

1 **Title**

2 Can biodiverse streetscapes mitigate the effects of noise and air pollution on human wellbeing?

3

4 **Author names and affiliations**

5 Jessica C. Fisher<sup>1</sup>, Eleanor Rankin<sup>1</sup>, Katherine N. Irvine<sup>2</sup>, Mark A. Goddard<sup>3\*</sup>, Zoe G. Davies<sup>1</sup>,  
6 Martin Dallimer<sup>3</sup>

7 <sup>1</sup>Durrell Institute of Conservation and Ecology (DICE), School of Anthropology and  
8 Conservation, University of Kent, Canterbury, CT2 7NR, UK

9 <sup>2</sup>Social, Economic and Geographical Sciences Department, James Hutton Institute,  
10 Craigiebuckler, Aberdeen, AB15 8QH, UK

11 <sup>3</sup>Sustainability Research Institute, School of Earth and Environment, University of Leeds,  
12 Leeds, LS2 9JT, UK

13 \*Current address: Department of Geography and Environmental Sciences, Northumbria  
14 University, Newcastle upon Tyne, NE1 8ST, UK

15

16 **Corresponding author**

17 Jessica C. Fisher: [J.C.Fisher@kent.ac.uk](mailto:J.C.Fisher@kent.ac.uk)

18

19 **Abstract**

20 Most of the global population are urban, with inhabitants exposed to raised levels of pollution.  
21 Pollutants negatively impact human wellbeing, and can alter the structure and diversity of  
22 ecosystems. Contrastingly, urban biodiversity can positively contribute to human wellbeing.  
23 We know little, however, about whether the negative impacts of pollution on wellbeing could  
24 be lessened for householders living on more biodiverse streets, as the complex interlinkages  
25 between pollution, biodiversity and wellbeing have rarely been examined. Here, we used  
26 structural equation modelling to simultaneously test whether biodiversity (actual and  
27 perceived) mediates the relationship between traffic-related pollution (noise, dB; nitrogen  
28 dioxide, NO<sub>2</sub>) or air pollution (PM<sub>2.5</sub>) and wellbeing (mental wellbeing, happiness). In summer  
29 2019, we conducted questionnaires and biodiversity surveys, and collected noise and air  
30 pollution data, from households (*n* = 282) across the streetscapes of Leeds, UK. Biodiversity  
31 (actual or perceived) showed no mediating effects. However, increased flowering plant  
32 richness was positively associated with mental wellbeing. Traffic-related pollution negatively  
33 affected pollinator and flowering plant richness, but not wellbeing. This could be because  
34 householders are not exposed to high levels of noise or NO<sub>2</sub> because they do not maintain front  
35 gardens on noisier streets. There was no measurable effect of air pollution on biodiversity or  
36 wellbeing. These findings shed light on the complex mechanisms through which biodiversity  
37 could improve human wellbeing. Enhancing the diversity of plant species in streetscapes would  
38 have a positive effect on wellbeing, further emphasising the important role that biodiverse  
39 urban streetscapes play in improving the liveability of cities.

40

41

42 **Keywords**

43 biodiversity; green infrastructure; gardens; mental health; particulate matter; structural  
44 equation modelling

45

## 46 **1. Introduction**

47 By 2050, approximately 68% of the global human population will reside in urban areas (United  
48 Nations, 2018). Urban living poses challenges for the physical health, mental health and  
49 wellbeing of town and city dwellers, particularly because of associated stressful lifestyles and  
50 exposure to elevated levels of pollution (Abbot, 2012; Peen et al., 2010; Roberts et al., 2019;  
51 WHO, 2006; Zhang et al., 2019). Indeed, noise pollution (e.g. road, rail, and air traffic) and  
52 air pollution (e.g. nitrogen dioxide, NO<sub>2</sub>; particulate matter, PM) are two of the three main risk  
53 factors for environmental disease burden in Europe (Hänninen et al., 2014). As such, city  
54 planning and urban design professionals are seeking to implement land-use planning initiatives  
55 that reduce the detrimental health and wellbeing impacts of pollution on the growing urban  
56 population (Giles-Corti et al., 2016).

57

58 Human wellbeing is known to improve with the presence of urban green infrastructure (e.g.  
59 parks, gardens, streetscape greenery), providing an opportunity for restoration, gaining distance  
60 from psychological demands, and reducing stress and fatigue (Kaplan and Kaplan, 1989; Ulrich  
61 et al., 1991). For instance, in a study of 51 European cities, city greenness was positively  
62 associated with improved self-reported quality of life (Giannico et al., 2021). While empirical  
63 research has demonstrated that individuals in greener neighbourhoods are happier and healthier  
64 (Ambrey and Fleming, 2014; Sarkar et al., 2018; Wang et al., 2020; White et al., 2013; Wood  
65 et al., 2017), it remains unclear what specific qualities or attributes of the ‘green’ (e.g.  
66 biodiversity) could underpin the positive effects (Dallimer et al., 2012; Wheeler et al., 2015).  
67 This concept is further complicated by a discrepancy between what attributes are objectively  
68 present, compared with what attributes people perceive to exist (Pett et al., 2016). For instance,  
69 Dallimer et al., (2012) found no relationship between actual butterfly or plant species richness  
70 and human wellbeing, but a positive one with perceived species richness for both taxa.  
71 Disentangling these differences has important implications for planning and policy  
72 recommendations aimed at maximising the beneficial effects of biodiversity on human  
73 wellbeing.

74

75 Streetscape biodiversity and front gardens are largely overlooked in nature-wellbeing research  
76 to date (Chalmin-Pui et al., 2019). However, they could theoretically offer many of the same  
77 benefits as back gardens, which provide important ecological resources for biodiversity  
78 (Baldock et al., 2019; Davies et al., 2009), and increased quality of life, emotional wellbeing

79 (Goddard et al., 2013), and restorativeness for people (Young et al., 2020). Indeed, Chalmin-  
80 Pui et al. (2021) showed that when ornamental plants were added to residential front gardens,  
81 householders experienced lower levels of stress, more positive emotions, relaxation and pride.  
82 Spano et al. (2021) showed that the presence of natural features in people’s homes, including  
83 views of greenery, can improve mental health and wellbeing. Streetscape greenery is also  
84 publicly viewable, experienced by neighbours and passers-by, potentially underpinning  
85 opportunities for more cohesive social interactions that subsequently improve human wellbeing  
86 for a wider range of people than just the householders themselves (Chalmin-Pui et al., 2021).

87  
88 In some streetscapes, traffic-related pollution could exert a considerable influence on  
89 biodiversity and wellbeing. The negative impacts of traffic-related pollutants (noise and NO<sub>2</sub>)  
90 on pollinators have been widely documented (e.g. disruption of communication, Morley et al.,  
91 2013; heightening of physiological stress, Davis et al., 2018). Noise pollution directly impacts  
92 human wellbeing, for example through sleep disturbance, which could lead to cardiovascular  
93 ill-health (Bai et al., 2020; Münzel et al., 2018). Roadside verge pollinators respond negatively  
94 to increased traffic, probably because of pollution and wind turbulence (Phillips et al., 2020).  
95 Air pollution can also reduce species-specific growth rates of urban vegetation (Honour et al.,  
96 2009). Concomitantly, higher levels of air pollutants (e.g. PM<sub>2.5</sub>) can directly decrease human  
97 wellbeing (e.g. emotional wellbeing, Zhang et al., 2019; depressive symptoms, Roberts et al.,  
98 2019). Although specific species of plants and trees can contribute to air pollution through the  
99 release of hydrocarbons and allergens, which can detract from wellbeing (see Hartig et al.,  
100 2014). Some elements of biodiversity have the potential to alleviate or offset the negative  
101 consequences of noise and air pollution on wellbeing. Streetscape trees and shrubs can scatter  
102 and refract noise levels at traffic-level frequencies (Fang and Ling, 2005; Han et al., 2018;  
103 Klingberg et al., 2017). Similarly, vegetation can passively screen and filter air, while the  
104 presence of leaves on some species actively absorb pollutants (Klingberg et al., 2017; Nowak  
105 et al., 2006). The extent to which pollutants can be obscured are dictated by vegetation  
106 characteristics such as height, width, and density (Abhijith et al., 2017).

107  
108 Urban biodiversity could play a pivotal role in the relationship between pollution and human  
109 wellbeing in neighbourhood streetscapes, particularly as pollutants will be exacerbated by  
110 roadside traffic. We therefore investigate how traffic-related pollution (noise pollution, dB, and  
111 nitrogen dioxide, NO<sub>2</sub>) as well as streetscape air pollution (particulate matter, PM<sub>2.5</sub>) impacts  
112 human wellbeing via the mediating role of biodiversity. Given the likely complexity of these

113 associations, we used parallel mediation models to simultaneously examine how both objective  
114 and perceived measures of biodiversity (pollinators, flowering plants and trees) influence the  
115 effect of pollution on residents' wellbeing. We hypothesise that (H1) higher levels of  
116 biodiversity will have a mediating effect, reducing the impact of pollution on human wellbeing,  
117 and (H2) higher perceived biodiversity will have a similar mediating effect. This research  
118 makes a novel empirical contribution to the small but growing evidence-base on streetscape  
119 biodiversity, pollution and human wellbeing. These mediation effects remain largely  
120 understudied to date, despite urbanisation accelerating worldwide, but could offer crucial  
121 evidence to inform the sustainable design of biodiverse and liveable cities.

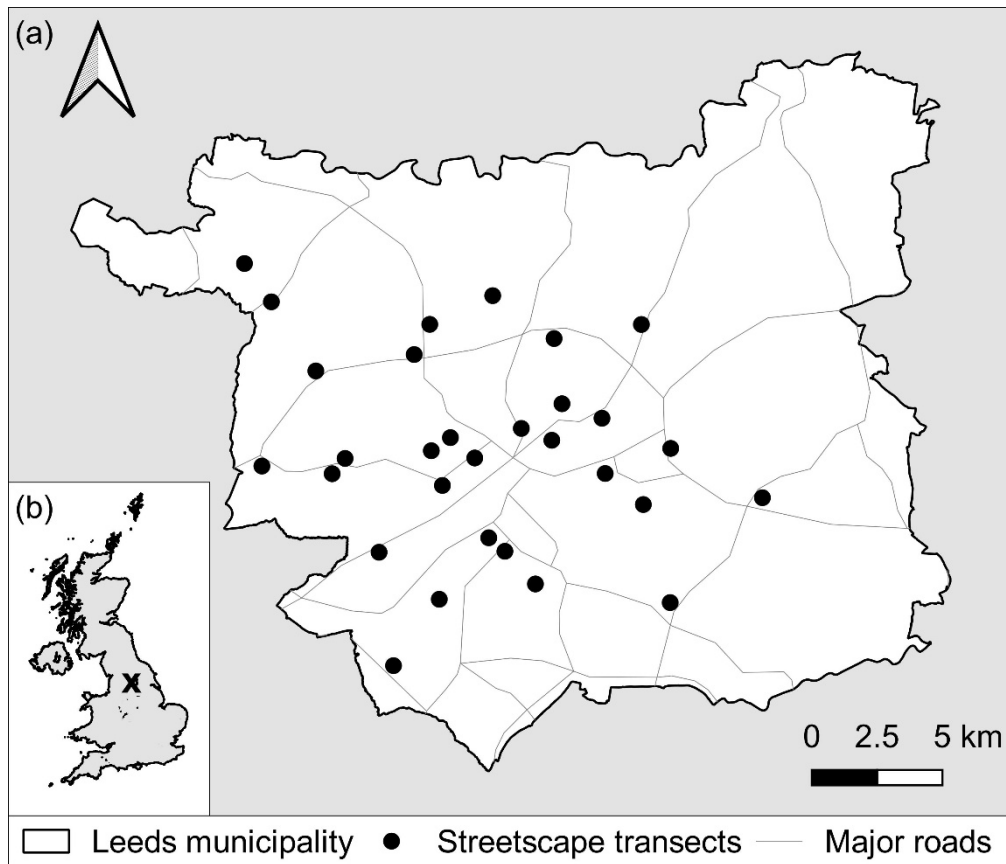
122

## 123 **2. Methods**

### 124 *2.1 Study system*

125 The research was conducted across the streets of Leeds, UK (53° 47' 59" N, 1° 32' 57" W; Fig.  
126 1), the fourth largest city in England (~552 km<sup>2</sup>). Leeds has a human population of ~790,000,  
127 which is very ethnically and culturally diverse (Office for National Statistics, 2018). Across a  
128 33 year period, Leeds has witnessed a 13% increase in impervious surfaces (Perry and Nawaz,  
129 2008). At the time of writing, urban planning policies and legislation largely overlook front  
130 gardens, as many are privately managed spaces (although planning permission is required for  
131 >5 m<sup>2</sup> of impermeable surface area) (Ellis and Lundy, 2016). Regardless, there is poor  
132 enforcement of this policy, and paving continues unabated. Motivations include reducing  
133 garden maintenance, as well as poor public transport links, which leads to increased car  
134 ownership and the subsequent need for parking spaces (London Assembly Environment  
135 Committee, 2005). As such, our study's focus on pollution, biodiversity, and human wellbeing  
136 at the level of the neighbourhood streetscape, could have implications for local sustainable  
137 urban planning initiatives.

138



139

140 **Fig. 1** Study area showing (a) the municipality of Leeds, its major roads (grey lines) and the  
 141 location of each of the 30 streetscape transects (black circles). (b) The location of Leeds (cross)  
 142 in the UK.

143

144 We used a hierarchical sampling design to capture variation in pollution, based on road size  
 145 and traffic capacity, as no systematic citywide data on pollution existed prior to initiating the  
 146 study. The vast majority of pollution in UK cities is derived from road transport (Department  
 147 for Business, Energy and Industrial Strategy, 2021). We therefore used road size as one way  
 148 of sampling across likely variation in noise and air pollution levels across streetscapes. Major  
 149 roads designated to provide large-scale transport links within or between major urban centres  
 150 (i.e. main “A roads”, including dual carriageways) were classified as ‘high’ pollution.  
 151 ‘Medium’ pollution roads were those intended to connect different areas within a region and  
 152 to feed traffic from major to smaller roads on the network (i.e. secondary “B roads” or roads  
 153 more than 4m wide) (Department of Transport, 2012). Finally, ‘low’ pollution roads were  
 154 considered as all other roads less than 4 m wide.

155

156 We selected 10 streetscapes from each of the three pollution categories, giving 30 in total.  
 157 Along each of these roads, a 200 m long streetscape transect was positioned so that all the

158 transects were located within a different ward (UK administrative areas) of Leeds, maximising  
159 spatial variation across the city. Each streetscape transect comprised all green infrastructure in  
160 residential front gardens and within the street itself (e.g. street trees, road verges, central  
161 reservations, all other vegetation). To further ensure sample independence, and diffusion of  
162 pollutants between sites of varying pollution categories, straight-line distances between the  
163 streetscape transects were at least 0.6 km, with the vast majority being >1 km apart. This  
164 distance is also greater than the forage range of most pollinator species in urban landscapes  
165 (Garbuzov et al., 2015; Langellotto et al., 2018). Each streetscape transect was selected to make  
166 sure the sampled households captured citywide variation in housing type (i.e. detached; semi-  
167 detached; terraced), which is indicative of the size of gardens (Loram et al., 2007) and  
168 sociodemographic/economic characteristics.

169

## 170 *2.2 Pollution*

171 Traffic-related pollution was captured using ambient nitrogen dioxide (NO<sub>2</sub>) and noise  
172 pollution. NO<sub>2</sub> concentration was measured using diffusion tubes. Three tubes were situated  
173 equidistant along each transect, positioned 2.5 m high on lampposts. Tubes were left in place  
174 for four weeks in May/June and three weeks in July/August 2019. An average concentration  
175 was calculated across all tubes and sampling periods. While this methodology is unable to  
176 capture incidences where NO<sub>2</sub> might be temporarily elevated (e.g. during rush hour), we were  
177 interested in the longer-term (rather than momentary) effects of traffic-related pollution on  
178 people's wellbeing in the streetscape where they lived, and thus compare between sites.

179

180 Air pollution on each streetscape transect was measured using PM<sub>2.5</sub> concentrations (µg/m<sup>3</sup>).  
181 This measurement of PM does not capture the size fraction typically emitted by vehicle  
182 tailpipes. The methods we employed are known to provide measurements suitable for relative  
183 spatial comparisons across a study system (e.g. Bush et al., 2001). However, the techniques are  
184 not recommended for carrying out internationally recognised monitoring of pollution levels  
185 (e.g. Ngo et al., 2019). As such, our findings should not be directly compared to publicly  
186 gathered data on air pollution concentrations across Leeds.

187

188 Particulate matter concentration (PM<sub>2.5</sub>) was recorded using the IQAir AirVisual Pro monitor  
189 (measuring range: 0.3-2.5 µm; accuracy to the nearest 1 µg/m<sup>3</sup>), and noise pollution (decibels;

190 dB A) were record using a Reed ST-8850 sound level meter (measuring range: 30-130 dB; 0.1  
191 dB resolution). Both particulate matter and noise level data were obtained by walking at a slow  
192 pace along both sides of the 200 m transect on two occasions, at different times of day (morning  
193 and afternoon), between May and August 2019. The measurement period was ~15 minutes in  
194 duration, and start times ranged from 09:39 to 17:21. The sound meter recorded one  
195 measurement per second, while the particular matter monitor recorded once every 10 seconds  
196 (equating to 600 values for noise, and 60 values for particulate matter, per transect per visit).  
197 There were no missing values. Median values were used to represent each streetscape transect.  
198 To minimise bias caused by variation in meteorological conditions that can affect air quality,  
199 streetscapes across all pollution level categories (high, medium and low) were sampled in  
200 groups on the same day or adjacent days with comparable weather conditions (Mues et al.,  
201 2012).

202

### 203 *2.3 Greenness*

204 To account for the known effect of neighbourhood greenness on wellbeing, we used the  
205 normalised difference vegetation index (NDVI) (Sarkar et al., 2018; Wang et al., 2020) and  
206 used it as a covariate in our analyses. NDVI was obtained from MODIS with no manipulation  
207 (MOD13Q1 Collection 6 satellite data 16-day composite at a 250 m spatial resolution; ORNL  
208 DAAC, 2018), ranging in from 0.15-0.79 in our dataset (there were no blue spaces in the  
209 vicinity of the streetscapes). For each streetscape transect we derived NDVI within a 0.25 km<sup>2</sup>  
210 polygon centred on the midpoint of the transect.

211

### 212 *2.4 Actual measures of biodiversity*

213 Pollinators, such as butterflies and bees, are a prominent component of urban biodiversity  
214 during the day. Streetscape pollinator richness and abundance were estimated using a pollinator  
215 transect sampling approach modified from Baldock et al. (2015). Each pollinator transect was  
216 2 m in height and 4 m in width, walked at a steady pace, following the boundary between the  
217 pavement and residential gardens (including road verges where present) along the side of the  
218 streetscape with the greater extent of green infrastructure for 200 m. All pollinators observed  
219 were recorded as one of 20 morphological functional groups (Supplementary Table A.1),  
220 giving a measure of morpho-functional group richness of pollinators. Pollinator transects were  
221 conducted in May and July 2019 when weather conditions were suitable. Flowering plant  
222 richness was estimated for each streetscape transect by identifying all plant species in flower



223 (excluding grasses, sedges and wind-pollinated forbs) across two survey visits. Tree richness  
224 was also assessed, based on all individuals  $\geq 2$  m in height.

225

## 226 *2.5 Questionnaire*

227 Human perceptions of biodiversity, wellbeing outcomes and covariates (with the exception of  
228 NDVI) were derived from a questionnaire administered *in situ* between June and August 2019.  
229 All 1033 households within the 30 streetscape transects were eligible to participate, with one  
230 questionnaire to be completed per household. Each streetscape transect was visited on at least  
231 three occasions, on both weekdays and weekends, and at different times (during the working  
232 day versus early evening) to maximise response rates. Only permanent household residents  
233 over the age of 18 were permitted to complete the questionnaire and only after informed  
234 consent was obtained. Ethics approval was granted by the University of Leeds Social Sciences,  
235 Environment and LUBS (AREA) Faculty Research Ethics Committee, reference AREA 18-  
236 165. The questionnaire was tested by focus groups of Leeds residents, comprising participants  
237 who were independent to the streetscape transect households. Focus groups allowed us ensure  
238 that questionnaire wording aligned with phrases that are used and understood by participants  
239 (e.g. ‘greenery’, ‘neighbourhood’, ‘noisy’, ‘street environment’).

240

## 241 *2.6 Perceived measures of biodiversity*

242 We asked householders about their perceptions of biodiversity in their streetscape, which was  
243 termed ‘street environment’ in the questionnaire. It was emphasised that the phrase ‘street  
244 environment’ covered all green infrastructure associated with front gardens and the street itself.  
245 Using five-point scales, participants were asked to estimate the total number of pollinating  
246 insect (*Fewer than 5, 5 to 9, 10 to 13, 14 to 19, 20 or more*), flowering plant (*Fewer than 10,*  
247 *10 to 30, 31 to 50, 51 to 99, 100 or more*) and tree species (*Fewer than 5, 5 to 10, 11 to 15, 16*  
248 *to 20, 21 or more*) across their streetscapes. Categories for the scales were based on numbers  
249 of species likely to be present, based on our previous research (Goddard et al., 2013).

250

## 251 *2.7 Wellbeing outcomes*

252 The two self-reported wellbeing outcomes we measured (Appendix A: Supplementary text)  
253 were mental wellbeing and happiness, using existing scales validated in nature-health research  
254 (van Herzele and de Vries, 2012; Houlden et al., 2017). Self-reported (rather than objective)  
255 measures of health and wellbeing are commonly used, and known to be both efficient and  
256 robust (Andrews et al., 1976; Lucas, 2018). We used the Short Warwick-Edinburgh Mental

257 Wellbeing Scale (SWEMWBS) (Stewart-Brown et al., 2009) to assess the primary components  
258 of mental health, including hedonic (feeling of positive emotions, satisfaction) and eudaimonic  
259 (functioning, relationships, sense of purpose) domains (Dolan and Metcalfe, 2012).  
260 SWEMWBS is a 7-item scale, where participants are asked to “*Tick the box that best describes*  
261 *your experience of each over the last two weeks*” and respond with one of five options (*None*  
262 *of the time, Rarely, Some of the time, Often, All of the time*). Scores are summed to produce an  
263 initial raw score, then transformed using a conversion table (Stewart-Brown et al., 2009) to  
264 give a final metric ranging from 7 (lowest possible wellbeing) to 35 (highest possible  
265 wellbeing). SWEMWBS has shown adequate validity and reliability in people with mental  
266 health diagnoses (e.g. depression, anxiety, schizophrenia) (Vaingankar et al., 2017). Happiness  
267 was evaluated using a single item (Fordyce, 1988), which asks participants asked “*In your life*  
268 *in general, how happy would you say you are?*”, and asked to respond on a continuous scale  
269 from 1 (*extremely unhappy*), to 10 (*extremely happy*). This single-item scale has shown good  
270 concurrent and convergent validity with positive wellbeing measures (optimism, self-esteem,  
271 positive affect; Abdel-Khalek, 2006).

272

### 273 2.8 Covariates

274 We sought to account for participant’s feelings of social cohesion, given it can play a  
275 considerable role in people’s mental health and wellbeing, and therefore influence people’s  
276 experiences of the world around them (Hartig et al., 2019; Markevych et al., 2017). Social  
277 cohesion was measured using five items, three positive (positive affect) and two negative  
278 (negative affect), drawn from the work by Sampson et al. (1997).

279

280 Human wellbeing can be affected by noise sensitivity, influencing how people respond to noise  
281 pollution and the restorative effects offered by biodiversity (Ojala et al., 2019), and is likely to  
282 vary between individuals. We evaluated self-reported noise sensitivity using four items taken  
283 from Weinstein’s (1978) noise sensitivity scale, each of which is context independent and  
284 correlates with the full scale in Weinstein (1978) (Heinonen-Guzejev et al., 2004). As such, we  
285 used four statements to represent noise sensitivity used in previous work (Okokon et al., 2015),  
286 and asked participants to respond on a five-point scale (*Strongly disagree, Disagree, Neutral,*  
287 *Agree, Strongly agree*). One item (“*I get annoyed when my neighbours are noisy*”), was adapted  
288 to relate to the participants’ streetscape (“*I get irritated when there is noise in my street*”). The  
289 scores are summed to create an overall measure of noise sensitivity ranging from 4 (low  
290 sensitivity) to 20 (high sensitivity) (Appendix A: Supplementary text).

291

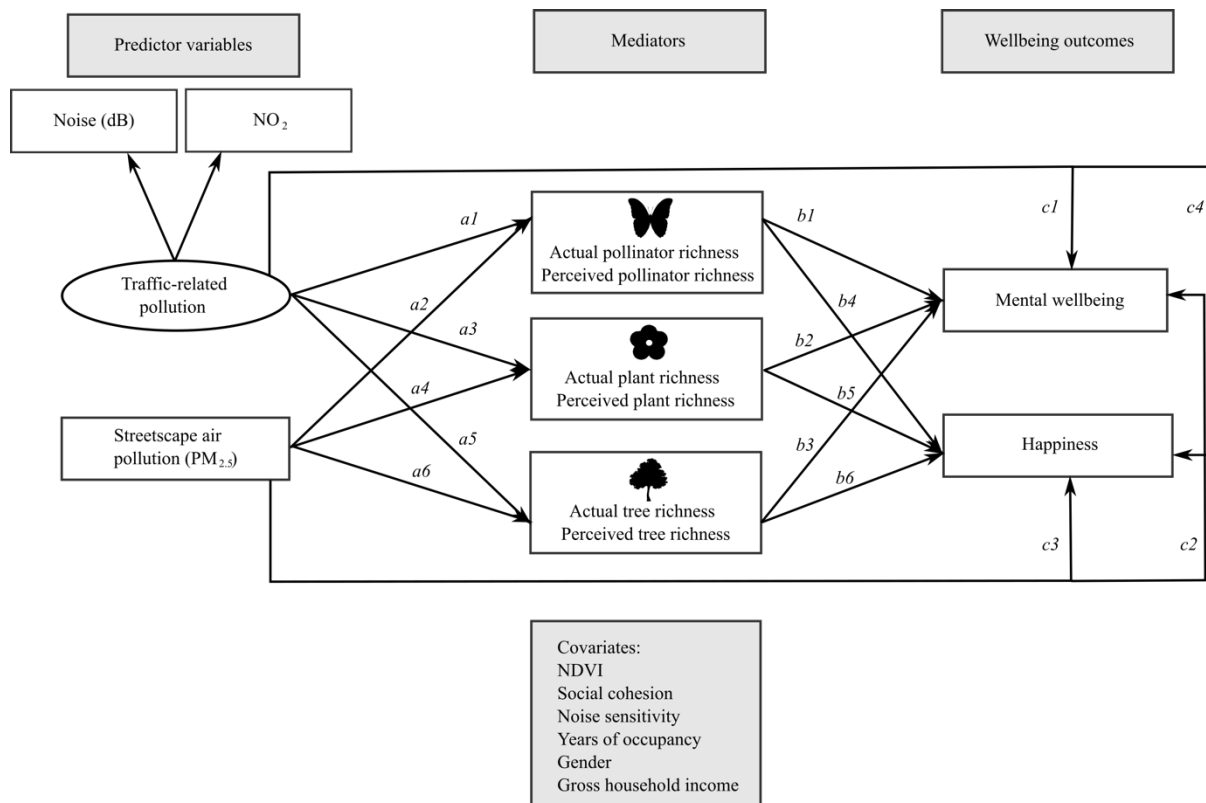
292 Additionally, we collected data on participant age, employment status, years of household  
293 occupancy, and gender to control for their potential effects on the mediators and wellbeing  
294 outcomes (Kendal et al., 2012; Dzhambov et al., 2018; Hartig et al., 2014). As many  
295 participants did not wish to share information about gross household income, we obtained  
296 median gross household income from the most recent UK census in 2011 (Office for National  
297 Statistics, 2018), using the most spatially resolved data available to cover the location of each  
298 streetscape transect ('Lower Layer Super Output Area' or LSOA).

299

### 300 *2.9 Analytical framework*

301 A series of parallel mediation models (Fig. 2) were constructed to test the influence of species/  
302 morpho-functional group richness (actual versus perceived) mediating the relationship between  
303 traffic-related (noise and NO<sub>2</sub>) and streetscape air pollution (PM<sub>2.5</sub>), with human wellbeing  
304 (mental wellbeing, happiness), while adjusting for covariates (NDVI, social cohesion, noise  
305 sensitivity, years of household occupancy, gender, gross household income). This approach  
306 simultaneously calculates regressions between the predictors and mediators (paths *a1 - a6*), the  
307 predictors and wellbeing outcomes while holding the mediators constant (direct effect, paths  
308 *c1 - c4*), and the mediators and wellbeing outcomes (*b1 - b6*). The indirect effect (*a\*b*)  
309 measures how the predictors influence the wellbeing outcomes as a result of the influence of  
310 the predictors on the mediators, which, in turn, also affect the wellbeing outcomes.

311



312

313 **Fig. 2** The parallel mediation model framework to assess the effects of predictors (traffic-  
 314 related and streetscape air pollution) on wellbeing outcomes (mental wellbeing and happiness),  
 315 via potential mediators (actual versus perceived richness for each taxa). Actual pollinator  
 316 richness was assessed by morpho-functional group richness. Richness of pollinators = butterfly  
 317 symbol, flowering plants = flower symbol, trees = tree symbol. *a* path = tested direct  
 318 associations between predictor and mediators. *b* paths = tested direct associations between  
 319 mediators and wellbeing outcomes. *c* paths = tested direct associations between predictors and  
 320 wellbeing outcomes. The indirect effect (*ab*) is the product of the *a* and *b* paths.

321

### 322 2.10 Data analysis

323 Analyses were performed using R Statistical Software version 3.6.0 (R Core Team, 2020). We  
 324 tested for associations between variables using Spearman's rank correlation tests for non-  
 325 normal data, and Kruskal Wallis tests for continuous and categorical data. We used a G-test to  
 326 examine whether our sample was comparable to census data for Leeds in terms of gender, age,  
 327 and ethnicity. Due to a large number of participants not disclosing their age ( $n = 31$ ), and given  
 328 that age was closely associated with years of household occupancy ( $r = 0.69, p < 0.001$ ), we  
 329 used the latter in our structural equation models to maximise our sample size and to account  
 330 for its likely influence on how people perceive the streetscape (Dzhambov et al., 2018). We  
 331 found a strong association between employment and gross household income ( $X^2 = 6.46, df =$

332 2,  $p < 0.05$ ), so used the latter due to its continuous nature and possible influence on the  
333 maintenance of front gardens within the streetscape (Kendal et al., 2012). Gender was treated  
334 as binary.

335

336 To create the parallel mediation models we used the ‘lavaan survey’ package in R designed for  
337 structural equation modelling (Oberski, 2014), which allows for the analysis of clustered  
338 sampling (i.e. households from the same streetscape) using cluster-robust standard errors. To  
339 improve statistical reliability, we removed three streetscapes where less than five households  
340 had completed the questionnaire. The final set of variables included in the models showed no  
341 evidence of multicollinearity based on Variance Inflation Factors, all of which were  $< 2.5$  (Zuur  
342 and Ieno, 2016). Data were then scaled and centred to stabilise variances and improve model  
343 fit.

344

345 A latent variable termed traffic intensity, indicated by noise pollution (dB) and  $\text{NO}_2$ , was used  
346 as a model predictor. Two separate models were run to test mediation of actual and perceived  
347 species/ morpho-functional group richness, respectively. Error variances between the three  
348 mediators in each model (pollinators, flowering plants, and trees) and two outcome variables  
349 (mental wellbeing and happiness) were free to covary due to the plausible associations between  
350 them (e.g. more pollinators are likely to be found where flowering plant richness is greater).  
351 Models were estimated using a maximum likelihood estimator and a Satorra-Bentler scaled test  
352 statistic that is robust to non-normality. We gauged the quality of our models using a  
353 combination of model fit indices (Hu and Bentler, 1999), employing a chi-square adjusted for  
354 clustered data (‘*pval.pFsum*’ function, Oberski, 2014), standardised root mean square residual  
355 (SRMR), root mean square error of approximation and its 95% confidence intervals (RMSEA),  
356 and the comparative fit index (CFI) to identify good model fit ( $X^2 p > 0.05$ ,  $\text{CFI} > 0.95$ ,  $\text{RMSEA}$   
357  $< 0.06$ ,  $\text{RMSEA}$  95% confidence intervals  $< 0.06$ ,  $\text{SRMR} < 0.08$ ) (Oberski, 2014; Barrett,  
358 1997; MacCallum et al., 1996; Hu and Bentler, 1999). We further tested for indirect effects by  
359 computing Monte Carlo confidence intervals (9999 replicates) (‘semTools’ package, Jorgensen  
360 et al., 2021) for our models where paths ‘*a*’ and ‘*b*’ were significant (MacKinnon et al., 2004;  
361 Preacher and Selig, 2012), thus accounting for nonnormality in the sampling distribution of the  
362 indirect effect (Fairchild and McDaniel, 2017).

363

### 364 **3. Results**

#### 365 *3.1 Descriptive statistics*

366 A total of 282 households (27.3% response rate) completed the questionnaire across the 30  
 367 residential streetscapes in Leeds, UK. Participant's ages ranged from 18 to 95, and 57% were  
 368 female (Table 1). Most participants (92%) had been living at their property for more than one  
 369 year. The sample was representative of the population of Leeds as a whole, based on gender,  
 370 age, and ethnicity (Table A.2).

371  
 372 **Table 1** Summary of predictors, wellbeing outcomes, mediators and covariates used in the  
 373 parallel mediation models (Fig. 2). Traffic-related pollution, air pollution, actual biodiversity  
 374 (pollinators, flowering plants, trees) and greenness (NDVI) were measured within 30  
 375 streetscapes in Leeds, UK. Wellbeing outcomes, perceived biodiversity (pollinators, flowering  
 376 plants, trees) and remaining covariates were derived from questionnaires from 282 households  
 377 across the 30 streetscapes. Actual pollinator richness was assessed by morpho-functional group  
 378 richness. Flowering plant richness values are sum from two visits per streetscape. Tree richness  
 379 values are the total number counted per streetscape. Perceived species richness values are the  
 380 average across all questionnaire responses within a streetscape. Gender was a categorical  
 381 covariate (see Table A.2). IQR = interquartile range.

	<b>Min</b>	<b>Median</b>	<b>Max</b>	<b>IQR</b>
<b>Predictors</b>				
Noise pollution (dB(A))	67.50	82.40	93.60	10.6
NO <sub>2</sub> (ppm)	6.11	36.43	80.99	29.53
Air pollution (PM <sub>2.5</sub> µg/m <sup>3</sup> )	2.26	4.69	8.83	3.17
<b>Wellbeing outcomes</b>				
Mental wellbeing	13.33	22.33	35.00	5.05
Happiness	1.00	8.00	10.00	2.00
<b>Mediators</b>				
Actual pollinator richness	0.00	7.00	14.00	3.75
Actual flowering plant richness	31.00	58.50	101.00	18.75
Actual tree richness	2.00	16.50	33.00	10.5
Perceived pollinator richness	1.25	2.81	4.00	0.82
Perceived flowering plant richness	1.58	2.50	3.50	0.79
Perceived tree richness	1.00	2.26	2.88	0.55
<b>Covariates</b>				

Greenness (NDVI)	0.15	0.50	0.79	0.29
Social cohesion	1.00	2.88	4.62	0.75
Noise sensitivity	4.00	13.00	20.00	3.00
Years of household occupancy	0.08	16.00	69.19	22.36
Gross household income (£)	11,553.00	31,334.00	51,598	15,707
			.00	.00

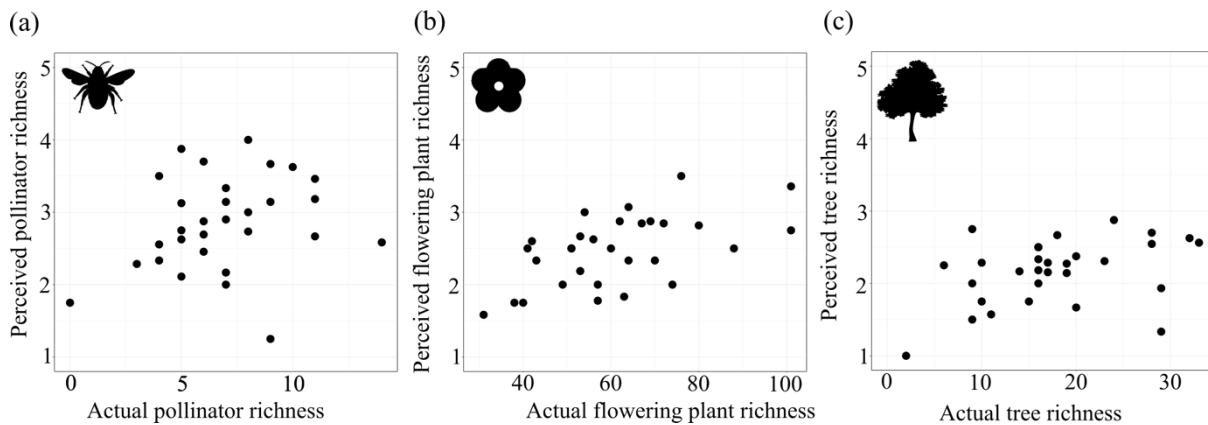
382

383 Measures of NO<sub>2</sub> and noise increased across the pollution level categories (low, medium, high)  
 384 used to stratify the study system (Supplementary Fig. A.1), indicating that the sampling effort  
 385 was broadly representative for these streetscape pollutants. The sampled PM<sub>2.5</sub> concentrations,  
 386 however, were highly variable across the pollution level categories.

387

388 Species richness for flowering plants and trees, and morpho-functional group richness of  
 389 pollinators, varied widely across the streetscape transects (Table 1). When compared with  
 390 perceived richness of pollinators, flowering plants and trees, there was no association for  
 391 pollinators ( $r = 0.29, p = 0.126$ ), a significant association for flowering plant richness ( $r = 0.54,$   
 392  $p = 0.002$ ) and tree richness ( $r = 0.37, p = 0.044$ ) (Fig. 3). Across the 282 participants, mental  
 393 wellbeing scores had a central tendency, whereas happiness scores were right-skewed.

394



395

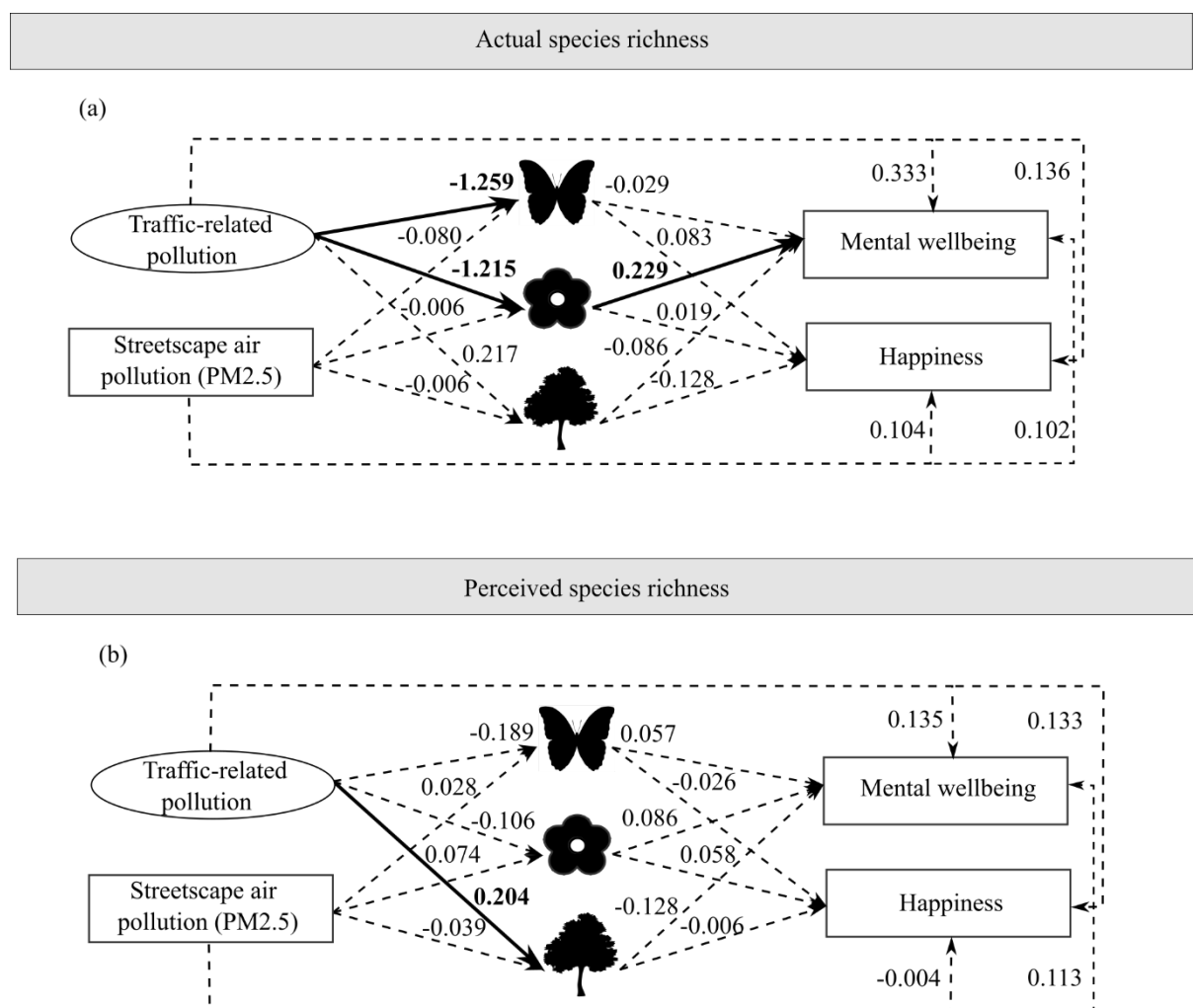
396

397 **Fig. 3** Association between actual and perceived richness of (a) pollinators = butterfly symbol  
 398 ( $r = 0.29, p = 0.126$ ) (b) flowering plants = flower symbol ( $r = 0.54, p = 0.002$ ), and (c) trees  
 399 = tree symbol ( $r = 0.37, p = 0.044$ ). Actual pollinator richness was assessed by morpho-  
 400 functional group richness. Values for perceived richness represent the mean score for each  
 401 streetscape.

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### 3.2 Parallel mediation models

The variances explained in each wellbeing outcome were between 8% and 12% in both models. Traffic-related pollution had an inverse association with actual pollinator and flowering plant richness, but no effect on tree richness (Fig. 4a). Streetscape air pollution had no effect on richness for any of the three taxa. Actual flowering plant richness had a positive effect on mental wellbeing but not happiness (Fig. 4a). We found no direct effects, nor any mediating effects (*ab* path; Fig. 2) of actual biodiversity, between traffic-related and streetscape air pollution, on mental wellbeing or happiness (Fig. 4a, Supplementary Table A.3).



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**Fig. 4** Parallel mediation models showing the effects of traffic-related pollution (latent variable representing noise and NO<sub>2</sub>) and streetscape air pollution (PM<sub>2.5</sub>) and on mental wellbeing and happiness, mediated by (a) actual richness, and (b) perceived richness (of pollinators, flowering plants, and trees, respectively). Actual pollinator richness was assessed by morpho-functional



418 group richness. Richness of pollinators = butterfly symbol, plants = flower symbol, trees = tree  
419 symbol. Latent variable represented by an oval, measured variables represented by rectangles.  
420 All models are adjusted for covariates (NDVI, noise sensitivity, years of household occupancy,  
421 gender, social cohesion, and gross household income). Plots display the unstandardised beta  
422 estimates after rows containing missing values were removed ( $n = 239$ ), with statistically  
423 significant estimates and respective paths highlighted in bold ( $p < 0.05$ ). Mediating effects (*ab*  
424 path) tested separately, and latent variable estimates and error covariances not shown for  
425 readability (Supplementary Tables A.3, A.4).

426

427 Models revealed that traffic-related pollution had a positive association with perceived tree  
428 richness, but no effect on perceptions of other taxa (Fig. 4b). Streetscape air pollution had no  
429 direct nor indirect effects on people's perceptions of pollinator, flowering plant or tree richness  
430 in their streetscapes, or their mental wellbeing or happiness (Fig. 4b; Supplementary Table  
431 A.4). Given that traffic-related pollution (noise and NO<sub>2</sub>) was inversely associated with  
432 flowering plant richness, which in turn was positively associated with mental wellbeing  
433 (significance of path *a* and path *b*; Fig. 4a), we computed Monte Carlo confidence intervals for  
434 the indirect effect, but found no evidence of mediation as the confidence interval crossed zero  
435 (-0.134, 0.057).

436

437 Amongst covariates (Supplementary Table A.3, A.4), years of occupancy had a negative effect  
438 on actual tree richness, compared with gross household income, which had a strong positive  
439 effect. Social cohesion was a strong predictor of perceived richness for all taxa. Both models  
440 suggested that participants with higher noise sensitivity reported lower happiness.

441

## 442 **4. Discussion**

443 Pollution can be detrimental to human wellbeing, contributing to the prevalence of  
444 psychological and physical health disorders amongst urban dwellers (Abbot, 2012; WHO,  
445 2006, 2018). For the first time to our knowledge, we test whether biodiversity (actual and  
446 perceived) could intervene in this relationship, using structural equation modelling to consider  
447 the likely complex associations. Our findings show traffic-related pollution (noise, NO<sub>2</sub>) can  
448 have detrimental impacts on pollinator and flowering plant richness, and increased flowering  
449 plant richness can benefit mental wellbeing. However, our approach revealed no statistical  
450 support for (H1) actual biodiversity will mediate the relationship between pollution and

451 wellbeing, or (H2) perceived biodiversity will also have a mediating effect, where neither  
452 measure of biodiversity showed any mediation, therefore unveiling further complexity in how  
453 pollution, biodiversity, and human wellbeing are associated.

454

455 An increase in actual flowering plant richness had a positive effect on the mental wellbeing of  
456 participants. This aligns with findings that increased flowering plant richness is associated with  
457 enhanced human wellbeing (reflection, distinct identity) (Fuller et al., 2007), that gardens rich  
458 in plant species are perceived as more restorative (Young et al., 2020), and that front gardens  
459 containing more diverse ornamental plants are related to reduced stress, improved motivation,  
460 and a sense of place (Chalmin-Pui et al., 2021). Researchers propose that these linkages could  
461 be explained by the emotional attachment participants have with the place and the familiarity  
462 of the species in question (Southon et al., 2017), as well as aesthetic factors such as colour  
463 (Hoyle et al., 2018), or smell (Pálsdóttir et al., 2021). In our study, social cohesion was a  
464 significant predictor of perceived richness of all three taxa. It possible that residents who spent  
465 more time in the streetscapes socialising with neighbours and passers-by, could also be more  
466 familiar with streetscape biodiversity. Equally, people may be spending more leisure time  
467 socialising where the streetscape is more biodiverse, as shown in the Netherlands (de Vries et  
468 al., 2013). These findings imply that biodiverse streetscapes could contribute to improved  
469 mental wellbeing through multiple biopsychosocial pathways (Hartig et al., 2014). The low  
470 levels of variance explained in the models imply that there are other variables influencing  
471 human wellbeing that were not captured within the scope of our study. For instance, McElroy  
472 et al. (2021) illustrate a rich network of individual, community, and place-based characteristics  
473 that are connected to mental wellbeing when measured using SWEMWBS, like financial  
474 difficulties and physical ill-health. While these can be addressed in population-level studies,  
475 they are difficult to account for at finer scales.

476

477 Increased traffic-related pollution negatively impacted actual pollinator and flowering plant  
478 richness. One potential explanation is that residents on highly polluted streets spend less time  
479 maintaining front gardens, deterred by the streetscape pollution, therefore reducing the richness  
480 of plants and subsequently pollinators (given their reliance on floral resources, Baldock et al.,  
481 2019). Further, pollinators themselves can be directly impacted by noise pollution (Morley et  
482 al., 2013; Davis et al., 2018; Leonard et al., 2018; Girling et al., 2013). The approximate hearing  
483 ranges of many invertebrate orders (e.g. Hymenoptera, Diptera, Lepidoptera, Hemiptera, and  
484 Orthoptera) are well below the frequency of noise exerted by traffic, which could therefore

485 disrupt their communication, behaviour, and eventually reproductive success (Morley et al.,  
486 2013). Research has also shown that traffic exhaust can degrade floral odours and subsequently  
487 interrupt pollinator foraging habits (Girling et al., 2013). To combat these effects, strategic  
488 streetscape planting regimes could be used to attenuate noise pollution, while also increasing  
489 habitat availability for pollinators. This is pertinent given the accumulating evidence of  
490 substantial declines in pollinators worldwide (Potts et al., 2010; Powney et al., 2019; Zattara  
491 and Aizen, 2021). Pollinators are also increasingly recognised as important by members of the  
492 public (Hall and Martins, 2020). As such, small-scale changes at the streetscape scale by city  
493 planners, local council, and residents, could contribute to their conservation.

494

495 Traffic-related pollution (noise and NO<sub>2</sub>) was negatively associated with increasing flowering  
496 plant richness (path  $a'$  in the structural equation model), and flowering plant richness positively  
497 impacted mental wellbeing (path  $b'$ ). We therefore expected mediation to be shown through a  
498 significant indirect effect ( $ab$ ) (i.e. plants act as a mediator between traffic-related pollution  
499 and mental wellbeing). Indeed, some plant species are known to act as a buffer to anthropogenic  
500 noise (industrial, traffic, construction, social) (Han et al., 2018) and intercept air pollutants like  
501 NO<sub>2</sub> (Abhijith et al., 2017). They are most effective when used in dense planting regimes,  
502 particularly when species have thick stems that act as a barrier (Ow and Ghosh, 2017), or  
503 complex foliage that can scatter and refract (Fang and Ling, 2005). In our study, the most  
504 commonly recorded plant species across the streetscapes in Leeds included the Leyland cypress  
505 (*Cupressus leylandii*), holly (*Ilex aquifolium*), and garden privet (*Ligustrum ovalifolium*),  
506 which support dense evergreen foliage. Despite this body of evidence, we did not find a  
507 mediating role for plant richness. Potentially this was due to a lack of statistical power (Aglar  
508 and Boeck, 2017), given the complexity of our models (Fairchild and McDaniel, 2017).  
509 However, it may also be that other metrics of plant biodiversity, such as abundance or  
510 vegetation structure, would be more appropriate. Regardless, our findings imply that further  
511 work might uncover such a mediating effect of plants between pollution and mental wellbeing.

512

513 We also found that traffic-related pollution was positively associated with perceived tree  
514 richness. This incidental finding is probably because participants' are not able to perceive  
515 actual tree richness accurately, or that tree planting regimes are uniform across the city of Leeds  
516 (we found no differences in tree richness between streetscape pollution level categories high,  
517 medium, and low). However, noise pollution was measured at breast height, and the structural  
518 characteristics of the vegetation was not a focus of this study, despite its known effects

519 (Abhijith et al., 2017). Further research on tree characteristics is therefore needed to disentangle  
520 their role in the relationship between pollution and human wellbeing. Nonetheless, our results  
521 emphasise the need to encourage diverse planting regimes across urban streetscapes to reduce  
522 traffic-related pollution. This is reinforced by the WHO (Europe), who recommend that road  
523 traffic noise pollution should be below 53 dB to maintain population health (WHO, 2018), a  
524 value well below what we measured in our study.

525

526 Streetscape air pollution (PM<sub>2.5</sub>) had no significant direct effect on people's mental wellbeing  
527 or happiness at the streetscape level. Despite PM<sub>2.5</sub> concentrations varying across the  
528 streetscapes, the range of measured values were within air pollution limits deemed acceptable  
529 by the WHO. PM<sub>2.5</sub> concentrations in our study were below 10 µg/m<sup>3</sup> in general, above which  
530 air pollution-related mortality events are known to increase significantly (WHO, 2006).  
531 Additionally, concentrations exceeding this level have been associated with decreased hedonic  
532 wellbeing (Zhang et al., 2019), and increased incidence of depressive symptoms (Roberts et  
533 al., 2019). However, we caution against drawing comparisons between our observations and  
534 those made by public bodies, and used in other studies (e.g. Roberts et al., 2019; Zhang et al.,  
535 2019), due to the temporal extent of observations. Furthermore, at the streetscape level,  
536 researchers have demonstrated that PM<sub>2.5</sub> concentrations are much reduced during the summer  
537 months (Gehrig and Buchmann, 2003). Nonetheless, we did identify several incidences when  
538 concentrations were well above >10 µg/m<sup>3</sup>, indicating that more localised pollution events  
539 could be occurring, but would require much finer-scale assessment to investigate any effects  
540 on wellbeing. These explanations could also explain why we did not detect an effect of PM<sub>2.5</sub>  
541 concentration on our biodiversity metrics across streetscapes as a whole.

542

## 543 **5. Conclusion**

544 The complex interlinkages between pollution, biodiversity, and human wellbeing are largely  
545 unexamined. Understanding the mechanisms through which pollution and biodiversity  
546 influence human wellbeing could help inform the development of strategic planning initiatives  
547 that maximise human quality of life. Through structural equation modelling, we were able to  
548 examine these potentially complex associations simultaneously. Our study makes a novel and  
549 timely contribution to the evidence about how traffic-related pollution on residential  
550 streetscapes can negatively impact biodiversity, and simultaneously how streetscape  
551 biodiversity can positively affect human wellbeing. However, we also demonstrate that, at

552 present, there is insufficient evidence to indicate that biodiversity itself offers a mediating  
553 effect between pollution and human wellbeing. These findings are applicable to cities  
554 worldwide, where elevated pollution levels, increasing populations, and stressful lifestyles  
555 pose detrimental impacts to human wellbeing. Several wider implications stem from this  
556 research, including the importance of streetscape greenery for those who stay closer to home,  
557 and its role as a habitat resource for pollinator conservation. As such, city planners, councils,  
558 and residents should strive to reduce traffic-related pollutants across the streetscapes of  
559 polluted cities, as well as encourage diverse planting regimes, with subsequent benefits for the  
560 health and wellbeing of urban dwellers worldwide.

561

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