Biodiversity and environmental stressors along urban walking routes

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1 Abstract

There is increasing focus on designing liveable cities that promote walking. However, urban walking routes can expose people to adverse environmental conditions that reduce health, well-being and biodiversity. Our primary objective is to assess how urban form is associated with environmental quality, including biodiversity, for people moving through urban spaces. We assess a range of environmental conditions that influence human health and biodiversity (temperature, noise and particulate pollution) and biodiversity of three taxa (trees, butterflies and birds) along 700m public walking routes embedded in 500m x 500m grid cells across three UK towns. Cells are selected using random stratification across an urbanisation intensity gradient. Walking routes in more built-up areas were noisier and hotter; noise levels further increased in areas with more industrial land-use and large roads. There was no evidence of vegetation mitigating noise or temperature, but there was some evidence that increased vegetation cover mitigated small particulate pollution. Walking routes in more built-up environments had lower butterfly, bird and native tree species richness, and reduced butterfly abundance. Large roads were associated with reduced bird species richness and increased noise was associated with reduced bird abundance. Most specific measures of vegetation in the surrounding matrix (median patch size, structural complexity at 1.5m resolution) were not detectably associated with biodiversity along walking routes, indicating minimal beneficial spill-over. Increased garden cover in the surrounding matrix was associated with less abundant and less species-rich butterfly communities. Our results highlight considerable heterogeneity in the environmental quality of urban walking routes and pedestrians' potential to experience biodiversity along these routes, driven by reduced quality in areas with more built cover. A greater focus is needed on mitigating adverse effects of specific features of the built environment (roads, industrial areas, noise)

surrounding walking routes to enhance the co-benefits of more biodiversity and healthier conditions for pedestrians.

2 Key words

green infrastructure; sustainable cities; walkability; pedestrian experience; luxury effect; habitat fragmentation

3 Introduction

Urbanisation alters many aspects of the environment, including temperature, air quality, hydrology, and habitat type and availability, which in turn affect the extent to which urban areas can support biodiversity and provide high-quality, healthy, environments for people (Beninde et al., 2015, Haines-Young and Potschin, 2010). For example, urban air pollution (World Health Organization, 2013), noise pollution (Hankey and Marshall, 2017, Xie and Kang, 2009) and high temperatures (Kovats and Hajat, 2008) lead to poor human health and premature mortality, and negatively impact many species (Salmón et al., 2018, Davis et al., 2018, Johnson et al., 2019). It is thus crucial to understand the consequences of urban form for the way in which people experience both the physical and biological environment if we are to plan, build and manage cities and towns in ways that support and sustain human wellbeing.

In the twentieth century, city design primarily focused on the efficiency of vehicle movement at the expense of effective planning for pedestrians (Lo, 2009). The 'walkability' of cities has, however, come under increasing scrutiny in recent years, given the now widely recognised health benefits of walking (Smith et al., 2008). The walkability of an area is concerned with the extent to which walking is supported by features of the urban environment (Shashank and Schuurman, 2019); a 'walkable' urban area provides routes that are convenient and health-promoting for pedestrians (Tight, 2018, Zuniga-Teran et al., 2017). Walkability has frequently been assessed based on the structural features of the urban landscape (e.g. "Density, Diversity and Design", Cervero and Kockelman, 1997, Lu et al., 2017), although there is no single agreed measure (Fonseca et al., 2021, Shields et al., 2021). Increasingly, measures of the experience of the environment are incorporated when assessing walkability (Zuniga-Teran et al., 2016, Zuniga-Teran et al., 2017), including consideration of thermal comfort and availability of amenity facilities (Labdaoui et al., 2021, Zuniga-Teran et al., 2017). Similarly, urban walking routes can have high concentrations of air pollution that limit their potential to promote health, thus reducing their walkability (Hankey and

Marshall, 2017, Howell et al., 2019, James et al., 2015). Although walkability measures are often integrative, detailed studies of the benefits of each aspect tend to be addressed separately and there are few studies addressing their interrelationships (James et al., 2017).

Despite the well-recognised health benefits of engaging with urban greenspace (World Health Organization, 2016), greenspace is not well integrated into measures of walkability. The term greenspace takes diverse meanings between and within disciplines (Taylor and Hochuli, 2017). In this study, greenspace refers to any vegetated area, public or private, deliberately planted or self-sown. Indeed, much of the research on walkability ignores greenspace (Tobin et al., 2022) although it can influence walking patterns and behaviour (Zuniga-Teran et al., 2017). As an example, greater tree cover can reduce the perceived walking time of a route (Jia and Wang, 2021). More specifically, the potential for pedestrians to experience biodiversity along walking routes is very poorly understood – despite evidence that exposure to biodiversity can enhance peoples' enjoyment and well-being (Douglas and Evans, 2022, Lai et al., 2019, Southon et al., 2018), and recognition that enhancing biodiversity can enrich the walking experience (Brierley and Cockett, 2017).

Spatial patterns in urban land cover and land use are very fine-grained (Norton et al., 2016). To reflect these, sampling strategies that focus on walking routes should capture biodiversity and environmental characteristics across a complex mixture of greenspace patches that vary in their type, size and composition, and are embedded within a matrix of variable built environments. Whilst environmental conditions are often measured in built-up areas and greenspace, such sampling does not always occur at scales that appropriately capture local and landscape-scale variation (Pope and Wu, 2014), or the experience of urban residents (Nerriere et al., 2005). Furthermore, we are not aware of any previous research that integrates field-based measurements of biodiversity and environmental conditions at the scales relevant to human experience along urban walking routes. Such studies are needed to enable urban planners to understand which attributes of urban environments determine conditions along these walking routes.

Here, we examine existing publicly accessible walking routes in urban areas and sample both environmental conditions and biodiversity on these routes. We aim to understand what features of urban environments are associated with environmental conditions that are known to benefit people's experience of the walking route – i.e. enhanced biodiversity and reduced urban stressors (air pollution, noise and heat). Specifically, we ask what characteristics of the urban environment in which walking routes are embedded can predict: (1) biodiversity (abundance and species richness of butterflies, birds and trees) and (2) environmental conditions (particulate matter, temperature, noise), on publicly accessible walking routes in diverse urban areas. This approach enables identification of features of the urban environment of joint benefit for pedestrians and biodiversity that are amenable to modifications through planning and design. Our sampling is focused on three adjacent towns in southern England, comprising an urban area of approximately 253 km^2 and containing most of the typical urban forms and development history found in the UK.

4 Methods

4.1 Study area and site selection

This study was conducted in three towns in southern England that share similar topographical and climatic features. Milton Keynes (52°0' N, 0°47' W; 89 km^2; population 229,941 in 2011 (Office for National Statistics, 2013)) is a planned 'new town' that was established in the 1960s and built up around villages dating from the medieval period. The extensive development is set around a grid network of roads and large areas of well-connected public greenspace. Luton (51°52' N, 0°25' W; 58 km^2; population 258,018) developed rapidly during the nineteenth century as an industrial centre; the inner town is dominated by terraced housing. Bedford (52°8' N, 0°27' W; 36 km^2; population 106,940) developed in the Middle Ages as a market town and has subsequently developed around a road network radiating from the town centre.

The combined urban areas of the three towns were overlaid with a 500 m x 500 m grid (referred to as 'tiles'). Survey areas were selected using random stratification to represent a range of urban forms. Each tile was categorised as one of 25 urban forms using remotely sensed data from airborne LiDAR (see section 4.4.2) and OS MasterMap® data to generate five categories of building cover (<5%; 5% to 9.99%; 10% to 14.99%; 15% to 19.99%; ≥ 20%) and five categories of vegetation cover 0.5 m or taller (10%; 10% to 19.99%; 20% to 29.99%; 30% to 39.99%; ≥40). MasterMap is a comprehensive mapping layer produced by the UK's Ordnance Survey. Five tiles of each urban form were randomly selected for surveying, ensuring no tiles were directly adjacent to each other, nor included unusual features (e.g. an airport). This yielded 112 survey tiles (<125 because there were fewer than five tiles in four of the urban form categories).

Transect routes were established along publicly accessible walking routes in order to sample biodiversity and environmental conditions in areas available for any mobile pedestrian to experience. The routes incorporated influence from private greenspace, including residential gardens, as many ran along footpaths adjacent to residential front gardens or along alleyways or linear parks adjacent to residential back gardens. Transect length was set at 700 m as this enabled us to sample the major land cover types within each tile with sufficient replication of individual sampling points (see below) whilst being sufficiently short to enable us to maximise the number of grid cells that could be covered. Data were collected during spring and summer 2014. Surveys of butterflies, trees and environmental conditions were undertaken along the transect routes. Due to their greater mobility, bird surveys were undertaken at four points across the tile, thereby capturing the full extent of the birdlife people might experience along the walking route. Three tiles were excluded from all analyses, as access issues meant a full complement of surveys could not be completed. An additional two tiles had complete surveys for bird points, but full transect walks could not be completed. So, a maximum of 107 tiles from a pool of 109 were surveyed for any taxon or environmental conditional, although some data were further excluded from analysis (for details, see: 4.2 Biodiversity surveys; 4.3 Environmental condition surveys).

4.2 Biodiversity surveys

Butterfly species richness and total abundance were recorded along each transect route (Table S1), in suitable weather conditions (defined following UKBMS, 2014). Surveys used the standard Pollard Walk method, i.e. a 5 m cube ahead of the surveyor (Pollard and Yates, 1993) on one visit between 27th June and 7th August 2014, starting between 10 am and 5 pm. This represents the period of maximum butterfly activity, and includes the flight periods of most butterfly species in the region (Balmer, 2014).

Tree data were collected at eight fixed survey points spaced 100 m apart along the transect routes. Trees (>2 m tall) within a 10-m radius of each survey point (which incorporated public and private land) were identified to species level (Table S2), and classified as native or exotic using definitions from PLANTATT (Hill et al., 2004) and Stace (2010), supported by expert advice. Tile-level tree abundance and species richness were calculated for all trees and only native trees.

Bird surveys were conducted at four publicly accessible points in each tile in May and June 2014, all points fell on, or very close to, the transect routes. Eight tiles were excluded from analysis because fewer than four points were surveyed due to access issues or coverage by unsuitable habitat (e.g. open water). At each point, two 10-minute counts (one in May and one in June) of all species seen and heard were conducted between 06:00 am and 10:00 am (Table S3). All birds detected within a 60-m radius of the points were recorded, and the maximum abundance of each species at each point across the two surveys was calculated as an estimate of true abundance of species that are more detectable both earlier and later in the season. Surveys were timed to maximise bird detectability; complete detection of large, vocal and territorial species (i.e. the vast majority of the bird community) present within the limited radius considered was expected. Pooled data across survey points were used to calculate indices of total abundance and species richness per tile.

4.3 Environmental condition surveys

Environmental recording equipment was mounted on aluminium backpack frames designed to avoid interference from the wearer's body. Noise and temperature data were recorded at the eight fixed survey points (see tree data description, above), plus two additional points, as environmental condition data were more time-efficient to collect than the tree data. Environmental data were collected under the same conditions as the butterfly surveys, and usually at the same time, and always on weekdays to minimise the influence of day of the week on the results. The additional points were in dominant land covers along the transect, or, when appropriate, in rarer land-cover types that were not sufficiently captured by the other survey points.

Noise was measured for 60 seconds, whilst standing still, at each of the ten survey points, using an Optimus Red Sound Level Meter (CR:160 series, Cirrus). We measured the A-weighted, equivalent sound level (LAeq, measured as dBA) which integrates sound energy over time and is weighted towards 500 Hz to 6 kHz, i.e. the frequencies to which human hearing is most sensitive (Cirrus Research plc, 2015) and within which the frequency range of the local avifauna's vocalisations fall (Xeno-canto Foundation and Naturalis Biodiversity Center, 2018). For each tile, we then calculated mean noise.

Temperature (°C) was recorded at each point using the median of two readings over a 30-second period from each of two Thermochron® ibuttons (model DS1923), following a two-minute stabilisation period (as per manufacturer guidelines). When modelling air temperature as a response variable, we expressed these measurements as a temperature anomaly relative to the median of readings during the transect walk period recorded at the nearest meteorological station (Wolfson station, Cranfield University; 52°04'27.4"N, 0°37'41.5"W; data recorded at five-minute intervals) which is located in a rural setting. A tile-level mean value was calculated across the point-level anomalies. The median temperature recorded at the Wolfson station for the period the transect was walked was used as a predictor variable in one model.

Particulate matter (particles per 0.1 cubic foot) was recorded along the entire transect at a rate of one reading per minute using a laser particle counter (Dylos DC1700). Readings separated small particles (>0.5 µm; small particulate pollution and bacteria, fungal spores, etc.) and large particles (>2.5 µm; larger particulate pollution and pollen). Eight tiles were dropped from analysis for particulate matter due to missing data. Tile-level mean concentrations from recordings along the full transect.

4.4 Characterising the urban environment

We used field-collected data and additional metrics derived from secondary data to characterise the urban environment within our focal survey tiles to use as predictors when modelling biodiversity and environmental conditions. These predictors characterised the built environment, the quality and extent of greenspace, abiotic stressors and socio-economic variables. Table 1 provides the rationale for the selection of the suite of predictor variables used when modelling each response variable.

Table 1: Predictor variables and rationale for inclusion in each linear model. Variables are grouped as: built cover metrics, greenspace metrics, abiotic stressor, socio-economic variable and control variables. The distribution of each variable is in Figure S1. For references supporting the rationale, refer to Table S4. When variables were transformed for analysis, the transformation used is provided in the second column.

Variable	Transformation?	Butterfly (richness, abundance)	Birds (richness, abundance)	Trees – all (richness, abundance)	Trees – native (richness, abundance)	Temperature (mean, standardised)	Noise (LAeq) (mean)	Particulates – Large PM (mean)	Particulates – Small PM (mean)	Rationale for inclusion
Built cover metrics										
Distance to urban edge (m)	Square root	X	X	X	X	X	X	X	X	A common measure of urbanisation, related to biodiversity patterns and environmental conditions in some cities and habitats.
Impervious surface cover (% cover)		X	X	X	X	X	X	X	X	A standard metric of urbanisation intensity and absence of greenspace.
Industrial land use (% cover)	Natural log						X	X	X	Industry is a source of noise and particulate pollutants directly and due to increased vehicle traffic in the vicinity.
Large road density (m/m^2)	Natural log	X	X				X	X	X	A measure of urbanisation intensity, fragmentation and expected to increase conditions (such as noise and pollution) that may adversely influence butterflies and birds. Large road density is used rather than total road density as smaller roads provide limited dispersal barriers for mobile taxa, and typically have smaller impacts on noise and pollution.
Road density (all roads) (m/m^2)				X	X					Trees are frequently planted along roads, especially smaller residential roads, thus a

										measure of total road density is more appropriate than a measure of large road density.
Greenspace metrics										
Median foliage height diversity (FHD)		X	X			X	X	X	X	A greater diversity of foliage heights indicates more structurally complex vegetation which is hypothesised to benefit bird and butterfly assemblages. More complex vegetation structures may also enhance removal of particulates and ameliorate temperature and noise. Not included in models of tree diversity due to circularity that would result.
Median patch size (m^2)		X	X	X	X	X	X	X	X	A measure of habitat fragmentation. Large and continuous areas of green cover are typically expected to be more effective at supporting biodiversity (at least for species specialising on the core habitat type), and potentially at reducing noise, particulate matter, and temperatures
Residential gardens (% cover)		X	X	X	X					Gardens can enhance habitat and food provision for butterflies and birds. Gardens can also influence tree cover and composition (with a greater number of non-natives) than in other urban land uses.
Tree diversity (Shannon index)						X	X	X	X	Diversity enhances the range of growth forms and range of species traits which may increase overall capacity to remove pollutants, and screen noise.
Abiotic stressor										

Noise (LAeq) (dBA)			X							Noise can disrupt bird vocal communications and change community composition. Current research suggests limited direct effects of noise on butterflies and no effects on trees.
Socio-economic variable										
Index of Multiple Deprivation		X	X	X	X					The luxury effect refers to observations that there is typically higher biodiversity in wealthier areas. Some taxa, however, find greater availability of key resources in less wealthy and less intensively managed areas.
Control variables										
Air temperature (°C)		X								Median air temperature recorded at the Wolfson weather station (Cranfield University; 52°04'27.4"N, 0°37'41.5"W) as matched field recorded data were not always available. The median rather than mean was used due to skew in the data. Will influence activity and thus detectability of endothermic butterflies but not our other taxonomic groups.
Day of season (day 1 = first transect visit, 27th June)		X				X	X	X	X	Takes seasonal variation into account. Not included for birds as data are aggregated in a manner that takes seasonal variation into account (see main text).
Hour of day						X	X	X	X	Takes diurnal variation in temperature and human activity patterns into account.
Interaction effects										
Median foliage height diversity (FHD) * Large road density								X		In models of noise levels and particulate matter (large and small), interactions with FHD and large roads were tested. Vegetation

										that is structurally diverse (high FHD) could mitigate adverse impacts of industrial land use and large roads on noise levels and particulate matter (large and small). The only significant interaction was between FHD and large road density, and was included in that model.

4.4.1 Built cover metrics

We used five metrics of the built-up environment that broadly relate to urbanisation intensity: (i) distance to the urban edge; (ii) percentage impervious surface cover; (iii) percentage cover of industrial land use; (iv) total road density; (v) total density of large roads. Road density, especially of large roads, provides an index of habitat fragmentation given the considerable barriers to animal movement generated by such infrastructure. Distance of the tile from the urban edge was calculated from the tile centroid to the nearest edge of the built-up area of the focal town. This was identified using 0.5 m resolution colour infrared (CIR) 2007 and 2009 aerial photography acquired from Landmap (http://landmap.mimas.ac.uk/). Percentage impervious surface cover was obtained from a land-cover layer (Grafius et al., 2016) that identified vegetation, buildings, water, and other non-vegetated (mostly paved) surfaces by classifying the CIR aerial images based on Normalised Difference Vegetation Index (NDVI) and using OS MasterMap® data. Industrial land-use percentage cover was acquired from a land-use layer generated from OpenStreetMap®, OS AddressBase Plus© and OS MasterMap® (for details see Section S1). Total road density and density of large roads were obtained from the OS Mastermap® Integrated Transport Layer (May 2016 release) (OS ITN). Large roads were defined as the OS ITN descriptors 'Dual Carriageways', 'Collapsed Dual Carriageways', and 'Slip Roads' and capture those with heavy traffic flows.

4.4.2 Greenspace metrics (quality and extent of vegetation)

We used four greenspace metrics: (i) median foliage height diversity; (ii) median patch size; (iii) residential garden cover (the outdoor area associated with private residences, including vegetated and unvegetated surfaces); (iv) tree species diversity. Percentage vegetation cover was not used as it is essentially the inverse of impervious surface cover.

Foliage height diversity (FHD) is a measure of structural complexity of vegetation (MacArthur and MacArthur, 1961), which is often a key determinant of urban biodiversity (Beninde et al., 2015). It was calculated from full-waveform LiDAR data, collected between June and September 2012, by processing to a high-resolution voxel map (1.5 m horizontal and 25 cm vertical) of canopy cover. This processing used the deconvolution method described in Hancock *et al.* (2017). The Shannon diversity index was then calculated for each vertical stack of canopy cover voxels (Goetz et al., 2007) to produce a 1.5 m resolution FHD map. LiDAR data were missing for 13 tiles, which were dropped from models that included FHD. The median FHD measure was calculated for green areas across the tile.

Median patch size provides an additional measure of fragmentation (in addition to road density) and size of urban habitat patches is frequently associated with biodiversity (Lepczyk et al., 2017), although this relationship can vary between taxa and ecosystems (Fahrig, 2003). We used Fragstats 4.2 (McGarigal et al., 2012, Grafius et al., 2018) to measure the size of individual greenspaces that were delimited using the land cover layer (see 'Built cover metrics'). Patches were defined using an 8-cell neighbourhood rule. Residential gardens can provide substantial resources for wildlife, although are often not managed to prioritise this function (Gaston et al., 2007). Residential garden percentage cover was calculated from the 'Mixed Surfaces' category in the OS MasterMap® Topography Layer. Tree species diversity was measured as the Shannon diversity index using our field data.

4.4.3 Other variables

Within urban areas, anthropogenic noise can be a strong determinant of avian species richness and community structure (Arévalo et al., 2022, Barbosa et al., 2020) so, when modelling avian richness and abundance, we used mean LAeq, as measured in the field, as a measure of this abiotic stressor. Socio-economic conditions can influence biodiversity via the luxury effect, where species richness tends to be higher in more affluent areas due to, for example, more, and higher quality, greenspace (Hope et al., 2003). Socio-economic conditions were measured using the Index of Multiple Deprivation (IMD), which is the official small-area deprivation measure for England. It is a weighted composite index of seven domains of deprivation (income, employment, education, health, crime, barriers to housing and services, living environment) calculated for Census Lower-layer Super Output Areas (LSOAs), reported in deciles (Office for National Statistics, 2015). As LSOAs did not spatially coincide with the survey tiles, we used the IMD (2012-13 Smith et al., 2015) covering the largest proportion of the tile. We also took three additional variables (date, ambient temperature and time of surveys) into account that could influence one or more of our response variables (Table 1; Table S4).

4.5 Statistical analysis

Statistical modelling was undertaken using R v3.4.2 (R Core Team, 2017). For each response variable, we constructed a generalised least squares (GLS) model using the nlme package (v. 3.1, Pinheiro et al., 2017), with the objective of determining which characteristics of urban environments are associated with biodiversity and environmental conditions along the walking routes. Predictor variables for each statistical model were selected *a priori* to test pre-specified hypotheses whilst taking other potentially influential variables into account and not over-specifying the models (Table 1; Table S4 for supporting citations). We adopted a full model approach, following Whittingham *et al.* (2006). All response variables were standardised (mean-centred and SD-scaled, following transformation where indicated, see Tables S5, S6 and Figures S2, S3) to assist interpretation of parameter estimates. Preliminary checks confirmed all models met the assumptions of normality (following transformation of response variables in some cases; Table 1) and that including predictors' polynomial terms did not improve model fit. We also conducted several robustness checks. First, for spatial autocorrelation (for details, see Section S2). Second, we ensured that correlations between predictor variables used in each model were below the threshold (0.7) at which model results can be unduly influenced by collinearity (Dormann et al., 2013), and checked the influence of using alternative predictors that were highly correlated with those included in our core models (for details, see Section S2). Finally, we checked for the influence of interactions between predictors (where interactions could plausibly arise). In models of noise and air pollution we checked for interactions between vegetation structural complexity (FHD) and industrial land use, and between FHD and large roads; we did this because structurally complex vegetation could mitigate adverse impacts of industrial land use and large roads on noise levels (Fang and Ling, 2003, Jang et al., 2015) and particulate pollution (Abhijith et al., 2017, Janhäll, 2015). Only the interaction between FHD and large roads on large particulate matter was significant and thus included in the final models.

5 Results

Across our study area we detected 22 butterfly species (all native; Table S1), 174 tree-species (41 native; Table S2), and 65 bird species (61 native; Table S3).

5.1 Which predictors are associated with biodiversity along walking routes?

Butterfly species richness and abundance, bird species richness and native tree species richness all decreased as impervious surface cover increased (Table 2A; Figure 1). In addition, bird species richness was negatively associated with the density of large roads, and bird abundance was negatively associated with noise levels (LAeq), an abiotic stressor (Table 2A; Figure 1). Butterfly richness and abundance were negatively associated with garden cover (Table 2A; Figure 1).

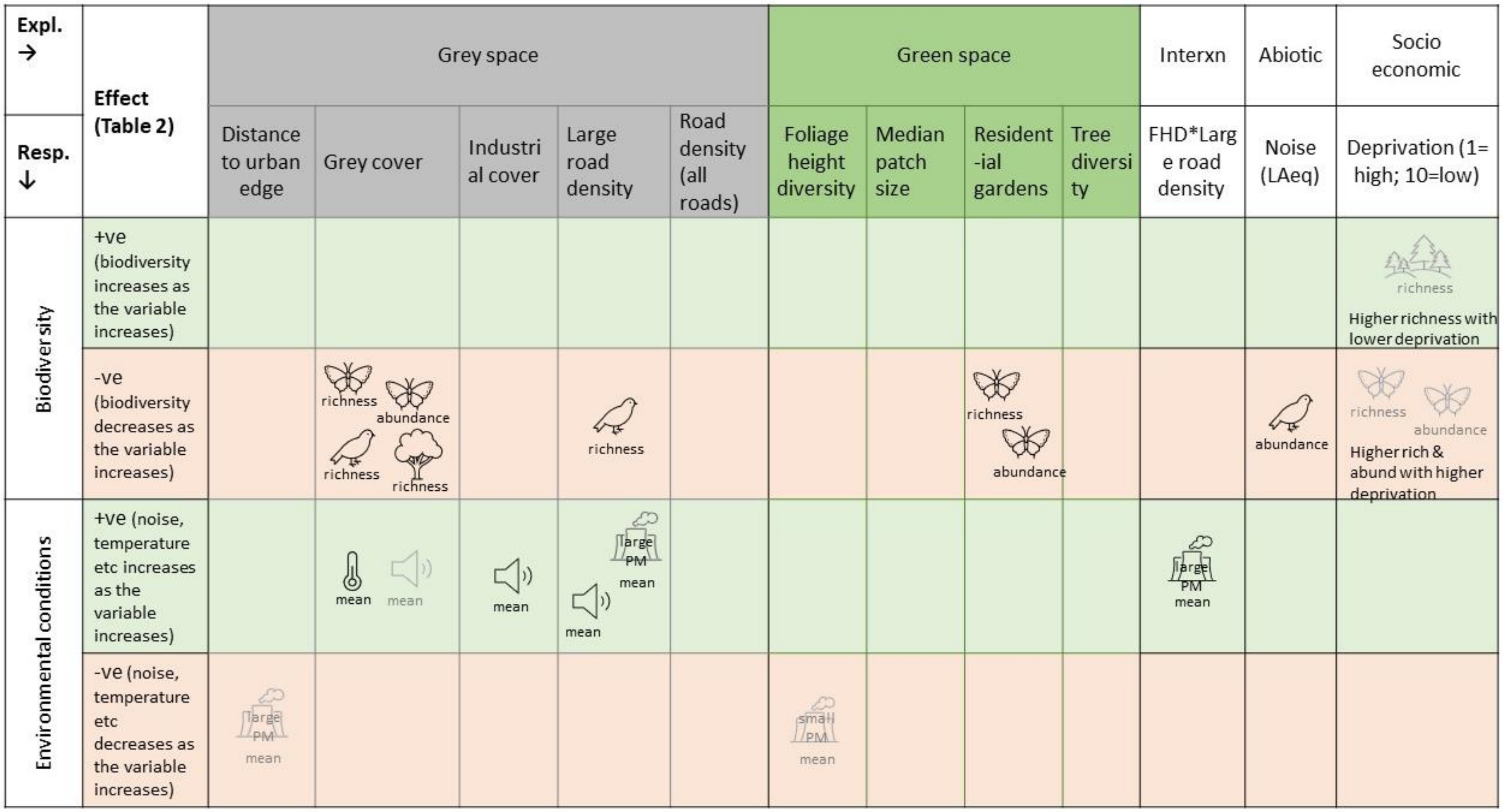
The socio-economic predictor variable (IMD) was not significantly ($p<0.05$) associated with any of our biodiversity outcome variables, but was marginally positively associated with total tree species richness ($p=0.065$), and marginally negatively associated with butterfly richness ($p=0.089$) and abundance ($p=0.061$).

Table 2: Linear model results. Table A shows the model output for the biodiversity response variables. Table B shows the model output for the environmental condition response variables. Distributions of these variables are available in Tables S5 and S6, Figures S2 and S3. Response variables are listed vertically, with the number of observations included in the model (n) and a measure of goodness-of-fit (Pearson's correlation coefficient (r)). Predictor variables are listed horizontally in the order presented in Table 1. Blank cells indicate that the predictor variable was not used in the model (see Table 1). The top value is the parameter estimate, with the standard error below, and the p-value listed third. Bold values in cells with dark grey shading are significant at $p<0.05$. Cells with light grey shading are significant at $p<0.10$. Models do not include spatial structure unless indicated; *Ratio spatial structure, **Linear spatial structure, ***Exponential spatial structure.

		Built cover metrics				Greenspace metrics			Abiotic	Socioeconomic	Control	
A	**Intercept**	**Distance to urban edge**	**Impervious surface cover**	**Large road density**	**Road density (all roads)**	**Foliage height diversity**	**Median patch size**	**Residential gardens**	**Noise (LAeq)**	**Index of Multiple Deprivation**	**Air temperature**	**Day of season**
Butterfly species richness (sqrt)	**1.921**	0.043	**-0.226**	-0.084		-0.062	-0.032	**-0.161**		-0.132	**0.258**	-0.034
n = 96; r = 0.51	**(±0.069)**	(±0.077)	**(±0.08)**	(±0.077)		(±0.078)	(±0.079)	**(±0.079)**		(±0.076)	**(±0.080)**	(±0.079)
	<0.001	0.583	**0.006**	0.278		0.433	0.685	**0.044**		0.089	**0.002**	0.668
Butterfly abundance (sqrt)	**3.163**	-0.098	**-0.528**	-0.113		-0.207	-0.172	**-0.416**		-0.366	**0.626**	-0.230
n = 96; r = 0.51	**(±0.175)**	(±0.195)	**(±0.202)**	(±0.195)		(±0.198)	(±0.201)	**(±0.199)**		(±0.193)	**(±0.203)**	(±0.198)
	<0.001	0.618	**0.011**	0.563		0.298	0.394	**0.039**		0.061	**0.003**	0.250
Bird species richness	**18.045**	0.338	**-1.788**	**-0.711**		-0.139	0.032	-0.343	0.160	0.187		
n = 89; r = 0.66	**(±0.276)**	(±0.311)	**(±0.334)**	**(±0.329)**		(±0.313)	(±0.328)	(±0.325)	(±0.343)	(±0.308)		
	<0.001	0.280	**<0.001**	**0.034**		0.658	0.924	0.296	0.641	0.547		
Bird abundance	**102.494**	4.903	-2.102	-3.544		-1.102	-1.293	4.600	**-6.901**	-3.249		
n = 89; r = 0.45	**(±2.698)**	(±3.039)	(±3.266)	(±3.211)		(±3.056)	(±3.207)	(±3.180)	**(±3.352)**	(±3.015)		
	<0.001	0.111	0.522	0.273		0.719	0.688	0.152	**0.043**	0.284		
Tree species richness	**9.114**	0.402	-0.714		-0.044		0.474	0.242		0.764		
n = 107; r = 0.30	**(±0.392)**	(±0.442)	(±0.531)		(±0.616)		(±0.438)	(±0.496)		(±0.409)		
	<0.001	0.365	0.181		0.943		0.281	0.627		0.065		
Tree abundance (sqrt)*	**7.059**	0.118	-0.324		-0.715		-0.004	-0.269		0.233		
n = 107; r = 0.38	**(±0.582)**	(±0.387)	(±0.382)		(±0.45)		(±0.329)	(±0.403)		(±0.334)		
	<0.001	0.761	0.399		0.115		0.989	0.506		0.488		
Native tree species richness (sqrt)	**1.999**	-0.021	**-0.216**		0.016		0.089	-0.047		0.019		
n = 107; r = 0.35	**(±0.064)**	(±0.072)	**(±0.087)**		(±0.101)		(±0.071)	(±0.081)		(±0.067)		
	<0.001	0.767	**0.014**		0.876		0.214	0.560		0.779		
Native tree abundance (ln)**	**2.773**	0.015	-0.136		-0.192		0.086	-0.193		0.054		
n = 107; r = 0.42	**(±0.176)**	(±0.158)	(±0.156)		(±0.185)		(±0.125)	(±0.164)		(±0.141)		
	<0.001	0.926	0.385		0.302		0.494	0.240		0.702		

		Built cover metrics				Greenspace metrics			Interaction	Control	
B	Intercept	Distance to urban edge	Impervious surface cover	Industrial cover	Large road density	Foliage height diversity	Median patch size	Tree diversity	FHD * large road density	Day of season	Hour of day
Temperature (°C) mean standardised*	2.104	0.118	**0.320**			-0.040	-0.101	0.000		**-0.341**	**0.330**
n = 96; r = 0.35	(±3.809)	(±0.116)	**(±0.105)**			(±0.113)	(±0.132)	(±0.104)		**(±0.107)**	**(±0.106)**
	0.582	0.311	**0.003**			0.723	0.447	0.999		**0.002**	**0.003**
Noise (LAeq) mean	**51.854**	0.154	0.902	**1.676**	**1.721**	-0.542	0.018	-0.375		-0.443	0.337
n = 96; r = 0.63	**(±0.419)**	(±0.465)	(±0.487)	**(±0.482)**	**(±0.470)**	(±0.476)	(±0.472)	(±0.45)		(±0.431)	(±0.439)
	<0.001	0.741	0.068	**0.001**	**<0.001**	0.258	0.971	0.407		0.307	0.445
Large PM mean (ln)	**8.904**	-0.065	-0.066	0.014	**0.136**	-0.041	-0.023	0.012	**0.124**	**0.178**	-0.050
n = 88; r = 0.66	**(±0.034)**	(±0.037)	(±0.04)	(±0.04)	**(±0.038)**	(±0.041)	(±0.042)	(±0.037)	**(±0.041)**	**(±0.035)**	(±0.035)
	<0.001	0.088	0.105	0.731	**0.001**	0.314	0.583	0.744	**0.003**	**<0.001**	0.156
Small PM mean (ln)***	**11.482**	0.101	0.004	-0.019	0.084	-0.127	0.013	0.014		**0.294**	**-0.143**
n = 88; r = 0.43	**(±0.184)**	(±0.086)	(±0.070)	(±0.072)	(±0.070)	(±0.065)	(±0.075)	(±0.063)		**(±0.079)**	**(±0.065)**
	<0.001	0.244	0.957	0.787	0.231	0.053	0.862	0.822		**<0.001**	**0.031**

Figure 1: Summary of responses (Resp.) of biodiversity variables and environmental conditions, to built cover and greenspaces, as well as abiotic conditions and socioeconomic variables (Expl.; full model results in Table 2). Not all variables are included in all models (Table 1). Box shading indicates the model coefficient direction: green=positive, orange=negative. Icons (from the Microsoft Office collection) indicate the p-value and response variable. Black icons: p<0.05; Grey icons: p<0.1. The single tree icon indicates native trees; the cluster of trees icon indicates all tree species; the water drop icon indicates humidity; the thermometer icon indicates temperature; the loudspeaker icon indicates noise; the smokestacks icon indicates particulate matter. Note that air temperature and day of season are taken into account for butterfly models, and day of season and hour of day were taken into account for all environmental condition models (Tables 1, 2, S4).



5.2 Which predictors are associated with environmental conditions along walking routes?

After taking into account time of day and date, walking routes located in more built-up areas were warmer, with an increase in percentage impervious surface cover of 10% points predicted to increase mean temperature by 3.2°C (Table 2B, Table S6, Figure 1). Noise levels increased with increased industrial cover and the density of large roads, and marginally significantly increased ($p=0.068$) with increased percentage impervious surface cover (Table 2B, Table S6, Figure 1). Both large and small particulate matter concentrations increased throughout our sampling period (from late June to early August). Large particular matter concentrations were higher along walking routes in locations with a greater density of large roads. There was a significant, positive interaction between large road density and foliage height diversity, indicating that the rate of increase in particulate matter as large road density increased was steeper in areas with more structurally complex vegetation. There was also a marginally negative, significant relationship ($p=0.088$) between large particulate matter and distance to the urban edge. Small particulate matter concentrations exhibited a marginally significant ($p=0.053$), negative relationship with foliage height diversity.

6 Discussion

We used field-based transect sampling to capture variation in the biodiversity and environmental conditions along urban walking routes at scales relevant to human experiences. We found that biodiversity (birds, trees, butterflies) increased and potentially adverse environmental conditions, such as noise and air pollution, decreased in areas with less built infrastructure. These findings were in line with expectations. The positive role of vegetation was less evident from our models, and residential gardens were negatively associated with butterfly biodiversity on walking routes. The methodological approach provides a potentially useful way of collecting data across a range of parameters influencing the conditions on walking routes, which is rarely undertaken.

6.1 Biodiversity

Walking routes with the greatest species richness of native trees, birds and butterflies occurred in grid cells with reduced impervious surface cover, and, by implication, higher amounts of vegetation. This is consistent with many previous studies (e.g. Chace and Walsh, 2006, Steenberg, 2018, Dallimer et al., 2012b, Kurylo et al., 2020). Overall, there was a somewhat poorer fit for the tree models (richness and abundance) than for other diversity measures. This could be due to the static nature of trees and their longer lifespans slowing their responses to changed conditions such as increased impervious surface cover (Le Roux et al., 2014), and more direct influences of human activity on tree species composition (via planting and felling decisions), compared to mobile taxa (Bourne and Conway, 2014). The dominant role of human planting decisions in determining patterns of non-native tree species richness may explain why only native tree species richness was associated with impervious surface cover and not all trees. The mental health and well-being benefits people derive from biodiversity can arise from increased abundance (Marselle et al., 2020, Cox et al., 2017) or richness (Cameron et al., 2020, Southon et al., 2018) of organisms. Relationships between biodiversity and well-being are, however, complex (Lovell et al., 2014, Pett et al., 2016); perceived increases in biodiversity can lead to increased enjoyment of the area, and sometimes increased well-being (Cameron et al., 2020, Southon et al., 2018, Douglas and Evans, 2022). Whilst people can sometimes accurately detect differences in species richness across locations, this is, however, not always the case (Cameron et al., 2020, Southon et al., 2018, Dallimer et al., 2012a), and the realised wellbeing benefits of increases in actual richness along walking routes need further exploration.

Walking routes had fewer bird species when located in areas with a higher density of large roads. This could be driven by adverse effects of roads on habitat fragmentation, as many bird species are reluctant to cross large roads (Johnson et al., 2017, Cox et al., 2016). Roads can also have adverse impacts on insect populations (Baxter-Gilbert et al., 2015) that comprise an essential component of most bird species' nestlings' diets. Notably, we found that, when taking road effects into account, walking routes embedded in noisier locations had reduced avian abundance. Noise is a well-known stressor of birds associated with roads that can influence community composition, abundance and behaviour (Francis, 2015, Merrall and Evans, 2020), but finding negative effects when accounting for roads suggests that other sources of urban noise pollution are also detrimental. Anthropogenic noise can affect detectability by surveyors (Cooke et al., 2020). Such effects are likely to be limited in our study, however, as bird surveys primarily occurred early in the morning when anthropogenic noise is limited. Most pedestrians using walking routes do so when anthropogenic activity and noise is high. Consequently, in noisy environments pedestrians are likely to experience a 'double jeopardy' with noise reducing the numbers of birds present, and reducing their ability to experience the songs of those present. This will thus limit their ability to gain enjoyment and other benefits that experiments indicate arise from experiencing avian vocalisations (Douglas and Evans, 2022).

Measures of the characteristics of greenspace in the area surrounding our focal walking routes were often not significantly associated with biodiversity along the walking routes. Foliage height diversity (measured at 1.5 m resolution in this study) is a measure of vegetation structural complexity, which is often associated with greater biodiversity (Beninde et al., 2015, Grafius et al., 2019), but was not significantly related to biodiversity of birds or butterflies in this study. The lack of a relationship with birds is surprising given that urban birds typically exhibit positive responses to increased woodland cover (Plummer et al., 2020). FHD has been the traditional measure of structural diversity (MacArthur and MacArthur, 1961), which has been shown to correlate with biodiversity in some biomes (Goetz et al., 2007), but some recent work suggests that it is not universally related to biodiversity at all scales and biomes (Ehbrecht et al., 2016). Therefore, while this study shows that FHD is not a strong model predictor, a number of studies published since the mapping of these sites was completed have proposed new metrics of structural complexity which may be stronger predictors of biodiversity (Ehbrecht et al., 2016, Liu et al., 2022). Patch size also typically correlates positively with biodiversity (Beninde et al., 2015, Grafius et al., 2019), but was not significant in any model in this study. Our results suggest that, for our focal taxa, biodiversity along the local area of the walking routes is not beneficially impacted by spill-over effects from nearby patches of suitable habitat in the surrounding landscape. Rather, the environmental conditions along the walking routes themselves may exert greater influence on the biodiversity that can be experienced on the routes. Other factors probably also contribute, for example urban biotas are dominated by generalist species (Evans et al., 2011) and animals with high dispersal ability (Jung and Threlfall, 2018, Martinson and Raupp, 2013). These are likely to be able to use the fragmented network of urban greenspace and associated matrix effectively, and thus reduce the influence of patch size and vegetation structural complexity in the surrounding area on biodiversity along the walking routes.

Walking routes in areas with more residential garden cover had lower butterfly richness and abundance, and increased garden cover was never positively associated with species richness and abundance. These results may appear surprising given that gardens can provide important resources for wildlife. Gardens, however, are not always managed to maximise wildlife benefits (Gaston et al., 2007), for example gardeners may intensively manage vegetation, keeping grasses short, and remove weeds that can provide crucial food or larval host plants for butterflies (Goddard et al., 2013, Öckinger et al., 2009), or replace areas of vegetation with impervious surfaces or other artificial structures (Perry and Nawaz, 2008). While gardens are beneficial for some taxa (Baldock et al., 2019), this is not the case for all taxa (Mata et al., 2017) and the benefits depend on garden features (Olivier et al., 2016). In the back gardens observed from the transects, lawn was a dominant feature, in line with UK trends (54% of gardens contain mown lawns, which occupy approximately 43% of garden area (Loram et al., 2008)). Paved areas are present in 86% of UK gardens (Loram et al., 2008), but, as observed on the transects, are a particularly prominent and growing feature of front gardens, consistent with wider UK trends (Perry and Nawaz, 2008). Both mown lawn and hard surfacing have limited value for butterflies. Our results suggest that whatever the locally positive effects of domestic gardens (Goddard et al., 2010), these seem to have limited spill over into the wider public landscape, as sampled by our walking routes. This may include limited spill over of biodiversity from back gardens to public-facing front gardens, parts of which were captured in our sampling.

The frequently observed luxury effect – species richness being higher in more affluent areas – is often attributed to such areas having more, and higher quality, greenspace (Chamberlain et al., 2020). Our analyses, which account for the amount of greenspace, do not find evidence for a luxury effect when considering bird species richness. Total, but not native, tree species richness was higher (marginally significantly) in less deprived areas – perhaps reflecting more planting of ornamental trees in affluent areas. Luxury effects have not previously been examined in butterflies (Leong et al., 2018) but, here, we find a suggestion of the opposite pattern – greater diversity (marginally significantly) in more deprived areas. This may be due to greater availability of butterflies' larval host plants in ruderal vegetation in either unmanaged, or derelict, land, which is likely to be commoner in deprived areas. These findings suggest the potential for positive biodiversity experiences being potentially available in all socioeconomic settings, but that the nature of these might differ, although more work is required to confirm these trends.

6.2 Environmental conditions

Environmental conditions that can negatively impact people's health (warmer temperatures in the summer, more noise and greater concentrations of particulate matter) occurred along walking routes embedded in locations with more impervious surface cover. Greater coverage of industrial areas and higher densities of large roads further increased noise levels, with road density also further increasing concentrations of large particulate matter. Small particulate matter did not show a significant response to built cover metrics, possibly because its concentration, even adjacent to pollution sources, is very variable due to the particles' capacity to disperse (Nerriere et al., 2005). Many of these built cover metrics are also negatively associated with biodiversity (see above), which can deliver substantial wellbeing benefits (Marselle et al., 2020), meaning that the adverse impacts of the urban features reflected in the metrics could both directly and indirectly reduce the benefits for pedestrians using walking routes embedded in such locations.

Greenspace characteristics had few detectable associations with environmental conditions, but, where present, these seem to be mitigating effects. Firstly, areas with less vegetation were hotter, illustrating the buffering effect of greenspace against the urban heat island at the landscape scale (Coutts and Harris, 2013). Secondly, more complex vegetation structure affected exposure to particulate matter. Specifically, increased foliage height diversity was associated with lower mean concentrations of small particulates, which is consistent with previous work suggesting that vegetation can be effective at trapping such particles (Abhijith et al., 2017, Janhäll, 2015, Brantley et al., 2014). A more complex result was the effect on large particle density of the interaction between large roads and foliage height diversity. Areas with more large roads and greater foliage height diversity had higher levels of large particulates. This is consistent with observations that dense tree canopies can trap pollutants within the street canyon, as a result of lower wind speeds (Wania et al., 2012) and complex wind flow patterns (Janhäll, 2015), increasing their local concentrations (Abhijith et al., 2017), and potentially outweighing effects of particle removal (Vos et al., 2013). Appropriate choices of species and planting design are therefore essential to ensure pollution reduction using vegetation (Chen et al., 2016).

6.3 Limitations and implications

Our conclusions relate to measured biodiversity in three example (but important) taxa, and to the particular conditions prevalent in the spring and summer. The first of these factors means that our assessment is of the potential experience of biodiversity along the walking routes. People's perception of biodiversity will vary with their existing knowledge of, and interest in, the natural world (Southon et al., 2018). The second means that our findings are primarily relevant to periods when vegetation is fully developed and biological activity is high. Environmental conditions, biodiversity experience and use of walking routes by pedestrians will be substantially different at other times, particularly in winter. Nonetheless this study both demonstrates an approach for developing an integrated assessment of factors determining the walkability of urban walking routes, and provides a clear analysis of their effects over the period of the year when pedestrian activity is likely to be at its maximum.

The purpose of this sampling strategy was to capture conditions across a wide range of urban forms and a variety of environmental conditions and biodiversity. The trade-off of this holistic approach is that the variables are not assessed in depth, in particular the collection of data at each site from one visit limits how well the sites were characterised. There is a trade-off between spatial and temporal replication and in this study we focused on spatial replication, and used statistical controls such as survey date and time in the models to account partially for short-term temporal variation.

The overarching finding from this study is that areas with lower amounts of grey space at the landscape scale are associated with more biodiversity and more favourable conditions for people. The shared drivers of variation in conditions for biodiversity and people can provide insight into designing or retrofitting urban areas at landscape scales to benefit people and biodiversity. A correlation matrix showing covariation between the biodiversity and environmental condition measures (Table S7) suggests the biodiversity measures are typically positively correlated with each other, albeit sometimes weakly. In contrast, there was less correlation amongst the environmental response variables, with only small and large particulate matter levels being significantly correlated. This suggests that while improving conditions for one type of biodiversity is likely to have positive effects on other components of the community, modifying the environment to reduce the effects of one environmental condition may not achieve change in other environmental conditions. Consequently, and despite the overarching negative impact of grey-space on numerous metrics, multiple mitigating actions are likely to be required to simultaneously enhance environmental conditions that pedestrians are exposed to along walking routes and enhance the biodiversity that is available for them to experience. The lack of positive spill-over effects from green-space features embedded in the wider landscape, suggests that such mitigating actions will need to include improving habitat features for biodiversity at very fine spatial scales in close proximity to walking routes.

6.4 Conclusions

Users of urban walking routes can experience substantial biodiversity that has the potential to enhance their environmental experience. This biodiversity, however, varies considerably depending on the intensity of urban development in the surrounding area and environmental conditions along the walking routes. There is limited evidence for beneficial spill-over effects on biodiversity along walking routes of features in the wider landscape typically associated with high quality habitats for biodiversity. We provide evidence for a double jeopardy for pedestrians arising from low quality urban environments (increased impervious surface cover, high density of large roads and industrial areas), as these directly increase exposure to factors detrimental to human health and have additional indirect impacts by reducing exposure to biodiversity that can promote human well-being.

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