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1 **Islands in cities: urbanization and fragmentation drive taxonomic and functional variation in ground**  
2 **arthropods**

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8 **Abstract**

9 The conversion of natural lands in urban areas is exponentially increasing worldwide, causing a major  
10 decline in biodiversity. Environmental alterations caused by urbanization, such as land conversion and  
11 isolation of natural patches, favour tolerant and generalist species, causing both species loss and  
12 replacement. In addition, selective pressure is exerted on particular functional traits, driving a functional  
13 homogenization or turnover of biotic communities. We sampled ground arthropods within the municipality  
14 of Turin (NW-Italy), wherein an isolated and a connected control subplot were repeatedly sampled at 15  
15 stations distributed along a gradient of increasing urbanization. Such a nested sampling design allowed us  
16 to investigate the taxonomic and the functional responses of carabids and spiders to both the urbanization  
17 level and patch isolation. First, we highlighted the dominant role played by species homogenization  
18 (*nestedness*) in explaining both taxonomic and functional variation in both groups of arthropods. Secondly,  
19 we showed that urbanization causes simultaneously functional homogenization and replacement in both  
20 carabid and spider assemblages, whereas patch isolation influences carabid species composition and  
21 homogenizes and shifts spider taxonomic and functional composition. Lastly, by relating community-  
22 weighted means of body length, dispersal capacity and trophic strategy to the urbanization and isolation  
23 gradients, we demonstrated that urbanization alters the trophic structure of both taxonomic groups and  
24 increases the average dispersal capacity of spiders. On the other hand, patch isolation affected the

25 functional composition of spiders only, reducing the body size and increasing dispersal capacity and the  
26 proportion of web-builder species. Our results demonstrate that both urbanization and patch isolation alter  
27 species composition by causing functional and taxonomic homogenization. In addition, they exert a strong  
28 filtering effect on community functional traits, increasing the proportion of phytophagous species in  
29 carabids, and increasing dispersal capacity and web-builders occurrence in spiders, while reducing spider  
30 body size.

31 **Keywords:** taxonomic homogenization; functional homogenization; carabids; spiders; traits

32

### 33 Introduction

34 Throughout recent human history, the conversion of natural lands in urban areas increased exponentially  
35 everywhere in the world (Seto et al., 2011), representing today one of the main anthropogenic impacts on  
36 natural ecosystems (Grimm et al., 2006). In particular, the process of urbanization induces major  
37 detrimental effects on the environment, as a result of the severe and fast changes in landscape structure  
38 and physical conditions (Parris, 2016). These alterations are common to all urban settlements, and they are  
39 expected to drastically affect biodiversity in all continents (McKinney, 2006). However, although an  
40 increasing number of studies investigated the relationship between urbanization and species richness, the  
41 underlying mechanisms that drive biodiversity patterns in urban areas are not fully understood (but see  
42 Shochat et al., 2006, 2010; Pickett et al., 2011; Turrini & Knop, 2015), with inconsistent results among the  
43 studied taxa (see Chace & Walsh, 2006; McKinney, 2008; Niemela & Kotze, 2009; Luck & Smallbone, 2010;  
44 Jones & Leather, 2012; Ramirez-Restrepo & MacGregor-Fors, 2017, Piano et al., 2019).

45 From an evolutionary point of view, urbanization represents a primary selective pressure on biotic  
46 communities, especially when considering the hindrance to the establishment and/or survival of certain  
47 species (McKinney, 2002). Such a strong environmental filter is expected to differentially affect species  
48 from the regional pool, by either favouring tolerant taxa or disfavouring the most sensitive ones, with  
49 subsequent changes in species composition, mostly due to species loss (Sadler et al., 2006; Magura et al.,  
50 2008; Niemelä & Kotze, 2009; Piano et al., 2017) or to species replacement (Sattler et al., 2010; Vergnes et  
51 al., 2014; Knop, 2016) along the urbanization gradients. As a result, both phenomena result in biotic  
52 homogenization, due to the loss of rare and specialised species, and the gain of widespread tolerant ones  
53 (McKinney, 2006).

54 In this context, a prerequisite to properly describe the mechanisms that drive variations in taxonomic and  
55 functional community composition ( $\beta$ -diversity) is the evaluation of the relative contribution of *turnover*  
56 and *nestedness*, whereby *turnover* measures the degree to which species are replaced by others—species  
57 replacement—and *nestedness* measures the degree to which communities of species-poor sites are a  
58 subset of those in species-rich sites—i.e. species loss (Ulrich, Almeida-Neto & Gotelli, 2009). To our

59 knowledge, this approach has never been applied to investigate ground arthropod communities in urban  
60 areas (but see Brice, Pellerin & Poulin, 2017 and Gianuca et al., 2017 as examples in other contexts).

61 According to the “habitat templet theory” (Southwood, 1977, 1988), the above-mentioned selection  
62 process may have major effects on particular functional traits, like morphological or physiological  
63 attributes, or it could affect the realized niche of a certain species (Webb et al., 2010). Interestingly, several  
64 studies confirmed this hypothesis, providing evidence that, by filtering species according to their life-history  
65 traits and ecological needs, urbanization often determines shifts in life-history community traits, i.e.  
66 functional replacement (e.g. Vergnes et al., 2014; Concepción et al., 2015; Gianuca et al., 2017; Merckx et  
67 al., 2018), or deletions of some functional groups, i.e. functional loss (e.g. La Sorte et al., 2014; Piano et al.,  
68 2017). As a consequence, urbanization may favour species with common traits conferring positive fitness  
69 under local conditions, causing the overall functional homogenization of urban communities (Olden et al.,  
70 2004).

71 The study of urban-rural gradients has been successfully used to highlight the effects of urbanization on  
72 biotic communities (McDonnell & Hahs, 2008). However, urbanization gradients are indirect and complex,  
73 since they include different types of disturbance acting at different spatial scales (Pickett et al., 2011; Parris,  
74 2016), and changes in community may depend on several factors associated with urbanization (Rebele,  
75 1994; Seto et al., 2011). For instance, urbanization correlates with a higher presence of impervious  
76 surfaces, which are known to have an effect on temperature, soil nutrient cycling and gas exchange and  
77 other physical or chemical parameters (Parris, 2016). In addition, the growth of urban areas is responsible  
78 for natural and semi-natural habitat fragmentation, which creates small and isolated residual patches,  
79 which are typically highly dynamic and impermanent ecosystems (Parris, 2016). Accordingly, since physical  
80 and landscape alterations are strongly correlated, even acting synergistically along urbanization gradients  
81 with similar demographic consequences, it is often difficult to distinguish their effects on biotic  
82 communities. Therefore, in order to isolate and describe their effects on biodiversity, a hierarchical  
83 approach is required, taking into consideration the nested nature of ecological systems (McDonnell & Hahs,  
84 2008).

85 In this research, we aimed to assess which mechanisms are involved in shaping ground arthropod  
86 assemblage composition in urban areas, focusing on the role of taxonomic/functional *turnover* and/or  
87 *nestedness*, and how functional traits respond to urbanization. Spiders (Araneae) and carabids (Coleoptera,  
88 Carabidae) were selected as model groups, because of their well-defined ecology and taxonomy, together  
89 with their well-known response to urbanization, both in terms of taxonomic and functional diversity (e.g.  
90 Niemelä & Kotze, 2009; Sattler et al., 2010; Vergnes, Le Viol & Clergeau, 2012; Vergnes et al., 2014; Piano et  
91 al., 2017; Buchholz et al., 2018). The assemblages of the two selected groups were investigated by means  
92 of a hierarchical sampling design, combining data from isolated and connected control environmental  
93 patches along an urbanization gradient. The inner areas of traffic roundabouts were selected as isolated  
94 plots, while control patches were established on green areas connected with the surrounding  
95 environmental matrix. Specifically, we investigated whether: i) *nestedness* and *turnover* components  
96 differentially explain total variation ( $\beta$ -diversity) in taxonomic and functional diversity; ii) the urbanization  
97 level and patch isolation affect taxonomic and functional composition of carabid and spider communities,  
98 both in terms of homogenization (*nestedness*) and shift (*turnover*); and iii) functional traits show differential  
99 responses to the urbanization level and patch isolation.

100

## 101 **Materials and methods**

### 102 *Sampling design*

103 The study was carried out in 15 sampling plots (Fig. S1), randomly selected along an urbanization gradient  
104 in the municipality of Torino (about 880,000 inhabitants, NW-Italy). The average distance between the  
105 centre of sampling plots was 8,878 m (sd =  $\pm 1,705.6$ ), ranging from 1,408 to 18,512 m. In order to test the  
106 effects of patch isolation on spiders and carabids, in each plot two sampling subplots were identified: i) one  
107 in an isolated patch, within a traffic roundabout; and ii) the other in the nearest green area, connected with  
108 the surrounding environmental matrix (connected control patch) (Fig. 1). Sampling plots consisted in  
109 circular areas of 150 m diameter (approximately 18,000 m<sup>2</sup>), centred on the roundabout. Roundabouts  
110 ranged from 6 to 30 m radius (i.e. approximately from 150 to 2,500 m<sup>2</sup> of surface area), and we evaluated

111 their age using historical aerial maps on Google Earth (<https://www.google.com/earth/download/ge/>).

112 Based on the year of construction, we equally distributed them among three age classes (1 = less than 10

113 years; 2 = 10 years (built in 2007); 3 = more than 10 years) (see Appendix A: Table S1). Given that both

114 taxonomic groups were not affected by these factors, namely the area and the age of the roundabouts, we

115 did not include this information in our subsequent analyses. Connected control subplots were placed in the

116 closest green area within the sampling plot, being represented by small green urban patches (1,000 m<sup>2</sup>),

117 urban parks or seminatural areas at the city borders (up to a few hundreds of hectares). The isolated patch

118 (roundabout) within a certain sampling plot was always smaller than the green area encompassing the

119 connected control subplot. In all sampling plots, subplots were located in comparable semi-natural

120 grassland habitats, maintained by regular mechanical mowing (see Fig. S2 for real examples). Roundabouts

121 were assumed to represent a good approximation of isolated patches, since they are completely

122 surrounded by roads, which likely constitute a selective barrier for our targeted arthropod groups.

123 The urbanization level was assessed in each sampling plot by extrapolating the impervious surface coverage

124 (ISA, Elvidge et al., 2007) from aerial images of the study area. We used the proportion (%) of impervious

125 surfaces as a proxy for the urbanization level, which was calculated from updated digital maps (year 2017)

126 by means of the QGIS software (QGIS Development Team, 2018) in a buffer of 1,600 m radius (see Fig. S1).

127 We tried as much as possible to avoid overlapping buffer areas. However, due to logistic constraints related

128 to permissions issued by the local authorities to access the areas of the roundabouts, our final design

129 implied an overlap of 9.6 km<sup>2</sup> between buffer areas, corresponding to about 8% of the total area covered

130 by the buffers (approximately 120 km<sup>2</sup>). Given that carabid beetles and spiders often respond to factors at

131 different scales (Braaker et al., 2014; 2017), we checked whether this spatial scale represents the best

132 option to evaluate the response of ground arthropods to urbanization in the investigated area (see

133 Supplementary Materials).

#### 134 *Data collection*

135 Ground-active arthropods were captured in each sampling subplot within a sampling section 30 m long

136 using three pitfall traps (5-10 m apart), placed at least 20 cm from the patch border, and at least 5 m apart

137 to increase catch efficiency (Ward, New & Yen, 2001). Plastic jars (diameter 90 mm, length 110 mm) were  
138 dug into the ground so that the upper edge of the glass was flush with the soil surface, and filled with 20 ml  
139 of 50% propylene glycol solution. All traps were emptied every three weeks, in three sampling sessions  
140 between the 16<sup>th</sup> of May and the 18<sup>th</sup> of July 2017. The collected material was preserved in a 70% ethanol  
141 solution. Spiders and carabids were sorted, identified and counted in the laboratory according to the keys  
142 provided by Pesarini & Monzini (2010) and Boeken et al. (2002) for carabids, and by Nentwig et al. (2018)  
143 for spiders. Data from pitfall traps collected in the same subplot were pooled together for subsequent  
144 analyses.

145 It should be pointed out that pitfall traps provide data on activity density rather than real abundances of  
146 species, thus overestimating extremely active species. Although the pitfall trap-based sampling method  
147 introduces some bias in relative species abundances, the extent of the bias should be similar for each  
148 sampling site.

#### 149 *Functional traits*

150 Body length, dispersal capacity and trophic requirements – hunting mode for spiders and trophic level for  
151 carabids – were considered as key functional traits, based on Buchholz et al. (2018).

152 Carabid species were assigned to two trophic groups (1 = zoophagous; 2 = phytophagous) according to  
153 Vanbergen et al. (2010), and to three dispersal groups based on wing development (1 = brachypterous; 2 =  
154 dimorphic; 3 = macropterous) according to Desender et al. (2008). The average body size was assigned to  
155 each species according to Desender et al. (2008). Spider species were assigned to functional groups  
156 reflecting their strategy of food provision, according to Cardoso et al. (2011). In order to obtain a  
157 comparable number of species among categories, we referred to a broader classification (1 = hunters; 2 =  
158 web-builders), without considering in detail the specific hunting strategies. Spiders were assigned to three  
159 dispersal groups (1 = non- or sporadic ballooners; 2 = ballooners at juvenile stages only; 3 = ballooners at  
160 juvenile and adult stages), based on Blandenier (2009), Bell et al. (2005) and Simonneau, Courtial and  
161 Pétilion (2016). Each species' male and female average body sizes were extrapolated from Nentwig et al.



162 (2018). We retained only female's data due to the high correlation between the two measures (Pearson's  
163 correlation test:  $r = 0.97$ ;  $P < 0.001$ ).

#### 164 *Data analysis*

165 All statistical analyses were performed using the *R* software (R Core Team, 2017).

166 Preliminary Analyses. In order to identify the best scale of response to the urbanization level for carabids  
167 and spiders, we calculated the degree of urbanization for each station in three progressive larger buffers,  
168 defined as circles with a 100 m (small scale), 400 m (medium scale) and 1600 m (large scale) radius - with  
169 the roundabout as the centre. For each taxonomic group, we proceeded with fitting three separated  
170 models, one for each scale of the urbanization level, after eliminating outliers in our dependent variables  
171 following the standard protocol for data exploration proposed by Zuur et al. (2009). We tested the response  
172 of the total abundance of carabids and spiders against the urbanization level, patch isolation and their  
173 interaction by means of Generalized Linear Mixed Models (GLMMs) (Zuur et al. 2009), performed with the  
174 function "glmer.nb" in the *lme4* package (Bates et al., 2015). We assumed a negative binomial distribution  
175 after checking for the overdispersion of our dependent variables. To account for the spatial and temporal  
176 dependency, a station (*PlotID*) and a session identifier (*Session*) were incorporated as random factors in the  
177 models. We selected the one with the lowest AICc and, given that the best response was observed at the  
178 large scale (see Appendix B: Table S2), our subsequent analyses were performed using the urbanization  
179 level measured in the buffer of 1,600 m of radius. We then tested whether the total abundance of carabids  
180 and spiders in the isolated patch is influenced by the area and the age of the roundabout and by the  
181 distance of the roundabout from the control patch with the same model structure. No corrections to the  
182 data from the isolated patches were introduced to keep into account the effects of the area and the age of  
183 the roundabout and the distance between the roundabout and the control patch, since both groups were  
184 not affected by these factors (see Appendix B: Table S3).

185 Multivariate statistics. Firstly, we investigated the variation in taxonomic and functional composition in  
186 spider and carabid assemblages by calculating taxonomic and functional  $\beta$ -diversity across the study area.  $\beta$ -  
187 diversity was here intended as dissimilarity among samples and it was calculated by means of the

188 complement of the Sørensen index, which ranges from 0 (samples are composed exactly by the same  
189 species or functional groups) to 1 (samples do not share any species or functional groups). We used the  
190 function “beta” in the *BAT* package (Cardoso, Rigal & Carvalho, 2015), which returns the overall  $\beta$ -diversity  
191 and the contribution of its components, namely *turnover* and *nestedness*.

192 Secondly, we investigated the effects of the urbanization level and patch isolation on taxonomic and  
193 functional composition of both groups with multivariate statistics, performed with the *vegan* package  
194 (Oksanen et al., 2018). We first converted the urbanization level into a categorical variable by assigning  
195 sampling plots to one of three categories of urbanization: low (ISA < 50%), intermediate (ISA = 50-70%) and  
196 high (ISA > 70%). We then created a functional matrix, measuring the community-weighted means (CWM)  
197 of trait values with the function “functcomp” of the *FD* package (Laliberté & Legendre 2010; Laliberté,  
198 Legendre & Shipley, 2014). We built a site-by-trait matrix by multiplying the site-by-species matrix with a  
199 species-by-trait matrix resulting in the CWM trait values, which represent the mean trait value of all species  
200 in the community, weighted by their relative abundances, for body length (CWM-BL), dispersal (CWM-Disp)  
201 and trophic composition (CWM-Trophic). Increasing values in CWM-Trophic indicates a shift towards  
202 phytophagous species in carabids, and towards web-builders in spiders.

203 Taxonomic and functional composition among the three urbanization levels, and between control and  
204 isolated patches, were visualized by means of a Principal Coordinate Analysis (PCoA). We then tested  
205 whether the urbanization level and patch isolation were responsible for either taxonomic or functional  
206 homogenization by means of the Test of Homogeneity for Multivariate Dispersion (Anderson, Ellingsen &  
207 McArdle, 2006). This test measures the distance of each site to its associated group median calculated on a  
208 site-by-site distance matrix and subjects these values to an ANOVA (9,999 permutations) to assess if the  
209 variance differed among groups. For taxonomic diversity, the site-by-site distance matrix was computed on  
210 the matrix of relative abundance of species using the Bray-Curtis distance, whereas the Gower distance was  
211 applied to the CWM matrix to obtain the site-by-site distance matrix for functional diversity. To detect  
212 possible shifts in taxonomic and functional composition among urbanization and isolation levels, we  
213 performed a Permutational Multivariate Analysis of Variance (PERMANOVA; Anderson, 2001), specifying

214 urbanization, isolation and their interaction as factors. Statistical significance was tested via 9,999 random  
215 permutations with the sampling session (*Session*) and the ID of each sampling plot (*PlotID*) as strata.

216 To identify which taxa are specifically linked to the urbanization level and patch isolation categories, an  
217 Indicator Species Analysis was performed through the calculation of the *IndVal* (Indicator Value) index  
218 (Dufrene & Legendre, 1997), using the function “multipatt” in the *indicspecies* package (De Caceres &  
219 Legendre, 2009).

220 Statistical models. The effects of the urbanization level, patch isolation and their interaction on community-  
221 averaged functional traits were tested with Generalized Linear Mixed Models (GLMMs) (Zuur et al., 2009),  
222 performed with the function “glmer” in the *lme4* package (Bates et al., 2015) assuming a normal error  
223 distribution. To account for the spatial and temporal dependency, a plot (*PlotID*) and a session identifier  
224 (*Session*) were incorporated as random factors in the models. The set of functional metrics employed in this  
225 analysis as dependent variables were extracted from the previously calculated functional matrices. In  
226 addition, in order to check for spatial autocorrelation, we calculated the observed Moran’s *I* for each  
227 model, comparing it with the expected values.

228

## 229 **Results**

230 A total of 215 out of 270 pitfall traps were retrieved during the whole sampling season. Overall, we  
231 collected a total of 1,722 carabids, belonging to 52 species, and 4,811 spiders, belonging to 66 species, with  
232 a mean of  $5.28 \pm \text{sd} = 3.49$  (min = 4 and max = 13 species per subplot) and  $6.86 \pm \text{sd} = 3.57$  (min = 1 and max  
233 = 19 species per subplot) species and  $21.8 \pm \text{sd} = 22.4$  (min = 1; max = 126) and  $34.9 \pm \text{sd} = 32.6$  (min = 3;  
234 max = 162) individuals per subplot respectively (see Appendix C: Tables S4-S5 for details on the recorded  
235 species of carabids; and Appendix D: Tables S6-S7 for details on the recorded species of spiders).

236 The pairwise dissimilarity (Fig. 2) among carabid communities (total  $\beta$ -diversity) within the sampling area,  
237 was on average higher for taxonomic ( $0.84 \pm \text{sd} = 0.15$ ) than for functional diversity ( $0.62 \pm \text{sd} = 0.12$ ). The  
238 contribution of the *nestedness* component to total  $\beta$ -diversity was higher than the *turnover* component in  
239 both taxonomic diversity (*turnover* =  $0.34 \pm \text{sd} = 0.26$ ; *nestedness* =  $0.51 \pm \text{sd} = 0.28$ ) and functional diversity

240 (*turnover* =  $0.12 \pm \text{sd} = 0.21$ ; *nestedness* =  $0.51 \pm \text{sd} = 0.28$ ). Similarly, total  $\beta$ -diversity was on average  
241 higher for taxonomic diversity ( $0.77 \pm \text{sd} = 0.17$ ) than for functional diversity ( $0.55 \pm \text{sd} = 0.21$ ) among  
242 spider communities within the sampling area and again the *nestedness* component showed a higher  
243 contribution than the *turnover* one for both taxonomic diversity (*turnover* =  $0.32 \pm \text{sd} = 0.24$ ; *nestedness* =  
244  $0.44 \pm \text{sd} = 0.27$ ) and functional diversity (*turnover* =  $0.10 \pm \text{sd} = 0.10$ ; *nestedness* =  $0.44 \pm \text{sd} = 0.27$ ).

245 The test of homogeneity for multivariate dispersion showed significant functional homogenization of both  
246 groups along the urbanization gradient (Table 1 and Fig. 3), whereas significant taxonomic homogenization  
247 was observed in isolated patches and urbanized plots for spiders (Table 1 and Fig. 4). The PERMANOVA  
248 revealed significant shifts in taxonomic and functional composition along the urbanization gradient for both  
249 groups. Spider taxonomic and functional composition was affected also by patch isolation and the  
250 interaction between urbanization and isolation, whereas carabid taxonomic composition showed a shift  
251 from connected to isolated patches (Table 2).

252 The indicator species analysis highlighted no indicator species for urbanization categories in both groups.  
253 Some indicator species were identified for connected control subplots, namely *Harpalus serripes* (*IndVal* =  
254  $0.562$ ;  $P = 0.001$ ) and *Harpalus tardus* (*IndVal* =  $0.385$ ;  $P = 0.043$ ) for carabids; *Trochosa ruricola* (*IndVal* =  
255  $0.549$ ;  $P = 0.008$ ), *Thanatus arenarius* (*IndVal* =  $0.461$ ;  $P = 0.036$ ) and *Pardosa tenuipes* (*IndVal* =  $0.366$ ;  $P =$   
256  $0.049$ ) for spiders. No indicator species were identified for isolated subplots.

257 The response of carabid community-averaged functional traits revealed a significant increase of the  
258 proportion of phytophagous species with increasing urbanization level (range: 0.01-0.50), whereas no  
259 significant results were recorded for CWM-BL (body size) and CWM-Disp (Dispersal) (Table 3 and Fig. 5).  
260 Regarding spiders, body size was negatively affected by patch isolation, whereas the interaction term  
261 showed a significant positive effect, underlying that CWM-body size of spiders in isolated patches increases  
262 with increasing urbanization (Table 3). Both dispersal and trophic groups were significantly affected by all  
263 terms (Table 3). CWM-Disp (dispersal) showed an increasing trend along the urbanization gradient, ranging  
264 from 2.06 to 2.64, and higher values in the isolated ( $2.26 \pm \text{sd} = 0.25$ ) than in the control ( $2.16 \pm \text{sd} = 0.34$ )  
265 subplots (Fig. 5), while the interaction factor displayed a negative effect. Regarding CWM-Trophic, the

266 relative proportion of web-builders significantly increased along the urbanization gradient (range = 0.27-  
267 0.82) and it was significantly higher in isolated ( $0.68 \pm \text{sd} = 0.27$ ) than in control ( $0.69 \pm \text{sd} = 0.30$ ) subplots,  
268 while a negative effect was observed for the interaction factor (Fig. 5).

269 Lastly, Moran's  $I$  revealed that CWM-Disp and CWM-Trophic, but not CWM-BL, of both carabids and spiders  
270 show significant spatial autocorrelation (Table 4).

271

## 272 **Discussion**

273 In this work, we analyzed how urbanization and fragmentation drive the *turnover* and *nestedness* patterns  
274 of taxonomic and functional diversity within two taxonomic groups of ground arthropods, namely carabids  
275 and spiders. Since we consistently sampled the same habitat type (i.e. semi-natural grasslands) exposed to  
276 the same management practices, we could clearly depict the effect of site location within the conurbation,  
277 without confounding factors due to local habitat characteristics.

278 Our results showed that taxonomic variation was higher than functional variation within both carabid and  
279 spider communities along the urbanization and isolation gradients. This is in accordance with the functional  
280 redundancy concept (Lawton & Brown, 1993) stating that multiple species perform similar roles in  
281 communities and ecosystems (e.g. Petchey et al., 2007). This would guarantee high ecological resilience to  
282 disturbance, which is particularly crucial for the maintenance of ecosystem functions in extremely altered  
283 habitats, like urban areas. Decomposing  $\beta$ -diversity into its components revealed that the same  
284 mechanisms drive variation in taxonomic and functional composition in the two taxa studied. Since  
285 *nestedness* was the dominant component, some sampling plots host a subset of species of other sites. We  
286 can hypothesize that both carabid and spider communities are composed of both specialized species,  
287 surviving only in few localities, and generalist species, which can exploit a broad range of ecological  
288 conditions. In the same way, some functional traits are removed from some sites without being replaced,  
289 with possible repercussions on ecosystem functionality.

290 Results of the test of homogeneity for multivariate dispersion showed that spiders were more affected by  
291 the urbanization level and patch isolation than carabids. Taxonomic homogenization (i.e. *nestedness*) was

292 observed only in spiders and it was explained by both the urbanization level and patch isolation. In  
293 addition, the role of isolation as a driver of taxonomic homogenization was further confirmed by the  
294 indicator species analysis that revealed indicator species for connected control subplots only. According to  
295 these results, we may hypothesize that connected control subplots could provide source populations for  
296 the isolated ones, guaranteeing their survival in disturbed habitats, likely recalling the island biogeographic  
297 model in fragmented landscapes (MacArthur & Wilson, 1967; Leibold & Chase, 2017). However, no  
298 indicator species were identified for the urbanization categories. Indicator species may not have been  
299 detected because they are too scarce and irregularly distributed among the subplots within each category.  
300 This result may however suggest that the observed *nestedness* pattern along the urbanization gradient may  
301 be due to varying relative abundances, instead of an effective substitution of species.

302 Regarding functional composition, the urbanization level causes functional homogenization in both  
303 communities. This may be due mainly to a filtering process, supporting the hypothesis that urbanization  
304 alters the functional community composition by shifting species composition towards more functionally  
305 homogeneous assemblages. Conversely, patch isolation did not significantly affect the functional  
306 composition of both carabid and spider communities, underpinning how the urbanization level alone may  
307 explain functional variation among samples. This corroborates the hypothesis that stochastic events, like  
308 dispersal-related processes or ecological drift, determine species richness in isolated patches for both  
309 carabids and spiders (Chase & Myers, 2011; Leibold & Chase, 2017). These outcomes parallel Concepción et  
310 al. (2017) who found functional homogenization with increasing urbanization in vascular plants and birds.  
311 On the contrary, our results are in contrast with those reported by Brice, Pellerin and Poulin (2017), who  
312 observed both taxonomic and functional differentiation with increasing urbanization in plant communities.  
313 These contrasting results are likely due to the fact that plant diversity often increases in urban habitats  
314 because of the unique abiotic conditions and disturbance regimes, favoring exotic species (McKinney,  
315 2008). In our case, we did not record any exotic species in the examined communities, thus, at least from  
316 this point of view, our results seem to reflect the filtering effect of urbanization without confounding  
317 factors. However, increasing differentiation might also emerge from differential human activities in urban

318 green spaces, facilitating certain species in some places or eliminating others elsewhere. This could have  
319 also happened in our case with spiders and carabid beetles, which could be easily transported in the soil or  
320 in plant pots as eggs, juvenile stages or even as adults.

321 The results of the PERMANOVA performed on taxonomic composition pointed out a combined effect of the  
322 urbanization level and patch isolation on the taxonomic shift—i.e. *turnover* component in both taxonomic  
323 groups. The role of urbanization as a driver of this process has already been pointed out by Knop (2016) for  
324 canopy insects, who observed true species replacement in terms of relative abundances in three insect  
325 groups, including ground beetles, when comparing rural against urban areas. Regarding functional  
326 composition, urbanization caused a shift in both examined groups, whereas patch isolation affected only  
327 spiders, and these trends are confirmed by the analysis of the response of functional traits.

328 Dispersal capacity significantly increased along the urbanization gradient for spiders. In particular, this  
329 increasing trend was clear in connected control subplots, while isolated subplots hosted highly dispersive  
330 species, regardless of the urbanization level. High dispersal capacity is essential in disturbed habitats in  
331 order to escape adverse environmental conditions and to enhance population survival in the area; vice-  
332 versa in stable, late-successional habitats more energy can be invested in reproduction, favoring the  
333 establishment of less dispersive species (Roff, 1975). Hence, the observed decline in low-dispersive species  
334 appears in accordance with the dynamic nature of urban environments, where human activities result in  
335 high turnover rates of suitable habitat patches (Parris, 2016). The observed high dispersal capacity of  
336 spiders in isolated subplots can be explained in the framework of the theory of island biogeography  
337 (MacArthur & Wilson, 1967), which predicts that isolation of suitable patches increases extinction rates.  
338 Therefore, only highly dispersive species can support viable populations in isolated patches, due to the  
339 continuous immigration of new individuals, increasing the average dispersal capacity of the community.

340 Surprisingly, we did not record any effect of the urbanization level or patch isolation on carabid dispersal  
341 capacity, in contrast with the available literature (Piano et al., 2017). The differential response obtained for  
342 the two examined arthropod groups might be due to their different dispersal mode, i.e. active dispersal in  
343 carabids and passive dispersal in spiders. The peculiar dispersal mode of spiders, i.e. ballooning, requires

344 particular physical conditions to occur (Weyman, 1993; Simonneau, Courtial & Pétilion, 2016) and  
345 dispersing individuals have no control over the flight direction (Compton, 2002). In heavily fragmented  
346 landscapes, the probability to reach isolated patches is therefore strictly related to the ballooning  
347 performance, which is maximized in highly dispersive species (Bonte et al., 2003). Conversely, carabid  
348 dispersal, either cursorial or by flight, is active and not constrained by the physical environment, therefore  
349 all species likely have the same probability to reach isolated patches.

350 The analysis of the trophic structure revealed how urbanization strongly acts on both examined taxa.  
351 Regarding carabids, we observed a significant replacement of zoophagous with phytophagous species.  
352 Phytophagous carabids are specialized on seeds from ruderal plants (Thiele, 1977; Honek et al. 2007;  
353 Honek, Martinkova & Saska, 2011), which typically occur in highly impermanent sites (Ribera et al., 2001).  
354 Phytophagous species can therefore be considered adapted to ruderal habitats (Vanbergen et al., 2010),  
355 likely supporting viable populations within urban sites in our study. Concerning spiders, the proportion of  
356 web-builders significantly increases with increasing the urbanization level in connected control subplots.  
357 Conversely, in isolated subplots, web-builder proportion is higher than connected control subplots  
358 regardless of the urbanization level, suggesting how the effect of patch isolation overrides urbanization in  
359 determining the trophic structure of spiders. This may be due to the fact that patch isolation may  
360 negatively affect preys of cursorial spiders (Hawn et al., 2018) with consequent negative effects on hunters  
361 (Gravel et al., 2011; Zalewski et al., 2018). However, it should be pointed out that this shift from hunters to  
362 web-builders in isolated patches might also be a consequence of the dispersal-based selection, since, in our  
363 study, web-builders include most of the highly dispersive species (i.e. linyphiids).

364 Contrary to our expectations, body size did not respond to the urbanization gradient in our study for both  
365 taxonomic groups, in accordance with Buchholtz et al. (2018). However, we observed a significant decrease  
366 of spider body size along the urbanization gradient in connected control patches, but not in isolated ones. It  
367 has been demonstrated that community-wide body-size shifts occur in urban communities as a  
368 consequence of increased temperatures due to the urban heat-island effect, but these shifts are mediated  
369 by the dispersal capacity of each taxon (Merckx et al., 2018). Thus, this lack of response may result from the



370 dispersal-mediated effect, since higher dispersal capacity of spider species recorded in roundabouts may  
371 dampen the urban heat-island effect on body size.

372 Results of the Moran's  $I$  test revealed that there is high spatial autocorrelation among sampling plots in  
373 terms of dispersal capacity and trophic composition, but not of body size, for both groups. These patterns  
374 underlie the role of the spatial arrangement of individuals in a landscape, which is rarely random (McGlenn  
375 et al., 2019). Instead, most individuals are spatially clustered or aggregated in some way, with  
376 repercussions also on the functional traits, as demonstrated by our results.

377

### 378 **Conclusions**

379 Overall, we here highlighted differential mechanisms underlying the selective pressure exerted by  
380 urbanization and fragmentation on ground arthropods, showing a more evident response in spiders than  
381 carabids. This is in accordance with literature, which highlighted how top predators are usually more  
382 sensitive to urbanization than lower trophic levels (Egerer et al., 2017; El-Sabawii, 2018).

383 We have demonstrated that *nestedness* more than *turnover* explains taxonomic and functional variation in  
384 the examined communities, indicating a loss of species and functionality among sampling plots. This  
385 homogenization process is mainly due to urbanization, which filters species based on their functional traits  
386 in both the examined groups.

387 In addition, the high spatial autocorrelation among sampling plots identified for dispersal capacity and  
388 trophic composition underlies that stochastic factors, e.g. source-sink dynamics, also play a role in driving  
389 the functional composition of examined communities. Stochastic events, like those caused by human  
390 activities, may cause the extinction of some species in the examined sampling plots, potentially  
391 compromising the species survival in the whole sampling area. This may have repercussions on ecosystem  
392 functionality if key stone species are removed from the landscape.

393 Our approach, thus, proved to be particularly useful to understand how different facets of urbanization  
394 affect biodiversity, providing a valuable framework to predict how biotic communities will respond to  
395 increasing anthropogenic pressures associated with urbanization. In the next future, further investigations

396 should be performed to highlight alterations at the ecosystem level in order to provide management  
397 suggestions aiming at reducing negative effects caused by urbanization.

398

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403

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407

408 **Appendix A. Supplementary data**

409 **Appendix B. Supplementary data**

410 **Appendix C. Supplementary data**

411 **Appendix D. Supplementary data**

412 **Supplementary data associated with this article can be found, in the online version, at XXXXX**

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602 **FIGURE CAPTIONS**

603 Fig. 1. Schematic representation of the nested sampling design adopted in this research. Sampling plots  
604 were placed along an urbanization gradient, each one being composed of two subplots (I = isolated; C =  
605 connected control). In each sub-plot, three pitfall traps were activated during the sampling period (black  
606 dots).

607 Fig. 2. Barplot representing the contribution of *turnover* (dark grey) and *nestedness* (light grey) to the total  
608 taxonomic and functional  $\beta$ -diversity for carabids (left panel) and spiders (right panel).

609 Fig. 3. Ordination of carabid (left) and spider (right) communities, according to the first two PCoA axes  
610 performed on taxonomic (upper panel) and functional (lower panel) composition. Ellipses represent  
611 standard deviations around the medians of urbanization levels (green = low; orange = intermediate; purple  
612 = high). Different symbols represent sampling subplots belonging to different urbanization levels (circles =  
613 low; squares = intermediate; triangles = high). The animal silhouettes are from PhyloPic  
614 (<http://www.phylopic.org>).

615 Fig. 4. Ordination of carabid (left) and spider (right) communities, according to the first two PCoA axes  
616 performed on taxonomic (upper panel) and functional (lower panel) composition. Ellipses represent  
617 standard deviations around the medians of isolation levels (blue = connected control; brown = isolated).  
618 Different symbols represent sampling subplots belonging to different isolation categories (circles =  
619 connected control; squares = isolated). The animal silhouettes are from PhyloPic  
620 (<http://www.phylopic.org>).

621 Fig. 5. Predicted CWM values (CWM-Trophic = trophic composition; CWM-BL = body length; CWM-Disp =  
622 dispersal) **and confidence intervals** along the urbanization gradient for carabids and spiders (light blue line =  
623 connected control patch data; orange line = isolated patch data). Only significant trends are reported. The  
624 animal silhouettes are from PhyloPic (<http://www.phylopic.org>).

Figure 1

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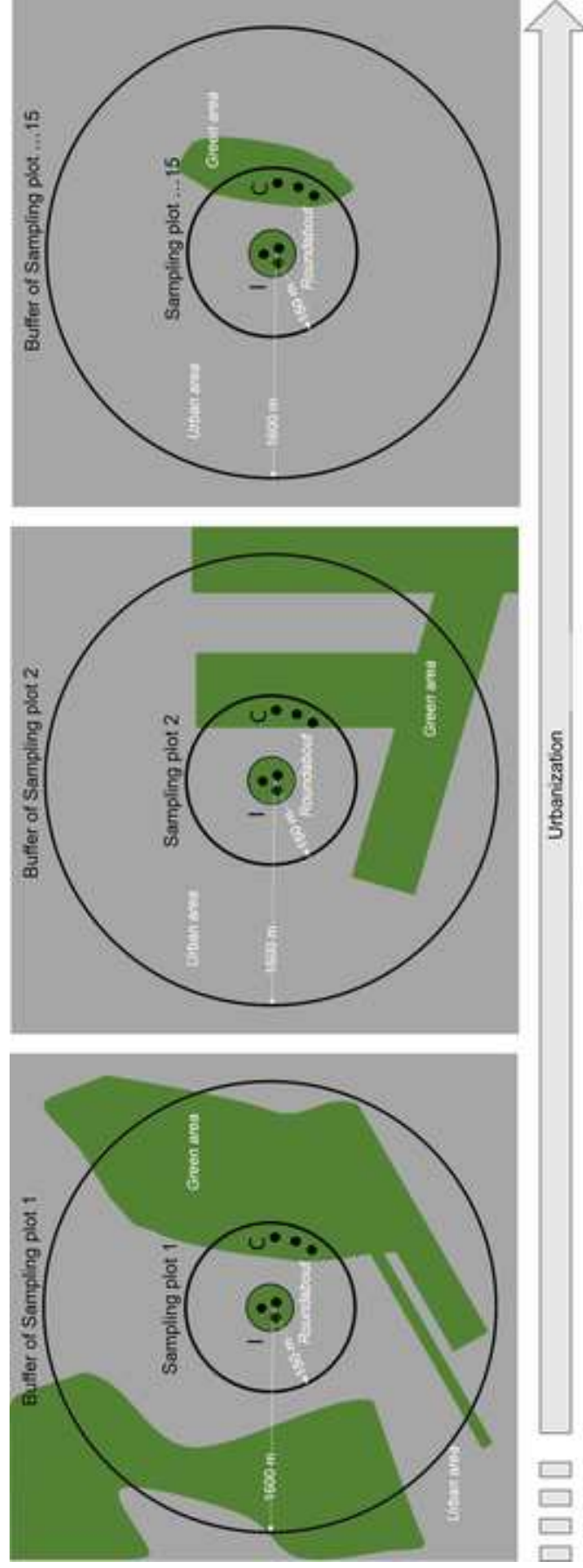


Figure 5

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