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

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ABSTRACT

Most of the global population are urban, with inhabitants exposed to raised levels of pollution. Pollutants negatively impact human wellbeing, and can alter the structure and diversity of ecosystems. Contrastingly, urban biodiversity can positively contribute to human wellbeing. We know little, however, about whether the negative impacts of pollution on wellbeing could be lessened for householders living on more biodiverse streets, as the complex interlinkages between pollution, biodiversity and wellbeing have rarely been examined. Here, we used structural equation modelling to simultaneously test whether biodiversity (actual and perceived) mediates the relationship between traffic-related pollution (noise, dB; nitrogen dioxide, NO_2) or air pollution ($PM_{2.5}$) and wellbeing (mental wellbeing, happiness). In summer 2019, we conducted questionnaires and biodiversity surveys, and collected noise and air pollution data, from households ($n = 282$) across the streetscapes of Leeds, UK. Biodiversity (actual or perceived) showed no mediating effects. However, increased flowering plant richness was positively associated with mental wellbeing. Traffic-related pollution negatively affected pollinator and flowering plant richness, but not wellbeing. This could be because householders are not exposed to high levels of noise or NO_2 because they do not maintain front gardens on noisier streets. There was no measurable effect of air pollution on biodiversity or wellbeing. These findings shed light on the complex mechanisms through which biodiversity could improve human wellbeing. Enhancing the diversity of plant species in streetscapes would have a positive effect on wellbeing, further emphasising the important role that biodiverse urban streetscapes play in improving the liveability of cities.

1. Introduction

By 2050, approximately 68% of the global human population will reside in urban areas (United Nations, 2018). Urban living poses challenges for the physical health, mental health and wellbeing of town and city dwellers, particularly because of associated stressful lifestyles and exposure to elevated levels of pollution (Abbot, 2012; Peen et al., 2010; Roberts et al., 2019; World Health Organization, 2006; Zhang et al., 2019). Indeed, noise pollution (e.g. road, rail, and air traffic) and air pollution (e.g. nitrogen dioxide, NO_2; particulate matter, PM) are two of the three main risk factors for environmental disease burden in Europe (Hänninen et al., 2014). As such, city planning and urban design professionals are seeking to implement land-use planning initiatives that reduce the detrimental health and wellbeing impacts of pollution on the growing urban population (Giles-Corti et al., 2016).

Human wellbeing is known to improve with the presence of urban green infrastructure (e.g. parks, gardens, streetscape greenery), providing an opportunity for restoration, gaining distance from psychological demands, and reducing stress and fatigue (Kaplan and Kaplan, 1989; Ulrich et al., 1991). For instance, in a study of 51 European cities, city greenness was positively associated with improved self-reported quality of life (Giannico et al., 2021). While empirical research has demonstrated that individuals in greener neighbourhoods are happier and healthier (Ambrey and Fleming, 2014; Sarkar et al., 2018; Wang et al., 2020; White et al., 2013; Wood et al., 2017), it remains unclear what specific qualities or attributes of the 'green' (e.g.

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biodiversity) could underpin the positive effects (Dallimer et al., 2012; Wheeler et al., 2015). This concept is further complicated by a discrepancy between what attributes are objectively present, compared with what attributes people perceive to exist (Pett et al., 2016). For instance, Dallimer et al. (2012) found no relationship between actual butterfly or plant species richness and human wellbeing, but a positive one with perceived species richness for both taxa. Disentangling these differences has important implications for planning and policy recommendations aimed at maximising the beneficial effects of biodiversity on human wellbeing.

Streetscape biodiversity and front gardens are largely overlooked in nature-wellbeing research to date (Chalmin-Pui et al., 2019). However, they could theoretically offer many of the same benefits as back gardens (e.g. de Bell et al., 2020), which provide important ecological resources for biodiversity (Baldock et al., 2019; Davies et al., 2009), and increased quality of life, emotional wellbeing (Goddard et al., 2013), and restorativeness for people (Young et al., 2020). Indeed, Chalmin-Pui et al. (2021) showed that when ornamental plants were added to residential front gardens, householders experienced lower levels of stress, more positive emotions, relaxation and pride. Spano et al. (2021) showed that the presence of natural features in people's homes, including views of greenery, can improve mental health and wellbeing. Streetscape greenery is also publicly viewable, experienced by neighbours and passers-by, potentially underpinning opportunities for more cohesive social interactions that subsequently improve human wellbeing for a wider range of people than just the householders themselves (Chalmin-Pui et al., 2021).

In some streetscapes, traffic-related pollution could exert a considerable influence on biodiversity and wellbeing. The negative impacts of traffic-related pollutants (noise and NO_2) on pollinators have been widely documented (e.g. disruption of communication, Morley et al., 2013; heightening of physiological stress, Davis et al., 2018). Noise pollution directly impacts human wellbeing, for example through sleep disturbance, which could lead to cardiovascular ill-health (Bai et al., 2020; Münzel et al., 2018). Roadside verge pollinators respond negatively to increased traffic, probably because of pollution and wind turbulence (Phillips et al., 2020). Air pollution can also reduce species-specific growth rates of urban vegetation (Honour et al., 2009). Concomitantly, higher levels of air pollutants (e.g. $PM_{2.5}$) can directly decrease human wellbeing (e.g. emotional wellbeing, Zhang et al., 2019; depressive symptoms, Roberts et al., 2019). Although, specific species of plants and trees can contribute to air pollution through the release of hydrocarbons and allergens, which can detract from wellbeing (see Hartig et al., 2014). Some elements of biodiversity have the potential to alleviate or offset the negative consequences of noise and air pollution on wellbeing. Streetscape trees and shrubs can scatter and refract noise levels at traffic-level frequencies (Fang and Ling, 2005; Han et al., 2018; Klingberg et al., 2017). Similarly, vegetation can passively screen and filter air, while the presence of leaves on some species actively absorb pollutants (Klingberg et al., 2017; Nowak et al., 2006). The extent to which pollutants can be obscured are dictated by vegetation characteristics such as height, width, and density (Abhijith et al., 2017).

Urban biodiversity could play a pivotal role in the relationship between pollution and human wellbeing in neighbourhood streetscapes, particularly as pollutants will be exacerbated by roadside traffic. We therefore investigate how traffic-related pollution (noise pollution, dB, and nitrogen dioxide, NO_2) as well as streetscape air pollution (particulate matter, $PM_{2.5}$) impacts human wellbeing via the mediating role of biodiversity. Given the likely complexity of these associations, we used parallel mediation models to simultaneously examine how both objective and perceived measures of biodiversity (pollinators, flowering plants and trees) influence the effect of pollution on residents' wellbeing. We hypothesise that (H1) higher levels of biodiversity will have a mediating effect, reducing the impact of pollution on human wellbeing, and (H2) higher perceived biodiversity will have a similar mediating effect. This research makes a novel empirical contribution to the small but growing evidence-base on streetscape biodiversity, pollution and human wellbeing. These mediation effects remain largely understudied to date, despite urbanisation accelerating worldwide, but could offer crucial evidence to inform the sustainable design of biodiverse and liveable cities.

2. Methods

2.1. Study system

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

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

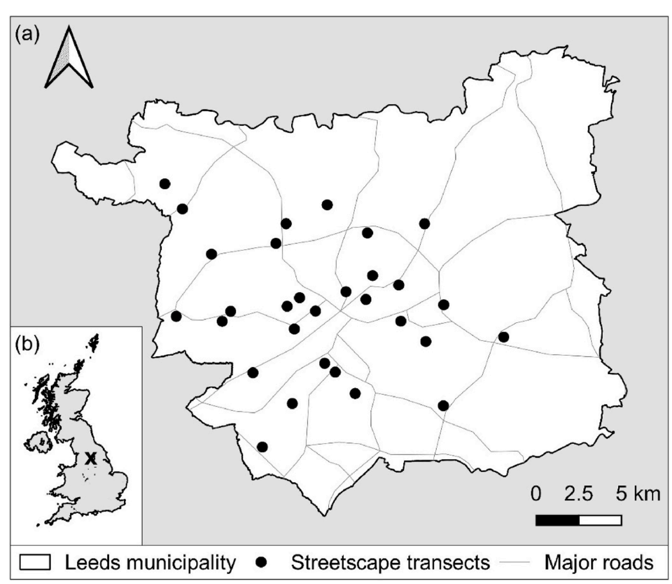

Fig. 1. Study area showing (a) the municipality of Leeds, its major roads (grey lines) and the location of each of the 30 streetscape transects (black circles). (b) The location of Leeds (cross) in the UK.

roads were considered as all other roads less than 4 m wide.

We selected 10 streetscapes from each of the three pollution categories, giving 30 in total. Along each of these roads, a 200 m long streetscape transect was positioned so that all the transects were located within a different ward (UK administrative areas) of Leeds, maximising spatial variation across the city. Each streetscape transect comprised all green infrastructure in residential front gardens and within the street itself (e.g. street trees, road verges, central reservations, all other vegetation). To further ensure sample independence, and diffusion of pollutants between sites of varying pollution categories, straight-line distances between the streetscape transects were at least 0.6 km, with the vast majority being >1 km apart. This distance is also greater than the forage range of most pollinator species in urban landscapes (Garbuzov et al., 2015; Langellotto et al., 2018). Each streetscape transect was selected to make sure the sampled households captured citywide variation in housing type (i.e. detached; semi-detached; terraced), which is indicative of the size of gardens (Loram et al., 2007) and sociodemographic/economic characteristics.

2.2. Pollution

Traffic-related pollution was captured using ambient nitrogen dioxide (NO_2) and noise pollution. NO_2 concentration was measured using diffusion tubes. Three tubes were situated equidistant along each transect, positioned 2.5 m high on lampposts. Tubes were left in place for four weeks in May/June and three weeks in July/August 2019. An average concentration was calculated across all tubes and sampling periods. While this methodology is unable to capture incidences where NO_2 might be temporarily elevated (e.g. during rush hour), we were interested in the longer-term (rather than momentary) effects of traffic-related pollution on people's wellbeing in the streetscape where they lived, and thus compare between sites.

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

Particulate matter concentration ($PM_{2.5}$) was recorded using the IQAir AirVisual Pro monitor (measuring range: 0.3–2.5 μm; accuracy to the nearest 1 μg/m^3), and noise pollution (decibels; dB A) were record using a Reed ST-8850 sound level meter (measuring range: 30–130 dB; 0.1 dB resolution). Both particulate matter and noise level data were obtained by walking at a slow pace along both sides of the 200 m transect on two occasions, at different times of day (morning and afternoon), between May and August 2019. The measurement period was ~15 min in duration, and start times ranged from 09:39 to 17:21. The sound meter recorded one measurement per second, while the particular matter monitor recorded once every 10 s (equating to 600 values for noise, and 60 values for particulate matter, per transect per visit). There were no missing values. Median values were used to represent each streetscape transect. To minimise bias caused by variation in meteorological conditions that can affect air quality, streetscapes across all pollution level categories (high, medium and low) were sampled in groups on the same day or adjacent days with comparable weather conditions (Mues et al., 2012).

2.3. Greenness

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

2.4. Actual measures of biodiversity

Pollinators, such as butterflies and bees, are a prominent component of urban biodiversity during the day. Streetscape pollinator richness and abundance were estimated using a pollinator transect sampling approach modified from Baldock et al., (2015). Each pollinator transect was 2 m in height and 4 m in width, walked at a steady pace, following the boundary between the pavement and residential gardens (including road verges where present) along the side of the streetscape with the greater extent of green infrastructure for 200 m. All pollinators observed were recorded as one of 20 morphological functional groups (Supplementary Table A.1), giving a measure of morpho-functional group richness of pollinators. Pollinator transects were conducted in May and July 2019 when weather conditions were suitable. Flowering plant richness was estimated for each streetscape transect by identifying all plant species in flower (excluding grasses, sedges and wind-pollinated forbs) across two survey visits. Tree richness was also assessed, based on all individuals ≥2 m in height.

2.5. Questionnaire

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

2.6. Perceived measures of biodiversity

We asked householders about their perceptions of biodiversity in their streetscape, which was termed 'street environment' in the questionnaire. It was emphasised that the phrase 'street environment' covered all green infrastructure associated with front gardens and the street itself. Using five-point scales, participants were asked to estimate the total number of pollinating insect (*Fewer than 5, 5 to 9, 10 to 13, 14 to 19, 20 or more*), flowering plant (*Fewer than 10, 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 to 20, 21 or more*) across their streetscapes. Categories for the scales were based on numbers of species likely to be present, based on our previous research (Goddard et al., 2013).

2.7. Wellbeing outcomes

The two self-reported wellbeing outcomes we measured (Appendix A: Supplementary text) were mental wellbeing and happiness, using existing scales validated in nature-health research (van Herzele and De Vries, 2012; Houlden et al., 2017). Self-reported (rather than objective) measures of health and wellbeing are commonly used, and known to be both efficient and robust (Andrews and Crandall, 1976; Lucas, 2018). We used the Short Warwick-Edinburgh Mental Wellbeing Scale (SWEMWBS) (Stewart-Brown et al., 2009) to assess the primary components of mental health, including hedonic (feeling of positive emotions, satisfaction) and eudaimonic (functioning, relationships, sense of purpose) domains (Dolan and Metcalfe, 2012). SWEMWBS is a 7-item scale, where participants are asked to "*Tick the box that best describes your experience of each over the last two weeks*" and respond with one of five options (*None of the time, Rarely, Some of the time, Often, All of the time*). Scores are summed to produce an initial raw score, then transformed using a conversion table (Stewart-Brown et al., 2009) to give a final metric ranging from 7 (lowest possible wellbeing) to 35 (highest possible wellbeing). SWEMWBS has shown adequate validity and reliability in people with mental health diagnoses (e.g. depression, anxiety, schizophrenia) (Vaingankar et al., 2017). Happiness was evaluated using a single item (Fordyce, 1988), which asks participants asked "*In your life in general, how happy would you say you are?*", and asked to respond on a continuous scale from 1 (*extremely unhappy*), to 10 (*extremely happy*). This single-item scale has shown good concurrent and convergent validity with positive wellbeing measures (optimism, self-esteem, positive affect; Abdel-Khalek, 2006).

2.8. Covariates

We sought to account for participant's feelings of social cohesion, given it can play a considerable role in people's mental health and wellbeing, and therefore influence people's experiences of the world around them ((Hartig et al., 2014; Markevych et al., 2017). Social cohesion was measured using five items, three positive (positive affect) and two negative (negative affect), drawn from the work by Sampson et al. (1997).

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

Additionally, we collected data on participant age, employment status, years of household occupancy, and gender to control for their potential effects on the mediators and wellbeing outcomes (Kendal et al., 2012; Dzhambov et al., 2018; Hartig et al., 2014). As many participants did not wish to share information about gross household income, we obtained median gross household income from the most recent UK census in 2011 (Office for National Statistics, 2018), using the most spatially resolved data available to cover the location of each streetscape transect ('Lower Layer Super Output Area' or LSOA; Experian Limited, 2007).

2.9. Analytical framework

A series of parallel mediation models (Fig. 2) were constructed to test the influence of species/morpho-functional group richness (actual versus perceived) mediating the relationship between traffic-related

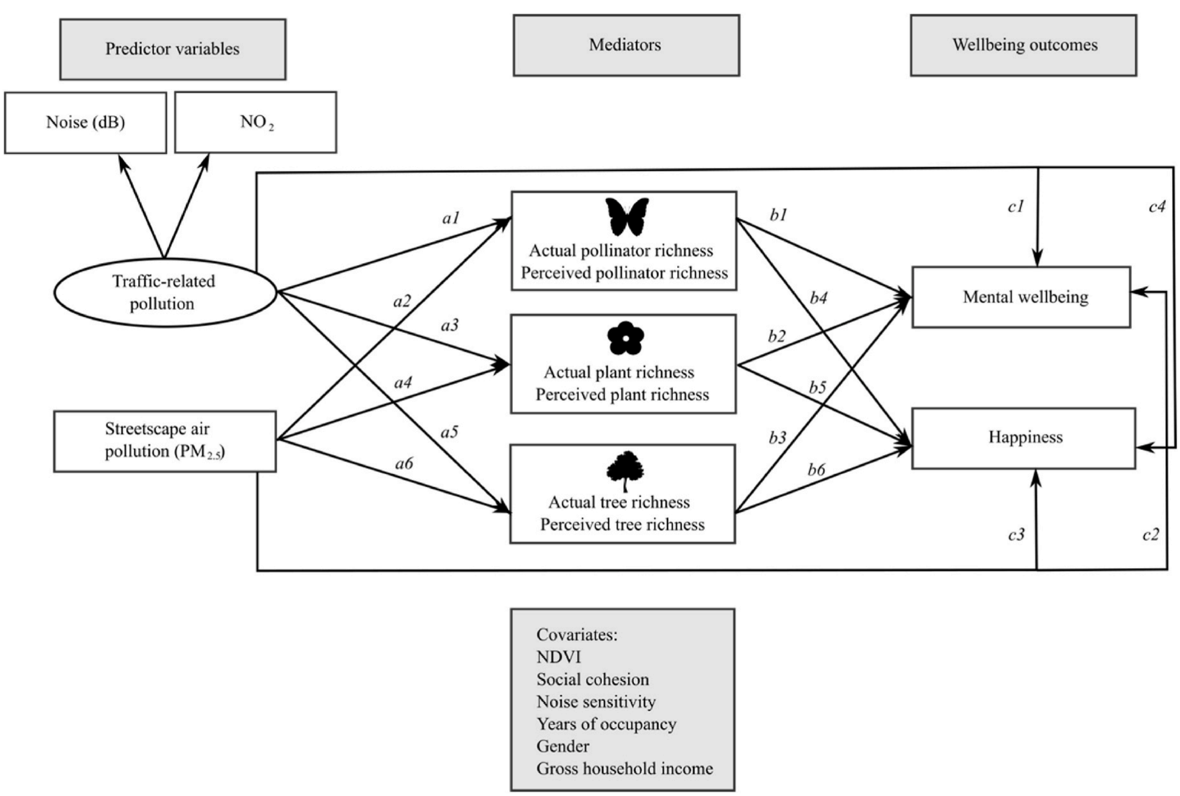

Fig. 2. The parallel mediation model framework to assess the effects of predictors (traffic-related and streetscape air pollution) on wellbeing outcomes (mental wellbeing and happiness), via potential mediators (actual versus perceived richness for each taxa). Actual pollinator richness was assessed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

(noise and NO_2) and streetscape air pollution ($PM_{2.5}$), with human wellbeing (mental wellbeing, happiness), while adjusting for covariates (NDVI, social cohesion, noise sensitivity, years of household occupancy, gender, gross household income). This approach simultaneously calculates regressions between the predictors and mediators (paths *a1 - a6*), the predictors and wellbeing outcomes while holding the mediators constant (direct effect, paths *c1 - c4*), and the mediators and wellbeing outcomes (*b1 - b6*). The indirect effect (*a*b*) measures how the predictors influence the wellbeing outcomes as a result of the influence of the predictors on the mediators, which, in turn, also affect the wellbeing outcomes.

2.10. Data analysis

Analyses were performed using R Statistical Software version 3.6.0 (R Core Team, 2020). We tested for associations between variables using Spearman's rank correlation tests for non-normal data, and Kruskal Wallis tests for continuous and categorical data. We used a G-test to examine whether our sample was comparable to census data for Leeds in terms of gender, age, and ethnicity. Due to a large number of participants not disclosing their age ($n = 31$), and given that age was closely associated with years of household occupancy ($r = 0.69$, $p < 0.001$), we used the latter in our structural equation models to maximise our sample size and to account for its likely influence on how people perceive the streetscape (Dzhambov et al., 2018). We found a strong association between employment and gross household income ($X^2 = 6.46$, df = 2, $p < 0.05$), so used the latter due to its continuous nature and possible influence on the maintenance of front gardens within the streetscape (Kendal et al., 2012). Gender was treated as binary.

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

A latent variable termed traffic intensity, indicated by noise pollution (dB) and NO_2, was used as a model predictor. Two separate models were run to test mediation of actual and perceived species/morpho-functional group richness, respectively. Error variances between the three mediators in each model (pollinators, flowering plants, and trees) and two outcome variables (mental wellbeing and happiness) were free to covary due to the plausible associations between them (e.g. more pollinators are likely to be found where flowering plant richness is greater). Models were estimated using a maximum likelihood estimator and a Satorra-Bentler scaled test statistic that is robust to non-normality. We gauged the quality of our models using a combination of model fit indices (Hu and Bentler, 1999), employing a chi-square adjusted for clustered data (*'pval.pFsum'* function, Oberski, 2014), standardised root mean square residual (SRMR), root mean square error of approximation and its 95% confidence intervals (RMSEA), and the comparative fit index (CFI) to identify good model fit (X^2 $p > 0.05$, CFI >0.95, RMSEA <0.06, RMSEA 95% confidence intervals <0.06, SRMR <0.08) (Oberski, 2014; Barrett, 2007; MacCallum et al., 1996; Hu and Bentler, 1999). We further tested for indirect effects by computing Monte Carlo confidence intervals (9999 replicates) ('semTools' package, Jorgensen et al., 2021) for our models where paths '*a*' and '*b*' were significant (MacKinnon et al., 2004; Preacher and Selig, 2012), thus accounting for non-normality in the sampling distribution of the indirect effect (Fairchild and McDaniel, 2017).

3. Results

3.1. Descriptive statistics

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

Measures of NO_2 and noise increased across the pollution level categories (low, medium, high) used to stratify the study system (Supplementary Fig. A.1), indicating that the sampling effort was broadly representative for these streetscape pollutants. The sampled $PM_{2.5}$ concentrations, however, were highly variable across the pollution level categories.

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

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

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

	Min	Median	Max	IQR
Predictors				
Noise pollution (dB(A))	67.50	82.40	93.60	10.6
NO_2 (ppm)	6.11	36.43	80.99	29.53
Air pollution ($PM_{2.5}$ μg/m^3)	2.26	4.69	8.83	3.17
Wellbeing outcomes				
Mental wellbeing	13.33	22.33	35.00	5.05
Happiness	1.00	8.00	10.00	2.00
Mediators				
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
Covariates				
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	51,598.00	15,707.00

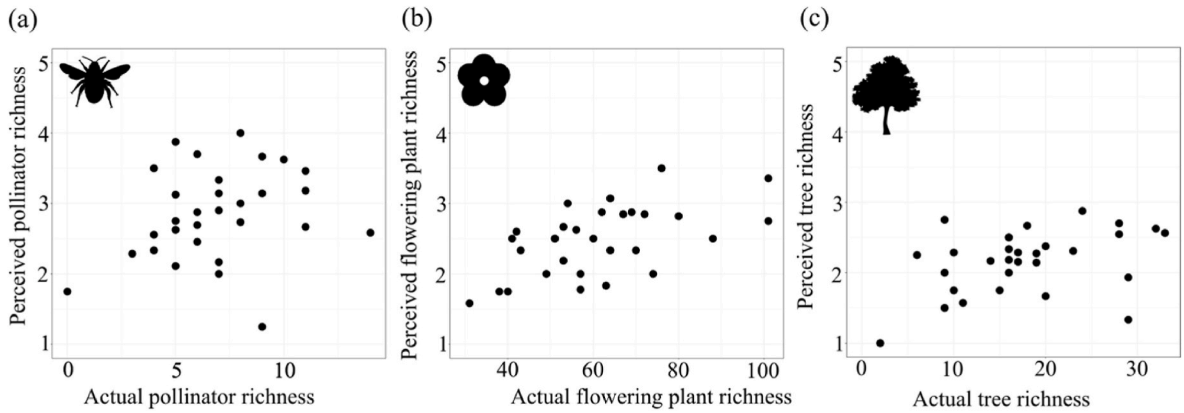

Fig. 3. Association between actual and perceived richness of (a) pollinators = butterfly symbol ($r = 0.29$, $p = 0.126$) (b) flowering plants = flower symbol ($r = 0.54$, $p = 0.002$), and (c) trees = tree symbol ($r = 0.37$, $p = 0.044$). Actual pollinator richness was assessed by morpho-functional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

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).

Models revealed that traffic-related pollution had a positive association with perceived tree richness, but no effect on perceptions of other taxa (Fig. 4b). Streetscape air pollution had no direct nor indirect effects on people's perceptions of pollinator, flowering plant or tree richness in their streetscapes, or their mental wellbeing or happiness (Fig. 4b; Supplementary Table A.4). Given that traffic-related pollution (noise and NO_2) was inversely associated with flowering plant richness, which in turn was positively associated with mental wellbeing (significance of path *a* and path *b;* Fig. 4a), we computed Monte Carlo confidence intervals for the indirect effect, but found no evidence of mediation as the confidence interval crossed zero (−0.134, 0.057).

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

4. Discussion

Pollution can be detrimental to human wellbeing, contributing to the prevalence of psychological and physical health disorders amongst urban dwellers (Abbot, 2012; World Health Organization, 2006; 2018). For the first time to our knowledge, we test whether biodiversity (actual and perceived) could intervene in this relationship, using structural

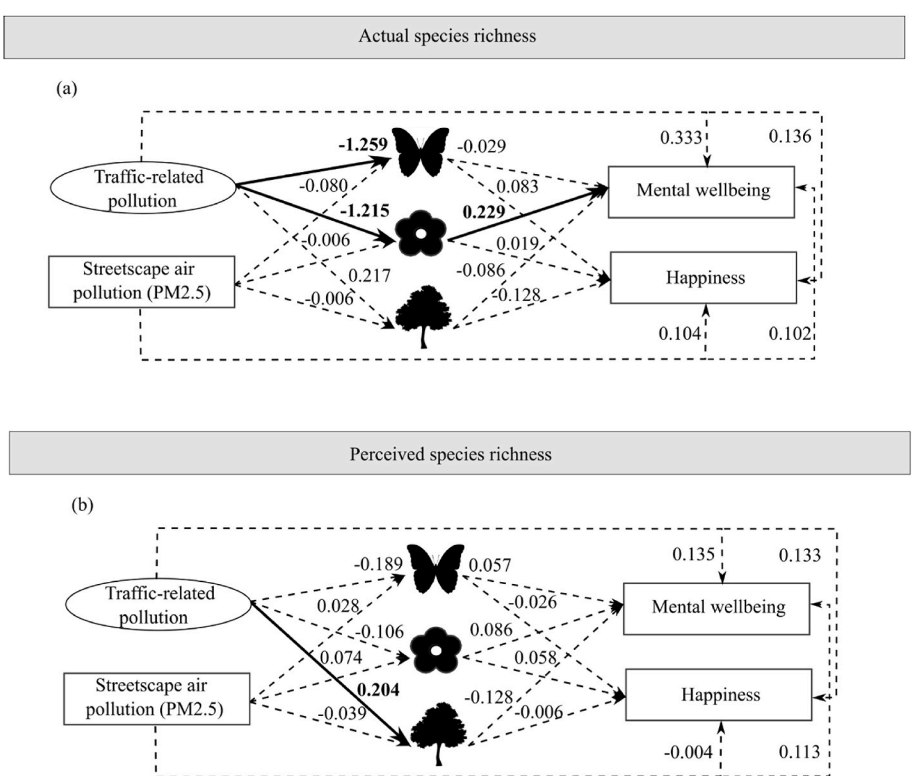

Fig. 4. Parallel mediation models showing the effects of traffic-related pollution (latent variable representing noise and NO_2) and streetscape air pollution ($PM_{2.5}$) and on mental wellbeing and happiness, mediated by (a) actual richness, and (b) perceived richness (of pollinators, flowering plants, and trees, respectively). Actual pollinator richness was assessed by morpho-functional group richness. Richness of pollinators = butterfly symbol, plants = flower symbol, trees = tree symbol. Latent variable represented by an oval, measured variables represented by rectangles. All models are adjusted for covariates (NDVI, noise sensitivity, years of household occupancy, gender, social cohesion, and gross household income). Plots display the unstandardised beta estimates after rows containing missing values were removed ($n = 239$), with statistically significant estimates and respective paths highlighted in bold ($p < 0.05$). Mediating effects (*ab* path) tested separately, and latent variable estimates and error covariances not shown for readability (Supplementary Tables A.3, A.4).

equation modelling to consider the likely complex associations. Our findings show traffic-related pollution (noise, NO_2) can have detrimental impacts on pollinator and flowering plant richness, and increased flowering plant richness can benefit mental wellbeing. However, our approach revealed no statistical support for (H1) actual biodiversity will mediate the relationship between pollution and wellbeing, or (H2) perceived biodiversity will also have a mediating effect, where neither measure of biodiversity showed any mediation, therefore unveiling further complexity in how pollution, biodiversity, and human wellbeing are associated.

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

Increased traffic-related pollution negatively impacted actual pollinator and flowering plant richness. One potential explanation is that residents on highly polluted streets spend less time maintaining front gardens, deterred by the streetscape pollution, therefore reducing the richness of plants and subsequently pollinators (given their reliance on floral resources, Baldock et al., 2019). Further, pollinators themselves can be directly impacted by noise pollution (Morley et al., 2013; Davis et al., 2018; Leonard et al., 2019; Girling et al., 2013). The approximate hearing ranges of many invertebrate orders (e.g. Hymenoptera, Diptera, Lepidoptera, Hemiptera, and Orthoptera) are well below the frequency of noise exerted by traffic, which could therefore disrupt their communication, behaviour, and eventually reproductive success (Morley et al., 2013). Research has also shown that traffic exhaust can degrade floral odours and subsequently interrupt pollinator foraging habits (Girling et al., 2013). To combat these effects, strategic streetscape planting regimes could be used to attenuate noise pollution, while also increasing habitat availability for pollinators. This is pertinent given the accumulating evidence of substantial declines in pollinators worldwide (Potts et al., 2010; Powney et al., 2019; Zattara and Aizen, 2021). Pollinators are also increasingly recognised as important by members of the public (Hall and Martins, 2020). As such, small-scale changes at the streetscape scale by city planners, local council, and residents, could contribute to their conservation.

Traffic-related pollution (noise and NO_2) was negatively associated with increasing flowering plant richness (path *a'* in the structural equation model), and flowering plant richness positively impacted mental wellbeing (path *b'*). We therefore expected mediation to be shown through a significant indirect effect (*ab*) (i.e. plants act as a mediator between traffic-related pollution and mental wellbeing). Indeed, some plant species are known to act as a buffer to anthropogenic noise (industrial, traffic, construction, social) (Han et al., 2018) and intercept air pollutants like NO_2 (Abhijith et al., 2017). They are most effective when used in dense planting regimes, particularly when species have thick stems that act as a barrier (Ow and Ghosh, 2017), or complex foliage that can scatter and refract (Fang and Ling, 2005). In our study, the most commonly recorded plant species across the streetscapes in Leeds included the Leyland cypress (*Cupressus leylandii*), holly (*Ilex aquifolium*), and garden privet (*Ligustrum ovalifolium*), which support dense evergreen foliage. Despite this body of evidence, we did not find a mediating role for plant richness. Potentially this was due to a lack of statistical power (Agler and De Boeck, 2017), given the complexity of our models (Fairchild and McDaniel, 2017). However, it may also be that other metrics of plant biodiversity, such as abundance or vegetation structure, would be more appropriate. Regardless, our findings imply that further work might uncover such a mediating effect of plants between pollution and mental wellbeing.

We also found that traffic-related pollution was positively associated with perceived tree richness. This incidental finding is probably because participants' are not able to perceive actual tree richness accurately, or that tree planting regimes are uniform across the city of Leeds (we found no differences in tree richness between streetscape pollution level categories high, medium, and low). However, noise pollution was measured at breast height, and the structural characteristics of the vegetation was not a focus of this study, despite its known effects (Abhijith et al., 2017). Further research on tree characteristics is therefore needed to disentangle their role in the relationship between pollution and human wellbeing. Nonetheless, our results emphasise the need to encourage diverse planting regimes across urban streetscapes to reduce traffic-related pollution. This is reinforced by the WHO (Europe), who recommend that road traffic noise pollution should be below 53 dB to maintain population health (World Health Organization, 2018), a value well below what we measured in our study.

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

5. Conclusion

The complex interlinkages between pollution, biodiversity, and human wellbeing are largely unexamined. Understanding the mechanisms through which pollution and biodiversity influence human wellbeing could help inform the development of strategic planning initiatives that maximise human quality of life. Through structural equation modelling, we were able to examine these potentially complex associations simultaneously. Our study makes a novel and timely contribution to the evidence about how traffic-related pollution on residential streetscapes can negatively impact biodiversity, and simultaneously how streetscape biodiversity can positively affect human wellbeing. However, we also demonstrate that, at present, there is insufficient evidence to indicate that biodiversity itself offers a mediating effect between pollution and human wellbeing. These findings are applicable to cities worldwide, where elevated pollution levels, increasing populations, and stressful lifestyles pose detrimental impacts to human wellbeing. Several wider implications stem from this research, including the importance of streetscape greenery for those who stay closer to home, and its role as a habitat resource for pollinator conservation. As such, city planners, councils, and residents should strive to reduce traffic-related pollutants across the streetscapes of polluted cities, as well as encourage diverse planting regimes, with subsequent benefits for the health and wellbeing of urban dwellers worldwide.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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