parasitism rates of the
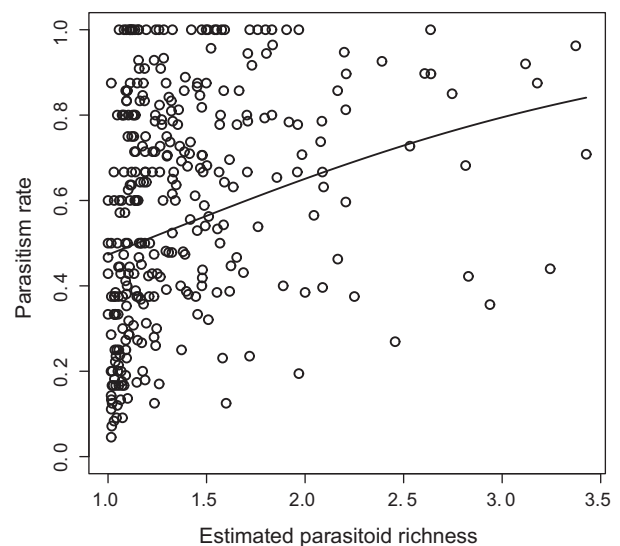

Fig. 4. The relationship between overall parasitism rates of *L. commelinae* and parasitoid richness estimated by rarefaction (GLMM: χ^2 = 69.57; df = 1,323; p < 0.0001; b = 0.73).

leaf miner *L. commelinae* mainly through changes in parasitoid species richness. Patch size had a net positive effect on parasitism rates of *L. commelinae* as a result of interplay between different indirect pathways involving host abundance and parasitoid species richness. Meanwhile, patch isolation negatively influenced parasitism rates through reductions in the number of parasitoid species. Furthermore, parasitism caused by particular parasitoid species also varied with spatial plant patch features.

Variations in parasitism rates of *L. commelinae* with size of plant patches partially depended on the availability of hosts. As expected from the resource concentration hypothesis (Root, 1973), larger patches of *C. erecta* sustained higher host abundance. This increment in host abundance brought about, on one side, a dilution effect on parasitism rates but on the other side, an increase in the number of parasitoid species which led to a strong positive effect in the proportion of parasitized larvae. Overall, richness mediated indirect effects of patch size on parasitism seem to be stronger than indirect effects mediated by host abundance. In fact, parasitoid richness effects on parasitism remained highly significant after removing the influence of host abundance.

A combination of regional processes linked to metacommunity dynamics, and local processes associated to parasitoid behavior could underlie the observed relationship between plant patch structure and the parasitism rates inflicted on the leaf miner by its parasitoid assemblage. Given the relatively ephemeral nature of *C. erecta* patches, processes such as local extinction and immigration (Leibold et al., 2004) could have been involved in determining the number of parasitoid species in relation to patch structure, finally affecting the host-parasitoid trophic interaction. At the same time, plant patch features could also be affecting parasitoid foraging behavior. In some situations, it might be easier for parasitoids to find the plant resource of the host than the host itself (Cappuccino, 1992; Sheehan and Shelton, 1989), which could lead to an accumulation of parasitoids and consequently higher parasitism rates on large and close plant patches.

In particular, olfactory and visual cues related to the host microhabitat could be playing a significant role for parasitoids during host location. In the urban environment, larger patches of *C. erecta* would produce a stronger achromatic contrast with the background (Fischer and Samietz, 2004; Fischer et al., 2003; Ne'eman and Kevan, 2001), thus being more easily detected by parasitoids from a distance and attracting a higher number of parasitoid species. However, it must be noticed that only three parasitoid species (albeit the most efficient ones), killed a higher proportion of leaf miner larvae in larger patches. Foraging responses to plant cover in this system seem to be variable and highly species-specific, as observed in other studies (Doak, 2000; Woodcock and Vanbergen, 2008).

The distance among *C. erecta* patches negatively affected total parasitism rates on the leaf miner, a finding consistent with the study of Peralta et al. (2011) who showed that larvae of the leaf miner in experimental plant patches near natural *C. erecta* plants experienced higher parasitism rates than those located more distant, despite the parasitoids being capable to exploit other available host species (Salvo and Valladares, 1999). Moreover, Denys and Schmidt (1998) reported isolation as the main reason for the decline in parasitism rates of herbivores on experimental mugwort plots along an urban gradient. Although these authors also found parasitoid species richness decreasing with isolation, they did not directly test the relationship between diversity and parasitism.

Lower parasitism rates in more isolated patches can be expected if fewer parasitoid species are capable to reach them or if movement of parasitoid individuals is limited across the urban matrix (Denys and Schmidt, 1998; Raupp et al., 2010). In the present study, some species within the parasitoid complex of *L. commelinae* caused lower parasitism rates at increasing patch isolation,

whereas the dominant parasitoid appeared to thrive on isolated patches, and parasitism by other species was independent of isolation. Variation in dispersal ability among species may explain these differential responses to habitat isolation (Roland and Taylor, 1997; Tscharntke and Brandl, 2004). However body size, a surrogate for dispersal ability, did not appear to play a significant role with regard to patch isolation in this system (Fenoglio et al., 2010). Other factors related to host location, dependence on other resources such as flowers and refuges for adults, or competition among parasitoids, might be involved in the differential response of species (Amarasekare, 2000; Doak, 2000; Kareiva, 1987).

Richer parasitoid assemblages ultimately caused higher mortality rates to *L. commelinae*, even after removing the influence of host abundance on parasitoid richness. This finding is consistent with the few studies on biodiversity-functioning performed with natural gradients of parasitoid diversity (Fenoglio et al., 2012; Tylianakis et al., 2006, 2008), although those works were focused on host communities. The underpinning drivers of this relationship have not been explicitly addressed in this study, but some scenarios can be envisaged. On the one hand, if all parasitoid species increased their level of parasitism in larger or less isolated patches, some niche differentiation regarding host exploitation could be expected in those patches supporting several parasitoid species, for instance some temporal segregation across host stages (Rodriguez and Hawkins, 2000) or differential response of parasitoid species to host density at leaf scale (Fenoglio, 2009). Alternatively, richer assemblages on large/less isolated plant patches may have a higher probability of including certain dominant and efficient species which would ultimately be responsible for the community pattern (Finke and Snyder, 2010; Ives et al., 2005). Although we found that individual parasitoid species responded differently to patch attributes, either mechanism could be involved and deserve further exploration.

Our results underline the value of plant patch structure itself for parasitoids, since patch spatial attributes of *C. erecta* were independent from urbanization level. Nonetheless, other variables derived from urbanization could be involved in the patterns here reported, e.g. effects of altered plant quality and/or changes in insect behavior product of ambient conditions (Raupp et al., 2010). Also, the presence of vertical and horizontal barriers, typical of urban habitats, could interfere with parasitoids challenging their access to resources. However, previous studies reported that parasitism of *L. commelinae* was not influenced either by walls of buildings (Peralta et al., 2011) or by the riverbed (Fenoglio et al., 2009).

5. Conclusions

All in all, our findings highlight the importance of local scale factors derived from urbanization, like vegetation patchiness, in the context of herbivore population control by parasitoids. Moreover, given the lack of urbanization effects on parasitism rates of *L. commelinae* at landscape scale (Fenoglio et al., 2009), the present results emphasize the need to consider multiple spatial scales when studying urban habitats, due to the high heterogeneity of these environments and the different dispersal capability of organisms (Bennett and Gratton, 2012; Savard et al., 2000).

Understanding parasitoid responses to the patchiness of herbivore resources in the urban context represents an initial step to allow preservation of these beneficial insects in such highly human-modified landscapes (Hunter and Hunter, 2008). It should be noticed that several of the parasitoid species here studied also attack *Liriomyza huidobrensis* (Fenoglio and Salvo, 2009), a worldwide pest of ornamental and horticultural plants commonly found in urban gardens. Therefore, larger or less isolated patches of *C. erecta* could represent an important source of parasitoids for the pest species.

Spontaneous urban vegetation has been advocated as contributing to preserve biodiversity and valuable ecosystems services like pest control (Robinson and Lundholm, 2012). Our study supports this idea and incorporates the spatial attributes of plant patches as factors influencing diversity and functioning of parasitoid communities in the urban environment, suggesting that natural biological control could be enhanced by preserving larger clumps of plants and minimizing the distance among them. Negative social conceptions on spontaneous vegetation could represent a challenge to the implementation of urban management plans for its active conservation (Tredici, 2010). However, we expect that the results of this work will promote further studies on the potential contribution of ruderal urban vegetation to key ecological processes in order to have a sounder basis for future management strategies.

Acknowledgements

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