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

Eleanor Rankin

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Abstract

Changes to the behaviours and priorities of the human population have led to an unprecedented number of urban inhabitants worldwide. Urban dwellers are exposed to high levels of air and noise pollution that negatively impact both human well-being and species diversity. However, research has shown that biodiversity present within urban areas can positively contribute toward human well-being. Despite this, no literature to date has examined the potential for urban biodiversity to mitigate or lessen the negative associations between air and noise pollution and mental well-being. This study used parallel mediation models to examine the complex interplay between these factors, testing whether species richness (actual and perceived) can mediate the relationship between air ($PM_{2.5}$) and noise (dB) pollution and well-being (mental well-being and happiness) across 30 streetscapes in the city of Leeds, UK. The results revealed that greater actual flowering plant richness reduced the negative impact of noise pollution on resident's self-reported well-being across streetscapes. Additionally, there was a direct negative association between noise pollution and flowering plant and pollinator richness. This could be because residents are deterred from gardening in streetscapes where noise pollution is high. However, noise pollution can also have a direct negative impact on pollinator species. No direct or mediated negative associations between air pollution and well-being measures were identified in this study. This thesis provides a valuable insight into the complex interplay between streetscape pollution, biodiversity and human mental well-being, which until now remained largely unexamined. Increasing plant diversity should be considered a priority within any future plans to reduce urban noise. In turn, critical gains in pollinator conservation can also be made. Trends in population growth, increased pollution and declines in mental health that characterise cities worldwide, mean that the findings presented here are applicable to a multitude of urban settings.

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Contents

Chapter 1. Introduction

1.1 The rise of urbanisation

The planet is undergoing the most radical and fast-paced transformation in humanity's history. Changes to the behaviours and priorities of the human population have led to an unprecedented number of urban inhabitants worldwide. Urban areas account for ~3% of global landcover yet harbour 56% of the human population (Liu et al., 2014). This latter percentage is projected to reach 68% by 2050, which would increase the number of urban dwellers by 2.5 billion (United Nations, 2018). Despite covering a relatively small proportion of Earth's surface, urban land is expanding more rapidly than any other land-use type (Marzluff and Ewing, 2001). The degree of urbanisation within the developed world, particularly across temperate regions, is already substantial (~80% throughout most of North America and Europe) (United Nations, 2018).

Urban settings can provide considerable benefit to society, such as improved healthcare, education, social equality, and increased housing and economic growth (United Nations 2018). However, where there is a lack of thorough and strategic planning, urban expansion can result in overcrowding, increased pollution and poor sanitation (United Nations, 2016). These characteristics also increase an area's susceptibility to severe damage following stochastic events, particularly those associated with climate change (e.g., extreme weather, epidemics). Furthermore, the World Health Organisation has described urbanisation as one of the foremost health challenges facing the 21st century (WHO, 2015), as cities continue to act as epicentres for chronic, non-communicable diseases (e.g., obesity, stress, poor mental health, declines in physical activity) and mental health conditions (e.g., depression, anxiety) (Dye, 2008; Peen et al., 2010).

1.2 Implications of urbanisation for biodiversity

Biodiversity is rapidly declining on a global scale. Since 1900, the average abundance of native species across most terrestrial habitats has fallen by at least one-fifth (IPBES, 2019). One million species of fauna and flora are now threatened by extinction, many of which have a trajectory of just a few decades (IPBES, 2019). Observations suggest that we are now undergoing the sixth mass extinction, which is, for the first time in history, propelled by anthropogenic behaviours (Barnosky et al., 2011). The threats to biodiversity are multi-faceted and often inextricably linked, typically involving one or more of the following: direct exploitation (Rosser and Mainka, 2002), climate change (Walther et al., 2002), invasive non-native species (Sala et al., 2000), land conversion/fragmentation (Brooks et al., 2002) and pollution (Hernández et al., 2016).

Urban expansion has profound implications for the natural world. The rapid development of urban land, particularly since the 1950s, has coincided with significant change to the world's ecosystems leading to environmental degradation, habitat loss and the depletion of natural resources (Liu et al., 2014; McNeill, 2000). Modelled future scenarios suggest that global urban growth could threaten an additional 290,000 km^2 of natural habitat by 2030 (McDonald et al. 2018). Urbanisation is now considered a leading cause of species extinction (Czech et al., 2000). Biological communities are unable to persist in urban landscapes where there is a lack of suitable habitat and where urban infrastructure, such as roads and buildings, isolate remaining patches of habitat (Beninde et al., 2015). Furthermore, the ecological footprint of urban settlements far exceeds the physical perimeters of towns and cities, as the demand for energy and agricultural production required to sustain them increase in tandem (Newman, 2006). Indeed, species which urbanisation pose the most threat of extinction to tend also to be highly threatened by agricultural activity (Czech et al., 2000). Moreover, due to the nature of urbanisation, the impacts are long lasting and intensify over time, rarely allowing opportunities for ecological succession (Marzluff and Ewing, 2001).

1.2.1 Urban greenspace

Urban greenspace can be defined as an area comprised of vegetation, which can encompass planned and managed natural and semi-natural landscapes (e.g., domestic gardens, urban parks/forests, street verges, allotments), in addition to unmanaged space (e.g., brownfields found in cities) (European Environment Agency, 2014; Taylor and Hochuli, 2017). Urban greenspaces can support endemic, threatened and specialist species and thus are deserving of conservation attention (Aronson et al., 2014; Ives et al., 2016; Soannes and Lentini, 2019). This counters evidence that biodiversity losses are inevitable along rural to urban gradients. For example, Buchholz et al., (2016) recorded a total of 600 species in the Berlin Weißensee Jewish cemetery, which included 25 plant, five bat and nine bird species of conservation concern, as well as one lichen species rarely found across the region. However, research has shown that size and composition of natural features remain fundamental to the viability of urban greenspace habitat (Beninde et al., 2015; Braaker et al., 2017; Lowry et al., 2013). Small patches of urban greenspaces (e.g., domestic gardens) can also facilitate the movement of species between larger public greenspace (e.g., urban forests) and the wider countryside, thereby improving habitat connectivity (Fernández-Juricic and Jokimäki, 2001; Rudd et al., 2002). Furthermore, the accumulation of residential gardens (typically heterogeneous in nature) across a single geographical area can be of more benefit to species that require larger swathes of habitat to fulfil niche requirements, than individual garden plots (Cannon et al., 2005; Knight et al., 2005). The contribution of domestic gardens to overall urban greenspace cover makes their existence and management critical for the preservation of many urban species (Baldock et al., 2019; Davies et al., 2009). For example, in the UK, an estimated 28.7

million trees, 4.7 million nest boxes and 3.5 million ponds are supported by domestic gardens (Davies et al., 2009).

1.2.2 Biotic homogenisation

Urbanisation not only extirpates many native species, but also results in the homogenisation of biotic communities, typically favoring fewer generalist non-native species (McKinney, 2006). Such patterns ensue when non-urban specialist species, reliant on narrow ranges of habitat and which generally occur in semi-natural environments, are replaced by species capable of adapting and exploiting the habitats which urban settings support. For instance, patterns of biotic homogenisation in European avifauna were found to correlate with increased urbanisation, leading to a decrease in species richness and the overall dominance of fewer species within city-dwelling avian communities (Clergeau et al., 2006). Urban pollinator assemblages were more homogenous compared to those observed in agricultural land and nature reserves in the UK (Baldock et al., 2015). In New York City, USA, 578 (approximately 43%) native plant species have been lost, while the city has gained 411 non-native species (DeCandido et al., 2004). While the shift from native to non-native species at local scales might appear to enrich some urban ecosystems, the overall implication for conservation is concerning as the native taxa are reduced and sometimes lost from the global species pool (Sax and Gaines, 2003).

1.2.3 Impacts of pollution on biodiversity

At fine spatial scales increased temperatures, artificial light, radiation and the accumulation of pollutants reduce the favorability of urban habitats for biodiversity (Grimm et al., 2008; Iserhard et al., 2019; New, 2015). For example, vehicle exhaust emissions (CO, NO, NO_2) disrupt the volatile recognition process in honeybees, thereby reducing their ability to recognise and retain memory of floral food resources (Leonard et al., 2019). Nitrogen oxides, typical roadside pollutants, accelerate senescence in herbaceous plants, in addition to other phenological changes such as delayed flower development (Honour et al., 2009). They have also been shown to disturb height growth in conifers and lead to the defoliation of trees (Ots and Rauk, 2001; Ots et al., 2009;).

When a sound exceeds the sensitivity threshold of a species it becomes “noise” (referred to as “noise pollution”) and can lead to disruptions in a species’ behaviours and abilities (Sordello et al., 2020). The primary source of noise pollution is, in many cases, a byproduct of transportation (e.g., cars, motorbikes, airplanes), although other anthropogenic sources, such as windfarms, fireworks and large group gatherings (e.g., festivals, sporting events) are also known to impact biodiversity (Rast et al., 2019; Shamoun-Baranes et al., 2011; Zwart et al., 2016). For example, noise pollution reduced sexual signaling in insects (Lampe et al., 2012) and anurans (Sun and Narins, 2005). In avian species, nest construction, territory defence and predator defence were impaired due to the energy trade-off between those behaviours and increased vigilance due to noise pollution (Gil and Brumm, 2013; Quinn et al., 2006; Zwart et al., 2016). The same trade-off has been observed in bats, leading to reduced foraging success (Luo et al., 2015). Direct noise pollution can also cause elevated stress, leading to physiological changes to birds such as decreased production of eggs and stunted chick growth (Kleist et al., 2018).

Urban heat islands occur when the temperature of a given built-up area is noticeably greater than that of the countryside directly surrounding it (Nakayama and Fujita, 2010; Santamouris, 2015). This phenomenon is caused by reduced vegetation and evotranspiration, increased volume of dark surfaces with low albedo that absorb and reradiate solar radiation, and the overall increase in human heat/energy output (Stone et al., 2010). Current studies suggest that heat island effects can increase temperatures by between 5 and 15 degrees Celsius (Santamouris, 2013). Urban heat islands can alter species richness and abundance, as well as cause changes to community composition (Yuan and Bauer, 2007). For example, many bee species are particularly intolerant to warming, as demonstrated by contractions northward in response to climatic warming (Kerr et al., 2015). Indeed, Hamblin et al. (2018) found that bee abundance declined by 41% for every degree Celsius increase in average site temperature. Urban trees display reduced photosynthesis and subsequent growth owing to urban stressors such as insufficient water. In the context of urban areas, water stress is an interactive factor which is exacerbated by the effects of urban heating (Meineke and Frank, 2018).

1.2.4 Pollinators

Pollinators are vital for many of the key ecosystem services (see sub-section 1.3 for definition) responsible for sustaining human health and well-being globally, including ecosystem regulation (e.g. hydrology), the reproduction of wild plants, crop production and food security (Klein et al., 2007; Potts et al., 2016). Subsequently, they bare substantial economic weight, worth ~€153 billion per annum (2005) (Gallai et al., 2009). Despite their importance, declines in the abundance of all key insect pollinator groups have been reported, including honeybees, bumblebees, solitary bees and hoverflies (Baldock et al., 2019). Habitat loss and fragmentation are key drivers of pollinator declines (Potts et al., 2010; Banaszak-Cibicka, 2012). Despite the general trend of pollinator declines alongside increased urbanisation (e.g., Bates et al., 2011; Hernandez et al., 2009), research has also demonstrated the ability of urban greenspaces to support remarkably high levels of pollinator richness and abundance. For example, 50% of the German bee fauna was recorded within the highly urbanised city of Berlin (Owen, 2010). Some studies have demonstrated positive effects of urbanisation on the richness of particular bee taxa, including bumblebees and cavity-nesting bees (Cane et al., 2006; Carre et al., 2009). Bumblebee (*Bombus terrestris*) nests across the gardens of suburban homes in the UK (~36 nests ha^{-1}) were comparable to the quantity recorded in rural habitats such as hedgerows (20 – 37 ha^{-1}) (Osborne et al., 2008). The enhancement of agricultural and peri-urban landscapes for pollinators has been a favourable conservation strategy in recent decades (Goddard, et al., 2010). However, due to the rapid expansion of urban areas, the improvement of urban greenspace for pollinators is also becoming increasingly recognised as an important component of conservation and restoration efforts (Goddard et al., 2010).

1.3 Urban greenspace and ecosystem services/disservices

Ecosystem services are the benefits that people derive, directly and indirectly, from ecological systems (Millennium Assessment, 2005). Within urbanised areas, urban greenspaces support many ecosystem services pivotal to society (Pauleit et al., 2010; Gómez-Baggethun and Barton, 2013). For example, urban greenspaces help to mitigate against urban heat island effects by regulating temperatures (Bowler et al., 2010). Vegetation, particularly trees, can provide shade during the summer, reflect solar radiation and absorb heat through evapotranspiration (Nowak et al., 2008). Indeed, the role of alleviating the heat load of urban areas is one of the most beneficial regulatory ecosystem services provided to cities by trees (McPhearson, 2011). Increased impervious surface cover (e.g., concrete, tarmac), which is characteristic of urban areas, exacerbates water runoff following precipitation and increases the risk of flooding. Vegetation limits this runoff by acting as a sponge and storing water in pore spaces (Pataki et al., 2011). The interception of precipitation by canopies further reduces flooding by slowing down rainfall and alleviating pressures placed on urban drainage systems (Pataki et al., 2011). Vegetation also improves surrounding air quality via the deposition of pollutants through photosynthesis (e.g., sulfur dioxide (SO_2), nitrogen dioxide (CO_2), carbon monoxide (CO) and particulate matter ($PM_{2.5}$) (Beckett et al., 2000; Lovett, 1994;). It also acts as a barrier to noise through diffraction and absorption, which lessens the burden of acoustic disturbance caused by roads (van Renterghem et al., 2012) (see sub-section 1.9).

The crucial role in which biodiversity plays in supporting provision of ecosystem services is well established within the literature (Mace et al., 2012; Norris, 2012). For example, large proportions of soil nutrient cycles are determined by the biological composition within the soil, and resilience to pests and environmental change is stronger in biological communities where diversity is greater (Hector et al., 2000; Cardinale et al., 2003). Indeed, the quality of ecosystem functioning has been found to be associated with increased species richness in several studies (Worm and Duffy, 2003; Balvanera et al., 2006). As such, biodiversity plays an important role in the delivery of ecosystem services.

On the other hand, there are also several ecosystem disservices associated with urban greenspaces. For instance, densely vegetated urban greenspaces can provide cover for criminal activity (Kondo et al., 2017), or lead to residents feeling unsafe walking at night (Tandogan and Ilhan, 2016). Urban greenspaces can support species which are vectors for disease, or which might be considered a nuisance (e.g., flies, ticks, mosquitos) (Dunn, 2010). Urban flora (e.g., some trees, grasses, flowering plants) can cause and aggravate allergies such as hayfever. One study conducted in New York identified significant associations between the extent of tree canopy and asthma and allergic sensitisation in African American and Dominican children (Lovasi et al., 2013). Furthermore, urban greenspaces are not equitably distributed, with their availability often disproportionately stratified by socioeconomic background (e.g., income, ethnicity) (Wolch et al., 2014). Housing price premiums associated with homes close to greenspaces can therefore lead to the gentrification and the displacement of low-income earners (Wolch et al. 2014).

1.4 Human well-being

Well-being is a multidimensional concept made difficult to define by the varied and complex determinants that underpin it, including genetic pre-dispositions, environmental factors, life conditions and opportunities, lifestyle and patterns of thoughts (Stiglitz et al. 2010). Indeed, there is a growing recognition that there is more to mental health and well-being than simply the absence of psychological disorders such as depression and anxiety, for example, the promotion of positive subjective well-being (i.e., how people view and feel about their own lives) (Diener et al., 2018). The assessment of people's well-being can be split up into objective and subjective measures.

1.4.1 Objective well-being

Objective well-being indicators are calibrated against an index of what is considered necessary for a quality standard of living, including variables such as material resources (e.g., income, housing) and social attributes (e.g., education, political voice, social networks) (Diener and Suh, 1997).

1.4.2 Subjective well-being

Subjective well-being measures require an individual to evaluate their own lives. Results are subjective not simply because they are self-reported, but because the individual is asked to rate how they feel, typically done using Likert-style scales (Hicks et al., 2013). Questions can be evaluative (e.g., life satisfaction), experiential (e.g., positive and negative affect or emotion) or eudaimonic (e.g., a sense of purpose and meaning in life) (Dolan and Metcalfe, 2012). It has been demonstrated that subjective well-being is comparable transnationally, regardless of societal variation (Dolan and Metcalf 2012). Indeed, recognition of the importance of standardised and comparable subjective well-being measures has increased (Samman, 2007). More recently, the United Nations Development Programme (UNDP) sought to utilise subjective assessments of people's own lives to identify indices of inequality that are in harmony with objective measures collected on national scales. Doing so recognises that inequalities are often a symptom of an unfair system rather than a cause (United Nations Development Programme, 2019). Increased subjective well-being is positively correlated with longevity (Diener and Chan, 2011).

1.5 Nature and human well-being

Research suggests that humans now spend less time outdoors than their predecessors, leading to a certain degree of "disconnection" from nature (Miller, 2005; Wilson, 1984;). This trend has been contextualised as an adaptation to contemporary urban lifestyles, such as office working and increased usage of digital media platforms (Kellert, 2018; Larson et al., 2019). For example, screen time has now reached nine hours per day in youth ages 13 to 18 across the USA (Rideout, 2015), a pattern reflected across the globe (Taylor and Silver, 2019). The "extinction of experience" phenomenon is a term used to describe this growing disconnect between humans and nature and posits that people are less exposed to nature due to a lack of time spent in greenspaces and, indeed, a lack of greenspaces (Pyle, 1993; Soga and Gaston, 2016). Examples of this widening gap are well documented. For instance, in the UK, youths were more competent at identifying cartoon characters than locally found biodiversity (Balmford, 2002).

The extinction of experience phenomenon is concerning for two overarching reasons. Firstly, it is not conducive to biodiversity conservation as "collective ignorance ultimately leads to collective indifference" (Prévot et al., 2018; Pyle, 2002), which is considered one of the fundamental obstacles to reversing environmental degradation (Miller, 2005; Prévot et al., 2018). Increased detachment of people from nature leads to a cycle of reduced support and interest in conservation initiatives, resulting in greater losses to biodiversity and a further worsening of the extinction of experience phenomena (Soga and Gaston, 2016). Secondly, lack of nature contact diminishes associated positive health and well-being outcomes. For example, time spent in greenspace has shown to reduce blood pressure (Hartig et al. 2003), lower mortality rates from cardio-vascular disease (Mitchell and Popham, 2008) and improve self-perceived general health (Kardan et al., 2015). Improvements to mental health following exposure to greenspace is also a widely studied outcome within the literature (Barton and Rogerson, 2017; Gascon et al., 2015; Martin et al., 2020). Even relatively short-term exposure to parks, urban forests, domestic gardens and other semi-natural environments can reduce symptoms brought on by stress and depression, restore attentional fatigue and improve self-esteem and perceived mental state (Barton and Pretty, 2010; Marselle et al., 2014, 2016; van den Bosch and Ode Sang, 2017).

1.6 Theories and concepts describing nature and human well-being relationships

1.6.1 Psychological restoration

One of the first attempts made to understand the relationship between well-being and nature was through the biophilia (or "love of life or living systems") hypothesis. This theory postulates the relationship between humans and nature as being innate by virtue of humans' coevolution alongside the natural environment (Kellert and Wilson, 1993), described as "the connections that human beings subconsciously seek with the rest of life". Critics of biophilia contend that the theory is biologically determinist (Bone, 2009) and thus neglects to account for influential environmental factors (Newton, 2007), for example those outlined in behaviorism and its encompassing theories of conditioning (Bandura, 1965). Determinism seeks to precede and limit cultural and social effects on the psyche, which research has shown to be greatly influential (Van der Veer, 1996). Therefore, it can be argued that many aspects of biophilia, and indeed the converse of biophobia, are in part learned ways of thinking (Bandura, 1965). For instance, a fear of spiders may be passed down to children as they learn to imitate observed behaviours and subsequently acquire the fear of spiders for themselves. The notion of learned behaviours extends into a cultural argument. For example, Noe and Snow (1990) observed significant differences in sensitivity levels for environmental concerns and natural environments between Hispanic and Non-Hispanic populations in the USA. Thus, a cultural effect was determined responsible. Moreover, some researchers argue that there is limited empirical evidence to support the biophilia theory (Joye and De Block, 2011; Joye and Van den Berg, 2011). However, amidst the contention, widespread interest in the evolutionary origin of nature-human well-being relationships has continued. More specific hypotheses grounded in the works of biophilia were proposed in the 1980s, including Attention Restoration Theory (ART, Kaplan and Kaplan, 1989) and Stress Reduction Theory (SRT, Ulrich, 1983). They are complimentary psycho-evolutionary theories that infer that nature provides more restorative qualities than urban/artificial environments. However, they differ fundamentally in terms of what drives one to a restorative state.

SRT posits that humans have a positive pre-cognitive response to natural scenes comparable to the fight-or-flight response found in non-human primates (Ulrich, 1983; Ulrich et al., 1991). Landscapes that contain vegetation, water and that are of modest depth or complexity are evolutionarily associated with general feelings of safety (from predators) and availability of resources, thus reducing the arousal of negative thought. As such, it is theorised that humans have not had sufficient time to evolve analogous responses to urban settings and are therefore hindered in their capacity to recover from stressful experiences. Indeed, evidence suggests that, on average, city dwelling residents suffer from greater levels of psychological stress compared to those who reside in rural areas (Dhingra et al., 2009; Lambert et al., 2015; Verheij et al., 2008).

On the other hand, ART refers more specifically to the struggles of recovering from mental fatigue in urban environments (Kaplan and Kaplan, 1989; Kaplan, 1995, 2001). For humans to focus on a particular task at hand (referred to as "directed attention"), the brain is required to exclude surrounding stimuli (e.g., noise, visual cues) and other cognitive activity which may manifest into conscious thought (e.g., remembering to purchase food from the supermarket). The mechanism responsible for this inhibitory response fatigues with use over a relatively short period of time, making it increasingly difficult to concentrate. This theory therefore hypothesises that being in view of, or immersed in natural environments, provides an opportunity to rest and restore cognitive function that allows for the re-focus of attention (referred to as "fascination" or "effortless involuntary attention"). By extension, attentional fatigue is considered a condition that elevates an individual's predisposition to stress. There is now a substantial body of SRT and ART literature demonstrating that natural environments have significantly greater restorative ability than entirely built-up settings (e.g., Chawla et al., 2014; Mennis et al., 2018).

1.6.2 Social cohesion

As highly sociable species, it is argued that humans possess an innate need to interact with conspecifics and to feel a sense of being part of a particular social group (Baumeister and Leary, 1995). An individual can thus gain psychological benefits from social support, including an improved sense of meaning, belonging, self-esteem and general companionship, all of which have been correlated positively with increased well-being and negatively with mental health disorders (Matthews et al., 2016; Nieminen et al., 2010; Shankar et al., 2015; Thoits, 2011). Indeed, both actual (e.g., social networks, social participation) and subjective social connectedness have shown to play an important role in facilitating positive mental health and well-being outcomes (Cornwell and Waite, 2009; Yu et al., 2015). As such, there is an increasing breadth of research which suggests that social connectedness (or "social cohesion") may mediate the relationship between well-being and nature by providing a setting through which to connect (Cartwright et al., 2018; de Vries et al., 2013; Maas et al., 2009).

Cartwright et al. (2018) showed that greater nature exposure and social connectedness over seven days both were associated with increased subjective well-being over the same time period. Interestingly, nearby nature, although not visitation frequency, acted as a moderator between social connectedness and subjective well-being. However, individuals with low social connectedness still presented with elevated levels of well-being and a decreased likelihood of depression if nature was reported close-by. It is therefore possible that, due to the similar processes and mechanisms which underlie the outcomes of both nature contact and social interaction (e.g., stress reduction, Ozbay et al., 2007; attention restoration, Baumeister et al., 2005; rumination, Nolen-Hoeksema et al., 1994; self-esteem, Leary, 2005); sense of belonging, Walton et al., 2012; mood and affect, Krach et al., 2010), that nature contact can buffer against negative well-being outcomes associated with low social connectedness.

1.6.3 Promotion of physical activity

Physical inactivity is one of four leading risk factors for global mortality (WHO, 2010). Environmental factors, such as high traffic volumes, lack of greenspace and footpaths, are known to exacerbate physical inactivity (An et al., 2018; Coombs et al., 2010). Increased and regular physical activity has been shown to improve neurocognitive development and mental well-being, in addition to other physical outcomes including improved cardiovascular health, reduced obesity, cancer and osteoporosis, the occurrence of which can also negatively impact on an individual's mental well-being (Owen et al., 2010). Numerous publications from a variety of countries have linked improved physical activity, recreational walking and deductions in sedentary time to the access and use of urban greenspaces in children, adults and geriatrics (e.g., Coombes et al., 2012; Ellaway et al., 2005; Lachowycz and Jones, 2013; Schipperijn et al., 2013), although this is not consistent throughout the literature (King et al., 2005).

The quality of greenspace may also impact people's levels of physical activity and therefore well-being outcomes. For example, attractiveness of greenspace related positively to walking behaviours for both exercise and leisure (De Vries et al. 2013). De Vries et al. (2013) also found that while total physical activity was not a mediator between well-being and greenspace, physical activities which could specifically take place within greenspace were (although such relationships were not as strong as with stress and social cohesion). This may be due to the concept of 'green exercise', which can be defined as a physical activity undertaken in natural/semi-natural environments (Barton and Pretty, 2010). Empirical evidence suggests that green exercise (e.g., running in the park) is more beneficial to psychological restorative processes than exercise carried out in other settings (e.g., running in urban areas or inside a gym) (Bodin and Hartig, 2003; Marselle et al., 2014).

1.7 Public health and conservation policy

Since 2010, approximately US$ 2.5 trillion has been lost per annum from the global economy due to the two most common mental health disorders, depression and anxiety (Bloom et al., 2012). This is projected to reach US$ 6 trillion by 2030 (Bloom et al., 2012). The opportunity to utilise urban greenspaces to address rising mental ill-health as an 'upstream' preventative method, considered more efficient and economical than treating issues 'downstream', is now increasingly recognised by policy makers. The implementation and management of urban greenspaces for this purpose may simultaneously help to achieve conservation goals and provide a strong argument for conservation itself (Dean et al., 2011). For example, in the UK, it was said that: "*safe, green spaces may be as effective as prescription drugs in treating some forms of mental illness*" (Faculty of Public Health 2010; p. 2). A total of 34 English conservation NGO's have since actively participated in lobbying for 1% of health spending to be directed toward nature-based solutions (Response for Nature Partnership, 2015). However, policy makers and urban planners cannot effectively implement urban greenspaces for the promotion of mental well-being without a clear understanding of what natural elements/characteristics are responsible for such outcomes (Marselle et al. 2015; Wheeler et al. 2015; Wood et al. 2018). For example, Southon et al. (2017) identified that people had a strong preference for urban meadows rather than manicured plant beds and herbaceous borders.

1.8 Biodiversity and well-being

Much of the existing nature-human well-being research has focused on how factors such as presence, size, accessibility and proximity of greenspace promote well-being (Bokalis et al., 2018). However, this viewpoint neglects to account for the overall ecological 'quality' of the greenspace available, accounting for factors such as diversity of landcover and species richness. Indeed, there is an accumulating body of research that demonstrates the positive mental well-being outcomes associated with increased species richness, some of which illustrate the superior effects of richness over just abundance. For example, Methorst et al., (2021) identified a significant positive relationship between bird and plant species and mental health. However, no such relationship was observed when accounting for abundance. Likewise, Fuller et al., (2007) found that increased measures of psychological well-being strongly correlated with higher plant and bird richness across urban greenspaces. While the researchers identified a positive correlation between well-being and the size of the greenspace, it was not as strong as with plant or bird species richness. In Italy, higher measures of biodiversity across urban and peri-urban habitats lead to increased perceived restorativeness and subjective well-being (Carrus et al., 2015). Wood et al. (2018) found that biodiversity enhanced the restorative benefit of urban parks and with minor influence from other variables. The notion that quality may be of equal, if not of greater, importance than the presence, size, accessibility and proximity of greenspace means that a shift in focus is vital for informing both effective public health and conservation policy (Taylor and Hochuli, 2015; Wood et al., 2018).

Despite the increase in literature demonstrating the positive benefits of increased biodiversity on mental well-being, the evidence is equivocal. Trends are inconsistent, which could be caused by methodological discrepancies in both well-being and biodiversity metrics (Markevych et al., 2017). Furthermore, there has been an observed disparity between perceived levels of biodiversity and those which objectively exist, determined by the often-limited biodiversity knowledge skills of the general public (Dallimer et al., 2012). Dallimer et al. (2012) found that while there was a lack of consistent relationship between actual species richness and well-being benefit, a correlation between perceived species richness and well-being was observed.

It may also be necessary to consider the role of people's own biodiversity preferences when accounting for observed discrepancies in reported subjective well-being, as suggested by Pett et al. (2016). People are a product of their lives, shaped by individual and unique experiences, concepts and spiritual/cultural beliefs (Proshansky et al., 1983). Components of each can influence a person's momentary emotional state when immersed in greenspace (Gopal et al., 2019). For example, people have been found to be happier in environments which they perceive to be scenic, regardless of whether those locations are urban instead of natural (Seresinhe et al., 2019). As a result, there may discord between factors that are beneficial to people's well-being and characteristics that conservations seek to change.

1.9 Pollution and people

Air pollution is the largest environmental cause of non-communicable disease (WHO, 2016). Annually, an estimated 10.2 million premature deaths are attributed to air pollution; a figure three times that caused by AIDs, malaria and tuberculosis combined (Vohra et al., 2021). Air pollution is of particular concern for urban dwellers as approximately 75% of globally produced CO_2 emissions originate in urban areas (Seto et al., 2012). Moreover, urban inhabitants are exposed to increased levels of noise pollution which, as stated by the WHO, is the third most hazardous environmental pollutant found in cities (WHO, 2016). Several studies have highlighted the potential health and well-being benefits gained by those residing in quieter areas, concluding that the former possess a better quality of life compared to those living in noisier areas (Shepherd et al., 2013).

From a physiological perspective, systemic and direct nose-to-brain routes trigger neuroinflammation and oxidative stress when airborne pollutants enter the bloodstream. Research has shown that exposure to environmental pollution may therefore play a role in triggering and exacerbating psychiatric disorders due to neuroinflammatory and excitotoxic processes (Khan et al., 2019). Exposure to nitrogen dioxide (NO_2) and particulate matter may disturb sleep quality (Billings et al., 2019). Sleep disturbances are known to increase a person's susceptibility to developing psychiatric disorders, such as depression, anxiety and schizophrenia, when experienced over a prolonged period.

Self-reported annoyance caused by high levels of noise diminished the restorative effects of neighbourhoods in Bulgaria and reduced physical activity, thus leading to poorer mental health (Dzhambov et al., 2018). These findings reflect conclusions drawn up from other studies (Foraster et al., 2016; von Lindern et al., 2016). Foraster et al. (2016) found that reduced physical activity triggered by noise related annoyance was a possible outcome of both night-time sleep disturbance and subsequent day-time sleepiness. On the other hand, Zijlema et al., (2017) did not find evidence that natural environments mediated the relationship between self-reported noise annoyance and particulate matter and cognitive function.

There is compelling evidence to suggest that anthropogenic noise can be buffered to a substantial degree by purposefully designed greenspace areas and green infrastructure, providing relief to urban residents (González-Oreja et al., 2010; Irvine et al., 2009). Roadside vegetation can be especially useful for abating surface transport noise (Han et al., 2018; Pathak et al., 2018) and is superior to synthetic barriers both psychologically and in absolute values of noise (Gallagher et al., 2015; Ow and Ghosh, 2017). Furthermore, synthetic barriers are often considered to be less economical than vegetation barriers as the costs associated with construction are high (Ohiduzzaman et al., 2016). Ow and Ghosh (2017) found that reductions in transport noise of up to 50% were observed when vegetation barrier densities increased from minimal to moderate, with only marginal improvements recorded beyond that density. Similarly, vegetation barriers for improving surrounding air quality via the deposition of pollutants have been evaluated in numerous numerical, field and wind tunnel studies and have provided extremely promising results (Al-Dabbous and Kumar, 2014; Hagler et al., 2012). While research can support the effective implementation of vegetation barriers, it is important to remember that the optimum physical structure of such barriers is typically context-dependent and must account for environmental variables (e.g., topography, air flow) as well as vegetation characteristics (Abhijith et al., 2017; Barwise and Kumar, 2020; Hewitt et al., 2020).

1.10 Urban gardens and streetscapes

Domestic gardens make up a substantial proportion of greenspace coverage within urbanised landscapes. Estimates vary from 16% in Sweden (Colding et al., 2006), to 36% in New Zealand (Mathieu et al., 2007), and between 25 – 50% in the United Kingdom (Loram et al., 2007). Figures can also be high in developing nations. For example, in the city of León, Nicaragua, estimations stretch as far as 86% (González-García and Sal 2008). In addition to supporting biodiversity conservation, gardens can also benefit the well-being of urban dwellers by acting as a primary source of nature contact. This is of particular importance for those who are less physically mobile (Finlay et al., 2015), and, indeed, anyone who lacks garden space as people do not typically compensate with more frequent visits to public greenspaces (Grahn and Stigsdotter, 2003). In the UK, one in eight households (12%) do not have access to a private or shared garden space, affecting more than 3.3 million families. This statistics rises to more than one in five (12%) across large cities such as London (ONS, 2020).

The vast majority of geospatial analyses exclude greenspaces less than one hectare in size, usually due to the spatial resolution of landcover datasets used for the analyses (Cox et al., 2017; van den Bosch and Ode Sang, 2017). Only a small amount of research has considered these overlooked small-scale greenspaces when measuring what "dose" of nature is needed to bring about positive outcomes. However, the few studies that have been conducted show that spending five hours in a garden per week could prevent 27% of depression cases (Cox et al., 2017), and garden owners consider positive well-being outcomes as a main benefit provided by gardens (Calvet-Mir et al., 2012). Results also scale proportionately to improved public health and well-being (Cox et al., 2017; Shanahan et al., 2015). More research into small-scale greenspaces specifically could potentially yield some important policy and practice outcomes.

There is a very limited understanding as to the social and ecological contribution or importance of front gardens and streetscapes (i.e., people's own front gardens, neighbouring gardens, street trees and public greenspace/green infrastructure such as road verges) (Chalmin-Pui et al., 2021). In a review by Wendelboe-Nelson et al. (2019) it was revealed that only 1% of the literature surrounding greenspace and well-being involved residential gardens. Likewise, while evidence suggests that residents who live in greener neighbourhoods are happier and healthier than those who do not (Wang et al., 2020; Wood et al., 2018b), very few studies have considered streetscapes specifically. Streetscapes are an important spatial scale to consider as they are akin to a viewshed from people's households and are the first point of contact before entering the home.

In the UK, approximately five million front gardens are absent of flora (~33%) and four and a half million (~25%) are completely paved over (Royal Horticultural Society, 2015). Front gardens are now comprised of three times less flora coverage than just one decade ago (Royal Horticultural Society, 2015). In part, this is due to the fees and regulations associated with roadside parking and the desire for low maintenance gardens, as well as the lack of skill required to manage greenspaces (London Assembly, 2005). Moreover, in the UK, planning law does not cover the protection of domestic gardens as they are not categorised as a stand-alone land-use type (Sayce et al., 2012).

1.11 Thesis scope, aims and outline

This thesis aims to explore the knowledge gaps surrounding biodiversity and human mental well-being associations at the streetscape level, using the city of Leeds, UK, as a study system. Urban biodiversity may help to lessen the negative impacts of pollution on human mental well-being across streetscapes, especially as air and noise pollution are heightened by road traffic. Therefore, this thesis specifically seeks to understand how species richness might mediate the often negative association between air and noise pollution and mental well-being. Given the likely complex interplay between these interactions, parallel mediation models are used to examine simultaneous effects. To my knowledge, this is the first time in the literature that a study has accounted for potential disparity between public estimations of species richness and actual measures of species richness by including both actual and perceived biodiversity measures (flowering plants, pollinators and trees) within this kind of modelling framework. Understanding how these variables (perceived and actual levels of air and noise pollution and biodiversity, and their impacts on subjective mental well-being) interact means taking steps towards creating cities which can benefit biodiversity conservation while also addressing rising trends in declining mental health (Patel et al., 2018). This is particularly pertinent as urban population growth is set to continue (United Nations, 2018).

Chapter 2. Methods

2.1 Study system

This research was conducted across a sample of urban streets in the city of Leeds, UK (53° 47′ 59″ N, 1° 32′ 57″ W; Fig. 1). The city is the fourth largest in England (~552 km^2) with a human population of ~790,000, which is highly ethnically and culturally diverse (Office for National Statistics, 2018). Leeds is typical of cities found in developed, temperate nations, as it includes a wide range of residential areas and housing types (Goddard et al., 2013). Urban greenspaces cover more than half of the city (~55%), while gardens alone account for 28% of the area (Baldock et al., 2019). However, over a 33-year period, impervious surfaces have increased by 13% across Leeds (Perry and Nawaz, 2008).

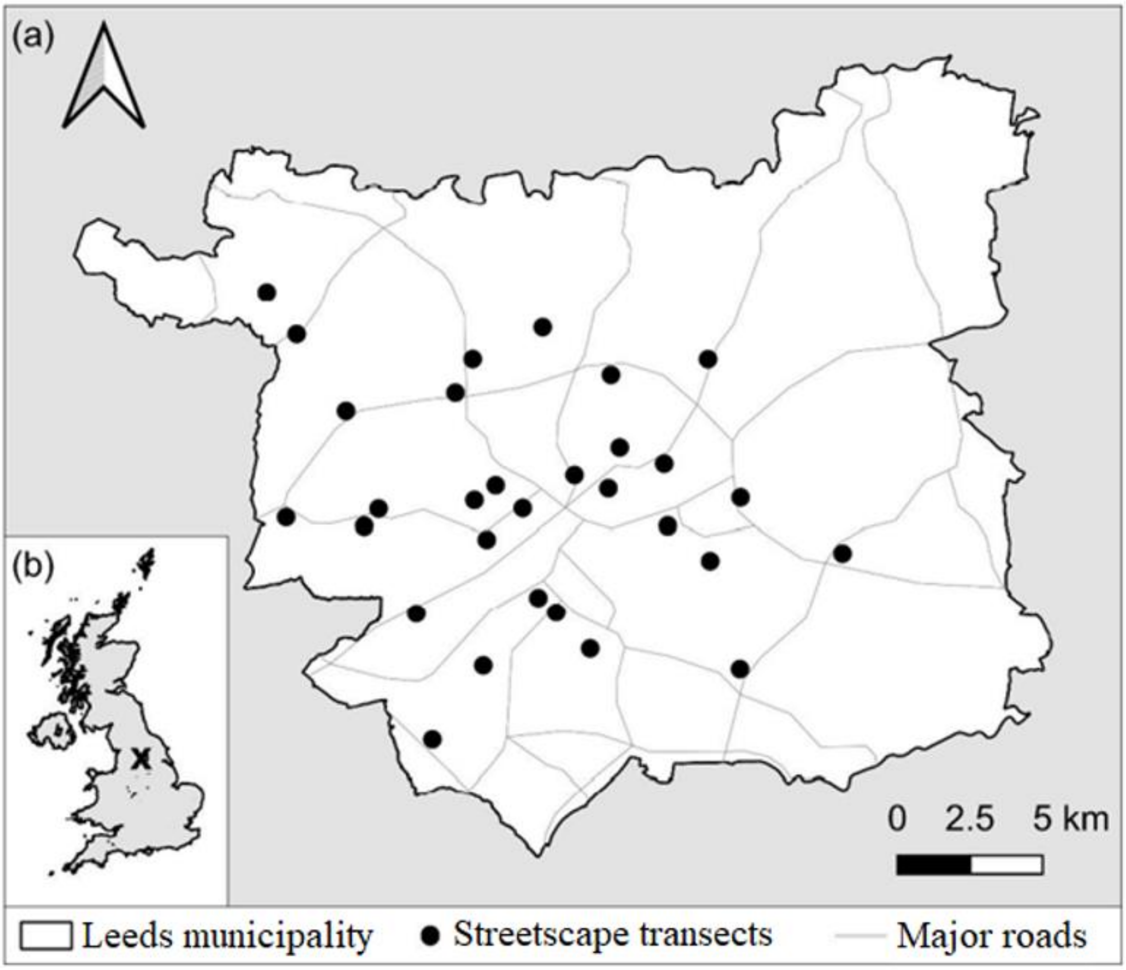


Figure 1. Study area showing (a) the municipality of Leeds, its major roads (grey lines) and the location of each of the 30 streetscape transects (black circles). (b) The location of Leeds (cross) in the UK.

In the UK, most air and noise pollution is caused by road transport (United Kingdom Department for Business, Energy and Industrial Strategy, 2018, p14) but, at the time of research, no systematic and accessible data had been collected on either for Leeds. This study therefore used a hierarchical sample design to capture air and noise pollution variation, using road size and traffic capacity as a proxy. Three pollution categories were identified based on road type. 'High' pollution roads included major roads designated to provide large-scale transport links within or between major urban centres, including dual carriageways (i.e., classified formally as "A roads"). 'Medium' pollution roads comprised roads intended to connect different areas and to feed traffic between major and smaller roads on the network, including any roads greater than 4 metres in width (i.e., "B roads"). 'Low' pollution roads were all other roads less than 4 metres in width.

Ten roads from each pollution category (30 in total) were selected (Fig. 1), with a 200 m long streetscape transect established down each one. Each streetscape was located in a different ward of Leeds to maximise spatial variation. Streetscapes encompassed all green infrastructure (GI) within the transect, which included GI within front gardens and GI situated outside of front gardens (e.g., street trees). Transects were at least 0.6 km apart, although the majority were over 1 km in distance, to maximise sample independence. Most urban pollinator species do not forage beyond distances of 1 km (Garbuzov et al., 2015; Langellotto et al., 2018). Streetscapes were selected to ensure that variation in housing type was represented (e.g., detached, semi-detached, terrace) as this is a good proxy for garden size and socio-economic status (Loram et al., 2007).

2.2 Noise and air pollution

Ambient nitrogen dioxide (NO_2) and particulate matter ($PM_{2.5}$) concentrations (μg/m^3) were measured at each streetscape transect. The data collection methods used are suitable for relative spatial comparisons across a study system (e.g., Bush et al., 2001), but are not suitable for carrying out internationally recognised monitoring of pollution levels (e.g., Ngo et al., 2019). Therefore, the findings cannot be compared to other air pollution data that has been gathered in Leeds collected by other studies, or to that collected from other cities.

Diffusion tube samples were used to measure NO_2 concentrations. Three diffusion tubes were situated on separate lampposts approximately 2.5 m high, within at least 10 m of the start/end of the transect and on alternating sides of the street (where possible) to maximise the distance between tubes. The diffusion tubes were left in place for four weeks in May/June and three weeks in July/August. An average concentration was calculated across all tubes and sampling periods.

$PM_{2.5}$ concentrations were recorded using the IQAir AirVisual Pro. While the AirVisual Pro is a low-cost monitor, the data derived from it allow discrepancies in concentrations between study streets to be evaluated (e.g., Ngo et al. 2019). Noise pollution was collected at breast height using a Reed ST-8850 sound level metre and was measured in decibels (dB). $PM_{2.5}$ and noise levels were obtained by walking at a slow pace along both sides of the 200 m transect across two independent visits between May and August 2019. To minimise potential bias that may have arisen due to variation in meteorological conditions (which is known to affect air quality; Yoshikado and Tsuchida, 1996), all streets from each of the respective pollution categories were sampled on the same or adjacent days. $PM_{2.5}$ concentration measurements were only taken on days with comparable and suitable weather conditions (Mues et al., 2012).

2.3 Greenness (NDVI)

The greenness of a neighbourhood is known to impact the psychological well-being of its residents (Sarkar et al., 2018; Wang et al., 2020). To quantify the overall 'greenness' of each transect, NDVI data were obtained based on MODIS (MOD13Q1 Collection 6 satellite data 16-day 185 composite at a 250 m spatial resolution; ORNL DAAC, 2018) satellite imagery at two spatial scales (0.25 km^2 and 2.25 km^2 centred on the transect midpoint) and used as a covariate within the parallel mediation models. The spatial scale of 0.25 km^2 was included as this is the finest spatial scale made available from the Modis vegetation indices data (MOD12Q1) and corresponds to the vegetation along the street as well as within the immediate vicinity. The 2.25 km^2 scale represents the vegetation cover in the local landscape more broadly as a crude measure of habitat availability for biodiversity. NDVI is a widely used method for quantifying green vegetation and does so by normalising green leaf scattering in Near Infra-red wavelength with chlorophyll absorption in red wavelengths (Myneni et al., 1995).

2.4 Actual measures of biodiversity

Actual species richness was recorded for pollinators, flowering plants and trees across each of the 30 streetscapes. Pollinators were included as they form a large component of urban biodiversity that is visible throughout the day, such as bees and butterflies. This study used an adapted sampling approach from Baldock et al. (2015). Pollinator transects were conducted by walking at a steady pace down each 200 m streetscape transect. Pollinators were observed and recorded if seen foraging on flowers or flying within the transect, within a 2 m height and 4 m width boundary. Transects ran between the pavement and residential gardens (including road verges if present) on the side of the streetscape where most GI was present. Pollinators were recorded as one of 20 morpho-functional groups (Supplementary Information S2). Data collection was conducted during May and July 2019 when weather conditions were suitable. Flowering plant richness was measured by noting down each observed plant within the 200 m transect, which was in flower at the time the transect was conducted. Grasses, sedges and windpollinated forbs were not included. Flowering plant data was collected over two separate visits. All tree species ≥ 2 m in height were recorded within each streetscape transect.

2.5 Questionnaire

Questionnaires (Supplementary Information S2) were administered *in-situ* face-to-face from June to August 2019. A total of 1033 households from all 30 streetscapes were eligible to complete one questionnaire per household. Each street was visited on three separate occasions, on different days (both weekends and weekdays) and at different times of the day (e.g., during the day and evening) to maximise response rates. Participants were required to be over the age of 18 to complete a questionnaire and informed consent was obtained prior to completion. Questionnaires were available to complete in English and Urdu (the most commonly spoken language in Leeds, second to English). Ethics approval was granted by the University of Leeds Social Sciences, Environment and LUBS (AREA) Faculty Research Ethics Committee, reference AREA 19-165 (University of Leeds was the institution leading the overarching project, rather than University of Kent). The questionnaire was piloted with a total of 21 Leeds residents who were not residing in any of the 30 streetscapes included within the study. Following the pilot, some wordings were changed to improve flow and clarity of the questionnaire.

2.5.1 Perceptions of biodiversity and pollution

Human perceptions of biodiversity were obtained by asking each participant to estimate the number of flowering plant, pollinator and tree species in their streetscape, which was termed ‘street environment’ in the questionnaire. It was explained in the questionnaire (Supplementary Information S1) that ‘street environment’ referred to all GI included within front gardens and the street itself (e.g., road verges). Participants were asked to estimate the number of species using a five-point scale for flowering plants (‘fewer than 10’, ‘10 to 30’, ‘31 to 50’, ‘51 to 99’, ‘100 or more’), pollinators (‘fewer than 5’, ‘5 to 9’, ‘10 to 13’, ‘14 to 19’, ‘20 or more’) and trees (‘fewer than 5’, ‘5 to 10’, ‘11 to 15’, ‘16 to 20’, ‘21 or more’). Categories were informed by previous research conducted on urban biodiversity in Leeds (Goddard et al., 2013).

Human perceptions of air pollution were obtained by asking each participant (Supplementary Information S1) how polluted they believed their streetscape to be, using a five-point scale as a response (‘very polluted’, ‘polluted’, ‘neutral’, ‘clean’, ‘very clean’). Noise pollution perceptions were obtained by asking participants to respond to four items (Supplementary Information S1). Each of these were answered using five-point scales, which were added together to form a composite score: (i) ‘completely unacceptable’, ’unacceptable’, ‘neutral’, ’acceptable’, ’completely acceptable’; (ii) ‘very quiet’, ‘quiet’, ‘Neutral’, ’noisy’, ‘very noisy’; (iii) ‘not at all annoying’, ’not annoying’, ’neutral’, ’annoying’, ’very annoying'; (iv) 'very relaxing', 'relaxing', 'neutral', 'stressful', 'very stressful'). Item (i) was reverse coded as the scale was inverted. Participants were also asked to list: *"What three main sounds do you hear in your street environment?"* to gain further insight into their streetscape soundscape.

2.5.2 Well-being outcomes

The questionnaire (Supplementary Information S1) used two different self-reported well-being outcomes: the Short Warwick-Edinburgh Mental Well-being Scale (SWEMWBS) and happiness. Both are validated scales commonly used in nature-health related research (Houlden et al., 2017; van Herzele and de Vries, 2012). SWEMWBS consists of seven affect items relating to the primary components of psychological well-being, including those of a hedonic (feeling of positive emotions, satisfaction) and eudaimonic (functioning, relationships, sense of purpose) nature (Stewart-Brown et al., 2009). Participants were asked to: "*Tick the box that best describes your experience of each over the last two weeks*" and responded using a five-point scale ('none of the time', 'rarely', 'some of the time', 'often', 'all of the time'). Scores for each item were summed to produce an initial raw score that was then transformed using the SWEMWBS conversion table to give a final score, which could have ranged from 7 to 35. The happiness scale used was adapted from Fordyce (1988) and asked participants: "*In your life in general, how happy would you say you are?*" on a scale of one to ten.

2.5.3 Covariates

An individual's sensitivity to noise is likely to influence how noisy and acceptable they believe their streetscape to be, and therefore may influence the restorativeness of streetscape biodiversity. To measure self-reported sensitivity to noise we used four items from Weinstein's (1978) noise sensitivity scale in the questionnaire (Supplementary Information S1). The four items have shown relative context independence while, in particular, the last item ("*I am sensitive to noise*") correlates with results from Weinstein's scale in its entirety in the adult Finnish population (Heinonen-Guzejev et al., 2004). Participants were asked to respond on a five-point scale. One item was adapted to represent the streetscape scale from: "*I get annoyed when my neighbours are noisy*" to: "*I get irritated when there is noise in my street*".

Given the influence that social cohesion has on well-being in the literature and, by extension influences people's perception of the world around them, we included a self-reported social cohesion scale (Sampson et al., 1997) in the questionnaire (Supplementary Information S1). Social cohesion was measured using five items, three of which were positive affect and two negative affect. The scores were then added together to create an overall social cohesion score.

Sociodemographic characteristics collected from the participants in the questionnaire (Supplementary Information S1) included: age, gender, ethnicity and employment status. Information pertaining to the household was also gathered, such as the number of years lived at the property, tenure status and number of permanent residents. Gross household income data was not requested due to the sensitive nature of the question. Instead, this was obtained from the most recent national census in 2011 (Office for National Statistics, 2018), using the most spatially resolved data ('Lower Layer Super Output Area') to cover each streetscape.

2.6 Statistical analysis

2.6.1 Parallel mediation models

Four parallel mediation models (Fig. 2) were constructed to test the mediating effect of species richness (actual and perceived) on the relationship between noise (dB) and air ($PM_{2.5}$) pollution on human well-being (SWEMWBS and happiness). The models were adjusted for covariates (NDVI, noise sensitivity, social cohesion, gender, years of household occupancy, gross household income). They simultaneously calculate regression analyses between predictors (dB, $PM_{2.5}$) and mediators (biodiversity actual, perceived) (paths a1 – a6), predictors and outcome variables (SWEMWBS, happiness) (direct effect, paths c1 and c2), and mediators and outcome variables (b1 – b3). The indirect effect (a*b) measures how predictors influence the wellbeing outcomes as a result of the influence of the predictors on the mediators, that, in turn, also affect the wellbeing outcomes.

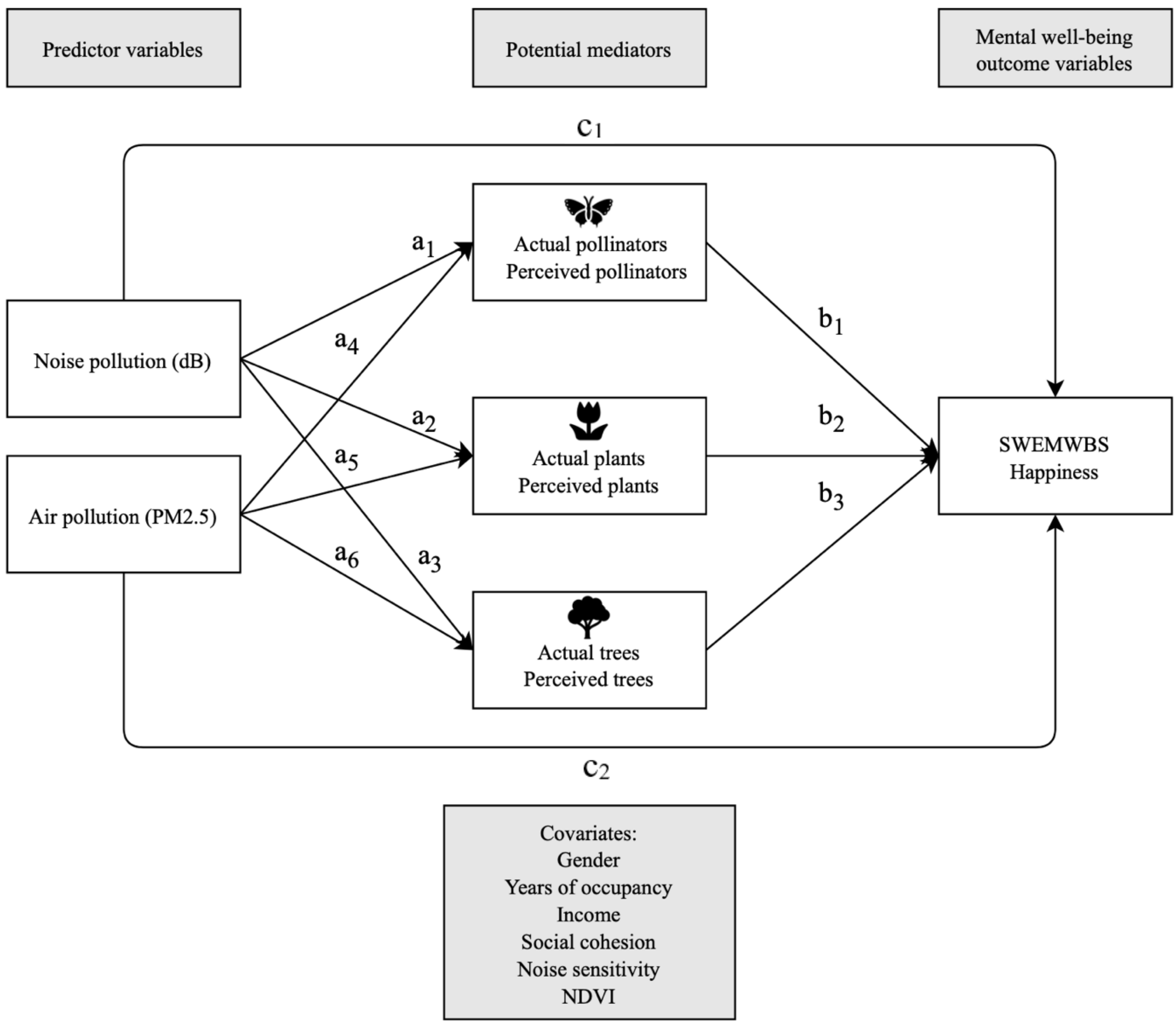


Figure 2. Conceptual diagram of the model framework. Parallel mediation models tests the effects of predictors (noise and air pollution) on well-being outcomes (SWEMWBS and happiness), via potential mediators (actual and perceived flowering plant, pollinator and tree richness). Actual pollinator richness was recorded at morpho-functional group richness level, rather than species richness. a paths = tested direct associations between predictor and wellbeing outcomes. b paths = tested direct associations between mediators and wellbeing outcomes. c paths = tested direct associations between predictors and wellbeing outcomes. The indirect effect is the product of the a and b paths (a*b).

Parallel mediation models were performed using R statistical software version 3.6.0 (R Core Development Team, 2020), using the ‘Lavaan survey’ package designed for structural equation modelling (Oberski, 2014). ‘Laavan survey’ allows for the analysis of clustered samples using cluster-robust standard errors (SE) (i.e., households from the same streetscape). Analyses accounted for ‘street code’ clustering of data by including it as a random effect. Three streetscapes with fewer than five completed questionnaires were removed from analyses to strengthen statistical reliability. Two were from the low pollution streetscapes and one from the medium pollution streetscapes.

Spearman's pairwise correlations (Fig. 3) and Kruskal Wallace tests were used to test for associations between categorical and continuous demographic variables. Due to a large proportion of the sample choosing not to disclose their age (n = 31), which strongly correlated with years of household occupancy ($r = 0.67$, $p <0.001$), the latter was used to maximise sample size. A strong association was identified between employment status and gross household income ($X^2 = 6.46$, df = 2, $p < 0.05$) using the 'het.cor' function on R, specifically designed for categorical and ordinal data. Gross household income was therefore included as a covariate to account for the possible influence it may have on regular garden maintenance (Kendal et al., 2012). Gender was treated as binary, and both years of household occupancy and income were scaled and centred to stabilise variances. Due to a strong correlation between dB and NO_2 ($r = 0.77$, $p = < 0.001$), NO_2 was not included in the models.

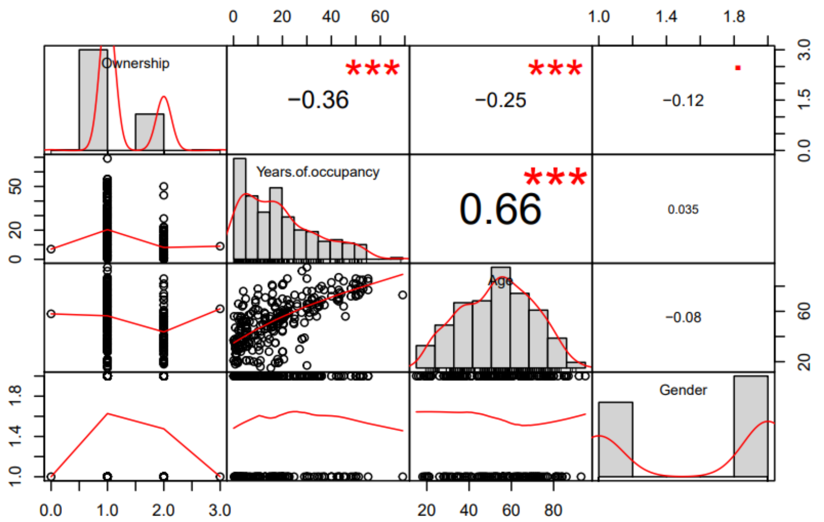


Figure 3. Spearman's rank correlation analyses used to identify associations between categorical covariates: ownership, years of occupancy, age and gender.

Four models were run in total, interchanging between perceived/actual biodiversity measures as mediators and SWEMWSB/happiness as outcome variables. The data showed no evidence of multicollinearity via Variance Inflation Factors, all of which were < 1.9. Models were estimated using a maximum likelihood estimator and a Satorra-Bentler scaled test statistic that is robust to non-normality in the model variables. The error variances between the three mediators were free to covary due to the plausible biological associations between them (e.g., more pollinators are likely to be found where flowering plant richness is greater). To check for model fit, a Chi-square p value adjusted for the random effect/clustered data was used ('pval.pFsum' function; Obserski, 2014) alongside other fit measure statistics including RMSEA (and its 95% CI), SRMR and comparative fit index (CFI). Good model fit was identified ($p > 0.05$, CFI > 0.95, 313 RMSEA < 0.06, RMSEA 95% confidence intervals < 0.06, SRMR < 0.08) (Hu and Bentler, 1999; MacCallum et al., 1996; Oberski, 2014; Barrett, 1997).

2.6.2 Bivariate analyses

Bivariate analyses were performed using R statistical software version 3.6.0 (R Core Development Team 2020). Spearman's rank correlations were used to test the associations between perceptions of flowering plant, pollinator and tree species richness/morpho-functional group richness and actual richness for each taxon. A spearman's rank correlation was also performed on perceptions and actual measurements of air ($PM_{2.5}$, NO_2) and noise pollution (dB). Pearson's correlations were used to test the associations between actual measures of flowering plant, pollinator and tree species richness/morpho-functional group richness and actual pollution readings ($PM_{2.5}$, NO_2, dB). Kruskal Wallis tests for non-normally distributed data were used to check for statistical differences in SWEMWBS, happiness, social cohesion and noise sensitivity scores between the three pollution categories. A Kruskal-Wallis test and Dunn's test were used to identify statistical differences between NO_2 and dB levels across the three pollution categories.

Chapter 3. Results

3.1 Descriptive and bivariate statistics

A total of 282 participants (27.3% response rate) from across all 30 streetscapes completed the questionnaire. Participant's ages ranged from 18 to 91 and 55.3% (n = 156) were female. At the time of data collection most participants (92%, n = 260) had been living at their property for at least one year. The demographic profile of the sample was representative of the overall population of Leeds, based on sex, age and ethnicity (Table 1).

Table 1. Demographic characteristics of the study sample (sex, age, ethnicity) taken from 282 questionnaires across 30 streets in Leeds and the 2011 national census population for Leeds (Office for National Statistics, 2018). G-tests (indicated in italics) used to measure data goodness of fit show that the study sample was representative of the overall population of Leeds.

Demographic Characteristic	N	%	Census %
Gender			
Male	116	41.1	49.1
Female	156	55.3	50.9
Prefer not to say	10	3.5	-
G = -2500.7, X^2 df = 1, p = 1			
Age			
16 – 24	19	6.7	18.8
25 – 29	10	3.5	9.7
30 – 44	50	17.7	25.3
45 – 59	76	26.9	21.6
60 – 64	17	6	6.5
65 – 84	66	23.4	15.4
85+	10	3.5	2.3
Prefer not to say	34	12	-
G = -2222.3, X^2 df = 6, p = 1			
Ethnicity			
White	205	72.7	81.1
White other	20	7.1	3.9
Asian	27	9.6	7.8
Black	7	2.5	3.4
Other	14	5	4.3
Prefer not to say	9	3.2	-
G = -2507.0, X^2 df = 4, p = 1			

NO_2 and dB pollution measures (Table 2) both increased across the three pollution categories (low, medium, high) (NO^2: $X^2 = 112.93$, df = 2, p < 0.001; dB: $X^2 = 229.46$, df = 2, p < 0.001) (Table 3), demonstrating that the sampling approach was representative of how polluted each streetscape was (Fig. 4). However, $PM_{2.5}$ measures were highly variable across pollution categories ($X^2 = 0.92$, df = 2. p = 0.62).

Table 2. Summary statistics detailing the median, interquartile range (IQR), mean, standard error (SE), minimum and maximum recorded values for the predictors, well-being outcomes, mediators and covariates examined in the study. Noise pollution, air pollution, actual biodiversity (pollinators, flowering plants, trees) and greenness (NDVI) were measured across all 30 streetscapes. Actual pollinator richness was recorded at morpho-functional group level, whereas flowering plants and trees were recorded as species richness. Well-being outcomes, perceived biodiversity (pollinators, flowering plants, trees) and remaining covariates were assessed via the questionnaire (n = 282).

	Median	IQR	Mean	SE	Min	Max
Predictors						
Noise pollution (dB)	82.40	13.65	59.56	0.50	67.5	93.6
Air pollution ($PM_{2.5}$ μg/m^3)	4.69	3.17	5.15	0.11	2.26	8.83
Well-being outcomes						
SWEMWBS	22.33	4.67	22.87	0.22	13.33	35
Happiness	8.00	2	7.90	0.10	1	10
Mediators						
Actual pollinator richness	7.00	4	6.91	0.17	0	14
Actual flowering plant richness	63.00	19	63.01	1.09	31	101
Actual tree richness	17.00	8.75	19.02	0.43	2	33
Perceived pollinator richness	3.00	2	2.90	0.07	1	5
Perceived flowering plant richness	2.00	1	2.51	0.07	1	5
Perceived tree richness	2.00	1	2.24	0.05	1	5
Covariates						
Greenness (NDVI)	0.50	0.28	0.48	0.01	0.15	0.79
Social cohesion	2.88	0.75	2.85	0.03	1	4.62
Noise sensitivity	13.00	3	12.74	0.14	4	20
Gender	-	-	-	-	-	-
Years of household occupancy	16	22.34	18.90	0.91	0.08	69.16
Gross household income	31334	16256	32062	602.32	11553	51598

Table 3. Results of Kruskal-Wallis test (indicated in italics) and the post hoc Dunn's multiple comparisons test, showing significant differences in NO_2 and dB readings across the three pollution categories (low, medium, high).

Comparison	Z	P
NO_2		
High – Med	7.53	< 0.001
High – Low	15.13	< 0.001
Low – Med	-7.35	< 0.001
$X^2 = 112.93, df=2, p < 0.001$		
dB		
High – Med	10.52	<0.001
High – Low	3.57	< 0.001
Low – Med	-6.71	< 0.001
$X^2 = 229.46, df=2, p < 0.001$		

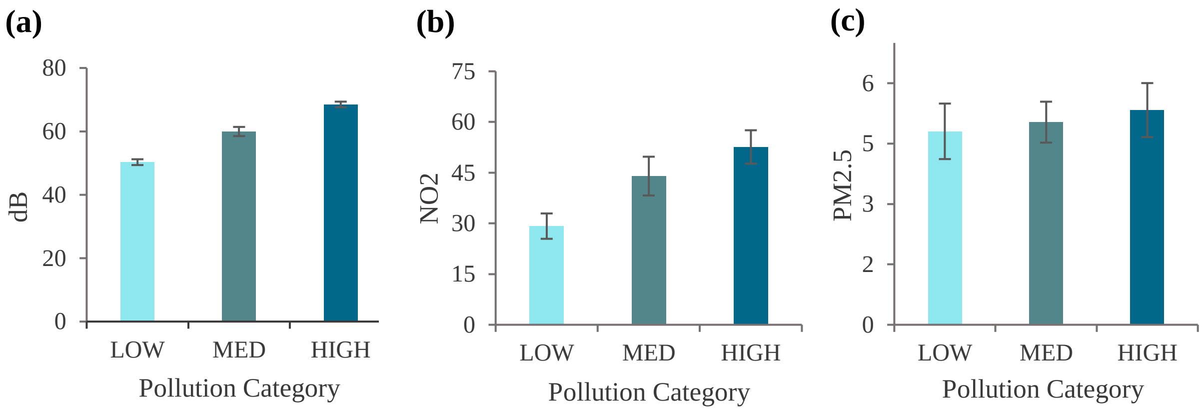


Figure 4. Median pollution levels and associated IQR in the low, medium and high pollution category streetscapes for: (a) noise (dB); (b) nitrogen dioxide (NO_2); and (c) particulate matter ($PM_{2.5}$).

Actual biodiversity richness (pollinators, flowering plants, trees) was variable across streetscapes (Table 1). Flowering plant richness had a significant negative association with NO_2 ($r = -0.54$, $p < 0.001$) and dB ($r = -0.35$, $p < 0.001$), but not with $PM_{2.5}$ ($r = 0.08$, $p = 0.178$). Pollinator richness also had a significant negative association with NO_2 ($r = -0.54$, $p < 0.001$) and dB ($r = -0.35$, $p < 0.001$), but not with $PM_{2.5}$ ($r = -0.03$, $p = 0.543$) (Fig. 5). No associations were observed between tree richness and any of the pollution measures.

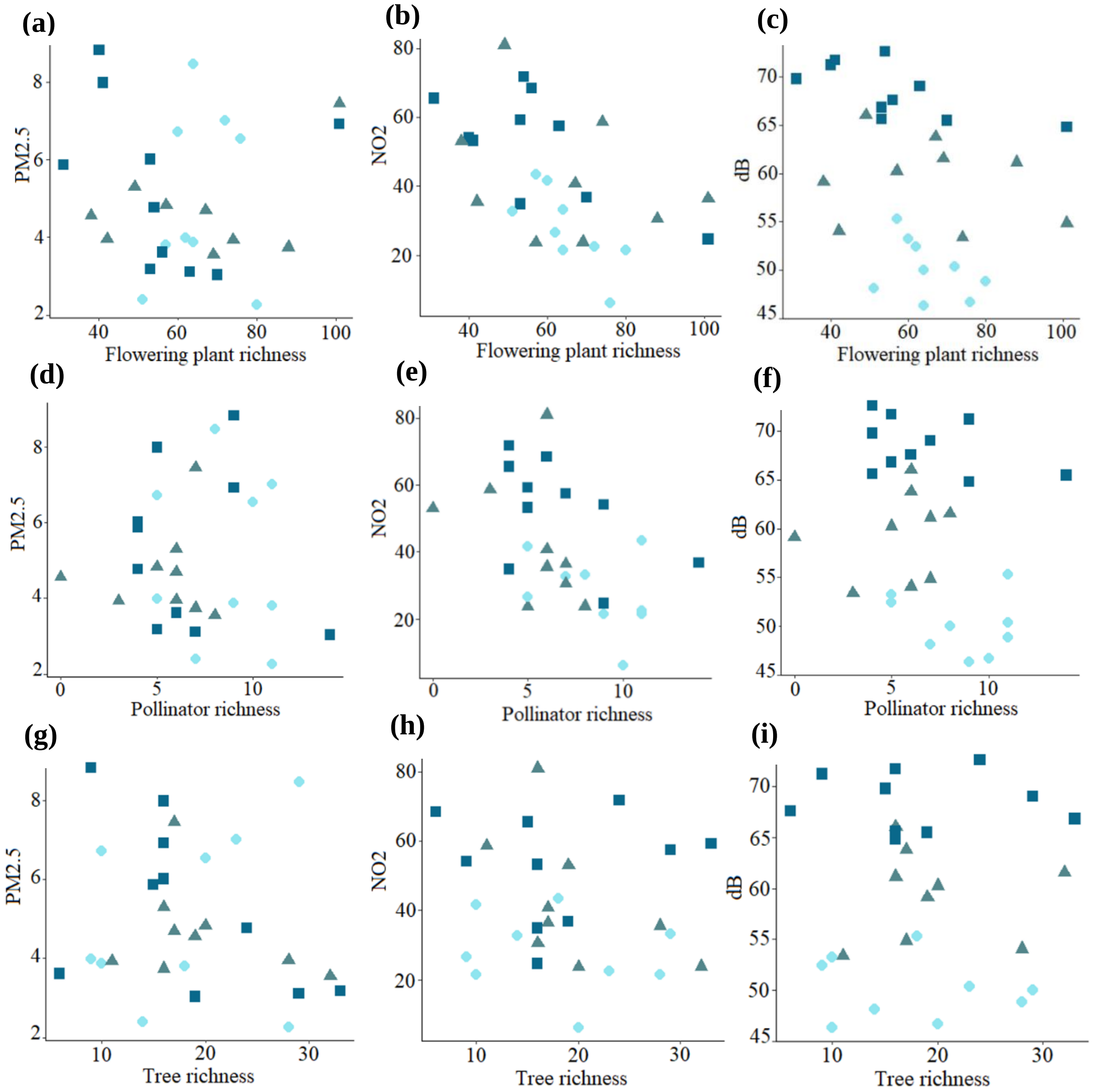


Figure 5. Associations between actual flowering plant richness and (a) particulate matter ($PM_{2.5}$); (b) nitrogen dioxide (NO_2); and (c) noise (dB) pollution. Associations between actual pollinator richness and (d) particulate matter ($PM_{2.5}$); (e) nitrogen dioxide (NO_2); and (f) dB. Actual pollinator richness was assessed by morpho-functional group richness. Associations between actual tree richness and (g) particulate matter ($PM_{2.5}$); (h) nitrogen dioxide (NO_2); and (i) noise pollution (dB). Each symbol is a mean value for each streetscape, with the shape indicating the pollution category of the streetscape: circle = low; triangle = medium; and square = high.

No association between participant perceptions of pollinator richness and actual pollinator richness ($r = 0.29$, $p = 0.126$) was recorded (Fig. 6). However, significant positive associations were observed between perceived and actual flowering plant ($r = 0.54$, $p < 0.05$) and tree ($r = 0.37$, $p < 0.05$) richness.

A significant negative association was observed between perceived air pollution and NO_2 ($r = -0.41$, $p < 0.001$), although no association was found with $PM_{2.5}$ ($r = -0.002$, $p = 0.970$). A significant negative association was also observed between perceived noise pollution and dB ($r = -0.20$, $p < 0.001$).

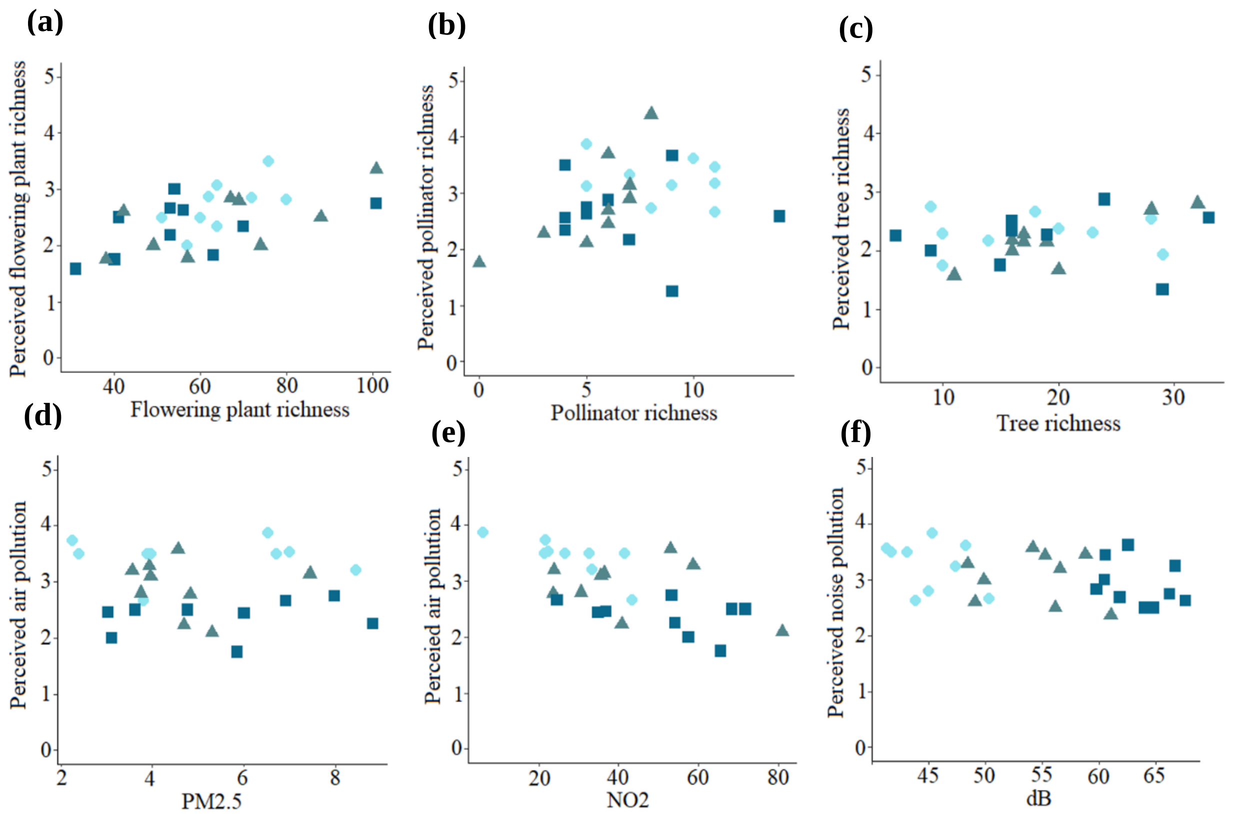


Figure 6. Associations between actual and perceived richness of: (a) flowering plants; (b) pollinators; and (c) trees. Actual pollinator richness was assessed by morpho-functional group richness, whereas flowering plants and trees were measured by species richness. Associations between perceived streetscape air pollution and (d) particulate matter ($PM_{2.5}$) and (e) nitrogen dioxide (NO_2). Associations between perceived streetscape noise pollution and (f) noise pollution (dB). Each symbol is a mean value for each streetscape, with the shape representing pollution category of the streetscape: circle = low; triangle = medium; and square = high.

Across the 282 participants, SWEMWBS and social cohesion scores had a central tendency, whereas happiness scores were right-skewed (Table 1). There were no significant differences in SWEMWBS ($X^2 = 1.49$, df = 2, $p = 0.481$) and happiness ($X^2 = 2.26$, df = 2, $p = 0.322$) scores between the three pollution categories (Fig. 7). Social cohesion demonstrated significance between pollution categories ($X^2 = 27.27$, df = 2, $p < 0.001$), whereby social cohesion scores decreased as the pollution category increased.

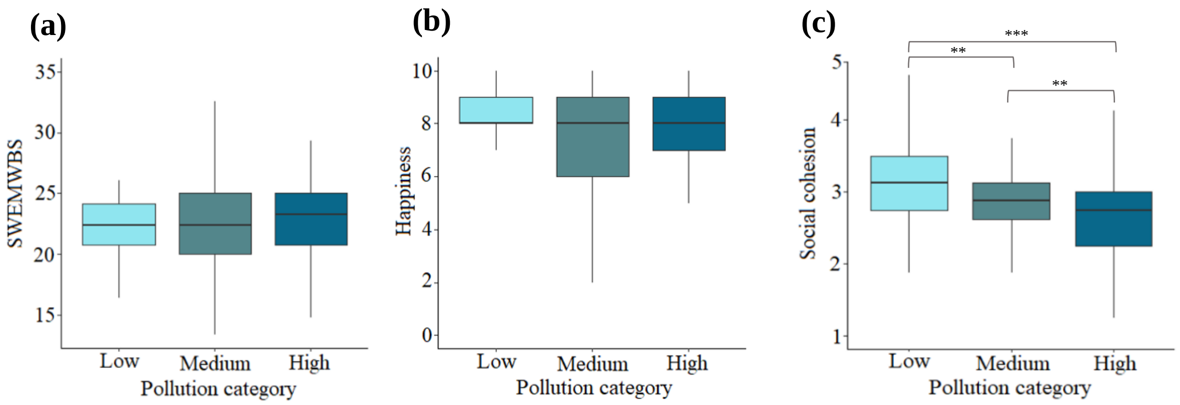


Figure 7. Box plots showing participant responses (n = 268), grouped by the low, medium and high pollution categories, to: (a) SWEMWBS; (b) happiness; and (c) social cohesion. No significant relationship was identified between SWEMWBS or happiness scores across pollution categories. Significant differences between pollution categories for social cohesion scores are indicated using symbols: ** = < 0.05; *** = < 0.001.

Participants most commonly reported hearing sounds that were anthropogenic in nature, such as cars and traffic (Table 4), across all pollution categories. However, 'traffic' was mentioned much less in low pollution streetscapes (n = 6) compared to medium (n = 34) and high (n = 41) ones. 'Birds' were reported most by participants living in low pollution streetscapes and least in high ones (n = 17 and 2 respectively). No significant differences were apparent in noise sensitivity scores across the three pollution categories ($X^2 = 2.09$, df = 2, p = 0.35) (Fig. 8).

Table 4. Types of sounds listed by participants when asked: "*What three main sounds do you hear in your street environment?*", grouped by pollution categories.

Sound Types	**Low**	**Medium**	**High**
Road vehicles	50	71	86
Aeroplanes	2	2	0
Birds	17	5	2
People	9	3	2
Music	1	0	1
Construction	0	0	2
Weather	0	1	0
Dogs	0	1	0

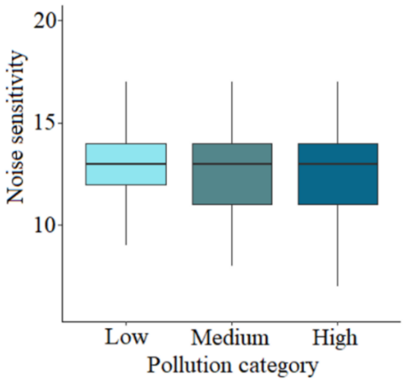


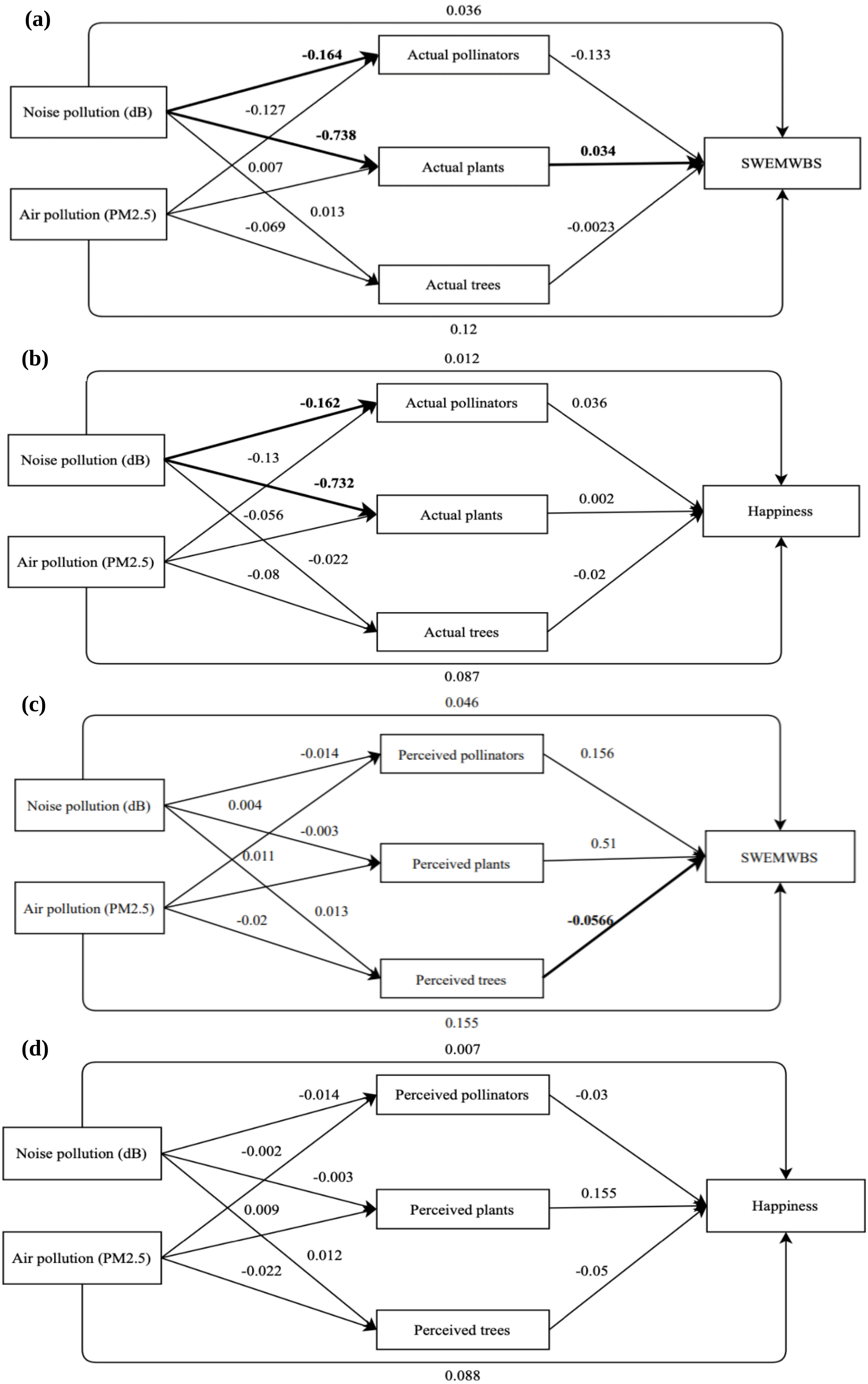
Figure 8. Box plot showing participant self-reported sensitivity to noise scores (n = 269), grouped by the low, medium and high pollution categories.

3.2 Parallel mediation models

The variance explained for the SWEMWBS and happiness well-being outcomes was 8% in each of the four models (Fig. 9). Noise pollution significantly and negatively influenced both actual pollinator and flowering plant richness, although this was not observed for tree richness. No such relationship was observed between air pollution and actual pollinator, flowering plant, or tree richness. Actual flowering plant richness had a positive effect on SWEMWBS outcomes, but not happiness (Table 5). Indirect pathways revealed that when noise pollution is increased, SWEMWBS decreases, mediated by decreasing actual flowering plant richness (Table 5). Air and noise pollution showed no direct effect on either SWEMWBS or happiness well-being outcomes.

People's perceptions of flowering plant, pollinator or tree richness was not influenced by the noise or pollution in their streetscape. Perceived flowering plant and pollinator richness had no direct effect on SWEMWBS or happiness scores either (Tables 7 and 8). However, perceived tree richness did have a significant and negative direct association with SWEMWBS score (Table 7).

Amongst covariates, gross household income had a significant positive effect on actual tree richness (β = 4.805, p = 0.004) and was positively associated with perceived pollinator and tree richness when happiness was used as a well-being outcome variable (Table 8). Social cohesion, across all models, was a significant positive predictor for perceived flowering plant, pollinator and tree richness.



(a)
0.036
Actual pollinators
-0.164
-0.133
Noise pollution (dB)
-0.127
-0.738
0.034
SWEMWBS
Actual plants
0.007
Air pollution (PM2.5)
0.013
-0.069
-0.0023
Actual trees
0.12
(b)
0.012
Actual pollinators
-0.162
0.036
Noise pollution (dB)
-0.13
-0.732
Actual plants
0.002
Happiness
-0.056
Air pollution (PM2.5)
-0.022
-0.08
-0.02
Actual trees
0.087
(c)
0.046
-0.014
Perceived pollinators
0.156
Noise pollution (dB)
0.004
-0.003
0.51
SWEMWBS
0.011
Perceived plants
Air pollution (PM2.5)
-0.02
0.013
-0.0566
Perceived trees
0.155
(d)
0.007
-0.014
Perceived pollinators
-0.03
Noise pollution (dB)
-0.002
-0.003
0.155
Happiness
0.009
Perceived plants
Air pollution (PM2.5)
0.012
-0.022
-0.05
Perceived trees
0.088

Figure 9. Parallel mediation models showing the effects of air pollution ($PM_{2.5}$) and noise pollution (dB) on (a) SWEMWBS mediated by actual pollinator (recorded as morpho-functional groups), flowering plant and trees richness, (b) happiness mediated by actual pollinator, flowering plant and trees richness, (c) SWEMWBS mediated by perceived pollinator, flowering plant and trees richness and (d) happiness mediated by perceived pollinator, flowering plant and trees richness. All models are adjusted for covariates (NDVI, social cohesion, noise sensitivity, gender, years of household occupancy and gross household income). Plots display the unstandardised estimates (n = 241) after rows containing missing values were removed, with statistically significant paths highlighted in bold ($p < 0.05$).

Table 5. Parallel mediation model testing the effect of noise and air pollution on SWEMWBS, mediated by actual richness of pollinators (assessed as morpho-functional group richness), flowering plants, and trees. Estimate values are standardised β values. Standard error = SE. Bold values indicate statistical significance ($p < 0.05$).

Model 1 **OUTCOME = SWEMWBS** **NOISE = ACTUAL** **BIODIVERSITY = ACTUAL**					
Fit statistics – Rubust					
Observations used		244			
Model fit test statistic		3.626			
Degrees of freedom		3			
P value from F test		0.306			
CFI (robust)		0.984			
SRMR		0.014			
RMSEA		0.041			
Lower CI		0.000			
Upper CI		0.164			
		Estimate	**SE**	**Z**	**P**
Actual pollinators					
Noise pollution	a_1	**-0.164**	0.059	-2.786	**0.005**
$PM_{2.5}$	a_4	-0.127	0.343	-0.37	0.711
Years occupancy		-0.139	0.246	-0.564	0.573
NDVI		-1.188	2.82	-0.421	0.673
Noise sensitivity		-0.013	0.066	-0.192	0.848
Social cohesion		-0.154	0.525	-0.293	0.769
Income		0.583	0.529	1.101	0.271
Actual plants					
Noise pollution	a_2	**-0.738**	0.312	-2.367	**0.018**
$PM_{2.5}$	a_5	0.007	2.249	0.003	0.998
Years occupancy		0.922	1.178	0.783	0.434
NDVI		12.938	19.63	0.659	0.51
Noise sensitivity		0.381	0.351	1.084	0.278
Social cohesion		2.953	2.666	1.107	0.268
Income		2.409	3.145	0.766	0.444
Actual trees					
Noise pollution	a_3	-0.013	0.223	-0.059	0.953
$PM_{2.5}$	a_6	-0.069	0.88	-0.079	0.937

Years occupancy		-1.033	0.453	-2.278	0.023
NDVI		-7.229	8.487	-0.852	0.394
Noise sensitivity		0.176	0.154	1.143	0.253
Social cohesion		-1.027	1.045	-0.983	0.326
Income		4.912	1.681	2.922	0.003
SWEMWBS					
Actual pollinators	b_1	-0.133	0.092	-1.439	0.15
Actual plants	b_2	**0.034**	0.013	2.663	**0.008**
Actual trees	b_3	-0.023	0.039	-0.595	0.552
Noise pollution	c_1	0.036	0.039	0.933	0.351
$PM_{2.5}$	c_2	0.12	0.163	0.739	0.46
Years occupancy		0.352	0.199	1.773	0.076
NDVI		-3.093	1.629	-1.898	0.058
Noise sensitivity		0.006	0.097	0.063	0.95
Social cohesion		0.592	0.342	1.729	0.084
Income		0.446	0.448	0.995	0.32
Gender		0.097	0.439	0.222	0.824
Mediation					
Noise_pollinators		0.022	0.015	1.428	0.153
$PM_{2.5}$_pollinators		0.017	0.039	0.43	0.667
Noise_plants		-0.025	0.008	-2.993	0.003
$PM_{2.5}$_plants		0	0.077	0.003	0.998
Noise_trees		0	0.005	0.058	0.954
$PM_{2.5}$_trees		0.002	0.02	0.08	0.936

Table 6. Parallel mediation model testing the effect of noise and air pollution on happiness, mediated by actual richness of pollinators (assessed as morpho-functional group richness), flowering plants, and trees. Estimate values are standardised β values. Standard error = SE. Bold values indicate statistical significance ($p < 0.05$).

Model 2 **OUTCOME = HAPPINESS** **NOISE = ACTUAL** **BIODIVERSITY = ACTUAL**					
Fit statistics – Rubust					
Observations used		251			
Model fit test statistic		4.931			
Degrees of freedom		3			
P value from F test		0.197			
CFI (robust)		0.977			
SRMR		0.015			
RMSEA		0.051			
Lower CI		0.000			
Upper CI		0.107			
		Estimate	**SE**	**Z**	**P**
Actual pollinators					
Noise pollution	a_1	**-0.162**	0.059	-2.788	**0.005**
$PM_{2.5}$	a_4	-0.13	0.334	-0.389	0.697
Years occupancy		-0.14	0.237	-0.593	0.553
NDVI		-1.088	2.785	-0.391	0.696
Noise sensitivity		-0.008	0.064	-0.129	0.898
Social cohesion		-0.148	0.513	-0.288	0.773

Income		0.557	0.519	1.074	0.283
Actual plants					
Noise pollution	a_2	**-0.732**	0.314	-2.329	**0.02**
$PM_{2.5}$	a_5	-0.056	2.212	-0.026	0.98
Years occupancy		0.636	1.134	0.561	0.575
NDVI		13.164	19.402	0.678	0.497
Noise sensitivity		0.385	0.345	1.114	0.265
Social cohesion		2.814	2.559	1.1	0.271
Income		2.227	3.182	0.7	0.484
Actual trees					
Noise pollution	a_3	-0.022	0.221	-0.098	0.922
$PM_{2.5}$	a_6	-0.08	0.871	-0.092	0.927
Years occupancy		-1.048	0.412	-2.546	0.011
NDVI		-7.327	8.526	-0.859	0.39
Noise sensitivity		0.186	0.157	1.185	0.236
Social cohesion		-1.042	1.005	-1.037	0.3
Income		4.805	1.672	2.873	0.004
Happiness					
Bees perceived	b_1	0.036	0.031	1.134	0.257
Plants perceived	b_2	0.002	0.006	0.279	0.78
Trees perceived	b_3	-0.02	0.021	-0.95	0.342
Noise pollution	c_1	0.012	0.019	0.672	0.501
$PM_{2.5}$	c_2	0.087	0.07	1.252	0.21
Years occupancy		0.095	0.091	1.038	0.299
NDVI		-1.121	0.969	-1.157	0.247
Noise sensitivity		-0.052	0.036	-1.44	0.15
Social cohesion		0.336	0.173	1.942	0.052
Income		0.372	0.214	1.735	0.083
Gender		0.054	0.192	0.28	0.779
Mediation					
Noise_pollinators		-0.006	0.005	-1.111	0.266
$PM_{2.5}$_pollinators		-0.005	0.012	-0.374	0.709
Noise_plants		-0.001	0.005	-0.277	0.782
$PM_{2.5}$_plants		0	0.004	-0.026	0.979
Noise_trees		0	0.004	0.097	0.922
$PM_{2.5}$_trees		0.002	0.017	0.093	0.926

Table 7. Parallel mediation model testing the effect of noise and air pollution on SWEMWBS, mediated by perceived richness of pollinators, flowering plants, and trees. Estimate values are standardised β values. Standard error = SE. Bold values indicate statistical significance ($p < 0.05$).

Model 3 OUTCOME = SWEMWBS NOISE = ACTUAL BIODIVERSITY = PERCEIVED	
Fit statistics – Rubust	
Observations used	241
Model fit test statistic	266.55
Degrees of freedom	3

P value from F test		0.908			
CFI (robust)		1			
SRMR		0.004			
RMSEA		0.000			
Lower CI		0.000			
Upper CI		0.019			
		Estimate	**SE**	**Z**	**P**
Perceived pollinators					
Noise pollution	a_1	-0.014	0.015	-0.924	0.356
$PM_{2.5}$	a_4	0.004	0.05	0.084	0.933
Years occupancy		-0.045	0.08	-0.565	0.572
NDVI		-0.483	0.607	-0.797	0.425
Noise sensitivity		0.049	0.038	1.286	0.199
Social cohesion		0.556	0.169	3.296	0.001
Income		0.265	0.119	2.226	0.026
Perceived plants					
Noise pollution	a_2	-0.003	0.014	-0.225	0.822
$PM_{2.5}$	a_5	0.011	0.048	0.224	0.823
Years occupancy		-0.065	0.095	-0.681	0.496
NDVI		0.163	0.587	0.277	0.781
Noise sensitivity		0.03	0.035	0.873	0.383
Social cohesion		0.995	0.169	5.883	0
Income		0.124	0.08	1.549	0.121
Perceived trees					
Noise pollution	a_3	0.013	0.011	1.133	0.257
$PM_{2.5}$	a_6	-0.02	0.036	-0.565	0.572
Years occupancy		-0.067	0.067	-1.007	0.314
NDVI		-0.077	0.342	-0.225	0.822
Noise sensitivity		-0.005	0.021	-0.254	0.8
Social cohesion		0.629	0.112	5.625	0
Income		0.153	0.063	2.449	0.014
SWEMWBS					
Bees perceived	b_1	0.156	0.257	0.608	0.543
Plants perceived	b_2	0.51	0.293	1.743	0.081
Trees perceived	b_3	**-0.566**	0.289	-1.96	**0.05**
Noise pollution	c_1	0.046	0.032	1.426	0.154
$PM_{2.5}$	c_2	0.155	0.158	0.979	0.328
Years occupancy		0.381	0.199	1.917	0.055
NDVI		-2.35	1.692	-1.389	0.165
Noise sensitivity		0.021	0.091	0.231	0.817
Social cohesion		0.448	0.418	1.073	0.283
Income		0.282	0.347	0.813	0.416
Gender		0.136	0.411	0.331	0.741
Mediation					
Noise_pollinators		-0.002	0.005	-0.471	0.638
$PM_{2.5}$_pollinators		0.001	0.008	0.083	0.934
Noise_plants		-0.002	0.007	-0.233	0.816

$PM_{2.5}$_plants	0.005	0.025	0.222	0.825
Noise_trees	-0.007	0.007	-1.108	0.268
$PM_{2.5}$_trees	0.011	0.023	0.508	0.611

Table 8. Parallel mediation model testing the effect of noise and air pollution on happiness, mediated by perceived richness of pollinators, flowering plants, and trees. Estimate values are standardised β values. Standard error = SE. Bold values indicate statistical significance ($p < 0.05$).

Model 4 **OUTCOME = HAPPINESS** **NOISE = ACTUAL** **BIODIVERSITY = PERCEIVED**					
Fit statistics – Rubust					
Observations used		248			
Model fit test statistic		0.421			
Degrees of freedom		3			
P value from F test		0.931			
CFI (robust)		1			
SRMR		0.004			
RMSEA		0.000			
Lower CI		0.000			
Upper CI		0.000			
		Estimate	**SE**	**Z**	**P**
Perceived pollinators					
Noise pollution	a_1	-0.014	0.015	-0.958	0.338
$PM_{2.5}$	a_4	-0.002	0.05	-0.035	0.972
Years occupancy		-0.017	0.083	-0.202	0.84
NDVI		-0.466	0.573	-0.814	0.416
Noise sensitivity		0.041	0.038	1.091	0.275
Social cohesion		0.564	0.172	3.274	0.001
Income		0.231	0.118	1.967	0.049
Perceived plants					
Noise pollution	a_2	-0.003	0.014	-0.235	0.814
$PM_{2.5}$	a_5	0.009	0.046	0.192	0.847
Years occupancy		-0.064	0.092	-0.698	0.485
NDVI		0.184	0.587	0.314	0.754
Noise sensitivity		0.027	0.035	0.764	0.445
Social cohesion		0.994	0.159	6.264	0
Income		0.112	0.082	1.361	0.174
Perceived trees					
Noise pollution	a_3	0.012	0.011	1.114	0.265
$PM_{2.5}$	a_6	-0.022	0.031	-0.725	0.469
Years occupancy		-0.063	0.064	-0.986	0.324
NDVI		-0.069	0.354	-0.196	0.845

Noise sensitivity		-0.009	0.021	-0.433	0.665
Social cohesion		0.631	0.109	5.798	0
Income		0.142	0.062	2.284	0.022
SWEMWBS					
Bees perceived	b_1	-0.03	0.124	-0.242	0.809
Plants perceived	b_2	0.155	0.103	1.507	0.132
Trees perceived	b_3	-0.05	0.145	-0.348	0.728
Noise pollution	c_1	0.007	0.017	0.408	0.683
$PM_{2.5}$	c_2	0.088	0.07	1.259	0.208
Years occupancy		0.113	0.087	1.292	0.196
NDVI		-1.036	0.873	-1.186	0.236
Noise sensitivity		-0.055	0.037	-1.496	0.135
Social cohesion		0.251	0.196	1.285	0.199
Income		0.293	0.152	1.927	0.054
Gender		0.052	0.197	0.262	0.793
Mediation					
Noise_pollinators		0	0.002	0.246	0.806
$PM_{2.5}$_pollinators		0	0.002	0.034	0.973
Noise_plants		0	0.002	-0.249	0.804
$PM_{2.5}$_plants		0.001	0.007	0.199	0.842
Noise_trees		-0.001	0.002	-0.353	0.724
$PM_{2.5}$_trees		0.001	0.003	0.326	0.745

Chapter 4. Discussion

Both air and noise pollution above a certain threshold can be detrimental to the health and psychological well-being of urban residents (WHO, 2006, 2018). On the other hand, biodiversity-rich greenspaces have shown to promote positive well-being outcomes (Nghiem et al., 2021; Young et al., 2020). Furthermore, the presence of greenspaces and green infrastructure can play an important role in reducing pollution and thereby protect urban residents (Barwise and Kumar, 2020; Ow and Ghosh, 2017). This thesis extends previous research on the associations between urban biodiversity, pollution and subjective well-being. Specifically, it enhances our knowledge regarding the role biodiversity might play in mitigating the increasingly recognised negative well-being outcomes associated with air and noise pollution exposure, with implications for urban policy and planning. For the first time to my knowledge, this research integrates both actual and perceived measures of taxa richness (flowering plants, pollinators, trees) into sophisticated parallel mediation models capable of examining potential simultaneous effects. The results revealed that greater actual flowering plant richness reduced the negative impact of noise pollution on resident's self-reported well-being across streetscapes. Interestingly, people's perceptions of the richness of different taxa did not mediate the relationship.

Increased actual flowering plant species richness had a significant and direct positive effect on resident's self-reported well-being across streetscapes. This is consistent with Chalmin-Pui et al. (2021), who found that adding ornamental plants to bare front gardens positively influenced the biochemical regulation of cortisol (stress hormone), reduced perceived stress levels improved motivation and a sense of place in residents. However, SWEMWBS scores did not improve, which is in discord to findings presented here. A greater presence of plant diversity has also been shown to enhance the restorativeness of allotments for gardeners (Young et al., 2020), and increased happiness and well-being levels of public greenspace visitors (Adjei and Agyei, 2015; Fuller et al., 2007).

In this study, social cohesion was a significant predictor of perceived species richness for all three taxa. This might be because a greater familiarity of streetscape biodiversity comes with spending more time within the streetscape itself, while socialising with neighbours. Indeed, existing research has found that species richness perception accuracy can increase alongside measured levels of eco-centricity (i.e., those who were more accurate were more connected to nature), as observed in urban meadows (Hoyle et al., 2018). Importantly, increased social cohesion requires long-term social interaction compared to a simple nodding acquaintance (Kawachi and Berkman, 2001). It is therefore possible that an individual who perceives greater levels of species richness within their streetscape is more inclined to spend more time socialising within that environment, as was found to be the case in the Netherlands (de Vries et al., 2013). Wang et al. (2021) also found that streetscape greenspace quality, rather than quantity, positively influenced mental well-being through building capacities, such as better levels of activity and social cohesion. These findings imply that more biodiverse streetscapes could contribute to the improved mental wellbeing of people via multiple biopsychosocial pathways (Hartig et al., 2014).

The results showed that the well-being measures used (happiness and SWEMWBS) were not directly influenced by actual tree and pollinator richness. There is a growing recognition of the role in which eye-level greenspace perspectives play on well-being (Wang et al., 2021). Eye-level greenspace has received far less attention than that of the over-head perspective (e.g., NDVI, LiDAR), mostly due to methodological limitations (Lachowycz and Jones, 2013; Markevych et al., 2017). Tree canopy is not necessarily always at eye-level, particularly through windows when inside the home and on the ground level. Contrary to this, Wolf et al. (2017) found that increased tree richness had a positive effect on various self-reported well-being measures. However, the study was an ex-situ video-based exercise that depicted trees in their entirety.

Participant estimates of pollinator richness did not correlate with actual richness in streetscapes, although perceived estimates of trees and flowering plants were a better reflection of actual richness. Dallimer et al. (2012) found that people could not correctly identify plant or butterfly richness within public urban greenspaces. This disparity between our study and Dallimer et al. (2012) may be due to the familiarity people have with their own streetscapes. Nonetheless, findings here support existing evidence of the public's limited ability to estimate pollinator richness, regardless of setting. The popularisation of the 'save the bees' campaign has helped to garner support and educate the public about the importance of bees at a macro-scale, highlighting their importance in food security (Dicks, 2019). However, this rarely translates to an understanding of diversity in bee species or other kinds of pollinators (Wilson et al., 2017).

When asked about the noises people hear in their streetscapes, answers pertaining to road vehicles/traffic were most common across all three of the pollution categories. This reflects the fact that, within the UK, noise pollution is primarily derived from road surface transport and is a core component of people's immediate soundscapes (United Kingdom Department for Business, Energy and Industrial Strategy, 2018, p14). Birds were the second most common noise heard, and this decreased in frequency as the pollution category of the streetscape increased. Birds were not a taxonomic focus of this study. However, given their noticeable presence, it is possible that they could exert some mediating effect on the relationship between pollution and well-being. Indeed, unsurprisingly, the public consider natural sounds like bird song to be favourable compared to road traffic (Viollon et al., 2002). Bird song has also been found to be better at reducing human stress than other natural sounds, such as running water or non-avian animals (Ratcliffe et al., 2013), and contributes to positive values associated with urban green space (Hedblom et al., 2014). However, when Fuller et al. (2007) examined the associations between psychological benefits and plant, bird and butterfly species richness, they found stronger support for an effect of the number of plant species than for birds.

The ability of vegetation to buffer against noise is improved when planting patterns include species with thick stems, or dense and complex leaf structure, better able to refract sound (Fang and Ling, 2005; Ow and Ghosh, 2017). Reductions in noise levels of up to 15 dB can be achieved (Tyagi et al., 2006). However, much of the existing literature has concentrated on the attenuation of noise via wide swathes of vegetation (e.g., 30 m), which are unrealistic and impractical in many urban settings. Nonetheless, some studies have found that far narrower vegetation belts can still produce significant reductions in noise. For example, vegetation heights of 3 and 5 m comprised of coniferous species have shown to attenuate noise from between 5 and 11 dB (Kragh, 1981; Ow and Ghosh, 2017). To sufficiently reduce noise pollution in urban settings, narrow vegetation belts can be complemented by a setback (the distance between a building/structure and a road) comprised of sparse to medium planting regimes (Ow and Ghosh, 2017). The most recorded plant species across the streetscapes in this study were those with highly dense, evergreen foliage, such as holly (*Ilex aquifolium*), garden privet (*Ligustrum ovalifolium*) and Leyland cypress (*Cupressus* × *leylandii*). Our results showed that greater flowering plant richness reduced the negative impact of noise pollution on resident's self-reported well-being across streetscapes, therefore highlighting the need to incorporate diverse planting regimes into future urban planning. This maps onto existing WHO guidelines that strongly advise that noise pollution derived from road transport be reduced to a maximum of 53 dB in the daytime and 45 dB in the evening, as traffic noise beyond this has proven to reduce health (WHO, 2018). Nevertheless, levels recorded in this study averaged 82.4 dB, with the maximum reaching 93.6 dB. Therefore, even the lowest recorded noise level (67.5 dB) within this study far exceeded WHO recommendations. Tee species richness did not impact noise levels. Several studies suggest that while noise is absorbed by tree trunks, other parts of the tree such as foliage, small stems and branches (i.e., the canopy) can be equally as important for noise attenuation (Fang and Ling, 2005). Noise recordings in this study were taken at breast height and therefore did not capture the noise attenuation ability of tree canopy, which may explain why tree richness did not have the same mediating effects on noise pollution as flowering plants. However, this is speculation as vegetation structure was not explored specifically in this study, but could be a potentially fruitful avenue for future research. Moreover, the residents are unlikely to plant more than a single row of trees in their garden due to space constraints and the desire to maintain natural light in the home, thereby limiting their noise reduction potential at a streetscape scale.

Actual flowering plant and pollinator richness decreased as noise pollution increased in this study. This could be because residents are deterred from gardening in streetscapes where noise pollution is high. Plant diversity may therefore be less in these areas and, by extension, the pollinators which rely on them (Baldock et al., 2019). Noise pollution can also have a direct negative impact on pollinator species. In comparison to light and air pollution, there is much less research into the effects of noise pollution on pollinators (Morley et al., 2013). However, the vastly diverse mix of auditory structures found across invertebrate species mean that hearing sensitivities can sit anywhere between 10 Hz to more than 100 kHz. Many invertebrates specifically communicate at frequencies below 10 kHz, which is well within the frequency spectrum of most anthropogenic noise (Morley et al., 2013). The vulnerability of pollinators to this pollutant is therefore clear and is supported by the small amount of literature available, which demonstrates impairments from noise to survival mechanisms and behaviours (e.g., Davis et al., 2018; Lampe et al., 2012). Improving urban habitat to intensify insect conservation is an urgent challenge in the face of global mass invertebrate decline (Cardoso et al., 2020; Hallmann et al., 2017). Diversely planted streetscape vegetation has the potential to buffer against noise pollution while also providing pollinators with higher quality and better-connected habitat.

Air pollution did not have a direct effect on well-being measures (SWEMWBS and happiness) across the streetscapes. Unlike noise pollution, air pollution ranges did not exceed those recommended by the WHO (> 10 µg/m3 annual mean; > 25 µg/m3 24 hour mean) (WHO, 2006). However, localised pollution events were recorded on a small number of occasions, where $PM_{2.5}$ concentrations did exceed 10 µg/m3. This may have been due to traffic associated with the time of day in which recordings were taken, or other *ad hoc* occurrences such as temporary traffic lights/road works.

4.1 The Covid-19 pandemic

Governments around the world were forced to enact policy measures to curtail transmission of the novel coronavirus SARS-CoV-2 following the World Health Organisation’s declaration of a global pandemic on 11th March 2020 (WHO, 2020). Restrictions were placed on transport and public mobility, resulting in over half the global population living under stay-at-home orders (with varying degrees of stringency) for one or more time-periods. These stay-at-home measures were typically coupled with social distancing orders. Although data collection for this study took place prior to the pandemic, several key points are worth raising that are pertinent to the research topic.

Firstly, a considerable proportion of workers swapped their office-based working environment for homeworking during the pandemic to comply with lockdown/social distancing measures (Bartik et al., 2020). Even as government restrictions begin to lift, many companies believe that working from home is set to become the long-term norm, as the upfront costs associated with remote systems have already been paid for and there are financial benefits associated with cutting back on office space (Bartik et al., 2020). As society adapts, understanding how the home as a working environment can be modified to enhance well-being, such as reducing noise pollution, is set to become a priority (Bouziri et al., 2020; Xiao et al., 2021). Secondly, when outdoor interaction with greenspace is impeded, viewing greenery through the window can be an important act of engagement to promote mental health. Being in view of greenspace through the window has shown to trigger micro-restorative episodes that promote healing (Jo et al., 2019), mental restoration (Lee et al., 2015) and stress recovery (Li and Sullivan, 2016) after a matter of hours or days. Over longer time periods, green window views have shown to elevate an individual's capacity to complete cognitive tasks and improve self-reported life and job satisfaction (Benfield et al., 2015; Chang et al., 2020; Shin, 2007). Indeed, during the pandemic, Wang et al. (2021) observed reduced depression and anxiety in home-schooled students who had window views with greater plant abundances.

Working from home may negatively impact an individual's sense of social cohesion as it can limit the opportunity to socialise with colleagues, which can be particularly detrimental to those who live alone (Tavares, 2017). Moreover, blurred boundaries between work and home life can make it difficult to recover from mental fatigue (Evanoff et al., 2020; Vander Elst et al., 2017). Several papers have highlighted the negative implications of prolonged stay-at-home measures on mental well-being outcomes (e.g., generalised anxiety disorder, major depressive disorder, loneliness) across countries including Canada, Greece and China (Dozois, 2021; Fountoulakis et al., 2021; Huang and Zhao, 2020). Socialising with those close to the home (e.g., neighbours) could be an increasingly crucial point of contact for many home-workers. This adds weight to concerns over the inequalities surrounding front garden access (ONS, 2018). Overall, the context of the pandemic adds weight to the importance of considering streetscapes as a spatial scale and biodiversity within future urban planning regimes.

4.2 Limitations

There are several limitations to this study that may have introduced bias into the results. For example, compared to some other research of this nature, the overall sample size is relatively small. Streetscapes with fewer than five completed questionnaires were removed from analyses, which included two low and one medium pollution category streetscapes. The number of streetscapes across categories was therefore disproportionate, with the greatest amount of representation falling within the high pollution category. All possible efforts were made to remain impartial while the questionnaires were being delivered, so to avoid influencing questionnaire responses. However, given that questionnaires were administered face-to-face, it cannot be ruled out that some individuals may have tried to respond in a manner that they thought the researcher might expect or prefer them to, regardless of how speculative this would have been on the respondent's part (Bowling, 2005).

There may also be some limitation to the study design itself. The categorisation of pollution levels based on road type may have introduced some bias due to the diffuse nature of particulate matter. It is possible that roads close-by to an A road, despite being classed as low pollution due to road width, might experience much higher pollution levels than you would expect as a by-product of the nearby A road. Indeed, some of the low pollution streetscapes experienced $PM_{2.5}$ levels higher than those recorded in high pollution streetscapes. As previously mentioned above, pollution readings were collected at breast height and therefore would have failed to capture the full extent of pollution reduction by trees.

People's perceptions of species richness have already been successfully utilised within the nature-health literature (e.g., Fuller et al., 2007; Dallimer et al., 2012; Fisher et al., 2021). However, it is worth bearing in mind that there are several factors that may influence people's perceptions. For example, an individual's eco-centricity (including ecological knowledge and pro-environmental behaviour) can positively influence identification skills (Southon et al., 2018) which, in turn, can affect an individual's ability to estimate the species richness of a given area (Dallimer et al., 2012). Research has also shown that amongst the general public, even those who value nature are unable to detect changes in species richness across parks and gardens (Schwartz et al., 2014). The accuracy of people's perceptions might also vary depending on the taxonomic group. For instance, Dallimer et al. (2012) found that participants were better able to identify bird species than either butterflies or plants. The amount of time an individual has spent living in an urban area may also impact people's perceptions of species richness and pollution. For example, an individual who has only ever resided in an area of high pollution may not perceive it to be highly polluted as they are accustomed to such levels. On the other hand, an individual who might have spent some time living rurally would presumably hold a different baseline for what they consider to be of high or low pollution, and this was not controlled for within the model. Perceptions may also be influenced by an individual's sensory system. Quantitative alterations in olfactory and auditory function, including impaired and enhanced performance, may distort the level of air and noise pollution experienced. In addition, anosmia (the loss or erosion of the ability to smell) can be caused by air pollution (Hudson et al., 2006; Guarneros et al., 2009; Zhang et al., 2021). Therefore, people living in highly pollution areas may over time experience decreased olfactory sensitivity, thus potentially distorting their perception of pollution.

5. Conclusion

This thesis provides a valuable insight into the complex interplay between streetscape pollution, biodiversity and human mental well-being, which until now remained largely unexamined. Given the findings presented here, governments, city planners and urban residents themselves ought to take action to reduce noise pollution across streetscapes to improve mental well-being. Increasing plant diversity should be considered a priority within any future plans to reduce urban noise. In turn, critical gains in pollinator conservation can also be made. Although recent years have seen the public take a greater interest in biodiversity and, specifically, the vital role played by pollinators, this does not necessarily always translate into conservation action (Wilson et al., 2017). In part, this could be attributed to a lack of detailed ecological knowledge, or failure to see what real-time benefits could be gained. As such, governments and local councils could offer fiscal incentives, coupled with educational ecological and pollution-wellbeing related guidance, as one way to increase planting. Additionally, more should be done to reduce noise pollution at the source. For example, by re-routing busy traffic which passes through residential areas, or by encouraging and improving public transport to minimise the quantity of vehicles on the road. Trends in population growth, increased pollution and declines in mental health that characterise cities worldwide, mean that the findings presented here are applicable to a multitude of urban settings. Additionally, this study offers particularly timely insights given that people are projected to spend more time at home within the coming years.

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Supplementary Information

Supplementary Information S1. The 20-morpho-functional groups of pollinators recorded across the 30 streetscape transects in Leeds, UK. Bumblebees (*Bombus* spp.) and butterflies constitute six and five morpho functional groups, respectively.

Morpho-functional group	Notes
Bumblebees (*Bombus* spp.):	
2 yellow bands, white/buff tail: *B.terrestris/B.lucorum*	
2 yellow bands, red tail: *B.pratorum*	
3 yellow bands, white tail: *B.hortorum/B.ruderatus (B.ruderatus* maybe melanic)	Group including all individuals belonging to the genus *Bombus*
Brown/Ginger: *B.pascuorum/B.humulis/B.muscorum*	
Black, red tail: *B.lapidarius/B.ruderarius*	
Tree bumblebee (*B. hypnorum*)	
Honeybee	*Apis mellifera*
Solitary bee	Group including bees from the Apoidea super-family except those from genus *Bombus* and *Apis mellifera*
Social wasp	Social wasps in the sub-family Vespinae
Other hymenoptera	All other hymenoptera, including solitary wasps (e.g. ichneumonids) and sawflies
Hoverfly	Group including all individuals from the Syrphidae family (Diptera)
Other fly	Group including all non-Syrphidae Diptera
Butterflies:	
White	Group including all white species from the family Pieridae
Red	Group including the red species from the family Nymphalidae; *Lycaena* spp.
Brown	Group including the brown/orange species from the family Nymphalidae
Blue	Group including the blue species from the family Lycaenidae
Yellow	Group including the yellow species from the family Pieridae
Moth	Day-flying moth species (*Tyria jacobaeae*; *Zygaena* spp.)
Beetle	All insects in the order Coleoptera
Bug	All insects in the order Hemiptera

Supplementary Information S2. Research questionnaire used to collect participant data, administered in 30 streetscapes across the city of Leeds, UK.

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Your street environment and quality of life questionnaire

This questionnaire explores your views about your street environment and your quality of life.

Please think about the street that your home is on. This includes everything you can see from your home to the opposite side of the street, such as gardens (including if they are paved over), parking spaces, yards, pavements, the road, trees, bushes or shrubs in the street, and verges and the plants and flowers that are on them, including any insects, birds or animals that might be there.

There are no right or wrong answers as we are interested in what you think

1. How would you best describe the area outside your home that faces the street environment that is still part of your property?
Tick all that apply

☐ There is no area outside my home that faces the street that is part of my property
☐ Somewhere to park a car
☐ An area of garden with lawn, flowers, shrubs, trees (or some of these)
☐ A gravel area
☐ A yard covered by concrete, paving or decking
☐ Other (Please state): ____________________________

2. In general when thinking about the sounds in your street environment, how would you describe them?
Circle one option per row

Completely unacceptable	Unacceptable	Neutral	Acceptable	Completely acceptable
Very quiet	Quiet	Neutral	Noisy	Very noisy
Not at all annoying	Not annoying	Neutral	Annoying	Very annoying
Very relaxing	Relaxing	Neutral	Stressful	Very stressful

3. What three main sounds do you hear in your street environment?
Please write in up to three sounds and tick one box per sound

Sounds I hear	Very unpleasant	Unpleasant	Neutral	Pleasant	Very pleasant
1					
2					
3					

1

1824

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4. In general how would you describe the level of air pollution in your street environment?
Circle one option

Very polluted	Polluted	Neutral	Clean	Very clean

5. In general, how would you describe the acceptability of air pollution in your street environment?
Circle one option

Completely unacceptable	Unacceptable	Neutral	Acceptable	Completely acceptable

6. About how many different species of tree would you say are in your street environment?
Circle one option

Fewer than 5	5 to 10	11 to 15	16 to 20	21 or more

7. About how many different species of flowering plants would you say are in your street environment?
Circle one option

Fewer than 10	10 to 30	31 to 50	51 to 99	100 or more

8. About how many different species of bees, butterflies and other insects that visit flowers would you say are in your street environment?
Circle one option

Fewer than 5	5 to 9	10 to 13	14 to 19	20 or more

9. To what extent do you notice nature out the windows of your home that face your street environment?
Circle one option

Not at all	Rarely	Sometimes	Often	Very Often

10. How much of the view out of the windows that face your street environment is of nature, such as plants, trees, bushes, flowers, grass?
Circle one option

None	Less than a quarter	Between a quarter and a half	Between a half and three quarters	More than three quarters	All

2

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11. Please indicate how much you agree with each statement about your street environment.
Tick one box per row

	Strongly disagree	Disagree	Neutral	Agree	Strongly agree
I look forward to coming to my street					
My street reflects the type of person I am					
Lots of things in my street remind me of past experiences					
My street is a desirable place to be					
My street feels almost like a part of me					
I have had a lot of memorable experiences in my street					
When I am in my street I feel that I belong there					
I am proud of my street					
I really miss my street when I am away from it for a long time					
I feel happy when I am in my street					
I gain pleasure from using my street					
I like my street					
My street has many advantages compared to other streets					
I am not satisfied with my street					

1826

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12. Please indicate how much you agree with each statement about your street environment.
Tick one box per row

(a) When I am out of doors in my street environment…

	Strongly disagree	Disagree	Neutral	Agree	Strongly agree
I can easily think about personal matters					
I gain perspective on life					
I have time to listen to what is on my mind					

(b) When I look at my street environment out of a window…

	Strongly disagree	Disagree	Neutral	Agree	Strongly agree
I can easily think about personal matters					
I gain perspective on life					
I have time to listen to what is on my mind					

13. On how many of the last 14 days did you do each of the following in your street environment? *Your street environment includes your garden/yard, pavements, trees, any plants, grass verges and road*
Tick one box per row

	None	1 or 2	2 or 3	3 or 4	5 or 6	7 or 8	9 or 10	11 or 12	13 or 14
Relax									
Observe nature									
Play games / sports									
Think about things									
Park a car/vehicle									
Walk or cycle to work, school, the shops, or another destination									
Walk, cycle or run/jog for exercise or enjoyment									
Tend plants, trees, shrubs or lawn/grass									
Talk to other people									
Other (please state): ________________									
Other (please state): ________________									

4

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14. Please indicate how much you agree or disagree with the following statements.
Tick one box per row.

	Strongly disagree	Disagree	Neutral	Agree	Strongly agree
People on this street are willing to help their neighbours					
This is a close-knit street					
People on this street can be trusted					
People on this street generally don't get along with each other					
People on this street do not share the same values					

The next questions are about you

15. In general, would you agree or disagree with the statement: Spending time out of doors is an important part of my life.
By 'out of doors' we mean open spaces in and around towns and cities, including parks, canals and nature areas; the coast and beaches; and the countryside including farmland, woodland, hills and rivers. It includes time spent close to your home, workplace, further afield or while on holiday in Britain.

Circle one option

Strongly agree   Agree   Neutral   Disagree   Strongly disagree

16. In your life in general, how happy would you say you are?
Circle one number

Extremely unhappy ______________________________ Extremely happy

1   2   3   4   5   6   7   8   9   10

17. Over the past 14 days to what extent have you felt:
Tick one box per row

	Not at all	A little	Somewhat	Quite	Very much
Irritated					
Tired					
Worn out					
Mentally exhausted					

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18. Below are some statements about feelings and thoughts. Please tick the box that best describes your experience of each over the last two weeks.
Tick one box per row

	None of the time	Rarely	Some of the time	Often	All of the time
I've been feeling optimistic about the future					
I've been feeling useful					
I've been feeling relaxed					
I've been dealing with problems well					
I've been thinking clearly					
I've been feeling close to other people					
I've been able to make up my own mind about things					

19. Please indicate how much you agree with the following statements.
Tick one box per row

	Strongly disagree	Disagree	Neutral	Agree	Strongly agree
I get irritated when there is noise in my street					
I am good at concentrating no matter what happens around me					
It is difficult for me to relax in a noisy place					
I am sensitive to noise					

20. On how many of the last 14 days did you do the following?
Tick one box per row

	None	1 or 2	2 or 3	3 or 4	4 or 5	6 or 7	8 or 9	10 or 11	13 or 14
Visit local parks or the countryside									
Spend time in your back garden (e.g. gardening, sitting, socialising, playing)									

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21. About your home:

a. Do you: ☐ Own ☐ Rent

b. How long have you lived in your current home? _______ Years ______ Months

c. How many bedrooms does your home have? ______________

d. How many people living at this home are:

Under 16 years _________ Over 16 years __________

22. What year were you born in? ________________ ☐ Prefer not to say

23. What gender do you identify with?
Tick one option

☐ Male ☐ Female ☐ Prefer not to say

24. How would you describe your ethnic group or background?
Tick one option

- ☐ White English/British/Scottish/Welsh
- ☐ White (other background)
- ☐ Any Asian background
- ☐ Any black background
- ☐ Any other ethnic background: _____________
- ☐ Prefer not to say

25. Which of the following best describes your employment status over the last 12 months?
Tick one option

- ☐ Employed
- ☐ Self-employed
- ☐ Unemployed
- ☐ Retired
- ☐ Prefer not to say
- ☐ Long-term sick/disabled
- ☐ Looking after home/family
- ☐ Student
- ☐ Other: _________________

26. Do you have a disability? Under the Equality Act 2010, a person has a disability if they have a physical or mental impairment, and the impairment has a substantial and long-term adverse effect on his or her ability to carry out normal day-to-day activities.
Tick one option

☐ Yes ☐ No ☐ Prefer not to say

☺ ☺ Thank you for taking the time to complete this questionnaire ☺ ☺