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1 **Insect ecology and conservation in urban areas: An overview of knowledge and needs**

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17 **Abstract**

18 1. Urban expansion across the globe profoundly impacts local biodiversity. The growing body of
19 urban ecology research on animals has largely focused on mammals and birds, whereas knowledge
20 of insect ecology and conservation in urban areas remains limited.

21 2. To anchor this Special Issue (SI) we have taken a broad approach to editorial and conducted a
22 structured literature search to set the scene. We provide here an overview of existing literature
23 reviews on urban insect ecology and conservation, indicate where the articles included in this SI
24 contribute to developing our understanding, and point to priority areas for further investigation.

25 3. Key themes in the growing literature (at individual, species, and/or community level) include: the
26 influence of habitat quality, quantity, and land use type on insect diversity; the impacts of
27 anthropogenic pollution (for instance heat, noise, light and chemicals); habitat connectivity and
28 changes in habitat structure; and impacts of urban density on genetic diversity. Insect diversity and
29 abundance broadly decline with urban density and loss of habitat. Beyond this, variation in
30 responses of different taxa, or in different regions, and methodological limitations of individual
31 studies make it challenging to identify general patterns.

32 4. Insect ecology and conservation research in urban environments should focus on: applying
33 ecological theory to understand variation in diversity patterns; investigating interactions between
34 climate change and urban contexts; identifying impacts of novel environments on insect biodiversity;
35 addressing methodological limitations and harmonising methodological approaches; and exploring
36 the influence of social and historical factors on urban insect biodiversity. Insect conservation must
37 also consider research into how best to communicate the value of urban insects to urban humans.

38

39

40 **Graphical Abstract**

- 41 • Global urban expansion is a threat to biodiversity, but knowledge of insect ecology and
42 conservation in urban areas remains limited.
- 43 • Key themes in existing literature include habitat influences, anthropogenic pollution threats,
44 road ecology and impacts on genetic diversity, but knowledge gaps and methodological
45 limitations make it difficult to identify general patterns.
- 46 • Priorities for future research include understanding variation in diversity patterns, climate
47 change interactions, impacts of novel environments, methodological limitations, and human
48 social factors that influence urban insect biodiversity.

49

51 **Introduction**

52 Conservation of insects in urban environments is a key global challenge (Luke et al., 2023). The
53 continued expansion of urban areas across the globe (Angel et al., 2011) has raised concerns about
54 how this impacts biodiversity and the field of urban ecology has grown rapidly in recent decades
55 (Niemelä et al., 2011). To date, urban animal studies have, however, largely focused on birds and
56 mammals predominantly in the cities of North America and Europe (Collins et al., 2021). Much less is
57 known of the ecology and diversity of invertebrates in urban environments, particularly for taxa and
58 systems other than insect pests or disease vectors (Rega-Brodsky et al., 2022). Improving knowledge
59 of this critical component of urban ecosystems is crucial to understanding functional ecology
60 processes, and to assess and manage impacts of urbanisation. In particular, understanding the
61 ecological filtering process in an urban context, essentially which species thrive in urban
62 environments and why (Gathof et al., 2022), is critical to inform context-specific urban management
63 strategies, actions, and policies that effectively support biodiversity and maintain the ecosystem
64 services that urban greenspaces can provide.

65 Historically, the ecology of most urban insect species, especially those dwelling in patches or
66 fragments of greenspace, has remained largely unexplored as research has focused on insects of
67 economic importance. To anchor this Special Issue (SI) we use a structured literature search to clarify
68 the knowledge gaps, provide an overview of the current literature on urban insect ecology and
69 conservation, indicate where the articles in this SI contribute, and identify priority areas for
70 investigation. We use the term insects throughout the text for consistency, but we also emphasise
71 the importance of studying invertebrates more broadly in urban environments. We include other
72 invertebrates where relevant and note that some studies, particularly in aquatic environments, focus
73 on invertebrates classified in functional groups, or at surrogate higher taxonomic resolutions (e.g.
74 Ferzoco et al. 2023; Hill et al. 2024).

75 **Approach: methodology, literature extraction and synthesis**

76 An exploratory literature search was completed in August 2023 using the Scopus database and used
77 the key words and combinations in Box 1. Our aim was to provide a rapid evidence assessment of
78 the information available in reviews of this field and this overview is thus representative rather than
79 exhaustive. The topics of urban insect ecology or conservation do also feature in useful general and
80 specific textbooks (e.g. New, 2015; Langellotto & Hall, 2020), but this search was confined to
81 academic journal publications. The overview we give here is also supported by the comments of
82 expert reviewers, the experience and conversations of the authors and the papers submitted to this
83 SI. The emergent themes are thus drawn from the literature and the research opportunities
84 identified arise from gaps proposed or perceived in these.

Box 1: Key words and combinations used to explore the Urban insect literature in the Scopus database

(TITLE-ABS-KEY (*urban OR city OR metropolis OR town OR village OR hamlet) AND TITLE-ABS-KEY (ecology OR conservation OR diversity) AND TITLE-ABS-KEY (insect* OR arthropod* OR *invertebrate OR archaeognatha OR blattodea OR coleoptera OR dermaptera OR diptera OR embioptera OR ephemeroptera OR grylloblattodea OR hemiptera OR hymenoptera OR lepidoptera OR mantodea OR mecoptera OR megaloptera OR neuroptera OR notoptera OR odonata OR orthoptera OR phasmatodea OR phthiraptera OR plecoptera OR psocoptera OR raphidioptera OR siphonaptera OR thysanoptera OR trichoptera OR zoraptera OR zygentoma OR bristletail OR cockroach OR termite OR beetle OR earwig OR *fly OR *flies OR mosquito* OR *spinner OR bug OR true AND bug OR cicada OR wasp OR *bee OR ant OR moth OR mantid OR mantis OR lacewing OR grasshopper AND planthopper OR leafhopper OR locust OR cricket OR katydid OR weta OR *lice OR louse OR flea OR thrips)))

85 **Overview of findings**

86 The search returned 4436 papers, from which we extracted 65 review or synthesis papers (Table 1
87 and Table S1). These 65 reviews fell thematically into the six broad areas that we then used to
88 structure this essay (Table 1), and which we illustrate with specific examples drawn from the full
89 search results, the articles submitted to this SI and other relevant literature.

90 **Table 1: Summary of the 65 reviews identified as relevant to urban invertebrate ecology.**

Broad topic	Specific topic	Number of reviews
General overview		9
Pest management		1
Sociology		2
Ecological interactions	Plant-arthropod	4
	General interactions	1
Biological framing	Genetics	1
	Spatial ecology	2
Habitat	Coastal	2
	Forest	2
	Freshwaters	13
Land use type	Urban agriculture	2
	Golf courses	1
	Grasslands	2
	Green roofs	4
	Wastelands	2
	Specific taxon	Beetles
	Dragonflies	1
	Lepidoptera	2
	Pollinators	4
	Tardigrades	1
Threats	ALAN (artificial light at night)	1
	Disease	1
	Invasion biology	2
	Noise	1
	UHI effect	1

91

92 Since Frankie & Ehler's (1978) comprehensive first review of insects in urban areas, only two further
93 reviews provide general overviews of this topic. These summarise general trends but lack a full
94 synthesis of the existing literature (McIntyre, 2000; Jones & Leather, 2012). Other general reviews
95 that emerged from our search included invertebrates as a small proportion of the broader urban
96 ecology focus on all wildlife (e.g. Collins *et al.*, 2021), or were limited to meta-analysis of research on
97 invertebrates that quantified the impact of urbanisation on species (e.g. Fenoglio *et al.*, 2020). As
98 urban invertebrate research has grown to include more taxonomic or functional groups of non-pest
99 insects, taxon-specific global reviews have also been compiled, as is the case for tardigrades
100 (delBarco-Trillo, 2019) and copro-necrophagous beetles (Ramirez-Restrepo & Halffter, 2016), which

101 have built a clearer picture of how these taxa are impacted by urban environments. The other
102 reviews we found focused on the responses of specific taxa in particular regions or subject to specific
103 disturbances, or, examined urban systems from the broader perspective of understanding the
104 impacts of land-use changes, but were not directly linked to understanding the ecology of species
105 and patterns of diversity in urban environments (Table 1, Table S1).

106

107 'Urban' and 'Urbanisation' are perceived and described somewhat variably across the literature. In
108 general, urban areas are viewed as being characterised by high human population densities and built
109 infrastructure with impermeable surfaces, which make these ecosystems distinctive in many
110 respects (e.g., climate, hydrology, habitat patchiness, biodiversity). Within this, individual urban
111 areas have very different characteristics and shapes. For example, they vary greatly in size, with
112 large metropolises and small villages whose contours are defined not only by urban sprawl, but also
113 on a political, social or economic basis (MacGregor-Fors, 2011). Additionally, urban density is
114 increasingly a descriptor seen in the urban ecology literature and serves complement descriptions of
115 urban size and scale. Analysis and definition of the term 'urban' is beyond the scope of this review,
116 but it is important to appreciate the variability in the use of the term (and related terms such as peri-
117 urban, non-urban, suburban etc.). This semantic variability makes direct comparison among studies
118 challenging across study systems and countries, or when attempting to extrapolate findings to other
119 urban contexts.

120 **Key emergent themes**

121 [Habitat quality, quantity and pattern - land use and movement ecology](#)

122

123 Urban areas are a complex mosaic of land covers, including largely inhospitable and impermeable
124 built environments (e.g., residential, commercial, industrial and infrastructural, including
125 transportation infrastructure), interspersed with green (e.g., remnant vegetation, street trees, parks,

126 gardens, road verges and roundabouts) and blue (e.g., lakes, rivers, ponds, wetlands, drains and
127 canals) spaces. The patchiness of natural or semi-natural habitat this creates in urban areas can
128 deliver a wide variety of habitats juxtaposed at small scales. While research has traditionally focused
129 on viewing urban environments as hostile or being barriers and obstacles for wildlife, there is now,
130 increasing evidence suggesting many species can thrive in these anthropogenic environments,
131 including some conservation-relevant species (Plummer *et al.*, 2024). The assessment of each of
132 these habitats, in terms of suitability for supporting insect diversity, depends on the taxonomic
133 identity of any species and the connectivity with other habitat patches (e.g. Braaker *et al.*, 2014; Su
134 *et al.*, 2015; Perry *et al.*, 2020). Home gardens, urban agriculture, road verges and unbuilt (green- or
135 brown-field) land often provide abundant resources and refugia for many insects (Jones & Leather,
136 2012; McKinney, 2021). Patch quality matters, though is taxon-subjective, and those with greater
137 heterogeneity can support greater insect activity, diversity and abundance (Helden & Leather, 2004;
138 Azhar *et al.*, 2024). These fragmented urban habitats can also experience varying and under-
139 recorded degrees of management and disturbance, for example mowing or unregulated use of
140 pesticides in residential areas mediate the environmental and ecological factors which influence
141 insect diversity in urban areas (Helden & Leather, 2004; Lowe *et al.*, 2019; Sanetra *et al.*, 2024).

142

143 For many insects inhabiting urban green spaces, the broader city can appear as a set of habitat
144 islands embedded within an inhospitable matrix (Johnson & Munshi-South, 2017; Fattorini *et al.*,
145 2018). The dynamics of their populations in cities may thus be affected by many of the same
146 ecological processes that drive spatially fragmented populations in other environments, where
147 dispersal limitation is of central importance (Hanski, 2001). As a result, isolated patches of green
148 space in urban areas often show reduced abundance and diversity of insects (Jones & Leather,
149 2012). Fattorini *et al.* (2018) assessed several predictions of Island Biogeography theory for urban
150 insects and found strong support for increased insect species richness in larger patches of green

151 space, and in patches better connected to rural areas or to other green spaces that can act as
152 potential sources of immigration. These effects are not universal however, highlighting potentially
153 complex spatial dynamics of urban environments (Casanelles Abella et al., 2024, Hill et al., 2024).
154 The degree of isolation between insect populations in urban areas is mainly defined by intervening
155 land use and distance between green spaces or to other non-urban habitats (Jones & Leather, 2012;
156 Fattorini et al., 2018). Beyond distance alone, different land cover type and physical barriers can
157 further affect insect movement in urban areas. Peralta et al. (2011) found that walls restrict
158 movement of a leaf miner, while Corcos et al. (2019) found that at a local scale (< 50 m), buildings
159 acted as barriers to movement and reduced species richness of tachinid parasitoids. Conversely, the
160 presence of ecological green corridors connecting green spaces considered as habitat patches
161 facilitates insect movement in urban areas and enhances the metacommunities of insects,
162 centipedes and spiders in these patches (Balbi et al., 2020, Vergnes et al., 2012, Xu et al., 2024).

163

164 Urban density may also select for dispersal-enhancing morphologies within populations or species
165 assemblages, which has been proposed for flight-related traits of aquatic beetles (Liao & Lin, 2024),
166 as well as body size of wild bees (Brasil et al., 2023) and macro moths (Merckx et al., 2018b). These
167 patterns, which are also considered a form of ecological filtration, may reflect changes in the costs
168 and benefits of dispersal in urban environments (Jones & Leather, 2012; Ancillotto & Labadessa,
169 2024), though may also reflect traits of founder individuals in low-dispersal systems and conclusions
170 are largely uncertain (see also genetics section below) (Federico, Guralnik & Belitz, 2024).

171

172 Anthropogenic Pollution: heat, noise, light and other contaminants

173

174 Beyond restricted availability of favourable habitats, cities are characterized by other land use
175 factors that can influence insect survival and reproduction. The **urban heat island effect (UHI)**,

176 caused by increased solar energy absorption by anthropogenic materials and surfaces (Phelan et al.,
177 2015), elevates average temperatures and reduces amplitudes of daily thermal profiles compared to
178 surrounding non-urban environments. The UHI effect is particularly pronounced at night, as artificial
179 surfaces release solar radiation accumulated during the day as heat, whereas in non-urban
180 environments this energy is largely absorbed by plants to contribute to primary production.
181 Appropriate climate and temperature are crucial for all ectothermic organisms, accordingly, strong
182 associations are reported between urban temperatures, the abundance and species richness of
183 many taxa (Hansen et al., 2005), and the variability of phenotypes identified within species among
184 populations dispersed along a gradient of urban intensity (Merckx et al., 2018a, b, [Ombugadu et al.](#)
185 [2024](#)). These patterns are the outcomes of physiological (Hamblin et al., 2017) and metabolic
186 (Chown & Duffy, 2015) opportunities and constraints, which shape individual life-history trait
187 expression, phenological timing and may affect immunity (Merckx et al., 2018a, b). A species'
188 phenological shift in response to warmer conditions is thus likely to alter the nature of its
189 interactions with the biotic environment, for instance with its resources or natural enemies
190 potentially even beyond the limits of cities (Zhang et al., 2004, Meineke et al., 2013, [Casanelles](#)
191 [Abella et al., 2024](#)).

192 **Anthropogenic noise** is omnipresent in urban environments and is another novel challenge for many
193 species. Noise in cities has increased since the industrial revolution as our societies shift towards
194 operating day and night. Almost half the studies of noise pollution to date have focused on urban
195 noise, including the impact of road noises (Jerem & Mathews, 2021). Vertebrates dominate these
196 impact studies and effects on invertebrates are little known (Morley et al., 2014; Jerem & Mathews,
197 2021). Yet, sound is involved in many invertebrate behaviours, particularly mate-finding and
198 courtship, and the ability of invertebrates to hear sounds over a wide range of frequencies, including
199 the spectrum of anthropogenic noise, suggests that these species are very likely to be negatively
200 impacted by noise pollution (Hoy & Robert, 1996).

201 **Artificial light at night (ALAN)** is another constraint for many species in-and-around urban areas
202 (Hölker et al., 2010, Stewart, 2021). The spatial dimension of ALAN is considerable as light pollution
203 extends beyond the geographical limits of urban ecosystems (Zapata et al., 2019) and is a global
204 phenomenon. Currently, 80% of the human population lives under light-polluted skies (Falchi et al.,
205 2016) and there is a predicted growth in global ALAN of 6% per year (Hölker et al., 2010). Over 60%
206 of invertebrates are nocturnal, and many diurnal species remain active during crepuscular hours. For
207 these, light pollution is likely to have a major impact (Hölker et al., 2010). Moreover, light affects not
208 only terrestrial organisms but also aquatic organisms. These effects can be direct, but also indirect
209 through trophic web modification as raised primary production can, in turn, affect higher trophic
210 levels (Zapata et al., 2019). The impact of light pollution on fireflies is well-documented, as not only
211 is light pollution likely to reduce the activity period of these nocturnal species, but it can also alter
212 the visibility of the light signals they produce to communicate with conspecifics and other organisms.
213 The ALAN effect has also been shown to increase their risk of predation by reducing the aposematic
214 effect of their luminescence, disrupt their orientation, and interfere with sexual communication
215 which then affects reproduction (Lewis et al., 2020; Vaz et al., 2021).

216 Urban areas are also prone to **chemical and biological pollution** from human activities. The impact
217 of urban water pollution on aquatic invertebrates is well documented (e.g. Windsor et al., 2019), but
218 there is scope for further research focused on novel chemical challenges and in diverse aquatic
219 environments across understudied regions of the world (Ferzoco et al., 2023). Numerous chemical
220 and biological pollution pathways in urban areas are evident, but these urgently need further
221 investigation to understand impacts on insect species and communities. Pollutants that require
222 attention include the effect of volatile organic compounds on olfactory learning and memory
223 (Leonard et al., 2019, Ryalls et al., 2022), potential impacts of pet parasiticides on urban freshwater
224 invertebrates (Preston-Allen et al., 2023), and impacts of soil pollution on a range of below and
225 above ground insects (Audusseau et al., 2017; Scott et al., 2023).

226 Anthropogenic alterations in disturbance regimes such as fire suppression in fire adapted systems,
227 flow regulation in urban rivers and other altered abiotic regimes may have important effects on
228 invertebrate populations but remain understudied.

229

230 Roads as both barriers and corridors

231

232 Roads and other transport infrastructures, such as canals or railway lines, are features of urban
233 landscapes and have received considerable attention for their influences on animal movement. The
234 effects of roads as barriers to animal movement are well established, particularly for vertebrates
235 (Fahrig & Rytwinski, 2009; Benítez-López et al., 2010). There is now increasing evidence that roads
236 may also influence the movement of some insects and other arthropods acting as both strong
237 barriers to insect crossing and as corridors facilitating movement along road verges (Muñoz *et al.*,
238 2015; Vиллемey *et al.*, 2018) .

239 Roads can fragment insect habitats and populations through various mechanisms: insects may avoid
240 roads, be directly obstructed by them, or be killed when trying to cross them. While crossing roads
241 can be associated with excess insect mortality from vehicle collisions (Munguira & Thomas, 1992;
242 Rao & Girish ,2007; Soluk et al., 2011), this is difficult to quantify as methodological differences
243 among studies prevent general conclusions and comparisons between taxa or regions (Muñoz et al.,
244 2015). The extent to which roads obstruct movement and contribute to population fragmentation is
245 highly variable and dependent on species' traits, which in turn interact with road and traffic
246 characteristics (Munguira & Thomas, 1992; Bhattacharya et al., 2003; Melis et al., 2010; Martin et
247 al., 2018; Fitch & Vaidya, 2021). For example, Muñoz et al. (2015) identified roads as major
248 impediments to movement of small or flightless insect species though mobile, flying species varied
249 in response, while Andersson et al. (2017) also confirm a stronger barrier effect of roads for smaller
250 bees and wasps.

251 The potential for road verges to act as corridors facilitating insect movement is increasingly
252 recognised. For example, while roads were barriers to transverse movement, pollinators used road
253 verges as corridors (Dániel-Ferreira et al., 2022). Movement across roads was not affected by traffic
254 intensity, but road verges with more diverse and abundant floral resources retained more insects
255 (Dániel-Ferreira et al., 2022). Our understanding of the importance of road verges as habitats and
256 corridors for pollinators, how they facilitate insect movement and dispersal, and the impact of
257 management remains limited (Villemey et al., 2018). Some studies comparing movements along
258 roads and road verges to movements in other habitats are contradictory (e.g., Brunzel et al., 2004;
259 Vermeulen, 1994) limiting generalisation about their role in dispersal. In general, work on the barrier
260 effects of linear transportation features (e.g. roads, railways etc.) on insect movement dominates
261 that on their role as potential corridors (Villemey et al., 2018).

262 [Impacts on genetic diversity](#)

263 Adaptations that arise from selective pressures of urban environments have been addressed
264 comprehensively elsewhere (Szulkin et al., 2020; Winchell et al., 2022). Here we consider effects of
265 the urban landscape on movement and dispersal, which are likely to translate into effects on gene
266 flow, with consequences for genetic structure and diversity of urban insect populations (e.g. Xu et
267 al., 2024). Overall, the fragmented nature of insect populations that rely on urban greenspaces is
268 predicted to result in high genetic differentiation among, and reduced genetic diversity, thus
269 reduced effective population size, within the local populations (Keyghobadi, 2007; Johnson &
270 Munshi-South, 2017). The effects of habitat fragmentation by roads on genetic differentiation and
271 diversity of urban insects is increasingly well-studied, with work on crickets (Holzhauer et al., 2006;
272 Vandergast et al., 2009) and beetles (Mader, 1984; Keller & Largiadér, 2003; Keller et al., 2004; Keller
273 et al., 2005; Holzman et al., 2009) supporting these predictions. These effects contrast with those for
274 some human commensals and parasites, for example human-biting mosquitoes, where urban roads
275 appear to facilitate gene flow, most likely through passive transport (Egizi et al., 2016; Schmidt et al.,
276 2017); the extent to which other invertebrates may experience passive transport through urban

277 environments is not clear and has received little attention. Other urban infrastructure (e.g.,
278 commercial, industrial and residential buildings and zones) was found to limit gene flow in many
279 insects, including bees, butterflies and odonates (Watts et al., 2004; Davis et al., 2010; Crawford et
280 al., 2011; Jha & Kremen, 2013; Jha, 2015), although such effects are not universal. For example,
281 urban infrastructure overall had limited influence on genetic structure in a globally rare butterfly
282 (Leidner & Haddad, 2010) and a woodland associated ant (Khimoun et al., 2020). Furthermore, many
283 studies group all urban infrastructure into a single land cover class and may miss more subtle effects
284 of different types of infrastructure, or different use and management of such infrastructure, on
285 insects.

286

287 An important ecological insight arising from genetic studies of parasite or commensal insects is that
288 the potential effects of increased gene flow via human transport may be countered by extinction-
289 recolonization events and bottlenecks driven by pesticide use and other control methods used in
290 urban settings. For example, urban populations of bed bugs tend to contain low levels of genetic
291 diversity and be strongly differentiated from each other, with no pattern of isolation by distance, all
292 consistent with bottlenecks resulting from strong founder effects (Saenz et al., 2012; Fountain et al.,
293 2014). The extent to which bottlenecks affect and shape morphological or genetic diversity in other
294 insects of urban environments is poorly known. Scale also has an important influence on genetic
295 patterns and processes in commensal species, as they often display low levels of genetic
296 differentiation at very fine scales where they may be able to disperse readily and rapidly (e.g., within
297 single buildings), but high levels of differentiation at larger scales (e.g., among buildings) as a result
298 of either reduced movement and gene flow (as in German cockroaches; Crissman et al., 2010) or
299 bottlenecks and founder events (as in bed bugs; Booth et al., 2012; Fountain et al., 2014). Patterns
300 of movement or genetic structure across different scales in other, non-commensal insects have
301 received little attention.

302

303 [The influences of non-native or invasive species](#)

304 Urban plant biodiversity has a high proportion of non-native species relative to less disturbed and
305 curated rural areas largely through human selection as urban planners and gardeners often choose
306 exotic plant species for their ornamental appeal (Aronson et al., 2015; Blair and Johnson, 2008,
307 Cadotte et al., 2017). Non-native plants can, however, function as insect resources and shelters and
308 may remain beneficial for many invertebrates despite having potential, if ecologically dominant or
309 invasive, to alter ecosystem function, cause sharp reductions or even local population extinctions in
310 other species (Levine et al., 2003; Moles et al., 2012; Pyšek et al., 2017). Although there is no
311 consensus at the scale of multiple taxa (Shochat et al., 2010), the increase in non-native plant
312 species diversity with urban density (Chen et al., 2014) does not appear to compromise the diversity
313 of native species (Cadotte et al., 2017), although the relative abundance of these two groups may be
314 altered along the gradient of urban density. We understand little about the effects of non-native
315 plants on insects and their interactions and these may be dependent on species identities and
316 environmental factors. For example, non-native plant species may be less attractive or palatable for
317 native herbivorous and florivorous insects (Helden et al., 2012; Grunzweig et al., 2015; Jain et al.,
318 2016), but some urban pollinator species can benefit from exotic plants, which extend the temporal
319 window of nectar and flower availability (Wenzel et al., 2020).

320 Invasive insect species are by definition good dispersers, a trait that is selected for by the habitat
321 patchiness of urban environments (see movement ecology above). Invasive and non-native
322 invertebrates themselves can also have damaging effects in urban environments. For example,
323 invasive social wasps can establish quickly and reach high densities in urban areas subsequently
324 threatening native insect species (Baker & Potter, 2020), and invasive ants can disrupt local plant-
325 insect interactions and facilitate outbreaks of native herbivorous species (Brightwell and Silverman,
326 2009).

327 **Priorities for urban insect ecology research and conservation**

328 Promoting insect conservation and diversity in urban areas presents many challenges, not the least
329 of which is a general unfamiliarity with, or caution of, insects in the urban human population. There
330 are signs that this may be changing, largely through the increased visibility and consciousness of
331 insect declines (Didham et al., 2020). Nevertheless, this overview of reviews helps us to identify
332 some key areas in which further research will promote the conservation and diversity of insects as
333 part of the urban fabric.

334 [Effects of pattern and scale - addressing a current research bias](#)

335 Urban systems present an opportunity to test and apply a range of ecological theories and concepts
336 that have emerged from work in natural systems, such as trait frameworks, metapopulation theory,
337 and biodiversity-ecosystem function links (e.g. Aronson et al., 2016; Grilo et al., 2022). Urban areas
338 encompass a range of interlinked ecosystems, and more work is needed to understand how
339 ecological patterns and processes vary among urban environments and when compared to natural
340 systems, and how urban patterns may scale up to macroecological processes (Hahs & Evans, 2015;
341 Lokatis et al., 2023).

342 Patterns and scales of urbanisation vary across countries and regions. The majority of research has,
343 however, been conducted in large cities or highly urbanised regions in North America and Europe
344 (Collins et al., 2021), which are not representative of the diversity of urban environments in other
345 parts of the world. This creates a knowledge bias and drives a need for more research on how urban
346 processes impact insect population dynamics and interactions in understudied countries and urban
347 forms, and how insect diversity, population dynamics and interactions vary in other patterns of
348 urban space relative to larger cities and connected metropolises. We both call for and encourage
349 this. Of the 18 articles in this SI, almost half are from understudied regions including analysis of
350 butterfly communities in urban Nigeria ([Ombugadu et al. 2024](#)) and both plant-associated arthropod

351 communities and bee diversity in Brazilian cities (Barao et al., 2024; Pereira, Zamoner & Gonçalves,
352 2024).

353

354 [The influences of climate change in urban areas](#)

355 A key goal for urban biodiversity research and conservation is to understand how climate change
356 impacts intersect with urbanisation. Heat waves in particular are predicted to become more
357 frequent, longer and more severe (Schoetter et al., 2015), yet impacts of temperature extremes and
358 associated water stresses on insects and their interactions are poorly understood (Harvey et al.,
359 2020). Understanding whether, and at what pace, insects adapt thermal tolerance will be critical to
360 predicting future urban ento-diversity. There will likely be high variation across species, populations
361 and communities, depending on thermoregulatory ability, behaviours, and microclimate associations
362 (e.g. Miler et al., 2020; Tüzün & Stoks, 2021), and there is a need for a diverse range of field and lab-
363 based studies to increase our understanding of these. Life cycle studies would help evaluate the
364 potential of species to adapt to future urban climates, and this is particularly relevant for species
365 with life cycles dependent on a range of habitats such as odonates (Raebel et al., 2012). One
366 methodological challenge to this is that climatic data are often not available at a resolution that
367 would allow researchers to study fine-scale variation in species assemblage or abundance. In this,
368 and in other areas such as pollution monitoring, the increasing availability of low-cost monitoring
369 devices and the expansion of meteorological urban networks will be transformative. Existing studies
370 that investigate the impact of the urban climate, and/or the UHI effect, often rely on a comparison
371 among a limited number of sites distributed along an urbanisation gradient, without considering
372 their spatio-temporal dimensions, fine-scale variations and covariance with the micro-local scale
373 (Kaiser et al., 2016, Pincebourde et al., 2016). Several works in this SI do explore elements of this
374 complexity as Omugadu et al. (2024) consider vegetation complexity and temperature regimes and
375 Azhar et al. (2024) investigate ant foraging as a function of microclimatic variables.

376 Less studied, but critical, is the reduced amplitude of temperatures over a diurnal cycle in cities,
377 especially during heatwaves as limited night-time cooling will limit nocturnal physiological recovery
378 in plants and animals. These effects linked to temperature are often coupled with risk of desiccation
379 (Karlsson & Wiklund, 1985, Gibbs & Matzkin, 2001), and more research is needed exploring the
380 interaction and relative impact of these two variables on urban biodiversity.

381 [Novel environments and ecological traps](#)

382 Urban environments provide many novel habitats and spaces for biodiversity. The ability of some
383 invertebrates to adapt to these is documented, including green roofs (Wang et al., 2022), golf
384 courses and lawns (Colding & Folke, 2009), and abandoned or waste lands (McKinney, 2021).
385 Successful insect use of these habitats, such as ornamental garden ponds (Hill et al., 2024) or urban
386 pavements (Noël et al., 2024), may be influenced by context-specific environmental, management
387 and social factors. Some studied habitats can be locally-specific to countries, regions or urban
388 typologies, and may be inherently tied to local social and political history. For example, artificial
389 canals (Lin et al., 2020), rooftop agriculture (Appolloni et al., 2021), and urban green infrastructure
390 (Liberalesso et al., 2020) differ in distribution, ecological history and policy support around the
391 world. Urban green infrastructure may be incidental or designed to follow local cultural norms, but
392 the need for these to meet multiple objectives, including biodiversity and active human use creates
393 novel constraints to insect populations. Another important area of research needed is to
394 understand better how, and under what circumstances, novel urban features create ecological traps
395 in which insect mortality may outweigh reproduction. The trap effects of ALAN are well-documented
396 (Stewart, 2021), but there are potentially many other ecological traps present in urban areas around
397 the globe, including solar panels or glass surfaces that can appear as water habitats to aquatic
398 insects (Horváth et al., 2010), or road corridors that can support dispersal but increase mortality risk
399 (Lehtonen et al., 2021).

400 [Methodological advances](#)

401 Addressing these priority research areas will require new methods or novel applications of existing
402 methods and will also depend on greater understanding of the effectiveness of standard and best
403 practice entomological trapping techniques (Montgomery et al., 2021). There is scope for research
404 to explore whether the effectiveness of trap types and sampling methodologies varies among urban
405 environments and relative to non-urban areas, or to develop new or modified methods that
406 minimise the risk of human impact on fragile insect populations. Use of environmental DNA (E-DNA)
407 techniques is becoming more accessible and evaluating its site-precision in the urban area is
408 essential to improving its utility and applicability (Roger et al., 2022).

409 Urban centres also provide new opportunities for crowd-sourcing methods and data collection
410 through citizen science participation, either through targeted projects designed for community data
411 collection (Plummer et al., 2023) or by collating and analysing online biodiversity records to
412 understand urban diversity trends (Pereira, Zamoner & Gonçalves et al., 2024). These stakeholder
413 techniques also serve to engage and habituate the wider public to the presence and value of insects.

414 [Social and historical influences](#)

415 Context matters and urban areas are inherently human-regulated spaces, yet urban ecology
416 research has largely focused on understanding biodiversity patterns in relation to environmental
417 attributes with limited treatment of the social, cultural and political factors that underpin and
418 moderate these. History of land use, planning and development, as well as current use and
419 socioeconomic factors, will influence the diversity possible at any given place or time (Leong et al.,
420 2018; Windsor et al., 2019), but these human influences are not commonly analysed or discussed in
421 studies. These human-mediated factors also need to be considered when comparing studies
422 between countries and regions and synthesising literature to identify patterns as management,
423 social and political histories of urban areas have strong influences on biodiversity that may not be
424 apparent through simple landscape attributes. For example, major European cities that have
425 themselves evolved over thousands of years (e.g. Rome, Athens) will potentially support very

426 different biodiversity patterns to the modern, higher density cities that have undergone radical
427 transformation and expansion (e.g. Singapore, Guangzhou), or to newer cities emerging with a firm
428 conscience of local vernacular (e.g. Kampala, Fez) or ecological function (e.g. 'sponge cities' in
429 China). These, and other influences such as wars or social conflict, potentially temper emergent
430 ecological patterns and our understanding of this is limited.

431 Landscape approaches are essential for urban conservation planning and future research should
432 consider how private and public land management together influence insect diversity in urban areas.
433 Much urban ecology research has focused on publicly accessible lands , although work in Sheffield
434 two decades ago revealed the critical contribution of private garden biodiversity and the importance
435 of their management (Gaston et al., 2004). Management, insect tolerance and planting approaches
436 vary widely on private land and will influence insect diversity and movement ecology beyond the
437 boundaries of the property. If this variation is to promote overall insect diversity, then it must be
438 better understood and effective communication with private urban landholders improved.

439 An additional challenge for urban insect conservation is to understand, manage and optimise trade-
440 offs between social, cultural and ecological factors. Habitat, resource or management attributes that
441 support insect diversity aren't always suitable for, or desirable in, human-used spaces. Weedy floral
442 resources may be ideal for insects but can be perceived as messy by people (Lowenstein et al., 2019)
443 and this impedes the many benefits of reduced mowing frequency (Wintergerst et al., 2021).

444 Elements such as carrion are an even greater challenge in urban areas (Barton et al. 2013).

445 Similarly, opportunities to apply Integrated Pest Management strategies in urban areas, which would
446 ultimately benefit humans and most insect species through reduced pesticide use, are hindered by
447 limited information and poor communication as well as varied regulation and data on household
448 pesticide use (Lowe et al., 2019). Urban beekeeping (keeping colonies of *Apis mellifera* L., an invasive
449 species in many parts of the world) is increasing in popularity around the world due to misleading
450 promotion of this to support bee conservation. Recent studies, however, suggest that a focus on

451 *Apis* honeybees does little to raise awareness of insect conservation generally (Smith & Saunders,
452 2016). The impacts of this practice on local pollinator communities are still poorly understood
453 (Graystock et al., 2016; Casanelles-Abella & Moretti, 2022; MacKell et al., 2023) and may negatively
454 impact the richness, abundance, and activity of wild bees, even in areas of *A. mellifera*'s native range
455 (Ropars et al., 2019; MacInnis et al., 2023). More research is critical to understanding the impacts
456 and potential misuse of honey bees as a conservation flagship species.

457 **In summary**

458 Urban ecology is a growing research field relevant to addressing many contemporary environmental
459 challenges, but knowledge of insect ecology and diversity in urban environments remains limited.
460 Many works indicate a homogenisation and decrease in functional type, species diversity and/or
461 insect abundance as urban density increases (in this SI see: Barao et al., 2024; Casanelles Abella et
462 al., 2024; Federico, Guralnik & Belitz, 2024; Rivest & Kharouba, 2024; Sanetra et al., 2024;
463 Svenningsen et al., 2024). This picture is, however, nuanced (e.g. Ancillotto & Labadessa, 2024;
464 Federico, Guralnik & Belitz, 2024; Ombugadu et al., 2024) and while urban areas can detrimentally
465 impact some insect taxa through the predominance of artificial environments, constrained habitats
466 and disturbances, other taxa can adapt and thrive in the mosaic of semi-natural and novel
467 environments, management variation and abundant resources (Plummer et al 2023; Curry et al,
468 2024; Hill et al., 2024; Nunes et al., 2024; Xu et al., 2024 2024). Lessons from metapopulation
469 ecology and a better understanding of patch quality for specific taxa will be critical to raising urban
470 insect diversity and conserving this as a stable ecosystem component (e.g Azhar et al., 2024; Liao &
471 Lin, 2024). There are pathways to understanding patch improvement, but above this, integrating
472 connectivity and opportunity for insects in urban areas more broadly and by design will contribute
473 further (e.g. Noel et al., 2024) as will translocations to restore extirpated diversity (e.g. Yagui,
474 Kearney & Hoffmann, 2024).

475 This overview provides information on areas in which further research is needed, essentially
476 providing guidance on the WHAT we need to know. Integral to being able to drive insect diversity
477 and conservation in urban areas is the understanding that these are, above all, human-dominated
478 areas and that people matter. Making greater use in research of citizen science techniques (eg.
479 [Plummer et al 2023](#); [Pereira, Zamoner & Gonçalves, 2024](#)) will then serve to underpin
480 communications that will support biodiversity and conservation aims and the ecosystem functions
481 on which we depend. The relationship between humans and insects in the urban environment
482 should be explored further using social science and stakeholder engagement techniques and will
483 then help us understand HOW we can achieve stable, insect-diverse urban areas.

484

485 This Special Issue has assembled a range of relevant papers. These and others will emphasise the
486 need for further research, but also for regular accessible syntheses to inform urban conservation
487 action.

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