



HAL
open science

Domestic gardens as favorable pollinator habitats in impervious landscapes

Marine Levé, Emmanuelle Baudry, Carmen Bessa-Gomes

► **To cite this version:**

Marine Levé, Emmanuelle Baudry, Carmen Bessa-Gomes. Domestic gardens as favorable pollinator habitats in impervious landscapes. *Science of the Total Environment*, 2019, 647, pp.420-430. 10.1016/j.scitotenv.2018.07.310 . hal-02444494

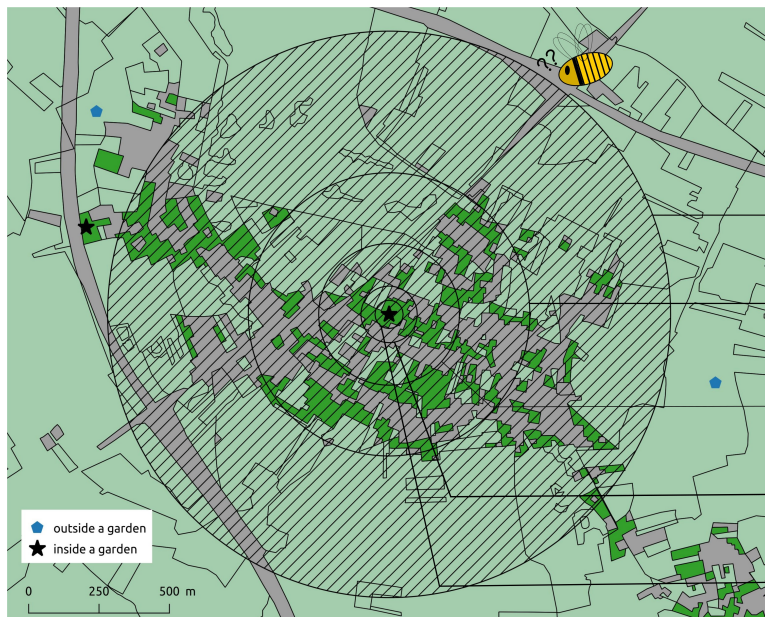
HAL Id: hal-02444494

<https://agroparistech.hal.science/hal-02444494v1>

Submitted on 9 Jan 2024

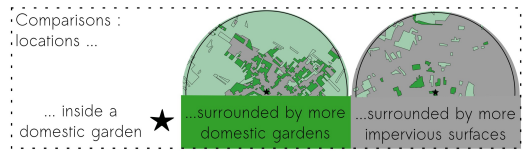
HAL is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.

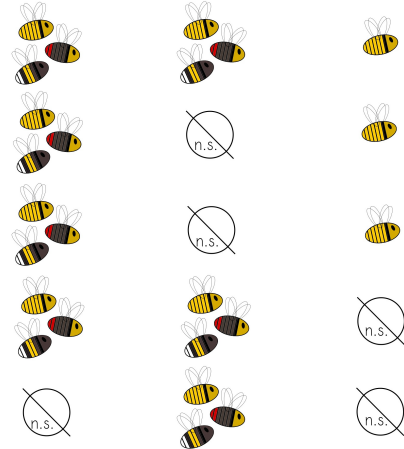


higher pollinator richness...
 lower pollinator richness...

... at 1000m :
 ... at 500m :
 ... at 250m :
 ... at 100m :
 ... at 50m :



Spatial scale



LANDSCAPE

LOCAL

Highlights

- Citizen science provided access to domestic gardens, understudied urban green spaces
- Impervious surfaces limit pollinators presence at landscape level
- Sufficient critical amount of gardens increased pollinator diversity at local scale
- Critical amount of gardens' knowledge may favor coordinated decisions by gardeners
- Pollinators may benefit from patches of domestic gardens in an urban matrix

1 **Abstract**

2 Urban expansion is correlated to negative biodiversity trends. The amount of impervious surfaces in
3 urban areas is a determinant of pollinator species assemblages. While the increase in urbanization
4 and impervious surfaces negatively impacts pollinators, cities also encompass urban green spaces,
5 which have a significant capacity to support biodiversity. Among them, domestic gardens that
6 represent a non-negligible fraction of green spaces have been shown to benefit pollinators.
7 Domestic gardens may form habitat clusters in residential areas, although their value at a landscape
8 scale is still unknown. Here, we investigate the combined effects of impervious surfaces and
9 domestic garden areas on pollinator richness. Due to the difficulty of accessing privately owned
10 domestic gardens, we chose to use citizen science data from a well-established French citizen
11 science program known as SPIPOLL. Using regression tree analysis on buffers located from 50m to
12 1000m around the data points, we show the importance of pollinators being in close proximity to
13 domestic gardens as locally favorable habitats that are embedded within a landscape, in which
14 impervious surfaces represent unfavorable areas. We highlight the inter-connection between local
15 and landscape scales, the potential for patches of domestic gardens in residential areas, and the need
16 to consider the potential of gardeners' coordinated management decisions within a landscape
17 context.

18

19 **Keywords:** pollinators, domestic gardens, impervious surfaces, citizen science

20 1 Introduction

21 Urban areas now contain more than half of the world's population and will continue to grow
22 (United Nations, 2014). As urban expansion leads to an increasing number of former natural and
23 semi-natural areas becoming urbanized, urbanization is now considered to be a major threat to
24 biodiversity (Grimm et al., 2008). General findings stress the negative impact of urban areas on
25 biodiversity through habitat loss, reduced habitat quality, and habitat homogenization (McKinney,
26 2008), as well as their impact on native species extinction (Czech et al., 2000). The increasing
27 proportion of impervious surfaces in urban areas is a possible proxy for urbanization level. It is also
28 a major determinant for several species assemblages, such as bees (Fortel et al., 2014; Geslin et al.,
29 2016) and amphibians (Parris, 2006), as well as for urban ecosystem functions, such as water
30 resources and flow regulation (Arnold and Gibbons, 1996).

31 Cities encompass urban green spaces (UGS), such as cemeteries, urban wasteland, public gardens,
32 community gardens, and domestic gardens (Lepczyk et al., 2017). These UGS account for a
33 variable percentage of a city's area, ranging from 2 to 46% in European cities (Fuller and Gaston,
34 2009). Their ability to support biodiversity has been recently acknowledged (Aronson et al., 2014;
35 Beninde et al., 2015), and there is now a call to effectively integrate UGS in biodiversity planning
36 and management to ensure their full inclusion in biodiversity conservation (Lepczyk et al., 2017).
37 They may constitute a diversification of land usages given the general impervious surface and thus
38 support increased levels of biodiversity (McKinney, 2008). Moreover, UGS benefit human health
39 and well-being (Tzoulas et al., 2007).

40 Domestic gardens are an understudied type of UGS, mainly because of their limited accessibility to
41 researchers and their supposed non-relevance to conservation (Cameron et al., 2012; Cook et al.,
42 2012; Goddard et al., 2010). Yet domestic gardens may account for a large part of UGS and are thus

43 worth considering in terms of their contribution to biodiversity conservation. Their estimated areas
44 in cities vary from 16% in Stockholm, Sweden (Colding, 2007) to 22-27% in the UK (Loram et al.,
45 2007) and 36% in Dunedin, New Zealand (Mathieu et al., 2007). Their distribution is heterogeneous
46 within cities and surrounding regions: in Flanders, there is a lower concentration of gardens in city
47 centers, but a higher proportion in the areas surrounding the centers and peri-urban areas
48 (Dewaelheyns et al., 2014). Various organisms have been found to benefit from urban or peri-urban
49 domestic gardens (Goddard et al., 2010), such as birds (Daniels and Kirkpatrick, 2006; van Heezik
50 et al., 2008) and invertebrates (Smith et al., 2006a, 2006b; Sperling and Lortie, 2010), including
51 pollinators (Pardee and Philpott, 2014).

52 In this study, we chose to focus on pollinators because of their role in ecosystem functioning (e.g.
53 Potts et al., 2010 but also Kleijn et al., 2015), but also because of their adaptation to urban
54 environments and the challenge associated with the low mobility of many small solitary bee species
55 (Greenleaf et al., 2007; Zurbuchen et al., 2010). We thus consider pollinator richness as a surrogate
56 for domestic gardens biodiversity. Indeed, grouped domestic gardens, i.e. patches, may represent
57 more favorable habitats to pollinators with small flight ranges, as they are able to take advantage of
58 the nearby resources, either inside the garden or in adjacent gardens (Lerman et al., 2018). As urban
59 perturbation is particularly high on small spatial scales and eliminates close living species
60 (McKinney, 2008), suitable habitat patches such as domestic gardens could serve as refuges for
61 pollinators. Hinners et al. (2012) found that resources are insufficient to maintain high pollinator
62 diversity in suburban habitats less than 80,000m², while in habitats around 200,000m², richness was
63 comparable to semi-natural areas: a threshold value for pollinator conservation might lie between
64 these two figures. However, these areas are already considerably greater than the average domestic
65 garden size, estimated to be 571m² in Belgium (Dewaelheyns et al., 2014) and 190m² in the UK
66 (Davies et al., 2009). Yet the peripheries of many Western cities consist of extended suburban areas

67 comprising residential areas with detached houses and private gardens. Consequently, the combined
68 surface of neighboring domestic gardens might attain the threshold value of habitat patch size.
69 When identifying actions to reduce the impact of urbanization on pollinators, the attained threshold
70 surface may be an indicator of their efficiency.

71 In this context, pollinators are an interesting choice when studying domestic gardens as their decline
72 is made visible to citizens, and individual actions in favor of pollinators are accessible to garden
73 owners. While urbanization adversely affects pollinators by destroying floral resources and nesting
74 sites (McKinney, 2002), the installation of “bee hotels” (artificial structures with materials that bees
75 can use as a nesting site, such as wooden blocks with holes, paper tubes, etc.) has a variable impact
76 on pollinator richness and abundance depending on the pollinator species (Gaston et al., 2005;
77 MacIvor and Packer, 2015). The planting of pollinator-friendly flowers likewise has a variable
78 impact depending on the chosen flower species and targeted insect species (Garbuzov and Ratnieks,
79 2014; Salisbury et al., 2015). A more precise determination of the scale of effect of domestic garden
80 patches on pollinator richness in peri-urban areas would make an important contribution to
81 biodiversity planning and management.

82 Because of their privately owned status, domestic gardens are not easily accessible and are thus less
83 often the subject of research compared to other types of UGS (Hernandez et al., 2009). While
84 obtaining regular access to gardens is difficult, long-term data gathering from domestic gardens is
85 still possible through citizen science programs. The French citizen science program known as
86 SPIPOLL was launched in 2010 (Deguines et al., 2012) by the National Museum of Natural History
87 (MNHN) and Office for Insects and their Environment (OPIE) with a focus on flower-visiting
88 insects, most of which are insect pollinators. Using a short protocol, SPIPOLL allows participants to
89 take photographs of insects seen on flowers and send them to an internet database. The collected

90 photographs result in an understanding of insects and their land-use preferences (Deguines et al.,
91 2012).

92 SPIPOLL is a nation-wide program, although we choose to focus on the Île-de-France region in this
93 study. Île-de-France is a densely populated region and is representative of urban areas in Western
94 industrialized countries with their organization around a metropolis. The Parisian metropolis is
95 located approximately in the center of the Île-de-France region and is surrounded by successive
96 urban belts with decreasing urbanization, with a higher urban concentration around transportation
97 networks (Fig. 1). Semi-detached or detached houses surrounded by domestic gardens are more
98 frequent on the Paris periphery. Overall, the region allows us to study an urbanization gradient with
99 a variable proportion of built-up, residential, and garden areas.

100 Determining a threshold for cumulative domestic garden areas that benefit pollinators relative to
101 urban impervious surfaces on a given scale could help urban planners in the decision-making
102 process. Moreover, regarding citizen science programs such as SPIPOLL, participants' knowledge
103 of this threshold and the geographic situation of their garden could help them to better appreciate
104 pollinator diversity as well as local pressures, and thus consider this diversity relative to the local
105 urbanization stage and processes, especially as the inhabitants may also experience these to some
106 extent.

107 The present study aims to understand the effect of urbanization and domestic gardens on pollinator
108 richness and their relative importance. It includes several spatial scales relevant to the flight
109 distances of pollinators and the size of domestic garden patches in peri-urban residential areas. Our
110 hypotheses are as follows: (1) the effects of gardens on pollinator richness will be limited in densely
111 urbanized areas, in which domestic gardens may not be determinants of pollinator richness; and (2)
112 in areas where domestic gardens do have an influence on pollinator richness, the latter will be
113 higher in areas with a low proportion of impervious surfaces and a high proportion of gardens.

114 2 Methods

115 The use of a French citizen science program focusing on pollinators allowed us to gather a large
116 amount of data from locations that are usually difficult to access, i.e. domestic gardens. As we used
117 citizen science data, several factors out of our control and irrelevant to this study may be related to
118 data variability, such as temperature and cloud coverage (see Data selection method section below).
119 Likewise, data location depended on the participants' decision to contribute to the program and
120 resulted in spatial autocorrelation. To account for these factors, we used a two-step methodology to
121 answer our research question. We initially standardized pollinator data via a first modeling step (see
122 Data standardization method section below). We then used the residuals of this first modeling step
123 to analyze the dependency of pollinators on the variables of interest: impervious surface proportion,
124 domestic garden surface proportion, and location of data points inside the gardens (see Regression
125 tree modeling method section below).

126 2.1 Diversity of flower-visiting insect data and standardization

127 2.1.1 Data selection

128 The SPIPOLL program started in 2010 and has already led to several publications (e.g. Deguines et
129 al., 2012; Deguines et al., 2016; Desaegher et al., 2018). The SPIPOLL protocol is as follows:
130 participants have to take photographs of insects visiting a given flower species over a period of at
131 least 20 min. They then identify the insects using an online key based mostly on morphological
132 features and upload one photograph per identified insect to the dedicated website
133 (www.spipoll.org). Identification is conducted at the level of morphotypes, representing groups of
134 individuals at six various taxonomic levels: a species, a species from a genus, a genus, a species
135 from different genera, several genera within a family, and a whole family. SPIPOLL encompasses
136 630 different morphotypes, of which 95.4% are considered to be pollinators. Thus, even if the
137 program takes into account all flower-visiting insects, it mostly focuses on pollinators, a term that

138 we will subsequently use. A set of all photographs taken during the same session is called a
139 collection. Collections are spatially located by participants when they upload the photographs, and
140 the flower species of the collection, or at least the flower family, is recorded. Because of the design
141 of the SPIPOLL program, collections are considered to be units of observation. They will therefore
142 be used as the unit for richness analysis in this study. Insect identifications are verified by several
143 professional entomologists (see Deguines et al., 2016). Additional information is also gathered:
144 wind intensity on a five-point scale from no wind to strong and continuous wind, temperature in
145 four categories (<10°C, 10-20°C, 20-30°C, >30°C), sky cloud coverage in four categories (0-25%,
146 25-50%, 50-75%, 75-100%), and presence of shade on the observed flower (yes/no). A full
147 description of the sampling protocol may be found in Deguines et al. (2012).

148 We obtained all data points recorded in Île-de-France from 2010 to 31 October 2017. We selected
149 data points with wind intensity belonging to the first four of five categories, temperatures of 10-
150 20°C, 20-30°C, and >30°C, and no shade on the flower. We excluded data points with an
151 observation period of less than 20 min. We only retained data points for which the flower belonged
152 to a family appearing more than 30 times in the dataset (Fig. 2). We also verified for suspicious data
153 points, i.e. those with a particularly high number of insects recorded compared to the observation
154 time. We finally obtained 2470 data points.

155 SPIPOLL participants are free to observe pollinators anywhere, not only in their gardens. We thus
156 identified data points recorded in domestic gardens using a Geographic Information System map
157 reporting land use categories over the Île-de-France region, the MOS database (Institut
158 d'Aménagement et d'Urbanisme, 2012). Data points located within the MOS category of
159 “individual habitat gardens” were labeled as “inside gardens” and the remaining as “outside
160 gardens,” creating a dichotomous garden local context variable (1/0). We also included data points
161 located in a 20m buffer zone around the MOS category of “individual habitat gardens,” as

162 participants have to locate the session on a map, which may lead to inaccurate locations if a large-
163 scale view is used.

164 As species identification was conducted at a morphological level, we calculated the morphotype
165 richness for each data point and standardized it per 20 min in order to account for variable
166 observation duration.

167 **2.1.2 Data standardization**

168 We observed spatial autocorrelation within data (see Supplementary Fig. S1), probably due to
169 participants' tendency to record collections around their habitation. As collections recorded near an
170 inhabitant's garden may have had quite a different context compared to collections recorded inside a
171 garden, we retained all data points instead of excluding closely located data points. In order to
172 manage autocorrelation, we built a generalized least square (GLS) model using the log of
173 morphotype richness per 20 min as the dependent variable, because of its skewed distribution. We
174 included latitude and longitude in an exponential spatial autocorrelation structure and added
175 weather variables (i.e. temperature, cloud coverage, wind intensity), temporal variables (i.e. hour,
176 week, year), flower family, and observer identity. As one aim of this study was to disentangle the
177 local effect of the collection location in a garden from the larger effects of land use around the
178 collection, we also used the dichotomous garden local context variable to indicate whether
179 collections were located inside (1) or outside (0) a garden. We built one GLS model including this
180 garden local context variable and another excluding it, hereafter respectively referred to as the
181 models standardized and unstandardized for local context. Latitude, longitude, week, and hour were
182 numerical variables. Cloud coverage, wind intensity, temperature, flower family, year, and garden
183 were categorical variables. We assumed that all these variables had a linear effect, while we added a
184 quadratic effect for hour and week based on the known seasonal and diurnal patterns of pollinators
185 (Herrera, 1990; Knop et al., 2017; Lefebvre et al., 2018). We initialized spatial autocorrelation

186 structure parameters after preliminary analysis of the data semi-variogram. We chose values that
187 could capture short range effects: 400m for range effect, i.e. accounting for a leverage of
188 autocorrelation effect with distance; and 0.9 for nugget effect, i.e. accounting for data which
189 structure has a shorter range than sampling interval. We used the GLS function provided by the
190 nlme package for R (Pinheiro et al., 2017). Residual spatial autocorrelation was verified using
191 Moran's I and along 100m steps for 20 intervals, which covered the full range of the chosen
192 environmental buffers.

193 2.2 Landscape structure and composition

194 We used a geographic information system (GIS) to treat land use data. We also used the MOS
195 database for general land use in Île-de-France (IAU). MOS covers the Île-de-France region with a
196 25m resolution and is regularly updated every 4-5 years. The present study uses the last updated
197 version from 2012. MOS categories include 81 items in the most precise version; we used a version
198 with 15 categories.

199 We extracted permeable areas and built impervious surfaces using Quantum-GIS (QGIS
200 Development Team, 2016) in several buffers around the collections' data points. Buffer sizes were
201 chosen based on the literature on wild bee flight distances (Zurbuchen et al., 2010): 0m, 100m,
202 250m, 500m, and 1000m. Forests, artificial open spaces, water, farming areas, semi-natural areas,
203 and domestic gardens were classified as permeable surfaces. Transport networks, facility areas,
204 collective and individual housing areas, activity areas, building sites, and landfills were classified as
205 built impervious surfaces (Fig. 3).

206 We computed correlations between the proportion of impervious surfaces, the proportion of
207 domestic gardens, and the inside or outside garden location (1/0). We used the Spearman correlation
208 method for correlations between the proportion of surface variables and the local context variable.

209 2.3 Regression tree modeling

210 We used the residuals from the models both standardized and unstandardized for local context to fit
211 the regression trees (see Supplementary Fig. S2 for the validation of residuals). Here, we chose to
212 fit the regression tree models to residuals instead of using an alternative method with spatial
213 autocorrelation to control for confounding variables, because the regression tree allows threshold
214 effects to be modeled. For each buffer and both sets of residuals, we repeated the regression tree
215 construction 500 times, each time using a randomly selected subsample of 2070 data points out of
216 2470. We then selected the most frequent tree structure and extracted all corresponding regression
217 trees from the pool of 500 (see Supplementary Tables S1 and S2). The variables included in the
218 regression trees were the proportion of impervious and garden areas. Regression trees using the
219 residuals unstandardized for local context also included the local garden context variable
220 (categorical: 0/1). We chose to retain both the domestic garden proportion and the dichotomous
221 local context garden variable, because even if a collection was recorded in close proximity to a
222 domestic garden, the precise location might, for instance, be a street or public space situated nearby.
223 These two collections would have different pieces of information related to the contrasting contexts:
224 for instance, management decisions taken by either an inhabitant or the municipality. To better
225 understand the contribution of all spatial scales to morphotype richness, we repeated the regression
226 tree construction while taking into account the proportion of garden and built-up areas on all scales
227 as well as the garden variable for the tree using residuals unstandardized for local context. To be
228 selected to define a node, a variable had to meet the 0.05 criteria of significance. We used the
229 partykit package from R (Hothorn and Zeileis, 2015; Hothorn et al., 2017).

230 We calculated the threshold values for nodes, the number of data points included in the terminal
231 nodes, and the associated mean morphotype richness as mean values for all regression trees

232 corresponding to the most frequent one. We compared the mean morphotype richness of nodes in all
233 selected trees using the Mann-Whitney-Wilcoxon test to assess the significance in the final tree.
234 We also retrieved the collections by terminal nodes for spatial scales, which appeared in the trees
235 that took into account all spatial scales. For each node, we calculated the proportion of insects
236 belonging to the different orders and compared the repartition of insect orders between terminal
237 nodes using chi squared tests for the main insect orders in the dataset: i.e. Hymenoptera,
238 Lepidoptera, Diptera, Hemipteran, and Coleoptera.
239 All statistical analyses were conducted in R (R Development Core Team, 2017).

240 3 Results

241 3.1 Diversity of flower-visiting insect data description

242 3.1.1 Dataset description

243 The overall dataset contained 2470 collections recorded by 261 citizen scientists on 18 different
244 flower families. On average, morphotype richness per 20 min was 7.51 +/- 6.12 morphotypes. A
245 summary of the observed insect orders and observed morphotypes per order is given in Table 1.
246 Considering each weather variable independently, collections were most frequently associated with
247 low and irregular wind intensity, 0-25% cloud coverage, or a temperature range of 10-20°C. When
248 all weather variables were combined, the greatest number of collections (17.7%) was made during
249 low wind intensity, with 0-25% cloud coverage, and a temperature range of 20-30°C. Asteraceae
250 flowers were the most sampled family of flowers, being found in 32.1% of collections. A detailed
251 distribution of collections according to flower family is provided in Figure 2.

252 3.1.2 Effects of standardization variables

253 Both standardized and unstandardized models had similar results. Both included a positive effect of
254 50-75% cloud coverage in comparison to 0-25% on the log of morphotype richness ($\beta=0.12$,
255 $p=0.0028$ and $\beta=0.12$, $p=0.0028$ in the models standardized and unstandardized for local context,

256 respectively). Temperatures between 20°C and 30°C had a significant positive effect compared to
257 those between 10°C and 20°C ($\beta=0.067$, $p=0.047$ and $\beta=0.067$, $p=0.047$ in the models standardized
258 and unstandardized for local context, respectively). The years 2011 to 2017 had a significant
259 positive effect on richness (β s between 0.26 and 0.50, all $p<0.01$ and β s between 0.26 and 0.50, all
260 $p<0.01$ in the models standardized and unstandardized for local context, respectively). Week had a
261 significant positive effect on richness ($\beta=0.87$, $p<0.0001$ and $\beta=0.87$, $p<0.0001$ in the models
262 standardized and unstandardized for local context, respectively), while its quadratic term had a
263 significant negative effect on richness ($\beta=-0.0015$, $p<0.0001$ and $\beta=-0.0015$, $p<0.0001$ in the
264 models standardized and unstandardized for local context, respectively). Latitude, longitude, and
265 other categories of factorial variables did not have a significant effect on morphotype richness.
266 Likewise, the local garden context variable did not have a significant effect compared to locations
267 situated outside gardens in the model standardized for local context.

268 Several flower families had a significant effect on morphotype richness in comparison to the
269 Asteraceae reference family. However, as this was not the main interest of the study, detailed
270 statistics on the effect of flower families will not be given here. In short, Apiaceae and Araliaceae
271 had a significant positive effect on morphotype richness compared to Asteraceae. Boraginaceae,
272 Caprifoliaceae, Crassulaceae, Fabaceae, Geraniceae, Lamiceae, Papaveraceae, Plantaginaceae,
273 Primulaceae, and Scrophulariaceae, which had a significant negative effect on morphotype richness.
274 Autocorrelation strongly decreased in both GLS model residuals and had only a minimal effect on
275 the first 100m interval (see Supplementary Fig. S3).

276 3.2 Relations between environmental variables

277 The proportion of impervious surfaces and proportion of domestic gardens were correlated at
278 several spatial scales: 50m ($p<0.0001$, $\rho=0.158$), 100m ($p<0.0001$, $\rho=0.0962$), and 1000m
279 ($p=0.00013$, $\rho=-0.0769$). The proportion of domestic gardens and location inside or outside a garden

280 (1/0) were correlated at the same spatial scales: 50m ($p < 0.0001$, $\rho = 0.872$), 100m ($p < 0.0001$,
281 $\rho = 0.715$), and 1000m ($p = 0.0011$, $\rho = -0.0659$). The proportion of impervious surfaces and location
282 inside or outside a garden (1/0) were correlated at several spatial scales: 50m ($p < 0.0001$, $\rho = 0.150$),
283 250m ($p < 0.0001$, $\rho = -0.178$), 500m ($p < 0.0001$, $\rho = -0.294$), and 1000m ($p < 0.0001$, $\rho = -0.402$). Other
284 correlations were not significant. A description of ranges for the proportion of impervious surfaces
285 and garden areas is given in Table 2.

286 3.3 Selected trees after bootstrapping

287 Selected trees were represented by 42.2% to 82.4% of the pool of 500 regression trees depending on
288 the buffer scale and the model standardized or unstandardized for local context. Details of all
289 regression tree structures observed in the pool of 500 trees are provided in Supplementary Tables S1
290 and S2 for the models standardized and unstandardized for local context, respectively.

291 Tree structures resulted from only one division, leading to trees with two nodes, or from two
292 divisions, a first-order division leading to two branches and a second-order division in one of the
293 branches, leading to a final number of three nodes.

294 3.4 Role of spatial scales and environmental variables in tree structures

295 In the regression trees using residuals from the model unstandardized for local context, the
296 consideration of all spatial scales resulted in the proportion of garden areas in the 50m buffer being
297 the first structuring variable at a threshold of 29.3 +/- 2.8% (Fig. 4). When focusing on the tree
298 using variables of the 50m buffer only, this variable remained the first structuring variable at a
299 threshold of 29.4 +/- 3.0% (Fig. 5A). Regression trees for 100m and higher buffers all resulted in
300 the local garden context variable (0/1) as a structuring variable. At the buffer spatial scale of 100m
301 (Fig. 5B), the garden context (0/1) was the first structuring variable, while the proportion of garden
302 area was the second-order division. At the buffer spatial scales of 250m and 1000m (Fig. 5C, E), the
303 garden context (0/1) was the first structuring variable, while the proportion of impervious surface

304 was the second-order division in the garden context 1 branch. This was reversed at the buffer spatial
305 scale of 500m (Fig. 5D).

306 As the local garden context variable appeared in all trees except two when using residuals from the
307 model unstandardized for local context, it may have masked the effect of the proportion of garden
308 area variable. Regression trees using residuals from the model standardized for local context helped
309 disentangle this.

310 In the regression trees using residuals from the model standardized for local context, taking all
311 spatial scales into account resulted in a tree similar to when the local garden variable was included:
312 the proportion of garden area in the 50m buffer was the first and only structuring variable at a
313 threshold of 29.7 +/- 3.8%. At a spatial scale of 100m, the proportion of garden area was also the
314 first structuring variable at a threshold of 22.6 +/- 2.24%. The branch with a low garden proportion
315 was further divided, again with the proportion of garden areas as a structuring variable at a
316 threshold of 6.38 +/- 1.43%. At spatial scales of 250m and higher, the proportion of impervious
317 surface was the first structuring variable. It was also the only structuring variable except at the
318 1000m spatial scale, which involved the proportion of garden areas in the low impervious surface
319 branch in a second-order division. Regression trees using residuals from the model standardized for
320 local context can be found in Supplementary Figure S4.

321 Table 3 summarizes the steps in the regression trees at which the proportion of the garden area, the
322 local garden context variable, and the proportion of impervious surfaces had an effect.

323 3.5 Dependence of pollinator richness on spatial scales and environmental 324 variables

325 Maximal morphotype richness was reached in nodes attached to the branch with a higher proportion
326 of garden areas or a lower proportion of impervious surfaces in the two-node trees (Fig. 5,
327 Supplementary Fig. S4). In the three-node trees, we observed that a node with maximal morphotype

328 richness could not always be identified at a 0.05 significance level. However, such a node could be
329 identified in trees with a spatial scale of 250m, 500m, and 1000m using residuals from the model
330 unstandardized for local context (Fig. 5) and in trees with a spatial scale of 100m and 500m using
331 residuals from the model standardized for local context. In all but one of the five concerned trees,
332 maximal morphotype richness was reached in a node resulting from a low proportion of impervious
333 surfaces combined with either a high proportion of garden areas or a location inside a garden (local
334 garden context variable = 1). In the last tree, using residuals from the model unstandardized for
335 local context at a spatial scale of 500m (Fig. 5E), maximal morphotype richness was reached in a
336 node resulting from a combination of location inside a garden and a high proportion of impervious
337 surfaces.

338 3.6 Repartition of orders between terminal nodes in trees with the domestic garden 339 proportion variable

340 The proportion of domestic gardens was a significant explanatory variable in regression trees taking
341 into account all spatial scales and using residuals from both models. It was significant at the 50m
342 spatial scale only in these regression trees, which had two terminal nodes according to a lower or
343 higher proportion of domestic gardens at 50m.

344 In both regression trees, the patterns of response of insect orders were similar. The terminal node
345 determined by a proportion of domestic garden less than 29.4% in the tree based on the
346 unstandardized model (compared to 29.7% in the tree based on the standardized model)
347 encompassed 2217 collections (compared to 2218 in the tree based on the standardized model),
348 while the proportions of insects belonging to Lepidoptera ($\chi^2=23.85$, $p<0.001$ and $\chi^2=23.4$,
349 $p<0.001$ for the unstandardized and standardized models, respectively) and Diptera ($\chi^2=20.9$,
350 $p<0.001$ and $\chi^2=21.9$, $p<0.001$ for the unstandardized and the standardized models, respectively)
351 were lower in this node compared to the node determined by a proportion of domestic gardens

352 exceeding 29.4% (29.7% in the standardized model). The low domestic garden proportion node also
353 encompassed a higher proportion of insects belonging to Coleoptera ($\chi^2=53.2$, $p<0.001$ and
354 $\chi^2=52.7$, $p<0.001$ for the unstandardized and the standardized models, respectively) compared to
355 the high domestic garden proportion node.

356 4 Discussion

357 The present study considered impervious surfaces and domestic garden areas together in order to
358 understand their relative effects on pollinator richness. As domestic gardens are difficult to access
359 and study, but nevertheless important places where various and numerous pro-biodiversity actions
360 may take place, understanding their role is relevant. By introducing a garden context variable, we
361 were also able to retain information about the local context of data points. Indeed, domestic garden
362 proportions were not always correlated to the garden context variable, and thus both variables
363 retained information about the situation of data points relative to domestic gardens.

364 4.1 Local and landscape determinants of pollinator richness

365 Maximal morphotype richness of pollinators was associated with data points that were located
366 inside gardens, surrounded by a higher proportion of gardens, surrounded by a lower proportion of
367 impervious surfaces, or a combination of these criteria. This is consistent with previous studies
368 investigating the effect of urbanization on pollinators: urban areas have negative effects on both
369 wild bee (Geslin et al., 2016) and moth richness (Bates et al., 2014) and more generally limit plant-
370 pollinator interactions (Desaegher et al., 2017). The mean morphotype richness of pollinators found
371 in this study – 7.51 morphotypes per 20 minutes – is also consistent with a previous measure based
372 on the SPIPOLL database and focusing on eight plant species: 4.47 morphotypes per 20 minutes
373 (Deguines et al., 2016).

374 We did not include finer scales of pollinator diversity in the regression tree analysis, as total
375 richness is an appropriate surrogate for diversity when dealing with citizen science data (Kremen et

376 al., 2011). The consistency of our results at different spatial scales nevertheless indicates a general
377 pattern of pollinator richness in the densely urbanized Île-de-France region, even if responses of
378 individual species may vary.

379 In spite of its high levels of urbanization, Île-de-France contains forests and farming areas as well as
380 semi-natural areas and other UGS. Farming areas usually have less pollinator richness than urban
381 residential areas (Baldock et al., 2015), while urban parks, natural areas, and urban remnants have
382 more pollinator richness than residential and domestic garden areas (Hostetler and McIntyre, 2001;
383 Matteson et al., 2013; Tommasi et al., 2004). We did not directly compare the impact of domestic
384 garden areas with that of other permeable land use areas known to favor pollinators, but our results
385 still provided insights into these variations: the high morphotype richness found in data points
386 inside gardens and within higher impervious surface proportion at the 1000m scale may indicate
387 that non-impervious surfaces are not all equivalent. We may here point to the role of farming areas.
388 At the 1000m scale, farming areas, which are negatively related to pollinator richness (Baldock et
389 al., 2015), may have been largely included in the buffers. Indeed, the Île-de-France region, apart
390 from dense urbanization, includes large areas of conventional wheat farming. A negative effect of
391 these farming areas might be related to the observed higher richness inside gardens and within areas
392 of higher impervious surfaces, i.e. away from farming areas. Such data points with high morphotype
393 richness inside gardens and within areas of higher impervious surfaces at the 1000m scale are not
394 numerous ($n=75 \pm 11$ depending on the bootstrapped tree). They may also represent a few small
395 areas of good habitat within a large matrix of non-favorable habitat, with the mean richness
396 reflecting the situation of a single almost inaccessible favorable patch (Steffan-Dewenter et al.,
397 2002).

398 4.2 Domestic gardens as favorable habitats within less favorable impervious 399 surfaces

400 Garden area effects, i.e. effects of being surrounded by domestic gardens, were only seen at smaller
401 scales, while impervious surface effects, i.e. effects of being surrounded by impervious surfaces,
402 were at larger scales. The garden variable, always representing a local effect, appeared to be a
403 strong structuring variable at almost all spatial scales. This variable conveys a complementary
404 significance regarding the proportion of domestic gardens, as both variables are not correlated at the
405 intermediate spatial scale, i.e. 250m and 500m. Being located in a domestic garden seems to be
406 always favorable at nearly all spatial scales, while being surrounded by domestic gardens exerted a
407 positive effect only at small spatial scales, i.e. 50m and 100m. Impervious surfaces seem to act as
408 strong determinants for pollinator richness, likely determining the richness found at a given place
409 according to pollinator affinity for impervious and urban areas (Deguines et al., 2012). At large
410 scales, a patch of favorable local habitat may not be reached if it is surrounded by an unfavorable
411 landscape, as shown for butterflies (Bergerot et al., 2011). Once a determined diversity of
412 pollinators is hosted by a region encompassing impervious surfaces, gardens are more likely to host
413 a greater proportion of this diversity compared to other areas in the same region.

414 Pollinator species perceive landscape structures at different spatial scales depending on their
415 dispersion or flight ranges (Keitt et al., 1997; Steffan-Dewenter et al., 2002; With et al., 1999).
416 Analyzing several Apidae groups relative to the features of numerous spatial scales, Steffan-
417 Dewenter et al. (2002) found solitary bee richness to be positively correlated to the proportion of
418 semi-natural areas up to 750m, contrary to honeybee and bumblebee richness, with this pattern
419 being reversed over 750m. Here, the 50m spatial scale appeared to be significant when all spatial
420 scales were analyzed together, while changes in the dominant structuring variables occurred at a
421 250m spatial scale when spatial scales were analyzed separately. The effects of local and landscape

422 levels are likely to rely on different species responses: local features, which disperse in larger
423 scales, may be quite important for some species.

424 The type of identification used in the SPIPOLL program does not allow this finer taxonomic scale
425 of pollinator diversity to be studied. It also results in a lower diversity of observable taxonomic
426 groups, which may have consequently led to lower variability in the data to some extent. This may
427 be viewed as positive, as we were nevertheless able to show significant and consistent patterns.

428 While citizen scientist protocols have been shown to overestimate rare species and underestimate
429 common ones, the amount of data collected is likely to compensate for this trend (Gardiner et al.,
430 2012), and the use of a computer-aided identification tool, here the SPIPOLL online identification
431 key, ensures reliable identification and thus consistent sorting of taxa across citizen scientists and
432 stable morphotype categories (Obrist and Duelli, 2010). The measures of morphotype richness
433 taken by citizen scientists have been previously validated for the detection of changes in abundance,
434 richness, or similarity over space and time (Kremen et al., 2011), allowing for ecological patterns to
435 be identified (Hochachka et al., 2012) and SPIPOLL citizen scientists to improve their knowledge
436 about pollinator identification while participating (Deguines et al., 2018). Morphotypes themselves
437 are relevant for the calculation of ecological indicators, such as pollinator diversity (Obrist and
438 Duelli, 2010). Moreover, the citizen science program has economic advantages, as the field work
439 needed in order to collect an equivalent pool of data would have required a large amount of money;
440 indeed, the SPIPOLL program has much lower costs, with estimated annual savings between
441 678,523€ and 4,415,251€ in France (Levrel et al., 2010). Our use of the SPIPOLL citizen science
442 database allowed us to gather a large amount of data (n=2470 data points), including inside
443 domestic gardens, which are less easily accessible (Hernandez et al., 2009).

444 4.3 Resulting thresholds and difficulties arising from their use

445 Regression trees allowed us to identify threshold effects and cascading thresholds as opposed to
446 linear effects. At small scales, the identified thresholds are likely to represent typical situations of
447 residential housing. For instance, the threshold for the proportion of garden areas is 29.7 +/- 3.8% at
448 50m and 15.7% at 100m, which may account for locations situated between gardens, houses, and
449 access roads. At small spatial scales, impervious surfaces and domestic gardens surfaces proportions
450 were correlated and impervious surfaces co-occurred with gardens location. This is likely to
451 indicate this parallel evolution of housing with dwellings and gardens on the one hand, and
452 residential infrastructure with access roads and other impervious surfaces on the other. The
453 threshold figures might seem smaller than what could be expected for the respective proportions of
454 detached houses, gardens, and access roads. This seemingly small figure may be compared to the
455 results pertaining to population conservation, which requires larger areas than those identified for
456 individual species conservation (Beninde et al., 2015; Smith, 2007). The identified threshold here is
457 only a minimum corresponding to the species observed by participants, which is a subset of total
458 diversity.

459 Considering an average garden size of 571m² (Dewaelheyns et al., 2014), the proportions of
460 domestic gardens in both 50m and 100m buffers represent approximately four gardens in the 50m
461 buffer and eight gardens in the 100m buffer. Assembled in a patch, these gardens may account for
462 coordinated individual actions in favor of pollinators in the neighboring gardens, either separated by
463 fences or situated in a row along a street. As a result, it would not require many inhabitants to
464 transform individual decisions with uncertain collective consequences into coordinated gardening
465 decisions in favor of the target organism, that is, to shift from the “tyranny of small decisions” to a
466 “resource by small gardening actions” (Dewaelheyns et al., 2016).

467 The threshold for the proportion of built-up areas with an effect on morphotype richness is variable
468 in regression trees, ranging from 250m to 1000m scales. Considering the variety of pollinators, it
469 may be that some key morphotypes respond overwhelmingly to certain spatial scales at different
470 thresholds, which would result in these inconsistencies.

471 We chose to use the proportion of garden and built-up areas as landscape indicators in this study.
472 Correlations exist between both proportions, but are not present at all spatial scales and are of low
473 value when present, i.e. less than 0.16. Thus, each indicator independently contributes information
474 in this study. A similar surface proportion of the indicator, regardless of which one, may point to a
475 situation with either numerous fragmented areas or a single large area: total surface and
476 fragmentation degree are thus undifferentiated (Andrén, 1994). Here, we cannot assess the
477 particular situation to which pollinator richness responded. Although threshold determination is
478 useful in statistical modeling when the non-linear effect is hypothesized, thresholds should
479 subsequently be manipulated with caution. To guide urban planning in favor of biodiversity, goals
480 and limitations associated with these thresholds need to be explicit from both scientific and
481 management perspectives (Beninde et al., 2015). Indeed, each threshold may be viewed from two
482 angles: either as a minimal requirement to be expanded or as a limiting value to be reached.

483 4.4 Orders favored at smaller scales in gardens

484 Finer scales of pollinator diversity may reveal a variability of responses to urbanization, as some
485 groups, such as cavity-nesting bees (Hernandez et al., 2009) and social small-bodied species
486 (Banaszak-Cibicka and Zmihorski, 2012), increase in cities, while others decrease: pollinators thus
487 have various levels of urbanophobia (Deguines et al., 2012). Here, we identified orders that were
488 favored or not by domestic garden surroundings at small spatial scales. Lepidoptera were favored
489 by domestic garden surroundings at the 50m spatial scale, which is consistent with their known
490 sensibility to urbanization (Ramirez-Restrepo et al., 2017) and negative affinity for urban land use

491 (Deguines et al., 2012). Likewise, Diptera have rather low affinities for urban land use (Deguines et
492 al., 2012), which is consistent with them being favored by domestic garden surroundings in this
493 study. Hymenoptera were not favored by domestic garden surroundings at the 50m spatial scale. As
494 insects belonging to this order are likely to have a positive affinity for urban land use (Deguines et
495 al., 2012), the proportion of domestic gardens taken into account in this study may not be sufficient
496 to reveal a positive effect of these less impervious surfaces (McKinney, 2008). Variability of species
497 responses within the order may also account for the absence of differences when taking into account
498 the entire order. Here, we also revealed that Coleoptera are not favored by domestic garden
499 surroundings, which may be surprising given the general negative effect of urban land use on
500 insects (McKinney, 2008; McIntyre, 2000), but this may also reflect certain management practices
501 in gardens.

502 Overall, our findings indicate that Lepidoptera and Diptera could benefit from combined decisions
503 in favor of insects at small spatial scales, coordinated between four and eight domestic gardens.
504 This also points to the necessity for broader communication about insects and pollinators, as they
505 are usually associated with negative feelings (Kellert, 1993), but can be recognized for utilitarian or
506 aesthetic values (Barrow, 2002). While Lepidoptera have aesthetic qualities (Barrow, 2002), this is
507 not the case with Diptera, for which more knowledge diffusion could help garden management in
508 their favor.

509 5 Conclusion

510 This study identified the less explored role of domestic gardens as places that structure favorable
511 features for pollinator habitat selection at the local scale after filtering due to urbanization. In line
512 with Goddard et al. (2010), we acknowledge that the consideration of garden owners' individual
513 management decisions as a whole is problematic as the scale of management is, at least, order-
514 dependent. Garden owners should be encouraged to focus on management favorable to species with

515 short dispersion and flight ranges, as these are most likely to be beneficial given that the positive
516 effect of gardens occurs at this spatial scale. Concerning specific orders, garden owners should be
517 encouraged to focus on management favorable to Lepidoptera and given more information about
518 Diptera. To outline propositions for conservation behavior at the domestic garden scale, our study
519 could be repeated with a smaller taxonomic resolution. For example, it could be extended to unique
520 morphospecies. Public policies will be more effective if they combine various management
521 approaches based on features that impact pollinator richness at different scales and favor
522 connectivity instead of isolated patches, as well as coordination between local actors, namely
523 gardeners, at the scale of garden patches. Regarding the SPIPOLL program, using morphotype
524 identification level, which is less precise than species identification level, is also linked to the
525 acknowledgment of participants' work. Indeed, using this dataset and communicating the results to
526 participants is a means to motivate citizen scientists to continue contributing to the program.

527 **Acknowledgments**

528 The authors acknowledge all SPIPOLL participants, the SPIPOLL database managers, as well as
529 two anonymous reviewers who provided helpful comments.

530 **Funding**

531 Île-de-France PSDR Dynamiques project, Labex BASC. M.L. is funded by the French Ministry of
532 Research and the FdV Doctoral School.

533 **References**

- 534 Andr en, H., 1994. Effects of habitat fragmentation on birds and mammals in landscapes with
535 different proportions of suitable habitat: A review. *Oikos* 71, 355.
536 <https://doi.org/10.2307/3545823>
- 537 Arnold, C.J.L., Gibbons, J.C., 1996. Impervious surface coverage: the emergence of a key
538 environmental indicator. *J. Am. Plan. Assoc.* 62, 243–258.

- 539 Aronson, M.F.J., La Sorte, F.A., Nilon, C.H., Katti, M., Goddard, M.A., Lepczyk, C.A., Warren,
540 P.S., Williams, N.S.G., Cilliers, S., Clarkson, B., Dobbs, C., Dolan, R., Hedblom, M., Klotz,
541 S., Kooijmans, J.L., Kuhn, I., MacGregor-Fors, I., McDonnell, M., Mortberg, U., Pysek, P.,
542 Siebert, S., Sushinsky, J., Werner, P., Winter, M., 2014. A global analysis of the impacts of
543 urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proc. R. Soc. B*
544 *Biol. Sci.* 281, 20133330. <https://doi.org/10.1098/rspb.2013.3330>
- 545 Baldock, K.C.R., Goddard, M.A., Hicks, D.M., Kunin, E., Mitschunas, N., Osgathorpe, L.M., Potts,
546 S.G., Robertson, K.M., Scott, A. V, Stone, G.N., Vaughan, I.P., Memmott, J., 2015. Where is
547 the UK 's pollinator biodiversity ? The importance of urban areas for flower- visiting insects.
548 *Proc. R. Soc. Biol. Sci.* 282, 20142849. <https://doi.org/10.1098>
- 549 Banaszak-Cibicka, W., Zmihorski, M., 2012. Wild bees along an urban gradient: Winners and
550 losers. *J. Insect Conserv.* 16, 331–343. <https://doi.org/10.1007/s10841-011-9419-2>
- 551 Barrow, L.H., 2002. What do elementary students know about insects? *J. Element. Sci. Educ.* 14,
552 51-56. <https://doi.org/10.1007/BF03173848>
- 553 Bates, A.J., Sadler, J.P., Grundy, D., Lowe, N., Davis, G., Baker, D., Bridge, M., Freestone, R.,
554 Gardner, D., Gibson, C., Hemming, R., Howarth, S., Orridge, S., Shaw, M., Tams, T., Young,
555 H., 2014. Garden and landscape-scale correlates of moths of differing conservation status:
556 Significant effects of urbanization and habitat diversity. *PLoS One* 9.
557 <https://doi.org/10.1371/journal.pone.0086925>
- 558 Beninde, J., Veith, M., Hochkirch, A., 2015. Biodiversity in cities needs space: A meta-analysis of
559 factors determining intra-urban biodiversity variation. *Ecol. Lett.* 18, 581–592.
560 <https://doi.org/10.1111/ele.12427>
- 561 Bergerot, B., Fontaine, B., Julliard, R., Baguette, M., 2011. Landscape variables impact the
562 structure and composition of butterfly assemblages along an urbanization gradient. *Landsc.*
563 *Ecol.* 26, 83–94. <https://doi.org/10.1007/s10980-010-9537-3>
- 564 Cameron, R.W.F., Blanuša, T., Taylor, J.E., Salisbury, A., Halstead, A.J., Henricot, B., Thompson,
565 K., 2012. The domestic garden - Its contribution to urban green infrastructure. *Urban For.*
566 *Urban Green.* 11, 129–137. <https://doi.org/10.1016/j.ufug.2012.01.002>
- 567 Colding, J., 2007. “Ecological land-use complementation” for building resilience in urban
568 ecosystems. *Landsc. Urban Plan.* 81, 46–55. <https://doi.org/10.1016/j.landurbplan.2006.10.016>
- 569 Cook, E.M., Hall, S.J., Larson, K.L., 2012. Residential landscapes as social-ecological systems: A
570 synthesis of multi-scalar interactions between people and their home environment, *Urban*
571 *Ecosystems* 15, 19–52. <https://doi.org/10.1007/s11252-011-0197-0>
- 572 Czech, B., Krausman, P.R., Devers, P.K., 2000. Economic associations among causes of species
573 endangerment in the United States. *Bioscience* 50, 593–601. [https://doi.org/10.1641/0006-3568\(2000\)050\[0593:EAACOS\]2.0.CO;2](https://doi.org/10.1641/0006-3568(2000)050[0593:EAACOS]2.0.CO;2)

- 575 Daniels, G.D., Kirkpatrick, J.B., 2006. Does variation in garden characteristics influence the
576 conservation of birds in suburbia? *Biol. Conserv.* 133, 326–335.
577 <https://doi.org/10.1016/j.biocon.2006.06.011>
- 578 Davies, Z.G., Fuller, R.A., Loram, A., Irvine, K.N., Sims, V., Gaston, K.J., 2009. A national scale
579 inventory of resource provision for biodiversity within domestic gardens. *Biol. Conserv.* 142,
580 761–771. <https://doi.org/10.1016/j.biocon.2008.12.016>
- 581 Deguines, N., Julliard, R., de Flores, M., Fontaine, C., 2012. The whereabouts of flower visitors:
582 Contrasting land-use preferences revealed by a country-wide survey based on citizen science.
583 *PLoS One* 7, e45822. <https://doi.org/10.1371/journal.pone.0045822>
- 584 Deguines, N., de Flores, M., Loïs, G., Julliard, R., Fontaine, C., 2018. Fostering close encounters of
585 the entomological kind. *Front. Ecol. Environ.* 16(4), 202–203. <https://doi.org/10.1002/fee.1795>
- 586 Desaegher, J., Nadot, S., Dajoz, I., Colas, B., 2017. Buzz in Paris: flower production and plant–
587 pollinator interactions in plants from contrasted urban and rural origins. *Genetica* 145(6), 513–
588 523. <https://doi.org/10.1007/s10709-017-9993-7>
- 589 Dewaelheyns, V., Rogge, E., Gulinck, H., 2014. Putting domestic gardens on the agenda using
590 empirical spatial data: The case of Flanders. *Appl. Geogr.* 50, 132–143.
591 <https://doi.org/10.1016/j.apgeog.2014.02.011>
- 592 Dewaelheyns, V., Kerselaers, E., Rogge, E., 2016. A toolbox for garden governance. *Land Use*
593 *Policy* 51, 191–205. <https://doi.org/10.1016/j.landusepol.2015.11.016>
- 594 Fortel, L., Henry, M., Guilbaud, L., Guirao, A.L., Kuhlmann, M., Mouret, H., Rollin, O., Vaissière,
595 B.E., 2014. Decreasing abundance, increasing diversity and changing structure of the wild bee
596 community (Hymenoptera: Anthophila) along an urbanization gradient. *PLoS One* 9.
597 <https://doi.org/10.1371/journal.pone.0104679>
- 598 Fuller, R.A., Gaston, K.J., 2009. The scaling of green space coverage in European cities. *Biol. Lett.*
599 5, 352–355. <https://doi.org/10.1098/rsbl.2009.0010>
- 600 Garbuzov, M., Ratnieks, F.L.W., 2014. Quantifying variation among garden plants in attractiveness
601 to bees and other flower-visiting insects. *Funct. Ecol.* 28, 364–374.
602 <https://doi.org/10.1111/1365-2435.12178>
- 603 Gardiner, M.M., Allee, L.L., Brown, P.M.J., Losey, J.E., Roy, H.E., Smyth, R.R., 2012. Lessons
604 from lady beetles: Accuracy of monitoring data from US and UK citizen science programs.
605 *Front. Ecol. Environ.* 10, 471–476. <https://doi.org/10.1890/110185>
- 606 Gaston, K.J., Smith, R.M., Thompson, K., Warren, P.H., 2005. Urban domestic gardens (II):
607 Experimental tests of methods for increasing biodiversity. *Biodivers. Conserv.* 14, 395–413.
608 <https://doi.org/10.1007/s10531-004-6066-x>

- 609 Geslin, B., Le Féon, V., Folschweiller, M., Flacher, F., Carmignac, D., Motard, E., Perret, S., Dajoz,
610 I., 2016. The proportion of impervious surfaces at the landscape scale structures wild bee
611 assemblages in a densely populated region. *Ecol. Evol.* 6, 6599–6615.
612 <https://doi.org/10.1002/ece3.2374>
- 613 Goddard, M.A., Dougill, A.J., Benton, T.G., 2010. Scaling up from gardens: biodiversity
614 conservation in urban environments. *Trends Ecol. Evol.* 25, 90–98.
615 <https://doi.org/10.1016/j.tree.2009.07.016>
- 616 Greenleaf, S.S., Williams, N.M., Winfree, R., Kremen, C., 2007. Bee foraging ranges and their
617 relationship to body size. *Oecologia* 153, 589–596. <https://doi.org/10.1007/s00442-007-0752-9>
- 618 Grimm, N.B., Grove, J.M., Pickett, S.T. a, Redman, C.L., 2008. Integrated approaches to long-term
619 studies of urban ecological systems. *Bioscience* 50, 571–584.
- 620 Hernandez, J.L., Frankie, G.W., Thorp, R.W., 2009. Ecology of urban bees: A review of current
621 knowledge and directions for future study. *Cities Environ.* 2, 360–376.
622 <https://doi.org/10.1002/9781118279519.ch20>
- 623 Herrera, C.M., 1990. Daily patterns of pollinator activity, differential pollinating effectiveness, and
624 floral resource availability, in a summer-flowering Mediterranean shrub. *Oikos* 58, 277–288.
- 625 Hinners, S.J., Kearns, C.A., Wessman, C.A., 2012. Role of scale, matrix, and native habitat in
626 supporting a diverse suburban pollinator assemblage. *Ecol. Appl.* 22(7), 1923–1935.
- 627 Hochachka, W.M., Fink, D., Hutchinson, R.A., Sheldon, D., Wong, W.K., Kelling, S., 2012. Data-
628 intensive science applied to broad-scale citizen science. *Trends Ecol. Evol.* 27, 130–137.
629 <https://doi.org/10.1016/j.tree.2011.11.006>
- 630 Hostetler, N.E., McIntyre, M.E., 2001. Effects of urban land use on pollinator (Hymenoptera:
631 Apoidea) communities in a desert metropolis. *Basic Appl. Ecol.* 2, 209–218.
632 <https://doi.org/10.1078/1439-1791-00051>
- 633 Hothorn, T., Hornik, K., Strobl, C., 2017. Package “party .”
634 <https://doi.org/10.1198/106186006X133933>>
- 635 Hothorn, T., Zeileis, A., 2015. partykit: A Modular Toolkit for Recursive Partytioning in R. *Journal*
636 *of Machine Learning Research*, 16, 3905-3909.
- 637 Keitt, T.H., Urban, D.L., Milne, B.T., 1997. Detecting critical scales in fragmented landscapes:
638 Detecting critical scales in fragmented landscapes. *Ecol. Soc.* 1, 4–17.
639 <https://doi.org/10.5751/ES-00015-010104>
- 640 Kellert, S.R., 1993. Values and perceptions of invertebrates. *Cons. Biol.* 7, 845–855.
- 641 Kleijn, D., Winfree, R., Bartomeus, I., Carvalheiro, L.G., Henry, M., Isaacs, R., Klein, A.-M.,
642 Kremen, C., M’Gonigle, L.K., Rader, R., Ricketts, T.H., Williams, N.M., Lee Adamson, N.,
643 Ascher, J.S., Báldi, A., Batáry, P., Benjamin, F., Biesmeijer, J.C., Blitzer, E.J., Bommarco, R.,

- 644 Brand, M.R., Bretagnolle, V., Button, L., Cariveau, D.P., Chifflet, R., Colville, J.F., Danforth,
645 B.N., Elle, E., Garratt, M.P.D., Herzog, F., Holzschuh, A., Howlett, B.G., Jauker, F., Jha, S.,
646 Knop, E., Krewenka, K.M., Le Féon, V., Mandelik, Y., May, E. a, Park, M.G., Pisanty, G.,
647 Reemer, M., Riedinger, V., Rollin, O., Rundlöf, M., Sardiñas, H.S., Scheper, J., Sciligo, A.R.,
648 Smith, H.G., Steffan-Dewenter, I., Thorp, R., Tschamtko, T., Verhulst, J., Viana, B.F.,
649 Vaissière, B.E., Veldtman, R., Westphal, C., Potts, S.G., 2015. Delivery of crop pollination
650 services is an insufficient argument for wild pollinator conservation. *Nat. Commun.* 6, 7414.
651 <https://doi.org/10.1038/ncomms8414>
- 652 Knop, E., Gerpe, C., Ryser, R., Hofmann, F., Menz, M.H.M., Trösch, S., Ursenbacher, S., Zoller, L.,
653 Fontaine, C., 2017. Rush hours in flower visitors over a day-night cycle. *Insect Conserv. Diver.*
654 11, 265-275. <https://doi.org/10.1111/icad.12277>
- 655 Kremen, C., Ullman, K.S., Thorp, R.W., 2011. Evaluating the quality of citizen-scientist data on
656 pollinator communities. *Conserv. Biol.* 25, 607–617. <https://doi.org/10.1111/j.1523-1739.2011.01657.x>
657
- 658 Lefebvre, V., Villemant, C., Fontaine, C., Daugeron, C., 2018. Altitudinal, temporal and trophic
659 partitioning of flower-visitors in Alpine communities. *Sci. Rep.* 8, 4706.
660 <https://doi.org/10.1038/s4158-018-23210-y>
- 661 Lepczyk, C.A., Aronson, M.F.J., Evans, K.L., Goddard, M.A., Lerman, S.B., Macivor, J.S., 2017.
662 Biodiversity in the city: Fundamental questions for understanding the ecology of urban green
663 spaces for biodiversity conservation. *Bioscience* 67, 799–807.
664 <https://doi.org/10.1093/biosci/bix079>
- 665 Lerman, S.B., Contosta, A.R., Milam, J., Bang, C., 2018. To mow or to mow less: Lawn mowing
666 frequency affects bee abundance and diversity in suburban yards. *Biol. Conserv.* 221, 160–174.
667 <https://doi.org/10.1016/j.biocon.2018.01.025>
- 668 Levrel, H., Fontaine, B., Henry, P.Y., Jiguet, F., Julliard, R., Kerbiriou, C., Couvet, D., 2010.
669 Balancing state and volunteer investment in biodiversity monitoring for the implementation of
670 CBD indicators: A French example. *Ecol. Econ.* 69, 1580–1586.
671 <https://doi.org/10.1016/j.ecolecon.2010.03.001>
- 672 Loram, A., Tratalos, J., Warren, P.H., Gaston, K.J., 2007. Urban domestic gardens (X): The extent &
673 structure of the resource in five major cities. *Landsc. Ecol.* 22, 601–615.
674 <https://doi.org/10.1007/s10980-006-9051-9>
- 675 MacIvor, J.S., Packer, L., 2015. “Bee hotels” as tools for native pollinator conservation: A
676 premature verdict? *PLoS One* 10, 1–13. <https://doi.org/10.1371/journal.pone.0122126>
- 677 Mathieu, R., Freeman, C., Aryal, J., 2007. Mapping private gardens in urban areas using object-
678 oriented techniques and very high-resolution satellite imagery. *Landsc. Urban Plan.* 81, 179–
679 192. <https://doi.org/10.1016/j.landurbplan.2006.11.009>

- 680 Matteson, K.C., Grace, J.B., Minor, E.S., 2013. Direct and indirect effects of land use on floral
681 resources and flower-visiting insects across an urban landscape. *Oikos* 122, 682–694.
682 <https://doi.org/10.1111/j.1600-0706.2012.20229.x>
- 683 McIntyre, N., 2000. Ecology of urban arthropods: A review and a call to action. *Ann. Entomol. Soc.*
684 *Am.* 93, 825–835. [https://doi.org/10.1603/0013-8746\(2000\)093\[0825:EOUAAR\]2.0.CO;2](https://doi.org/10.1603/0013-8746(2000)093[0825:EOUAAR]2.0.CO;2)
- 685 McKinney, M., 2002. Urbanization, biodiversity, and conservation. *Bioscience* 52, 883–890.
- 686 McKinney, M.L., 2008. Effects of urbanization on species richness: A review of plants and animals.
687 *Urban Ecosyst.* 11, 161–176. <https://doi.org/10.1007/s11252-007-0045-4>
- 688 Obrist, M.K., Duelli, P., 2010. Rapid biodiversity assessment of arthropods for monitoring average
689 local species richness and related ecosystem services. *Biodivers. Conserv.* 19, 2201–2220.
690 <https://doi.org/10.1007/s10531-010-9832-y>
- 691 Pardee, G.L., Philpott, S.M., 2014. Native plants are the bee’s knees: local and landscape predictors
692 of bee richness and abundance in backyard gardens. *Urban Ecosyst.* 1–19.
693 <https://doi.org/10.1007/s11252-014-0349-0>
- 694 Parris, K.M., 2006. Urban amphibian assemblages as metacommunities. *J. Anim. Ecol.* 75, 757–
695 764. <https://doi.org/10.1111/j.1365-2656.2006.01096.x>
- 696 Pinheiro, J.I., Bates, D., DebRoy, S., Sarkar, D., R Core Team, 2017. nlme: Linear and Non linear
697 Mixed Effects Models. R package version 3.1-131, <https://CRAN.R-project.org/package=nlme>
- 698 QGIS Development Team, 2017. QGIS Geographic Information System. Open Source Geospatial
699 Foundation. URL <http://qgis.osgeo.org>
- 700 R Core Team, 2017. R: A language and environment for statistical computing. R Foundation for
701 Statistical Computing, Vienna, Austria. URL <http://www.R-project.org>
- 702 Potts, S.G., Biesmeijer, J.C., Kremen, C., Neumann, P., Schweiger, O., Kunin, W.E., 2010. Global
703 pollinator declines: Trends, impacts and drivers. *Trends Ecol. Evol.* 25, 345–353.
704 <https://doi.org/10.1016/j.tree.2010.01.007>
- 705 Ramirez-Restrepo, L., MacGregor-Fors, I., 2017. Butterflies in the city: A review of urban diurnal
706 Lepidoptera. *Urban Ecosyst.* 20, 171–182. <https://doi.org/10.1007/s11252-016-0579-4>
- 707 Salisbury, A., Armitage, J., Bostock, H., Perry, J., Tatchell, M., Thompson, K., 2015. Enhancing
708 gardens as habitats for flower-visiting aerial insects (pollinators): Should we plant native or
709 exotic species? *J. Appl. Ecol.* 52, 1156–1164. <https://doi.org/10.1111/1365-2664.12499>
- 710 Samnegård, U., Persson, A.S., Smith, H.G., 2011. Gardens benefit bees and enhance pollination in
711 intensively managed farmland. *Biol. Conserv.* 144, 2602–2606.
712 <https://doi.org/10.1016/j.biocon.2011.07.008>

- 713 Smith, P.G.R., 2007. Characteristics of urban natural areas influencing winter bird use in southern
714 Ontario, Canada. *Environ. Manage.* 39, 338–352. <https://doi.org/10.1007/s00267-005-0028-2>
- 715 Smith, R.M., Gaston, K.J., Warren, P.H., Thompson, K., 2006a. Urban domestic gardens (VIII):
716 Environmental correlates of invertebrate abundance. *Biodivers. Conserv.* 15, 2515–2545.
717 <https://doi.org/10.1007/s10531-005-2784-y>
- 718 Smith, R.M., Warren, P.H., Thompson, K., Gaston, K.J., 2006b. Urban domestic gardens (VI):
719 Environmental correlates of invertebrate species richness. *Biodivers. Conserv.* 15, 2415–2438.
720 <https://doi.org/10.1007/s10531-004-5014-0>
- 721 Sperling, C.D., Lortie, C.J., 2010. The importance of urban backgardens on plant and invertebrate
722 recruitment: A field microcosm experiment. *Urban Ecosyst.* 13, 223–235.
723 <https://doi.org/10.1007/s11252-009-0114-y>
- 724 Steffan-Dewenter, I., Münzenberg, U., Bürger, C., Thies, C., Tschardt, T., 2002. Scale-dependent
725 effects of landscape context on three pollinator guilds. *Ecology* 83, 1421–1432.
726 [https://doi.org/10.1890/0012-9658\(2002\)083\[1421:SDEOLC\]2.0.CO;2](https://doi.org/10.1890/0012-9658(2002)083[1421:SDEOLC]2.0.CO;2)
- 727 Tommasi, D., Miro, A., Higo, H.A., Winston, M.L., 2004. Bee diversity and abundance in an urban
728 setting. *Can. Entomol.* 136, 851–869. <https://doi.org/10.4039/n04-010>
- 729 Tzoulas, K., Korpela, K., Venn, S., Yli-Pelkonen, V., Kaźmierczak, A., Niemela, J., James, P., 2007.
730 Promoting ecosystem and human health in urban areas using green infrastructure: A literature
731 review. *Landsc. Urban Plan.* 81, 167–178. <https://doi.org/10.1016/j.landurbplan.2007.02.001>
- 732 United Nations, 2014. *World Urbanization Prospects*, United Nations.
733 <https://doi.org/10.4054/DemRes.2005.12.9>
- 734 van Heezik, Y., Smyth, A., Mathieu, R., 2008. Diversity of native and exotic birds across an urban
735 gradient in a New Zealand city. *Landsc. Urban Plan.* 87, 223–232.
736 <https://doi.org/10.1016/j.landurbplan.2008.06.004>
- 737 With, K.A., Caderet, S.J., Davis, C., 1999. Movement responses to path structure in experimental
738 fractal landscapes. *Ecology* 80, 1340–1353. [https://doi.org/10.1890/0012-9658\(1999\)080%255B1340:MRTPSI%255D2.0.CO%253B2](https://doi.org/10.1890/0012-9658(1999)080%255B1340:MRTPSI%255D2.0.CO%253B2)
- 740 Zurbuchen, A., Landert, L., Klaiher, J., Müller, A., Hein, S., Dorn, S., 2010. Maximum foraging
741 ranges in solitary bees: Only few individuals have the capability to cover long foraging
742 distances. *Biol. Conserv.* 143, 669–676. <https://doi.org/10.1016/j.biocon.2009.12.003>

1 FIGURES

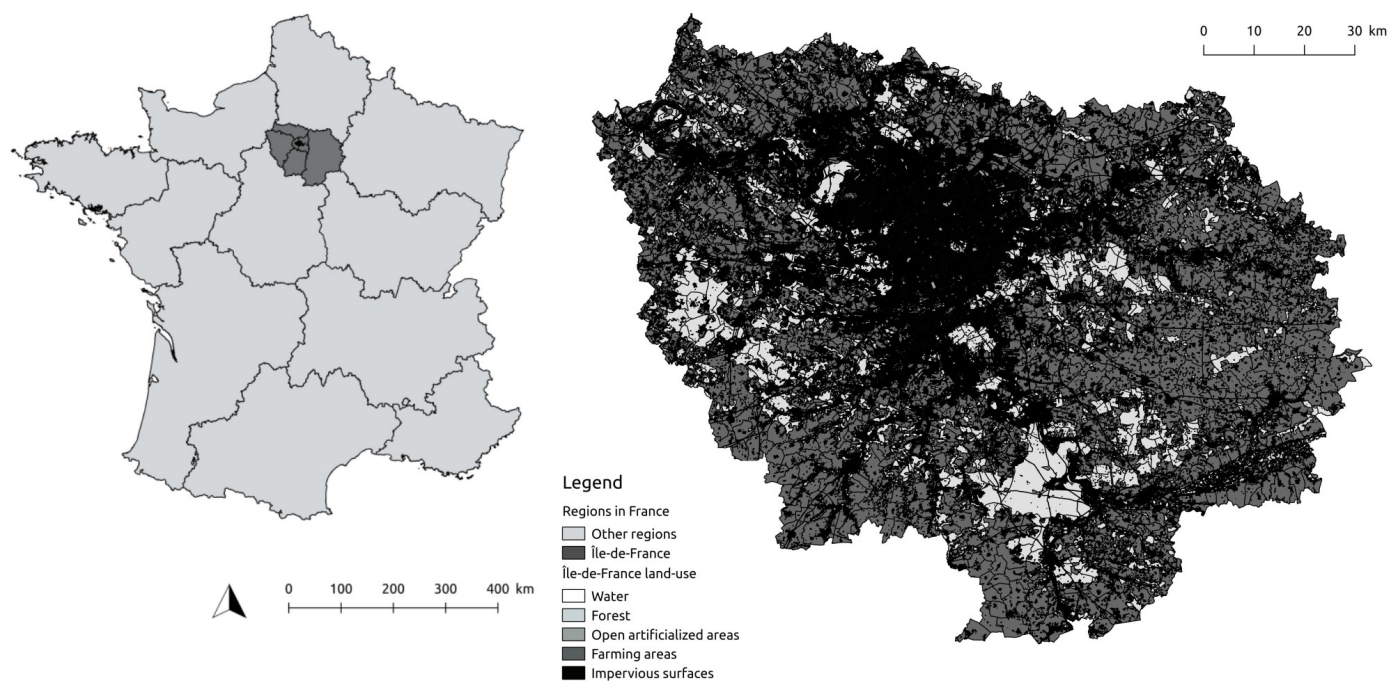
2 Figure 1. Location of the studied region in France and land use in Île-de-France.

3 Figure 2. Flower families which are present in 30 recorded collections and more, ordered by the
4 number of collections in which they are present.

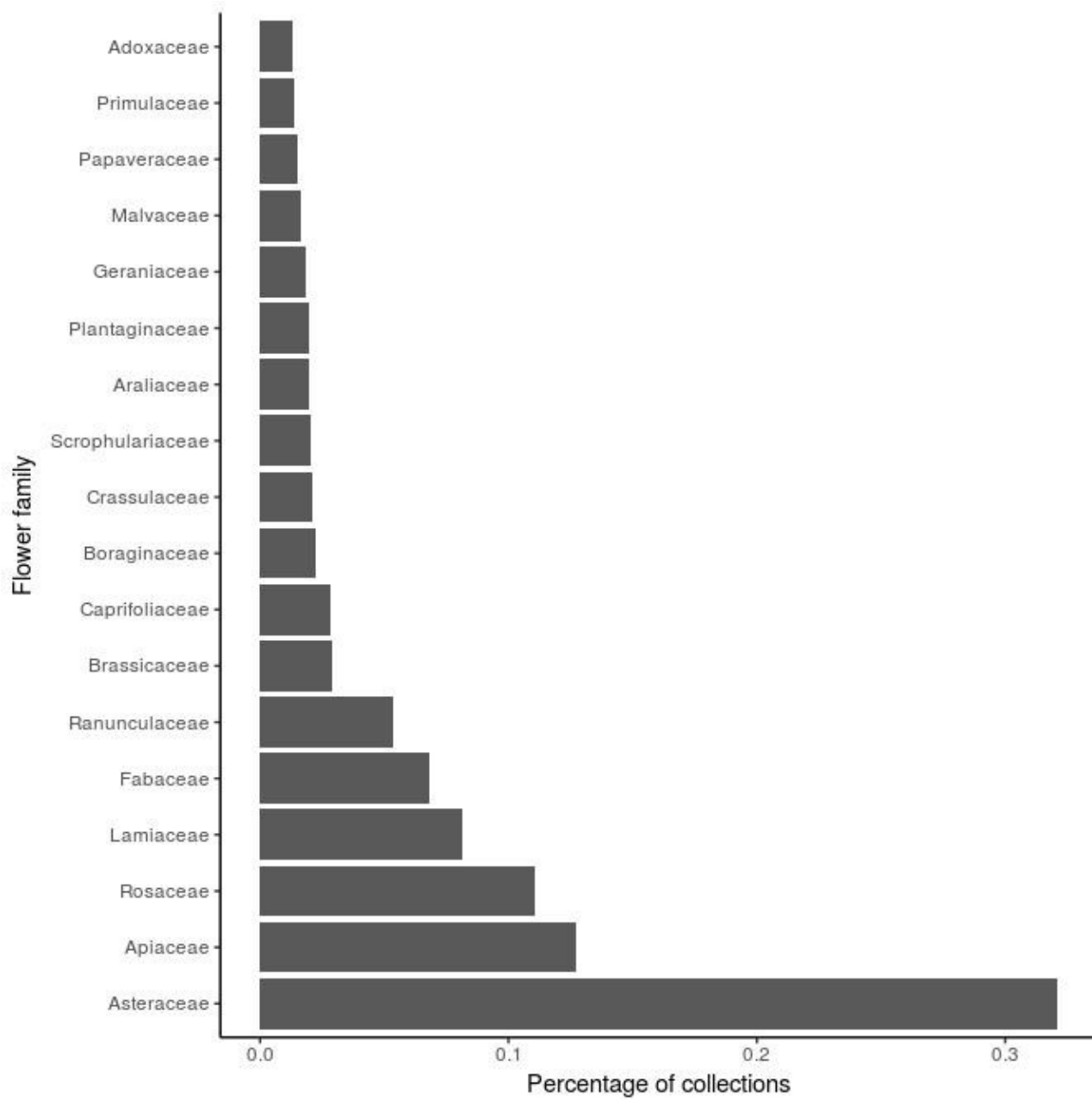
5 Figure 3. Map extract showing impervious surfaces, domestic gardens, and other non-impervious
6 areas. An example of the SPIPOLL collection location with a 250m buffer.

7 Figure 4. Regression tree for the model unstandardized for local context (i.e., excluding the garden
8 context variable) with all spatial scales taken into account and a synthesis of all corresponding trees
9 (n=244) from the pool of 500 drawn trees. Variables for nodes are shown in circles. Threshold
10 values of the variables are given on the formed branches. Terminal nodes are numbered, and the
11 associated mean +/- sd for morphotype richness is stated. Significant p-values for the difference
12 between mean morphotype richness are indicated.

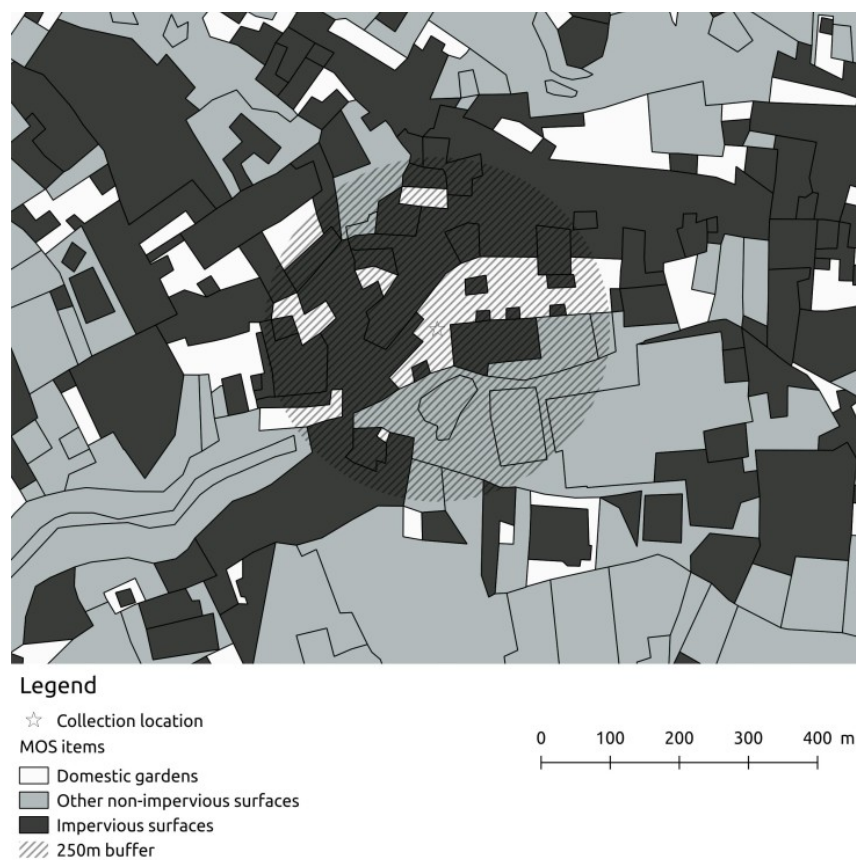
13 Figure 5. Regression trees for the model unstandardized for local context (i.e., excluding the garden
14 context variable). A: 50m buffer; B: 100m buffer; C: 250m buffer; D: 500m buffer; E: 1000m
15 buffer. Variables for nodes are shown in circles with the associated p-value. Threshold values of the
16 variables are given on the formed branches. Terminal nodes are numbered, and the associated mean
17 +/- sd for morphotype richness is provided. Significant p-values for the difference between mean
18 morphotype richness are indicated below each tree.



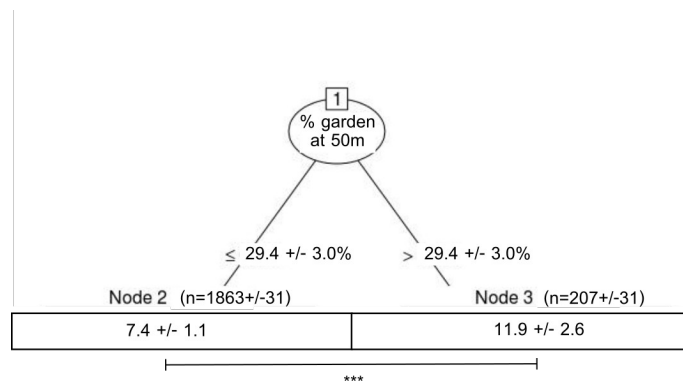
20 Figure 1. Location of the studied region in France and land use in Île-de-France.



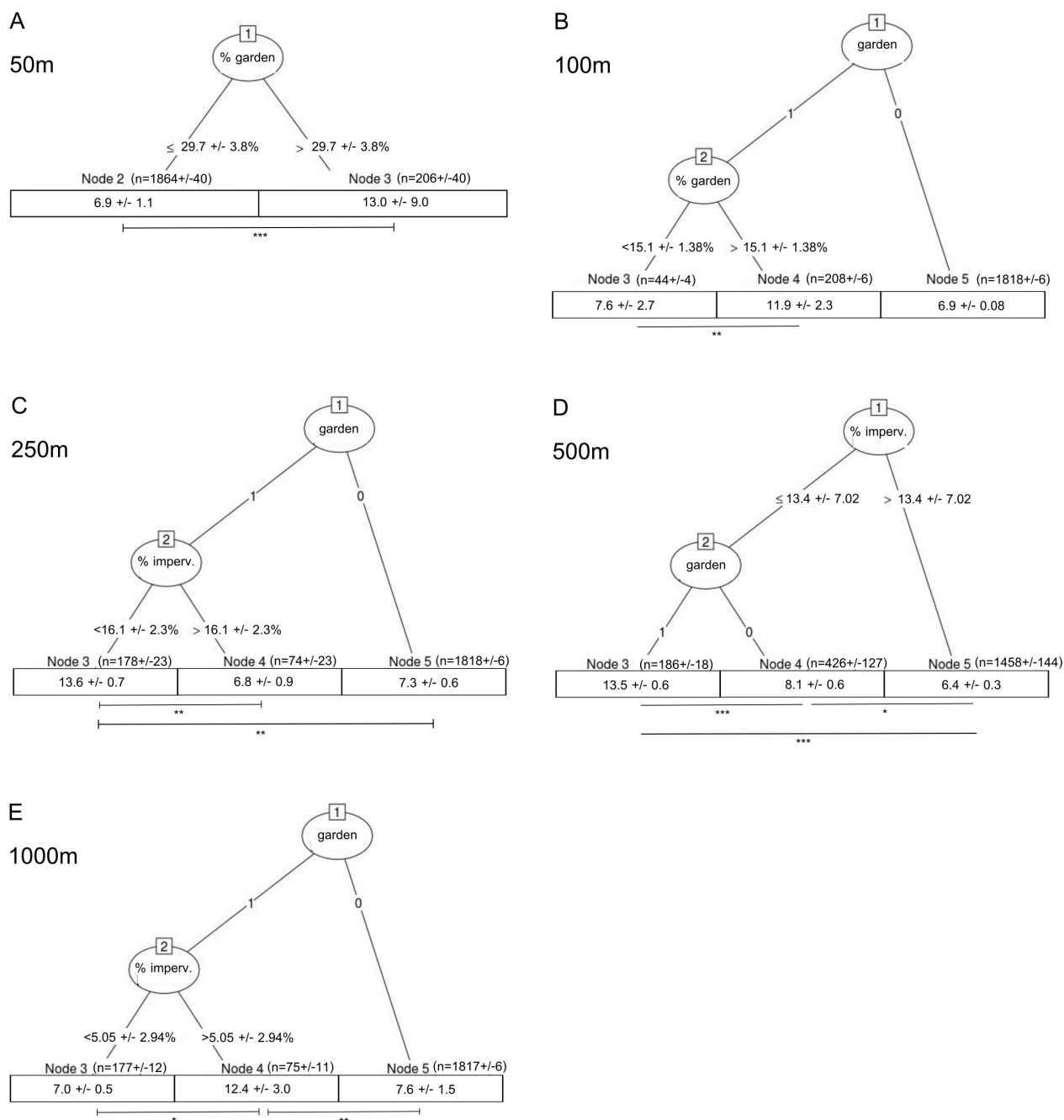
22 Figure 2. Flower families which are present in 30 recorded collections and more, ordered by the
23 number of collections in which they are present.



25 Figure 3. Map extract showing impervious surfaces, domestic gardens, and other non-impervious
26 areas. An example of the SPIPOLL collection location with a 250m buffer.



28 Figure 4. Regression tree for the model unstandardized for local context (i.e., excluding the garden
29 context variable) with all spatial scales taken into account and a synthesis of all corresponding trees
30 (n=244) from the pool of 500 drawn trees. Variables for nodes are shown in circles. Threshold
31 values of the variables are given on the formed branches. Terminal nodes are numbered, and the
32 associated mean +/- sd for morphotype richness is stated. Significant p-values for the difference
33 between mean morphotype richness are indicated.



35 Figure 5. Regression trees for the model unstandardized for local context (i.e., excluding the garden
 36 context variable). A: 50m buffer; B: 100m buffer; C: 250m buffer; D: 500m buffer; E: 1000m
 37 buffer. Variables for nodes are shown in circles with the associated p-value. Threshold values of the

38 variables are given on the formed branches. Terminal nodes are numbered, and the associated mean
39 +/- sd for morphotype richness is provided. Significant p-values for the difference between mean
40 morphotype richness are indicated below each tree.

1 TABLES

2 Table 1. Description of the data set according to orders morphotypes belong to, proportion of
3 morphotypes belonging to each order, number of different morphotypes seen in each order and
4 number of morphotypes levels appearing in the data set (over 7 levels).

5 Table 2. Description of the variables for the proportion of garden areas and impervious surfaces at
6 all spatial scales.

7 Table 3. Summary of variable effects in the regression trees at the different spatial scales. Mention
8 of 1 or 2 indicates if the variable was used in order to define a first order division in the tree of a
9 second order division, i.e. inside an already defined branch of the tree. Results for trees without the
10 local garden context variable are shown in brackets.

- 11 Table 1. Description of the data set according to orders morphotypes belong to, proportion of
12 morphotypes belonging to each order, number of different morphotypes seen in each order and
13 number of morphotypes levels appearing in the data set (over 7 levels).

Order	% of morphotypes belonging to this order	Number of different morphotypes seen	Number of morphotypes levels appearing
Blattodea	0.015	1	1
Coleoptera	13.9	87	5
Dermaptera	0.078	1	1
Diptera	34.9	73	5
Ephemeroptera	0.0098	1	1
Hemiptera	4.96	20	4
Hymenoptera	40.4	78	6
Lepidoptera	4.61	69	5
Mecoptera	0.14	1	1
Nevroptera	0.33	2	1
Opilion	0.1	1	1
Orthoptera	0.61	2	2

14 22

15 Table 2. Description of the variables for the proportion of garden areas and impervious surfaces at
16 all spatial scales.

Spatial scale :	50m	100m	250m	500m	1000m
	Proportion of garden areas (%)				
Mean +/- sd	4.65 +/- 12.4	3.90 +/- 8.89	0.565 +/- 1.62	0.308 +/- 1.06	0.178 +/- 0.621
Min	0	0	0	0	0
Max	76.1	43.9	27.7	18.5	15.8
	Proportion of impervious surfaces (%)				
Mean +/- sd	40.1 +/- 38.3	39.5 +/- 35.3	44.8 +/- 31.7	41.1 +/- 32.3	41.6 +/- 31.2
Min	0	0	0	0	0
Max	100	100	100	100	98.2

17

18 Table 3. Summary of variable effects in the regression trees at the different spatial scales. Mention
19 of 1 or 2 indicates if the variable was used in order to define a first order division in the tree of a
20 second order division, i.e. inside an already defined branch of the tree. Results for trees without the
21 local garden context variable are shown in brackets.

	Local →		→		→ Landscape
Spatial scale	50m	100m	250m	500m	1000m
Context	-	1 (-)	1 (-)	2 (-)	1 (-)
Garden area	1 (1)	2 (1, 2)	-	-	- (2)
Impervious surface	-	-	2 (1)	1 (1)	2 (1, 2)

22