Environmental factors affecting insect pollinators and pests at urban farmlands: a case study with honey bees and cabbage white butterflies

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Rapid urbanization results in a significantly increased urban population, but also the loss of agricultural lands, thus raising a concern for food security. Urban agriculture has received increasing attention as a way of improving city citizens' food accessibility and local people's livelihood. Although vegetable-dominant small urban farmlands are relatively common in China, little is known about environmental factors associated with insects that could affect ecosystem services at these urban farmlands, which in turn influence agricultural productivity. Using Asian honey bee (*Apis cerana*) and cabbage white butterfly (*Pieris rapae*) as examples, I investigated how environmental features within and surrounding urban farmlands affected insect pollinator (bee) and pest (butterfly) abundance in a megacity of China during winters. I considered environmental features at three spatial scales: fine (5m-radius area), local (50m-radius area), and landscape (500m-raidus and 1km-radius areas). While the abundance of cabbage white butterfly increased with local crop diversity, it was strongly negatively associated with landscape-scale crop and weed covers. Asian honey bees responded positively to flower cover at the fine scale. Their abundance also increased with local-scale weed cover but decreased with increasing landscape-scale weed cover. Although farmland fragmentation, i.e., patch density of farmland within a landscape did not affect cabbage white butterfly, the abundance of Asian honey bees tended to decrease with increasing farmland fragmentation. These results suggest that cultivating too diverse crops at urban farmlands can increase crop damage; however, the damage may be alleviated at farmlands embedded in a landscape with more crop cover. Retaining small amount of un-harvested flowering crops and weedy vegetation within a farmland, especially less fragmented farmland can benefit honey bees when natural resources are scarce.

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18 **Abstract**
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20 agricultural lands, thus raising a concern for food security. Urban agriculture has received
21 increasing attention as a way of improving city citizens' food accessibility and local people's
22 livelihood. Although vegetable-dominant small urban farmlands are relatively common in China,
23 little is known about environmental factors associated with insects that could affect ecosystem
24 services at these urban farmlands, which in turn influence agricultural productivity. Using Asian
25 honey bee (*Apis cerana*) and cabbage white butterfly (*Pieris rapae*) as examples, I investigated
26 how environmental features within and surrounding urban farmlands affected insect pollinator
27 (bee) and pest (butterfly) abundance in a megacity of China during winters. I considered
28 environmental features at three spatial scales: fine (5m-radius area), local (50m-radius area), and
29 landscape (500m-raidus and 1km-radius areas). While the abundance of cabbage white butterfly
30 increased with local crop diversity, it was strongly negatively associated with landscape-scale
31 crop and weed covers. Asian honey bees responded positively to flower cover at the fine scale.
32 Their abundance also increased with local-scale weed cover but decreased with increasing
33 landscape-scale weed cover. Although farmland fragmentation, i.e., patch density of farmland
34 within a landscape did not affect cabbage white butterfly, the abundance of Asian honey bees
35 tended to decrease with increasing farmland fragmentation. These results suggest that cultivating
36 too diverse crops at urban farmlands can increase crop damage; however, the damage may be
37 alleviated at farmlands embedded in a landscape with more crop cover. Retaining small amount
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39 fragmented farmland can benefit honey bees when natural resources are scarce.

40

Introduction

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42 Urban expansion in many countries is often accompanied by the loss of agricultural lands (Bren

43 d'Amour et al., 2017). The world's built-up areas are a quarter of total agricultural lands;

44 however, the expansion of built-up areas is faster than urban population growth and takes place

45 on productive agricultural lands, leading to reduction in global crop production (Seto &

46 Ramankutty, 2016; Bren d'Amour et al., 2017). Moreover, the proportion of global population in

47 urban areas is forecast to increase from 55% in 2018 to 68% by 2050 (United Nations, 2019).

48 Approximately 90% of the increase would occur in Asia and Africa, which are also the hotspot

49 of future urban expansion. These changes, i.e., growing urban population and agricultural land

50 loss, raise concerns for not only food security but also for the livelihoods of smallholders,

51 especially in developing countries (IFPRI, 2017; Huang et al., 2020).

52 Urban agriculture has received increasing attention as one of the practices that may

53 improve the accessibility of food, the livelihood of local people such as farmers and rural

54 migrants, and human well-being (Deelstra & Girardet, 2000; De Bon et al., 2010; Mok et al.,

55 2013; Martellozzo et al., 2014). One recent study also shows that small-scale urban agriculture

56 can bring high yields (McDougall et al., 2018). Urban agriculture can be broadly defined as

57 agricultural production, such as vegetables, fruits, ornamental plants, and other dietary products,

58 occurring in urban environments that include both inner city and city fringe areas (Mougeot,

59 2000). In China, small-scale vegetable farming is common in cities, especially at vacant lots and

60 city fringe areas. Urban agriculture, also called peri-urban agriculture in some cases, has been

61 supported by the Chinese government policy "Vegetable Basket" for decades (Zong et al., 2021).

62 Like other countries, urban expansion has been a major driver causing farmland loss in China

63 (Tu et al., 2020). Although farmland protection policy started in mid-1990s has alleviated the

64 loss to some extent, urban development frequently occurs on highly productive farmlands near

65 urban edge areas, often leaves reclaimed but less productive farmlands as an offset for the

66 development, and increases farmland fragmentation (Liang et al., 2015; Huang et al., 2019; Tu et

67 al., 2021). In this situation, understanding ecological factors that could affect agricultural

68 production at urban farmlands is crucial to promote the sustainability of these farmlands.

69 Insects are well known for their role in ecosystem service (e.g., pollination and pest

70 control) and disservice (e.g., crop damage) in agricultural landscapes (Losey, 2006; Kremen &

71 Chaplin-Kramer, 2007; Oliveira et al., 2014; Omkar, 2018). Their effects on crop yields have

72 been widely studied. Over 70% of main global crops benefit from animal-mediated pollination

73 (Klein et al., 2007) and 9-25% of staple food crops are estimated to be lost due to animal pests

74 (Oerke, 2006), in which insects are the major group in both cases. Bees are considered the most

75 important pollinators. For example, in USA, 11% of the agricultural gross domestic product in

76 2009 depended on pollination, mainly contributed by bees (Lautenbach et al., 2012). Bee

77 abundance affects yields of certain crops such as oilseed rape more than pesticides do (Catarino

78 et al., 2019 and references therein). Diverse wild bees support pollination of multiple crops in

79 urban gardens (Lowenstein et al., 2015). A number of factors, e.g., quantity and quality of floral

80 resources, semi-natural vegetation, crop diversity such as crop compositional or configurational

81 heterogeneity and landscape complexity, influence the abundance and diversity of bees (Klein et
82 al., 2012; Scheper et al., 2013; Potts et al., 2003; Priyadarshana et al., 2021; Raderschall et al.,
83 2021).
84   In contrast to bees, cabbage white butterflies (e.g., *Pieris rapae* and *P. brassicae*) are
85 recognized as agricultural pests because their larvae, cabbageworms, feed on the family
86 Brassicaceae crops (cruciferous crops) and can severely damage these crops. For example,
87 cabbage white butterflies caused about 70% yield loss in cruciferous crop in Meghalaya, the
88 northeastern state of India (Singh et al., 2014). While presence or abundance of Brassicaceae
89 crops is positively associated with cabbageworm density, this effect may be amplified if the host
90 plants are congregated as a large pure stand (resource concentration hypothesis; Root, 1973) or
91 sparsely distributed (resource diffusion hypothesis; Yamamura, 1999). The type of surrounding
92 crop has an impact on cabbageworm density (Maguire, 1984) and distance between cruciferous
93 patches influences egg density of cabbage white butterfly (Fahrig & Paloheimo, 1988). Floral
94 resource availability can also be important to the adult butterflies (Curtis et al., 2015).
95 Here, using Asian honey bee (*Apis cerana*) and cabbage white butterfly (*Pieris rapae*), I
96 investigated how environmental features within and surrounding urban farmlands in a megacity
97 of China influenced the abundance of these two species, during winter in which Brassicaceae
98 crops are of particular prevalent. I focused on farmlands largely cultivating vegetables, which are
99 dominant agricultural products of urban farming. I expected that the percentage of flowering
100 plants and Brassicaceae crops would positively affect honey bees and cabbage white butterflies,
101 respectively. Given that these farmlands are in urban areas mostly filled with built-up structures,
102 landscape weedy vegetation (spontaneous vegetation in vacant lots or parks) as well as available
103 farmlands could also be critical to both species. Crop diversity could negatively affect cabbage
104 white butterflies as found in the effect of crop diversification on insect pests in general (Hooks &
105 Johnson 2003; Beillouin et al., 2021). Alternatively, the effect may not be significant because
106 urban farmlands in this study region often contain more diverse crops than in rural agricultural
107 lands, and predator/parasitoid abundance is low in winter.
108

Materials & Methods

Study area

111 Guangzhou is the capital city of Guangdong Province, China (Fig. 1) and home to ~18 million
112 peoples. It also lies within the Indo-Burma Biodiversity hotspot region, the most urbanized area
113 among biodiversity hotspot regions in China (Güneralp & Seto, 2013). Guangzhou has a
114 subtropical monsoon climate: warm and dry winter and hot and humid summer, with a mean
115 annual temperature of 22.2°C (https://en.climate-data.org). During the past two decades,
116 farmlands located in city center area and between the center and boundary areas have drastically
117 decreased due to urban expansion and economic development (Shi & Shi, 2020).
118   I selected 33 farmlands in 2020 and 57 farmlands (23 from previous year) in 2021
119 (Fig.1). Several farmlands chosen in 2020 could not be surveyed in 2021 due to development,
120 abandonment, and too-early growth stage of crops (very small seedlings). The size of all

121 farmlands was $\geq$ 7,850m^2. Within a farmland, I established a 50m-radius area (site, hereafter)
122 that was also used for bird surveys in another study (Lee et al., 2022). All samplings were
123 performed within the 50m-radius area during January each year. Cabbage white butterfly was
124 surveyed at 90 sites (32 in 2020 + 58 in 2021) across two winters. Bee sampling was conducted
125 at 58 sites (25 sites in 2020 and 33 sites in 2021). Nearest distance between the centers of sites
126 surveyed for cabbage white butterfly was 1.87 ± 1.09km (mean ± standard deviation) in 2020
127 and 0.82 ±1.18km in 2021.
128   I did not distinguish between urban and peri-urban agriculture, including farmlands
129 located in the inner city as well as close to the city boundary (Mougeot, 2000). However, I
130 emphasize that all sites were in urban areas; built-up cover within a 500m-radius area of sample
131 sites averaged 61 ± 13.8%, consistent with the criterion of urban areas containing > 50% build-
132 up (Marzluff, 2000).
133
134 Cabbage white butterfly
135 Cabbage white butterfly (CWB) was surveyed twice ($\geq$ 12 days apart) using a transect method:
136 one survey in the morning (between 9-11:30am) and one in the afternoon (between 1:00-
137 4:00pm). During each survey, I placed randomly two 50m-line transects along a walk path within
138 a site. However, if Brassicaceae and non-Brassicaceae crop areas were clearly divided, I placed
139 one transect close to more Brassicaceae crops and the other near non-Brassicaceae crops. I also
140 changed the locations of transects between surveys. I walked the transect at a rate of 5m per
141 minute, and counted CWB individuals detected within a 10m x 5m x 5m imaginary window (5m
142 to each side of the transect, 5m from the ground, and 5m in front). Although a 5m width is
143 commonly used, I expanded the width of the imaginary window because CWBs were often
144 active slightly farther from the edge of path and moved along the row crops, which were not easy
145 to access because of irrigated water and farmer's attendance at some sites. Special care was taken
146 not to count the same individuals twice while walking along the transect. During the first winter,
147 I also carried out another survey using an area search method to verify the transect method. I
148 quickly walked around 50% area of a site, e.g., north-east-south side for 1 minute, and counted
149 CWB. This scanning process was repeated three times at different sides of the site, with 3
150 minutes between surveys. Data collected from transect method and area search methods were
151 highly correlated: Pearson's correlation (r) between maximum counts of each data was 0.94 (P <
152 0.001). Thus, I assumed that the transect survey data adequately represented CWB abundance in
153 this study.
154
155 Bee sampling
156 To capture bees, I used three-colored (blue, white, and yellow) pan traps (Campbell & Hanula,
157 2007). Three sample stations were established > 30m apart from each other within a site. Sample
158 stations were placed randomly from the center of site. However, most locations were restricted
159 by presence/absence of flowering plants (crops and weeds), farmer's permission, and
160 accessibility (avoiding deep furrows filled with water). I installed a set of three-colored pan traps

161 at each sample station. I attached pan traps to a pole at ~ 0.8m high, which was tall enough to
162 allow bees to see them, and placed them near flowering crops or weeds if they were present. Pan
163 traps were 1/2-2/3 filled with soapy water (10 drops of dish soap per liter) to remove the surface
164 tension. All traps were retrieved 2 days after installation. While I brought all bees captured to a
165 lab, washed, and preserved in 70% ethyl alcohol for identification, I counted only Asian honey
166 bees (Apis hereafter) because they were the main bee species captured and abundant enough to
167 conduct analysis.
168

Environmental data at fine and local scales

169
170 Local scale environmental data were collected within a site, that is, "site" represents the local
171 scale of this study. At each site, crop and non-crop features such as built-up structures (storage
172 house and path), herbaceous weeds, trees/shrubs, and open water (small pond and water channel)
173 were identified and marked on a printed satellite image downloaded from Google Earth
174 (https://www.google.com/earth/). The images were taken between 2018 and 2020. Many
175 farmlands were composed of highly diverse crop species, but each crop was cultivated in similar-
176 sized rows. Thus, I divided the site into several blocks. I identified crops and counted the number
177 of rows of each crop within a block. I summed all counts by block. In a block, I calculated the
178 relative frequency of each crop from the sum and converted that to "area" by multiplying it by
179 the size of the block. Sum of the crop's area across all blocks represented total area of the crop
180 within the site. All non-crop features and blocks were delineated in ArcGIS using georeferenced
181 Google Earth images as a base map.
182 I calculated the percentage of weedy vegetation (weed50) and cruciferous crops
183 (Brassicaceae; the mustard family). Crop diversity (cropdiv50) was calculated as a Shannon-
184 Wiener diversity index. While over 120 crops were found across sites, several crops were very
185 minor or variants of the same crop, sharing similar biological and ecological characteristics. I
186 also found cases of misidentifications when crops look similar, especially when they belong to
187 the same genus or family. Thus, I used the family of crop to calculate cropdiv50 (Table S1 for
188 the list of crop genera). At the family level, all sites had ≥ 2 crops and approximately 95% of
189 sites contained at least 6 crops, averaging 13.9 ± 5.4, and ranging from 2 to 24.
190 For Apis, crop surveys were also performed at a fine scale, a 5m-radius area surrounding
191 a sample station. Within the 5m-radius area, crops were identified and the percentage of each
192 crop was visually estimated. Similar to cropdiv50, I calculated fine-scale crop diversity based on
193 crop species (cropdiv5). In 2021, the percentage of flowering plants (flower) including crops and
194 weeds was also estimated at the fine scale.
195

Landscape data

196
197 I choose two landscape-scale sizes by considering flight distances of CWB and Apis (Jones et
198 al., 1980; Dyer & Seeley, 1991), the logistics of creating a land cover map, and the matrix
199 context of study sites, selecting 500m-radius and 1km-radius areas surrounding the center of site.
200 The 500m-radius area was used to examine compositional aspect of the landscape matrix in

201   which a farmland was embedded, and the 1km-radius area was used to characterize the spatial
202   configuration and fragmentation of farmlands within a landscape.
203   To generate the land cover map, I downloaded satellite images from Google Earth and
204   georeferenced them. Using the georeferenced images as a base map, I delineated 3 land cover
205   types within the 500m-radius area in ArcGIS: vegetable-dominant farmland (crop cover), weedy
206   vegetation (mostly spontaneous herbaceous vegetation in vacant lots and construction sites), and
207   built-up structure (building, road, any impervious surface). I then calculated the percentage of
208   farmland or crop cover (crop500) and weedy vegetation (weed500). I also delineated farmlands
209   within the 1km-radius area and calculated patch density (pd1000) and edge density of farmlands
210   (edge1000) using Fragstats v 4.2.1 (McGarigal et al., 2012). Although these indices may be the
211   simplest measure of the spatial configuration of habitat patch (McGarigal & Marks, 1995), they
212   can affect diversity and abundance of pollinators and butterflies by increasing edge habitats or
213   facilitating movements between patches (Flick et al., 2012; Hass et al., 2018; Martin et al., 2019;
214   among others). Edge density is the same as the total edge length in this study because landscape
215   sizes were identical across all sites.
216
Statistical analysis
218   I pooled 2 years of data together for CWB; even if the same site was surveyed both years, each
219   year's data were considered independent because local scale features such as crop diversity and
220   amount of weedy vegetation differed between two winters at the same site. Any potential bias
221   associated with this approach, i.e., year effect was also examined before final analysis (see
222   below). Among environmental variables, edge1000 was highly correlated with pd1000 and
223   crop500 (r = 0.83 and 0.68, respectively, P < 0.001 in CWB data), which increased the variance
224   inflation factor of these variables. Thus, edge1000 was not included in analyses.
225   I selected the maximum count of CWB between two transects and summed it over two
226   visits at each site. The sum was used as the "abundance" of CWB. I log(x+1)-transformed the
227   abundance of CWB to minimize potential bias caused by high abundance values and
228   overdispersion that can affect type I error. I first tested whether survey year (2020 or 2021) had
229   an effect on CWB abundance. I compared two generalized linear models (GLMs) with Gamma
230   distribution: intercept-only model vs model with a year variable. The AICc (Alkaike Information
231   Criterion adjusted for small sample size) value of the intercept-only model was lower (< 2) than
232   the other model, suggesting CWB abundance was not significantly affected by year-related
233   variations. Thus, I constructed 4 GLMs without "year" variable: null model (intercept-only
234   model), local model (3 local variables — Brassicaceae, cropdiv50, weed50), landscape model (3
235   landscape variables — crop500, weed500, pd1000), and full model (3 local and 3 landscape
236   variables). These four models were compared based on their AICc values. If there were no
237   competing models to a top model (model with a lowest AICc), that is, if other models had Δ
238   AICc (AICc difference from top model) ≥ 4, the top model was used to make inferences.
239   Otherwise, all models with Δ AICc < 4 were considered as plausible models (Burnham &

240 Anderson, 2002) and their parameter estimates were model-averaged ("MuMIn" package;
241 Bartoń, 2022).
242   I summed Apis individuals caught in a set of pant traps at each sample station, which was
243 considered as a unit for the analysis. The sum represented the abundance of Apis per sample
244 station. Environmental data for Apis included fine-scale data. However, flower covers at the fine
245 scale was collected in 2021 only and thus I performed two separate analyses: the one with the
246 data of two years (2020 and 2021) and the other with 2021 data only. I built a total of 8 GLMs
247 with negative binomial distribution for each analysis using a combination of scales (TableS2).
248 The survey year had a significant effect on Apis abundance and thus year was incorporated into
249 all models that used both 2020 and 2021 data. The abundance data of Apis had zero values;
250 while zero-inflated Poisson and negative binomial distributions can be used in this case, I chose
251 negative binomial distribution because of overdispersion in the GLMs with zero-inflated
252 Poisson. Model selection and averaging were carried out as for CWB.
253   Spatial dependence of bee and butterfly counts was also examined by conducting
254 Moran's I test on the residuals of the full models of CWB and Apis ("ape" package; Paradis &
255 Schliep, 2019): $P > 0.1$ in all cases, indicating that spatial dependence was negligible. I checked
256 other regression model assumptions such as homoscedasticity and overdispersion ("DHARMa"
257 package; Hartig, 2022) and did not find any cases of significant violation. The values of variance
258 inflation factor were < 2.5, suggesting little issue of multicollinearity.
259
Results
261 A total of 1189 CWB individuals was observed across all sites. Mean abundance was 13.21
262 butterflies per site with a SD of 10.16, ranging from 1 to 39. The abundance of CWB was
263 strongly associated with a combination of local and landscape variables: the full model showed
264 the lowest AICc, and Δ AICc of other models was > 9 (Table 1).
265   Crop diversity had a positive effect on the abundance of CWB, whereas percent cover of
266 farmland and weedy vegetation within the landscape affected negatively (Fig. 2 and Table S3).
267 The fragmentation of farmlands, i.e., patch density, was not associated with CWB abundance
268 given wide 95% confidence intervals across 0 and very low estimate (TableS3). Although
269 percent cover of cruciferous crops tended to have a positive impact on CWB abundance, the
270 effect was not as strong as other significant variables (TableS3).
271   For two winters, 265 Apis individuals were captured. While more individuals were
272 caught in 2021 than 2020, Apis abundance varied by sample stations: 1.15 ± 2.11 (mean
273 abundance $\pm$ SD), ranging from 0 to 14 in 2020, and 1.82 ± 2.03, ranging from 0 to 11 in 2021.
274 Of 8 GLM models, 4 models were selected as plausible models in 2021 data analysis and 2
275 models in combined years data analysis (Table 2).
276   Effects of weedy vegetation on Apis abundance were consistent across two winters given
277 that the 95% confidence intervals did not include 0 or slightly overlapped with 0. The effects
278 also depended on scale: the abundance of Apis was affected by weedy vegetation cover
279 negatively at the landscape scale but positively at the local scale (Fig. 3 and TableS3). In

280 combined years data analysis, increasing percent cover of farmland at the landscape scale was
281 positively associated with increasing Apis abundance. Apis abundance was also low in a
282 landscape with more fragmented farmlands. The percentage of flowering plants had a strong
283 positive effect on Apis abundance in 2021: its coefficient was ≥ 1.5 higher than other variables
284 associated with Apis, suggesting the amount of floral resource at the fine scale could be critical
285 to Apis (Fig. 3).
286

Discussion

288 The results of this study reveal variation in environmental factors at relevant spatial scales
289 associated with a host-specific pest, i.e., cabbage white butterflies, and a pollinator, i.e., Asian
290 honey bees. Cabbage white butterfly was more related to crop cover at the landscape scale
291 (negatively), whereas Asian honey bee was strongly related to flower cover at the fine scale and
292 weedy vegetation at the local scale (positively). Both species also responded negatively to weedy
293 vegetation cover at the landscape scale. These patterns can provide insight into how small urban
294 farmlands can be managed to improve ecosystem service, i.e., pollination by bees, while
295 minimizing ecosystem disservice, i.e., crop damage by cabbage white butterflies.
296   The resource concentration hypothesis predicts high abundance of host-specific
297 herbivorous insects at their host plant rich patches, i.e., dense, large, or pure stands of host plants
298 (Root, 1973). I expected a positive relationship between cabbage white butterfly abundance and
299 percent cover of Brassicaceae, and I found a tendency for this relationship. However, landscape-
300 scale crop cover was the environmental variable most strongly associated with the abundance of
301 cabbage white butterfly. Certainly the presence or absence of Brassicaceae in surrounding
302 farmlands could influence CWB distribution. If Brassicaceae cover is low in a landscape with
303 otherwise high crop cover, we might expect a negative association. Although I lack data on
304 Brassicaceae cover at the landscape scale, local Brassicaceae cover was not correlated with
305 landscape crop cover. Also, almost all farmlands surveyed contained Brassicaceae: of 90 sites,
306 44 included > 20% Brassicaceae at the local scale and only **7** had < 5% Brassicaceae. Thus, the
307 negative response of cabbage white butterfly is unlikely related to the amount of landscape
308 Brassicaceae cover. Matteson and Langellotto (2012) found that cabbage white butterfly spends
309 less time in urban gardens in a landscape with more green spaces, which could lead to low
310 detection of cabbage butterfly in these gardens. I often observed more cabbage white butterflies
311 and their mating, landing, and oviposition behaviors at farmlands isolated in a landscape
312 dominated by built-up structure. Considering that crop and weedy vegetation covers are part of
313 green spaces, the responses of cabbage white butterfly to both covers parallels the previous
314 finding. Increasing crop covers in a landscape likely diffuses cabbage white butterflies, lowering
315 abundance per farmland. It may also interrupt visual cues and consequently have a negative
316 impact on their ability to search host and floral resources because cabbage white butterflies
317 depend on vision to locate these resources (Hern et al., 1996).
318   The positive effect of crop diversity is somewhat unexpected. Polyculture stands often
319 show lower density or higher mortality of pest insects compared to monoculture stands (Altieri et

320   al., 1978; Russell, 1989; Letourneau et al., 2011; Iverson et al., 2014; among others). The
321   diversification of cruciferous crops through mixing or intercropping can effectively control pest
322   insects due to higher abundance and richness of parasitoids/predators, visual camouflage effect,
323   and masking effect of host plant odors, among other mechanisms (Finch & Collier 2000; Hooks
324   & Johnson, 2003). The positive effect I observed may be affected by the season of survey.
325   Although winter is relatively warm and dry in southern China, the abundance and richness of
326   predator and parasitoid insects are lower than other seasons such as spring (unpublished data).
327   Lowenstein and Minor (2018) reported different trends in abundance between herbivores
328   including cabbage white butterflies and predator/parasitoid insects at urban gardens and farms in
329   Chicago, USA: areas with high predators and parasitoids tended to have low herbivores.
330   Compared to large-scale rural agricultural lands, urban farmlands have also more diverse crops
331   despite their small size. Brassicaceae crops are cultivated at most of these farmlands in the study
332   area. This creates a spatially and temporally heterogeneous environment due to variations in
333   harvest time and growth stage between crops, even between Brassicaceae crops. These
334   conditions may benefit cabbage white butterflies by providing host and floral resources, shelters,
335   or mating chances for a longer period during the winter season. I also noticed seasonal variations
336   in crop composition, although crop diversity at each study site did not change significantly
337   between seasons. For example, in spring, the proportion of Brassicaceae crops declines but other
338   crops (especially vine crops, e.g., bean and gourd family crops) increase and flowering weeds are
339   abundant. Cabbage white butterflies may respond differently to crop diversity in spring, but the
340   extent of possible seasonal variations in the responses of cabbage white butterflies requires
341   further research.
342   Bee populations are strongly regulated by food resource availability (Roulston & Goodell,
343   2011). The establishment of semi-natural vegetation, especially planting flowering herbaceous
344   plants at field edges (i.e., flower strips) is often recommended to promote crop pollination and
345   yield by insect pollinators (Blaauw & Isaacs, 2014; Williams et al. 2015; Shutter et al., 2018; but
346   see Nicholson et al., 2018). One recent meta-analysis shows that crop pollination decreases with
347   increasing distance to floral resources, indicating the importance of floral resources to pollinators
348   (Albrecht et al., 2020). Crop diversity and semi-natural vegetation at landscape scales also
349   increase bumble bee density in wheat-dominant agricultural areas (Raderschall et al., 2021). The
350   result of the current study, i.e., the strong effect of percent cover of flowering plants on bee
351   abundance, partly supports the general trend. While I do not have data on the diversity and
352   abundance of flowering plants at local and landscape scales, the result highlights the benefit of
353   even small amount of flowering crop and weed to honey bees in urban environments, especially
354   during the winter in which natural floral resources may be scarce. This could also explain the
355   different effects of weedy vegetation cover at local and landscape scales. The subtropical climate
356   in southern China enables framers to cultivate vegetable crops year-round. While some weedy
357   vegetation showed withering in winter, the condition was less severe at urban farmlands than
358   some vacant lots (e.g., construction sites and abandoned lands), which is likely affected by

359 differences in water availability. There were also more weed plants blooming at urban farmlands
360 based on my observation.
361   One potential concern is the quality of floral resources in these urban farmlands because
362 most common weed plants flowering in the study area are non-native plants, particularly *Bidens*
363 species. There are few studies comparing nutritional values of nectars to bees between crops and
364 weed plants in this region. However, honey bees are generalists, visiting a wide range of
365 flowering plants, and require different diets to maintain colony health (Requier et al., 2015).
366 During times of food shortage, weed plants and non-native plants can be an important
367 component of the diets of *A. melifera* and *A. cerena,* respectively (Requier et al., 2015; Koyama
368 et al., 2018). *Apis melifera* also forages on flowers of *Bidens* species (Kajobe, 2006). Floral
369 resource use and preference by bees are significantly associated with dominant plants and
370 affected by season as floral resource availability changes throughout the year (Lowe et al., 2021).
371 Most flowering plants at sample sites in winter are weed plants and several crops that belong to
372 the genera *Brassica*, *Chrysanthemum*, and *Allium*. Weed plants can be one of major floral
373 resources to Asian honey bees in winter regardless of their nutritional values. However, with
374 seasonal variations in the diversity and abundance of floral resources, Asian honey bees may
375 prefer certain crops or weed plants in other seasons such as spring, when trees and shrubs as well
376 as more crops bloom. The degree of effects of flower strips on local bee abundance and diversity
377 can also depend on the interaction between characteristics of flower strips and floral resources
378 available in surrounding landscape (Scheper et al., 2015). Similarly, it is possible that the strong
379 response of Asian honey bees to fine-scale flower cover could be influenced by the total amount
380 of floral resources within farmland and landscape, which will vary by season.
381
382 **Conclusions**
383 Overall, the findings of current study provide a scientific basis for urban planners, policy makers,
384 and farmers to consider in the management of urban farmlands across scales to enhance the
385 sustainability of these farmlands. At the farmland (local) scale, cultivating too diverse crops
386 would not be recommended as it can increase the risk of crop damage by cabbage white
387 butterfly. However, the risk can be alleviated at farmlands embedded in a landscape with more
388 overall crop cover, which may partly benefit Asian honey bees as well. Retaining non-crop area
389 such as weedy vegetation patches or small portion of flowering plants including crops within a
390 farmland can be an effective practice to maintain Asian honey bee population and promote
391 pollination. It also benefits birds as local weedy vegetation has a positive effect on winter bird
392 diversity at urban farmlands (Lee et al., 2022). With growing urban expansion, farmlands in
393 China have been significantly converted into built-up structure, leading to a decline in net
394 primary productivity of cropland (He et al., 2017) despite farmland protection policy restrictions
395 on development. Farmlands in a city are not exceptional. In major cities in China, new
396 development largely spreads from the edge of city or occurs at old villages within a city (Tu et
397 al., 2021), lowering crop cover in surrounding landscapes and isolating farmlands (Liang et al.,
398 2015). Policy makers and urban planners must consider preserving farmlands, especially at the

399 city fringe or suburb for permanent agricultural activities. Priority should be given to farmlands
400 embedded in landscapes with relatively high crop covers. It is also critical that city government
401 monitors potential environmental problems associated with urban farmlands. For example, most
402 of these farmlands, especially ones next to main districts where development and population are
403 concentrated, use waste water. The average contents of heavy metals in agricultural soil in
404 Guangzhou and Foshan cities are higher than background values of Guangdong province (Xiao
405 et al., 2020). While the levels of metals are still lower than national standard values, the
406 ecological risk of heavy metal pollution in these cities is not negligible. Combined with further
407 research on seasonal variations in the patterns found in current study, the assessment of
408 environmental quality of urban farmlands can contribute to improve the sustainability of urban
409 agroecosystem.
410

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415

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Model selection results of cabbage white butterfly based on Akaike information criterion value adjusted for small sample size (AICc).

Note a big difference in AICc values (Δ AICc) between top model and other models. Null model is an intercept-only model, whereas full model includes both local- and landscape-scale variables. Abbreviation: DF, degree of freedom; LogLik, log-likelihood.

Model	DF	LogLik	AICc	Δ AICc	AICc weight
Full	8	-81.506	180.8	0.00	0.988
Landscape	5	-89.493	189.7	8.91	0.011
Local	5	-95.622	202.0	21.17	0.000
Null	2	-113.276	230.7	49.90	0.000

Summary of model comparisons of Asian honey bee.

Null model is an intercept-only model, whereas full model includes variables of all three scales. A combination of two spatial scales is denoted with "+": for example, Fine+Local model contains variables at fine and local scales. Note that fine-scale flower cover was considered in 2021 only. Models in "Both" year are constructed using two years of data, i.e., 2020 and 2021. Abbreviation: DF, degree of freedom; LogLik, log-likelihood; Δ AICc, AICc difference from top model.

Year	Model	DF	LogLik	AICc	Δ AICc	AICc weight
2021	Full	9	-163.529	347.1	0.00	0.444
	Fine+Local	6	-167.580	348.1	0.99	0.270
	Fine+Local	4	-170.216	348.9	1.78	0.182
	Fine+Landscape	7	-167.387	350.0	2.92	0.103
	Local+Landscape	7	-172.985	361.2	14.12	0.000
	Local	4	-176.948	362.3	15.24	0.000
	Landscape	5	-176.131	362.9	15.83	0.000
	Null	2	-179.720	363.6	16.49	0.000
Both	Local+Landscape	8	-280.330	577.5	0.00	0.582
	Full	9	-279.815	578.7	1.20	0.320
	Landscape	6	-284.734	582.0	4.44	0.063
	Fine+Landscape	7	-284.620	583.9	6.38	0.024
	Local	5	-288.486	587.3	9.79	0.004
	Null	3	-291.062	588.3	10.73	0.003
	Fine+Local	6	-288.123	588.8	11.21	0.002
	Fine	4	-291.054	590.3	12.81	0.001

Figure 1

Study sites established in the city of Guangzhou, Guangdong Province, China.

All sites are located in the Indo-Burma Biodiversity Hotspot (Indo-Burma) areas in Guangzhou. Number of sites used for each taxon and year are variable.
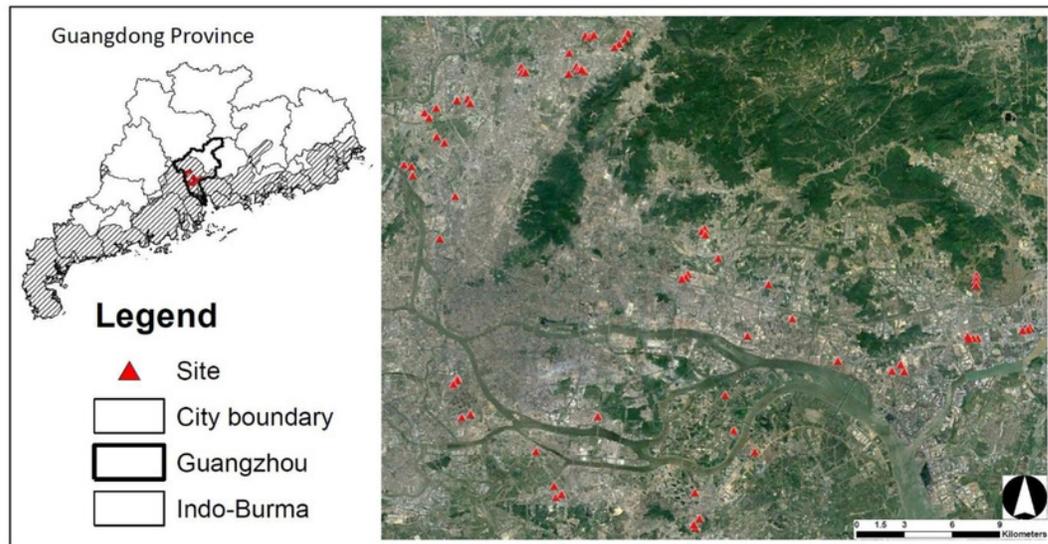
Figure 2

Significant relationship between the abundance of cabbage white butterfly and environmental variables: local crop diversity (cropdiv50, A), landscape crop cover (crop500, B), and landscape weedy vegetation cover (weed500, C).

Y-axis represents predicted abundance that is log(x+1) transformed. X-axis shows standardized values of each environmental variable.
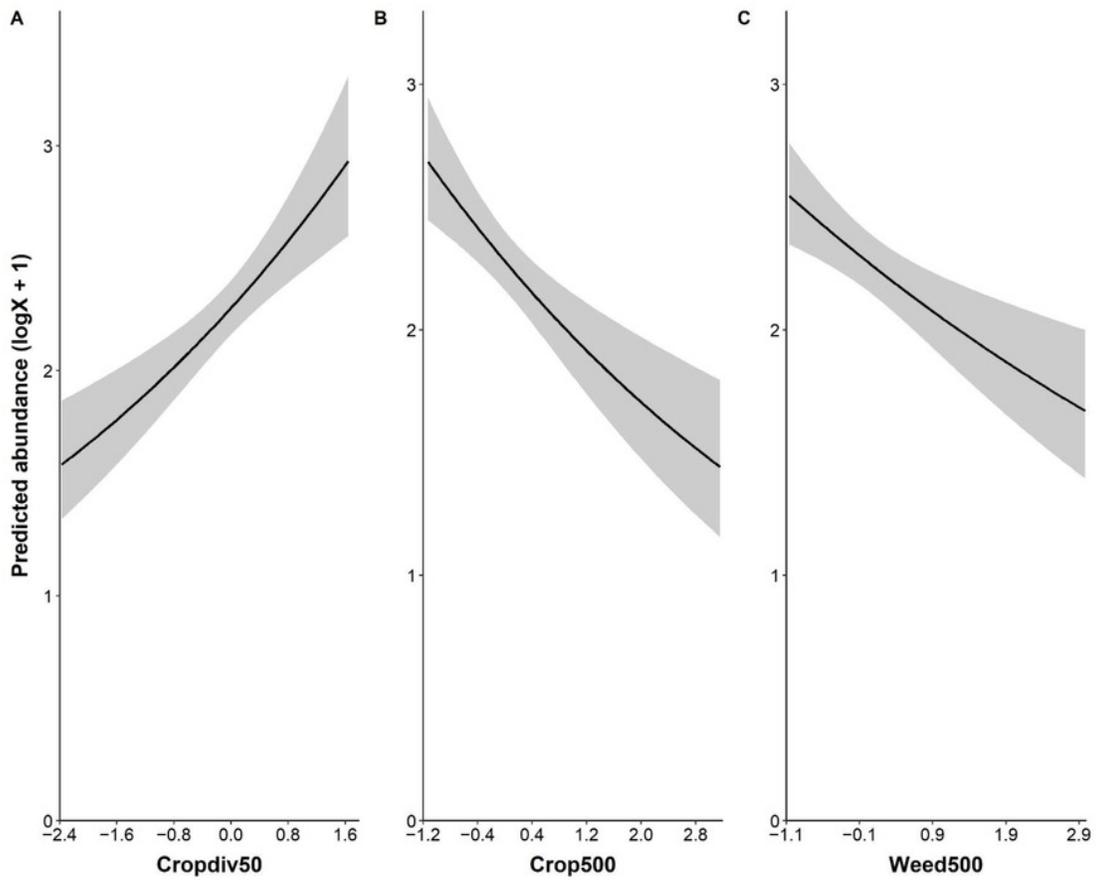
Figure 3

Model averaged parameter estimates and their 95% confidence intervals (error bars on the graph) from the analysis of Asian honey bee using 2021 data (A) and two years of data (B).

X-axis shows all environmental variables at each of three spatial scales: fine scale variables, flower (flower cover) and cropdiv5 (crop diversity); local scale variables, cropdiv50 (crop diversity) and weed50 (weedy vegetation cover); landscape scale variables, crop500 (crop cover), weed500 (weedy vegetation cover), and pd1000 (patch density).