

Butterfly biodiversity in the city is driven by the interaction of the urban landscape and the species traits: a call for contextualised management

Clàudia Pla-Narbona (✉ claudia.planarbona@gmail.com)

CREAF: Centre de Recerca Ecologica i Aplicacions Forestals

Constantí Stefanescu

CREAF: Centre de Recerca Ecologica i Aplicacions Forestals

Joan Pino

CREAF: Centre de Recerca Ecologica i Aplicacions Forestals

Francisco J. Cabrero-Sañudo

Complutense University of Madrid: Universidad Complutense de Madrid

Enrique García-Barros

Autonomous University of Madrid: Universidad Autonoma de Madrid

Miguel L. Munguira

Autonomous University of Madrid: Universidad Autonoma de Madrid

Yolanda Melero

University of Reading School of Biological Sciences

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Abstract

Context

Urbanisation is an environmental filter for many species that leads to community homogenisation, with a few species inhabiting isolated patches (gardens) embedded in the urban landscape. Promoting biodiversity in urban areas requires understanding which and how species traits allow species to survive the urban landscape.

Objectives

The objective of this study was to assess how species traits and landscape factors combine to allow species functional groups live in the city, by means of the use of suitable patches in the urban landscape.

Methods

We used butterfly count data collected by volunteers in 24 gardens of Barcelona city, for 2018 and 2019. Species were clustered in functional groups according to their traits. We applied a multinomial choice model to test the effect of the landscape on the different functional groups.

Results

Three functional groups appeared in the city while a fourth, containing most sedentary specialist species, was filtered out. Although the observed groups had similar species richness, abundances varied depending on urban landscape characteristics. Specialist sedentary and medium mobile species were all favoured by patch connectivity; while the presence of mobile generalist species was only enhanced by habitat quality. Our results indicate that butterfly communities are more diverse in highly connected gardens.

Conclusions

Our study highlights the need of contextualised management with actions accounting for the species functional groups, rather than a management focused on general species richness. It evidences that urban landscape planning must focus on improving connectivity inside the city in order to diversify the community composition.

Introduction

The worldwide urban cover is expected to triple its area (Seto et al. 2012) and to continuously increase at least until 2050 (Chen et al. 2020). As cities continue to expand, they need to play a key role in the conservation of biodiversity by becoming suitable for a larger number of species of flora and fauna (Desrochers, Kerr and Currie 2011; Convention on Biological Diversity 2012). Accordingly, several studies have investigated the relationship between landscape urbanisation and species richness and abundances, showing diverging conclusions with negative relationships (e.g. in invertebrates, arthropods, birds and mammals; Biard et al. 2017; Lagucki et al. 2017; Ramírez-Restrepo and MacGregor-Fors 2017), positive (e.g. in invertebrates; Hill et al. 2017) or a lack of relationship (e.g. in arthropods; Christie and Hochuli 2009). The lack of a consensus transcends taxonomy and generalities have yet to be identified (Piano et al. 2020). Yet, urbanisation means that many species will increasingly encounter conditions they have not experienced before in their evolutionary history. Since species with similar life-history and ecological traits (i.e. those that define species' ecological niches) respond similarly to specific perturbations (Murray et al. 2011; Wong and Candolin 2015; Melero et al. 2016), urbanisation can result in fitness loss for many species if they show traits maladapted to their new conditions (i.e. previously adaptive traits in the natural being maladaptive in the urban environment). In contrast, individuals from species with flexible traits might be able to survive and populate urbanised areas.

Consistent with the expectation that only a subset of traits allow persistence in urban environment, recent studies have shown an overall decline of trait richness (Sol et al. 2020), and an overrepresentation of some species traits in the urban communities (often traits linked to wide niche breadth; e.g. dispersal ability, Melero et al., 2020; thermophilia, Piano et al. 2017; or reproductive rate, Santini et al. 2019). However, the mechanisms underlying this filtering of traits in the urban landscape have not been yet elucidated, since no empirical research has gone beyond descriptive associations of species richness and abundances with few species traits (Parris 2018).

Whether in the natural or in the urban environment, species richness and abundance are the consequence of a series of local dynamics. Many animal species rely on environmental cues to settle in suitable patches in fragmented habitats, whether it is the natal patch or an alternative one, but the urban landscape may provide little or misleading information to individuals. Indeed, the urban landscape is mostly composed of a built-up matrix, which reduces the abundance and connectivity of suitable habitat patches, having a stronger negative impact on the species distribution and abundances than the matrix of (semi)natural areas (Aronson et al. 2017; Melero et al. 2020; Oliveira, Lion, and Cardoso 2018). Besides, urban green areas (i.e., the potentially suitable patches) are usually small, isolated, and influenced by strong edge effects from the surrounding built matrix (e.g., Verbeylen et al. 2003; Lizée, Manel, Mauffrey, Tatoni and Deschamps-Cottin 2011; Braaker, Ghazoul, Obrist and Moretti 2014). Overall, whether individuals of a species, and hence its populations, succeed or fail to use suitable urban patches may depend on the interaction of a certain (but unknown) combination of specific traits and landscape factors.

In this study, we analyzed how species traits and landscape factors combine to allow individuals of a series of species to use suitable patches in the urban landscape. Specifically, we assessed how the

probability of an individual to be in a urban patch depends on the habitat quality, size and functional connectivity of the patches, and on the mixture of the species traits (species were clustered into functional groups to avoid correlations between traits; see Materials and methods; Eskildsen et al. 2015; Middleton-Welling, Wade, Dennis, Dapporto, and Shreeve 2018). We chose the above mentioned landscape variables because they are key for species presence (e.g. Melero et al. 2020) and also manageable features of the landscape. Butterflies were used as model species because they are recognised indicators of ecological and biodiversity impact (Pollard et al. 1996; Thomas 2005) and respond quickly to habitat transformations (Krauss et al. 2010; Ubach et al. 2020). Besides, butterflies comprise a range of traits representative of insects and other short-lived species, guaranteeing a wide range of ecological variations and assuring reliable ecological predictions (Thomas 2005; Carnicer et al. 2013; Eskildsen et al. 2015). Our study is based in empirical count data gathered by a volunteer-based citizen-science project started in 2018 consisting on regular surveys of butterflies in a series of green areas (gardens and parks, gardens hereafter) within the city of Barcelona.

We hypothesised that (i) for all functional groups, the probability of an individual butterfly to be present in a garden will grow with the increase of garden habitat quality, size and connectivity; yet (ii) the effect of these landscape variables will increase in importance from wide to narrow breadth niche traits (i.e., from generalists, highly mobile and high reproductive rate species, to specialists with limited mobility and low reproductive rate). In fact, we expected individuals at the extreme of these restrictive traits to be rare or even absent in the city.

Materials And Methods

Study area and data collection

The study area was the city of Barcelona and its immediate conurbation, consisting of natural, (semi)natural and urban area, resulting in a total of 12,600 ha of study area. The city of Barcelona reaches ca. 6,600 ha (52.4% of the study area) with a dense urban matrix; the rest of the study area corresponds to the south-eastern part of the Natural Park of Collserola, with 3,500 ha (27.8% of the study area), known to be important for its species source effect (Melero et al. 2020); and to the built-up area from the neighbouring cities and suburbs, with 2,500 ha (19.8% of the study area; Fig. 1).

Sampling was done by volunteers involved in the citizen science program “urban Butterfly Monitoring Scheme” or uBMS (ubms.creaf.cat), consisting of regular adult butterfly monitoring surveys from March to the end of September covering the flight period for most species in the area. Surveys were done along 300 m transects within selected gardens following the ‘Pollard walk’ standard methodology (Pollard 1988). A total of 39 volunteers covered 25 gardens in 2018, first year of the project, and 24 gardens during 2019 (total number of different gardens = 27). Our statistical methodology (Multinomial choice model) allows for unequal sampling; yet, following a conservative approach, we avoided low surveyed gardens in our analyses by selecting gardens monitored weekly, biweekly in two non-consecutive weeks or once per month, at least during one of the study years. This selection resulted in a total of 24 selected

gardens. The 24 selected gardens comprised 318 ha of the total 335 ha of public and private gardens included in the program. The size of studied gardens ranged from 0.49 to 68.69 ha (mean = 13.77 15.23 ha; Fig 1).

Landscape variables

The studied gardens were first delineated using current (2018) orthoimages of the Cartographic Institute of Catalonia. Then, their land cover composition was inferred from a land-cover map (a 5-m cell grid) obtained from the combination and reclassification of the Copernicus Urban Atlas (<https://land.copernicus.eu/local/urban-atlas>), the land cover map of Catalonia MCSC 2015 (<https://www.creaf.uab.es/mcsc/>), and the Spanish SIOSE Land Cover Map (<https://www.siose.es/>).

To gather garden habitat quality, we used a proxy based on the similarity of the garden vegetation in relation to the (semi)natural areas where butterfly species richness is higher (catalanbms.org/ca/habitats/). We did so based on previous research pointing the higher importance of native flora for many butterfly species (Graves and Shapiro 2003). Vegetation per garden was checked *in situ*, and categorized into values from zero to three according to the following criteria: 0 for no-vegetation; 1 for non-native spontaneous and ornamental vegetation; 2 for isolated native plants embedded by non-native vegetation; 3 for continuous native vegetation. We then calculated habitat quality of each garden as the weighted mean of the categories, i.e., the sum of the (numbered) categories multiplied by their covered area divided by the total area of the garden, i.e.:

$$Q_i = \frac{\sum(C_{i,j} * A_{i,j})}{A_i}$$

where Q_i is the habitat quality of the garden i , C_j is the vegetation category j (0-3), A_{ij} is the area of that category j at each garden i , and A_i is the total area of the garden (see Online resource 3 and 4a).

Garden size was assessed based on three different metrics: (i) total garden area, including vegetated and non-vegetated areas; (ii) open vegetated area including grasslands, shrubs, crops and ornamental vegetated areas, known to be important for butterfly specialists of open areas (Ubach et al. 2020); and (iii) closed vegetated area, including forests and isolated urban trees, important for butterflies adapted to closed areas (Suggitt et al., 2012; Ubach et al., 2020; see Online resource 3 and 4b-c).

Connectivity was calculated for each surveyed garden in relation to all vegetated areas throughout the study site including both the surveyed and non-surveyed gardens as well as the adjacent natural areas. We used the 5-m land cover grid categorized as vegetated (i.e., belonging either to open or closed vegetation categories) and non-vegetated categories. We then calculated the connectivity of gardens based on the number and distance of vegetated cells around them. Given the importance of the urban matrix in the dispersal of individual butterflies (Lizée et al. 2011; Melero et al. 2020) we used a modification of the Hanski index (Hanski 1998) that includes the impedance of each land cover category:

$$S_i = \sum_i^n A_i \sum_{j \neq i} p_j * \exp(-\alpha * d_{ij})$$

where S_i is the connectivity of the garden i , as before A_i is the total vegetated area of the garden, p_j is each vegetated cell j outside the garden i in area units (0.0025 ha), and d_{ij} is the cost distance from garden i to each vegetated cell j . Distance cost (d_{ij}) is the Euclidean distance (in m) to each vegetated cell j , weighted by a resistance value assigned to each land cover category based in previous studies (Lizée et al. 2011; Fernández Chacón 2014; Lizée et al. 2016; see Online resource 5 and 6). To account for the effect of the scale on butterfly species richness and abundance, we used the model parameter for adjusting the maximum dispersal distance to three contrasting values (200, 500, 1,000 m) to obtain a connectivity metric at these three scales (see Online resource 3 and 4d-e). Larger scales surpassed the limits of the city (Fig. 1).

All seven patch and landscape variables were tested for significant correlations, leading to the selection of five variables: garden habitat quality, open vegetated area, closed vegetated area, and connectivity at 500 m and at 1,000 m buffer (see Online resource 7).

Species traits, functional grouping and statistical modelling

We chose seven species traits known to be important in behaviour and ecology of butterfly and invertebrates in general (Stefanescu et al. 2011; Melero et al. 2016; Moretti et al. 2017). These were adult habitat specialisation, larval specialisation, voltinism (i.e., the number of annual generations), overwintering stage, species temperature index, mobility ability, and habitat openness preference (full description in Table 3 of Online resource 1: Appendix 1).

These seven traits were used to group the species into functional groups based on the co-occurrence of traits within species (Eskildsen et al. 2015; Middleton-Welling et al. 2018). Clustering was done using all butterfly species known to occur in the Mediterranean area of Catalonia, to which the study area belongs, to later extract the cluster of each of the observed species in the sampled gardens to feed the modelling analysis. The total of butterfly species in Catalonia is 152, excluding those rarely seen (i.e., species with < 20 observations along the last 24 years; catalanbms.org, Vila, Stefanescu, and Sesma 2018).

Hierarchical clustering was used to group the 152 Catalonian butterfly species according to the similarities between species-specific values of their traits (species listed in Online resource 10). We applied a Hill & Smith principal component analysis, which performs a multivariate analysis with mixed categorical and continuous variables (Hill and Smith 1976). Best clustering method and optimal number of clusters were then determined following the criteria of elbow and silhouette graphics (for exhaustive explanation see Online resource 1: Appendix 1). Clustering was done using package ADE4 in R 3.5.3 (Dray and Dufour 2007; R Core Team 2016).

We used multinomial discrete choice models (multinomial regression models) to analyse the probability of an individual to be present in a specific garden. Multinomial models account for the ability of individuals to make choices based on collected information; therefore assuming that an individual i (from a total of N individuals) will select a patch (i.e., gardens in our study) from a finite number of alternatives due to the characteristics of the chosen patch, the alternative non-chosen patches and of the individual (e.g., Melero, Cornulier, Oliver, and Lambin 2018; Vardakis, Goos, Adriaensen, and Matthysen 2015). Thus, for each detected individual within the 24 gardens, the response variable is the garden where the individual has been observed (the choice) and the explanatory variables of this choice relate to the characteristics of the gardens (garden habitat quality, open vegetated area, closed vegetated area, connectivity at 500 m buffer and connectivity at 1,000 m buffer) and of the individual (the functional cluster it belongs to) (see model details in Online resource 2: Appendix 2).

The model was fitted in a Bayesian framework using a Markov Chain Monte Carlo (MCMC) procedure, with the Package 'MNP' (Imai and van Dyk 2005). All independent continuous variables were standardised to reduce standard deviation. We settled the Gaussian with "infinite" variance, which are the default non-informative priors and drew 1,500,000 MCMC Gibbs samples, discarding the first 500,000, and retaining one iteration in 100 for storage-saving purposes and predictions. Finally, we ran two independent MCMC chains, starting from over-dispersed values, to diagnose their convergence with the Gelman-Rubin statistic from package 'coda' (see Online resource 8; Gelman and Rubin 1992).

Results

Functional grouping of butterfly species

The optimal number of functional groups for the 152 Catalanian species was four, using the divisive clustering method (see Online resource 1: Appendix 1). Cluster C1 ($N = 62$, hereafter C1-sedentary specialists) enclosed the least mobile, with lower reproductive rate and the most specialist butterflies in both the larval and adult stages. This cluster included species that overwinter in the egg or larval stage. Cluster C2 ($N = 58$, C2-sedentary intermediate specialists) included species with low mobility, mostly overwintering in the larval stage (with only 15 species overwintering in the pupal and egg stages). Species in this cluster were less specialists for nectar resources than those in C1 and were either uni- or multivoltine. Cluster C3 ($N = 19$, C3-forest specialists) was the only cluster composed by forest specialist species (i.e., with preference for closed vegetated areas), that were also multivoltine, generalist species with intermediate mobility. The last cluster C4 ($N = 13$, C4-mobile generalists) enclosed highly mobile, multivoltine, the most generalist (both in larval and adult stage) species, and species with high temperature tolerance (Table 1).

From the pool of 152 species that were used to create the clusters, 36 species were observed in the gardens of the study area, amounting to a total of 3545 individuals. None of the species in group C1-sedentary specialists were seen in the city; but 14 species and 412 individuals (11.6% of the total observations) of group C2-sedentary intermediate specialists, 10 species and 1,241 individuals (35%) of

C3-forest specialists, and 12 species, and 1,892 individuals (53.4%) of group C4-mobile generalists were observed (Species listed in Online resource 11).

Landscape effects on contrasting functional groups

The probability of an individual butterfly to be in a garden varied between gardens and functional groups, yet the average probability was 0.042 (0-0.361, median = 0.02). C2-sedentary intermediate specialists and C3-forest specialists appeared both with a lower median probability of ca. 0.01, while C4-mobile generalist butterflies were the most frequent in the gardens with a median probability of 0.02 (see Online resource 9).

C2-sedentary intermediate specialists were positively affected by large scale connectivity (1 km), such as one unit of this large-scale connectivity increased ca. 33 % the probability of a C2-butterfly to appear in a garden. Meanwhile, C3-forest specialists were more likely to be in gardens with forested areas (i.e., closed vegetated areas) and less in open areas. Connectivity also increased their probability but at a lower spatial scale (500 m). Lastly, the most frequent, i.e. C4-mobile generalist butterflies, were independent of the landscape except for a negative effect of the garden habitat quality, our proxy based on (semi)natural native vegetation (Table 2; Fig. 2).

Discussion

Using a volunteer based project over 24 urban gardens, we provided new evidence on the effects of the urban landscape on species presence in urban patches, and how this relation is mediated by species traits summarised in functional groups (clusters), a combination that has been widely overlooked in research (but see Delgado de la Flor et al. 2020). While the functional C1-sedentary specialists was not seen in the city, all other clusters had similar number of species in the city; i.e. species richness of C2-sedentary intermediate specialists, C3-forest specialists and C4-mobile generalists was similar with 14, 10 and 12 species, respectively. Yet, both the raw abundances and the probability of their individuals to be present in the gardens decreased with the reduction of species mobility, the reproductive rate and the trophic generalisation. In fact, C4 was 4.5 and 1.5 times more abundant than C2 and C3 respectively.

We expected more suitable, larger and better connected gardens to favour the presence of all species independently of the functional group, yet this was not completely detected. Garden connectivity to vegetated areas (including other gardens and the semi(natural) areas) was the limiting factor for the individuals of C2-sedentary intermediate specialists. The importance of connectivity increased in its effect and its spatial range (500 to 1,000 m) from mobile generalist species to sedentary specialists (i.e., it had no significant effect on C4-mobile generalist, but an increasingly significant one for C3-forest specialist species and C2); supporting the key role of connectivity for less mobile species in urban environments (Braaker et al. 2017; Delgado de la Flor et al. 2017; Melero et al. 2020). Indeed, for these two clusters, connectivity was the only (for C2) or main factor (for C3) involved in their presence; despite C3-forest specialist butterflies preferring closed than open vegetation (e.g., *Pararge aegeria* (Linnaeus 1758) and *Gonepteryx rhamni*). The functional group C4-mobile generalists was the less influenced by

the landscape characteristics and the most likely to be present in any garden, as expected since their traits are common in urban areas. The dominance of mobile generalist species in urban gardens, whether as city dwellers or transients, has also been observed in arthropods (Delgado de la Flor et al. 2017; Melero et al. 2020; Delgado de la flor et al. 2020), birds (Bonier et al. 2007; Croci et al. 2008), and mammals (Santini et al. 2019). Despite the low effect of the urban landscape on these species, there was a negative effect of the garden habitat quality. We associate this result to our definition of habitat quality because we gave low quality values to ornamental non-native plants. However, generalist species frequently feed on some of these plant species, both at the larval and at the adult stage; for instance (Graves and Shapiro 2003), species of the genera *Vanessa* (Fabricius 1807), *Pieris* (Schrank 1801), *Iphioclis* (Hübner 1807), and *Papilio* (Linnaeus 1758) are highly attracted by non-native plants frequently used as ornamental in urban gardens (e.g. *Buddleja davidii* (Franchet 1887), in Barcelona). Therefore, ornamental non-native vegetation can provide good habitat for generalist species that we failed to capture in our definition of habitat quality.

Overall, while there was not a unique combination of garden characteristics favouring all functional groups at once, our results indicate that butterfly communities will be more diverse in highly connected gardens, the most likely to harbour less abundant species (C2-sedentary intermediate specialists and C3-forest specialists). Increasing the presence of these two groups, especially C3, will also require combining open and closed vegetation covers in the gardens. In fact, habitats with diverse vegetation structure can provide microhabitats suitable for a higher number of species (Sarhou et al. 2014; Sivakoff et al. 2018). Therefore, it is the combination of the impedance of the urban matrix and garden isolation (i.e. connectivity), and the vegetation characteristics that seems to not only shape butterfly richness as previously seen (Lizée et al. 2016; Tzortzakaki, Kati, Panitsa, Tzanatos, and Giokas 2019) but also the community composition of the species based on their traits.

Worryingly, the C1-sedentary specialists were not present in any of the gardens, and as such we could not analyse the trait-landscape interactions affecting them. However, it has been shown that garden isolation from urbanisation represents an ecological barrier that impedes colonization of those species that lack the traits to overcome dispersal challenges such as mobility or larval trophic specialisation (Braaker et al. 2017; Sivakoff et al. 2018; Melero et al. 2020). The loss of richness with the lack of C3 species and reduced abundances of C2 and C4, comes with an overall community homogenization and impoverishment of urban areas which has also been observed in other studies (Beninde, Veith, and Hochkirch 2015; e.g. in birds and butterflies, Aronson et al. 2014; Olivier et al. 2016). In fact, our study revealed that because of these factors there were almost half (0.58) times less species in the city than in the surrounding (semi)natural area.

Conclusions For Practice

Our findings evidence that increasing urban biodiversity requires diversifying the community composition, accounting for the key traits of the species and how the landscape differently affects them. To do so, management actions need to be targeted to increase the presence and abundance of those species with

no or less presence and abundances in the cities. Therefore, most efforts of management should be concentrated at improving connectivity inside the city to facilitate those migration events between suitable patches and promote population establishment in those patches, at least until these species break the colonization barriers imposed by the urban matrix (Melero et al. 2020). Further research could account for how the different species use the urban green areas and flora (e.g., breeding and feeding species) and how this relates to the species traits, but further data is needed to elucidate this.

Declarations

Funding

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Conflicts of Interest

The authors declare no conflict of interests.

Ethics approval

The study did not include capturing individuals. Hence, no ethical approvals were needed.

Consent to participate

Not applicable.

Consent for publication

All authors agree on the publication of the current version.

Availability of data and material

The datasets generated during the study will be available in the Dryad repository upon acceptance.

Code availability

The code of the model generated during the study will be available in the Dryad repository upon acceptance.

Authors' contributions

YM conceived the idea and designed the methodology; YM, CS, MLM, EGB, FCS and JP initiated the project, with YM coordinating the volunteers that gathered butterfly data; CP did the field work; CP performed the clustering, supervised by CS, JP and YM; CP and JP calculated the landscape variables; CP and YM did the multinomial model, interpreted the results and wrote the manuscript. All authors edited the manuscript and gave final approval for publication.

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Tables

Table 1 Species traits within each functional group resulting from the ordination analysis of seven different trait categories according to divisive hierarchical clustering. N relates to the number of species in each cluster from the pool of 156 Catalanian species. Traits with relevant weight per cluster are marked in bold. For those traits with mixed categories the main ones appear first, and the secondary ones in brackets

Species trait	C1-sedentary specialists (N=62)	C2-sedentary intermediate specialists (N=58)	C3-forest specialists (N=19)	C4-mobile generalists (N=13)
Species specialisation index	Specialist	Intermediate	Medium Generalist	Generalist
Larval specialisation	Specialist	Intermediate	Intermediate	Generalist
Voltinism	Univoltine ^[1]	Both	Multivoltine ^[2]	Multivoltine
Overwintering stage	Larva, Egg / (Pupa)	Larva / (Pupa, Egg)	Adult, Pupa / (Any, Larva)	None, Pupa / (Larva)
Species temperature index	Low	Intermediate / High	Intermediate	High
Mobility ability	Very low	Low	Medium	High
Habitat openness preference	Open	Intermediate / Closed	Closed	Intermediate / Open

[1] Except of three multivoltine species: *Carcharodus baeticus* (Rambur, 1839), *Lycaena tityrus* (Poda, 1781) and *Melitaea trivia* (Denis & Schiffermuller, 1775).

[2] Except of five univoltine species: *Gonepteryx rhamni* (Linnaeus, 1758), *Anthocharis cardamines* (Linnaeus, 1758), *Nymphalis antiopa* (Linnaeus, 1758), *N. polychloros* (Linnaeus, 1758), and *Libythea celtis* (Laicharting, 1782).

Table 2 Estimated coefficients, their mean, SD (Standard deviation) and 97.50% credible interval bounds for the probability of an individual to be present in a garden in relation to the standardised landscape covariates. Individual-specific covariates (i.e., clusters) are included in choice-specific covariates (i.e., landscape variables) with a ‘_CX’ (where X is clusters’ number).

Significant effects are marked in bold

Cluster	Covariates	Estimate	SD	2.50%	97.50%
C2-sedentary intermediate specialists	Habitat quality	0.04	0.04	-0.05	0.10
	Open vegetated area	0.05	0.04	-0.05	0.13
	Closed vegetated area	-0.06	0.08	-0.20	0.09
	C500	-0.21	0.15	-0.48	0.07
	C1k	0.34	0.17	0.10	0.73
C3-forest specialists	Habitat quality	0.04	0.03	-0.02	0.11
	Open vegetated area	-0.19	0.05	-0.27	-0.08
	Closed vegetated area	0.09	0.04	0.01	0.16
	C500	0.30	0.12	0.02	0.50
	C1k	-0.045	0.062	-0.14	0.12
C4-mobile generalists	Habitat quality	-0.30	0.11	-0.52	-0.11
	Open vegetated area	0.00	0.20	-0.38	0.38
	Closed vegetated area	0.30	0.14	-0.02	0.56
	C500	-0.53	0.42	-1.03	0.55
	C1k	-0.39	0.41	-1.35	0.09

Figures

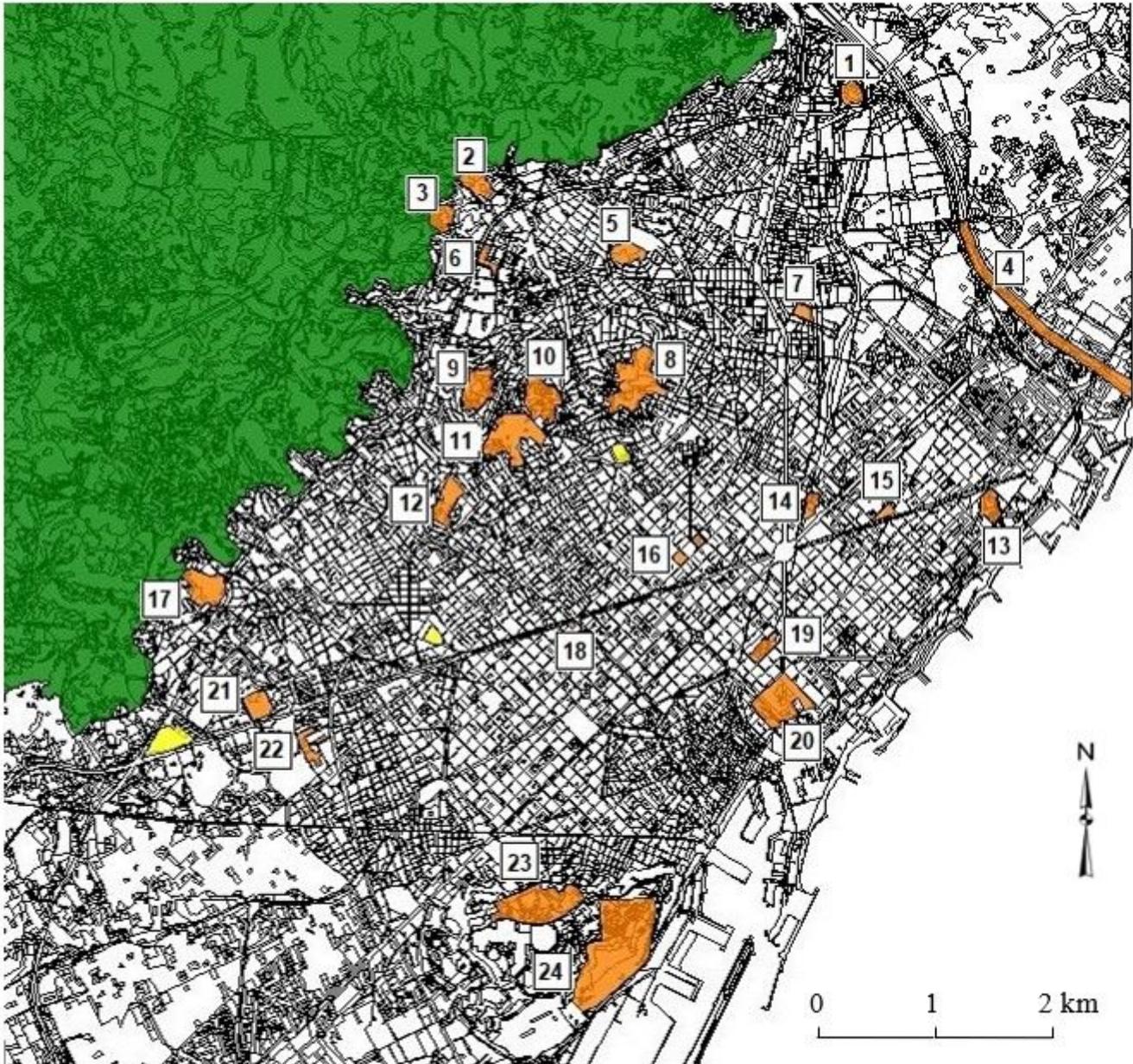


Figure 1

Study area including the urban area of Barcelona, the adjacent sector of the Collserola Natural Park and the neighbouring towns. In green, the Natural Park; in orange the 24 urban gardens studied with their ID numbers (see Online resource 3). In yellow, the three discarded gardens

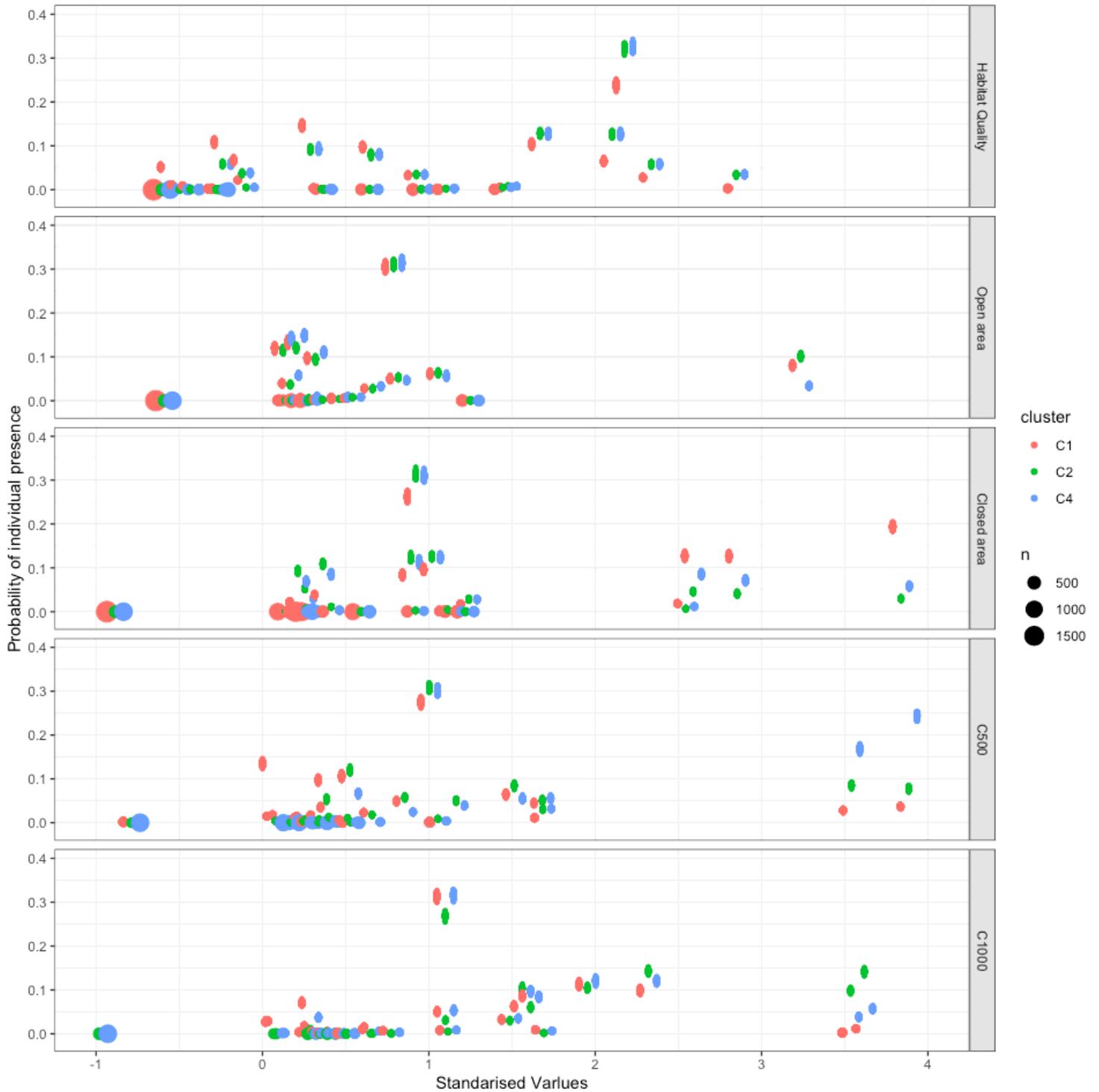


Figure 2

Predicted probability of presence for each individual butterfly (coloured circles) in each garden in relation to the: habitat quality, open vegetated area, closed vegetated area, connectivity at small-scale (i.e., 500 m) and connectivity at large-scale (i.e., 1 km) (standardised values). In green C2-sedentary intermediate specialists, in blue C3-forest specialists and in red C4-mobile generalists. To reduce superposition a +0.05 jitter was applied to the covariate axis for the clusters C2 and C3. For each cluster, each individual observed ($n = 3\ 545$) is multiplied by the 24 potential gardens, giving a total of 0.12, 0.35 and 0.53

probabilities. Predictions for each variable were based on fixing all the other covariates values at their mean. The size of the circles relates to the number of observations from one (smallest circles) to 100 observations (largest circles)

Supplementary Files

This is a list of supplementary files associated with this preprint. Click to download.

- [Onlineresource1011.xlsx](#)
- [Onlineresource1Appendix1.docx](#)
- [Onlineresource2Appendix2.docx](#)
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