# Environmental Justice and Private Urban Gardens: A Critical Analysis of Practices in Salford, Greater Manchester

Doctor of Philosophy

University of Salford

Katherine Louise Scaletta

## Abstract

While it is well accepted that the quality of a greenspace determines its ability to provide health and wellbeing benefits, research and policy overlook the spaces which residents have their most direct and frequent contact with nature – urban household gardens. In the UK, 85 per cent of households have access to a garden; but, there is a disparity in the amount of vegetation and biodiversity found in these spaces. This imbalance needs addressing. Missing in the literature is a detailed account of household gardens, exposing the unequal opportunities different socio-economic groups have within their gardens.

The novel use of Google Earth to collect household garden data overcame well-documented issues of accessing gardens. Data were collected on size and landcover using five categories: plants, shrubs, trees, lawn, and paved areas (resolution ≤1m) for 6881 gardens across Salford, Greater Manchester. An economic analysis was undertaken to compare the value of gardens to public greenspace and perform a cost-benefit analysis for residents.

Mean garden size in the City of Salford was 139 $m^2$, smaller than previously reported, with a configuration of approximately 80 per cent paved and lawn combined and 20 per cent vegetation. Mean feature diversity in gardens was found to vary significantly between socio-economic groups suggesting some receive far greater health and wellbeing benefits from their higher quality gardens.

This work, along with the valuation of gardens, contributed to the Urban Pioneer – a national DEFRA initiative – directly advocating the value of gardens to key stakeholders and the need for equal access to quality greenspaces. The overall aim was to undertake a critical exploration and analysis of the value of private gardens, with a key focus on the City of Salford, evaluating the size and content of urban household gardens to determine factors which contribute to the heterogeneity of the resource and their associated benefits. Resulting in four research objectives and three research questions which are outlined below.

# List of Contents:

## List of Figures:

## List of Tables:

# Acknowledgement:

My greatest acknowledgement and deepest gratitude must go to Prof. Philip James without whom I would never have believed I could do this. His support, knowledge, and work ethic have inspired my academic career and will continue to do so. Philip, you have taught me a great deal and I remain deeply grateful for your friendship.

I would like to give great thanks to Dr. Michael Hardman for remaining my supervisor for seven years and seeing this journey through to the end. I am also incredibly grateful for his flexibility and expertise; I feel I have learnt a lot under his supervision.

I have had the pleasure to collaborate with Prof. Richard Armitage and Dr. Matthew Dennis, for which I am thankful. Their research knowledge and creative thinking inspired me to expand into other areas of the field.

Many thanks to my family who have supported me for so long. To my mum for her unwavering support and encouragement, for her understanding and commitment to help, and in the latter years for her child-minding services. To my dad for his financial support and his ‘proud daddy’ talks which kept me motivated.

To James who has ridden this rollercoaster with me for seven years, sick bag in hand when I needed him every step of the way. And finally to my son, Sylas, who has in equal parts made it harder to complete but more worthwhile in doing so.

Thank you all and thank you to the reader for taking the time to read my work.

# Chapter One: Introduction

## 1.1 Environmental justice in urban greenspaces

A hallmark of the twenty first century's urban landscape will be the geographic proximity of the concentration of affluence and poverty (Harris, 2019). A prediction by Douglas Massey states 'just as poverty is concentrated spatially, anything correlated with poverty is also concentrated' (Massey, 1996, pg. 407). Within urban areas, economically poorer communities will be increasingly confined to and concentrated in economically and environmentally poorer neighbourhoods, resulting in an unprecedented density of material deprivation (Ainamani, Gumisiriza, Bamwerinde, & Rukundo, 2022; Harris, 2019). Whilst these more historic perspectives focus on poverty, the term inequality, defined as the unequal and/or unjust distribution of resources and opportunities among members of a given society (Koh, 2019), is a broader concept of injustice. In contemporary research this inequality has been termed environmental injustice, a social movement disproportionately affecting disadvantages communities, as is discussed below.

The Environmental Justice Movement emerged in North Carolina, USA in the early 1980s, due to a local dispute over toxic waste dumping near a neighbourhood of African-American people (Lehtinen, 2009). Since then, understanding of environmental justice has broadened from the original concern with pollution to become a community approach that is focused on the central role of socio-environmental networks and co-associations, both local and trans-local, in developing the ideas and relations that generate what is considered just and acceptable (Lehtinen, 2009). In parallel to the Environmental Justice Movement, the concept of just sustainabilities also grew. Within the environmental realm this concept is largely accredited to Julian Agyeman who was inspired by the more far reaching book published by Wilkinson and Pickett (2009), The Spirit Level: Why Equality is Better for Everyone (Agyeman, 2013). Wilkinson and Pickett (2009) argue many health and social problems ranging from medicine, education, and crime are most common in disadvantaged areas of society and are more common in unequal societies. Further, Agyeman (2013) introduces and highlights the need for just sustainabilities within the realm of environmental research. He defined just sustainabilities as 'The need to ensure a better quality of life for all, now and into the future, in a just and equitable manner, whilst living within the limits of supporting ecosystems' (Agyeman, 2013, p. 7). There is a need for environmental benefits to be evenly accessible to all, similarly to environmental injustice, but, equally important within just sustainabilities is the need for a sustainable approach in doing so.

Today, environmental justice covers a range of aspects, from the uneven distribution of pollutant emitting facilities to the quantity and quality of greenspace. Within the literature, it has been demonstrated that, compared with more wealthy communities, poorer and ethnic minority communities are often exposed to higher levels of air pollution and below average air quality (Gurgatz et al., 2016; Vaz, Anthony, & McHenry, 2017; Hernandez, Collins, & Grineski, 2015; Li, Han, Lam, Zhu, & Bacon-Shone, 2018) and often have less access to quality urban greenspaces (Apparicio, Pham, Seguin, & Dube, 2016; Burt, Feng, Mavoa, Badland & Corti, 2014; Wustemann, Kalisch, & Kolbe, 2017). From an urban planning perspective, there is a lack of representation of ethnic minority members, for example in 2015 only 6.3 per cent of architects and town planners in the UK were from minority backgrounds (Rishbeth, Ganji & Vodicka, 2017), illustrating a lack of representation and cultural diversity in urban planning. There is a need to understand the multiple, multi-directional pathways through which socio-environmental variables influence the provision of urban ecosystem services and associated benefits which, in turn, influence personal health and wellbeing (Bagstad et al., 2013; Wilkerson et al., 2018). Ecosystem services are:

> 'the benefits that people obtain from ecosystems. They sustain and fulfil human life…' (Tallis, Guerry, & Daily, 2013).

There exists a complex, multi-directional relationship between the biophysical supply of ecosystem services, the demand for services by people, and the benefits people receive from these services, as illustrated in Figure 1.1 (Wilkerson et al., 2018). The linkages in Figure 1.1 show that socio-environmental factors within towns and cities affect ecosystem services by, among other things, both influencing the management of greenspaces, and, in turn, their service supply, and, by altering a person's needs and activities, their demands. In addition, along each of these pathways, ecosystem services can feed-back to influence socio-environmental variables, for example greener communities having higher house prices (Jim & Chen, 2006; Kumagai & Yamada, 2008; Larson & Perrings, 2013). This situation is further exacerbated as those communities which are more socially disadvantaged, in addition to often having less greenspace provision, suffer greater from population pressure, and there are often fewer opportunities to travel out of the area to enjoy the environment (Apparicio, Pham, Seguin, & Dube, 2016; Jones, Hillsdon, & Coombes, 2009; Mears et al., 2019; Wustemann, Kalisch, & Kolbe, 2017). Evidence suggests those in disadvantaged socio-economic classes are more in need and more sensitive to the positive effects the natural environment can have on health and wellbeing (James, et al., 2015; Maas et al., 2006). Therefore, as discussed above, provision of and access to urban greenspaces which promote good health may be an important pathway to environmental justice (Mitchel & Popham, 2008).

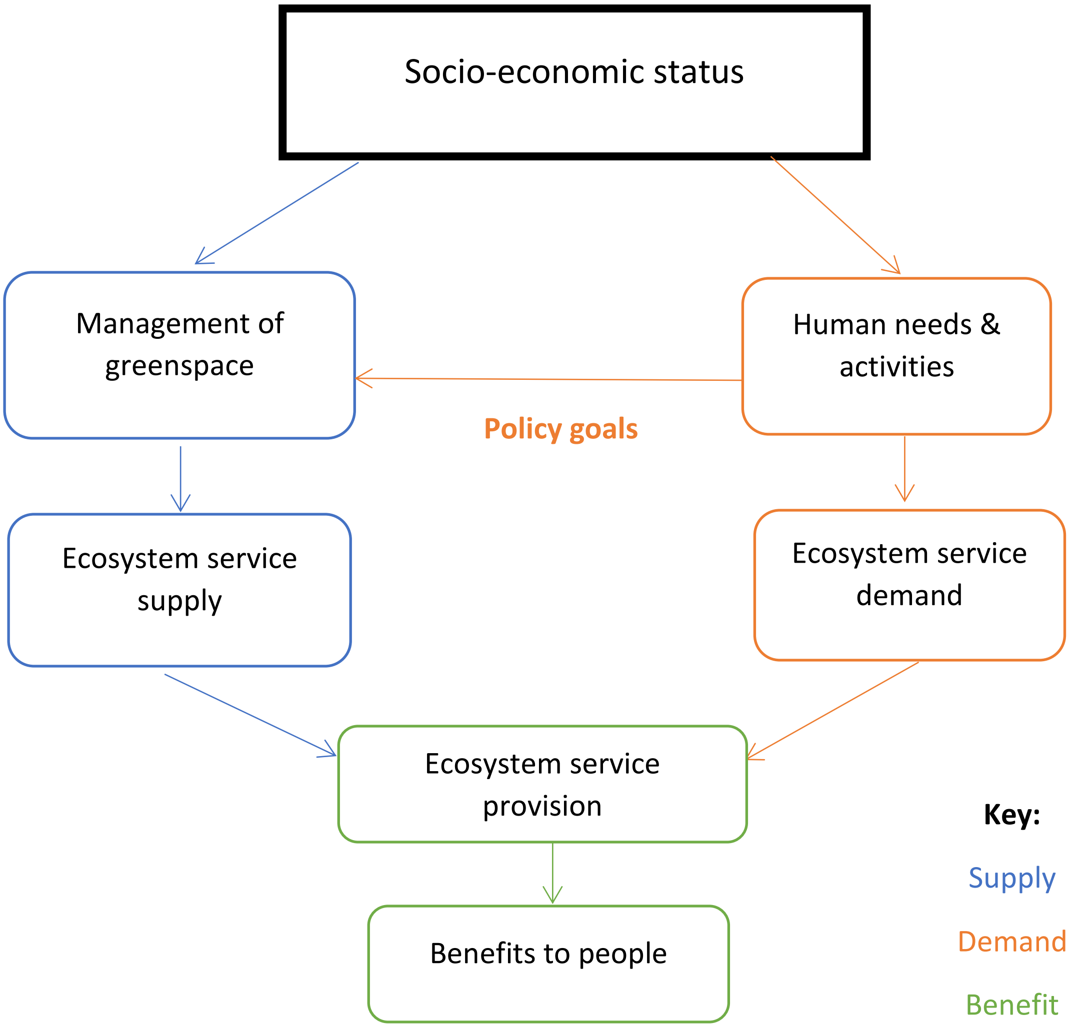



Figure 1.1: An illustration of how socio-economic status affects the flow of ecosystem services in an urban socio-environmental system (adapted from Wilkerson et al., 2018).

Urban green infrastructure (UGI) is defined as:

'urban areas which consist of vegetated green surfaces, such as parks, trees and small forests, grasslands, but also private gardens or cemeteries' (European Environment Agency, 2021).

While, currently, research into how UGI can benefit people is focused on public spaces (Murkin, Shiode, Shiode, & Kidd, 2023) there is little published material that considers another important socio-environmental variable associated with the provision of green space – the private, domestic garden. Household gardens, defined as 'the area adjacent to a domestic dwelling, which itself is either owned or rented. A key element is that the resident/s have autonomy over the garden…' (Cameron et al., 2012, p. 4), are often an urban resident's most frequent contact with the natural world (Goddard et al., 2013). Due to the private, small-scale character of household gardens they have often lain beyond the scope of land use statistics, spatial and green structure planning, and many environmental policies, being included less frequently than public space (Dewaelheyns, Rogge, & Gulinck, 2014; Perry & Nawaz, 2008; Samus, Freeman, Dickinson, & van Heezik, 2022).

Socio-economic factors have been shown to affect the vegetation cover within private urban greenspace. Grove et al. (2006) found a combination of both house age and lifestyle behaviour variables were the best predictors of both private land tree (pseudo-$R^2$ = 0.34) and private land grass (pseudo-$R^2$ = 0.32) cover in Baltimore, Maryland. Vegetation cover has been shown to be similarly positively correlated to socio-economic status (Kendal, Williams, & Williams, 2012; Shanahan, Lin, Gaston, Bush, & Fuller, 2014), as well as with plant species richness (Chamberlain, Henry, Reynolds, Caprio, & Amar, 2019; van Heezik, Freeman, Porter, & Dickinson, 2013), and plant diversity (Bigirimana, Bogaert, De Canniere, Bigendako, & Parmentier, 2012; Chamberlain et al., 2019; Luck, Smallbone, & O'Brian, 2009). There is a growing body of evidence to suggest urban greenspace can contribute to the mitigation of health inequalities often experienced by socially disadvantaged neighbourhoods (Brown et al., 2018; Maas et al., 2009; Mears et al., 2019) as well as reduce social isolation (Howarth, Rogers, Whithnell, McQuarrie, 2019). Maxwell and Lovell (2017) concluded in an evidence statement for DEFRA (Department for Environment, Food, & Rural Affairs) that biodiversity is critical to the delivery of ecosystem goods and services, in particular cultural services, essential to human health and wellbeing. Cultural ecosystem services are defined as 'the non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences' (Church, Burgess, & Ravensoft, 2011, p. 639). Therefore, equal use and access to these greenspaces is paramount and research within the field has grown over the last 20 years (Howarth, Lawler, & de Silva, 2021).

In the late 2000s research conducted by Davies et al. (2009) found 87 per cent of households in the UK have access to a household garden. Within the same decade, household gardens were reported to contribute between 22–36 per cent of the total urban area of cities across the UK (Gaston, Warren, Thompson, & Smith, 2005; Mathieu, Freeman, & Aryal, 2007), illustrating their considerable collective geographic scope. Since then, in the city of Manchester, UK, Baker, Smith and Cavan (2018) found approximately 21 per cent of UGI was gardens, and within these gardens only 50 per cent was green, permeable land cover. Policy, such as the Green and Blue Infrastructure Strategy for Manchester, has previously assumed all garden space was green cover and therefore permeable, so given this overrepresentation there is a need for contemporary data to correct it and give a more accurate image of urban ecosystem services (Baker, Smith, & Cavan, 2018). Crucially, there is a need for a more extensive and up-to-date account of garden extent within a city which has a more varied landscape than that of the city of Manchester. Due to their ease of access and large collective area, household gardens are an important UGI and ought to be considered alongside publicly owned and managed green space when looking at the socio-environmental system of a town or city.

There is a gap within environmental injustice research to further investigate the affect socio-economics may have on the vegetation within a private garden. As there is strong evidence for a positive relationship between socio-economics and vegetation cover within public and private land (Kendal, Williams, & Williams; 2012; Shanahan, Lin, Gaston, Bush, & Fuller, 2014), it is expected that similar results will be seen for the frequency of vegetation types, a measurement not yet reported within the literature. The diversity of features, a common measure of garden quality as discussed in Section 2.6, within a household garden is also yet to be researched within the realm of environmental injustice, yet it is the diversity and quality of a garden which directly produced the benefits to residents (Pretty, Peacock, Sellens, & Griffin, 2005). Ethnic minority communities have reported viewing quality public greenspaces as a luxury that they cannot afford (Birch et al., 2020), and it is likely private greenspaces are similar.

## 1.2 The extent, content, and form of urban household gardens

In the 2016 UK focussed report *Gardens and Health: Implications for policy and practice*, the Kings Fund state as one of their three aims 'to demonstrate how gardening interventions have an important place in the National Health Service (NHS) and wider health care system' (Buck, 2016, p 5). Perhaps the most significant step forward came in January 2018, when the UK government's Department of Food and Rural Affairs' (DEFRA) published its long awaited 25-Year Environment Plan (25-YEP) – a plan that sets out a strategy to improve the natural environment of England within a generation. Within the 25-YEP, household gardens are noted to have an important role in healthcare, in encouraging contact with nature in children, and being an important part of green infrastructure in the city (DEFRA, 2018; Horticultural Trades Association, n.d.). Defra established The Urban Pioneer in Greater Manchester in 2018 to test deliverables practically within its 25-YEP in an urban setting. By linking with and contributing to The Urban Pioneer an opportunity was opened for the research reported in this thesis to inform UK government of the contribution of household gardens to the social-environmental system of Greater Manchester.

Data are needed on the stock of household gardens to develop all-inclusive policies that include this resource (Dewaelheyns, Rogge, & Gulinck, 2014). Such data are needed to influence policy makers and stakeholders when deciding on initiatives, especially due to the environmental injustices highlighted above (Garcia-Antunez, Lindgaard, Lampinen, & Olafsson, 2023). Further, size and quality of greenspace are important factors when assessing their health benefits to people (Dillen, Vries, Groenewegen, Spreeuwenberg, 2012). Therefore, it is important to have up-to-date, accurate data on the collective size and content of urban gardens, as well as what causes variation within them. Recently published articles within the field commonly cite city-wide garden extent accounts from Loram et al., (2007) as a more up-to-date account is lacking (van Heezik, Freeman, Davidson, & Lewis, 2020; Garcia-Antunez et al., 2023). Urban form, defined as 'the physical characteristics that make up built-up areas, including the shape, size, density, and configuration of settlements' (Williams, 2014, p 6), has been reported to influence household garden size and thus content (Whitford, Ennos, & Handley, 2001; Loram, Tratalos, Warren, & Gaston, 2007; Cavan et al., 2018).

As shown by Grove et al. (2006), both house age and residents characteristics have contributed significantly to the content of a garden. Similarly, house age had a weak positive correlation with canopy >3 m tall increased in both area ($r^2 = 0.13$, $p = <0.03$) and proportional contribution ($r^2 = 0.12$, $p = <0.05$) (Loram, Warren, & Gaston, 2008). Baker and Smith (2019) found the proportion of useable greenspace differed substantially between Victorian terraced housing, 34 per cent, and all other property types: non-Victorian terraced 63 per cent, semi-detached 62 per cent, and detached 64 per cent. Furthermore, there is evidence to suggest garden size is getting smaller in contemporary developments, and in some housing developments gardens are omitted completely (Chalmin-Pui et al., 2021; Douglas et al., 2019). Conversely, some publications have noted the increase in the provision of household gardens in towns and cities during the twentieth century throughout the UK (Bhatti & Church, 2004; Cillers, Siebert, Davorn, & Lubbe, 2012). This contradiction highlights the importance of accurately reporting garden extent; whilst individual garden size has decreased urban density has increased (Wellmann, Schug, Haase, Pflugmacher, & Linden, 2020) resulting in a larger collective garden size in some cities.

Yet, a comprehensive comparison of garden size and content between different development eras is lacking within the literature, with the studies above not giving a comprehensive account. Some argue as urban densification increases there is little space for gardens, and that they should be viewed as a lesser priority to public spaces in urban planning (Wellmann et al., 2020). But, gardens can have vital health and wellbeing benefits (James, Banay, Hart, & Laden, 2015), as discussed below. Thus, urban form is another variable worth investigating when assessing how household gardens can contribute to the improvement of resident's health and wellbeing.

## 1.3 Valuing private urban gardens and the impacts on residents

Household gardens are the most heavily used outdoor space and represent the most frequent contact with nature for most people (Dunnett & Qasim, 2000). Gardening as a leisure activity increased in popularity during the post war period in the UK. More recently, post-COVID-19 pandemic interest in gardening and food growing has skyrocketed, largely due to the increased time spent at home, equating to approximately an extra 3 million gardeners in 2020 in the UK (Sams, 2020). Understanding the value in the health and wellbeing benefits green spaces can provide often takes a qualitative approach. Interviews have been used previously to provide insight into: data on duration and reasoning behind residents gardening in Sheffield (Dunnett & Qasim, 2000); the link between nature and perceived wellbeing in Sheffield (Fuller et al., 2007); participants thoughts on urban nature in Sheffield (Birch et al., 2017); young people's views on how urban nature supports their mental health in Sheffield (Birch et al., 2020); and provide data on perceived health and garden use post-pandemic in Scotland (Corley et al., 2021).

Within England, there are two large-scale national annual surveys which capture data relating to gardens and gardening: English Monitor of Engagement with the Natural Environment survey (MEME) (n = 15,390 total) and the Taking Part survey (n = 10,000 annually), yet only one publication has analysed the data. Bell et al. (2020) used MEME to highlight the variation in gardening engagement between age and socio-economic groups, reporting overall greater self-assessed health and wellbeing in residents who used their garden with those 65 years and over or in socio-economic group AB (higher & intermediate managerial, administrative, professional occupations) benefitting most. By combining the two surveys, large-scale data can be analysed on gardening popularity and access, including finer analysis on the socio-economic status of gardeners and their views on their garden. This thesis was the first to collate and compare these data, highlighting an environmental injustice on a national scale (Section 6.4).

Personal investment in the context of this thesis is viewed as the amount of time and money invested by residents into their household gardens. The value of easily accessible private green space, such as gardens, has received little attention in recent literature. Previous research has explored participation in the 1990s in England (Bhatti & Church, 2004); gardening frequency in Sheffield (Gaston et al., 2005) and gardening duration in England (Dunnett & Oasim, 2000). However, there is no recent study analysing the monetary value of such spaces, which is important resource as outlined by the Fields in Trust (2018) report Revaluing Parks and Green spaces. Within this thesis, the analysis of time invested by residents is explored using secondary sources, the Taking Part Survey 2015/16 – 2019/20 and MEME 2009-2019 national surveys. To explore the amount of money invested by residents, data from the Horticultural Trades Association (HTA) (2018) was used. Previous publications of such figures are lacking, Gaston et al. (2005) estimated that between 1999–2000 spend on garden products was valued at £2.62 billion, with a mean household spend of £183 per annum, though this was based off a small sample size and is similarly out-dated. There is a gap in the literature for an up-to-date report on the level of investment residents undertake in their gardens. Further, from this a cost-benefit analysis of household gardens and their benefits to people's health and wellbeing would be possible to showcase the monetary value of these urban greenspaces. This work was of interest to the Urban Pioneer, an environmental initiative led by the Environment Agency in Greater Manchester and was included within their reporting on gardens potential (Urban Pioneer, 2019).

## 1.4 Overall aim, research objectives, and research questions

The overall aim of this thesis is to undertake a critical exploration and analysis of the value of private gardens, with a key focus on the City of Salford, evaluating the size and content of urban household gardens to determine factors which contribute to the heterogeneity of the resource and their associated benefits. Resulting in four research objectives and three research questions which are outlined below.

**Research Objectives:**

There is a far greater body of evidence for environmental injustice within public spaces than within the private realm (Sections 2.1.1 and 2.1.2). As people often have their most direct and frequent contact with nature within their private gardens, injustice within this setting is of great importance. Thus, more evidence is needed. Specifically, the relationship between the variation in garden size and feature diversity and the socio-economic factors of the household has yet to be investigated. It has been shown greenspaces, including gardens, can mitigate health inequalities (Section 2.1.2) and whilst environmental injustice has begun to be considered more in UK policy (Section 2.2), with DEFRA and the NHS working in partnership, there is still a need to build the evidence base.

Previous literature on the extent of urban gardens at city-level are out-dated, often using methods which could be improved upon today (Section 2.3). Research into garden content is similarly outdated (Section 2.4), with only a handful of investigations that were either small-scale or limited within the content they included. In addition, a review of the urban form literature (Section 2.5) shows property type directly effects the size and content of a household garden, but little is published on the effect of development age on gardens. Therefore, there is a gap in the knowledge to conduct a large-scale investigation into how property type and development age affect urban garden size and content in the UK.

Residents gain multiple health and wellbeing benefits from their gardens (Section 2.6.1). Both the view and use of a quality, diverse greenspace can reduce chronic stress and improve health and wellbeing; but UGI research and environmental policy largely focus on these benefits within the public realm.

**Research Objectives:**

*Objective 1: To compare the size and diversity of OAC Supergroups household gardens in the City of Salford.*

*Objective 2: To determine whether household gardens as a collective resource are equally distributed across districts of Greater Manchester.*

*Objective 3: To assess the change in garden size and diversity in housing developments over time in the City of Salford.*

*Objective 4: To evaluate the economic value of urban household gardens compared to that of public urban greenspace.*

**Research Questions:**

RQ1: In a time when environmental equality is highly topical, is there a disparity in the size and diversity of urban household gardens resulting in an injustice in the access to resources between different socio-economic groups?

RQ2: Does the extent of urban household gardens vary in neighbouring Greater Manchester districts and is there evidence of newer housing developments having smaller gardens in the City of Salford?

RQ3: How do the benefits within urban household gardens compare to those in public UGI, and are these benefits equally accessible?

## 1.5 Research map

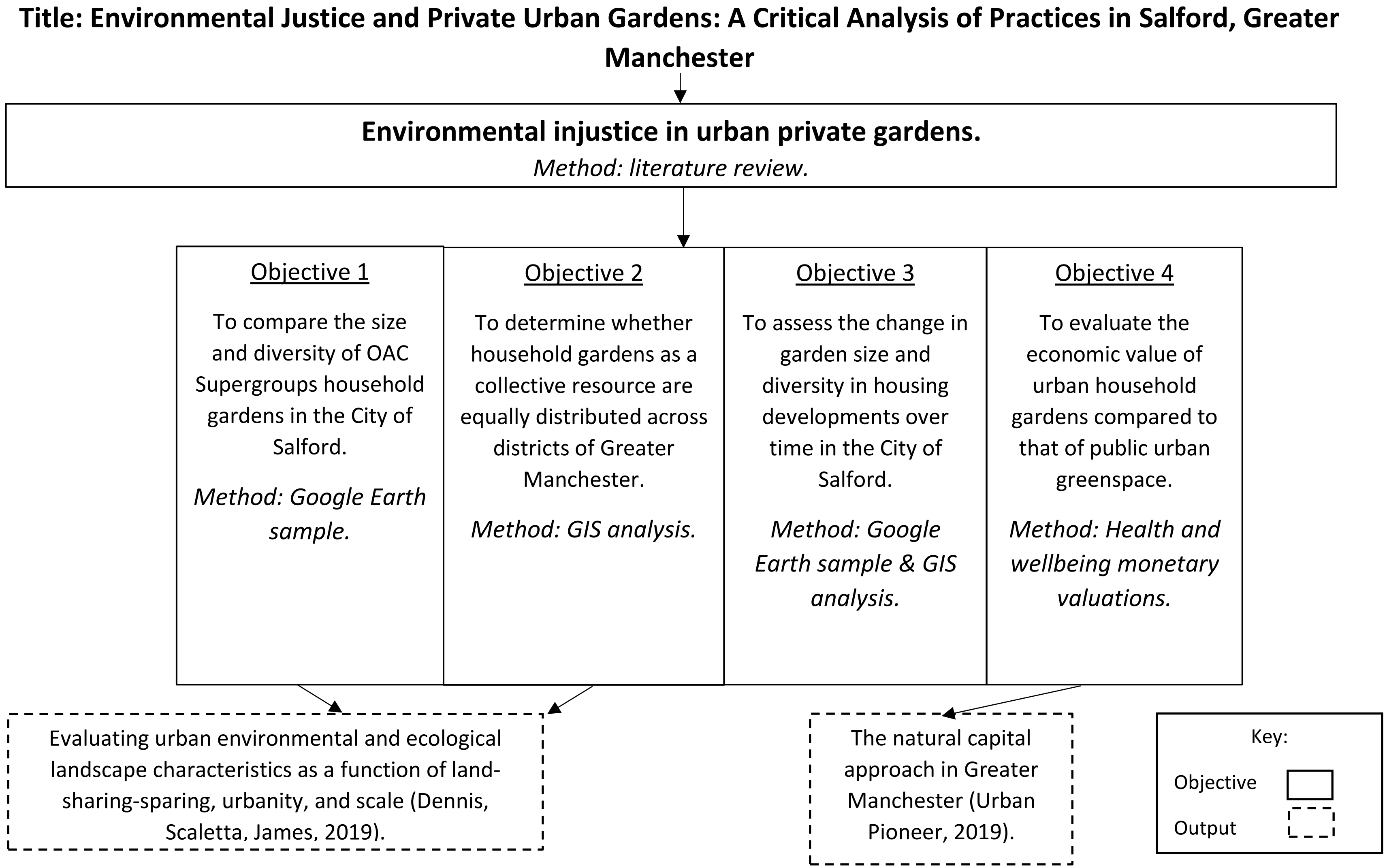



Figure 1.2: A schematic to show how the methods and objectives are linked.

## 1.6 Thesis Outline:

To understand fully the key concepts needed in this thesis a literature review was undertaken, presented in Chapter Two. It begins by exploring the idea of environmental injustice (Section 2.1), specifically within a public setting (Section 2.1.1) and then a private one (Section 2.1.2) exploring the effect of socio-economics on the extent and quality of the benefits received by urban greenspaces. Next, the increasing importance of household gardens within UK policy is explored (Section 2.2). It is highlighted that although steps have been taken, for example with DEFRA's 25-Year Environment Plan, more evidence is needed to improve the reach of such policies. The existing literature on household garden extent (Section 2.3) and characteristics (Section 2.4) are then evaluated; followed by investigating the relationship between urban form (Section 2.5) and garden form and function. After exploring these factors, socio-economics and urban form, the benefits of gardens are presented (Section 2.6) – divided into health & wellbeing (Section 2.6.1) based around the theories of Stress Recovery Theory (Ulrich et al., 1991) and Attention Restoration Theory (Kaplan, 1995) and environmental benefits (Section 2.6.2) based around their ecosystem services. Finally, the variation in the level of personal investment, time and money, residents undertake in their gardens is explored (Section 2.7).

Next, presented in Chapter Three are the methodology and tools used to test these hypotheses. The Chapter begins with an exploration of research philosophy (Section 3.1). Then, a detailed account and defence of the research location (Section 3.2) followed by an overview of the approach to methods (Section 3.3). Then, to gain a more detailed understanding of this variation at a city-level, target Output Areas within different socio-economic Supergroups were sampled, obtaining data on garden size and content (Sections 3.4 – 3.9). To further this exploration, engagement data from the RSPB Big Garden Birdwatch were analysed to investigate variation within the socio-economic Supergroups (Section 3.7). Aerial photography was then analysed to quantify the garden cover across Greater Manchester and within its ten districts (Sections 3.8 and 3.9). Health and wellbeing calculations using the estimates presented in Mourato et al., (2010) (Section 3.10) are then presented for the use and view of a garden and the use of a public greenspace. By calculating an up-to-date estimate of spend on gardens (Section 3.11) these could then be used in a cost-benefit analysis.

Chapter Four is the first of three results chapters in this thesis. The focus is exploring environmental injustice in household gardens in Salford, Greater Manchester. Section 4.3 presents radar graphs depicting a typical garden for each of the seven different socio-economic Supergroups, which show great differences in the amount of vegetation cover and garden features between different societal groups. Section 4.4 then uses these same groups and secondary data from the RSPB to map residents' engagement in a national bird watching survey, used as a proxy for garden engagement generally. Finally, a summary (Section 4.5) of this Chapter explores the luxury effect which is exposed using the data in this thesis and further highlights the societal groups most at risk of environmental injustice.

Chapter Five is the second research chapter focusing on the surrounding urban form influencing environmental injustice within gardens. It begins by giving a landcover overview of the study area, Greater Manchester, and a justification for the use of the City of Salford (Section 5.2). Presented next is an in-depth investigation into the size and features of Salford's urban household gardens, comparing these to the literature shows the City of Salford to have, on average, smaller gardens than previous research suggests (Section 5.3). Further, garden size is shown to have a significant effect on content as explored in Section 5.3.3. The effect of surrounding urban form (property type, development age, and density) on garden size and content is then explored (Section 5.4). A summary of the chapter in Section 5.5 highlights the need for this contemporary and representative land cover data for policy makers, as well as the effect smaller gardens in newer developments could have on residents' health.

Chapter Six is the final research chapter. Section 6.2 uses data on health and wellbeing from the 2010 NEA report, updated to account for inflation, to provide 2018 monetary valuations for household gardens. It estimates a value for the use of a garden by residents, the use of an urban park, and for having a view over a greenspace, concluding gardens could save the NHS the equivalent of approximately 8 per cent of its budget annually. Next, secondary data on residents' monetary investment is presented (Section 6.3) followed by a discussion on how this investment has increased post-pandemic.

Chapter Seven provides a discussion combining work from the thesis as a whole, split into three sub-sections: how does the socio-economic situation of residents affect their gardens (Section 7.2); the extent, content, and form of urban household gardens (Section 7.3); and valuing private urban gardens and the impacts to residents (Section 7.4). The last chapter, Chapter Eight, then gives an overview of the contributions to knowledge from this thesis (Section 8.1), the limitations of the research (Section 8.2), and finally recommendations for future research (Section 8.3).

# Chapter Two: Literature Review

## 2.1 Introduction

This chapter begins with a critical analysis of key environmental justice research, within both the public and private settings, with the latter urban greenspaces the central focus of this thesis. The literature review then proceeds to provide an overview of household gardens within UK policy; including the Urban Pioneer, a DEFRA initiative which this research directly contributed to. This overview of policy highlights the lack of inclusion of private gardens compared with public UGI. The chapter then reflects on the current research on garden extent in the UK with literature estimating landcover between 22 – 27 per cent (Gaston et al., 2005; Loram et al., 2007; Tratalos, Fuller, Warren, Davies & Gaston, 2007). The content of these urban gardens has scarcely been recorded, resulting in the overall aim of this thesis, to undertake a critical exploration and analysis of the value of private gardens, with a key focus on the City of Salford, evaluating the size and content of urban household gardens to determine factors which contribute to the heterogeneity of the resource and their associated benefits. Resulting in four research objectives and three research questions which are outlined below.

Following from this is an overview of urban form and the impact the surrounding form can have on the size and content of a garden. This review on the urban form found a gap in the knowledge to investigate the change in garden size from a temporal perspective which was a resulting objective of this thesis. Green exposure has been linked to a variety of health-related outcomes including reduced chronic stress, reduced obesity, and reduced psychiatric morbidity (James, Banay, Hart, & Laden, 2015). The potential health and wellbeing benefits of greenspaces, both public and private, are discussed next within the literature review to highlight the importance of these spaces to users and residents. Finally, a discussion around the level of personal investment, time and money, residents invest into their gardens is presented. The review of the literature resulted in an overall research aim, four research objectives and three research questions which are shown in Section 1.4.

## 2.2 Environmental Justice

Environmental justice is a broad, inter-disciplinary term which has grown in popularity in contemporary research. As shown in Chapter One injustice, either by access or exposure, can be found in any aspect of the biophysical supply of ecosystem services and their associated benefits. As such, the term has been subject to a number of definitions. Defined by Maantay (2007, p.33) as:

‘the disproportionate exposure of communities of colour and the poor (or other vulnerable groups) to pollution, and its concomitant effects on health and environment, as well as the unequal environmental protection and environmental quality provided through laws and policies’.

A more wide-ranging and accessible definition may be environmental justice is:

‘the fair treatment and meaningful involvement of all people regardless of race, colour, national origin, or income with respect to the development, implementation, and enforcement of environmental laws, regulations and policies’ (United States Environmental Protection Agency, n.d).

The first definition reflects a common focus of environmental injustice literature; Aschener et al. (2021) in a contemporary overview of the topic highlight the focus on the disproportionate effects of pollution on different groups in society. The second definition has a greater focus on the difference in the treatment and opportunities of such groups, a research standpoint less frequently taken. For the research reported in this thesis, the second definition is more relevant as it is viewing injustice through a lens of fair treatment and involvement rather than exposure to polluting substances. Justice, in the context of environmental justice, is defined as the fairness of decision making and/or decision outcomes (Philips & Sexton, 1999). Kelly, Reif, and Wing (2016) point out that environmental justice has also been coined environmental racism in the 1980s. In the same article those authors also suggest that environmental justice is now the favoured term as it is viewed as more inter-disciplinary, producing popular movements of research in the social sciences, public health, and law, and, thus, this is the terminology used in this thesis.

Environmental injustice occurs when different societal groups have unequal access and gain unequal benefits from ecosystem services. Ecosystem services are defined as ‘the benefits provided by ecosystems that contribute to making human life both possible and worth living’ (UK NEA, n.d.). These services play fundamental roles in achieving the sustainable development of social-ecological systems and are often classified as either provisioning, regulating, or cultural services (Costanza, 2008; Shen et al., 2023), with some also including a fourth supporting services category (UK NEA, n.d.). Provisioning services directly result in goods/benefits; regulating services moderate/maintain other environmental features that benefits society such as climate regulation; cultural services are the non-material benefits that people receive from experiencing nature; and supporting services underpin all others by providing physical structure and ecological niches (Moss, Evans, & Atkins, 2021). See Figure 2.1 for more examples of ecosystem services. Natural capital is the term used for the stock which generates ecosystem services. Natural capital is the living and non-living components of ecosystems that contribute to the generation of goods and services of value for people (Guerry et al., 2015). Current global economic systems reward short-term production and consumption of commodities at the expense of stewardship of natural capital (Guerry et al., 2015). A better understanding of the important role of natural capital is needed through a long-term focus on integrating the concept within policy and decision making on a global scale (Guerry et al., 2015). In the 21st this is becoming increasingly recognised (Guerry et al., 2015).

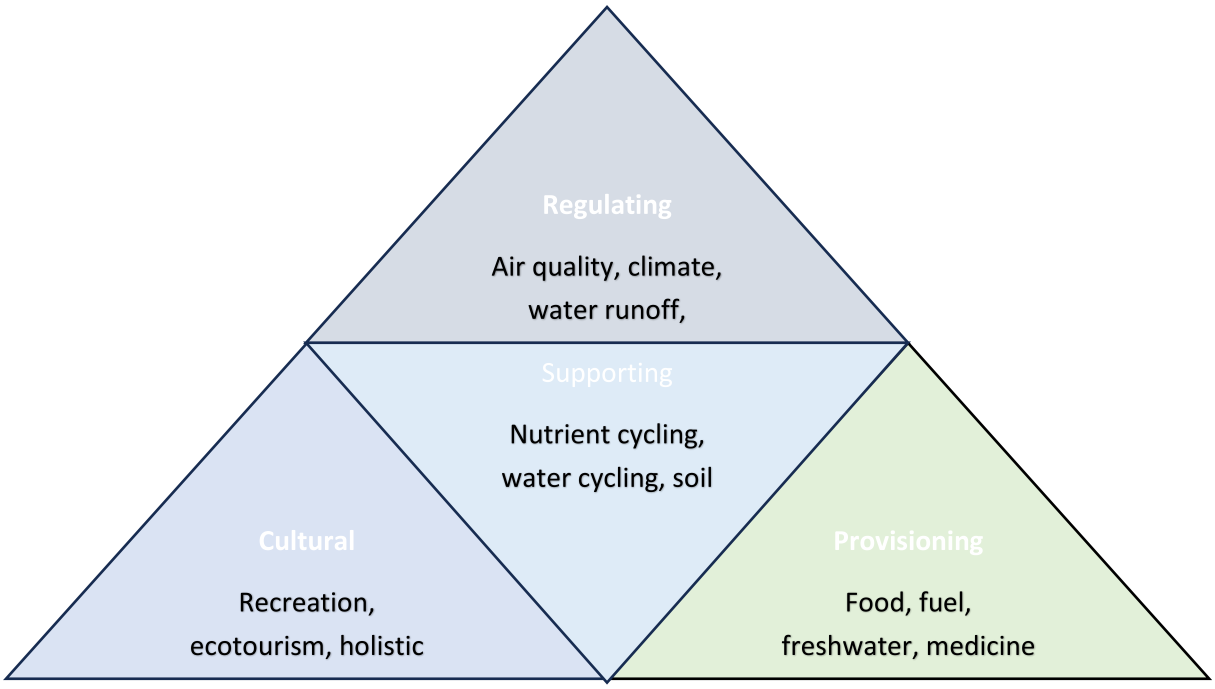



Figure 2.1: Examples of each of the four types of ecosystem services.

Within this section, the literature on environmental justice has been split into two subsections: within a public setting and within a private setting. In the public setting there is a much greater body of evidence on the subject: within public parks (Birch et al., 2020);

pollutant emitting facilities (Fong, 1994; Vaz et al., 2017); greenspace provision (Apparicio, Pham, Seguin, & Dube, 2016; Jones, Hillsdon, & Coombes, 2009; Mears et al., 2019; Wustemann, Kalisch, & Kolbe, 2017); vegetation cover (Luck, Smallbone, & O'Brian, 2009); and biodiversity (Carrus et al., 2015; Methorst, Bonn, Marselle, Bohning-Gaese, & Rehdanz, 2021) to give some examples. There is a larger body of literature examining social disparity within community gardens (Burt, Mayer, & Paul, 2021; Ramsden, 2020) and often participants in community gardens do not reflect the social and ethnic diversity of the whole community (Christensen, Malberg Dyg, & Allenberg, 2018). For example, in Copenhagen, Denmark 87 per cent of community garden users had obtained a higher education compared to 41 per cent of the surrounding, non-using neighbourhood members, likely due to barriers with language and annual membership costs (Christensen et al., 2018). However, in the private setting there is less evidence, largely due to difficulties with access during research (Bell et al., 2020; Maantay, 2007; Vaz et al., 2017). Previous literature has demonstrated environmental injustice within garden vegetation cover (Grove et al., 2006; Kendal, Williams, & Williams, 2012; Shanahan, Lin, Gaston, Bush, & Fuller, 2014); and biodiversity (Chamberlain, Henry, Reynolds, Caprio, & Amar, 2019; Kinzig, Warren, Martin, Hope & Katti, 2005; Lerman & Warren, 2011; van Heezik, Freeman, Porter, & Dickinson, 2013). Missing from the literature within the private realm is an account of socio-economic patterns on garden size and on garden feature diversity.

### 2.2.1 In a public setting

Urban greening has grown in popularity in recent years with landscape planners (Liotta, Kervinio, Levrel, & Tardieu, 2020). Target 11.7 of the Sustainable Development Goals (United Nations) states that cities need to 'provide universal access to safe, inclusive and accessible, green and public spaces…' by 2030 (United Nations, n.d.). As of 2020, 47 per cent of the global urban population lived within 400 m walking distance of open public spaces (United Nations, n.d.); given their diverse benefits to people it is an injustice that over half of the urban population are living further than a 10-minute walk away from accessible greenspace. Further, this statistic does not give detail on the quality of such greenspaces, an important factor for their value.

One contemporary example of environmental injustice within the realm of public health can be seen in the report published by Public Health England in June 2020(Public Health

England, 2020). Within which it is stated that the COVID-19 pandemic has not created health inequalities within BAME groups (Black, Asian and minority ethnic), but exposed and exasperated them. 'BAME groups tend to have poorer socio-economic circumstances which lead to poorer health outcomes' (Public Health England, 2020, p. 6), and these differences in circumstance are often linked to environmental factors (Vaz, Anthony, & McHenry, 2017). Kingsley et al. (2022) state the exposure of significant health inequalities, food insecurities, and limited access to greenspace were consequences of the upheaval caused by Covid-19. On a more global scale, all of the seventeen United Nations Sustainable Development Goals (SDGs), to be reached by 2030, fit the concept of linking environmental quality with social equality (Agyeman, 2020). In the 2020 update statement on the progress of the Sustainable Development Gaols (SDGs) in the UK, it was noted the impacts of Covid-19 within the education sector would be hardest felt by the most marginalised, with a total of 1.3 billion children being out of school over the pandemic (Sugg, 2020). The two examples of BAME groups and marginalised students within schools illustrate how the UK government is acknowledging injustice within the public sector and are beginning to seek to address the inequalities.

An understanding of the need for environmental quality has been demonstrated amongst young adults in the UK; Birch et al. (2020) conducted interviews and used workshops to establish the views of 24 participants aged 17 – 27-years-old on their views on urban nature in Sheffield. Reoccurring themes from the interviews were ideas around nature being something that occurred outside the city in landscapes where there were vast openings and mature trees present, and not within the city itself. When asked out-right about the benefits of nature to mental health and wellbeing, participants generally where more sceptical stating nature is no 'cure-all' for illness. Interestingly, participants noted the deterioration of their local public parks and how this negatively affected their desire to spend time there; when asked about travelling to a park which felt cleaner and safer one respondent stated it was a 'luxury I can't afford'. Ironically, at a similar time to when this research was conducted, Sheffield council received some bad press for the unnecessary and dishonest felling of thousands of trees within residential areas (BBC, 2020). The study by Birch et al. (2020) provides a valuable snapshot of the contemporary views of young people regarding urban nature, a key point highlighted from this research is the fact that young people are aware of the quality of the greenspace, and further that they see high quality space as a luxury. However, the small sample size and time-consuming nature of interviews are common drawbacks of greenspace quality research (Liu et al., 2023). To rectify this, and increase the evidence base within street GS, Liu et al. (2023) used a neighbourhood approach (n = 137) increases in total vegetation cover, used as an indicator for quality, were associated with lower prevalence of hypertension. However, a key factor in the quality of GS is maintenance which is increasing difficult due to continued urbanisation (Liu et al., 2023).

The need for spatial consideration of injustice has been an issue of growing concern since the 1970s (Carey & Harvey, 1975; Certoma & Martellozzo, 2019). Spatial injustice, defined as intentional and focused emphasis on the spatial or geographical aspects of justice and injustice, is a key consideration within many social science disciplines (Fincher & Iveson, 2012; Rauhut, 2018; Yazar & York, 2023). One of the earliest writers on environmental injustice was Douglas Massey (1996) who showed inequality could be characterised as a spatial phenomenon with clear segregation between where groups of the privileged and the poor could be found within an urban area. Massey (1996) described the 'typical poor Latin American of the twenty-first century…' as living in a large city as opposed to the rural existence of such a community in the 1970s (Massey, 1996, p. 396). Massey describes a similar trend amongst poor communities in North America with increasing numbers concentrating in urban areas, 56 per cent in 1970s to 72 per cent in 1990s. Further, Massey predicted that between 2010-2020 the Global South would cross a line whereby the majority of the population lives in cities, a prediction we now know to be true (United Nations, 2014; World Health Organisation, n.d.). Massey states because of this the twenty-first century will experience unprecedented spatial separation of classes with poor communities being concentrated in urban areas: 'because there is no precedent for a reversal of urbanisation once it has begun, the future of human poverty almost certainly lies in cities' (Massey, 1996, p. 400). This historical perspective illustrates a shift in the predominant spatial extent of poorer communities, from rural to urban. It is likely this shift intensified environmental injustice as poorer communities move into urban areas, but do not have the financial powers to determine their environmental surroundings and thus potential gains (Vaz et al., 2017). Therefore, research into environmental injustices in the twenty first century should focus on the urban landscape, but the bulk of the studies which do this are within a public land setting.

Environmental injustice negatively affects the health of low-income and minority communities, in part, by disproportionate exposure to negative environmental impacts and social pressures (Vaz et al., 2017). Much of the literature surrounding negative impacts of environmental injustice focuses on the geographic location of pollutant emitting facilities and how these are often within racialized and socio-economically disadvantaged areas (Fong, 1994; Vaz et al., 2017). Another prominent area of environmental injustice research is child asthma, as this remains one of the most prevalent chronic childhood diseases among racial/ethnic minority and poorer children, particularly in urban environments. Asthma can also be exacerbated by chronic stress caused by social and environmental injustices (Chen et al., 2006), further, this stress has been shown to negatively impact concentration in school and relationships with family members, as well as increase the risk of adolescent depression (Basch, 2011; Moonie, Sterling, Figgs, & Castro, 2008). Such health disparities are often inherited through generations (Andersson, 2016; Harris, 2019; Leopold & Leopold, 2018); which deepens both individual and family disadvantage and perpetuates population-level inequality (Harris, 2019). This uneven distribution of environmental impacts, both positive and negative, is termed urban landscape preferentialism (Vaz et al., 2017).

Similar to research within the field of pollutant emitting facilities, much research regarding injustice is on a large-scale, focusing on the unequal exposure to various pollutants by a certain societal group. To identify neighbourhoods that are exposed to the highest and lowest rates of environmental injustice, Vaz et al. (2017) combined the Canadian 2011 census data with Toronto's public health ChemTRAC database. Using household income, home ownership, education level, and unemployment they found the communities residing within the areas of lowest air quality, often due to proximity of major transport routes, were generally those who were economically limited and/or of a minority ethnicity. Further, they state household circumstance, particularly income, is a key limiting factor in having the freedom to choose a neighbourhood and that those with a greater income can out-bid others for the best neighbourhoods. Within the Greater Toronto area, the location of their study, Vaz et al. (2017) argue this disparity has been historically overlooked by the government and industry resulting in an unfair housing market and unregulated pollution emitting practices. Similar findings have been shown globally: in Paranaguá, Brazil where higher risk levels from air pollution correlate with lower household income (Gurgatz et al., 2016); Hong Kong, China where a positive relationship was illustrated between poor air quality and social deprivation (Li, Han, Lam, Zhu, & Bacon-Shone, 2018); and Houston, Texas with high hazardous air pollutants risks positively related to economic restraints of residents (Hernandez, Collins, & Grineski, 2015) to name a few examples.

Environmental injustices focusing on large-scale ecosystem benefits such as clean air have been well documented, as has the uneven distribution of greenspace equity between social classes within public greenspaces (James, Banay, Hart, & Laden, 2015). Households which are more socially disadvantaged often have less greenspace provision, have greater population pressure, and often have fewer opportunities to travel out to enjoy the environment (Apparicio, Pham, Seguin, & Dube, 2016; Jones, Hillsdon, & Coombes, 2009; Mears et al., 2019; Wustemann, Kalisch, & Kolbe, 2017). Yet, groups with a lower level of education and those with socio-economic status have been shown to be more sensitive to physical environmental characteristics negatively impacting their health (James, et al., 2015; Maas et al., 2006). Some argue, therefore, that access to greenspace is an important environmental justice (Boone, Buckely, Grove, & Sister, 2009) and that development needs to be both sustainable and equally accessible to benefit society best (Agyeman, 2020). Conversely, Barbosa et al. (2007) and Mears et al. (2019) found the more disadvantaged neighbourhoods in Sheffield, UK, had better access to greenspaces. However, when including the quality of greenspace into analysis, Mears et al. (2019) found no significant difference between neighbourhoods. This contradiction is often due to the plasticity of society, as explained further below.

Studies into public greenspace provision can be conflicting, as with Mears et al., (2019) and Barbosa et al., (2007): disadvantaged neighbourhoods do not always contain fewer or lower quality UGI. In the US, economic depression and government-led schemes have explained unexpected socio-economic equality in the access to greenspaces (Boone et al., 2009; Wolch et al., 2005). In the UK, urban parks were largely established in the mid-nineteenth century to improve working-class lifestyles. For example, Peel Park, the City of Salford established in 1846 was the first public park in the UK funded entirely by local people, featuring a museum and a library for the ‘self-improvement’ of users (Salford City Council, n.d.). Many of these public spaces remain unchanged, thus their distributions are largely affected by land-use planners in the Victorian era. Kuras et al. (2020) terms this the ‘Legacy Effect’ whereby past spatial patterns of social inequality continue to shape the modern urban landscape. Results of two studies in Sheffield have shown that more income-deprived households live closer to public greenspaces than those who are less disadvantaged which reflects the change in socio-economic status of Victorian developments in that city (Barbosa et al., 2007; Mears et al., 2019). Other studies, however, have found no significant difference between greenspace provision and socio-economic status (Mavoa et al., 2015; Kimpton, 2017). Further, the opposite conclusion has also been reported: in Montreal, Canada significantly less vegetation cover is found in and around low-income city blocks (Apparicio et al., 2016); availability of greenspace was substantially lower in areas of lower income residents in five cities in Australia (Burt, Feng, Mavoa, Badland & Corti, 2014); and overall, income was found to have a positive influence on the availability of greenspace across 53 cities in Germany (Wustemann, Kalisch, & Kolbe, 2017). In a global meta-analysis of 84 cases from 34 cities, Kuras et al. (2020) found the majority of studies found positive relationships between socio-economic status and biodiversity (63 per cent), only 12 per cent of studies found a negative relationship and 25 per cent showed a neutral one.

Similarly, socio-economic variables can affect vegetation cover when assessing urban greenspace as one resource, both public and private land combined. In south-eastern Australia, 20 years of socio-economic change and 15 years of vegetation change within gardens from 1986 to 2006 were tracked in 32 residential neighbourhoods of approximately 200 houses, based on Census Collection Districts (Luck, Smallbone, & O’Brian, 2009). Stratified random sampling was used to select sample neighbourhoods based on housing density and income level. Data on vegetation cover were from the Australian Commonwealth Government National Carbon Accounting System satellite imagery database. The purpose of that study was to compare the effectiveness of using biophysical and socio-economic variables to explain spatial variation in vegetation cover. Time-lag models containing socio-economic data from previous years explained more vegetation variance than models based on contemporary characteristics only. In 1991 the contemporary model produced $R^2 = 0.148$ $p = 0.052$ compared to the time lag model $R^2 =$

$0.234$ $p = 0.021$, and in 2006 the contemporary model produced $R^2 = 0.309$ $p = 0.002$ compared to the time lag model $R^2 = 0.431$ $p = 0.001$. Luck et al. (2009) suggest this provides strong support for allowing a time-lag for the response of vegetation to socio-economic characteristics, though no recommendation for the duration of the lag is given and the correlation coefficient is weak. Therefore, in future research this time-lag needs to be considered, perhaps by repeating greenspace experiments to add in a temporal consideration. Three socio-economic variables were found to have relationships with vegetation cover: housing density had the highest vegetation cover at medium density, while education and the percentage of non-Australian born residents both had positive relationships, see Figure 2.2, with vegetation increasing in higher educated neighbourhoods. Due to the spatial resolution of the data, it was not possible to distinguish between public and private land in this study.

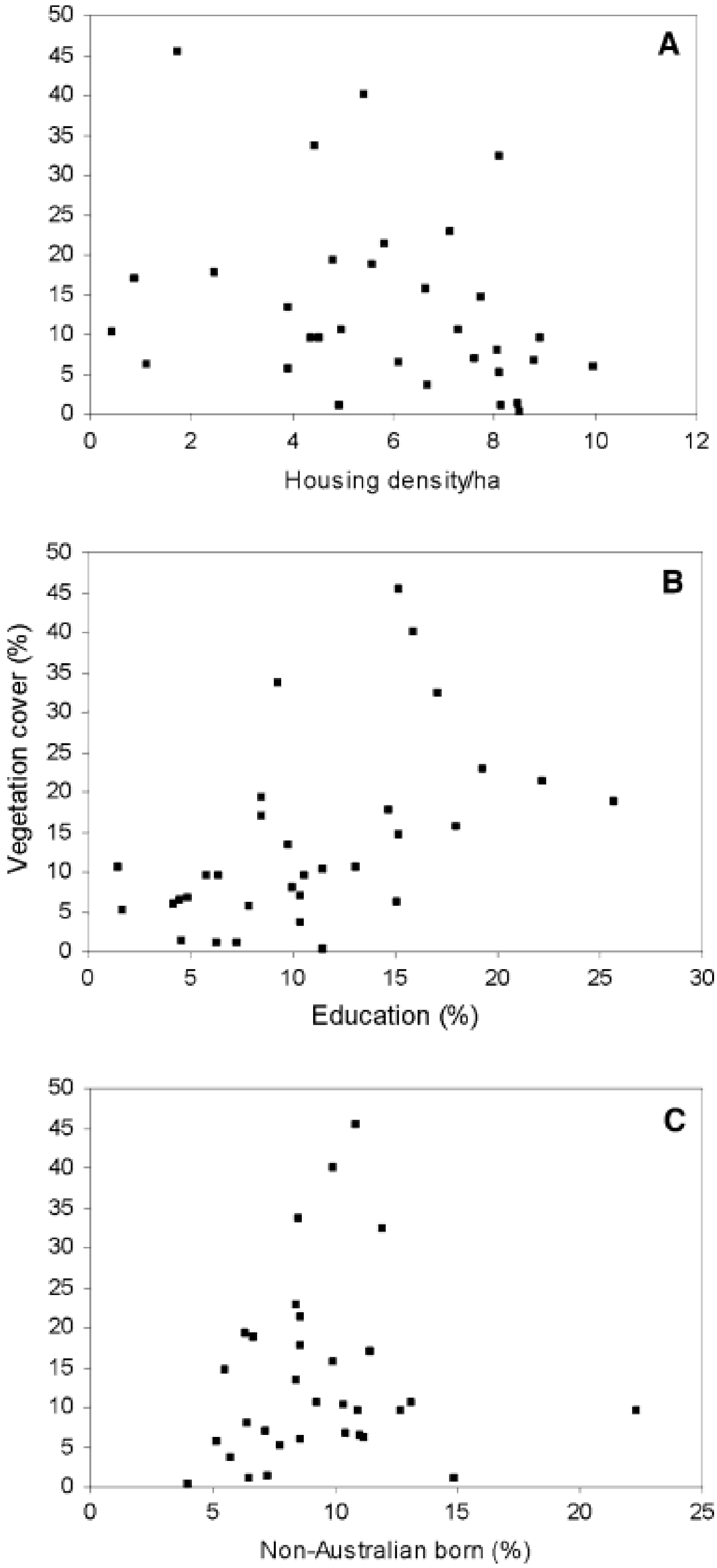



Figure 2.2: Relationships between vegetation cover and: A) housing density B) education C) percentage of non-Australian born residents, (Luck et al., 2009, p. 614).

Further, positive associations have been found between human health and biodiversity. Higher bird and plant species richness was found to be associated with the perceived wellbeing of greenspace visitors, reported during semi-structured interviews with 312 users in Sheffield, UK (Fuller, Irvine, Devine-Wright, Warren, & Gaston, 2007). Similarly, vegetation cover and afternoon bird abundance in urban greenspaces were found to be positively associated with lower prevalence of self-reported depression, anxiety, and stress in an online survey of 1023 urban dwelling adults in Southern England (Cox et al., 2017). The positive relationship between increased species richness and human health has also been shown in Germany (Methorst, Bonn, Marselle, Bohning-Gaese, & Rehdanz, 2021) and Italy (Carrus et al., 2015). In Canada, semi-structured interviews with 50 home gardeners who promoted biodiversity within their gardens reported benefits of social interaction, connectedness to nature, reduced stress and anxiety, and attention restoration (Raymond, Diduck, Buijs, Boerchers, & Moquin, 2018). The same study does also note a counterpoint. Gardening for biodiversity, often a wilder approach to management, frequently results in an untidy garden which puts many off the practice due to social pressures. However, although this viewpoint is noteworthy the health and wellbeing benefits arguably would outweigh these social and ethical conundrums (Raymond et al., 2018).

In order to make trade-offs between different urban land uses, it is essential to quantify the value of ecosystem services provided by different UGI (Perino, Andrews, Kontoleon, & Bateman, 2014). Much research has focused on publicly accessible greenspace, either through distance or quality (Belcher & Chisholm, 2018; Stormberg, Ohrner, Brockwell, & Lio, 2021). For example, in England those living in areas with greater proportions of public UGI had significantly higher mental wellbeing scores than areas with fewer UGI (Houlden, Weigh, & Jarrvis, 2017). Nature-based activities are increasingly gaining momentum as a cost-effective, easy, low-risk preventative and therapeutic intervention (van den Berg, 2017), with multiple examples of the social return on investment (SROI) in public UGI in recent literature. Social return on investment is a widely used approach to evaluate the wider impact of investment in relation to public benefits (Bosco, Schneider, & Broome, 2019). For example, in Glasgow in 2011/12 59 health walk projects had the associated benefits to participants with a benefit ratio for every £1 spent £8 was generated in wellbeing benefits (Allen & Balfour, 2014; Carrick, 2013). Similarly, the Green Gym project estimate a return of £2.55 for every £1 invested in England (Yerrelll, 2008). Healthcare professionals have increasingly been turning to these ‘green prescriptions’ making this intervention more integrated in contemporary everyday healthcare (van den Berg, 2017) although there is a lack of such research within private UGI.

Environmental injustice can be seen as a structural place-based inequality which creates a cycle of social and environmental injustice overtime (Andersson, 2016; Harris, 2019; Leopold & Leopold, 2018). There is a large bias, however, towards literature researching injustices within public spaces. This is largely due to a lack of data within the private setting (Maantay, 2007; Vaz, Anthony, & McHenry, 2017). There is no simple, universal methodology for measuring spatial preferentialism (Certoma & Martellozzo, 2019). Yet, as environmental injustice is inherently a spatial matter, spatial analysis and mapping should be employed to visualize and interpret patterns of inequality in urban areas (Vaz et al., 2017). Many studies use a geographic information system (GIS) to explore the spatial extent of inequalities (Vaz et al. 2017; Certoma & Martellozzo, 2019). Though, some highlight that there are limitations of using a GIS for environmental injustice and health research, mainly that of data deficiency both for the environmental and demographic parameters (Maantay, 2007; Vaz et al., 2017). Further, Certoma & Martellozzo (2019) note a distinct lack of quantitative analysis within the field. With health inequality in England estimated to cost up to £29.8 billion in 2022 (All-Party Parliamentary Group, 2022), more research into these injustices in private spaces are needed both to improve public health equality and the national economy. The widespread and systematic social and economic inequalities within the healthcare system show a clear social-class gradient with lower life expectancy in the most disadvantaged neighbourhoods (Allen & Balfour, 2014) (Figure 2.3), and with gardens often being a person's most direct and frequent contact with nature this UGI warrants further exploration.

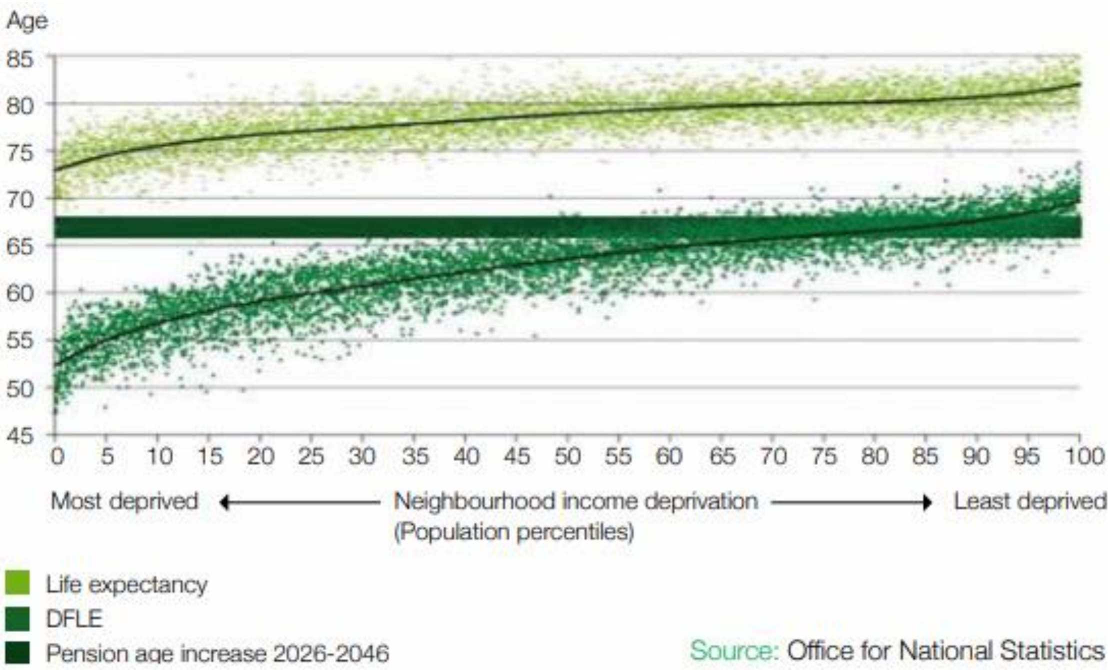



Figure 2.3: Life expectancy and disability-free life expectancy (DFLE) at birth compared with a person's neighbourhood income level, England, 1999-2003 (Allen & Balfour, 2014, p. 10).

## 2.2.2 Within a private setting

Environmental injustice may have been observed for centuries, but direct assessments are limited in terms of their subset of cities and taxonomic groups (Leong, Dunn, & Trautwein, 2018). Furthermore, there is a larger gap for this area of research within a private land setting (Brindley, Jorgensen, & Maheswaran, 2018; Haase, Janicke, & Wellmann, 2019). Research into private gardens is lacking, moreover, quantitative, city-wide studies are few and far between (Haase et al., 2019; Samus et al., 2022). Collectively, household gardens are an important UGI, it is in these settings people often have their most direct and frequent contact with the natural world. Household gardens account for a large part of urban greenspaces, and yet are understudied compared to other types of green infrastructure (GI) (Leve, Baudry, & Bessa-Gomes, 2019). In England, a target of 300,000 new homes annually was set out by the Conservative government in 2019; a target which is yet to be met, in part due to the Covid-19 pandemic, with 216,000 homes supplied in 2020/21 (Barton, Wilson, & Booth, 2022). It is this surge in new housing, along with the lack of published data, which support the need for a more in-depth and widescale account of urban household gardens.

The limited evidence of environmental injustice within the private setting, household gardens for example, is largely due to restrictions in data collection (Maantay, 2007; Vaz et al., 2017; Bell et al., 2020). Therefore, novel approaches are often needed to illustrate the unequal distribution of environmental benefits. This has been conducted by comparing the differences in gardening investment between different socio-economic groups. This comparison can be found in Section 2.8 of this literature review.

Investigating the demographics of gardeners is key, in parallel with their personal investment in their gardens, to understanding how the multiple benefits of gardens can be accessed by all (Kettle, 2012; Kirkpatrick & Davison, 2018). Wilkerson et al. (2018) use Maslow's (1943) hierarchy of needs to illustrate how socio-economic factors influence different types of needs and subsequently how these may also affect demand (Figure 2.4). Maslow (1943) categorises need according to five levels: physiological, safety, love/belong, esteem, and self-actualisation where those at the base are more fundamental. This hierarchy is a widely accepted framework across psychology and sociology (Wilkerson et al., 2018). Wilkerson et al. (2018) argue as people increase in socio-economic advantage their demand for ecosystem services related to fundamental physiological needs like food supply will decrease due to less dependency on their environment as a result of more disposable income. On the reverse, poor urban residents in South Africa who use their garden space for food production may be more reliant on ecosystem services for those fundamental needs, whereas some richer residents may choose to use their space for relaxation and aesthetic services (Cilliers et al., 2013).

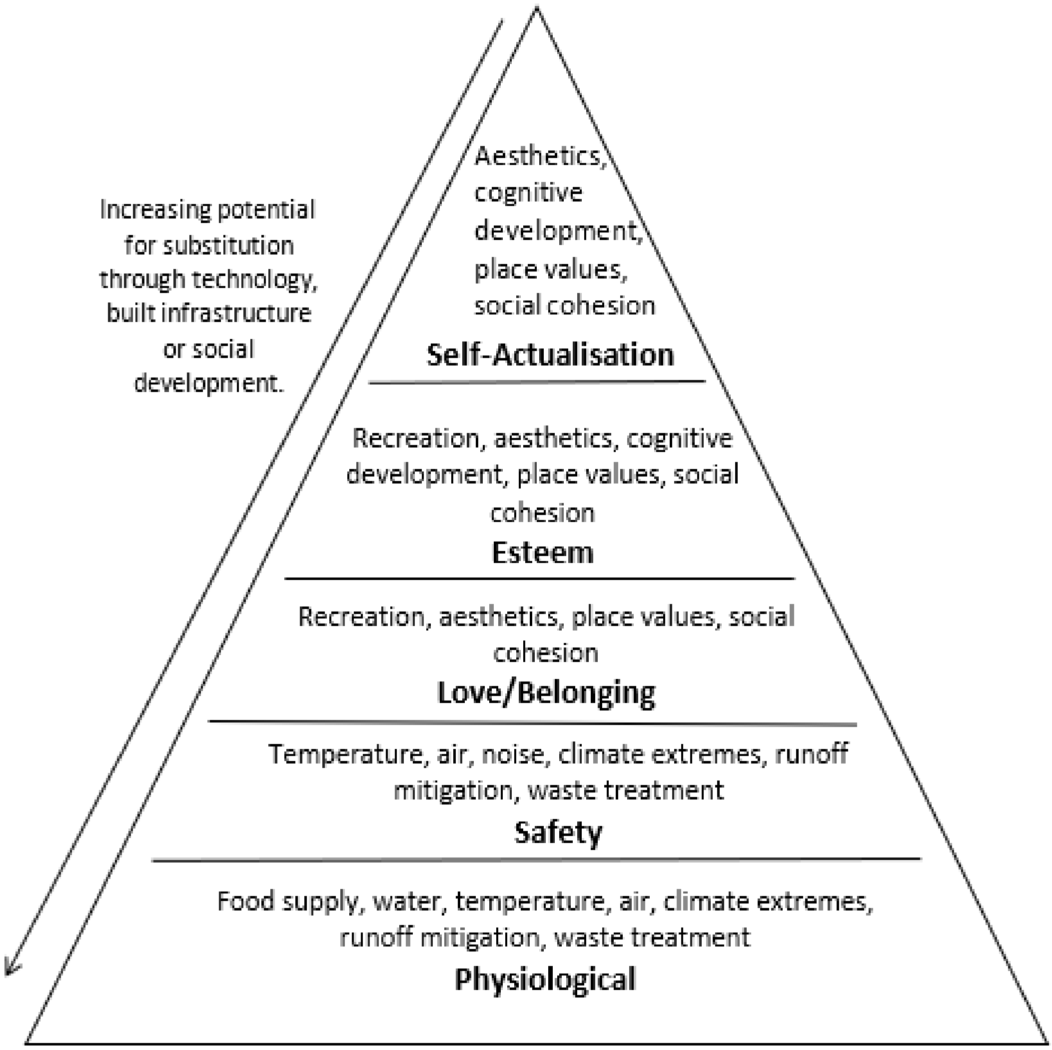



Figure 2.4: Maslow’s hierarchy of needs and how the delivery of ecosystem services may differ between differing socio-economic sectors of a population (redrawn from Wilkerson et al., 2018, p. 106).

Socio-economic factors have been shown to affect the vegetation cover within private urban greenspace. Grove et al. (2006) assessed which of three social variables – population density, lifestyle behaviour, and social stratification – best describe variation in vegetation cover (tree and grass cover) in public, private, and riparian land in Baltimore, Maryland. In the study social stratification refers to socio-economic status and the lifestyle behaviour variable incorporates a number of descriptive characteristics including household composition, mobility, ethnicity, and housing characteristics. It was hypothesised by Grove et al. (2006) that lifestyle behaviour would have the greatest bearing on private land due to an individual’s desire to express their membership of a given lifestyle group and the desire for an inner feeling of prestige. Data were collected from the Potential Rating Index for Zipcode Market database. Vegetation data were derived from satellite imagery. Thirty-two regressions were performed combining various social variables with each land type, as well as median house age, to determine which combinations best described vegetation cover variation. It was found a combination of house age and lifestyle behaviour variables were the best predictors of both private land tree cover (pseudo-$R^2 = 0.34$) and private land grass cover (pseudo-$R^2 = 0.32$). This highlights an important limitation of the luxury effect often used to explain the relationship between socio-economic factors and urban vegetation variation (Grove et al., 2006). The luxury effect, a social-ecological concept largely accredited to Hope et al. (2003) argues that areas with greater wealth have greater species diversity, a concept that has since been shown globally (Leong, Dunn, & Trautwein, 2018; Whitford, Ennos & Handley, 2001). It is important to include descriptive characteristics such as ethnicity, income, and education to explain variation due to the fluidity of society. One study has shown socio-economic status does not independently influence a person's use of greenspace if it is readily available, as results were statistically significant irrespective of age, sex, or economic status (Grahn & Stigsdotter, 2003). This again supports the need for a more descriptive and inclusive approach to explaining variation caused by socio-economic factors; a point also highlighted from an urban planning perspective by Liotta et al. (2020). There is a need for a more comprehensive approach to urban planning that incorporates environmental justice amongst urban greening initiatives (Calderon-Argelich et al., 2021; Sholry, Connolly, & Anguelovski, 2020).

Similarly, vegetation cover in household gardens has been positively correlated to socio-economic status in a number of other studies. In Brisbane, Australia a total of 592, 1 km x 1 km grid squares were created to characterise both public and private land by socio-economic status using data from the Index of Relative Socio-economic Disadvantage (IRSD) (Shanahan, Lin, Gaston, Bush, & Fuller, 2014). Shanahan et al. (2014) used satellite imagery to record percentage tree cover within each grid square, recording both total tree cover and exclusively native tree cover. The study was limited in terms of the scope of the urban form considered as it only included detached dwellings, these being the most dominant form in the area. Both total tree cover and native tree cover were found to have a positive correlation with the socio-economic index, $R^2 = 0.31$, $p = <0.001$ and $R^2 = 0.47$, $p = <0.01$

respectively. The index used combines income, education, and ethnicity to create an IRSD value for each grid square, placing neighbourhoods on a continuous scale. Similarly, Kendal, Williams, and Williams, (2012) tested household income, proportion of graduates, mean resident age, proportion of renters, and population density variables against tree cover variation in residential front yards in Ballarat, Australia (n = 128) and drew a similar conclusion. The proportion of graduates was found to have the strongest positive relationship with tree cover in gardens. The authors' conclusion was that social stratification along a continuous model, and not a lifestyle behaviour model, is the best predictor of tree cover variation. This contradicts the findings of Grahn and Stigsdotter, (2003) and Grove et al. (2006) where, as stated above, it was found that descriptive characteristics including a range of socio-economic factors best described vegetation cover variation, not a continuous model. This contradiction requires further exploration. Furthermore, the effect of a time lag was found to be significant by Luck et al. (2009), this temporal element is missing from Shanahan et al. (2014) which may contribute to the weak correlations in the results.

Socio-economic factors have also been shown to explain variation in biodiversity across household gardens. The luxury effect can be attributed to two main factors – either wealthier households have the freedom of mobility to choose to purchase in more aesthetically pleasing neighbourhoods or they simply have more disposable income to invest in increasing species richness within their private land (Chamberlain, Henry, Reynolds, Caprio, & Amar, 2019; van Heezik, Freeman, Porter, & Dickinson, 2013). This positive association between plant diversity and wealthier neighbourhoods has been demonstrated in New Zealand (van Heezik et al., 2013), Australia (Luck, Smallbone, & O'Brian, 2009), South Africa (Chamberlain et al., 2019), and Burundi (Bigirimana, Bogaert, De Canniere, Bigendako, & Parmentier, 2012). A study combining data from two urban districts, in New Zealand and Germany, used a citizen science method (n = 261) to assess the link between biodiversity, nature connectedness and mental wellbeing in gardens during the COVID-19 lockdowns (Samus et al., 2022). Participants in the study self-reported the size of their garden, the abundance of 16 physical and 13 flora features, their gardening participation rate, and their mental wellbeing. It was found stronger nature connectedness was associated with higher feature richness and nature connectedness had positive effects on mental wellbeing by increasing the frequency of positive emotions (Samus et al., 2022). Feature richness was also significantly affected by garden size in both cities (Samus et al., 2022). However, as highlighted by the researcher a main limitation of the study is the subjective nature of citizen science which is not as comprehensive or accurate as surveys by trained ecologists (Samus et al., 2022).

Fewer studies have exposed the luxury effect on animals than on plants (Leong et al., 2018). One way humans can indirectly affect mobile taxa such as birds is by choosing which plant species are within their gardens and eliminating others – being active landscape managers of their own private habitats (Leong et al., 2018). The abundance and diversity of native bird species has been shown to be positively correlated to increasing socio-economic status in Phoenix, USA (Kinzig, Warren, Martin, Hope & Katti, 2005, Lerman & Warren, 2011) and Vancouver, Canada (Melles, 2015), as well as higher overall bird diversity with increasing wealth in Chicago, USA (Loss, Ruiz, & Brawn, 2009). Birds living in urban environments depend upon plant species for food, roosting, and nesting; those choosing these located within gardens would, therefore, be directly affected by the residents' management choices. As there is greater evidence for the luxury effect on vegetation, it can be assumed this is a contributing factor to the uneven distribution of birds highlighted above.

Within a private setting, the composition of private gardens has been shown to directly affect bird species abundance and the number of breeding pairs in Poland (Kos, Bujoczek, & Bujoczek, 2021). The presence of birds in gardens can have a two-way effect on residents. Residents increase wildlife encouraging activities (such a bird feeding) when more birds are present. Then, the increase in feeding would increase the number of birds, and so on (Kos et al., 2021). To investigate how differing garden composition affects bird species composition, Kos et al. (2021) conducted 34 in-person surveys on a variety of urban garden types in Southern Poland. The sample size was small, and only included detached and small farm-like dwellings. Gardens were sampled non-randomly to allow for different compositions to be chosen. A total of 48 bird species were recorded along with a number of garden features including: tree density, plant species, the presence of artificial and natural lawn, the presence of animals both agricultural and domestic, bird feeders, and building walls for nesting. Tree density was found to significantly increase the number of breeding pairs and the number of birds present in the gardens in autumn and winter. The highest species composition was also found in gardens which were most structurally rich, creating a large number of niches for a variety of bird species (Kos et al., 2021). Kos et al. (2021) conclude that with the increasing homogeneity of gardens seen in the urbanising Western world the damaging effect on urban birds is likely to increase as gardens become increasingly more uniform and housing developments more dense.

Fair access to greenspaces is paramount for the whole of society to benefit (Brown et al., 2018; Maas et al., 2009; Mears et al., 2019). However, socio-economic factors have been shown to affect time spent and frequency of gardening, the vegetation cover within gardens and public spaces, and the species richness within both public and private land. There is a gap within environmental injustice research to understand the effect socio-economics may have on a residents access to a quality private garden. As there is strong evidence for a positive relationship between socio-economics and vegetation cover within public and private land (Kendal, Williams, & Williams; 2012; Shanahan, Lin, Gaston, Bush, & Fuller, 2014), it is expected similar results will be seen for the frequency of vegetation types and for the quality or diversity of a garden, measurements not yet reported within the literature.

The diversity of features within a household garden is yet to be researched within the realm of environmental injustice. Evidence suggests biodiversity is strongly associated with household characteristics (Chamberlain et al., 2019; van Heezik, Freeman, Porter, & Dickinson, 2013); thus, feature diversity within a garden would likely have similar results. As will be demonstrated below, Section 2.6, the quality and diversity of a green view has been shown to have multiple positive effects on human health and wellbeing. Therefore, it is important that variations due to the socio-economics of the household are explored to expose the currently under-researched environmental injustice within gardens. Data on the diversity of a household garden are key, both the vegetation and feature diversity, as these can directly influence the quality of the greenspace and the benefits residents receive. As mentioned within the first paragraph of this literature review, the definition of environmental justice focuses on viewing injustice through a lens of accessibility. As gardens are often an urban residents most accessible resource, their contribution is vital. Ethnic minority communities have reported viewing quality public greenspaces as a luxury that they cannot afford (Birch et al., 2020); it is likely a private garden would be viewed similarly: an unacceptable environmental and social injustice. The next section will explore how UK policy has sought to mitigate the environmental injustices seen in private greenspaces, and what more could be done.

## 2.3 Household gardens in UK policy

The UK's Department of Food and Rural Affairs (DEFRA) published their 25-Year Environment Plan (25-YEP) in January 2018, outlining their vision to 'leave our environment in a better state than we found it' (DEFRA, 2018, p. 6). Such policies are motivated by the growing global recognition of the importance of urban greenspaces to people (Liotta et al., 2020). DEFRA's initial plan indicates the need to 'scope out how we could connect people more systematically with greenspace … for preventative and therapeutic purposes' (DEFRA, 2018 p. 73). The 25-YEP was met with reserved optimism by stakeholders. The World Wildlife Fund (2018) and Wildlife and Countryside Link (2019) described the plan as a strong start and applaud the Government's green intentions, but both express concerns regarding the length of time before action is taken. Further, while the UK's exit from the European Union can be viewed as an opportunity to establish new environmental policy without the restraints of the European Union (Cowell, 2018), it has been an undeniable, time-consuming set-back.

The plan has been visionary in the instigation of cross-departmental approaches between DEFRA and the National Health Service (DEFRA, 2018; NHS, 2019). The promotion of health and wellbeing feature heavily throughout the report. The main emphasis on actions, however, are within the realm of public GI and private greenspaces are under-represented. The term 'park' is used 37 times throughout the report; in comparison 'garden' is used 10 times. Six of the uses of the term 'garden' are regarding: household garden gardening relating to healthcare (Table 2.1, A and B); encouraging contact with nature in children (Table 2.1, C and D); and being an important part of green infrastructure in a city (Table 2.1, E and F). The other four relate to public gardens and peat-free practices. Chapter 3: Connecting people with the environment to improve health and wellbeing, states a number of actions the report pledges to take. Within these the public realm is mentioned within the areas of green prescribing, volunteering, community forests, care farming, Parks Action Group, GI improvement with local authorities and urban tree-planting. In comparison, household gardens are not mentioned within any proposed action. Quotation D (Table 2.1) states the importance of an alternative urban greenspace in the absence of a household garden, but as 87 per cent of households have access to a garden (Davies et al., 2009), this bias is arguably an oversight.

Table 2.1: The quotations of the seven times household gardens are mentioned within the 25-Year Environment Plan (DEFRA, 2018).

| Quote Reference: | Page Number: | Quotation: |
|---|---|---|
| A | 72 | ‘Some health professionals have adopted a practice known as ‘green prescribing’, a type of social prescribing where nature-based interventions are used to treat people with health conditions. Examples of interventions include gardening, conservation, care farms and green gyms.’ |
| B | 73 | ‘We will consider how NHS [National Health Service] mental health providers in England can establish new working arrangements with environmental sector organisations to offer appropriate therapies – such as gardening, outdoor exercise and care farming – in natural settings to people with mild to moderate mental health conditions and who may be struggling to overcome loneliness and isolation.’ |
| C | 75 | ‘Playing and learning outside is a fundamental part of childhood, and helps children grow up healthy. Some children are lucky enough to have a family garden; others will not, and it is important that we find other ways to give them better access to the great outdoors.’ |
| D | 75 | P75 ‘The initiatives we outline below are designed to encourage and support outdoor activities, particularly where a child has no access to a family garden.’ |
| E | 79 | P79 ‘The Mersey Forest programme in the North West is creating greenspaces. As England’s largest Community Forest, the Mersey Forest partnership have planted more than nine million trees creating a 1300 $km^2$ network of woodlands, open spaces, urban gardens and street trees in some of the most disadvantaged areas of Merseyside and Cheshire.’ |
| F | 147 | ‘Around 60 per cent of Kingsbrook will be green infrastructure, including 250 acres [101 hectares] of accessible, wildlife-rich open space, orchards, hedgehog highways, newt ponds, tree-lined avenues, fruit trees in gardens, bat, owl and swift nesting boxes and nectar-rich planting for bees.’ |

In March 2019 the first annual progress report was published, providing an update on progress of the 25-YEP from January 2018-March 2019 (DEFRA, 2019). Within its first year the plan has focused on policy, arguably a necessity due to the uncertainty around Brexit. The progress made is again focused on the realm of public GI, with few direct mentions of household gardens. Regarding biodiversity net gain, it states '… new houses meet the needs of people, while contributing to ecological recovery and enriching the quality of local greenspaces. […] delivery of much needed housing and infrastructure will not be at the expense of the environment' (p. 3). Although this does highlight a new focus on UGI when development planning, it fails to comment on how household gardens may change in the future. The report outlines additional funding for three main greenspace initiatives, all of which are within public spaces: £13.1 million has been provided to support urban parks and GI, £10 million to provide disadvantaged children with better access to nature both in and out of school, and a further £10 million has been announced for urban tree planting (DEFRA, 2019). This is also the case for wider funding streams, which focus explicitly on expanding UGI within the public realm, for example The National Lottery and Esmee Fairbairn Foundation.

Throughout the 2019 progress report household gardens are only mentioned twice. The first, within the 'enhancing biodiversity' goal it states 'by strengthening biodiversity we can better protect the nations animals, cultivated crops, wild plants, trees, forests, amenity plants, gardens, and ecosystems from pests, diseases, and non-invasive species' (p. 58). Although this does recognised gardens as an important asset in biosecurity and native species conservation, it does not link them to public health and wellbeing as the original report stated. The second mention, however, does include gardens as an important way people can contribute to the 25-YEP. The Year of Green Action is an initiative set up to provide a focal point for businesses and individuals to learn about their impact on the environment and how they can reduce it throughout 2019. As part of the initiative a 'demonstration garden' has been established to 'illustrate actions individuals can take to contribute to the 25-YEP goals of thriving plants and wildlife, biosecurity, enhanced engagement with the natural world and the sustainable use of natural resources' (p. 70–71).

Household gardens continued to be rarely mentioned in subsequent updates. In fact, in the two annual updates published March 2020 and March 2021 household gardens were not mentioned. The reports heavily focused on climate change mitigation, with updates on clean air, waste recycling, and biosecurity (DEFRA 2020; DEFRA 2021). In the 2021 progress report, a £5.77 million cross-governmental project was outlined to test nature-based social prescribing running Oct 2020 to April 2023, highlighting the changing attitudes of policymakers towards the importance of using the environment as a healthcare tool. But, the potential for gardens was missed.

Urban greenspaces supported 2.1 million people to meet their weekly physical exercise needs, equating to 90,000 extra Quality of Life Years in 2017 valued at £1.4 billion in saving to the healthcare system (DEFRA, 2021). Quality of Life years or Quality-adjusted life years (QALY) are ‘a measure of the state of health of a person or group which benefits, in terms of length of life, are adjusted the reflect the quality of life. One quality-adjusted life year is equal to one year of life in perfect health’ (NICE, n.d.). But, access to UGI is not equal with barriers caused be a number of demographic factors which in turn exacerbate disparities in quality of life (DEFRA, 2021). Yet, in the follow-on long-term plans to improve the access to the natural environment, household gardens are not mentioned. The long-term plans include: helping people improve their health and wellbeing by using green spaces; encourage children to be close to nature; planting 1 million urban trees; connect more people of all backgrounds with national landscapes (DEFRA, 2021). Each of these long-term plans links to the need for environmental justice, yet gardens are not mentioned as a metric to encourage people into nature. With gardens being often the most accessible UGI, particularly for those without funds to travel, it is an oversight.

The March 2022 annual report of the 25-YEP gave a similar update, with progress being made ‘enhancing beauty, heritage, and engagement with the natural environment (DEFRA, 2022, p. 7) in public UGI but no mention of private gardens. The green social prescribing programme launched seven test and learn sites with 1,500 referrals to nature-based activities; the success of the programme has led to national upscaling (DEFRA, 2022). An additional £30 million was declared to be invested over a 3-year period to support health, wellbeing, and the environment with a focus on expanding green routes (DEFRA, 2022). The annual progress reports on the 25-YEP show a clear trend in the priorities of stakeholders: public UGI. Although gardens were mentioned after the first year, with regards to biodiversity and sustainability, they were no longer included in further updates. Whilst the funding into public UGI is vital, this thesis has shown and will continue to show the importance of household gardens in mitigating the same health and wellbeing issues being tackled by UGI.

As part of the 25-YEP, four, three-year pioneer programmes were established by DEFRA to test new tools and methodologies for investing in and managing the natural environment. One of these four, the Urban Pioneer, was established in Greater Manchester in 2018. The vision of the Urban Pioneer was:

> 'To make a clear and evident contribution to Greater Manchester's natural environment, engaging and connecting people with nature, maximising their health and economic benefits through investment in the environment, creating sustainable growth and a good quality of life.' (retrieved from webpage, 05/09/2019).

The Urban Pioneer was led by the Environment Agency and consisted of partners from across the public and private sectors as well as academia. These included Lancashire Wildlife Trust, Bruntwood, City of Trees, Forestry Commission, Greater Manchester Combined Authority, National Trust, Natural England, Royal Horticultural Society, Salford Council, University of Salford, and University of Manchester (Nature Greater Manchester, n.d.).

In its infancy the Urban Pioneer did not mention household gardens within their accounting of greenspaces, nor were they a focus within their gathering of evidence or proposed actions. This thesis has contributed such evidence which has been included within The Natural Capital Approach in Greater Manchester 2019 (Urban Pioneer, 2019). The My Backyard project, a two-year project that took place between 2016–17, has also become a contributor to the Urban Pioneer. It aimed to develop a new understanding of the benefits that household gardens provide to the residents of the City of Manchester, focusing on physical ecosystem benefits (Cavan et al., 2018). The My Backyard project reported figures relating to rainfall absorption and surface temperature, whereas the project reported in this thesis focuses on a detailed sample of gardens across Salford to build a detailed picture of landcover and garden content. . The My Backyard Project is discussed in greater detail in Section 2.5.

5. The Urban Pioneer was a success in that it has acted as a catalyst to a natural capital approach to environmental policy and initiatives on a national scale. It was also successful in

encouraging a collaboration of several organisations, the outcome being a coherent and inclusive account of services using a thematic, understandable approach to delivery (Environment Agency, 2020). At a more local scale, the Urban Pioneer directly fed into the GMCA 5-year Environment Plan for Greater Manchester 2019-2024. Within which increasing residents' engagement within their gardens, particularly for encouraging wildlife and tree planting, is mentioned as an important action to improve the local natural environment (GMCA, 2019).

The UK government published their new guidelines for planning in 2020 in a white paper entitled 'Planning for the Future' (Ministry of Housing, Communities & Local Government (MHCLG), 2020) within which the then Prime Minister, Rt Hon. Boris Johnson MP, called for 'radical reform (of the planning system) unlike anything we have seen since the Second World War … that actively encourages sustainable, beautiful, safe and useful development' (MHCLG, 2020, pg. 6). One aspiration of the new system is to support home ownership 'helping people and families own their own beautiful, affordable, green and safe homes, with ready access to better infrastructure and green spaces' (MHCLG, 202, pg. 14). The White paper goes on to state 'for our children and grandchildren, our reforms will leave an inheritance of environmental improvement – with environmental assets protected, more green spaces provided…' (MHCLG, 2020, pg. 21). The above statements are examples of how the UK government continues to make the environment a priority, but as with much of policy mentioned in this section, household gardens have minimal mention. Gardens are, however, mentioned within the context of development of renewal areas. 'Local authorities could continue to consider the case of resisting inappropriate development of residential gardens' (MHCLG, 2020, pg. 24). Increasingly, front gardens have been being paved over as car ownership becomes more of a norm and manicured gardens a thing of the past, with multiple detriments to the environment including increased local flooding (Smith et al., 2011; Warhurst, Parks, McCullock, & Hudson, 2014).

As has been demonstrated, the 25-YEP, and in its infancy the Urban Pioneer, focused on public green infrastructure. As a public or collective good, parks and other community greenspaces are a non-excludable resource managed by a political process (Savas, 2000).

Thus, governing bodies have authority over management and can choose to implement new strategies, for example those of the 25-YEP. In contrast, household gardens are an excludable resource as residents have autonomy of access and management. In economics, public parks are known as a quasi-public good, they are subjected to a degree of rivalry, extra consumers reduce space for others (Riley, n.d.). It is important that environmental stakeholders do not exclude gardens from policy or initiatives, the current administration has demonstrated this intent within the White Paper discussed above. While it may not be possible to directly manage gardens, they are an important resource to reduce rivalry and practices within the public sector will often influence the private one. Urban policies often lack appropriate strategies to protect urban gardens despite their potential to support spatial, social, and environmental justice (Ferrari, et al., 2023).

## 2.4 Garden Amount

Widescale migration into urban landscapes and the resulting increase in population density means household gardens are an ever more vital greenspace in the land-use of a city. In Stockholm, Sweden it is estimated that 16 per cent of landcover is household gardens (Colding, 2007). In Edinburgh, Belfast, Leicester, Oxford, and Cardiff (UK) household gardens contribute between 22 and 27 per cent of the urban area within relevant local authority administrative boundaries (Gaston et al., 2005; Loram et al., 2007; Tratalos, Fuller, Warren, Davies & Gaston, 2007). In Belgium, gardens take up 42 per cent of urban greenspace in the Brussels Capital Region (Van de Voorde, Vlaeminck, & Canters, 2008). In New Zealand 36 per cent of the total urban area are household gardens (Mathieu et al., 2007). Although there is variation in how the landcover of household gardens is reported, and many publications are out-dated, it is clear they are a significant spatial component of urban areas and are an important resource in urban areas globally.

In Great Britain, whilst approximately 5 per cent of urban land cover is publicly accessible greenspace, almost 30 per cent is private garden cover (Bell et al., 2020). Household gardens were named the third most important factor when purchasing a house, location and property size being first and second (Mason, 2017). Having a garden or outdoor space is often a key factor in house pricing (Adams & Tiesdell, 2012): 62 per cent of clients surveyed valued this commodity when searching for their perfect property (Mason, 2017). In the UK, garden size is getting smaller (Chalmin-Pui et al., 2021), with some new developments even omitting gardens completely instead investing in other forms of greenspace (Douglas et al., 2019). City-level data on garden extent in the UK is out-dated, with current publications being dated in 2005 and 2007 (Gaston et al., 2005; Loram et al., 2007; Tratalos, Fuller, Warren, Davies & Gaston, 2007). Recent publications in the field have used these dated garden extent estimates (Garcia-Antunez., et al., 2023; van Heezik et al., 2020), showing the need for a more contemporary overview. Also, what has yet to be published is an account comparing the extent of gardens in neighbouring regions; this would give an indication of how the resource varies within a similar topographic location. This thesis includes such an account for the districts of Greater Manchester.

## 2.5 Garden Characteristics

Gaston et al. (2005) published the first detailed audit of features within private gardens in the UK. The research highlighted the scarcity of quantitative data on garden content stating it is likely due to problems of systematically obtaining data on such a fragmented and inaccessible resource. To rectify this, the authors used a telephone survey of 250 households combined with Ordnance Survey mapping. In the survey, participants were questioned on: whether they had a garden; the number of domestic cats they owned; the presence of a pond, a bird nest-box, and/or a compost heap; the number of trees >3 m; and the time spent working on the garden in an average summer week. The mean area of a domestic garden was reported at 151 $m^2$. From these results, it is estimated that 60 per cent of the total area of gardens in Sheffield is lawn. Regarding garden content, 14 per cent of gardens had ponds, 26 per cent had nest boxes, 29 per cent had compost heaps, and 48 per cent had trees more than 3m tall. A similar estimate was given by Hessayon & Hessayon (1973) for the lawn coverage in gardens across Britain. These figures on garden composition were the first published for a city region in the UK. However, this research is now out-dated and was based on a small sample size. Estimates produced from a sample size of 250 are arguably not robust enough to produce a model for city-wide garden composition given the heterogenous nature of gardens.

To extend the geographic extent of the Sheffield study conducted by Gaston et al., (2005), Davies et al., (2009) took a large-scale multifaceted approach using both mapping and surveys. Combining 12 datasets from seven urban and suburban areas across the UK, they collated previous surveys, data from telephone surveys, GIS analysis, postal questionnaires, and garden visits to create a nation-wide inventory of garden resources. This national study combined datasets with sample sizes ranging from 250 gardens to 19,913 households. From their results, mean garden size was 190 $m^2$. Additional findings on the occurrence of features within gardens were: 51 per cent provided supplementary food for birds; 16 per cent had a bird nest box; 10 per cent had a pond; and 54 per cent contained at least one tree >3 m, equating to, on average, 11 per cent tree cover within a garden. As the study was conducted with a focus on wildlife engagement and the activities residents were undertaking it does not give a full picture of their natural capital stock. As noted by Davies et al. (2009) previous garden content research has been done investigating: the biodiversity of individual gardens; the presence and abundance of particular taxonomic groups; and the occurrence of multiple taxonomic groups in multiple gardens. Further research is required to understand how the habitat features provided within gardens vary and what affects, to both their users and the environment, this may have.

Previous work by Gaston et al. (2005) and Davies et al. (2009) have provided some insight into the extent and characteristics of household gardens in the UK. While the survey size of Gaston et al. (2005) was small, their results for mean household garden size and occurrence of trees above 3m tall were similar to those reported by Davies et al. (2009) who used a far more representative sample. However, both studies focused on wildlife engagement activities and did not record household garden content beyond total household garden size, trees over 3m tall, and lawn cover (the latter only by Gaston et al., 2005), and, as previously stated, the sample was small. Further, neither study assessed variation caused by socio-economic characteristics, so cannot be used to draw conclusions regarding the environmental injustice within the private urban greenspace sector. Within the research field there is a gap in the knowledge which balances the depth of detail needed within the content of household gardens and the breadth of sample size needed to capture their heterogeneity, a gap this thesis aims to contribute to.

## 2.6 Urban Form: affecting garden form and function

The urban form of an area has been reported to contribute to the great variation in both form and function of household gardens (Dewaelheyns, Rogge, & Gulinck, 2014; Gaston et al., 2005; Loram, Tratalos, Warren, & Gaston, 2007; Loram, Warren, & Gaston, 2008). Urban form is 'the physical characteristics that make up built-up areas, including the shape, size, density and configuration of settlements' (Williams, 2014, p. 6). A better understanding of the relationship between urban form and land cover could assist with both development policy and in the creation of more ecologically friendly cities (Conway & Hackworth, 2007). Some contemporary urban development has focused on a 'compact city' approach, aiming to centralise services and reduce land take (Williams, Burton, & Jenks, 2000). This approach reduces costs in infrastructure benefiting sustainability goals by increasing public transport demand and by freeing up city budget for spend on environmental gains (Haughton & Hunter, 2003). But, this increasing densification has come at a cost as in the UK to facilitate the need for more urban housing the average house size has decreased (Office of Deputy Prime Minister, 2002) and urban land prices have inflated (Williams, Burton, & Jenks, 2000). In England, 4600 hectares of household garden space was converted to other uses between 2013–2017 (Brindley et al., 2018). Tratalos et al., (2007) states likely ramifications of this are the deterioration of ecosystem services, declines in biodiversity, and a reduction in the quality of life in residents. Therefore, an ecological dimension of urban land-use planning is increasingly important (Arshad & Routray, 2018).

The field of research into urban form, also termed urban morphology, is broad with an extensive history. Larkham (2006) provided a detailed history of research within the field pre-2000s within Great Britain. The first urban area or 'townscape' was a result of the Second World War which led to a new approach in land planning (Larkham, 2006). Rapid population growth in the 19$^{th}$ century resulted in large areas of dense, poor quality terraced housing. Since the second half of the 20$^{th}$ century much of the older dense housing has been cleared and replaced by diversified housing and high-rise apartments or flats (Britannica, n.d.; Kostourou, 2021). A modern-day urban landscape contains a great deal of variation in housing densities, property types and ages, which has produced a heterogeneity of gardens attracting urban form researchers.

Micromorphology – studies of form at the level of elements of individual houses – have been shown to influence environmental stock (Whitehand, Morton & Carr, 1999). Globally, the age of houses positively influenced neighbourhood tree and understory cover in Australia (Lin et al., 2017) and garden size doubled from terrace to semi-detached to detached housing in Belgium (Dewaelheyns et al., 2014) and Australia (Gill et al., 2008). Yet, studies into household gardens in the UK are lacking, in particular those which assess the extent and quality of the resource (Baker & Smith, 2019; Cameron et al., 2012). This lack of data has led to the Green and Blue Infrastructure Strategy for Manchester over-estimating the proportion of permeable land in household gardens across Manchester, UK, it was assumed 100 per cent of gardens were greenspace, but this estimation has been shown to be high with others reporting 50 per cent green landcover with the other half being occupied by hard landscaping features such as patios, decking and artificial lawns (Baker, Smith, & Cavan, 2018; Baker & Smith, 2019). The cumulative impact of widescale paving on flood risk has been well-documented (Kelly, 2016; Perry & Nawaz, 2008; Scott, Shannon, Hardman, & Miller, 2014).

Previous research on the effects of urban form on gardens has taken two different approaches to methodology: small-scale surveys or large-scale GIS analysis. Loram, Warren, & Gaston (2008) conducted 267 in-field surveys of household gardens across five UK cities: Edinburgh, Belfast, Cardiff, Leicester, and Oxford. They measured each garden to the nearest 0.5 m to find the total garden area, then researchers drew a scaled map of the garden estimating the land cover types. Land use richness is recorded as the number of land use types per garden (listed in Table 2.2). Mown grass and patio were found to have the greatest mean cover with 85.5 $m^2$ and 38.6 $m^2$ respectively (Table 2.2), a trend seen across all house types. A strong, positive, linear correlation was reported between garden size and the land use richness in three of the five cities (Belfast $r = 0.66$, $df = 53$, $p = <0.0001$; Cardiff $r = 0.61$, $df = 52$, $p = 0.0001$; Leicester $r = 0.73$, $df = 47$, $p = 0.0001$). The occurrence of some land uses, in particular mown grass, trees >3 m, uncultivated, pond, vegetable patch, greenhouse, and unmown grass were also positively related to garden size ($p = <0.01$), thus influencing diversity within gardens. House age was found to have a weak effect on a small number of land use richness features: cultivated boarders, sheds, and compost heaps. With increasing house age, canopy >3 m tall increased in both area ($r^2 = 0.13$, $p = <0.03$) and proportional contribution ($r^2$ = 012, p = <0.05). However, the study size was relatively small – approximately 50 houses from each city were selected which equated to a total sample size of 267. Also, the selection of sample houses was arguably biased. The authors used houses derived from contacts within each city. While these were derived from a wide cross-section of employment levels the selection was still biased towards people more attuned to academic research and disregards those who are unemployed or retired. Further, vegetation data were omitted from the mean area land cover data (Table 2.2) as count data were recorded for trees categorised by height. As a result, the land cover data cannot provide an extensive account of the composition of a garden.

Table 2.2: The mean area (and ± standard error) of individual land use types within gardens, across all house types over five cities. Other features represent unique features found in some properties, for example a railway carriage. In total 18 features were recorded, no data were given for Cultivated border, Trees >3 m (as this was count data), Uncultivated, Vegetable patch, and Linear features (Loram, Warren, & Gaston, 2008, p. 368).

| **Land Cover Type** | **House Type** | | | |
|---|---|---|---|---|
| | **All House Types ($m^2$)** | **Detached ($m^2$)** | **Semi-Detached ($m^2$)** | **Terraced ($m^2$)** |
| Garden area | 197.5 ± 11.4 | 315.0 ± 25.1 | 188.5 ± 14.4 | 93.5 ± 7.2 |
| Patio | 38.6 ± 1.9 | 48.8 ± 3.6 | 38.0 ± 2.7 | 29.4 ± 3.3 |
| Mown grass | 85.5 ± 7.2 | 126.1 ± 11.6 | 76.0 ± 8.4 | 31.4 ± 3.6 |
| Sheds | 6.7 ± 0.4 | 8.6 ± 0.8 | 7.2 ± 0.8 | 5.0 ± 0.7 |
| Path | 17.1 ± 1.3 | 27.4 ± 3.1 | 13.9 ± 1.5 | 12.6 ± 1.6 |
| Compost | 2.1 ± 0.4 | 3.1 ± 1.3 | 1.9 ± 0.4 | 1.1 ± 0.3 |
| Gravel/Bark chipping | 20.4 ± 3.0 | 32.4 ± 8.2 | 14.7 ± 2.8 | 16.1 ± 4.5 |
| Decking | 14.2 ± 1.6 | 5.8 ± 2.1 | 19.5 ± 2.6 | 10.3 ± 2.5 |
| Pond | 4.6 ± 0.6 | 4.9 ± 1.1 | 3.9 ± 0.6 | 5.9 ± 2.2 |
| Other Features | 4.2 ± 1.1 | 6.8 ± 2.8 | 3.4 ± 1.0 | 1.2 ± 0.2 |
| Greenhouse | 5.8 ± 0.5 | 6.9 ± 0.9 | 4.4 ± 0.5 | 7.1 ± 1.1 |
| Garage | 18.4 ± 1.1 | 19.5 ± 2.2 | 17.2 ± 1.7 | 19.0 ± 1.5 |
| Water butt | 1.1 ± 0.1 | 0.8 ± 0.3 | 1.2 ± 0.2 | 1.0 ± 0.2 |

The My Backyard project, a two-year project that took place between 2016–17, aimed to develop a new understanding of the surface run-off and social benefits that household gardens provide to residents of the City of Manchester (Cavan et al., 2018). Data were collected using an online survey completed by residents, with over 1000 responses, and was validated using aerial images. Additionally, computer modelling was used to investigate how gardens effect cooling and the absorption of rainfall. The results showed the City of Manchester had a total household garden area of 24 $km^2$, with only 12 $km^2$ of this greenspace. The modelling produced estimates of up to 6 per cent less rainfall absorption and up to a 4°C increase in surface temperature than previously estimated. Household gardens of detached and terraced houses constitute 80 per cent and 40 per cent greenspace respectively. The study does not investigate whether these differences in greenspace are solely due to property type, or whether socio-economic factors influence natural capital.

Taking a large-scale mapping approach, Loram et al. (2007) used five major cities across the UK to explore variation in garden size according to property type. A minimum sample size of 500 gardens from each of the five cities was used (Edinburgh, Leicester, Oxford, Cardiff, and Belfast). Although they reported terraced houses as the most numerous property type across all cities, semi-detached (Leicester, Oxford, and Cardiff) or detached (Edinburgh and Belfast) houses made the greatest total contribution to garden area. They found, on average, garden size doubled from terrace, to semi-detached, to detached houses, with combined median garden sizes recorded at 84.0$m^2$, 213.5$m^2$, and 364.8$m^2$ respectively. In addition, there was a weak to moderate positive correlation between individual garden area and the ground area of the house ($r^2$=0.24-0.57, df=513-557, $p<0.001$). Edinburgh and Oxford were regarded as the most affluent cities and these reported the largest mean garden size, 194.5 $m^2$ and 209.7 $m^2$ respectively.

In their study across five cities in the UK (Edinburgh, Glasgow, Leicester, Oxford, and Sheffield), Tratalos et al. (2007) found high density urban developments are associated with poor environmental performance, recorded through rainwater run-off and carbon sequestration. They surveyed three sites, each containing 2000 households, from each of the five cities to cover inner city, outer suburbs, and a mid-way point of urban forms. The proportion of property type (detached or semi-detached) was found to have significant relationships with carbon sequestration ($r^2$=0.55 p=<0.05), cover by greenspace ($r^2$=0.29, p=<0.05), and cover by gardens ($r^2$=0.5, p=<0.05) at Output Area level independent of housing density. 'Output Areas (OAs) are the lowest level of geographical area for census statistics and were first created following the 2001 Census' (ONS, 2021a), an individual

Output Area has a mean population of 300 and a minimum of 100 (Table 3.2). The relationship between the increased proportion of detached housing and increased cover by gardens has been reported previously by Loram et al. (2007). However, comparing the effects of the proportion of property type within an Output Area with the proportion of greenspace cover and the amount of carbon sequestration has novel implications. As these results were found independent of housing density it suggests housing type had a direct impact on ecological performance. This study illustrates the effective use of Output areas as geographic segments for analysing landcover relating to urban form.

Baker and Smith (2019) identified four property types to assess variation in garden size: detached, semi-detached, modern terraced, and Victorian terraced (pre-1920) in Leicester, UK. Using satellite imagery from the Worldview-2 satellite sensor at approximately 2 m resolution and Ordinance Survey MasterMap they classified garden landcover as either grass, shrubs, non-vegetative, or building surfaces. It was found the proportion of useable greenspace (grass and shrub categories combined) differed substantially between Victorian terraced housing, 34 per cent, and all other property types: non-Victorian terraced 63 per cent, semi-detached 62 per cent, and detached 64 per cent (Figure 2.5). Baker and Smith (2019) suggest this is a possible indication of environmental deprivation within household gardens in the UK. This study provided a unique analysis into how property type and development era can affect household garden content. Comparing results in Table 2.2 taken from Loram et al. (2008) shows gardens in Leicester had significantly less green area, with green area in this study only including grass as vegetation was not included in the study. Green area in Loram at al. (2008) were reported as: detached 40 per cent, semi-detached 40 per cent, and terraced 34 per cent. Interestingly, terraced housing in Loram at al. (2008) and Victorian terraced in Baker and Smith (2019) both have 34 per cent green area.

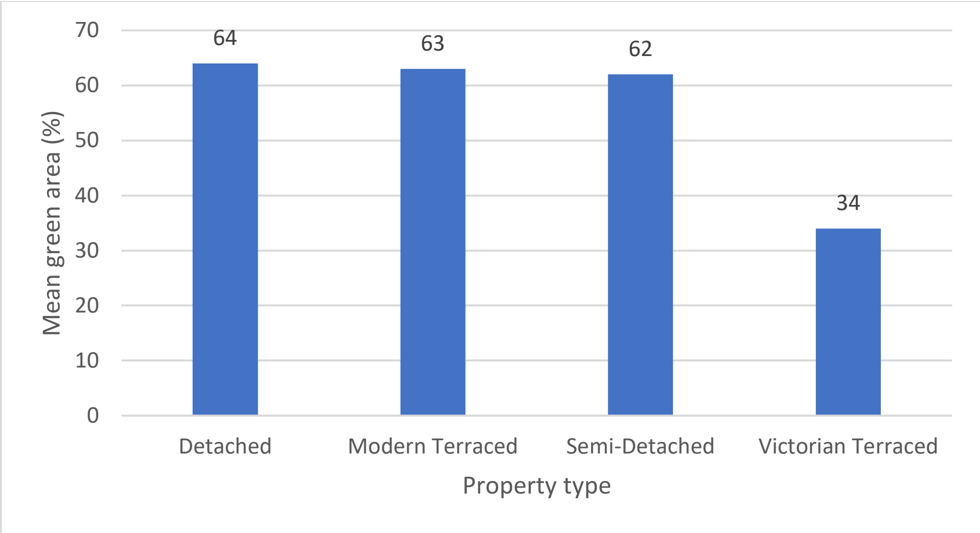



Figure 2.5: Mean useable green area (combined grass and shrubs) per property type (Baker & Smith, 2019, p. 143).

Baker and Smith (2019) have shown the development era of a property to be a key factor in the proportion of useable greenspace within gardens, evidenced by the significant difference between pre- and post- 1920s terraced (Figure 2.5). Yet, the other two classifications, semi-detached and detached, do not account for a temporal change, a key limitation of this study. Also, despite having four categories of landcover, there was only a discussion of the implications for useable greenspace within gardens and not the diversity of features or vegetation cover. There is no national policy in England concerning the size of gardens in new developments, rather this is determined by individual local authorities. Many choose to use the standard set out by the London Housing Design Quality and Standards which states ‘a minimum of 5 m$^2$ of private outdoor space should be provided for one-to-two person dwellings and an extra 1 m$^2$ … for each additional occupant (Mayor of London, 2010, p. 54). However, other local authorities have larger requirements, such as the Essex Planning Officers Association (2018) at 100 m$^2$. Further, some set the standard in length, with garden size for a three-bedroom semi-detached at 10 m in length according to Rother District Council, (2017). The City of Salford does not give clear guidance on a minimum outdoor space, with their Local Development Plan focusing on diversified development with different house sizes at a density high enough to meet growing demand (Salford City Council, 2023).

In summary, to build on the work of Tratalos et al. (2007) and Baker and Smith (2019) further investigation is needed into how micromorphology's, property type and development age for example, affect the content and thus quality of household gardens as a resource for residents' benefit. To achieve this, a large sample size is required due to the heterogenous nature of household gardens, but at a scale whereby individual land cover can be recorded.

## 2.7 Benefits of household gardens

### 2.7.1 Health and Wellbeing

Household gardens provide both ecosystem and health benefits to residents and the wider community (see Figure 2.6); see Cameron et al. (2012); Douglas, Lennon, and Scott (2017); and Soga, Gaston, and Yamaura, (2018) for an extensive review of population level affects. Fewer studies assess the benefits of individual private household gardens to their residents, likely due to their heterogeneity and a lack of data (Brindley et al., 2018; Siftova, 2021). Often, studies assess UGI as a collective, though many of the benefits can also be attributed to gardens (Raymond et al., 2018). For the purpose of this study a number of these benefits are reviewed below.

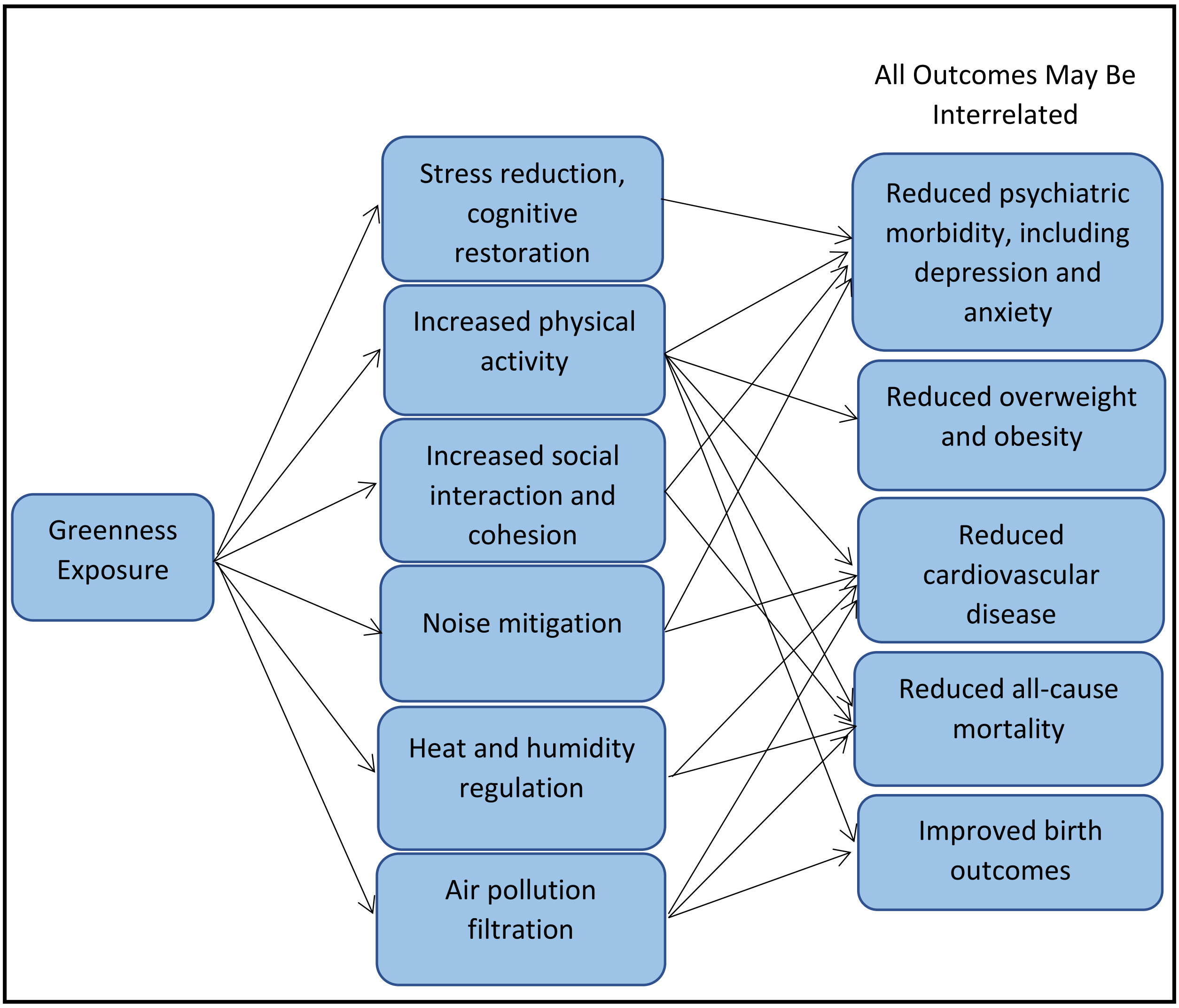



Figure 2.6: Pathways through which greenness may affect health (redrawn from James, Banay, Hart, & Laden, 2015, p. 132).

The World Health Organisation (2004) states depression is the largest contributor to the burden of fatal disease in middle- and high-income countries, by increasing the risk of several somatic illnesses such as stroke, diabetes, and obesity (Penninx, Milaneschi, Lamers, & Vogelzangs, 2013). There is evidence to suggest increasing urbanisation, in part, contributes to an increase in depression. For example, a population study of adult men and women in Sweden (n = 4.4million) found those living in areas with greater urbanisation reported an increased risk of developing depression and psychosis, a trend more profound in those living alone or with low educational attainment (Sandquist, Frank, & Sandquist, 2004). There is also a large, growing body of evidence to suggest the saturation of greenspace within the local area can reduce the effects of stressful life events (van de Berg, Maas, Verheij, & Groenewegen, 2010) and improve perceived mental health and life satisfaction (Chiesura, 2004; van de Berg, Maas, Verheij, & Groenewegen, 2010; Bertram & Rehdanz, 2015; Huang, Yang, Lu, Huang, & Yu, 2017; Soga, Gaston, & Yamaura, 2017; Houlden, Albuquerque, Weich, & Jarvis, 2019).

Ultimately, the mental health benefits to residents largely centre around two theories – Psychophysiological Stress Recovery Theory (SRT) (Ulrich et al., 1991) and the Attention Restoration Theory (ART) (Kaplan, 1995) – which explain how physiological measures of stress and fatigue are effectively reduced in a natural environment. SRT draws on the evolutionary psychology of humans assuming they are biologically prepared to react positively towards natural, vegetation-rich environments, assuming this reaction is still prevalent today (Ulrich et al., 1991; Joye & Dewitte, 2018). On the other hand ART views attention as a limited resource and claims natural environments can facilitate the recovery of an attention depleted state (Joye & Dewitte, 2018). Studies have shown visual contact with greenspaces can have these beneficial reactions. Having a window view populated by trees reduced blood pressure in 112 young adults after driving and performing demanding tasks (Hartig, Evans, Jamner, & Davis, 2003). Similarly, being presented with pictures while exercising of greenspaces categorised as 'pleasant', both urban and rural, had significantly positive effects on self-esteem and blood pressure (Pretty, Peacock, Sellens, & Griffin, 2005). Furthermore, this has been demonstrated by Ulrich (1984) in reducing postoperative hospital stays, Tennessen and Cimprich (1995) in the effects of direct attention on university students, Kaplan (2001) with increased residents' satisfaction in apartments with natural views, and Moran (2019) relieving mental fatigue in prisoners. It should be noted, however, in a recent review of both SRT and ART Joye and Dewitte (2018) question the methodology and thus validity of such experiments. They note most studies fail to include a control group when comparing participants reactions from a 'stressed' state to a calmed one. Also, a central principle in ART is that of fascination – being triggered by natural phenomena such as waterfalls, snow patterns, and sunsets – however, when subjecting participants to nature, researchers will often use simplistic greenspaces thus omitting the feeling of fascination. These criticisms do highlight the potential limitations of SRT and ART; other environmental psychology theories have also been presented such as topophilia – exploring the bond between people and place – (Staats, Gatersleben, & Hartig, 1997) and arousal theory – a heightening of brain activity caused by surroundings – (Oliveria, Roca, & Leitao, 2010). However, it is not within the scope of this thesis to explore all psychologies in detail. It is the focus of this research to explore the variation in household gardens, and specifically how a more varied landscape may provide greater stress-reducing benefits to residents (Bell et al., 2020; Brindley et al., 2018).

In their study assessing the happiness of residents of green urban areas, White, Alcock, Wheeler, and Depledge (2013) included both 'greenspace' and 'gardens' as each contributed to 33 per cent and 32 per cent landcover respectively. They drew on the British Household Panel Survey to obtain data on participants' self-reported psychological health using two metrics: the General Health Questionnaire (GHQ) ($n=12,818$) and a life satisfaction score ($n=10,168$). Psychological health scores were compared when participants moved to a new house within the 18-year study period to test areas with differing abundances of greenspace while using the same survey population. It was expected that those who moved to areas with more greenspace would have a lower GHQ score (measuring mental distress) and a greater life satisfaction score. Results supported this hypothesis when testing for a 1 per cent increase in greenspace: GHQ: $b = -0.0043$ $p = <0.001$ and life satisfaction: $b = 0.002$, $p = 0.003$. Although the coefficients are small, the researchers note this improvement in psychological health is based on a small increase in localised greenspace cover and a more substantial change would produce a greater increase in an individual's health. They state:

> Compared with living in a LSOA (Lower Super Output Area) [a geospatial statistical unit with a minimum population of 1000 with a mean size of 1,500 used in England and Wales to facilitate the reporting of small area statistics] with greenspace one standard deviation below the mean (48 per cent greenspace), living in an LSOA with greenspace one standard deviation above the mean (81 per cent greenspace) was associated with a 0.14 reduction in GHQ and a 0.07 increase in life satisfaction (White et al., 2013, p. 924).

White et al., (2013) re-ran the analysis excluding household gardens and found the relationship between psychological health and greenspace remained significant. Interestingly, the GHQ coefficient became less negative $b = -0.0023$ $p = 0.003$ and life satisfaction less positive $b = 0.0011$ $p = 0.007$ suggesting a weaker relationship between health and greenspaces when excluding household gardens. It was not within the remit of this study to test the effect of just household gardens.

Brindley et al. (2018) investigated the effect of household garden size on self-reported health using secondary data resources in England. Data used were collected by the UK Office for National Statistics 2011, 25,766 urban LSOAs (2001 and 2011) were cross-referenced with self-reported health scores. Domestic garden size was taken from the Generalised Land Use Database from the Office for National Statistics, data were only available from 2005. A higher prevalence ratio was recorded for the LSOAs with the smallest average garden sizes when compared to those with the largest average garden size; 1.13 in 2001 (95 per cent confidence interval (CI) between 1.12-1.14) and 1.12 in 2011 (95 per cent CI between 1.11-1.13). Within the same garden size grouping, LSOAs with the highest income deprivation had the highest prevalence ratio for self-reported health in 2001, 1.72 (95 per cent CI 1.64-1.79) in the smallest gardens and 1.31 (95 per cent CI 1.21-1.42) in the largest gardens, data for 2011 not shown. Employment and education deprivation showed similar patterns but with less significant results. This study illustrated the potential for household gardens to influence a person's health on a national scale. However, Brindley et al. (2018) highlight several drawbacks and limitations of the research. The land-use data used was dated either four years after or six years before the health data; further it gave a mean garden size for each LSOA which contain on average 650 houses meaning no account was taken for size variation within the LSOA. There was also no distinction between the quality of the gardens, a factor known to greatly affect their potential benefits (Hartig et al., 2003; Pretty et al., 2005). The role of ecological fallacy is also highlighted (i.e., a mistaken conclusion drawn about individuals based on findings from groups to which they belong), LSOAs contain on average 1500 people and categorising each using arbitrary units for deprivation is problematic as the results may not hold at the individual level. With these limitations in mind, this study was a trailblazer for research into household gardens and health, illustrating the strong benefits they may have on residents. Brindley et al. (2018) call for further research in this area, stating the causality needs confirmation and the evidence base growing.

Similarly, Bell et al. (2020) reported the effect of having access to a garden on resident's health by using secondary data collected 2009-2016 from the English Monitor Engagement with the Natural Environment survey by Natural England. The assessment of wellbeing was based on two questions, for evaluative wellbeing participants were asked ‘Overall, how satisfied are you with life nowadays?’ and for eudaimonic ‘Overall, to what extent do you feel that the things you do in your life are worthwhile?’ ($n = 5173$. Respondents with access to a garden had greater odds ratios of self-reported; evaluative wellbeing, 1.43 ($p = < 0.001$), eudaimonic wellbeing, 1.15 ($p = < 0.05$), and meeting physical activity guidelines, 1.24 ($p = < 0.05$) compared with those with no garden space. Yet, when including whether residents participated in gardening activities into the analysis eudaimonic wellbeing was no longer significant.

The physical act of gardening, in an allotment, community or household garden, can have both mental and physical benefits to residents.  Wood, Pretty, and Griffin (2016) measured self-esteem, mood, and general health in 136 allotment gardeners and in a control population of 133 non-gardeners. Significant improvements in self-esteem and mood were recorded after one session of allotment gardening. The research suggests a <30-minute session of gardening per week produced positive affects in mental wellbeing, multiple uses in a seven-day period did not increase the effects. The researchers also note the social benefits of community gardening, social interaction and community cohesion, though these would not necessarily be as prevalent in household gardening unless done as neighbourly activity. Although, Kingsley et al. (2022) found in a global survey of 3743 gardeners, that gardening during the Covid-19 pandemic was a vital source of social interaction especially between family members. Also, Chalmin-Pui et al. (2021) found respondents who gardened daily in the UK had greater self-reported wellbeing than those who partook less regularly ($n = 6015$).  In Australia, Fjaestad et al. (2023) conducted a survey of 4919 46 – 80-year-olds and found those who gardened for ≥150minutes per week were more likely to report better mental wellbeing and life satisfaction.

Therapeutic horticulture has also been shown as an effective treatment for individuals with clinical depression (Gonzalez, Hartig, Patil, Martinsen, & Kirkevold, 2010) and an effective way of decreasing depression and anxiety and increasing self-identity in middle-aged women (Kim & Park, 2018). Dayson, Bashir, Bennett, and Sanderson (2016) report a potential estimated decrease in A&E attendees, outpatient appointments, and impatient admissions of up to 20-21 per cent in the UK as a result of social prescribing initiatives like therapeutic horticulture. A reduction of this size is a considerable benefit to a strained NHS and results in a potential saving of £1.98 for every £1 invested (Dayson et al., 2016). See Sempik, Aldridge, and Becker, (2003) and Nicholas, Giang, and Yap (2019) for an extensive review. As a result of the review conducted, Sempik et al. (2003) highlighted the need for more hard evidence of therapeutic horticulture. In response to this, Hinds and Camic (2013) published a review of the papers published since Sempik et al. (2003). It was found most studies since 2003 had used questionnaires to obtain quantitative 'hard data' results. There is now a large body of evidence in support of therapeutic horticulture world-wide (Hinds & Camic, 2013; Maxwell & Lovell, 2017; Soga, Gaston, & Yamaura, 2018; Howarth, Lawler, & de Silva, 2021), and as discussed in Section 2.2 it is being used increasingly as a national healthcare tool. However, while the body of evidence is improving, Hinds and Camic (2013) still argue there is a need for well-designed controlled trials which explore specific active components of this nature-based therapy.

Whilst research within household gardens remains scarce, more recently front gardens have gained attention in their potential for increasing resident's wellbeing. Decreased perceived stress and a reduction in chronic stress indicated by diurnal cortisol profiles were reported by Chalmin-Pui et al. (2020) in a study based on 38 front gardens in Salford. Previously bare front gardens were landscaped with ornamental plants, after a 3-month period a higher proportion of residents had healthy diurnal cortisol patterns as well as claiming several socio-cultural benefits. Further, Chalmin-Pui et al. (2021) found participants of an online gardening motivation survey (n = 6015), in a separate, national study, felt their wellbeing was positively affected by the amount of vegetation cover within their front gardens. Increasing vegetation cover by 10 per cent increased a residents wellbeing score by 26 per cent (95 per cent CI between 22-30 per cent). Despite their proven health and wellbeing benefits, front garden vegetation has been rapidly declining in the UK (Chalmin-Pui, Griffiths, Roe, & Cameron, 2019). One in four front gardens are fully paved, equating to three times less plant cover than ten years ago (RHS, 2015).

In summary, both the presence of greenspace (Bertram & Rehdanz, 2015; Chiesura, 2004; Houlden et al., 2019; Huang et al., 2017; Soga et al., 2017; van de Berg et al., 2010) and the use of local greenspace (Gonzalez et al., 2010; Wood et al., 2016) have been shown to benefit an individual's mental and physical wellbeing. On the other hand, it should noted that contact with nature has been shown to have negative impacts such as environmental degradation and the displacement of people which negatively affects feelings of identity and autonomy (Russell et al., 2013), though these issues are more pertinent to public GS. Still, utilising the natural environment as a population health intervention is a key strategy for promoting wellbeing, thus it is essential the UK government have a collaborative and interdisciplinary approach to policy (Cook, Howarth, & Wheater, 2019).

The fundamental concepts around SRT and ART can be attributed to household gardens, quality and thus diversity within these land types can have the same effects on residents looking out onto gardens as seen in previous studies on views onto public land (Hartig et al., 2003; Pretty et al., 2005). Twedt et al. (2016) found more informal, naturalistic gardens had stronger restorative potential and visual appeal for residents and Ambrose et al. (2020) reported higher well-being scores associated with vegetable cultivation compared to ornamental gardens. During the unprecedented COVID-19 global pandemic, the importance of these concepts gained great attention, resulting in a resurgence of research within the field. In Bulgaria, an online survey of 323 university students found those undertaking lockdown ($17^{th}$ May – $10^{th}$ June 2020) self-reported less depression and anxiety symptoms when greenery was present either as house plants or from green views outdoors (Dzhambov et al., 2021). A study conducted across six European countries found that individuals expressed a great need to spend time in an urban green space during the pandemic, and that they were seen as essential for respite and relaxation as well as for exercise (Ugolini et al., 2020). In Scotland, higher frequency of garden usage during the lockdown was associated with improved self-reported physical, emotional, and mental health in a study of 171 retirees (Corley et al., 2021). Household gardens are primarily managed by the residents who have autonomy over decision making regarding the contents and, hence, the potential stress reducing benefits they could derive from their garden.

### 2.7.2 Environmental

UGI, such as household gardens, provide essential ecosystem services beyond those effecting people as discussed above, ecosystem services were defined and discussed in Section 1.1. As well as mitigating the effects of climate change, UGI provides other regulating and supporting ecosystem services. An extensive review of these is given by Cameron et al. (2012), Kwiatkowski et al. (2014) for the UK National Ecosystem Assessment

(UK NEA) technical report, and Camps-Calvet, Langemeyer, Calvet-Mir, & Gomez-Baggethun, (2016). It is not a focus of this thesis to extensively discuss and review environmental ecosystem services, but a brief overview is presented below.

'The ecosystem goods and services that could potentially be derived from Urban greenspace are substantial' (Kwiatkowski et al., 2014, p. 362): as stated in the UK NEA the urban landscape is vital for the services it provides. The UK NEA technical report highlights a decline in the condition and accessibility of UGI between 1970s and 2000 (Kwiatkowski et al., 2014), a lack of funding and the sale of playing fields are given as key causes. The UK NEA arose in 2007 from findings of the Millennium Ecosystem Assessment 2005, which showed the importance and degradation of ecosystem services worldwide. The UK NEA and NEA Follow On 2014 ignited ecosystem service research within UGI in the UK.

UGI provide supporting services such as habitat provision (Gomez-Baggethun & Barton, 2013) and regulating services such as air purification (Baro et al., 2014) making them an important factor in climate regulation (Camps-Calvet et al, 2016). Further, Gill et al. (2007) suggest a 10 per cent increase in UGI would negate the 4°C predicted increase in air temperature over the next 80 years in Manchester, UK. Garden vegetation, particularly trees, can provide flood mitigation services by both absorbing surface run-off and intercepting intense precipitation; thus, reducing peak flow and demand on urban drainage systems (Xioa & McPherson, 2002; Armson, Stringer, & Ennos, 2013). Conversely, impermeable surfaces such as patios or paths increase rainwater run-off and the likelihood of flooding (Ellis, 2002; Perry & Nawaz, 2008). Because of this, since October 2008 the UK government has made it law to obtain planning permission for the use of impermeable materials on over 5 $m^2$ of the front garden (Environment Agency, 2008). Urban GI, household gardens in particular, also act as a hub for species diversity within an urban environment; providing important supporting services resulting in a high level of biodiversity (Loram, Thompson, Warren, & Gaston, 2008; Young, Frey, Moretti, & Bauer, 2019).

## 2.8 Personal investment into household gardens

Personal investment into household gardens in the UK has scarcely been reported within scientific literature and most studies which have produced estimates are now out-dated and use secondary data from large-scale, government-led surveys where gardens are not the focus. Yet, such studies are needed to be able to quantify the amount of time and money spent within gardens, both for evidence of their importance in policy but also to understand the level of benefits derived from the amount of time spent using the resource. As mentioned above, the lack of research into environmental injustice within private gardens fuels the need for a better understanding of the differences between socio-economic groups and their levels of investment within their gardens. Bhatti and Church (2004) used The General Household Survey for England to provide an estimate of 48 per cent adult participation in gardening throughout the decade of the 1990s. Since then, more detailed research into the frequency (Gaston et al., 2005) and duration (Dunnett & Oasim, 2000) of gardening have been conducted. Both studies use data which are now out-dated and sample one city, Sheffield, meaning their results only represent a small sub-section of the UK. More recently, Bell et al. (2020) have published a national-scale review, the first of its kind for in the last twenty years.

Gaston et al. (2005) recorded varying levels of garden management in participants when surveying 250 dwellings across Sheffield, UK: 40.5 per cent gardened more than once a week, 34.2 per cent once a week, and 25.2 per cent less than once a week. The duration of the gardening was not recorded as part of this study. To calculate a mean across the 223 dwellings that had gardens, Gaston et al. (2005) assumed one hour of gardening on each occasion, giving a mean of 1.28 hours per week per garden. Assuming gardening occurs only in months of mild weather (45 weeks of the year), it is estimated 10 million hours of work is invested into gardens across Sheffield each year. Gaston et al. (2005) reported national spend in the UK on garden products between 1999–2000 to be valued at £2.62 billion, with an average household spend of £183 per annum. An estimate for the total spend on gardens across Sheffield in 1999-2000 was given as £19.2 million per year.

Dunnett and Qasim (2000) used 376 completed questionnaires, sent to residents from urban houses with gardens across 48 streets in Sheffield, to obtain information on the roles and values people ascribe to gardens.  Of the 25 questions, three directly related to personal investment which broadly covered: duration of time spent gardening, reasons for gardening, and the activities carried out by residents within their gardens. On average, adults under 35 years of age spent up to 1 hour per week gardening, while 35–45-year-olds spent 2–4 hours, and those over retirement age spent 5 hours or more per week gardening.

Though data were not reported, Dunnett and Qasim (2000) state fewer residents of semi-detached houses than expected valued the opportunity to be creative and express their personalities, while in terraced houses more than expected did so. Garden size was found to affect the activities residents enjoyed within their garden: in larger gardens (100 $m^2$ – 800 $m^2$) growing fruit and vegetables, exercise, and relaxation scored highly whereas in terraced housing significantly more residents reported liking nothing about gardening. The results of this study are limiting in their explanation of the value of gardens to people. Firstly, the small sample size and geographic scope make upscaling of the results difficult and unrepresentative. Second, very few numeric results are reported in the paper, much of the results section is descriptive and, therefore, limiting in providing evidence for causes of variation in peoples engagement within their garden.

Private gardens are a relatively neglected resource within environmental-health research (Dennis & James, 2017; Bell et al., 2020), yet their use has been shown to be increasingly popular with some residents in the UK. In a more contemporary study to Bhatti and Church (2004), Bell et al. (2020) use data from the English Monitor of Engagement with the Natural Environment survey, a national survey (n = 7814) conducted via face-to-face interviews by Natural England. They used annual data from 2009/10 – 2015/16 from questions regarding the presence of a garden, the participation of residents gardening/relaxing in a garden, and four self-reported measures of general health and wellbeing. Analyses were conducted at the LSOA level, and to compare socio-economic status between respondents the Ipsos MORI social grade classification tool was used, categorising the highest occupational level respondent within a household from AB (higher managerial roles) to DE (unskilled manual roles). It was found 69.7 per cent of 16–34-year-olds reported having access to a garden, compared with 79.9 per cent 35–64-year-olds and 85.6 per cent of those aged 65 years and over. Most respondents in all socio-economic groups had access to a garden, but only 6.3 per cent of group AB had no access compared to 17.1 per cent of group DE. Regarding garden engagement, 48.5 per cent of those of 65 years gardened or relaxed within their garden, compared to 23.8 per cent of 16-34-year-olds using their garden for both gardening and relaxing. Yet, just relaxing within their garden (not partaking in manual gardening) was more popular with the younger age category than any other. Within the AB group, 52.1 per cent of people used their garden for both activities (gardening and relaxing) compared to 30.1 per cent of the DE group. Overall, respondents with access to a garden reported significantly greater self-assessed health and wellbeing than those without, regardless of socio-economic characteristics.

The most up-to-date data on gardening engagement is from two key government-led national surveys: Monitor of Engagement of the Natural Environment (MENE) (also used in Bell et al., 2020) and the Taking Part survey. The Taking Part survey is one national database available for assessing the amount of time people invest in their gardens. The face-to-face survey had between 7000 – 10,000 respondents each year across England, conducted by the Department of Culture, Media & Sport. Participants were asked to highlight activities completed during their free time, within which gardening was an option, data are reported as proportion of respondents within each demographic category. Data are freely available online, though data on gardening were not provided in 2016/17 due to a substantially smaller sample size (H. Smart, personal communication, 30th September, 2020). Presented below is the proportion of people who garden by age group and deprivation category, followed by a breakdown of each region in England, and finally split by urban and rural residents, all for the years 2015-2020. Due to the Covid-19 pandemic the data were not collected for 2020/21 as in person field work was prohibited (DCMS, 2021). The survey was reinstated for 2021/22 with its closing date for participants in November 2022, but data are not due to be published until July 2023 making the data included below the most up to date.

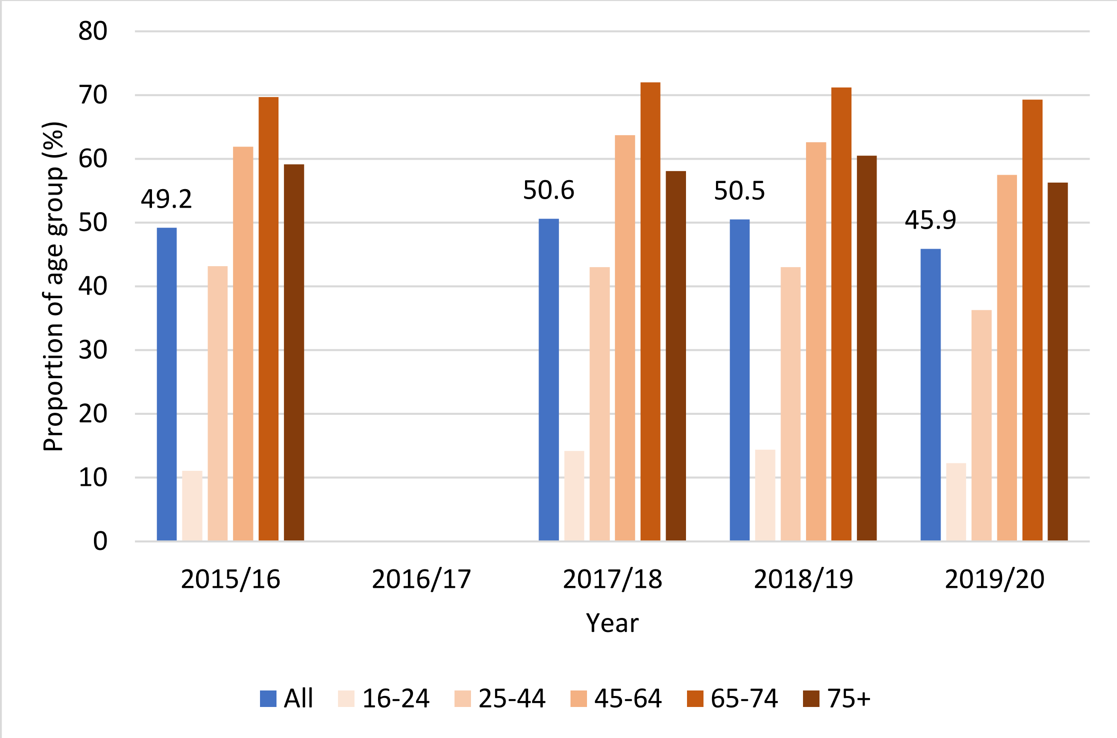



Figure 2.7: Proportion of respondents who garden as a free time activity in England. Blue = all age groups ≥ 16-year-olds. Orange = age categories (Taking Part Survey England, various years – no data are available for 2016/17).

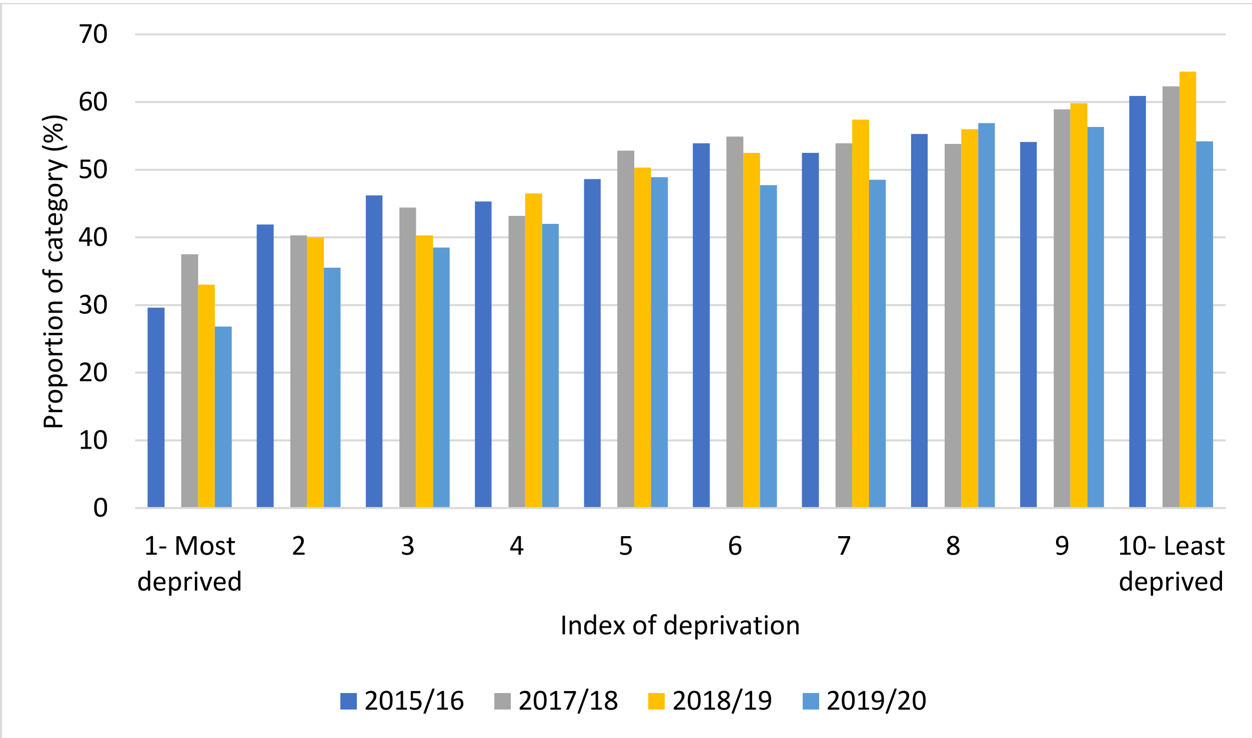



Figure 2.8: The proportion of each deprivation category who garden over four years (Taking Part Survey, various years).

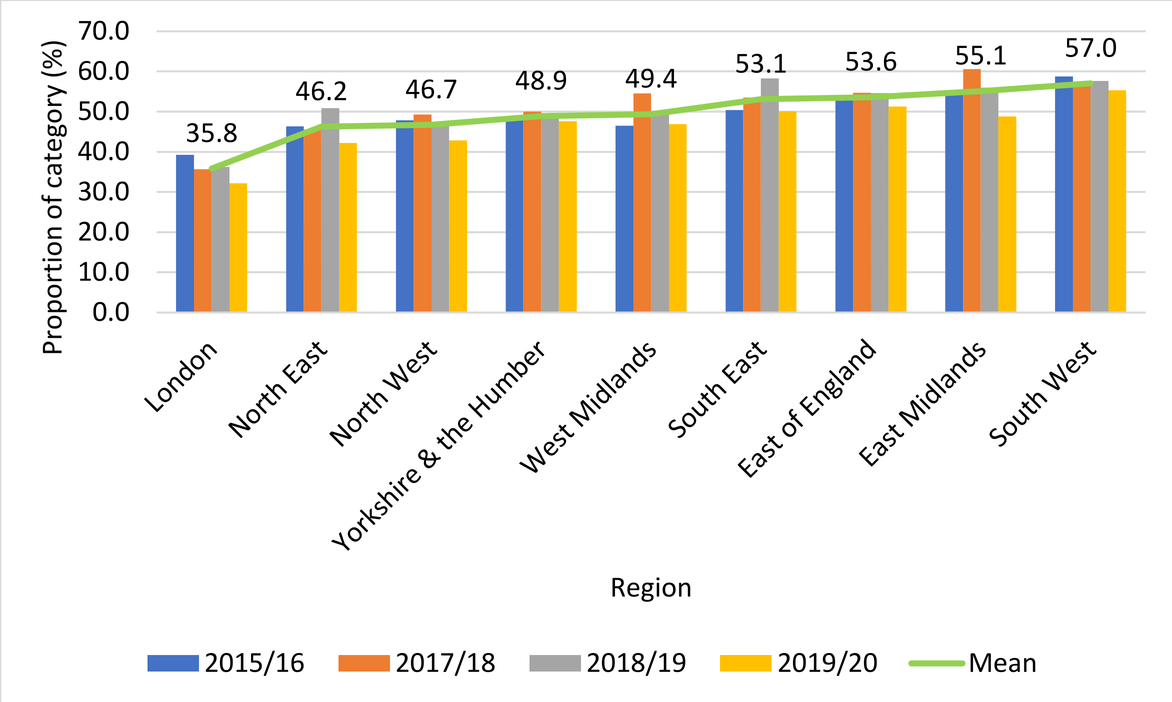



Figure 2.9: The proportion of each region who garden across four years and the mean over the four years for each category (in green). Data labels depict the mean for each region across the four years (Taking Part Survey, various years).

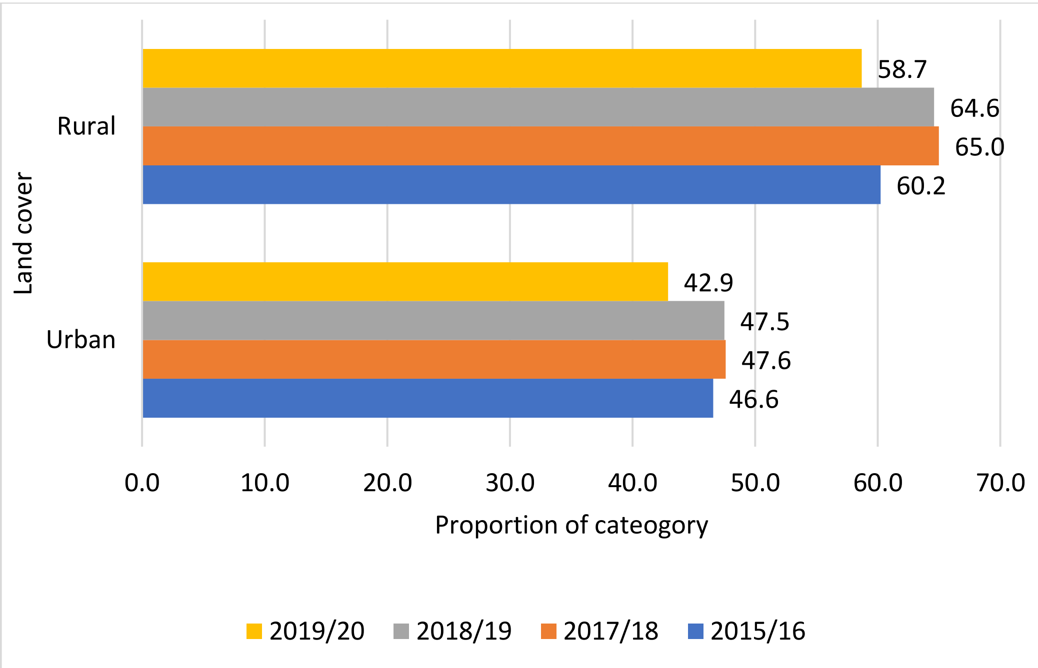



Figure 2.10: The proportion of the urban and rural categories nationally who garden as a free-time activity across four years (Taking Part Survey, various years).

Within the literature, gardening participation split by age group in the 1990s showed 61 per cent of 60–69-year-olds identified gardening as a leisure activity compared to 21 per cent of 20–24-year-olds (Bhatti & Church, 2004). Figure 2.7 shows 65-74-year-olds had the highest level of participation and 16-24-year-olds had the lowest, like that shown by Bhatti & Church (2004). This trend was consistent across all four years recorded in Figure 2.7. Residents over retirement age have been shown to garden for longer durations, five hours or more weekly, than younger generations, adults under 35 years spending up to one hour on average (Dunnett & Oasim, 2000). Socio-economic status was also a variant in a resident's likelihood to garden. Figure 2.8 uses the Index of Multiple Deprivation, a widely used national system ranking households according to numerous demographic and socio-economic factors (Ministry for Housing, Communities & Local Government, 2019), to illustrate gardening participation by socio-economic status. All four years show a similar trend of increasing gardening participation with decreasing deprivation status.

Gardening participation fluctuated between 45 – 50 per cent across the four years (Figure 2.7), overall, the mean national gardening participation is 49 per cent combining all years (Figure 2.7). Four regions shown in Figure 2.9 are below this national average. Participation varies widely between regions but remains reasonably consistent within each region over the four years. The North West, Yorkshire and the Humber, and the West Midlands are the three regions closest to the mean gardening participation. Gardening participation rate differs between areas which are urban and rural, as shown in Figure 2.10. Rural households have a consistently higher rate with mean participation being 62 per cent whereas in urban areas 46 per cent respectively. When deciding on a research location for this thesis, these regional data were considered. The North West is considered one of the UK's most varied landscapes with both rural and highly urban areas (Gill et al., 2008; Radford & James, 2013; Rothwell et al., 2010). The near average participation rate and the varied landscape make the North West a representative study location for a more detailed investigation on household garden content within this thesis.

The MENE survey also had a broad definition of gardening but included further questions on engagement. The survey captured a total of 15,390 respondents over a four-year period. Below these data are presented and discussed. Firstly, is the data for Q7 of the survey on garden access, split by age, socio-economic group, and ethnicity. Then, the data on residents' attitudes to their gardens, Q8 of the survey, as well as a socio-economic breakdown of key statements.

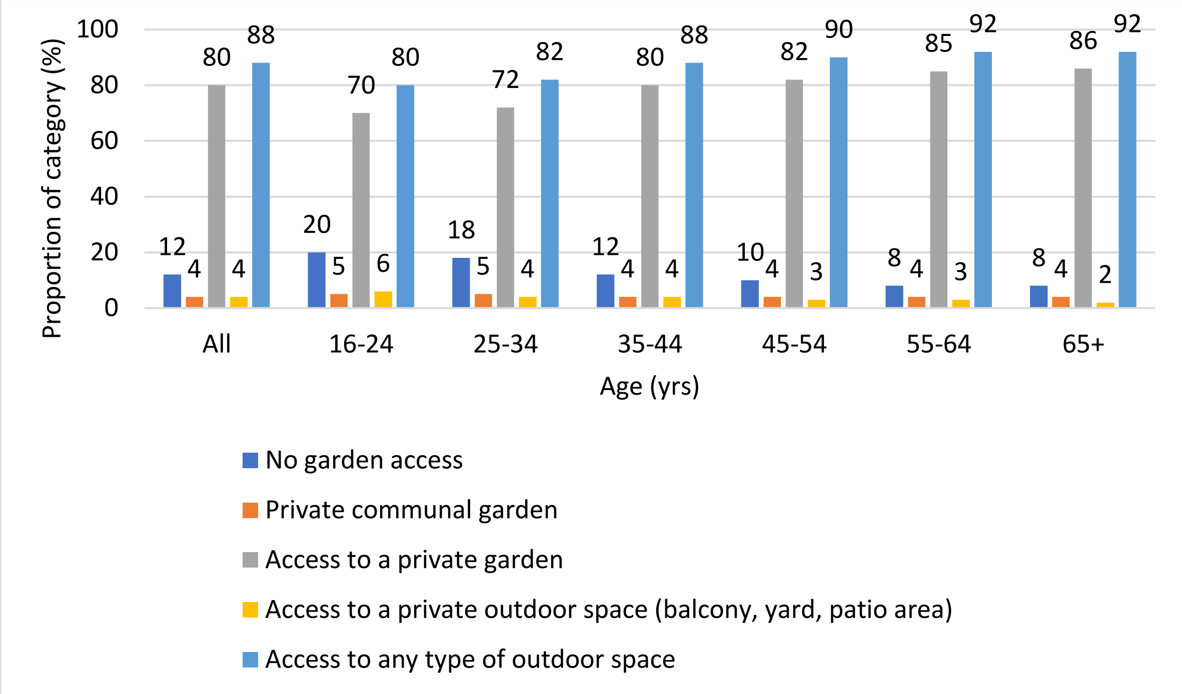



Figure 2.11: The proportion of each age category (≥16yrs) and their access to a private greenspace (MENE, 2014 - 2019).

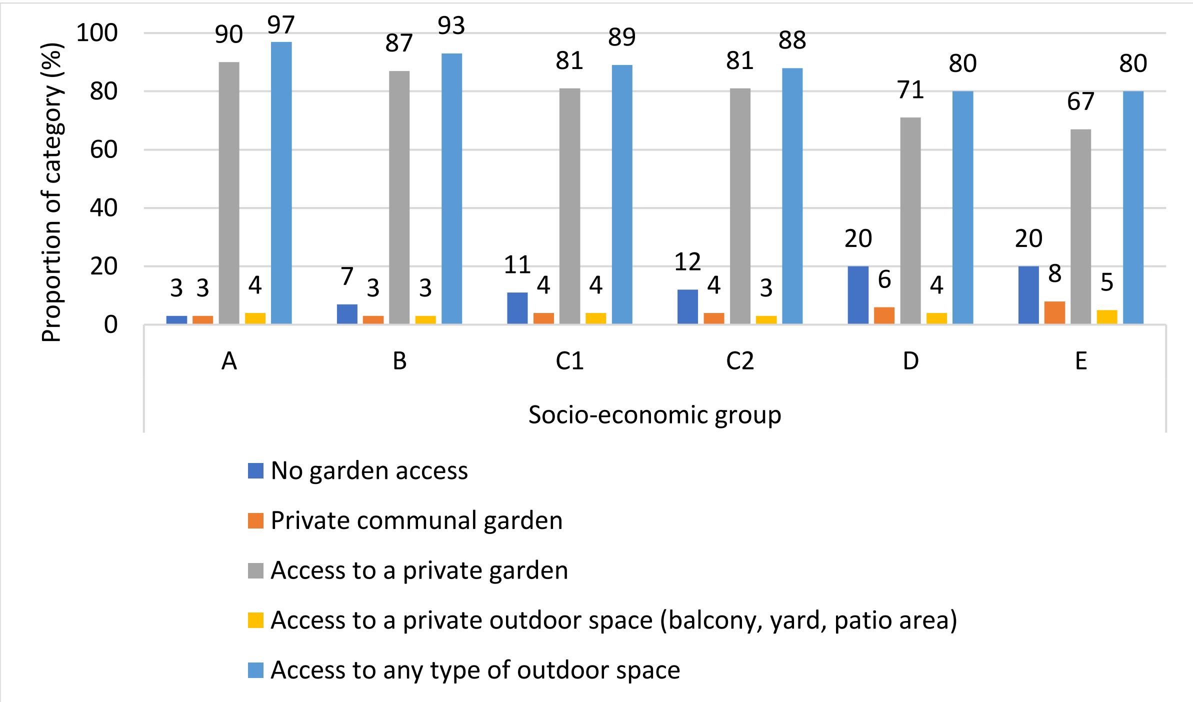



Figure 2.12: The proportion of each socioeconomic group and their access to a private greenspace. Socioeconomic categories based off National Readership Survey (NRS) social grade system (MENE, 2014 – 2019; Wilmshurst & Mackay, 1999).

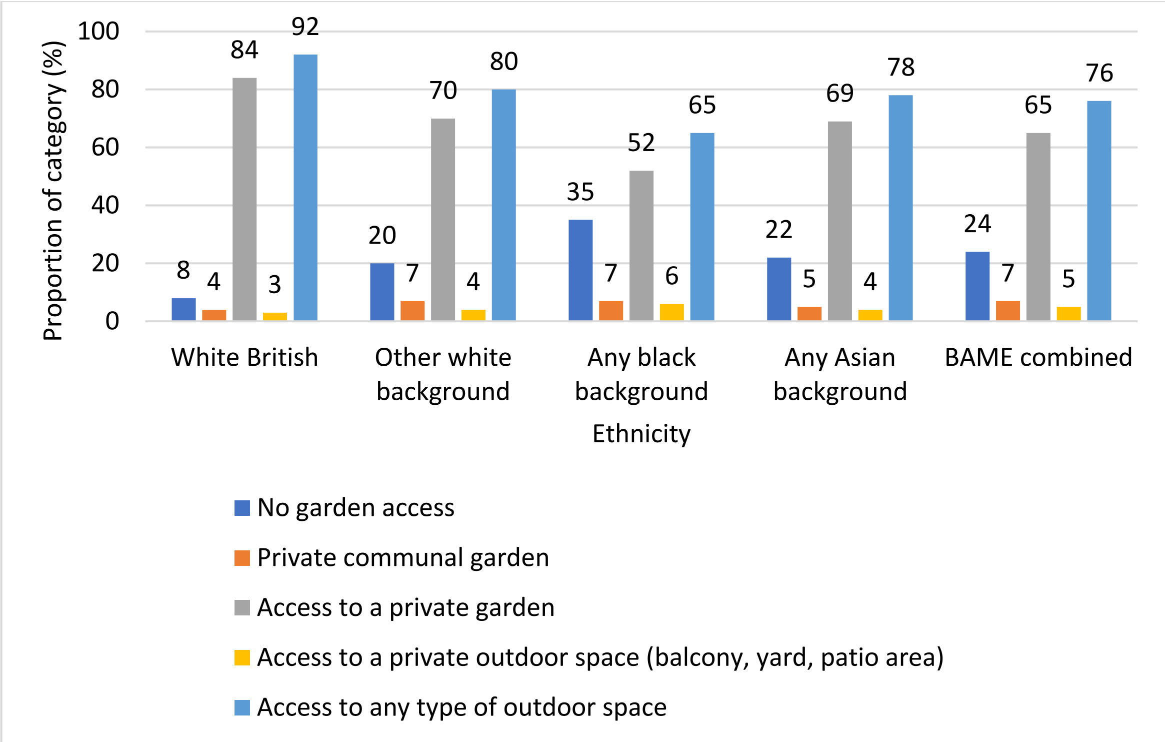



Figure 2.13: The proportion of each ethnicity category and their access to a private greenspace (MENE, 2014-2019).

According to data in Figure 2.11, 88 per cent of adults have access to an outdoor space. The age range with the smallest proportion of access is 16–24 yrs, with 20 per cent of people not having any access to outdoor space. Similarly, 16–24 yrs had the smallest proportion of adults with access to a private garden at 70 per cent, the average for all age groups being 80 per cent. Data in Figure 2.7 showed 16–24 yrs was the smallest proportion of self-reported gardeners too. Figure 2.12 splits the garden access data by socio-economic group using the National Readership Survey (NRS) social grade system. In short, categories are defined as: A – upper middle class with high managerial roles; B – middle middle class with intermediate managerial roles; C1 – lower middle class with supervisory roles; C2 – skilled working class with skilled manual roles; D – working class with semi-skilled or unskilled manual roles; E – non-working such as pensioners and unemployed. Access to any type of outdoor space (Figure 6.6) decreases from group A at 97 per cent to group E at 80 per cent. Outdoor access steadily decreases from A–C2 but falls dramatically from C2 to D and E which are both at 80 per cent. A similar trend is seen with access to a private garden, with groups D and E having 71 and 67 per cent respectively compared with 90-81 per cent in groups A–C2. This highlights an injustice in greenspace access for these two groups of society, working class and non-working. Finally, Figure 2.13 splits access by ethnicity illustrating White British have the highest proportion of access to any outdoor space 92 per cent and to a private garden 84 per cent.

Table 2.3: The proportion of respondents who answered Yes/No to statements regarding their attitude towards their gardens (MENE, 2014-2019).

| **Statement** | **% of respondents** | |
|---|---|---|
| | **Yes** | **No** |
| My garden is an important place to me | 47 | 53 |
| I enjoy gardening | 41 | 59 |
| My garden is too small | 14 | 86 |
| My garden is a place where children can play | 31 | 69 |
| I enjoy the trees in my garden | 30 | 70 |
| I enjoy the grass\plants in my garden | 41 | 59 |
| I encourage wildlife in my garden | 36 | 64 |

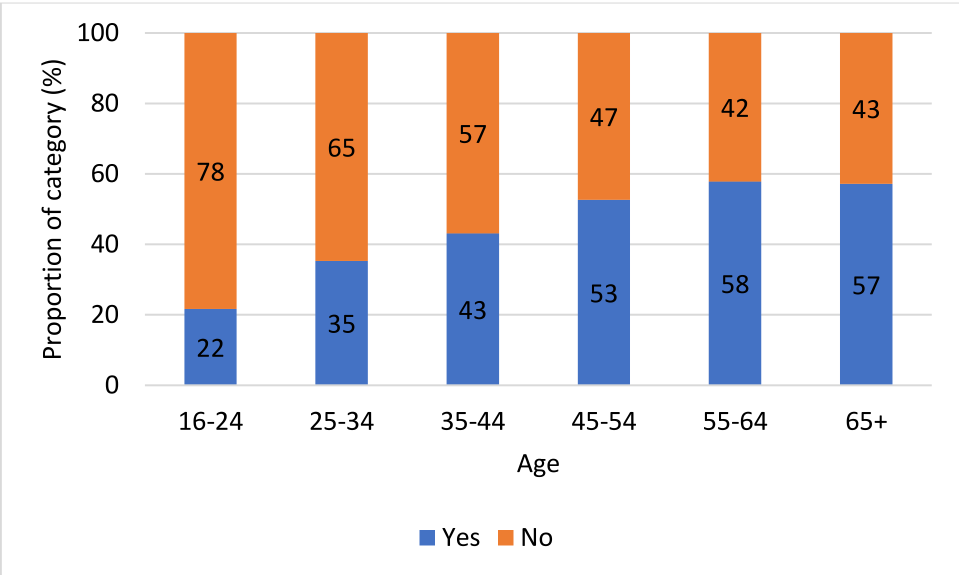



Figure 2.14: The proportion of each age category and their answer to the statement ‘My garden is an important place to me’ (MENE, 2014–2019).

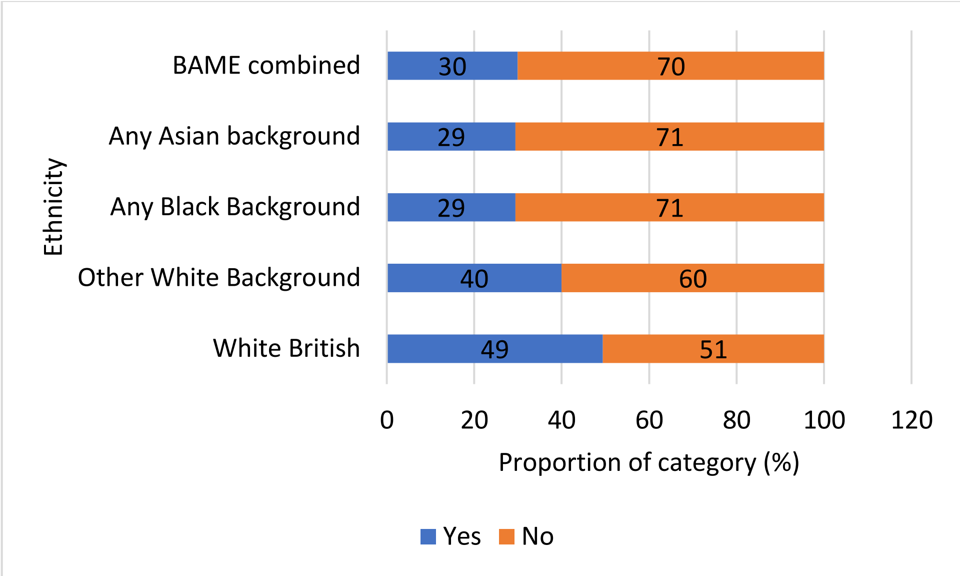



Figure 2.15: The proportion of each ethnicity category and their answer to the statement 'My garden is an important place to me' (MENE, 2014–2019).

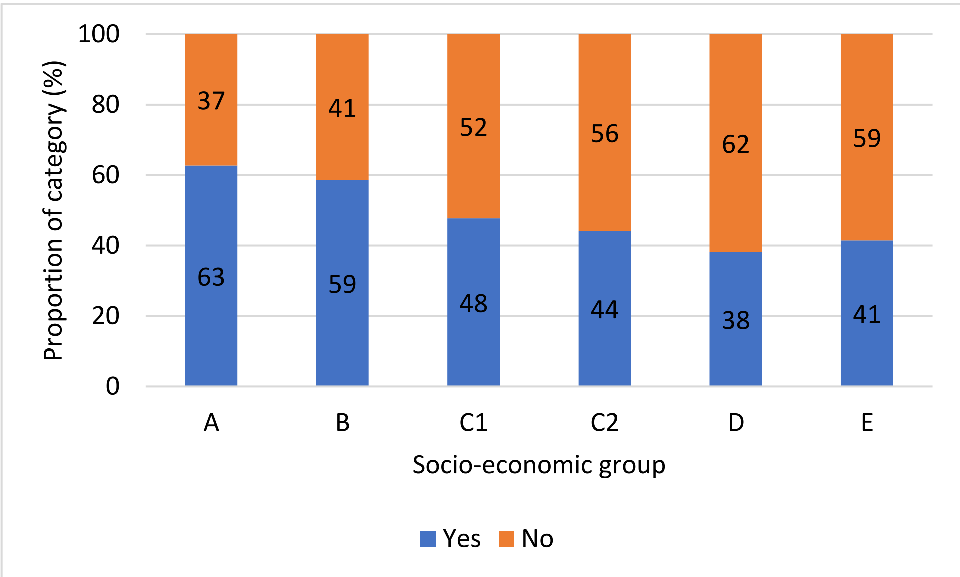



Figure 2.16: The proportion of each socio-economic group and their answer to the statement 'My garden is an important place to me' (MENE, 2014–2019).

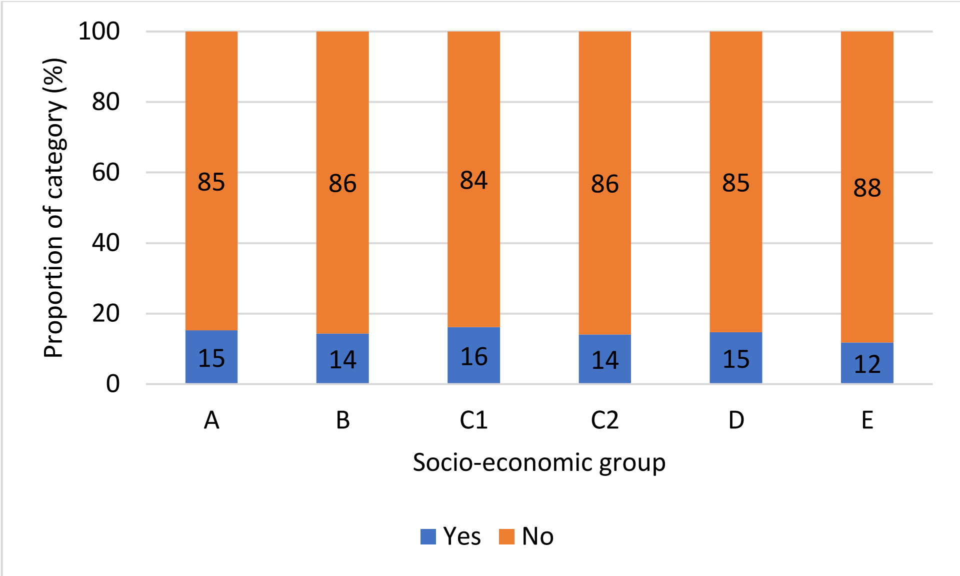



Figure 2.17: The proportion of each socio-economic group and their answer to the statement ‘My garden is too small’ (MENE, 2014–2019).

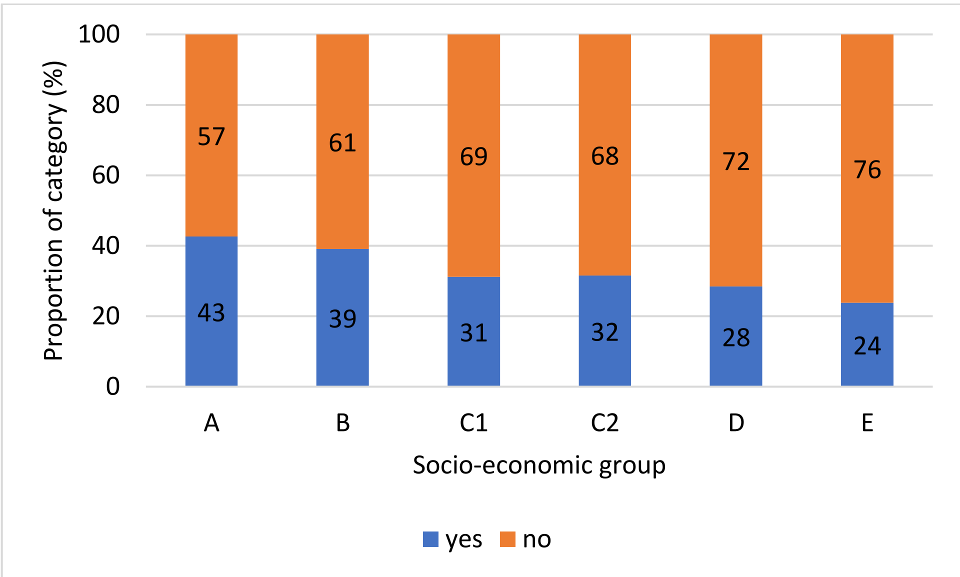



Figure 2.18: The proportion of each socio-economic group and their answer to the statement ‘My garden is a place where children can play’ (MENE, 2014–2019).

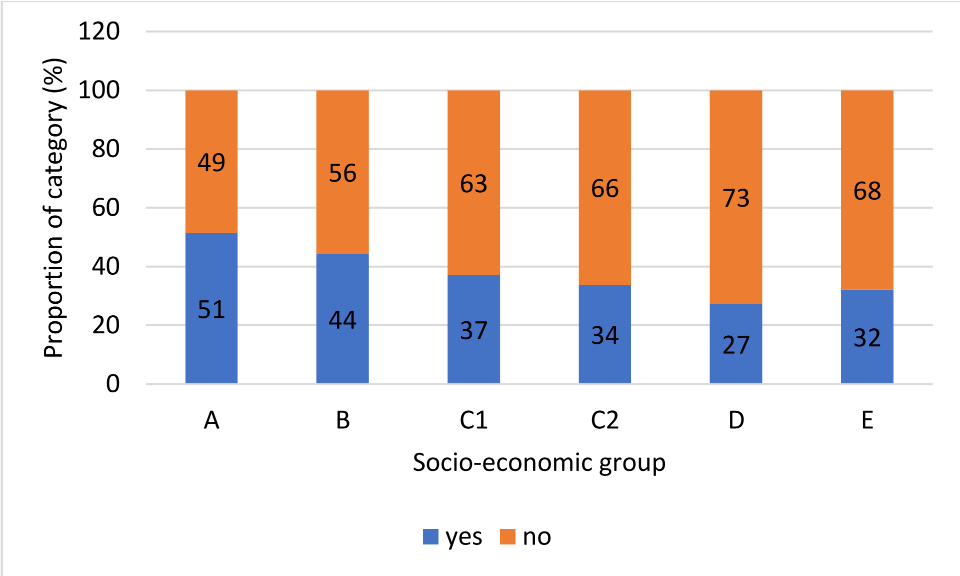



Figure 2.19: The proportion of each socio-economic group and their answer to the statement 'I encourage wildlife in my garden' (MENE, 2014–2019).

The data presented in Table 2.3 shows the proportion of respondents who answered Yes/No to several statements about their attitudes towards their gardens. It shows nearly half of the respondents view their garden as an important place, 47 per cent. Yet, the proportion of respondents who enjoy gardening is lower at 41 per cent. Only a small number of respondents view their garden as being too small, 14 per cent. Although 69 per cent said their garden was not a place for children to play; yet private gardens were found to be a key location for outdoor play for children in Scotland (Olsen, Caryl, McCrorie, & Mitchell, 2022). Whilst it is unclear why the number of children utilising their gardens is low in Table 2.3, the data suggest the lack of suitability for children is not down to size. Most respondents stated they do not enjoy the trees in their garden, 70 per cent, though less do not enjoy their grass/plants, 59 per cent. This suggests residents do appreciate vegetation in their gardens, but smaller and easier to maintain flora. Finally, only 36 per cent of respondents actively encourage wildlife into their gardens. This statistic is lower than previously reported, with Davies et al., (2009) stating 51 per cent of participants provided supplementary food for birds.

As the nature of this thesis is to expose environmental injustice, a number of these statements were then broken down further using the socio-economic parameters available. The first statement was 'My garden is an important place to me' (Figures 2.13 – 2.19). Figure

2.13 gives a break down by age category to this statement. It shows the likelihood of garden importance increases with age, only 22 per cent of 16–24-year-olds answers yes compared to 57 per cent of 65 years and over. An ethnicity breakdown for the same statement shows 49 per cent of those with a White British background view their garden as important (Figure 2.15). Yet, the proportion of those with non-White ethnicities who do so is lower, 30 per cent for ethnic minorities combined. The MENE survey uses the term BAME to describe an ethnic group, therefore it is used in this thesis when quoting data, however, the according to Gov.uk (2021) the correct term is ethnic minority. Finally for this statement, Figure 2.16 gives a breakdown of responses using the NRS social grading for socio-economic status. The proportion of respondents who view their garden as important was highest with groups A and B, 63 and 59 per cent respectively. Less than half of the respondents in each group C1, C2, D, and E view their garden as important. The lowest proportion being in group D, 38 per cent.

The data in Figure 2.16 give a breakdown of the respondents NRS social grade to the statement 'My garden is too small'. As stated above and in Table 2.3, overall, most respondents did not think their garden was too small, 86 per cent. This trend was similar across all social groups, with Yes responses only varying between 12 and 16 per cent. There was, however, more significant variation in the responses to the statement 'My garden is a place where children can play'. The Yes responses reduced steadily from 43 per cent for group A to 24 per cent in group E. As garden size is not a common issue with respondents, it is likely garden content is the restricting factor. Groups D and E, with 28 and 24 per cent Yes responses respectively, are working class and non-working communities, thus will have less disposable income. These data are highlighting an injustice whereby children of these families are not able to play in their gardens due to a lack of resources fuelled by monetary limitations. Finally, Figure 2.18 illustrates the NRS social grade breakdown for the statement 'I encourage wildlife in my garden'. Again, the data show a steady decrease in the number of Yes responses from 51 per cent in group A to 27 per cent in group D. Interestingly, group E has more Yes responses than group D at 32 per cent. Although this increase is marginal it could indicate an increase in wildlife engagement due to this social group being non-working residents who may have more spare time, though this is conjecture.

The MENE survey was stopped in 2019 and has been replaced by the People and Nature Survey for England. It began in April 2020 and is an online survey for adults (16yrs and over) with approximately 25,000 respondents each year (ONS, 2022). For year one (2020–2021) it reported 79 per cent of those who had access to a garden spent time in it at least once a week; for year two this decreased slightly to 76 per cent. Over both years, one in three people with garden access said they used it consciously for their mental health and well-being. Again, over both years, 8 per cent of respondents to the survey said they did not have access to a garden. This proportion without garden access in the People and Nature survey was smaller than the data for 2014 – 2019 in the MENE survey, which reported 12 per cent of respondents did not have garden access (Figure 2.11).

Overall, combining the two datasets from the Taking Part and MENE surveys provided an extensive overview of garden use, accessibility, and attitudes. Across England, an average of 49 per cent of all resident's garden as a free time activity (Figure 2.7). Both garden access (Figure 2.11) and gardening (Figure 2.7) were more likely in the older age categories, 65 years and over. Access to a private garden was found to decrease with NRS social group (Figure 2.12), highlighting an injustice in urban greenspace access for working class and non-working residents. Similarly, gardening as a free-time activity was lowest in the most deprived IMD categories (Figure 2.8). Further, children of working class and non-working families were also found to not have access to the same quality garden space as those with middle class parents, with approximately 26 per cent not having suitable gardens for play (Figure 2.17). These data show there is unequal opportunities for residents to gain the health and wellbeing benefits outlined in Section 2.6 of this thesis dependant on their socio-economic status.

## 2.9 Key outcomes from the literature

Resulting from the review of the literature was the overall aim of this thesis: to undertake a critical exploration and analysis of the value of private gardens, with a key focus on the City of Salford, evaluating the size and content of urban household gardens to determine factors which contribute to the heterogeneity of the resource and their associated benefits. Due to this heterogeneity, and the small sample sizes of previous literature, a key aspect of the research is a large sample size. Previous literature has presented household garden accounts of 250 houses (Gaston et al., 2005) and 267 (Loram, Warren & Gaston, 2008). Further, such detailed studies of garden size and content have yet to explore the affect the socio-economic profile of the household has, and thus the associated potential benefits to residents. Greenspace diversity has been shown to be an important factor in the potential health and wellbeing benefits in several studies (Apparicio et al., 2016; Baker & Smith, 2019; Birch et al., 2020; Cameron et al., 2012; and Dillen et al., 2012.), but not previously reported in household gardens through the lens of household deprivation.

In addition to the socio-economic status of the household, the urban form of a property has been shown to cause considerable variation in the size and content of household gardens (Dewaelheyns et al., 2014; Gill et al., 2008; Lin et al., 2017), but studies within UK cities are lacking (Baker & Smith, 2019; Cameron et al., 2012). Baker and Smith (2019) have shown the proportion of useable greenspace was substantially lower in Victorian terraced housing compared to non-Victorian terraced, semi-detached, and detached housing. But, this study only compared one development era. This potential indication of environmental deprivation warrants further investigation (Baker & Smith, 2019).

# Chapter Three: Methodology and Methods

## 3.1 Research Philosophy

Philosophy provides the general principles for theoretical thinking within research. Two main branches of philosophy are important in the natural and social sciences: ontology and epistemology (Moon & Blackman, 2014). Ontology 'describes the nature of reality, that is, the assumptions that we hold about the physical world' (Jonasson, 1991, p. 8). While epistemology 'is the study of the nature of knowledge and thought' (Jonasson, 1991, p. 8). Or, as described by Moon and Blackman (2014), ontology is concerned with what exists for people to know and epistemology with how people create knowledge. Moon and Blackman (2014) argue an interdisciplinary approach between the natural and social sciences is vital to understand the contemporary ways in which social, political, economic, and institutional factors affect the natural world.

Ontology is important in the natural sciences because it helps to establish a degree of certainty for the researcher (Moon & Blackman, 2014). Ontology exists along a continuum (Nwokah, Briggs & Kiabel, 2009). At the two extremes lie realism and relativism (Nwokah et al., 2009; Moon & Blackman, 2014), see Table 3.1. A realist ontological viewpoint holds that one single reality exists (Moon & Blackman, 2014): everything is black or white. A relativist ontological viewpoint holds that reality is constructed within the human mind (Moon & Blackman, 2014): a shade may be black to one person but grey to another. To illustrate the difference, Proctor (1998) compared whether wilderness is universally defined, measured, and experienced (realism) or if individuals define, measure, and experience wilderness differently (relativism). The argument that wilderness is relatively subjective is more apparent today; as Proctor (1998) writes there may be many different definitions of wilderness, and in contemporary society very few places are truly 'wild'. To some a place that appears unmanaged, for example a wildflower meadow, may be considered wild; but to others it may not, in this example the wildflowers may have been planted by people and thus is this truly wilderness? From an ontological perspective the research in this thesis is that of a critical realist, which lies between the two ends of the continuum in Table 3.1. Critical realism is concerned with understanding the causal mechanisms which exist to create the world we live in.

Table 3.1: The continuum of core ontological viewpoints (adapted from Nwokah et al., 2009).

| Reality as a concrete structure | Reality as a concrete process | Reality as a contextual file of information | Reality as a realm of symbolic discuss | Reality as a social construction | Reality as a project of human imagination |
|---|---|---|---|---|---|
| **Realist** | ⟵ | | | ⟶ | **Relativist** |

Epistemology is concerned with validity, scope, and methods. It influences how researchers frame their work in the discovery for knowledge (Moon & Blackman, 2014). At the two ends of the epistemological continuum are objectivist and subjectivist viewpoints. Objectivist researchers aim to remain detached, reducing the impact of their values and interpretations as much as possible in the generation of knowledge (Pratt, 1998). Objectivists seek methods to test reality by collecting and analysing evidence to explore claims about the real world (Patton, 2002). Whereas subjectivist epistemology holds that knowledge is dependent on how people perceive and understand reality (Moon & Blackman, 2014). As a mid-point between the two, constructivism assumes that different individuals construct meaning of the same object in different ways: how an individual engages with the world is based on their cultural, historical, and social perspectives (Moon & Blackman, 2014). Moon and Blackman (2014) claim constructionist research can best enable governments and stakeholders to design responses to conservation problems with the highest success (Moon & Blackman, 2014; Wayland, Fischer, McGowan, Thirgood, & Milner-Gulland, 2010).

Philosophical perspectives are important in research as they reveal assumptions made by the researcher dependant on their standpoint, and these assumptions drive their choice of methods (Moon & Blackman, 2014). Within inter-disciplinary research, different viewpoints must often be brought together to generate a critical examination of what we can know, what we can learn and how this knowledge can affect the way research is conducted (Moon & Blackman, 2014). Section 2.8 above sets out the four objectives to be tested within this thesis, taking an objectivist approach to research and decision-making. This deductive investigation using quantitative methods aims to keep the quest for knowledge objective, as is the viewpoint of the researcher. The paradigm of environmental injustice is arguably subjective, with different researchers drawing on different experiences and definitions to determine what is unequal. For example, the different definitions presented in Section 2.1 show injustice is inherently more subjective and open to variation, as much research is involving the general population (Barger, Perez, Canelas & Linnenbrink-Garcia, 2018). The methods used to investigate these objectives remain quantitative, but the discussion around subsequent findings are further towards the subjectivist standpoint along the continuum illustrated in Table 3.1. A constructivist viewpoint was key when interpreting data through an injustice lens. As this research is an interdisciplinary approach between social science and urban ecology, the search for knowledge is not fixated within one paradigm. This approach is taken to generate new insights that are not visible from one paradigm alone.

## 3.2 The research location

The research reported in this thesis was conducted within the conurbation of Greater Manchester. Greater Manchester is in the North West of England, one of nine regions of England (ONS(a), n.d.). The North West is home for approximately 13 per cent of England's population. Nearly 40 per cent of people in the North West reside in Greater Manchester. In 2020 the population of Greater Manchester was estimated at 3,350,000 (Citypopulation, 2020), spread across ten metropolitan districts. Greater Manchester is the second most populated urban area in the UK with a population density of 4,051 people per km², London was ranked first with 5,630 people per km² (ONS, 2011). Within Europe Greater Manchester has the 16th largest population of any metropolitan area (Citypopulation, 2020). Greater Manchester was the location of the Urban Pioneer, a DEFRA initiative 2018-2022 established to test new tools and methodologies for investing in and managing the natural environment related to their 25-year environment plan (DEFRA, 2018; Nature Greater Manchester, 2021). The Urban Pioneer was established to explore links between the environment, society, and the economy to enhance the natural environment through improved decision making, expanding Greater Manchester's ecosystem services whilst promoting prosperity (Nature Greater Manchester, 2021). This thesis has contributed evidence to the initiative, illustrating its widely recognised capabilities. The configuration of Greater Manchester's ten districts makes it ideal for the testing and scaling up of the methodology. Gill et al. (2008), for example, chose Greater Manchester for their study on landscape planning due to its full expression of urban environmental characteristics, a full range of neighbourhoods, and various building forms. Giezen and Pellerey, (2021) also note there is a gap in the knowledge to explore the upscaling of greening strategies in both the public and private realm, whilst this thesis does not provide green strategies it does build on the evidence base needed to support them.

The City of Salford, one of the world's first industrial cities, owes much of its historical success to the Manchester Ship Canal; running through the cities of Salford and Manchester, this canal brought great prosperity in Victorian times and was paramount during both world wars for supplies (Owens, 1983). In the 1840s Friedrich Engels described the City of Salford as 'one large working man's quarter', but in modern day it is a thriving mix of waterfront, urban, and countryside environments (Manchester History, n.d.). The City of Salford had an estimated population of 245,600 in 2015 (ONS(b), 2020). The City of Salford was chosen for the urban household garden auditing aspect of this thesis as almost one third of the Lower Super Output Areas (LSOAs) in the City of Salford are amongst the 10 per cent most disadvantaged areas of England. Further, it contains areas which fall into every category along the Index of Multiple Deprivation. Figure 3.2 shows the City of Salford has a spread of LSOAs within different deprivation deciles, making it representative of Greater Manchester as a whole. This ensured the inclusion of a range of household circumstances and socio-economic statuses. Also, as a post-industrial centre Salford showcases a multitude of housing types and development ages ranging from the affordable terraced housing of the industrial revolution to the family homes built due to £44 million invested in the City of Salford housing 2003–2006 (Salford City Council, 2008).

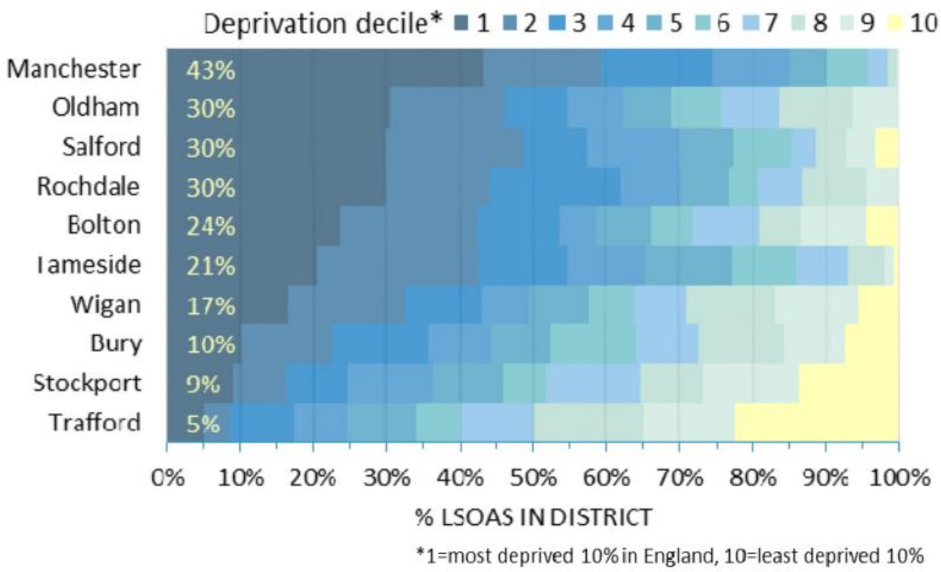



Figure 3.2: The percentage of LSOAs within each deprivation decile by district in Greater Manchester (Manchester City Council, 2019). Percentage in yellow along the left-hand side is the percentage of LSOAs in the 1st (most deprived) decile within each district.

## 3.3 Approach to methods

The methods chosen in this thesis were quantitative; this approach was taken due to the philosophical stance of the researcher (discussed in Section 3.1) and as a response to the gaps in the knowledge found through the review of the literature in Chapter Two, these gaps from a methodological perspective are summarised in this section below. Bagstad et al., (2013) and Wilkerson et al., (2018) note the need to understand the multiple pathways through which socio-environmental variables influence the health and wellbeing benefits to people, with evidence suggesting those in disadvantaged socio-economic classes are more in need of and more sensitive to the positive effects the natural environment can have (James, et al., 2015; Maas et al., 2006; Mitchel & Popham, 2008). To assess this variation of benefits to people within gardens, a large-scale approach was key to this thesis due to the heterogeneity of gardens (Brindley et al., 2018; Dennis et al., 2019; Lin & Fuller, 2013; Siftova, 2021). To achieve an in-depth understanding of this complex issue, a case study approach to research was taken at a city-wide scale using secondary data collection followed by descriptive data analysis. A case-study approach was taken as this is the most appropriate approach to data collection when in-depth explanations of social behaviour are sought after. A table of the secondary sources used is presented below (Table 3.2).

Table 3.2: The secondary datasets presented in this thesis and how they were used.

| **Secondary dataset** | **How it was used** |
|---|---|
| 2011 Area Classification for Output Areas | Classify socio-economic groups |
| Google Earth imagery | Household garden sample |
| RSPB Big Garden Birdwatch | Birdwatching engagement |
| OS MasterMap Greenspace Layer | Greater Manchester Garden extent |
| EDINA UK Buildings Layer | Classify urban form |
| EDINA UK Land Layer | Classify land use |
| NEA wellbeing valuations | Wellbeing valuations |
| Horticultural Trades Association spend | Garden spend |
| Taking Part Survey | Personal investment into gardens |
| Monitoring of Engagement with the Natural Environment | Personal investment into gardens |

The sample of 6881 household gardens in the City of Salford used a quantitative method outlined in Sections 3.5-3.7 below and the data were used to investigate objectives one and three. The OAC classification provided socio-economic data on a national scale at the smallest spatial unit available, Output Area. Further, previous similar research has been limited either by sample size due to the nature of in-person data collection (Gaston et al., 2005) or by the focus of secondary datasets resulting in key garden features being omitted (Davies et al., (2009). To achieve the large-scale landcover data in this thesis, at a fine scale to capture individual features, Google Earth satellite imagery was used. These methods allowed for the large sample size of 6881 gardens, with the fine detail of garden content which qualitative methods such at interviews or questionnaires would not.

Objective two was a spatial investigation (Section 3.5) to provide more up to date evidence of the collective size of urban household gardens. Current publications on city-wide garden extent are dated in 2005 and 2007 (Gaston et al., 2005; Loram et al., 2007; Tratalos, Fuller, Warren, Davies & Gaston, 2007). Recent publications in the field have used these dated estimates (van Heezik et al., 2020; Garcia-Antunez et al., 2023), showing the need for a more contemporary overview as provided by this thesis.

To investigate objective four secondary quantitative data were used to estimate a monetary valuation of gardens (Section 3.12). This was done to reflect the increase in popularity of social return on investment research (Carrick, 2013; Schoen, Caputo, & Blythe, 2020; Yerrelll, 2008), allowing gardens to be included within this realm and compared to other healthcare initiatives. These valuations were also included within the Urban Pioneer, as outlined in Section 7.4. The estimates use QALY, from the National Ecosystem Assessment report on Economic Analysis of Cultural Services (Mourato et al., 2010) (method outlined in Section 3.11), a measure frequently used when assessing benefits to people (Brazier et al., 2002; DEFRA, 2021; Drummond et al., 1997; Martin, Claxton, Lomas, & Longo, 2023).

## 3.4 Socio-economic datasets

To compare the size and diversity of OAC Supergroups household gardens in the City of Salford, objective one of this thesis, households were categorised by socio-economic characteristics using the 2011 Area Classification for Output Areas (OAC), established using data from the 2011 national census (Gale, Singleton, Bates, & Longley, 2016). Table 3.2 outlines the hierarchies used within this section for the geographic units and descriptive groups. The OAC is a hierarchical classification consisting of three tiers, Supergroups, Groups, and Subgroups for output areas across England, Scotland, and Wales. Area classifications are made to illustrate the characteristics of an area for demographic structure, household composition, housing, socio-economic characteristic, and employment patterns. Sixty variables are used to derive the OAC. A full list of all classification groups can be found in Table 3.3.

Table 3.2: A breakdown of the geographic and descriptive characteristics hierarchies used within this project. (NHS Business Definitions, n.d.; ONS(c), n.d.; ONS, 2015). The minimum and mean population estimates provided for the Output areas were estimates used for classification in 2015.

<table>
<tr><th>Hierarchy</th><th>Unit</th><th>Classification</th><th>Definition</th></tr>
<tr><td rowspan="3">Geographic Hierarchy</td><td rowspan="3">Output Areas</td><td>Middle Super Output Area (MSOA)</td><td>Built from multiple LSOAs, used for area statistics.<br>Minimum population = 5000.<br>Mean population = 7200.</td></tr>
<tr><td>Lower Super Output Area (LSOA)</td><td>Built from multiple OAs, used for small area statistics.<br>Minimum population = 1000.<br>Mean population = 1500.</td></tr>
<tr><td>Output Area (OA)</td><td>Used for finer level analysis.<br>Minimum population = 100.<br>Mean population = 300.</td></tr>
<tr><td rowspan="3">Descriptive Characteristics Hierarchy of the UK population</td><td rowspan="3">Classification Groups (OAC)</td><td>Supergroup</td><td>Eight Supergroups formed from national census data (2011).</td></tr>
<tr><td>Group</td><td>26 middle tier Groups. Supergroups contain 2-4 Groups each.</td></tr>
<tr><td>Subgroup</td><td>76 bottom tier Subgroups. Groups contain 2-4 Subgroups each.</td></tr>
</table>

Table 3.3: Output area classifications (ONS, 2015).

<table>
<tr><th>Super-group</th><th>Group</th><th>Subgroup</th><th>Super-group</th><th>Group</th><th>Subgroup</th></tr>
<tr><td rowspan="7">Rural Residents</td><td rowspan="4">Farming Communities</td><td>Rural Workers and Families</td><td rowspan="6">Urbanites</td><td rowspan="3">Urban Professionals and Families</td><td>White Professionals</td></tr>
<tr><td>Established Farming Communities</td><td>Multi-Ethnic Professionals with Families</td></tr>
<tr><td>Agricultural Communities</td><td>Families in Terraces and Flats</td></tr>
<tr><td>Older Farming Communities</td><td rowspan="3">Ageing Urban Living</td><td>Delayed Retirement</td></tr>
<tr><td rowspan="3">Rural Tenants</td><td>Rural Life</td><td>Communal Retirement</td></tr>
<tr><td>Rural White-Collar Workers</td><td>Self-Sufficient Retirement</td></tr>
<tr><td>Ageing Rural Flat Tenants</td><td></td><td>Suburban Achievers</td><td>Indian Tech Achievers</td></tr>
</table>

| | | | | | |
|---|---|---|---|---|---|
| | Ageing Rural Dwellers | Rural Employment and Retirees | Suburbanites | | Comfortable Suburbia |
| | | Renting Rural Retirement | | | Detached Retirement Living |
| | | Detached Rural Retirement | | | Ageing in Suburbia |
| **Cosmopolitans** | Students Around Campus | Student Communal Living | | Semi-Detached Suburbia | Multi-Ethnic Suburbia |
| | | Student Digs | | | White Suburban Communities |
| | | Students and Professionals | | | Semi-Detached Ageing |
| | Inner-City Students | Students and Commuters | | | Older Workers and Retirement |
| | | Multicultural Student Neighbourhoods | **Constrained City Dwellers** | Challenged Diversity | Transitional Eastern European Neighbourhoods |
| | Comfortable Cosmopolitans | Migrant Families | | | Hampered Aspiration |
| | | Migrant Commuters | | | Multi-Ethnic Hardship |
| | | Professional Service Cosmopolitans | | Constrained Flat Dwellers | Eastern European Communities |
| | Aspiring and Affluent | Urban Cultural Mix | | | Deprived Neighbourhoods |
| | | Highly-Qualified Quaternary Workers | | | Endeavouring Flat Dwellers |
| | | EU White-Collar Workers | | White Communities | Challenged Transitionaries |
| **Ethnicity Central** | Ethnic Family Life | Established Renting Families | | | Constrained Young Families |
| | | Young Families and Students | | | Outer City Hardship |
| | Endeavouring Ethnic Mix | Striving Service Workers | | Ageing City Dwellers | Ageing Communities and Families |
| | | Bangladeshi Mixed Employment | | | Retired Independent City Dwellers |
| | | Multi-Ethnic Professional Service Workers | | | Retired Communal City Dwellers |

| | | | | | |
|---|---|---|---|---|---|
| | Ethnic Dynamics | Constrained Neighbourhoods | | | Retired City Hardship |
| | | Constrained Commuters | Hard-Pressed Living | Industrious Communities | Industrious Transitions |
| | Aspirational Techies | New EU Tech Workers | | | Industrious Hardship |
| | | Established Tech Workers | | Challenged Terraced Workers | Deprived Blue-Collar Terraces |
| | | Old EU Tech Workers | | | Hard-Pressed Rented Terraces |
| Multicultural Metropolitans | Rented Family Living | Social Renting Young Families | | Hard-Pressed Ageing Workers | Ageing Industrious Workers |
| | | Private Renting New Arrivals | | | Ageing Rural Industry Workers |
| | | Commuters with Young Families | | | Renting Hard-Pressed Workers |
| | Challenged Asian Terraces | Asian Terraces and Flats | | Migration and Churn | Young Hard-Pressed Families |
| | | Pakistani Communities | | | Hard-Pressed Ethnic Mix |
| | Asian Traits | Achieving Minorities | | | Hard-Pressed European Settlers |
| | | Multicultural New Arrivals | | | |
| | | Inner City Ethnic Mix | | | |

The OAC was used in this research as it gave national data for socio-economics at the smallest geographic scale (Output Area), and is commonly used within demographics research (Dearden, Lloyd, & Catney, 2019; Druckman, Sinclair, & Jackson, 2008; Walford & Armitage, 2020). Alternatively, the Index of Multiple Deprivation (IMD) may have been used, however, these data are only available at the larger Lower Super Output Area level. Also, the IMD places deprivation along a linear gradient. As had been shown in Chapter 2, the relationship between deprivation and UGI is not always linear. Therefore, a categorical approach was taken for this study. The use of the OAC data from 2011 has a drawback of being six or seven years before the household garden sample data (collected 2017/18), but at the time of research it was the most up-to-date version of this dataset, although now the

2021 census has provided new data. In a study comparing the accuracy of proxy measures of socio-economic status for university admissions, Jerrim (2020) found IMD and OAC both correlated moderately with income, r = 0.48 and r = 0.41 respectively, and income deprivation, r = 0.47 and r = 0.46. However, OAC is available at the smaller scale OA and its categorical approach makes comparisons between societal groups more meaningful and descriptive as there is a large amount of supporting background information published.

## 3.5 Sampling technique

To select the sample size and location of Output Areas a stratified sampling technique was used. Stratified sampling is the best way for the researcher to guarantee a sample where appropriate proportions have been randomly selected from within specific strata (Denscombe, 2014; Iliyasu & Etikan, 2021), in this case OAC. According to Table 3.4, for a population of 233,933 (the City of Salford) at a 95 per cent confidence level with 1 per cent margin of error 20,499 individuals should be sampled. Dividing this by the mean number of people within an OA means a sample of 68 OAs would be needed for a representative sample. Sixty-eight OAs were used in analysis equating to 6881 household gardens sampled. Sample Output Areas were chosen based on the frequency at which they were found within the City of Salford (Table 3.5). Output Areas within Groups were chosen using the shape ID in ArcGIS and a random number generator. Nine Groups were not included in the study: two because they did not occur in Salford; five due to low abundance and, therefore, not being selected in the stratified sampling method; and two due to a lack of houses as they only contained apartment blocks (Table 3.5).

Table 3.4: Deciding on sample size for population sampling (Denscombe, 2014).

| Number in the population | Required sample size[1] | | |
|---|---|---|---|
| | 5% margin of error | 3% margin of error | 1% margin of error |
| 50 | 44 | 48 | 50 |
| 100 | 80 | 92 | 99 |
| 250 | 152 | 203 | 244 |
| 500 | 217 | 341 | 475 |
| 1000 | 278 | 516 | 906 |

| | | | |
|---|---|---|---|
| 5000 | 357 | 879 | 3288 |
| 10,000 | 370 | 964 | 4899 |
| 100,000 | 383 | 1056 | 8763 |
| 1 million | 384 | 1066 | 9513 |
| 10 million | 384 | 1067 | 9595 |

Table 3.5: The OAC Supergroups and Groups with the number of Output areas within each in the City of Salford and how many were sampled. Those Groups in bold were not included due to low abundance.

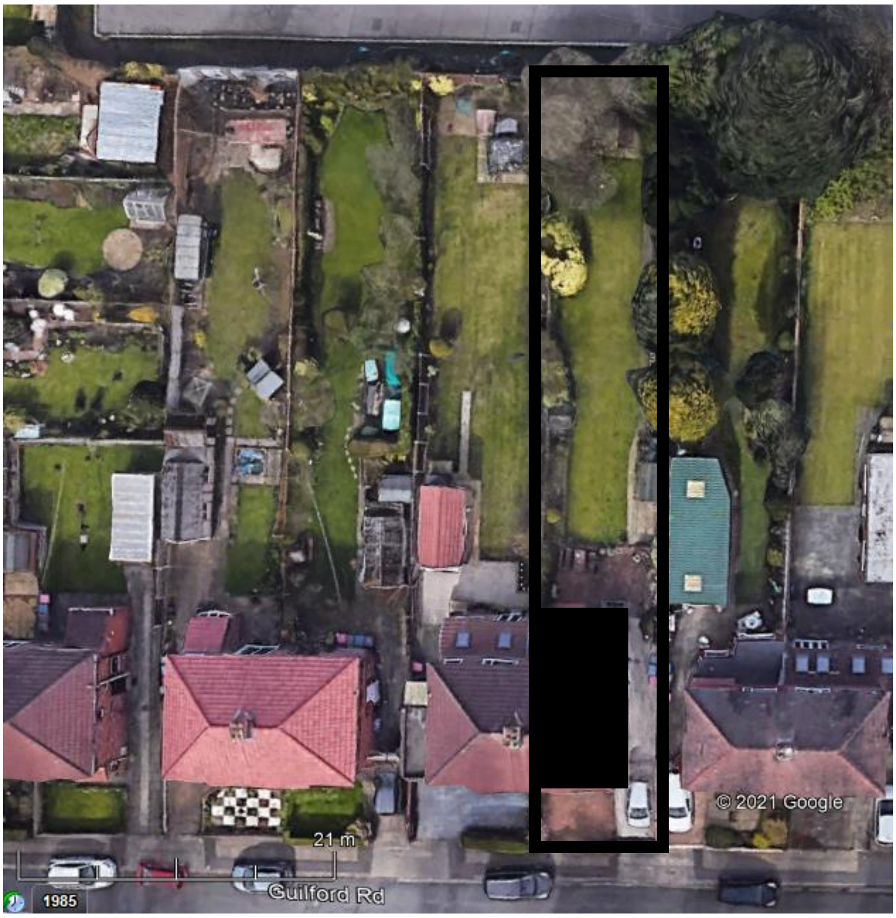
| **Supergroup** | **Group** | **No. of Output Areas** | **OA Sample** |
|---|---|---|---|
| Rural Resident | **Farming Communities** | 2 | 0 |
| | **Rural Tenants** | 0 | 0 |
| | **Ageing Rural Dwellers** | 0 | 0 |
| Cosmopolitans | Students Around Campus | 11 | 1 |
| | **Inner-City Students** | 38 | 0 |
| | Comfortable Cosmopolitans | 8 | 1 |
| | **Aspiring and Affluent** | 8 | 0 |
| Ethnicity Central | Ethnic Family Life | 15 | 1 |
| | **Endeavouring Ethnic Mix** | 2 | 0 |
| | Ethnic Dynamics | 23 | 2 |
| | **Aspirational Techies** | 2 | 0 |
| Multicultural Metropolitans | Rented Family Living | 73 | 7 |
| | **Challenged Asian Terraces** | 5 | 0 |
| | **Asian Traits** | 5 | 0 |
| Urbanites | Urban Professionals and Families | 78 | 7 |
| | Ageing Urban Living | 47 | 4 |
| Suburbanites | Suburban Achievers | 20 | 2 |
| | Semi-Detached Suburbia | 104 | 10 |
| Constrained City Dwellers | Challenged Diversity | 107 | 10 |
| | Constrained Flat Dwellers | 14 | 1 |
| | White Communities | 22 | 2 |
| | Ageing City Dwellers | 21 | 2 |
| Hard-Pressed Living | Industrious Communities | 37 | 3 |
| | Challenged Terraced Workers | 36 | 3 |
| | Hard-Pressed Ageing Workers | 27 | 3 |
| | Migration and Churn | 99 | 9 |
| | **Total** | **804** | **68** |

## 3.6 Household garden sample

Satellite imagery sourced from Google Earth was used to record a baseline of the assets within household gardens across Salford. The UK imagery is updated twice a month, with a full database update annually. Google Earth provides analysis at a resolution less than 1m, this is far lower than other landcover data typically ranging from 30 m to 80 m (Malarvizhi et al., 2016). At this resolution it is possible to see individual features such as buildings and water bodies (Malarvizhi et al., 2016). In addition, Google Earth is a free resource, obtaining similar, high-resolution images can be very expensive (Rahman, Aggarwal, Netzband, & Fazal, 2011) particularly for large-scale research. Eskandari and Sarab (2022) ground-tested the accuracy of using Google Earth to classify landcover in the Zagros Forest, Khuzestan, comparing data from 270 plots it was found Google Earth had a 92 per cent accuracy level to ground reality. Further, Pulighe, Baiocchi, and Lupia (2016) found Google Earth was highly accurate when ground-truthing images to test horizontal accuracy in the city centre of Rome, Italy, with overall positional accuracy close to 1 m at the 95 per cent confidence level. Depending on the area, the imagery used in this study were from March 2017 April 2018. Photo manipulation tools, zoom, change photo orientation, and street view, were used to provide the clearest, most detailed image of each individual space.

Data for landscape content of gardens has not been collected in this way previously, thus a novel methodology was produced as outlined below. Data were collected by the same researcher for all sites. Data were collected from images taken during months when canopy cover was greatest (spring/summer). Figure 3.3 illustrates how features were identified. Buildings, for example sheds or garages, were not included within the recording of garden size or content as this project focused on natural features and feature diversity. A polygon drawn onto the software and pasted into earthpoint.us gave the total area of each space in $m^2$. Individual mature trees and shrubs were counted. The percentage cover was then estimated for each category: trees and shrubs, plants, lawn, and pave to the nearest 5 per cent. Trees & shrubs percentage cover were calculated together as it was sometimes difficult to distinguish between canopies. Using the example in Figure 3.3, the largest features would be recorded first, pave estimated at 40 per cent and lawn also at 40 per cent. The remaining 20 per cent would be divided between the other three vegetation features. Plants would be given 5 percent, and the remaining 15 per cent to trees and shrubs. If a feature was visible, but only occupied a small space (<5 per cent) it was rounded up to 5 per cent and the space taken from its nearest neighbour feature. For example, if a garden only contained lawn and two shrubs, the shrubs were estimated at 3 per cent cover and the lawn at 97 per cent, the garden would be recorded as 5 per cent trees & shrubs and 95 per cent lawn. This was done to ensure the highest level of feature diversity was recorded. Statistical testing was done using a combination of MiniTab 18 and Microsoft Excel software.

This approach to data collection negated any disputes with gaining access to private gardens, an issue highlighted multiple times within the literature (Bell et al., 2020; Dewaelheyns, Rogge, & Gulinck, 2014; Maantay, 2007; Perry & Nawaz, 2008; Vaz et al., 2017). It allowed for the large sample size missing from current research (Brindley et al., 2018; Dennis et al., 2019; Lin & Fuller, 2013), and data collection completed by one researcher would have less subjective decision-making when classifying landcover when compared to a citizen science approach as used in Cavan et al. (2018). The data, however, have a few noteworthy limitations. From a temporal perspective, the household sample data provide a snapshot of the gardens in 2017–18. There is also a level of researcher bias; but this was reduced where possible by the same researcher classifying all 6881 gardens and was necessary to obtain data at this fine scale. Also, distinguishing between individual canopies was not always possible, resulting in tree and shrub percentage land cover being grouped to reduce discrepancies.


© 2021 Google
21 m
1985
Guilford Rd

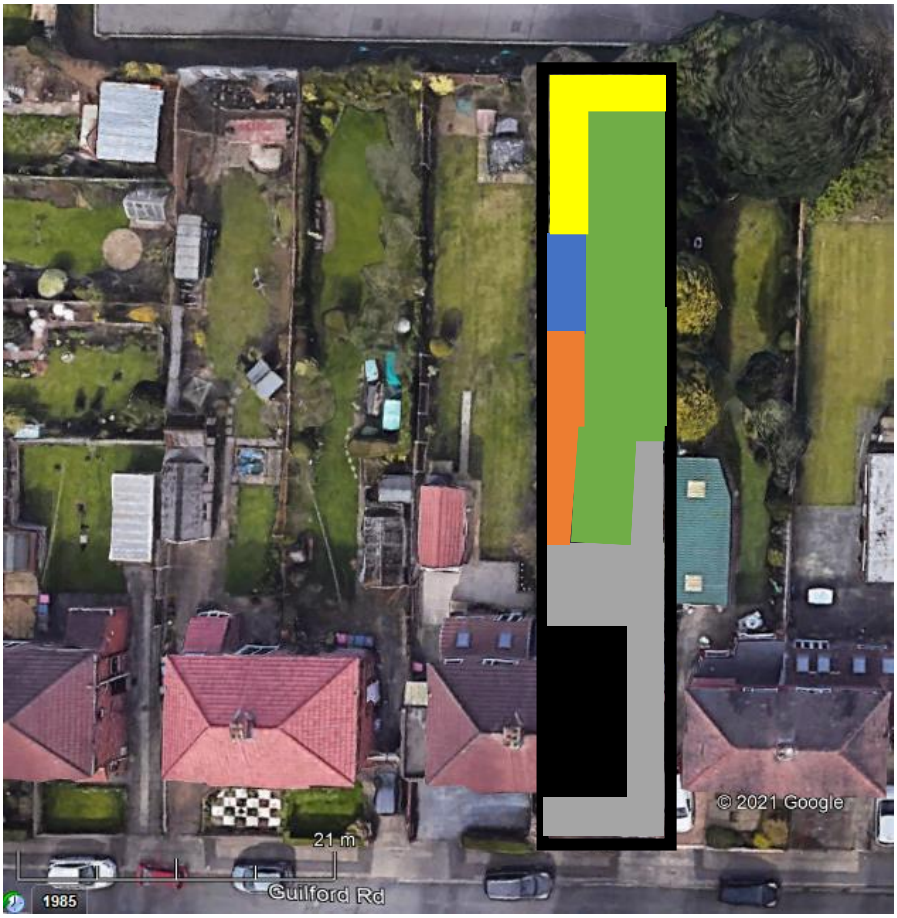

© 2021 Google
21 m
1985
Guilford Rd

Figure 3.3: An image taken from Google Earth of a typical semi-detached house within this study. Top: the space within the property which would have been recorded. Bottom: The features picked out by the researcher, orange = plants, grey = pave, green = lawn, yellow = individual tree, and blue = individual shrub. Note: photo manipulation tools were used to better identify features (Google Earth, 2019).

## 3.7 Variation in the observation of bird sightings within gardens

The Royal Society for Protection of Birds (RSPB) gave access to a dataset of presence/absence data of bird species recorded within gardens. Launched in January 2019, the RSPB Big Garden Birdwatch encouraged residents to record the birds seen within their gardens across the UK; data were recorded by residents for one hour between the 29$^{th}$ – 31$^{st}$ January 2019. This secondary data was overlaid onto a shapefile containing the OAC Supergroups across the City of Salford within Arc GIS Pro. Using the Union toolset, the data were combined to calculate: the total number of birds recorded, the number of households who participated, and a mean bird abundance for each Supergroup.

## 3.8 Garden extent

For objective two 'To determine whether household gardens as a collective resource are equally distributed across districts of Greater Manchester, greenspace data were obtained from the OS Matermap Greenspace Layer (Ordnance Survey, 2017) for the North West, downloaded from the Edina Digimap service. Local authority boundary data were clipped to a shapefile containing the spatial extent of Greater Manchester using the Geoprocessing toolset in ArcMap 10.5. The area covered by household gardens, as defined in the OS Mastermap Greenspace dataset, within each region of Greater Manchester was tabulated using the tabulate area tool in the ArcMap 10.5 zonal statistics toolset. For this study, these data are presented for all ten districts within Greater Manchester. The resolution of the dataset was 3m. This allowed for an accurate account of the spatial extent of household gardens across a large area, Greater Manchester. It was at a finer scale to previous work undertaken by the Green Infrastructure and the Health and Wellbeing Influences on an Ageing Population (GHIA) (2016-2019) funded by the Natural Environment Research Council, the Arts and Humanities Research Council, and the Economic and Social Research Council. The GHIA project had a resolution of 10m (Dennis et al., 2018). The work in this thesis produced an account, by district, of the extent of household gardens at a 3m resolution.

## 3.9 Urban Form

For objective three, 'to assess the change in garden size and diversity in housing developments over time in the City of Salford' shapefiles containing property data were downloaded from EDINA digimap 2020. The UKBUILDINGS layer provided the footprints of buildings across Great Britain, dividing individual buildings into residential, non-residential, and mixed use. These data were combined with the UKBUILDINGS AGE layer which provides information on the era of the housing development split into five categories: Historic (up to 1914), Interwar (1918-1939), Post-war (1945-1959), Sixties/Seventies (1960-1979), and Modern (1980-current). Within each sample OA, chosen as outlined in sections 3.5 and 3.6, the era of development was matched to the record on the individual property's garden features. The UKBUILDINGS AGE layer was developed as part of a pilot scheme, made accessible to members for a short period ending July 2020, to create a unique database to help understand the age, structure, characteristics, and use, of commercial, public, and residential buildings across the UK (Edina Digimap, personal communication, 31st March 2020). As such, to the knowledge of this researcher, it has not yet been used for publication.

The UK LAND layer provided land use classification for housing density. Land use is split into 25 categories, the four of interest here are: low density residential with amenities, medium density residential with high streets and amenities, high density residential with retail and commercial sites, and urban centres – mainly commercial/retail with residential pockets. Due to very low abundance within the sample areas and their similarity in building density, high density and urban centres were grouped into one classification for analysis. Residential building density is included in the EDINA UKLAND database. The classifications are defined manually based on the number of buildings within the local area (B. Tyrell, personal communication, 24 August 2020). It is completed by a trained employee within the EDINA Digimap team, a resource utilised by planners, the telecoms industry, government organisations, and researchers (Osborne & Alvares-Sanches, 2019) making it a valued and trusted resource. Combining these datasets gave the development era, housing density, garden size, and garden feature (garden size and feature taken from the household sample) for each individual house within the 6881 sample. These characteristics were then analysed to investigate the change in garden size and diversity over time.

### 3.10 Health and wellbeing valuations

To investigate objective four, ‘to evaluate the economic value of urban household gardens compared to that of public urban greenspace’, an estimate for the monetary value of the health and wellbeing benefits of household gardens was created based on the UK National Ecosystem Assessment report on Economic Analysis of Cultural Services (Mourato et al., 2010). These data are based on a web survey on respondent’s general and physical health (n=1851) and the geo-located survey data are used to estimate the physical and mental effects associated with different UK habitats, one of which being household gardens. For general and physical health, the RAND SF-36 Health survey was employed (RAND Health Care, n.d.), a leading general health measure comprising of 36 items scored to record quality of life measures (Mourato et al., 2010). In addition, questions were asked regarding views of greenspaces and water from residents’ homes, a factor shown to be important in reducing chronic stress (Section 2.6), and the frequency of use of domestic gardens. Demographic variables including gender, age, qualifications, work status, religiosity, income, house price, and postcode were all controlled for in the original dataset.

The RAND SF-36 health measure can detect changes in health within a population (Hemmingway et al., 1997). As such, it is possible to estimate a monetary value of the health benefits associated with the number of people benefitting from both using greenspaces and having greenspace views (Mourato et al., 2010). To do so, a preference weighted utility score was created from the RAND SF-36 to create an index which can then be used to generate Quality Adjusted Life Years (QALY) (Brazier et al., 2002). A Quality Adjusted Life Year is a measurement of health benefits which combine length of life with quality of life (on a scale from zero meaning death to one meaning full health) to assess the value of medical interventions (Drummond et al., 1997). These QALY were then associated with the environmental changes ‘Having a view over greenspace’ and ‘Use of non-countryside greenspace’ based on the results from the RAND SF-36.

A tentative monetary value was then placed on a QALY using as an anchor the value of the prevention of a non-fatal injury (which ranges from a short hospital stay through to permanent paralysis). QALY were estimated ranging from a median of £6414 to £21,519 per individual per annum depending on age of the individual and their health (Mason et al., 2009). This approach was chosen as the environmental changes in question, for example views of greenery, are most likely to impact quality of life, not extend life expectancy as some valuations assume (Mourato et al., 2010).

The UK NEA report (Mourato et al., 2010) provides a range for the estimated economic value per person of each environmental good. For the purpose of this study, both the lowest and highest value has been reported. To calculate 'Use of Garden' the lowest and highest economic values were multiplied by the number of adults (>16yrs) who were likely to garden (50.5 per cent according to Taking Part Survey 2018/19) in 2018. The population of each district of Greater Manchester was reported for 2011, and then multiplied by 4.2 per cent to account for population increase up until 2018. Population increase was taken as the average increase seen in England over that period (ONS, 2019b). This was then multiplied by 1.26 to account for inflation from the publishing of the values in 2010 (Bank of England, 2019). A similar method was taken for the 'Use of non-countryside greenspace', whereby the adult population (>16yrs) in 2018 who frequented public parks (57 per cent according to Taking Part 2015/16) was used. The question regarding frequenting parks was no longer included in the Taking Part survey after this date, therefore the estimate for 2015/16 was used. The changes shown in Chapter Six for the proportion of people who garden show the difference between years 2015-2018 were minimal at 1.3 per cent. This suggests a similar, small increase may have occurred for people frequenting parks; but, to avoid an overestimation, the conservative estimate of 57 per cent was used. For the parameter 'Having a view over greenspace' greenspace was interpreted as exclusively a household garden. It is unclear from the report what was specified as a 'greenspace view' and whether this would include household gardens. However, to be able to report on an economic estimate, it was assumed that a household with a garden had a 'greenspace view'. The number of households (ONS, 2016) with a garden was 87 per cent (Davies et al., 2009). This figure was multiplied by the lowest and highest economic value and then by 1.26 for inflation (Bank of England, 2019).

The economic valuations, however, are tentative results. The UK National Ecosystem Assessment report on Economic Analysis of Cultural Services states an important caveat regarding the difficulties in making such estimates. For example, there are multiple interpretations of valuing QALY and no solid consensus on how to do so (Tilling et al., 2009;

Willis, 2005). Contemporary research within the field has favoured a social return on investment approach, with literature on nature-based health intervention activities in public UGI becoming more popular (Allen & Balfour, 2014; Bosco, Schneider, & Broome, 2019; Carrick, 2013; van den Berg, 2017; Yerrelll, 2008) and in policy (DEFRA, 2022). But there is a lack of such research within private gardens, and there has yet to be an account comparing the benefits of public UGI, private UGI, and green views using the same methodology: a gap which this thesis aims to fill. Further the research has been conducted using secondary data from the UK NEA, boasting a large sample size and a reputable source, meaning these estimations, although tentative, are an important resource. This methodology was not repeated in the UK NEA Follow On, which focused on collecting data on gardening engagement, discussed further below (Section 3.13-3.14) (Church, Fish, Haines-Young, Mourato, Tratalos, 2014). Whilst the estimates do have some drawbacks, being based on data from 2010 and the interpretation of 'view over a greenspace' the most noteworthy, they do provide an interesting comparison and tangible evidence of the potential benefits of gardens. Comparing them to similar SROI in Section 7.4 shows the estimates to be sensible and comparable to that in UGI.

## 3.11 Personal investment: money

The most up to date figures available, provided by the RHS, on national garden spend were from the Horticultural Trades Association (HTA) 'The Economic Impact of Ornamental Horticulture and Landscaping in the UK' 2018 which provided annual figures for 2017. The figures reported were obtained from Kantar TGI's product-by-product sales data, they were triangulated and checked using data from ONS Family Spending, Kantar, and HTAs own survey evidence. Data on gardening services were estimated using ONS Annual Business Survey combined with the ONS Annual Population Survey to encompass self-employed workers.

The HTA reported figures for annual garden spend on goods at UK and North West level, and for annual garden spend on services at UK level for 2017. To calculate figures for the spend on goods in Greater Manchester and its districts the total spend for North West was divided by the number of adults over 16 years old who were likely to garden, estimated at 50.6 per cent of the population (Taking Part Survey 2017/18). This per person spend was then multiplied by the total adult population in 2018. Figures were then multiplied by 1.03

to account for a 3 per cent inflation rate from 2017-2018 (Bank of England, 2019). A similar method was used for the total spend on services in the North West, Greater Manchester and its districts, using the estimate reported by the HTA for the UK in 2017. Garden spend estimates were calculated for 2018 to match the garden sample data which were collected between March 2017 and April 2018. With the largest number of houses being audited towards the latter end of this timeframe, it was decided spend data would be reported for 2018.

## 3.12 Data analysis

Data from the household garden sample (n=6881) were analysed in MiniTab 18 and Microsoft Excel. Data distribution was non-normal; therefore, Spearman's rank correlations were used to test the strength of relationships of variables throughout Chapter Five (Figure 5.3 – 5.7).

# Chapter Four: Environmental injustice in urban greenspaces.

## 4.1 Introduction

Environmental injustice in urban greenspaces (see sections 2.1 and 2.2) occurs within both public (Apparicio et al., 2016; Birch et al., 2020; Burt, Mayer, & Paul, 2021; Fong, 1994; Jones, Hillsdon, & Coombes, 2009; Mears et al., 2019; Methorst et al., 2021; Vaz et al., 2017; Wustemann, Kalisch, & Kolbe, 2017) and private land (Bell et al., 2020; Chamerlain et al., 2019; Kendal, Williams, & Williams, 2012; Kos, Bujoczek, & Bujoczek, 2021; Maantay, 2007; Shanahan et al., 2014; van Heezik et al., 2013; Vaz et al., 2017). The issues of unequal access and greenspace quality fuelled by social class and ethnicity is now at the forefront of current research (see section 2.7.1). Presented first in this chapter is a case study of the City of Salford to explore the differing household gardens across a city. Data were collected by OAC Supergroup to provide key socio-economic data at the small geographic scale of Output area, a novel approach within the field. Lastly, the variation in bird observations across the City of Salford are examined using the same OAC Supergroups to enable a comparison of gardening engagement activities between different societal groups. The research presented within this chapter investigate objective one of this thesis: to compare the size and diversity of OAC Supergroups household gardens in the City of Salford.

## 4.2 Socio-economic Parameters

Previous studies have highlighted the need for a multi-dimensional approach to categorising socio-economic status in research (Section 2.1). As discussed in Section 3.6 the IMD is a commonly used source for such research in England. But it is limited in its geographic scope as data are reported at the larger LSOA scale. For this thesis, the smaller OA was chosen as it was a main aim of the research to gather a substantial database of gardens at a finer scale. Therefore, OAC Supergroups were chosen to provide the socio-economic data, a brief description of which can be found in Table 4.1. Further justification can be found in Sections 3.6 and 3.7.

Table 4.1: A brief overview of each Supergroup taken from the ONS pen portraits (ONS, 2011).

| Supergroup | Brief overview |
|---|---|
| Constrained City Dwellers | Social renting and overcrowding, low level qualifications, high unemployment. |
| Cosmopolitans | Communal living and private renting, high ethnic mix, young adults without children, high proportion of students. |
| Ethnicity Central | Communal living and private renting, high ethnic mix, young adults, high unemployment. |
| Hard Pressed Living | Socially renting, high proportion of UK born residents, low level qualifications, high unemployment. |
| Multicultural Met. | Private and social renting, high ethnic mix, families with children, below average qualifications, high unemployment. |
| Suburbanites | Home-owners, middle to retirement age, higher level qualifications, low unemployment. |
| Urbanites | Private renting, lower rates of ethnic mixing, low unemployment. |

Data for a total of 6881 gardens were collected across the City of Salford, split across 68 OAs and seven OAC Supergroups. Presented below is the median garden size for each Supergroup (Table 4.2) followed by radar graphs depicting the mean percentage cover of garden features (Figure 4.1). Garden features chosen, as outlined in Section 3.8 were trees & shrubs, plants, lawn, and pave. Also shown within Figure 4.1 is the presence/absence data for each feature.

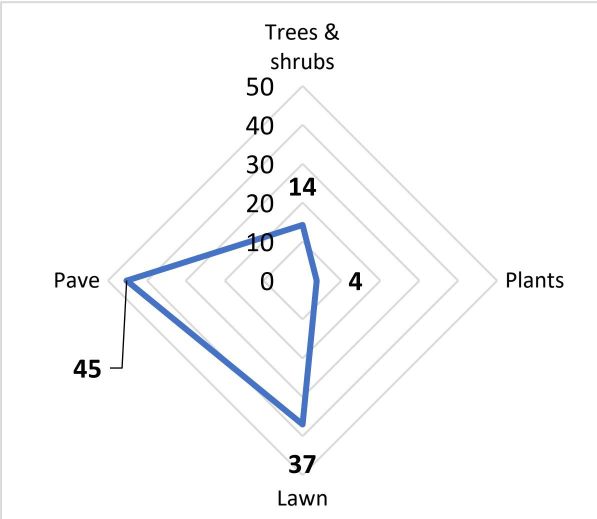
Table 4.2: Mean, median, and range of garden size within each Supergroup and overall.

| Supergroup | Mean Garden Size ($m^2$) | Median Garden Size ($m^2$) | Range ($m^2$) |
|---|---|---|---|
| Suburbanites | 213.93 | 176.00 | 17.00 – 1725.00 |
| Urbanites | 214.59 | 149.00 | 19.00 – 1527.00 |
| Multicultural Metropolitans | 83.30 | 63.00 | 9.00 – 396.00 |
| Ethnicity Central | 131.05 | 122.00 | 29.00 – 656.00 |
| Cosmopolitans | 44.80 | 20.00 | 20.00 – 445.00 |
| Constrained City Dwellers | 111.74 | 99.00 | 10.00 – 869.00 |
| Hard-Pressed Living | 145.91 | 141.00 | 15.00 – 576.00 |
| Overall | 139.00 | 129.00 | 9.00 – 1725.00 |

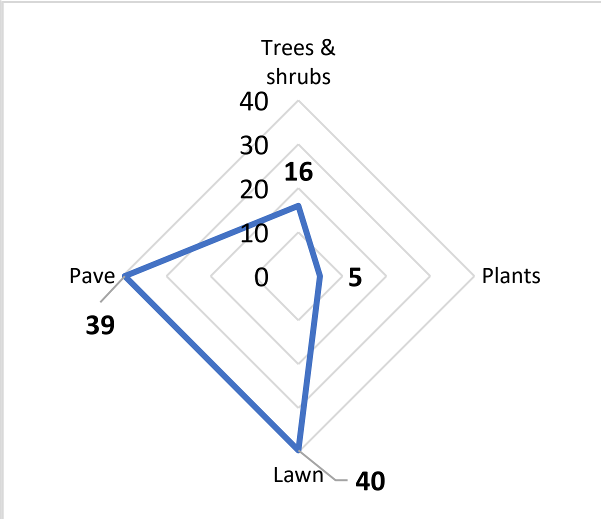
Data in Table 4.2 are non-normal; therefore, the median better reflects the central tendency for each Supergroup. Median garden size was similar between Suburbanites, Urbanites, and Hard-pressed Living, 176 $m^2$, 149 $m^2$, and 141 $m^2$ respectively. Ethnicity Central and

Constrained City Dwellers were also similar, 122 m² and 99 m². The two Supergroups with the smallest median garden size were Multicultural metropolitans at 63 m² and Cosmopolitans at 20 m². Garden size is directly linked to house type with average size approximately doubling from terraced, to semi-detached, to detached houses. This trend is illustrated in Chapter 5, as well as within the literature (Dewaelheyns et al., 2014; Gill et al., 2008). Suburbanites are typically characterised as homeowners in semi-detached or detached properties, whereas Urbanites are more likely to be found in flats or terraced houses as private renters (ONS, 2011). Within the study area Suburbanites had a mode house type of semi-detached (n=853); but Urbanites also had a mode of semi-detached (n=623). Within the Pen Portrait description of Urbanites it states the Supergroup is most likely found in urban areas of Southern England (ONS, 2011). As house prices are generally cheaper further North in England (ONS, 2021b), this may explain why in the City of Salford Urbanites more commonly inhabit semi-detached houses rather than smaller terraced or flats. Multicultural Metropolitans and Cosmopolitans within the study area both had a mode house type of terraced, n=889 and n=164 respectively. The Pen Portraits describe both Supergroups as more likely to inhabit terraced housing, both private and social renting, with a higher proportion of communal establishments. Within the City of Salford the data suggests this is the case. Additionally, the University of Salford and close proximity to other major universities in the City of Manchester will increase the amount of communal living.

The mean for each Supergroup is also reported in Table 4.2, as previously stated the data are not normally distributed, but previous literature has reported the central tendency as the mean garden size and therefore it has been included as a comparison only. Previously, mean garden size within England was reported as 151 m² (Gaston et al., 2005), 190 m² (Davies et al., 2009), and 197.5 m² (Loram et al., 2008). Comparing to the mean garden sizes recorded, Hard-pressed Living 146 m² or Suburbanites 214 m² were closest to the national average garden size.



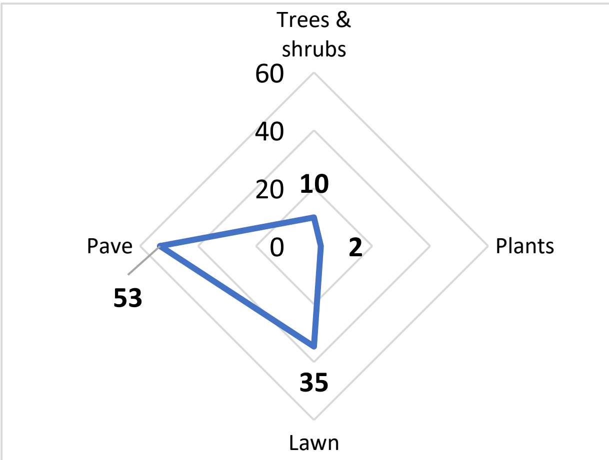

A) Urbanites



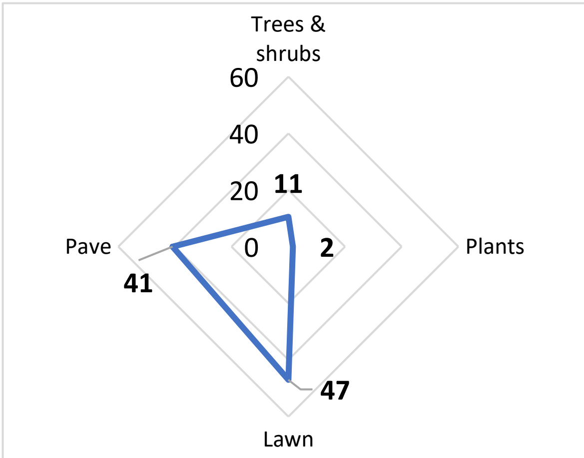

B) Suburbanites

| Mean presence/absence | | | | |
|---|---|---|---|---|
| **Trees** | **Shrubs** | **Plants** | **Lawn** | **Pave** |
| 0.27 | 0.56 | 0.28 | 0.73 | 1.00 |

| Mean presence/absence | | | | |
|---|---|---|---|---|
| **Trees** | **Shrubs** | **Plants** | **Lawn** | **Pave** |
| 0.36 | 0.73 | 0.34 | 0.91 | 0.99 |



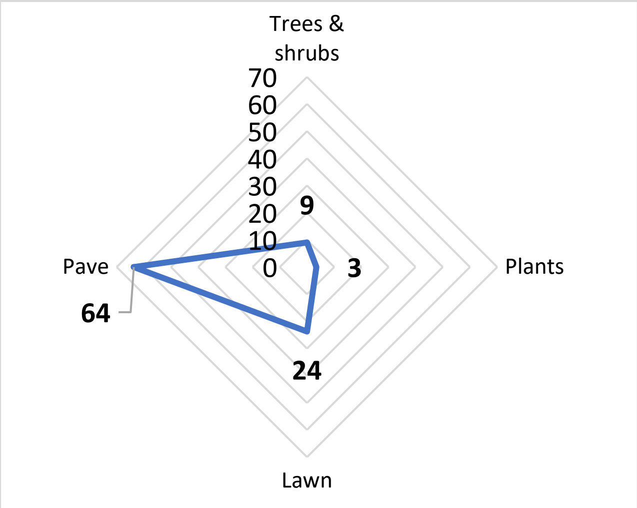

C) Multicultural Metropolitans



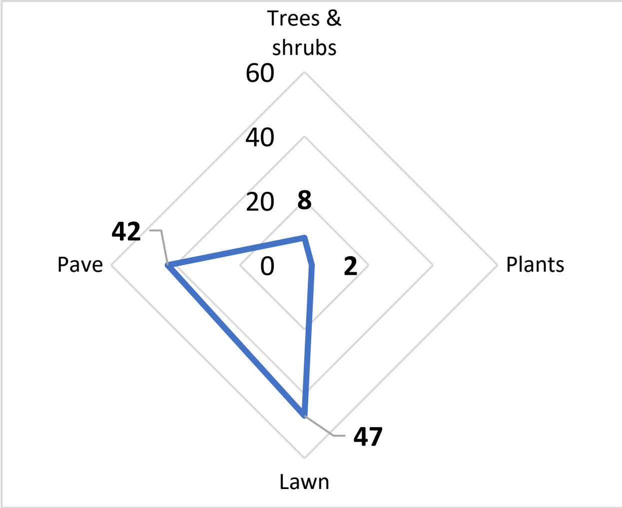

D) Ethnicity Central

| Mean presence/absence | | | | |
|---|---|---|---|---|
| **Trees** | **Shrubs** | **Plants** | **Lawn** | **Pave** |
| 0.16 | 0.29 | 0.13 | 0.43 | 1.00 |

| Mean presence/absence | | | | |
|---|---|---|---|---|
| **Trees** | **Shrubs** | **Plants** | **Lawn** | **Pave** |
| 0.26 | 0.39 | 0.16 | 0.89 | 1.00 |



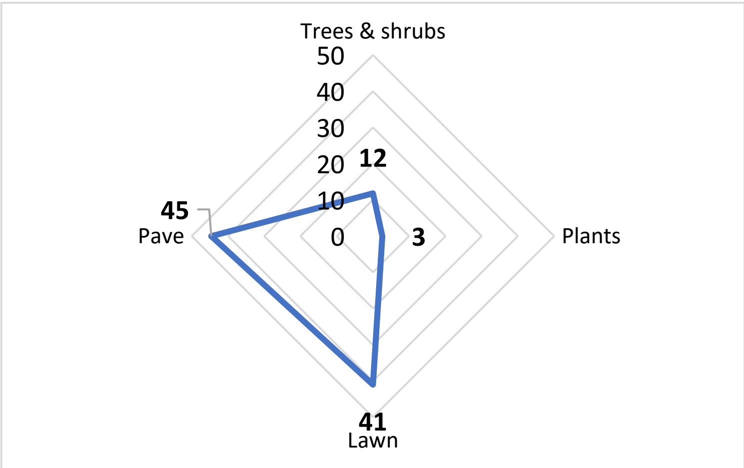

E) Cosmopolitans




F) Constrained City Dwellers

| Mean presence/absence | | | | |
|---|---|---|---|---|
| **Trees** | **Shrubs** | **Plants** | **Lawn** | **Pave** |
| 0.05 | 0.79 | 0.63 | 0.21 | 1.00 |

| Mean presence/absence | | | | |
|---|---|---|---|---|
| **Trees** | **Shrubs** | **Plants** | **Lawn** | **Pave** |
| 0.16 | 0.37 | 0.21 | 0.78 | 1.00 |




G) Hard-Pressed Living

| Mean presence/absence | | | | |
|---|---|---|---|---|
| **Trees** | **Shrubs** | **Plants** | **Lawn** | **Pave** |
| 0.19 | 0.50 | 0.17 | 0.76 | 1.00 |

| Supergroup | Mean Feature Abundance (features per garden) |
|---|---|
| Suburbanites | 3.32 |
| Urbanites | 2.84 |
| Multicultural Metropolitans | 2.01 |
| Ethnicity Central | 2.69 |
| Cosmopolitans | 2.68 |
| Constrained City Dwellers | 2.52 |
| Hard-Pressed Living | 2.62 |

Figure 4.1: The mean percentage cover of trees & shrubs, plants, lawn, and pave shown in the radar graphs and the mean presence/absence for each category shown in the tables, split by Supergroup, with the overall mean feature abundance in the final table. A) Urbanites B) Suburbanites C) Multicultural Metropolitans D) Ethnicity Central E) Cosmopolitans F) Constrained City Dwellers G) Hard-Pressed Living. Trees and shrubs were recorded together for cover but separately for presence/absence data.

Figure 4.1 illustrates some difference in the configurations of the gardens belonging to each Supergroup. Radar graphs A, B, C, D, F, and G all have a similar overall shape depicting a similar percentage cover of pave and lawn (equating to approximately 80 per cent of the total garden size) and a small amount of vegetation cover (approximately 20 per cent) dominated by tree and shrub canopy. However, graph C Multicultural Metropolitans has a greater proportion of pave, although it does follow a similar overall configuration. Graph E Cosmopolitans has a largely different shape depicting a much greater proportion of pave than lawn (approximately a 3:1 ratio) and a small amount of vegetation. Cosmopolitans had the smallest gardens (median 20 $m^2$) with Multicultural Metropolitans having the second smallest gardens (median 63$m^2$) (Table 4.2). A comprehensive account of garden land cover has not previously been published: Loram et al. (2008) recorded 43 per cent grass cover and 20 per cent patio cover (see Table 2.2), but this account did not include vegetation. Samus et al. (2022) recorded 16 physical and 13 flora garden features through a citizen science study, but the data collection method was arguably highly subjective, and the sample size was small (n = 261). Whilst the lawn cover in Loram et al. (2008) is comparable to the radar graphs above, the patio is half that of the pave cover in Figure 4.1. However, the sum of all the impermeable land cover in Loram et al. (2008) is 56 per cent (patio, sheds, gravel, decking, greenhouse, and water butt) which is similar to the range in Figure 4.1.

The uneven distribution of vegetation within greenspaces has been exposed within the literature, more commonly in public rather than private spaces. In Australia, Luck et al. (2009) found those within higher income socio-economic communities had greater vegetation cover, combining both public and private land. Again, in Australia, tree cover $R^2$ = 0.31 correlated with increased socio-economic status (Shanahan et al., 2014); the proportion of graduates within an area was also found to correlate with tree cover, a potential proxy for socio-economic status. Apparicio et al. (2016) reported significantly less vegetation cover in low-income city apartment blocks in Montreal. Within the private realm, Grove et al. (2006) found lifestyle behaviour to describe best the variation in vegetation cover, $R^2$ = 0.34 for tree cover and $R^2$ = 0.32 for grass in Maryland. Investigations into this relationship within England are lacking.

Figure 4.2 shows Suburbanites had the largest combined percentage vegetation cover at 21 per cent and Constrained City Dwellers the smallest at 10 per cent. Whilst the OAC Supergroups are not placed on a socio-economic scale of deprivation like the studies above, comparing vegetation cover in this way exposes a significant difference between societal groups and garden composition. Further, comparing the mean overall feature abundance, Suburbanites had the highest at 3.32 features with Urbanites second at 2.84 features. Hard-pressed living, Cosmopolitans, Ethnicity Central, and Constrained City Dwellers all had similar mean feature abundance ranging from 2.69 to 2.52 features. Feature abundance records the number of features present, features included were trees, shrubs, plants, lawn, and pave, therefore the feature abundance is on a scale from 1 to 5. As the quality of greenspace directly affects its benefit potential (Section 2.6) this highlights an injustice between the health and wellbeing services of gardens to their residents. The data suggest Suburbanites may receive far greater benefits, including reduced chronic stress (Chalmin-Pui et al., 2020), from their gardens due the greater diversity of landcover and higher percentage of greenery than residents within a different Supergroup.

Jerrim (2020) classified those with the Supergroups Constrained City Dwellers and Hard-pressed Living along with those in the Groups Ethnic family living, Endeavouring ethnic mix, Ethnic dynamics, and Challenged Asian terraces as the most disadvantaged within the population. The Groups Challenged terraces and Endeavouring ethnic mix do not occur in the City of Salford so were not included in this study. Constrained City Dwellers had a median garden size of 99 m², a mean feature abundance of 2.52 features per garden, and a mean vegetation landcover of 10 per cent. Hard-Pressed Living had a median garden size of 141 m², a mean feature abundance of 2.62 features per garden, and a mean vegetation landcover of 15 per cent. Whilst the gardens of residents within Hard-pressed Living are smaller and less diverse than some, Suburbanites and Urbanites for example, they are arguably not the most environmentally deprived. Constrained City Dwellers have the smallest proportion of vegetation within their gardens at 10 per cent and the second lowest feature diversity of 2.52 features per garden.  Multicultural Metropolitans have the second smallest gardens (median 63 m²), have the lowest feature abundance at 2.01 features per garden, and a mean vegetation cover of only 12 per cent. This suggests these residents are deprived of quality private greenspaces.

All gardens shown in Figure 4.1 were dominated by Pave and Lawn land cover. Lawn exceeded Pave cover in Suburbanites, Ethnicity Central, and Constrained City Dwellers, but only marginally. Pave cover ranged from 39 – 64 per cent of the total garden area. This large area of impermeable land will have a considerable impact on localised flooding, exacerbated by the recent trend in paving over front gardens (Smith et al., 2011; Warhurst, Parks, McCullock, & Hudson, 2014). In addition, the recent trend of replacing living lawns with artificial grass is cause for concern, with the synthetic polymer grass producing significantly more runoff which is comparable to that of hot rolled asphalt (Simpson & Francis, 2021). The City of Salford is inherently a high-risk flood area, with flat topography and within the catchments of the Irwell and Glaze Brook rivers (Salford City Council, 2008). As a result of climate change, surface water flooding is predicted to become more frequent and widespread with one climate model suggesting a 56 per cent increase in rainfall by 2080 leading to an increase in run-off up to 82 per cent in Greater Manchester (Handley & Carter, 2006). Many UK cities recognise the risk increased paving has on ecosystem services including flood vulnerability (Kazmierczak & Cavan, 2011; Warhurst et al., 2014). Although artificial grass cannot be determined through the methods in this thesis, the data in Figure 4.1 illustrate the need for at the least paving to be included in the cumulative impact of impermeable landcover in gardens, an under-researched area of the urban planning community (Scott, Shannon, Hardman & Miller, 2014; Simpson & Francis, 2021).

## 4.3 Variation in the observation of bird sightings within gardens

To further the investigation into how different Supergroups vary, their gardening engagement was analysed using raw secondary data from the 2019 RHS Big Garden Birdwatch. Firstly, the number of participating households is mapped (Figure 4.2), then Table 4.3 gives the estimated population, number of participating households, and the participation percentage for each Supergroup within the City of Salford, to compare differences in the level of engagement. Bird abundance across the city is then mapped (Figure 4.3) and presented as a mean bird abundance by Supergroup (Table 4.4).

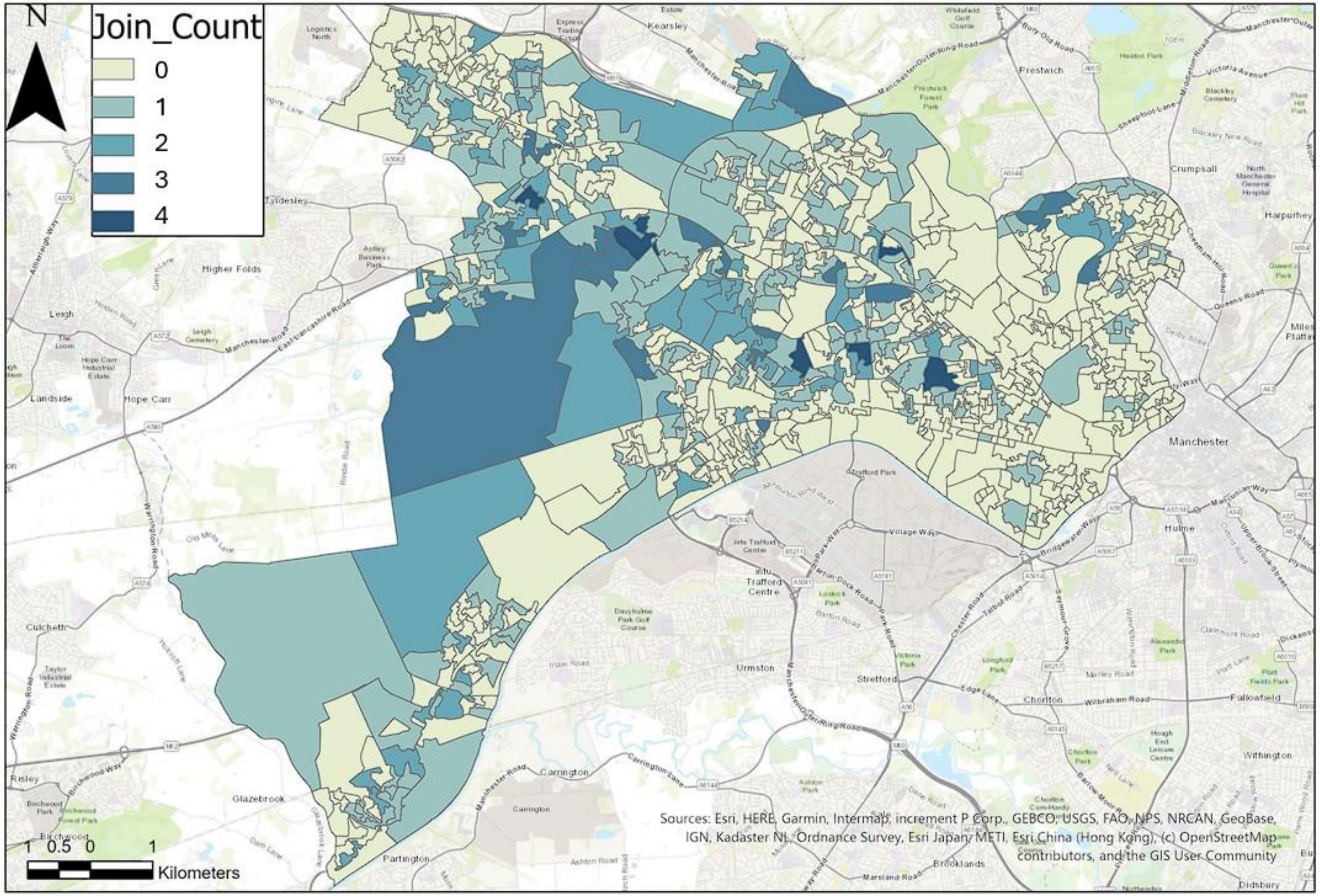



Figure 4.2: The City of Salford, split into Output areas, shaded with the number of participating households in the RSPB Big Garden Birdwatch.

Table 4.3: Split by Supergroup, the estimated population, number of participating households, and the participation percentage for each Supergroup, for the City of Salford (RSPB, 2019).

| Supergroup | Population | No. of participating households | Participating percentage |
|---|---|---|---|
| Rural Residents | 440 | 4 | 0.91 |
| Cosmopolitans | 17,269 | 5 | 0.03 |
| Ethnicity Central | 11,524 | 4 | 0.03 |
| Multicultural Metropolitans | 27,095 | 24 | 0.09 |
| Urbanites | 40,539 | 99 | 0.24 |
| Suburbanites | 37,753 | 158 | 0.42 |
| Constrained City Dwellers | 40,705 | 40 | 0.10 |
| Hard-Pressed Living | 58,608 | 83 | 0.14 |

Figure 4.2 illustrates the spread of participants of the RSPB Big Garden Birdwatch across the City of Salford. In total 417 households participated, with these distributed randomly across the city. However, Figure 4.2 does show there to be fewer participating households closer to Manchester city centre, to the East of Figure 4.2. Dividing these participants by Supergroup shows significant differences (Table 4.3). Rural Residents had the greatest participating percentage at 0.91 per cent, although very few residents of the City of Salford occupy this Supergroup. Rural Residents are the category most likely to occupy jobs within the agriculture, forestry, and fishing industries, which may make them more inclined to engage within their gardens (ONS, 2015). There is also a greater proportion of older residents, who have been shown to have higher rates of gardening engagement (Chapter 6). Suburbanites had the next highest percentage at 0.42 per cent as well as the highest number of participants, despite only ranking 4$^{th}$ for highest population. Suburbanites are again likely to be older residents, either of retirement age or middle-aged parents. However, residents have more desk-orientated jobs such as finance and administration (ONS, 2015). As discussed in Section 2.7.1, residents attitudes to their gardens post-pandemic have changed; people are more appreciative and choose to invest more time in their private greenspaces. The RSPB Birdwatch also indicated this trend, in 2019 there were approximately 500,000 people partaking across the UK but in 2022 this rose to approximately 700,000 people.

Using this survey as a proxy for overall gardening engagement highlights the increase in residents care and attention in their gardens from pre- to post-pandemic, assuming those partaking in the survey are likely encouraging wildlife into their gardens. Four Supergroups, Cosmopolitans, Ethnicity Central, Multicultural Metropolitans, and Constrained City Dwellers were identified as having the lowest participation, all ≤ 0.1 per cent.

The RSPB Garden birdwatch is arguably a niche activity, not undertaken by all who would engage in wildlife friendly gardening. Therefore, the percentage participation shown in Table 4.3 was always below one per cent of the population with Supergroups in the City of Salford.

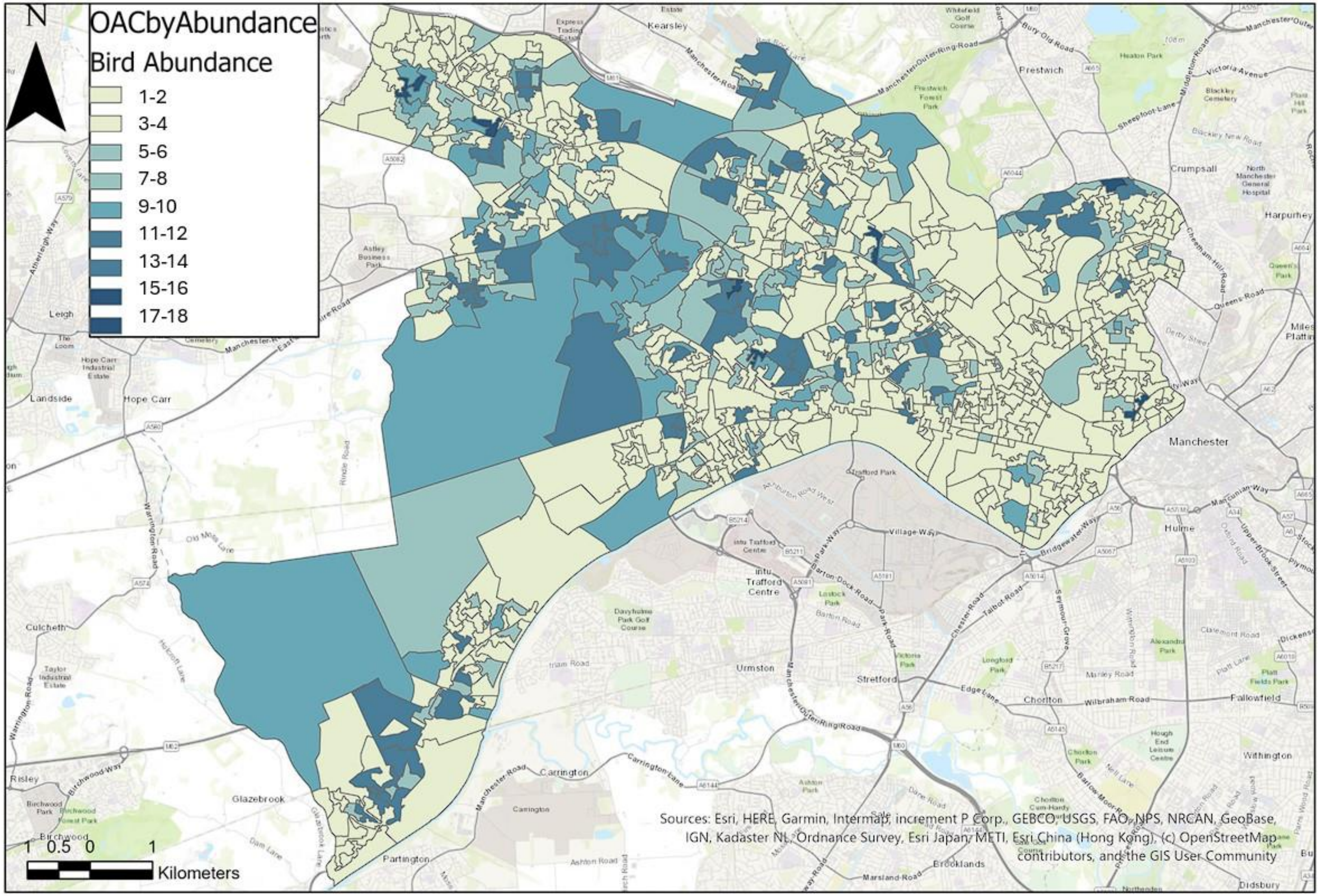



Figure 4.3: The city of Salford, split into Output areas, shaded with the total bird abundance recorded during the RSPB Big Garden Birdwatch.

Table 4.4: The median bird abundance split by Supergroup in the City of Salford (RSPB, 2019).

| Supergroup | Median bird abundance | Range |
|---|---:|---:|
| Rural Residents | 9 | 8 – 18 |
| Cosmopolitans | 9 | 7 – 15 |
| Ethnicity Central | 9 | 5 – 11 |
| Multicultural Metropolitans | 8 | 2 – 13 |
| Urbanites | 8 | 1 – 18 |
| Suburbanites | 8 | 1 – 19 |
| Constrained City Dwellers | 7 | 2 – 14 |
| Hard-Pressed Living | 8 | 1 – 15 |

As has been shown above, participation in the survey was skewed with Rural Residents and Suburbanites having the greatest participation percentage (Table 4.3). This led to the question of whether the data on bird abundance was also skewed. Similarly, to participation (Figure 4.2), there was a lower total bird abundance recorded closest to the centre of the City of Manchester, to the East of Figure 4.3. Excluding this highly urbanised centre, abundance was randomly distributed across the City of Salford.

Table 4.4 shows the median bird abundance by Supergroup, as the data show a non-normal distribution this central tendency was chosen. Median bird abundance was greatest in Rural Residents, Cosmopolitans, and Ethnicity Central at 9 and lowest in Constrained City Dwellers at 7. The ranges in Table 4.4 show a large variation in bird abundance within each Supergroup, with cross-over between each category. This is likely due to the small sample size, as shown in Table 4.3 the three Supergroups with the highest median bird abundance had ≤ 5 participants. Constrained City Dwellers had the second lowest mean feature abundance 2.52 (Figure 4.1) and the lowest median bird abundance 7 (Table 4.4) suggesting a possible trend of less diverse gardens hosting fewer birds. A similar trend has been shown within the literature with increased vegetation diversity resulting in higher bird abundance (Mayorga, Bichier, Philpott, 2020; Parker, Yom-Tov, Alon-Mozes, Barnea, 2014; van Heezik et al., 2013), but a comparison between bird abundance and feature diversity within a garden is lacking. Further, bird abundance has been shown to improve residents health and wellbeing in England (Cox et al., 2017; Fuller et al., 2007). Therefore, the potential relationship between bird abundance and garden feature abundance exposed in this thesis may be worth further exploration.

## 4.4 Summary

This chapter contains data and analysis to investigate objective one of this thesis – to compare the size and diversity of OAC Supergroups household gardens in the City of Salford. Median garden size was found to range from 20 – 176 $m^2$ in the seven Supergroups analysed in the City of Salford (Table 4.2). Previously, mean garden size was reported between 151 – 197.5 $m^2$ (Davies et al., 2009; Gaston et al., 2005; Loram et al., 2008). Whilst mean is not the correct central tendency for the data in this thesis it is used to compare to previous literature and shows Suburbanites (214 $m^2$) and Urbanites (215 $m^2$) had mean garden sizes greater than the averages previously reported. These two Supergroups with the largest gardens also had the highest mean feature abundance, Suburbanites 3.32 features per garden and Urbanites 2.84 features per garden, and the largest combined percentage vegetation landcover, Suburbanites 21 per cent and Urbanites 18 per cent.

Chapter Four has highlighted the significant differences between gardens of some Supergroups; as shown above residents in Suburbanites and Urbanites are able to access larger, more diverse gardens which will benefit their health and wellbeing through their diverse green views (Chalmin-Pui et al., 2021; Hartig et al., 2003; Kaplan, 1995; Pretty et al., 2005; Twedt et al., 2016; Ulrich et al., 1991). Multicultural metropolitans and Constrained City Dwellers are arguably the two most at risk of environmental injustice within their gardens as they are smaller and less diverse than other Supergroups. Comparing Supergroups in this way gives a unique perspective on injustice within private UGI; it clearly shows a disparity which needs addressing and gives stakeholders evidence for what communities to target for interventions, discussed further in Section 7.2.

# Chapter Five: The extent, content, and form of urban household gardens.

## 5.1 Introduction

Throughout Chapter Five, the physical factors that affect urban household gardens are analysed, also termed the urban form. Urban form is ‘the physical characteristics that make up built-up areas, including the shape, size, density, and configuration of settlements’ (Williams, 2014, p. 6). Collective garden landcover within an urban environment in the UK has been estimated at between 22 and 30 per cent (Bell et al., 2020; Gaston et al., 2005; Loram et al., 2007; Tratalos, Fuller, Warren, Davies & Gaston, 2007), however, these estimations are dated and there has yet to be a comparison of garden cover within neighbouring districts. Section 5.2 of this chapter addresses this research gap and objective two of this thesis: to determine whether household gardens as a collective resource are equally distributed across districts of Greater Manchester. Then, using the garden sample data as outlined in Sections 3.7 and 3.8, an overview of a typical garden in the City of Salford (Section 5.3.1) and of all gardens surveyed (Section 5.3.2) is presented (n = 6881). Followed by an exploration of the affect garden size has on content (Section 5.3.3). This research fills the need outlined in Section 2.4 for a contemporary account of garden size and content. Finally, Section 5.4 investigates urban form parameters to address objective three of this thesis: to assess the change in garden size and diversity in housing developments over time in the City of Salford (Section 5.4).

## 5.2 Garden extent

Landcover data were taken from the OS Mastermap Greenspace Layer (Ordnance Survey, 2017) for the North West, downloaded from the Edina Digimap service, see Section 3.10 for a full methodology. Table 5.1 shows the total area of each, the total area of household gardens as a collective UGI, and the garden cover as a percentage. Districts in Table 5.1 are listed in order of descending percentage garden cover.

Table 5.1: Greater Manchester household garden landcover.

| District | Total area ($km^2$) | Household garden ($km^2$) | Garden cover (%) |
|---|---|---|---|
| Stockport | 126.06 | 26.57 | 21.07 |
| Manchester | 115.65 | 23.49 | 20.31 |
| Trafford | 106.04 | 20.52 | 19.35 |
| Salford | 97.19 | 14.47 | 14.88 |
| Tameside | 103.17 | 14.44 | 14.00 |
| Bolton | 139.80 | 18.96 | 13.56 |
| Bury | 99.48 | 13.35 | 13.42 |
| Wigan | 188.19 | 22.20 | 11.80 |
| Oldham | 142.36 | 13.30 | 9.34 |
| Rochdale | 158.08 | 14.63 | 9.25 |
| Greater Manchester | 1276.03 | 181.93 | 14.26 |

Within Greater Manchester the percentage of garden cover ranges from 9.25 per cent in Rochdale to 21.07 per cent in Stockport. Previous research by Cavan et al. (2018) found the collective size of household gardens in the City of Manchester to be 24 $km^2$, a similar result to that reported in Table 5.1 (23.49 $km^2$). Greater Manchester reported similar figures to those previously recorded within the literature (Gaston et al, 2005; Loram et al., 2008), indicating it is representative of other UK cities. The median garden cover across all ten regions of Greater Manchester is 14 per cent. In Salford, garden cover is 14.88 per cent; it is therefore a region which is highly representative of Greater Manchester as a whole, in addition to its varied landscape as outlined in Section 3.3. At a district level the data in Table 5.1 highlight an interesting pattern, the two least disadvantaged districts (determined in Figure 3.2), Trafford and Stockport, are within the top three for garden cover. Yet, two of the most disadvantaged districts (determined in Figure 3.2), Rochdale and Oldham are within the bottom three for garden cover. This exposes a potential luxury effect between the districts of Greater Manchester, with deprivation here being defined as the districts with the highest percentage of LSOAs in decile 1 of the IMD (Greater Manchester Poverty Action, n.d.).

## 5.3 Household garden sample

The previous chapter focused on the differences in urban household gardens through the lens of the residents, using socio-economic status to compare size and content and expose injustices. As is shown in Chapter Two, gardens have also been shown to vary in size and content due to their urban form. Therefore, this section uses the same house garden sample data as Chapter Four (see Sections 3.6-3.7 for a full methodology); but compares differences caused by the physical characteristics of size, property type, development age and density for the City of Salford.

### 5.3.1 An overview of the data: a typical garden

This section uses all 6881 gardens sampled in the household garden sample to create a picture of a typical garden. This large sample size was used to capture the heterogeneity of gardens, giving a more representative account than previous research (see Section 2.4) by accounting for both breadth and depth in the data. It is unique in its approach as previous research has exclusively had a wildlife engagement focus, not a landcover and deprivation one as in this thesis, thus the data captured fill this research gap. Total garden area in the sample ranged from 9m² – 1725 m².

Table 5.2: The mean and median garden features plus the range across all gardens sampled, rounded to the nearest whole number.

| | No. mature trees | No. shrubs | Area of trees + shrubs (m²) | Area of plants (m²) | Area of lawn (m²) | Area of pave (m²) | Total garden area (m²) | Permeable (%) |
|---|---|---|---|---|---|---|---|---|
| **Median** | 0 | 1 | 7 | 0 | 48 | 51 | 129 | 50 |
| **Range** | 0 – 24 | 0 – 36 | 0 – 696 | 0 – 195 | 0 – 862 | 0 – 493 | 9 – 1725 | 0 – 100 |
| **Mean** | 1 | 4 | 18 | 5 | 56 | 61 | 139 | 47 |

Table 5.2 gives the median number of key features, trees and shrubs, the median area of major landcover classes (defined in Sections 3.7 and 3.8), the median total garden area, and the median percentage of permeable land within a typical garden in Salford. The mean has

also been included for comparison to the literature, but the data are not normally distributed and so median is the correct central tendency for analysis. There was a large range in garden size within the sample, 9m² – 1725 m², giving a median of 129 m² and a mean of 139 m². This garden area was smaller than previously reported in the literature. Gaston et al. (2005) recorded a mean area of 151 m² in gardens across Sheffield, variation around the mean was not given, though the range of gardens in Sheffield was reported at 0 m² – 1073 m² (n = 250), smaller than in Salford. Davies et al. (2009) reported a mean of 190 m² from seven urban and suburban areas across the UK with a CI 95 per cent of 173.0 m² – 207.8 m², confidence interval cannot be reported for Table 5.2 as distribution is non-normal. Loram et al. (2008) reported a mean of 197.5 m² (standard error ± 11.4) across five UK cities.

When compared to Greater Manchester, the City of Salford as a region had a collective percentage garden cover larger than six of ten other GM regions (Table 5.1). The large range in garden size shown in Table 5.2 is explored further in Section 5.3.2. Three landcovers in Table 5.2 also had significantly large ranges: trees & shrubs 0 – 696 m² with approximately only 10 per cent of gardens ≥50 m²; lawn 0 - 862 m² with approximately only 18 per cent ≥ 100 m²; and pave 0 – 493 with approximately only 15 per cent ≥ 100 m².

Impermeable land – the area of pave – had a median of 51 m² and a mean extent of 61 m² in the City of Salford (Table 5.2). Comparing to the literature, the sum of all the impermeable land cover in Loram et al. (2008) is 86.8 m² (patio, sheds, gravel, decking, greenhouse, and water butt). For the same study, combining just patio (38.6m²) and path (17.1m²) gives a total of 55.7 m² (Loram et al., 2008), a similar total to that of this thesis. Yet the mean total garden area of the City of Salford 139 m² was smaller than that in Loram et al. (2008) 198 m² suggesting the City of Salford has more pave cover. The same study by Loram et al., (2008) reported a mean area of 85.5 m² for mown grass in gardens across five UK cities; yet the data in Table 5.2 shows a mean lawn area of 56 m². The median area for lawn and pave cover in Table 5.2 show little variation between the two land cover categories. But Loram et al. (2008) found the mean area of lawn to be considerably larger than that of pave and patio combined. The garden composition for urban gardens in the City of Salford therefore shows a greater amount of pave and a lesser amount of lawn cover than previous literature. As has been shown above total garden size varies considerably, thus the size of different landcovers is expected to be varied. Missing from previous literature is a comprehensive account of garden composition using percentage cover as shown below.

Figure 5.1 illustrates the typical garden landcover using percentage cover. This is the first complete account of urban garden landcover presented as percentage cover within the literature. Trees and shrubs were grouped together when recording percentage cover due to inability to confidently distinguish between canopies (discussed further in Sections 3.7 and 3.8). Figure 5.1 shows vegetation (plants, trees & shrubs combined) only accounts for 16 per cent of a gardens cover. Vegetation rich environments have been shown to have stress reducing effects (Hartig et al., 2003) and to reduce mental fatigue (Moran, 2019). Increasing vegetation cover in front gardens has been shown to increase residents self-reported wellbeing score by 26 per cent (Chalmin-Pui et al., 2021). Therefore, the City of Salford gardens are underperforming, and residents would receive greater benefits from increasing their vegetation cover and diversity.

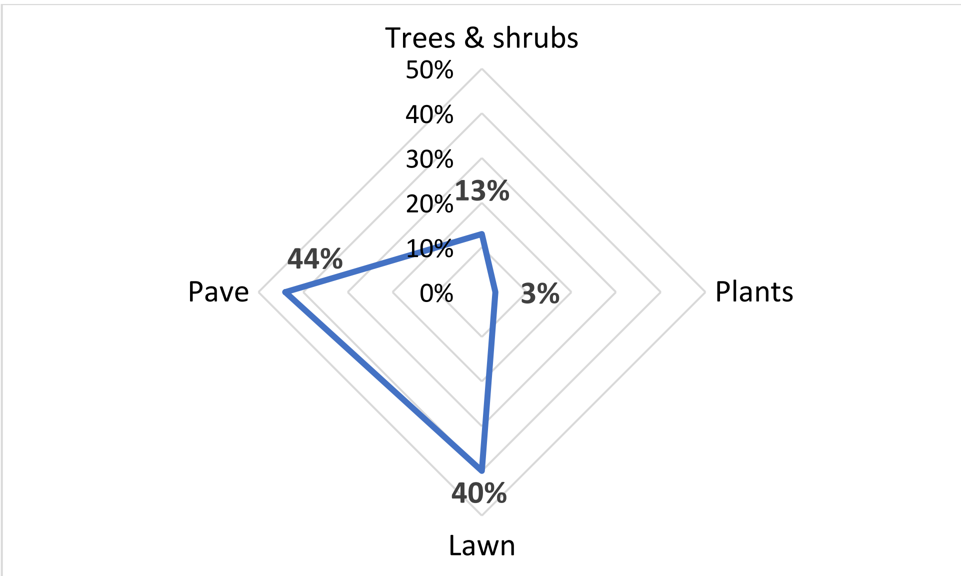


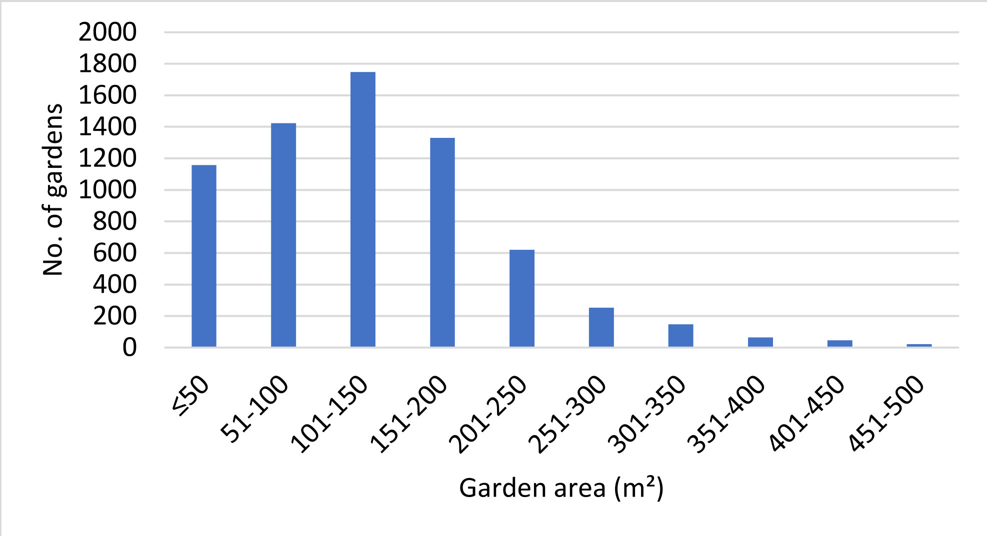

Figure 5.1: The mean percentage cover of each of the four categories across all gardens sampled.

Within the literature some aspects of urban garden landcover have been published, but a comprehensive account is missing. Davies et al., (2009) found, on average, 11 per cent tree cover within a garden, but did not report on other landcover features. Figure 5.1 reports a similar land cover for trees and shrubs combined. Gaston et al., (2005) reported an estimate of 60 per cent of the total area of gardens in Sheffield is lawn; yet Figure 5.1 shows only 40 per cent land cover to be lawn, similarly to above lawn cover has likely been overestimated in previous studies. Permeable land accounts for 56 per cent of the total garden cover in Figure 5.1, a similar green landcover to that described by Baker, Smith, and Cavan, (2018) and Baker and Smith, (2019). Loram et al. (2008) reported the mean area of several land cover features (Table 2.2), but other than mown grass vegetation data were not reported, and trees were recorded as count data. So, a comprehensive account of land cover is not possible.

### 5.3.2 An overview of the data: all gardens

The following data present an overview of all 6881 gardens sampled. First, the range in garden size is discussed. In the previous Section 4.3 garden area was discussed to compare differences in garden size between socio-economic groups, in the subsequent Section 5.3.2 garden area is presented from an urban form perspective and not split by residents characteristics. Then, the occurrence of each of the five features. Thirdly, the relationship between garden size and key environmental parameters is explored. Finally, results have been scaled up to provide data on gardens across the City of Salford and Greater Manchester.

A)




B)

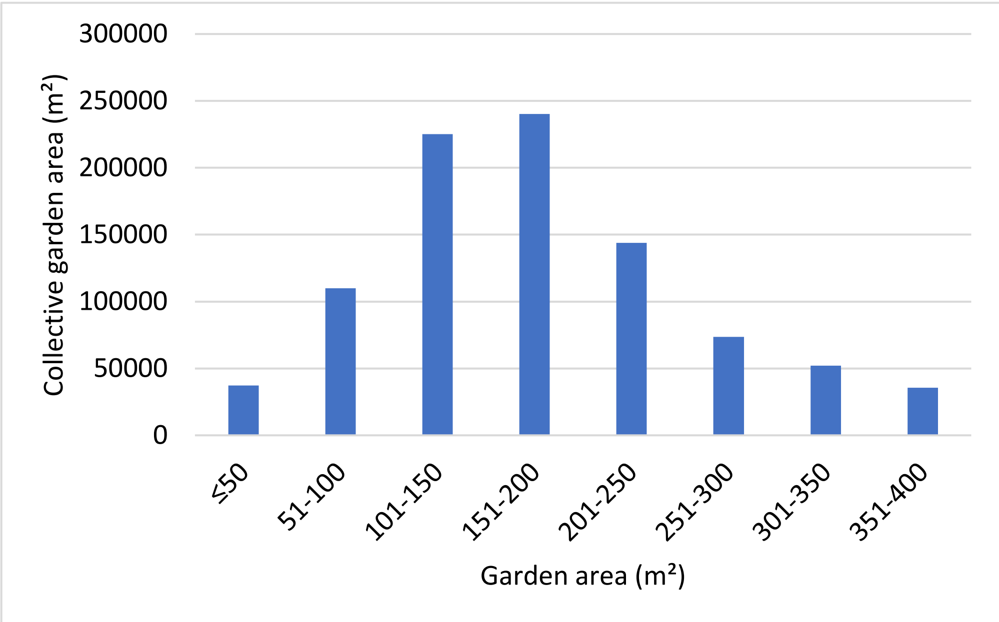



Figure 5.2: A) household garden size divided into groups up to 500 m². B) the total area covered by gardens within each group up to 500 m².

The range of household garden size was 9 m² – 1725 m². Figure 5.2 shows the spread of gardens up to 500 m² which includes 6814 gardens in total (n = 6881). Garden size was included up to 500 m² as above this the number of gardens within each group was minimal. Loram et al. (2007) also found smaller gardens (≤400 m²) contributed disproportionately to the total garden area of each city, being more numerous than larger gardens in four of the five UK cities surveyed. Interestingly, the city most comparable in size to Salford in the literature, Belfast, has gardens more uniformly spread, with those <50 m² occurring more frequently than the other five cities. In Belfast gardens of >800 m² contributed a greater proportion of total garden area than in any other city, suggesting a large range in garden size similar to that in Salford. The greatest number of gardens in the City of Salford were within the 101 – 150 m² group; however, the largest cumulative area was 151 – 200 m². Loram et al. (2007) found that gardens 200 – 400 m² contributed the greatest proportion of total garden area in each of the five cities. Brindley et al. (2018) found garden size to have a positive impact on resident's health on a national scale, suggesting the more frequent smaller gardens in the City of Salford may be impacting health at a community level.

Table 5.3: The percentage occurrence of the five landcover categories across all gardens sampled (n = 6881).

| | **Trees** | **Shrubs** | **Plants** | **Lawn** | **Pave** |
|---|---|---|---|---|---|
| **Occurrence (% of gardens)** | 22.0 | 50.1 | 22.7 | 72.9 | 99.6 |

The values in Table 5.3 show the percentage of gardens within which the feature occurred. For example, trees occurred in 22.0 per cent of gardens. Previous research has recorded the occurrence of trees to be much greater; Gaston et al. (2005) found trees (>3m) occurred in 48 per cent of gardens in Sheffield, and in Davies et al. (2009) 54 per cent of gardens contained at least one tree (>3 m) across the UK. Shrubs occurred in approximately half of all gardens and lawn in approximately 75 per cent. Pave was recorded in almost 100 per cent of the gardens sampled, only 24 of 6881 did not have any pave present. Loram, Warren, and Gaston, (2008) recorded patio (either concrete, paving stone, or bricks) in 86 per cent of gardens across the five UK cities. This very high prevalence of impermeable pave, coupled with the increase in popularity of artificial lawn (Francis 2018; Brooks & Francis, 2019), is of concern to the ecosystem services community as a potential flood risk, particularly in catchment areas like the City of Salford (Smith et al., 2011; Warhurst, Parks, McCullock, & Hudson, 2014). The need to monitor this trend is discussed further in section 7.2.

Table 5.4: Scaled up findings for the City of Salford and Greater Manchester based off the number of dwellings (excluding flats) with a garden.

| Region | No. of dwellings with a garden | Total area (km²) | No. mature trees | No. shrubs | Area of trees + shrubs (km²) | Area of plants (km²) | Area of lawn (km²) | Area of pave (km²) | Total garden area (km²) |
|---|---|---|---|---|---|---|---|---|---|
| **Salford** | 84,133 | 97 | 84,133 | 336,532 | 1.7 | 0.4 | 5.2 | 5.3 | 13 |
| **Greater Manchester** | 862,417 | 1276 | 862,417 | 3,449,668 | 17.2 | 4.3 | 53.5 | 54.3 | 130 |

Data in Table 5.4 are calculated using the mean values in Table 5.2 from all gardens sampled. Total garden area for Greater Manchester has been calculated at 130 km² in Table 5.4 and 182 km² in Table 5.1; this difference is due to the different methods used. Table 5.1 measures the actual garden cover, whereas Table 5.4 is estimated using the mean garden area and multiplied by the number of dwellings with a garden. Therefore, the data in Table 5.1 are a more accurate measure, however the comparison illustrates figures presented in Table 5.4 are likely to be the most conservative values for collective garden content.

It is estimated that there were 11.3 million trees across Greater Manchester in 2018, this includes trees found in a variety of habitats from woodlands to along the roadside (City of Trees, 2018). The data in Table 5.4 suggests approximately 8 per cent of these are in household gardens.

### 5.3.3 Variation caused by garden size

The following section uses data from all 6881 gardens surveyed as outlined in Sections 3.7 and 3.8 and examines variation from an urban form perspective. The data are collectively analysing gardens from all socio-economic Supergroups, whereas where garden features are discussed above (Section 4.3) they are comparing differences in gardens between socio-economic groups. It is expected increasing garden size would result in an increase in feature abundance and vegetation abundance, as it is logical to presume more space would result in a greater number of features (Loram et al., 2008). The following graphs test this relationship in the Salford dataset. Garden size has been transformed to a logarithmic scale due to the large variation in the dataset, (range 9 m² – 1725 m²), and to illustrate the relationships more clearly.

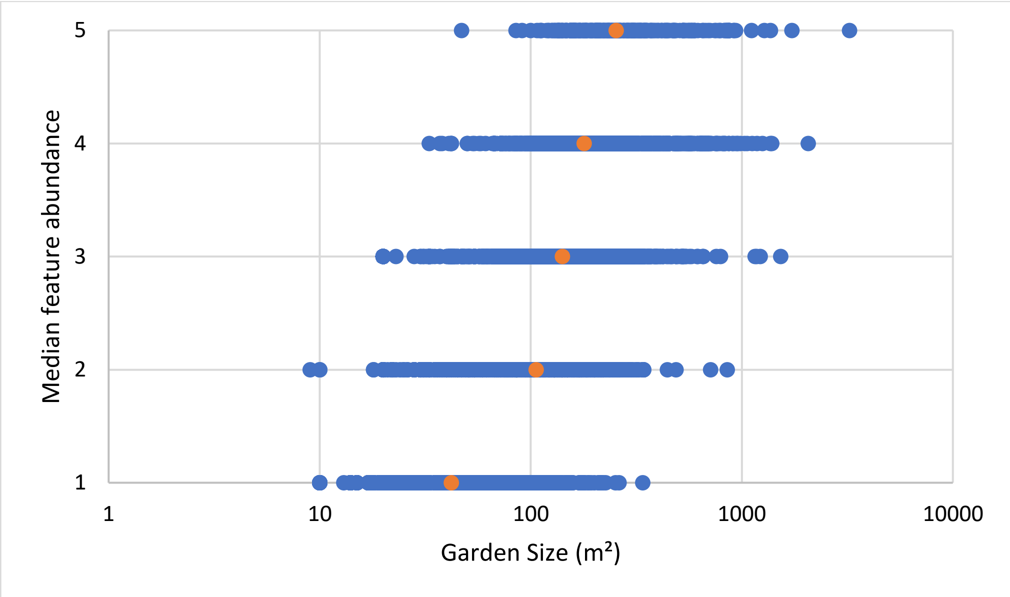
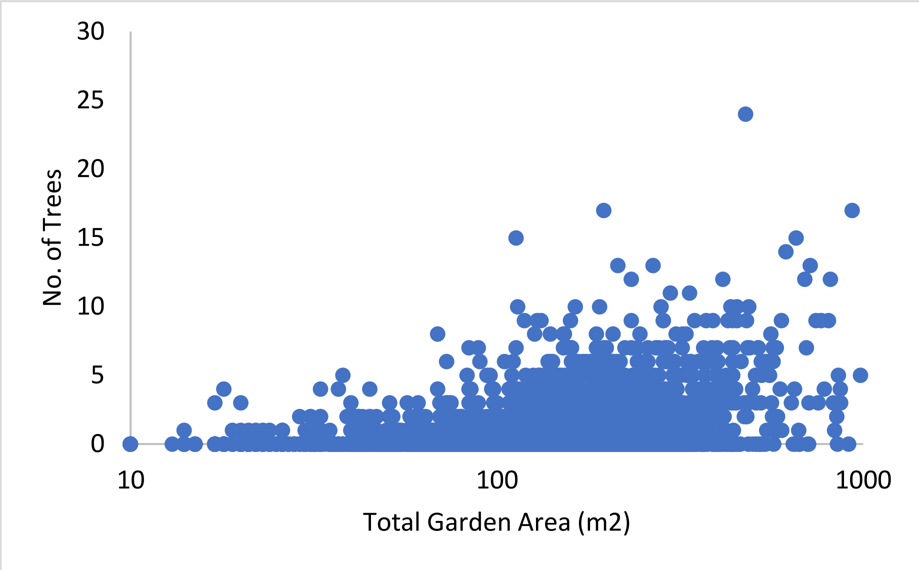



Figure 5.3: Total garden size (m²) and median feature abundance across all gardens. Orange points show the median garden size for each feature abundance category. Spearman's rank correlation $r_s = 0.59$, $p = <0.05$.

Feature abundance is the presence/absence data for the five key features outlined in Section 3.8: trees, shrubs, plants, lawn, and pave. There is a positive linear correlation between garden size and feature abundance when combining all gardens ($r_s = 0.59$ $p = \leq 0.05$), meaning as garden size increases the mean feature abundance increases. A logarithmic scale has been used on the x-axis due to the data spanning numerous orders of magnitude, from 9 – 1725 m². Median values for each feature abundance category were: 1 = 42 m², 2 = 106 m², 3 = 142 m², 4 = 179 m², 5 = 254 m². Loram et al. (2008) reported a similar strong, positive, linear correlation between garden size and the land use richness in three of the five cities. Land use richness in this study was out of 12 features (see Table 2.2 for a list of features). Belfast ($r_s = 0.66$, df = 53, $p = <0.0001$), Cardiff ($r_s = 0.61$, df = 52, $p = 0.0001$) and Leicester ($r_s = 0.73$, df = 47, $p = 0.0001$) all reported a similar relationship to that in Figure 5.3, as garden size increases feature abundance also increases.

A)




B)

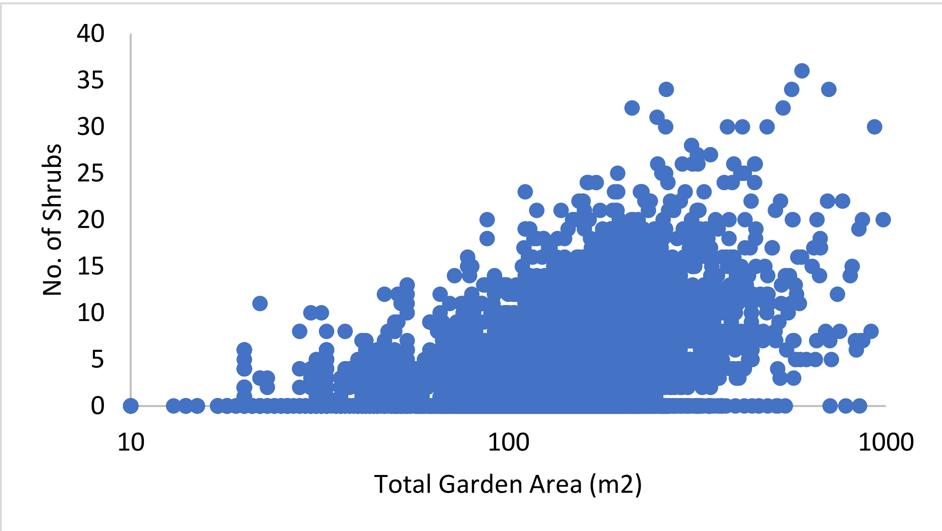
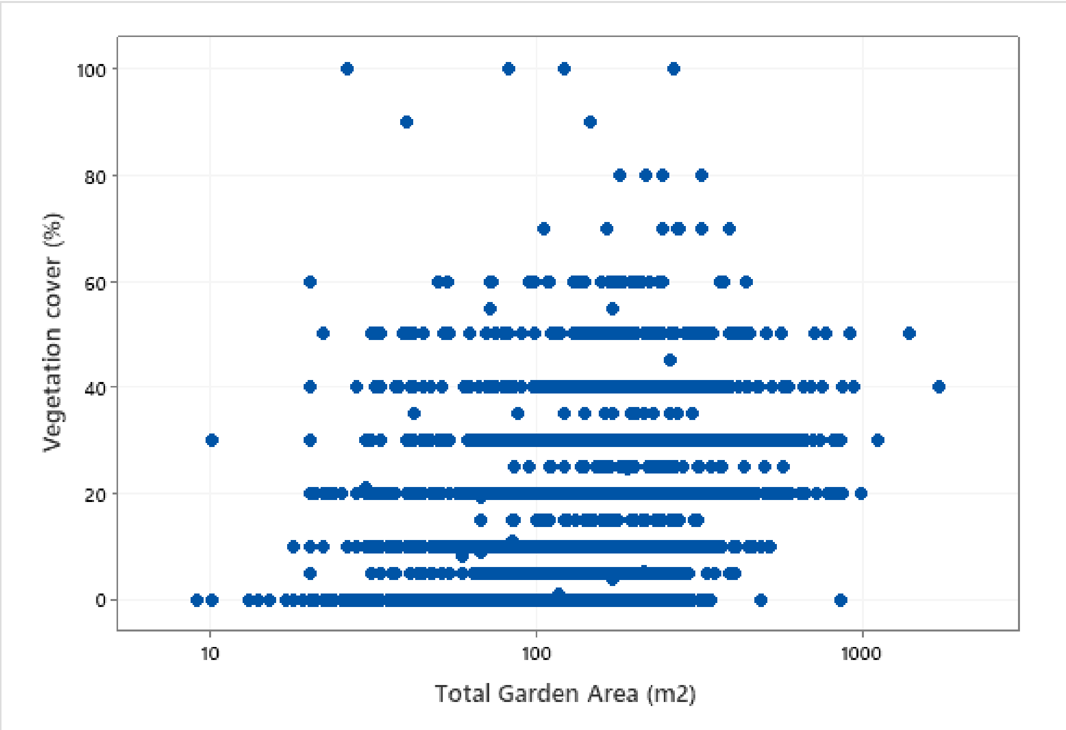



Figure 5.4: The number of mature trees (A) and number of shrubs (B) with increasing garden size ($m^2$) $\log_{10}$. Spearman's Rank mature trees $r_s = 0.35$ $p = \leq 0.05$, shrubs $r_s = 0.45$ $p = \leq 0.05$.

Figure 5.4 shows increasing garden size results in an increase in the abundance of both mature trees and shrubs, with the relationship stronger in larger gardens (>100 $m^2$) While both result in a positive, linear correlation, mature trees ($r_s = 0.35$ $p = \leq 0.05$) had a weaker relationship than shrubs ($r_s = 0.45$ $p = \leq 0.05$), though both correlations are weak. An outlier at 80 shrubs and a garden area of 266 $m^2$ was removed from Figure 5.4 and the analysis, though this did not affect the overall correlation coefficient but skewed Figure 5.4. The literature also illustrates this increase in trees with larger gardens. Loram et al. (2008) found the number of trees increased with increasing garden size across all five canopy classes, <0.5 m ($r_s$ = 0.32), 0.5-1 m ($r_s$ = 0.42), 1-2 m ($r_s$ = 0.42), 2-3 m ($r_s$ = 0.50), >3 m ($r_s$ = 0.51), all p values = ≤0.05, with the relationship getting stronger in taller trees. However, what has not been previously reported is the effect on shrub-like vegetation with garden size. As Figure 5.4 suggests this has been shown to follow a similar relationship. Whilst the correlation in Figure 5.4 and reported by Loram et al. (2008) were weak, it does illustrate the number of trees and shrubs has been shown to increase with garden size in two separate studies.

A) Vegetation cover



B) Green cover

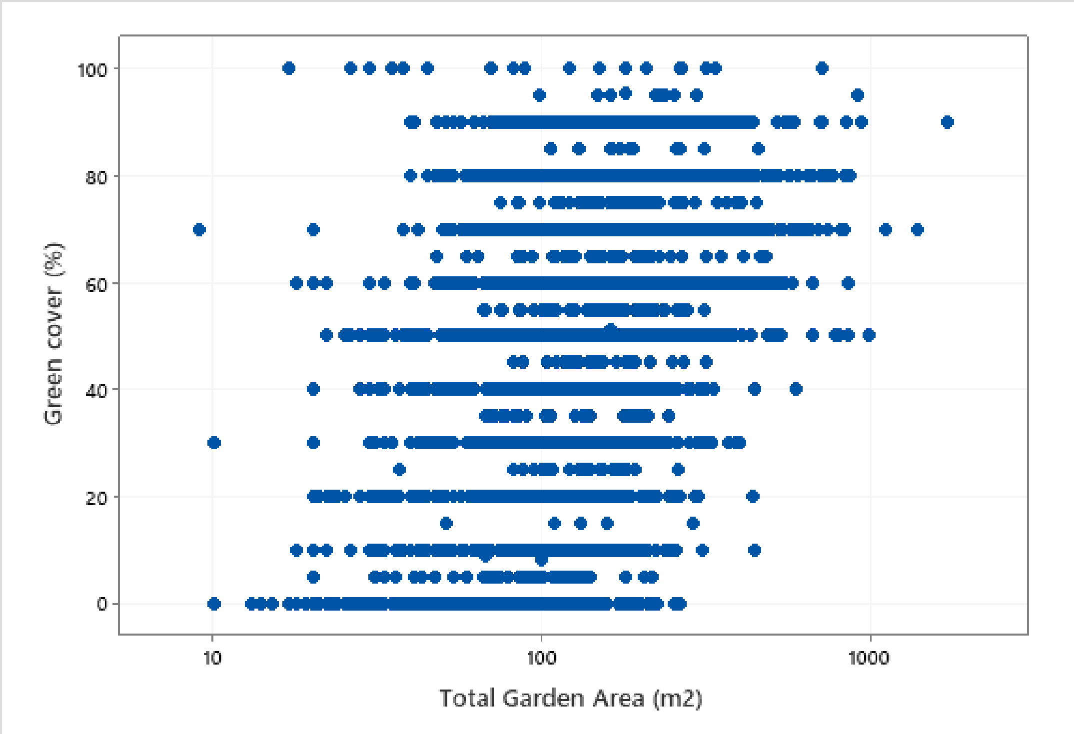



Figure 5.5: Percentage vegetation cover (trees, shrubs & plants) (A) and percentage green cover (trees, shrubs, plants & lawn) (B) with increasing garden size (m²) $\log_{10}$. Spearman's Rank veg. cover $r_s = 0.41$ p = ≤ 0.05, green cover $r_s = 0.54$ p = ≤0.05.

Green cover (trees, shrubs, plants, and lawn) and vegetation cover (excluding lawn) were graphed separately as suggested by Loram, Warran, and Gaston, (2008). Both parameters showed a positive linear relationship with garden area, as garden size increases so does the percentage of green and vegetation cover, though the correlations were weak. Loram, Warran, and Gaston, (2008) found the proportion of mown lawn to have a much weaker positive relationship with garden area ($r_s = 0.19$ p = ≤0.05), whilst other features relating to vegetation (cultivated boarder, uncultivated, vegetable patch, and unmown grass) were all found to have non-significant relationships.

Overall, household gardens in the City of Salford were smaller (median of 129 m²) than previous research suggested (Gaston et al., 2005; Loram et al., 2008; Davies et al., 2009). Gardens within the City of Salford had a larger area of paved surfaces and a smaller area of lawn, with little difference between the two land cover types opposing previous research by Loram et al. (2008). The larger area of paved land cover found in the City of Salford is of greater flooding concern as the city has a flat topography, sitting in the catchment of both the Irwell and Glaze Brook rivers (Salford City Council, 2008), an issue discussed further in Section 7.2. The research presented in Section 5.3 filled a gap in the literature for a comprehensive account of percentage landcover, and in doing so highlighted the lack of vegetation cover in Salford gardens (16 per cent cover) which will likely restrict health and wellbeing benefits to residents (Chalmin-Pui et al., 2021).

Collectively, garden size varied greatly; with gardens <500 $m^2$ contributing disproportionately to the overall size of gardens. Trees occurred less frequently in gardens in Salford than previous research suggested; only 22 per cent contained at least one tree in comparison to 54 per cent (Davies et al., 2009) and 48 per cent (Gaston et al., 2005) previously recorded. Scaling up these findings to Greater Manchester, it suggests 8 per cent of the city's trees are found in household gardens.

Garden size was found to have the expected relationship with garden content, that is, the larger the garden the more diverse the content. A linear relationship with feature abundance, like that recorded by Loram et al. (2008), highlighted garden size as a possible limiting factor in the benefits residents can receive from their gardens. A similar relationship was found when increasing garden size with the number of trees, number of shrubs, percentage green cover, and percentage vegetation cover. This relationship had not before been recorded in the abundance of shrubs, and vegetation cover was found to be stronger in the City of Salford than previous studies (Loram et al., 2008). with shrubs not previously being recorded in the literature.

## 5.4 Urban form parameters

### 5.4.1 House type

From the literature, urban form has been shown to affect household garden form and function (section 2.5). In the UK, previous research has shown garden size and content to vary between different housing types: terraced, semi-detached, and detached (Loram et al., 2007; Loram et al., 2008). However, house age has not previously been explored as a factor for varying garden size and content. Development age, property type, and housing density were all considered separately in analysis as sample sizes when considering them collectively were too uneven for reliable results. For example, the development age with the smallest sample had 794 houses, but when combining age and density the smallest sample was 53. This relationship is explored below using the same household garden sample data (see Sections 3.7-3.8, n = 6881), investigating objective three of this thesis: to assess the change in garden size and diversity in housing developments over time in the City of Salford.

Table 5.5: The mean, median, and range in garden size across all houses sampled (n = 6881), rounded to the nearest whole number.

| **Property Type** | **Mean ($m^2$)** | **Median ($m^2$)** | **Range ($m^2$)** |
|---|---|---|---|
| Detached | 257 | 202 | 56 – 1725 |
| Semi-Detached | 170 | 156 | 34 – 914 |
| Terraced | 83 | 72 | 9 – 852 |

Household garden size in the City of Salford decreases from detached, to semi-detached, to terraced; all three property type groups are significantly different from one another (Mann Whitney U Test p = <0.05). Garden size ranged from 56 – 1725 $m^2$ in detached, 34 – 914 $m^2$ in semi-detached, and 9 – 852 $m^2$ in terraced. Both mean and median central tendencies are reported in Table 5.5, data are non-normal but for comparison mean has also been included. Interestingly, both the mean and median figures in Table 5.5 are smaller than previous publications. Loram et al. (2008) recorded mean garden size across five UK cities (Edinburgh, Leicester, Oxford, Cardiff, and Belfast) at detached 315 $m^2$, semi-detached 188.5 $m^2$, and terraced 93.5 $m^2$ in their small-scale study. In their initial, more large-scale approach, Loram et al. (2007) reported median garden sizes of 84 $m^2$ for terraced, 213.5 $m^2$ for semi-detached, and 364.8 $m^2$ for detached for the same five cities. Compared to these figures, gardens in terraced houses in the City of Salford were similar in size to other cities in the UK, but semi-detached and detached garden sizes were smaller. The housing development trend in the City of Salford appears to be favouring terraced housing, excluding residence of multiple occupancy, much of the recent development has been terraced housing. Salix Homes in conjunction with the council announced a large development of 157 houses, mostly terraced, to be completed in summer 2023 (Salix Homes, 2021). Although on this site the developers specify a public greenspace with 86 trees planted, there is no mention of garden size, which from the pictures of initial development look minimal. As discussed in Section 2.5, the City of Salford does not clearly give guidance on a minimum outdoor space (Salford City Council, 2023), and with gardens having numerous benefits to residents, for example the reduction of chronic stress, it is important that private garden space is not overlooked.

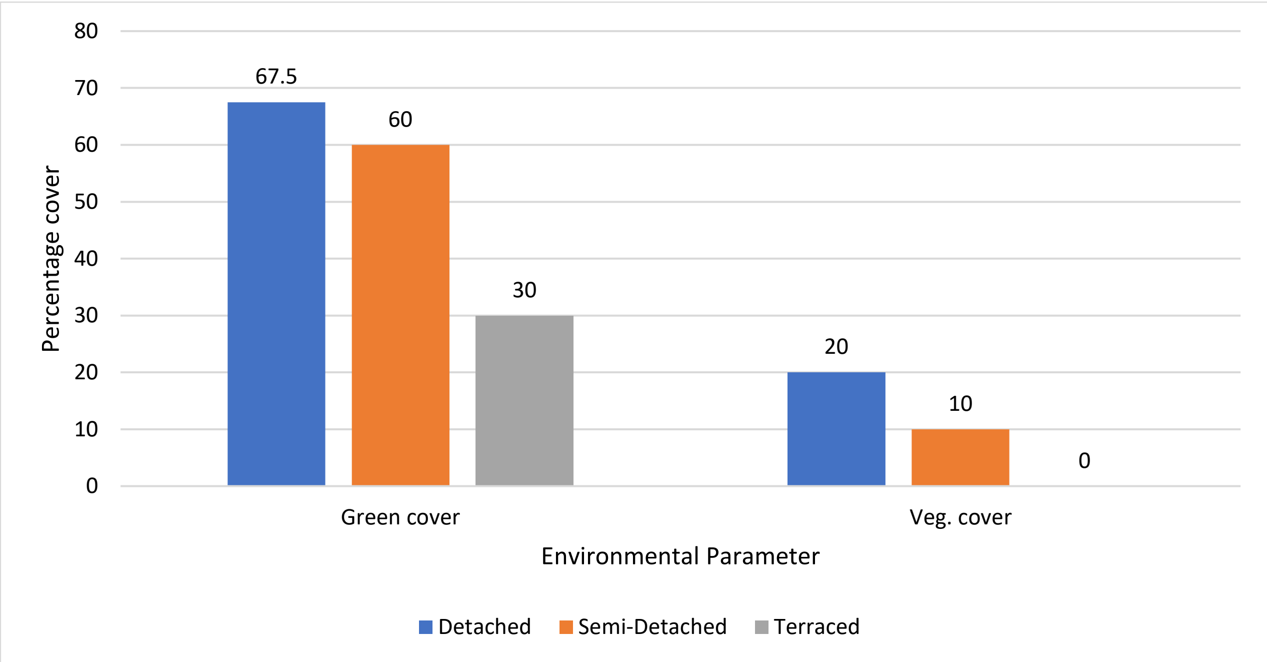
Table 5.6: The number and proportion of each property type across all houses sampled in the City of Salford (n = 6881). Proportions rounded to the nearest whole number.

| Property Type | No. of houses | Proportion of properties (%) |
|---|---|---|
| **Detached** | 582 | 8 |
| **Semi-Detached** | 3272 | 48 |
| **Terraced** | 3027 | 44 |

Across Salford, semi-detached and terraced housing contributed the greatest proportion of houses sampled. Previous research has shown terraced housing to contribute 50 per cent of housing (Loram et al., 2007). Detached houses were the smallest contributing property type in the City of Salford at 8 per cent, this has also been shown across the UK: Belfast 11.7 per cent; Leicester 10.8 per cent; Oxford 11 per cent and Cardiff 13.2 per cent (Loram et al., 2007). The proportion of property type has been shown to have a direct effect on ecological performance; both carbon sequestration ($r_s$ =0.55 p=<0.05), and greenspace cover ($r_s$ =0.29, p=<0.05), increased with increasing proportions of detached or semi-detached housing at Output Area level, independent of housing density (Tratalos et al., 2007). This suggests cities with a high proportion of terraced housing, such as Salford, will have lower ecosystem service capabilities for carbon sequestration and rainwater runoff.

Table 5.7: The median and range for each property type and five environmental parameters (table) with median percentage green cover and median vegetation cover illustrated underneath (bar chart) (n = 6881).

| | **No. of shrubs** | Range | **No. of trees** | Range | **Green cover (%)** | Range | **Veg. cover (%)** | Range | **Feature Abundance** | Range |
|---|---|---|---|---|---|---|---|---|---|---|
| **Detached** | 6 | 0 – 36 | 0 | 0 – 24 | 67.5 | 0 – 100 | 20 | 0 – 80 | 4 | 1 – 5 |
| **Semi-Detached** | 3 | 0 – 34 | 0 | 0 –15 | 60 | 0 –100 | 10 | 0 – 100 | 3 | 1 – 5 |
| **Terraced** | 0 | 0 – 28 | 0 | 0 – 12 | 30 | 0 – 100 | 0 | 0 – 100 | 2 | 1 – 5 |




The median quantity of shrubs and feature abundance both decreased from detached to semi-detached to terraced housing. As shown in Table 5.5 median garden size also decreased between each property type: 202 m², 156 m², 72 m². Median feature diversity and number of shrubs were both shown to have positive linear relationships with garden size (Figure 5.3 and 5.4) suggesting it is the size of the garden affecting these parameters. Similarly, green cover and vegetation cover were shown to have positive linear relationship with garden size (Figure 5.5). Green cover (trees, shrubs, plants, and lawn) and vegetation cover (excluding lawn) were presented separately as suggested by Loram, Warran, and Gaston, (2008). The largest difference between property types was in median green cover, with detached and semi-detached having 67.5 and 60 per cent respectively but terraced only having 30 per cent. The data in Table 5.7 show terraced gardens to contain very little vegetation and feature diversity, suggesting these gardens are the least beneficial to residents. A similar relationship was reported by Loram, Warran, and Gaston (2008), when combining all five cities, terraced housing was found to have significantly lower land-use richness than the other two property types (two-way ANOVA f = 23.7, df = 2,240, p = <0.001).

### 5.4.2 Density and age

The previous section reported on the investigation of property type as an urban form variant on garden size and content. The focus of this section is on differences in garden size and content between housing density (high, medium, and low) and development age (Historic – Modern) (n = 6881). Development age is categorised as: Historic (up to 1914), Interwar (1918-1939), Post-war (1945-1959), Sixties/Seventies (1960-1979), and Modern (1980-current), see Section 3.11 for a full methodology.

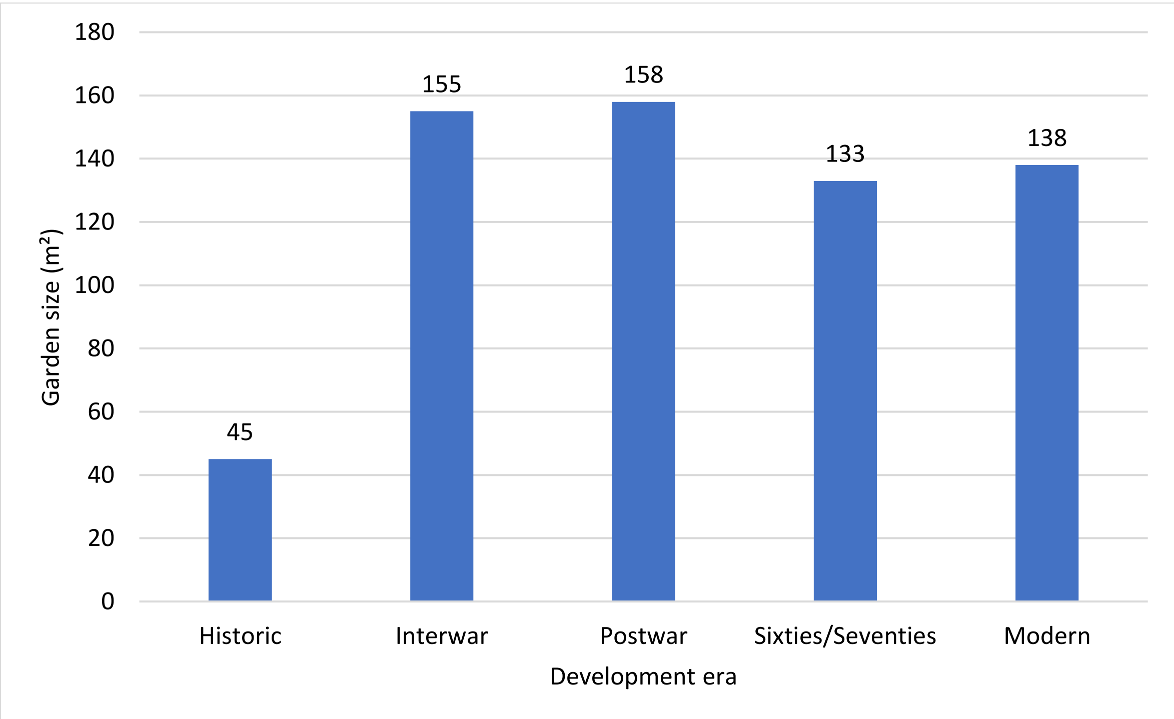



Figure 5.6: Median garden size for all property types and housing density split by development age.

Garden size was found to have a weak positive relationship with development age ($r_s$ = 0.38 p = ≤0.05). Figure 5.6 illustrates garden size increased in developments from Historic 45 $m^2$ to Interwar 155 $m^2$ to Postwar 158 $m^2$. But garden size declined in developments in the Sixties/Seventies 133 $m^2$, with median size increasingly slightly in modern houses 138 $m^2$. UK planning policy revised in 2000 reinforced the need to build at high building density on brownfield sites, strongly advising against new development densities falling below 30 dwellings per hectare (Bibby, Henneberry, & Halleux, 2018). Further, as discussed in Section 2.5 there is no universal standard for garden size in new developments in Salford. Between 2001 – 2011 33,000 dwellings were built in England on land which was previously a residential garden; this activity, known as garden infill, again greatly reduces the amount of garden land which can have disastrous local environmental effects (Bibby et al., 2018). The Ministry of Housing, Communities & Local Government published a National Design Guide in 2021 to showcase good practice for beautiful, enduring, and successful developments nationally. Within the report the need for a garden space is mentioned several times, including giving multiple examples of different ways garden space can be planned into development; yet, there is no tangible recommendation for the size of such spaces (Ministry of Housing, Communities & Local Government, 2021). Similarly, in their report on development plans 2018 – 2037 Salford City Council aim to build an additional 32,680 dwellings, 8,450 being houses (Salford City Council,. n.d.). But, within this report there is no mention of intended residential garden size, or even whether these dwellings will have a private garden space. The current trend of paved front gardens (Smith et al., 2011; Warhurst, Parks, McCullock, & Hudson, 2014) and the evidence given in Figure 5.6 that gardens are smaller in modern developments than those built in the beginning of the twentieth century suggest gardens as an important UGI are threatened.

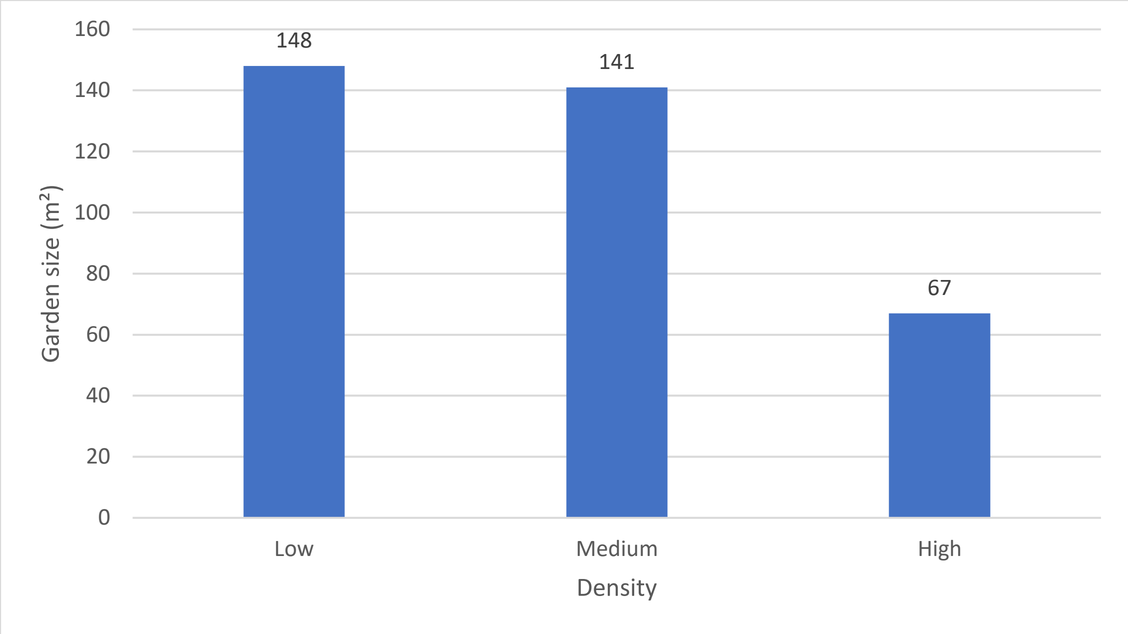



Figure 5.7: Median garden size for all property types and development era split by density.

A Spearman's Rank correlation shows a weak negative relationship ($r_s = -0.38$ p = ≤0.05) between garden size and housing density. The greatest difference in garden size was in high density housing with a median size of 67 m² compared to medium 141 m² and high 148 m². Data in Figure 5.7 indicate a high-density approach, as is championed by UK planning policy (Bibby, Henneberry, & Halleux, 2018), significantly increases the chances of small garden spaces.

Table 5.8: Spearman's rank correlations for density and development age and environmental parameters (vegetation cover, feature abundance, number of trees, and number of shrubs). P = <0.05. Density scale - 1 = Low 3 = High. Era scale – 1 = Historic 5 = Modern.

| | **Vegetation cover** | **Green cover** | **Feature abundance** | **No. of trees** | **No. of shrubs** |
|---|---|---|---|---|---|
| **Density** | -0.13 | -0.31 | -0.27 | -0.11 | -0.17 |
| **Age** | 0.14 | 0.42 | 0.31 | 0.12 | 0.11 |

Table 5.8 suggests, although significant, vegetation cover, number of trees, and number of shrubs only have weak correlations with housing density and age. Loram, Warren & Gaston, (2008) found with increasing house age >3 m tall canopy increased in both area ($r_s = 0.13$, p = <0.03) and proportional contribution ($r_s = 0.12$, p = <0.05). The results data in Table 5.8 suggest the reverse effect on number of trees, with a weak positive corelation between tree

abundance and newer houses ($r_s$ = 0.12). However, the test statistics are weak. Development age had the strongest relationship with green cover and feature abundance $r_s$ = 0.42 and $r_s$ = 0.31 respectively. Similarly, density had the strongest relationship with green cover and feature abundance $r_s$ = -0.31 and $r_s$ = -0.27. Yet, canopy cover was shown to have no significant difference between density level in a city-wide approach in Salford, Manchester, and Trafford (Dennis, Scaletta, James, 2019).

Overall, median garden size was found to increase from terraced 72 m² to semi-detached 156 m², and to detached 202 m² (Table 5.5): a similar relationship to previous publications (Loram et al., 2008). Semi-detached housing is the dominant property type in the City of Salford (48 per cent), closely followed by terraced (44 per cent), whilst detached housing had a proportion of only 8 per cent (n = 6881) (Table 5.6). The only parameter to not decrease with property type was number of trees (Table 5.7), with number of shrubs, vegetation cover, green cover, and feature abundance all having positive linear relationships with property type, being more common in detached than terraced housing. Development age had a weak positive relationship with garden size; median size increased from Historic to Postwar housing but levelled off during the Sixties/Seventies (Figure 5.6). Finally, housing density and garden size also had a weak positive relationship with the most significant difference being between median garden size in highly dense areas (67 m²) compared to that in medium (141 m²) or low (148 m²).

## 5.5 Summary

This chapter set out to investigate the physical factors that affect urban household gardens and how these may influence the size and quality of the private UGI residents can benefit from. To do this, firstly an up-to-date account of the collective extent of gardens was needed. Table 5.1 provides this information for Greater Manchester, showing garden land cover varies between districts from 9 – 21 per cent. These data also highlighted the City of Salford as a representative district for further research as the percentage garden cover was closest to the mean across Greater Manchester.

It was found median garden size across all gardens was 129 m², with 50 per cent permeable surface (Table 5.2). This high proportion of impermeable pave is a contributing factor to localised flooding (Ellis, 2002; Perry & Nawaz, 2008) and illustrates the need for the 2008 law change to require planning permission for the use of impermeable materials of 5 $m^2$ (Environment Agency, 2008). The overall composition of a garden was 44 per cent pave, 40 per cent lawn, and 16 per cent plants, shrubs, and trees (Figure 5.1). Linking back to Chapter Four, when considering Stress Recovery Theory (outlined in Section 2.6) this lack of vegetation-rich landcover is likely limiting the health and wellbeing benefits residents are receiving (Hartig et al., 2003; Moran, 2019); Chalim-Pui et al., (2021) found increasing vegetation cover in front gardens increased self-reported wellbeing by 26 per cent.

Garden size was found to have more than tripled in housing developments from the 1914 to 1958, from a median size of 45 $m^2$ to 158 $m^2$. Median size then decreased to 133 $m^2$ in the 1960s, increasing slightly to 138 $m^2$ since the 1980s onwards. Overall median garden size for the household sample (in 2018) was 129 $m^2$ suggesting garden size may have decreased to below the 1960s average. The data suggest overall garden size has decreased since the 1960s. When considering garden content (Table 5.8) and development age, however, the data are less clear. There was a positive linear relationship between green cover and feature abundance $r_s = 0.42$ and $r_s = 0.31$ respectively, but test statistics were moderate. Baker and Smith (2019) found Victorian terraced housing to have substantially less green cover than other property types (non-Victorian terraced, semi-detached, and detached). But, in this study no other property type considered the development age, only Victorian terraced. Whilst this thesis and Baker and Smith (2019) provide some evidence of garden content being affected by development age, it is not conclusive and further data collection would be needed.

As garden size is currently largely unregulated within the UK, with individual local authorities providing guidelines but not strict regulations (Essex Planning Officers Association, 2018; Salford City Council, 2023), it is a key area where change could make a large difference. Improved understanding of the effects urban form has on land cover could assist with policy development (Conway & Hackworth, 2007) and a standardised policy may negate some of the disparity in gardens exposed throughout this thesis.

# Chapter Six: Valuing private gardens and the impacts on residents.

## 6.1 Introduction

The two previous chapters have investigated urban household gardens and their residents in the City of Salford, Greater Manchester. They have used different causes of variation, socio-economics and urban form, to explain the heterogeneity of gardens. To further this and to fill a gap in the literature exposed in Section 2.7, Chapter Six focuses on monetising these benefits (Section 6.2) on a local and national scale and comparing them to public urban greenspace to increase the evidence base supporting household garden investment. This data were published as part of the Urban Pioneer (see Section 2.2) and investigate objective four of this thesis: to evaluate the economic value of urban household gardens compared to that of public urban greenspace.

## 6.2 Health and wellbeing valuations in Greater Manchester

The health and wellbeing benefits of gardens and gardening have been published extensively (Section 2.6). Yet, in Great Britain 12 per cent of households had no access to a private or shared garden during the Covid-19 lockdown (ONS, 2020b). And environmental policy continues to overlook household gardens (DEFRA 2020; DEFRA 2021) as the accessible, vital resource they are to residents. This thesis aims to improve the evidence base on the importance of gardens, and to provide an estimate of the economic value of household garden use, the value of a garden view, and the value of the use of a greenspace. Both the lowest and highest economic estimates (Table 6.1) were used from the NEA report 2010 (Mourato et al., 2010), the full methodology for which is discussed in Section 3.11. This methodology of monetising these parameters was not repeated in the NEA Follow On report. To calculate the use of a garden value the number of people (>16 yrs of age) who were likely to garden (50.2 per cent of the population in 2018) was multiplied by the lowest and highest estimates, £171 and £575 per person respectively, adjusted for inflation as outlined in the methods. A non-countryside greenspace refers to parks, recreation sites, and cemeteries (Mourato et al., 2010). It was calculated based on the number of people (>16 yrs of age) who frequented parks (57 per cent in 2015/16) monthly or more, based on £112 and £377 estimated value per person. An attempt was made to quantify the value of having a view over a greenspace by interpreting greenspace exclusively as a garden. Therefore, the lowest and highest economic estimates (£135 and £452) were multiplied by the number of households with a garden (87 per cent).

Table 6.1: An estimation of the monetary value of the health and wellbeing benefits associated with: 'use of a garden', 'use of non-countryside greenspace', and 'having a view over greenspace'. Districts within Greater Manchester are ordered by decreasing percentage garden cover to match Table 4.7.

| **Country/Region/District** | **Health and Wellbeing - value estimates for 2018 (£ millions)** | | |
|---|---|---|---|
| | **Use of Garden** | **'Use of non-countryside greenspace'** | **'Having a view over greenspace'** |
| UK | 5847–19,660 | 4348–4637 | 4113–13,774 |
| North West | 638–2144 | 474–1596 | 461–1545 |
| Greater Manchester | 246–827 | 183–616 | 174–583 |
| Stockport | 25.9–87.2 | 19.3–64.9 | 19–62 |
| Manchester | 46.0–154.8 | 34.2–115.3 | 32–108 |
| Trafford | 20.7–69.7 | 15.4–51.9 | 14–48 |
| Salford | 21.4–72.0 | 15.9–53.6 | 17–56 |
| Tameside | 20.1–67.5 | 14.9–50.2 | 15–39 |
| Bolton | 25.3–85.2 | 18.8–63.4 | 18–59 |
| Bury | 16.9–57.0 | 12.6–42.4 | 12–31 |
| Wigan | 29.1–97.8 | 21.7–72.8 | 21–71 |
| Oldham | 20.6–69.2 | 15.3–51.5 | 14–46 |
| Rochdale | 19.4–65.1 | 14.4–48.5 | 13–45 |

The values presented in Table 6.1 are economic estimates of the health benefits of three parameters of greenspaces devised by the author. The valuations estimate the amount of money saved by health services. In 2018 the budget for the NHS in England was approximately £125 billion (Kingsfund, 2020). Using the lowest estimate for each, the use of a garden, the use of a greenspace, and having a view over a greenspace are each worth approximately 4 per cent of the national NHS annual budget for 2018. Within the private greenspace setting, if each resident with a household garden utilised them frequently and benefitted from their diverse views, the NHS could save 8 per cent of its budget, approximately £10 billion annually. This potential saving is the amount the NHS currently spends on diabetes illnesses annually (GMCA, 2016a).

The monetary valuations, however, are tentative and as outlined in Section 3.11 they have some drawbacks. Most noteworthy is the lack of a solid consensus around the use of QALY (Tilling et al., 2009; Willis, 2005), and that the estimations used in Table 6.1 were calculated in 2010. But, QALY are still used as a valid measure within the medical field, both in the UK (Martin, Claxton, Lomas, & Longo, 2023; Spencer et al., 2022) and in Europe (Fischer et al., 2023). An alternative approach to monetise environmental benefits would be SROI, as used for health walks in Glasgow (Allen & Balfour, 2014; Carrick, 2013) and the Green Gym project throughout England (Yerrell, 2008). But, there is a lack of SROI estimates within private UGI as this approach is more suited to environmental interventions which have a strong social aspect (Moron & Klimowicz, 2021) which gardens typically do not. As mentioned, the estimates in Table 6.1 are using figures published in 2010 (Mourato et al., 2010), to the knowledge of the researcher there has not been a more contemporary account of similar figures which allow all three aspects, the use and the view of a garden and the use of a greenspace, to be compared. It was not within the scope of this thesis, nor the researcher, to publish such an account. Yet, these estimates do give a more up to date reflection of the potential benefits of gardens to residents. Following the methodology outlined in Section 3.11 they provide data for 2018; the year was chosen to be in-line with the household sample in Chapter Four and Five.

Another alternative valuation method is contingent valuation; a widely used method in environmental cost-benefit analysis and environmental impact (Venkatachalam, 2004). Contingent valuation aims to estimate benefits of public goods in nature by assessing their value through a survey of willingness to pay (Venkatachalam, 2004). Calculating the willingness to pay will assist in revealing respondents willingness to contribute to a particular environmental goal and is important to understanding public perception (Nur-Shafiza, Syamsul-Herman, Wan-Norhidayah, 2023). Contingent valuation allows for the reporting of both use and non-use benefits of greenspaces and can therefore help researchers capture a full range of values (Latinopoulos, Mallios, Latinopoulos, 2016), similar to those presented in Table 6.1. Contingent valuation is commonly used in valuing public UGI as an effective method in assessing the value of environmental change to inform decision making (Brander & Koetse, 2011; Exposito, Espinosa, & Villa-Damas, 2021; Lynch, Spencer & Edwards, 2020). A contingent valuation method was not conducted as part of this thesis as the collection of primary data was outside the remit of this research, also the survey methodology needed for such a valuation does not align with the philosophical stance of this researcher. Therefore, the secondary values provided by the NEA in Table 6.1 were used to provide a comparison of value of gardens and public UGI.

In 2016, Greater Manchester became the first region in England to control its own health and social care budget, with decisions being made by The Greater Manchester Strategic Partnership (BBC, 2016). In 2016 the budget to be shared across all 10 Greater Manchester districts was £6 billion (BBC, 2016). Locally, the use and view of a garden is worth approximately between £420 – £1470 million annually (the sum of the minimum and maximum values for G.M in Table 6.1). As the monetary value in Table 6.1 are calculated using QALY (Section 3.11) they represent a measure of money saved by the health service by the use and view of a garden, this equates to between just less than 8 per cent to 25 per cent of GM's health and social care budget. This estimated worth is equal to the expenditure by the NHS in GM on long-term conditions linked to poor mental health and wellbeing (GMCA, 2016b). The Greater Manchester Combined Authority estimated the Greater Manchester financial deficit for the health and social care system to be approximately £2 billion in 2020/21 (GMCA, 2016b). This deficit is presumably now higher due to the Covid-19 pandemic. As shown above and by the benefits of gardens outlined in Section 2.6, the use and view of a quality garden could be one way of reducing this shortfall.

Table 6.2: The natural capital estimate of the health benefits of outdoor exercise and exposure to nature for built up areas and gardens, as well as estimated numbers of people, in the UK as collected by a Natural England national survey (Monitor of Engagement with the Natural Environment and People and Nature Survey, Natural England, 2022). 2018 numbers are in bold as this is the year other valuation data are reported for (Table 6.1).

| | 2016 | 2017 | 2018 | 2019 | 2020 |
|---|---|---|---|---|---|
| **Health Benefits - outdoor exercise (£million)** | 2901 | 3529 | **4175** | 3997 | 3288 |
| **Health Benefits - exposure to nature (£million)** | 3840 | 4016 | **4121** | 4249 | 2398 |
| **Number of People - outdoor exercise** | 7,216,783 | 8,359,772 | **9,349,732** | 8,987,289 | 11,479,745 |
| **Number of People - exposure to nature** | 11,585,505 | 12,117,749 | **12,434,791** | 12,819,804 | 7,234,509 |

Since 2009 Natural England have undertaken the Monitor of Engagement with Natural Environment (MENE) survey to capture, on a national scale, how people (≥16yrs are engaging with the environment (Natural England, 2022). Table 6.2 provides the natural capital account for built up areas and gardens 2016 – 2020 in the UK. Natural capital is the world's stock of natural assets, it is from this stock that humans derive a wide range of services (ecosystem services) which make human life possible (World Forum on Natural Capital, n.d.). Table 6.2 estimates 14 per cent of the UK population in 2018 were conducting outdoor exercise and 19 per cent were gaining benefits from being exposed to nature. Combining the figures for outdoor exercise and exposure to nature gives a health benefit estimate of 8296 (£million) for 2018 in built up areas and gardens from Table 6.2. This figure falls on the lower side of the estimate in Table 6.1 of 5847–19,660 £million for the use of a garden, though it does show using the most conservative value within this estimation is a realistic minimum for monetising the benefits of gardens.

## 6.3 Personal investment: money

To test how the level of personal investment into gardening varies the following steps were taken. The estimates for household garden spend were calculated by the author using raw data from the Horticultural Trades Association from 2018. Spend on goods incorporates the following: tools and equipment, ornaments and furniture, fertilisers and chemicals, sheds and greenhouses. Spend on services ranges from routine maintenance to larger-scale landscaping.

Table 6.3: An estimation of the spend on household garden goods and services for 2018 for the UK, Greater Manchester, and its districts. Districts within Greater Manchester are ordered by decreasing percentage garden cover (Horticultural Trades Association, 2018).

| Country/Region/District | Spend on household garden goods (2018) (£ million) | Spend on household garden services (2018) (£ million) |
|---|---|---|
| UK | 7725 | 2472 |
| North West | 717 | 270 |
| G.M. | 285 | 107 |
| Stockport | 30 | 11 |
| Manchester | 53 | 20 |
| Trafford | 24 | 9 |
| Salford | 25 | 9 |
| Tameside | 23 | 9 |
| Bolton | 29 | 11 |
| Bury | 20 | 7 |
| Wigan | 34 | 13 |
| Oldham | 24 | 9 |
| Rochdale | 22 | 8 |

Nationally, the UK spent approximately £10.1 billion (the sum of spend of goods and services in Table 6.3) on gardening goods and services in 2018. The total gross disposable household income in the UK in 2018 was £1402 billion (after taxes); therefore, gardening goods and services equate to less than one per cent of this figure. In Great Britain (GB), the average annual spend on house improvements was £22 billion in 2018, including both property and garden renovations (Leaders, 2019); therefore, using the data in Table 6.3, almost half of this spend was on household gardens. Annual household spend, including all sectors, for 2018 was approximately £30,446 (ONS, 2020). Annual household spend on gardening goods and services for the same year was approximately £385, just over 1 per cent (HTA, 2018). Gardens and gardening have increased in popularity post Covid-19 pandemic (Horticulture Week, 2020); and this is also shown in the increase in national spend. Residents across Great Britain spent approximately £16 billion on their gardens 2020-21 (Independent, 2021), an approximate 50 per cent increase in 3 years excluding Northern Ireland.

The Covid-19 lockdown undoubtably changed the way many valued their gardens. An article published in the Property Reporter in June 2020, on the 100th day of national lockdown, illustrates the change in attitude residents had towards their gardens. LV insurance found 74 per cent of Britain's with a garden agreed that they changed how they used it during lockdown, with increases in both the time and money being spent (Property Reporter, 2020). The mean spend on a garden during lockdown was reportedly £125 per person, with millennials spending £213 per person on average (Property Reporter, 2020). This age group had the highest spend of any, likely attributed to their young families with one in five saying they invested to provide an outdoor play area for their children. The estimation per person on garden spend for 2018 (used in Table 6.3) generated within this thesis was £192.50. This figure and those reported in the Property Reporter are similar; though estimations are for two different years their similarity gives credit to the estimates given in Sections 6.2 and 6.3 of this thesis. Millennials also held their garden in a higher regard for its ability to improve overall wellbeing compared to the national average, 45 per cent and 39 per cent respectively (Property Reporter, 2020). Interestingly, Britain's who claimed on the government furlough scheme, designed to support employers to retain and continue to pay staff while businesses were closed (Clark, 2021), spent on average £50 more on their gardens than the national average and were more likely to add greenery to their living spaces, with two thirds of the population doing so (Property Reporter, 2020).

Taking the lowest estimates from Table 6.1, a single resident receives a value of £306 in health and wellbeing benefits from using and viewing their garden. The average household contains two adults (ONS, 2019). Therefore, each household receives £612 worth of benefits, yet spends £385 on gardening goods and services. So, for every £1 spent by residents on household gardens the national health care system is saving £1.60 in health and wellbeing benefits. Based on the assumption that money spent on household gardens leads to both residents spending time gardening and the garden becoming a more diverse and an aesthetically pleasing place. For comparison SROI analysis of community schemes can illustrate the potential of gardens. The Green Gym project established by British Trust for Conservation Volunteers estimated for every £1 invested between 2005 and 2009 £2.55 was saved in treating illnesses relating to physical activity (Yerrell, 2008). The return on investment is likely greater in community projects such as Green Gyms as they have a sociability factor which gardens do not, making them unsuitable for SROI (Moron & Klimowicz, 2021).

In summary, a reflection on how garden spend has increased provides hard evidence of the increase in popularity of gardening. UK spend on gardens in 1999–2000 was estimated at £2.62 billion (Gaston et al., 2005), in 2018 UK garden spend was estimated at £10.1 billion (Table 6.3), and in 2020-21 Great Britain's garden spend was estimated at £16 billion (Independent, 2021). Whilst these three sources used different methods and two are for the UK and one for GB, they illustrate the trend in growth of residents investment in their gardens. Understanding the amount of money spent on gardens and the return this provides is vital. Having such estimations build the evidence base on why gardens are an important UGI, encouraging stakeholders, such as governmental bodies, to invest in them which is currently lacking (DEFRA 2020; DEFRA 2021).

## 6.6 Summary

Chapter Six takes a quantitative approach to understanding how residents value their private gardens. This approach was taken to address the gap in the knowledge of understanding the variation the socio-economic status of residents has on their gardening engagement. Research has extensively used qualitative methods to investigate the interplay between gardening preferences and socio-economics (Garcia-Antunez et al., 2023): for example gardening knowledge (Goddard et al., 2013); pro-environmental attitudes and beliefs (van Heezik et al., 2013); and motivations (Loram et al., 2011).

In this chapter objective four of this thesis was investigated: to evaluate the economic value of urban household gardens compared to that of public urban greenspace. Using the most conservative values in Table 6.1 as calculated by the author, the use of a garden was estimated to have a health and wellbeing value of £5847 million and use of a non-countryside greenspace of £4348 million. Therefore, not including the potential value of a garden view, household gardens were estimated to be more valuable to urban residents' health and wellbeing than public spaces. These data were published as part of the Urban Pioneer to increase the evidence base of household gardens in environmental policy (discussed further in Section 7.4). Each of the use of a garden, the view of a garden, and the use of a non-countryside greenspace were worth the equivalent of approximately 4 per cent of the national NHS budget, thus combining the two parameters for a garden shows its greater potential worth. The prospective monetary savings for the NHS, both locally and nationally, from residents utilising their household gardens could be substantial. Nationally, Table 6.1 estimates a potential saving of approximately £10 billion, equivalent to 8 per cent of the national healthcare budget. Within Greater Manchester, the equivalent to between one quarter and nearly three quarters of the financial deficit in the healthcare system could be saved by people utilising their gardens for health and wellbeing benefits.

# Chapter Seven: Discussion

## 7.1 Introduction

Environmental injustice is defined as ‘the fair treatment and meaningful involvement of all people … and (the) enforcement of environmental laws, regulations and policies’ (United States Environmental Protection Agency, n.d). It is a topic which has gained greater traction in contemporary society as there is an increased social awareness of issues of equality, fuelled, in part, by the Black Lives Matters and Times Up campaigns in America (Time 2018; Time, 2021) and in the UK (BBC, 2021). The impacts are seen in a multitude of settings from media to policy, within offices to public parks. This thesis sits within the realms of social science and urban ecology, and as such assesses variation within the access, size, and content of the urban private greenspace household gardens. The topic of household garden use has been evermore popular post-pandemic, with the Covid-19 lockdowns seeing unprecedented garden popularity within the UK (Kingsley et al., 2022; Property Reporter, 2020). Yet, the resource of household gardens is not distributed evenly, as is exposed throughout this thesis.

## 7.2 How does the socio-economic situation of residents affect their gardens

The first research question presented for this thesis (Section 1.4) was: in a time when environmental equality is highly topical, is there a disparity in the size and diversity of urban household gardens resulting in an injustice in the access to resources between different socio-economic groups? It is well documented that the size and quality of a greenspace limits the health and wellbeing benefits to people, with household gardens as no exception (Apparicio et al., 2016; Burt et al., 2014; Dillen et al., 2012; Wustemann, Kalisch, & Kolbe, 2017). But these spaces have been documented to be viewed as a luxury to some, often those who are from less economically developed communities (Apparicio et al., 2016; Birch et al., 2020; Burt et al., 2014). However, not all research has supported the luxury effect; Mavoa et al., (2015) and Kimpton (2017) found no significant difference between vegetation cover and socio-economic status for example.

The investigation into the disparity of garden size and diversity required a detailed exploration to assess the luxury effect at a finer geographic detail. This thesis utilised data collected from 6881 gardens across the City of Salford equating to 68 sample Output Areas (OAs), this sample size was chosen according to the guide outlined by Denscombe (2014) for a representative sample (see Sections 3.6-3.8 for a full methodology). There has been a call for a more descriptive approach to assessing socioeconomics within UGI to incorporate the complexity and fluidity of society (Liotta et al., 2020; Sholry, Connolly, & Anguelovski, 2020; Calderon-Argelich et al., 2021). Therefore, the OAC was chosen to provide detailed socio-economic information at the fine geographic scale of OA; these demographic data are commonly used in the field (Dearden, Lloyd, & Catney, 2019; Druckman, Sinclair, & Jackson, 2008; Walford & Armitage, 2020) and have been shown to be an accurate representation of deprivation (Jerrim, 2020). Satellite imagery of each garden was used from Google Earth, this overcame the usual limitation of gaining access as highlighted by Maantay, (2007); Vaz et al. (2017); and Bell et al. (2020). As well as being a free resource allowing for a large sample size with limited resources that was at a high enough resolution to pick out individual features (Rahman et al., 2011).

Garden size has not previously been assessed as a form of environmental injustice, despite its potential as an easily accessible greenspace for urban residents. Three Supergroups out of seven had median garden size below the overall average for the City of Salford 129 m²: Constrained City Dwellers 99 m², Multicultural Metropolitans 63 m², and Cosmopolitans 20 m². Suburbanites had the largest median garden size 176 m², this was expected as residents within this Supergroup are typically semi-detached homeowners (n=853) and garden size has been found to increase with property type (see Chapter Five and Dewaelheyns et al., 2014; Gill et al., 2008). Cosmopolitans have a higher proportion of renting and living in communal establishments with a higher proportion of students than other Supergroups. Multicultural Metropolitans have a high proportion of renters, both social and private, and are often families with children and high unemployment. Constrained City Dwellers are also social renters with high unemployment.

Interestingly, Hard-Pressed Living, although categorised as having lower education attainment and occupying manual jobs and retail (ONS, 2011), also have a high proportion of semi-detached housing in the City of Salford (n=1227). Importantly, within Northern England this is often socially rented properties (ONS, 2011). The City of Salford has a high proportion of social housing, and an even greater demand for it. There were 6,094 households on the waiting list for social housing in the City of Salford in 2019, with only around 1,000 households being placed annually (Salford City Council, 2020). The city's strategy to combat this is to increase the number of high-rise residential homes in the city (Salford City Council, 2020). Within the same housing strategy 2020 – 2025 document outlining these development plans there is no mention of how these residents will access outdoor space.

As median garden size was found to vary significantly between Supergroups (Table 4.2), garden content was compared, as discussed in Chapter Two it is the size and quality of the greenspace which results in varying health and wellbeing benefits. Radar graphs drawn for each of the seven Supergroups (Figure 4.1) illustrates the mean percentage land cover of lawn, pave, trees & shrubs, and plants. The data presented in Figure 4.1 highlighted two main areas where gardens differed between Supergroups: the amount of vegetation cover and the mean feature diversity. Suburbanites had the largest vegetation cover at 21 per cent compared to Constrained City Dwellers at 10 per cent. Comparing mean feature abundance, (included features being lawn, pave, trees, shrubs, and plants equating to a total score of five), Suburbanites had the highest at 3.32 with Urbanites second at 2.84. Hard-pressed living, Cosmopolitans, Ethnicity central, and Constrained City Dwellers all had similar mean feature abundance ranging from 2.52 to 2.69 features, with Multicultural Metropolitans lowest at 2.01. Jerrim (2020) classified Constrained City Dwellers and Hard-pressed Living as the most disadvantaged Supergroups. Constrained City Dwellers had the smallest proportion of vegetation within their gardens and the second lowest feature diversity, suggesting these residents are also disadvantaged of quality private greenspaces (Figure 4.1).

From Figure 4.1, the most at risk societal groups for an inequality in the size and diversity of gardens are Cosmopolitans and Multicultural Metropolitans due to their small garden sizes (20 $m^2$ and 63 $m^2$ respectively) and limited feature diversity (2.68 and 2.01 respectively), and Constrained City Dwellers due to its small vegetation cover (10 per cent) and feature diversity (2.52). Only one of the two Supergroups Jerrim (2020) categorised as the most disadvantaged are included in this list: Constrained City Dwellers. According to the Pen portrait descriptions of the Supergroups provided by the ONS, Constrained City Dwellers are more likely to live in flats and social rented housing, be residents with lower qualification levels than the national average and have a high level of unemployment (ONS, 2015). High-rise flats were not included in the data collection for this thesis, although houses which were converted into flats were. The mode house type for Constrained City Dwellers in the City of Salford was Terraced at 61 per cent of houses. Constrained City Dwellers are more likely to be over 65 years of age, have older children over 14yrs, and be single or divorced (ONS, 2015). Multicultural Metropolitans has a high ethnic mix with residents likely to rent terraced housing both social and private, there is an above average proportion of young families (children < 14yrs), unemployment is above average and fewer households have multiple vehicles (ONS, 2015), suggesting less disposable income. The mode house type for Multicultural Metropolitans was also Terraced at 86 per cent of houses. Cosmopolitans are more likely to live in flats or private renting in highly urban areas, it has a high ethnic mix and residents are likely to be young adults with no children either employed or full-time students. The University of Salford has approximately 22,000 students (University of Salford, n.d.) with many living across the surrounding city in their 11 official accommodation options and an assortment of private renting options. The limited diversity of Cosmopolitans gardens' is likely due to their age, 16–24-year-olds are the least likely group to garden as a free-time activity (Chapter Six), and their lack of permanence within the area with many being students likely prevents them from investing in their gardens.

Overall, comparing these three Supergroups shows two have many similarities, Constrained City Dwellers and Multicultural Metropolitans, whilst Cosmopolitans have a very different demographic. Cosmopolitans are mainly students and young people, unlikely to be interested in gardening (Chapter Six), and more likely to be temporary visitors to the city instead of permanent residents. Therefore, garden size and content are likely to be unimportant to them, though post-pandemic these views may have changed (Dzhambov et al., 2021). Constrained City Dwellers and Multicultural Metropolitans however have a high proportion of terraced housing, rented accommodation, and unemployment. Both also consist of high proportions of families, with Constrained City Dwellers having children over 14 yrs. and Multicultural Metropolitans being younger families. Nationally, only 31 per cent of residents feel their garden is a place where children can play (Table 6.4). Further, residents who are working class or non-working were the least likely socio-economic groups to say their gardens were suitable for children (Figure 6.11), supporting the idea that these groups are experiencing an injustice within the size and quality of their private greenspace.

A similar trend between all three target Supergroups, Cosmopolitans, Multicultural Metropolitans, and Constrained City Dwellers is higher proportions of rented housing. As with Cosmopolitans, it is unsurprising residents in rented housing are less likely to invest in their gardens, with their accommodation having less permanence. Mee, Instone, Williams, Palmer, and Vaugan (2013) highlight how a tenants relationship with urban ecologies such as gardens are mediated by lease contracts and efforts towards sustainability measures are restricted by landlords and policies. According to the Evening Standard (2022) the number of 26–41 year-olds buying houses has dropped by 11 per cent in five years, further, the Financial Times (2022) reported 36 per cent fewer houses for sale at the start of 2022 in the UK compared to 2020. If renting properties is increasingly becoming more popular, and renters are less likely to invest in their gardens as has been suggested in this thesis it could result in a trend of unmanaged and thus unused gardens by this demographic. Figure 7.1 shows the percentage of adults reporting ill health and their tenancy status, owner-occupiers and private renters both have greater proportion of people in ill health in non-decent homes than decent ones (Tinson & Clair, 2020) suggesting their home may be a contributing factor. Tinson and Clair (2020) state 25 per cent of private rented properties are classified as non-decent, a non-decent home is one which fails to require thermal comfort, modern facilities, or contains hazards. The state of rented housing has been given as an example as academic research into the gardens of renters specifically is lacking, this thesis suggests this gap needs addressing. Although encouraging private landlords to manage their gardens would be difficult, though arguably necessary, social renting should be more accessible. As shown in this thesis these renters are often the owners of less diverse gardens, which limits the environmental and health and wellbeing benefits (Chapter Two) they can provide, perhaps these guidelines need improving. Chapter Six sets out the potential monetary savings the use and view of a garden can have, for example in Greater Manchester approximately £420 - £1470 million, almost one quarter of the financial deficit of the health and social care system. If councils invested in improving and maintaining the gardens in social rented housing, they could exploit this potential saving, whilst improving the natural capital stock of urban areas.

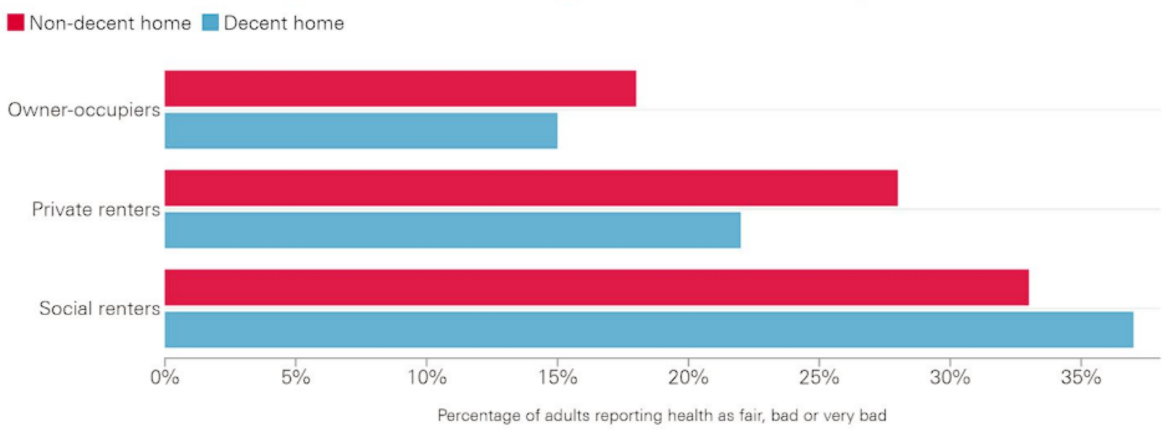



Figure 7.1: The percentage of adults reporting health as fair, bad, or very bad in the English Housing Survey and their house tenancy status (Tinson & Clair, 2020).

The disproportionate vegetation cover between gardens of different Supergroups is an important example of the disparity of access to quality private greenspace. As discussed above at the city-wide level, more vegetation in wealthier communities has been shown in Australia (Luck et al., 2009; Shanahan et al., 2014), in Montreal (Apparicio et al., 2016), in private gardens in Maryland (Grove et al., 2006) and in Greater Manchester (Figure 4.1 of this thesis). There is also a large body of literature on the topic in North America (Leong et al., 2018); whilst in England studies directly on the luxury effect in vegetation are lacking, though several show it in birds (Leong, et al., 2018). This disparity needs highlighting as equal use and access to greenspace is paramount for a thriving society (Brown et al., 2018; Maas et al., 2009; Mears et al., 2019). Further, a lack of quality spaces means a missed opportunity to benefit from the potential health and wellbeing advantages such as increased life satisfaction (Kaplan, 2001), relieved mental fatigue (Moran, 2019), reduced blood pressure (Pretty et al., 2005), and reduced chronic stress (Chalmin-Pui et al., 2020). Moreover, research has shown those in the most socio-economically disadvantaged situations are often those most in need of nature-based interventions (Chapperro, Benzeval, Richardson, & Mitchell, 2018; Gibson, Pollard, & Moffatt, 2021); yet, they are less able to access green amenities (Fixsen & Barrett, 2022).

A quality greenspace (as defined in Section 2.6) is one with diversity (Birch et al., 2020; Chamberlain et al., 2019; van Heezik, Freeman, Porter, & Dickinson, 2013), therefore high vegetation cover and feature diversity are important. Larger gardens where also shown to increase residents self-reported health scores (Brindley et al., 2018; Bell et al., 2020), making this an important factor too. The first research question of this thesis was based on exploring these concepts within urban gardens, specifically investigating whether there was a disparity between socio-economic groups in the size and quality of their greenspaces. It was found Multicultural Metropolitans and Constrained City Dwellers were evidenced to have smaller, less diverse gardens and thus would not have the same potential to benefit from them as other Supergroups within the City of Salford.

## 7.3 The extent, content, and form of urban household gardens

The second research question presented for this thesis (Section 1.4) was: Does the extent of urban household gardens vary in neighbouring Greater Manchester districts and is there evidence of newer housing developments having smaller gardens in the City of Salford? The comparison of collective garden extent was relevant to both research questions one and two; therefore, it was discussed above in Section 7.2. In short, percentage garden cover ranges from 9.25 to 21.07 per cent total land cover across Greater Manchester (Table 5.1). Rochdale and Oldham had the smallest garden cover, with 9.25 and 9.34 per cent respectively; whilst Stockport and Manchester had the highest, 21.07 and 20.31 per cent respectively. Internationally, landcover data have shown similar or greater results for the percentage of garden cover within urban regions: in Sweden 16 per cent (Colding, 2007); New Zealand 36 per cent; and similarly in Belgium 42 per cent of greenspace is gardens (Van de Voorde et al., 2008). In the UK, previous research found garden cover to range from 22 – 27 per cent in five different cities (Gaston et al., 2005; Loram et al., 2007; Tratalos, Fuller, Warren, Davies & Gaston, 2007). This thesis provides the first comparison of garden cover between neighbouring districts. It highlights garden cover ranges more significantly than previous publications have demonstrated (Gaston et al., 2005; Loram et al., 2007; Tratalos, Fuller, Warren, Davies & Gaston, 2007). The median percentage garden cover across Greater Manchester was 14 per cent, collective garden cover in the City of Salford was 14.88 per cent, making it a representative choice of Greater Manchester as a whole, providing justification for its use as the research location for this thesis.

### 7.3.1 Garden Composition

Household gardens are commonly an important factor for house buyers (Mason, 2017); it is unsurprising, therefore, that this feature is also a key factor in house price (Asam & Tiesdell, 2012). Hedonic pricing is 'used in the determination of the economic value for an ecosystem service or external factor that may influence the market price of a good or asset' (Corporate Finance Institute, 2021). There is a growing body of literature within the area of hedonic pricing in urban greenery, within which gardens play a key role and are beginning to be recognised (Brander, & Koetse, 2011; Li, Jiang, Ke, Nie, & Wu, 2021; Nazir, Othman, Nawawi, 2015). Using gardens as a variable in hedonic pricing in the UK shows a property valuation of approximately £11,000 more for houses of the same size with a garden, on average a 5 per cent increase (Pocock & Shaw, 2021). In a study of 30 towns and cities, Pocock and Shaw (2021) found the difference in value to be greatest in Walsall, West Midlands at 16 per cent higher, Sunderland at 15 per cent and Liverpool at 11 per cent were second and third in the ranking. Similar research has shown internationally distance from and/or quality of a greenspace increase house prices: Spain (Morancho, 2003); Malaysia (Nazir, Othman, & Nawawi, 2015); Houston, Texas (Sohn, Kim, Kim, Li, 2020); and Shenzhen, China (Li, Jiang, Ke, Nie, Wu, 2021). Yet, there is some evidence to suggest garden size is getting smaller in the UK (Chalmin-Pui et al., 2021), with some new developments even omitting gardens completely (Douglas et al., 2019).

Whilst the presence of a garden is the important factor in hedonic pricing, it is the quality or content and the size which provide the important health, wellbeing, and environmental benefits (Bell et al., 2020; Chalmin-Pui et al., 2020; Brindley et al., 2018). This thesis conducted the most comprehensive account of English urban garden content in current literature, recording data for 6881 individual houses. Previous similar accounts have surveyed 250 (Gaston et al., 2005) and 267 (Loram, Warren & Gaston, 2008) houses; or those with a more large-scale approach have not recorded the level of feature detail in this thesis (Loram et al., 2007; Tratalos et al., 2007; Cavan et al., 2018; Baker & Smith, 2019). It found the median garden size for the City of Salford gardens was 129 m², 50 per cent of this is permeable landcover (Table 5.2). There was a large range in garden size across the City of Salford, 9 – 1725 m². The mean garden size in the City of Salford (139 m²) was smaller than that recorded by Gaston et al., (2005) in Sheffield of 151 m² but had a similar large range 0 – 1073 m². The City of Salford's mean garden size was also smaller than that reported by

Davies et al. (2009) who presented data from seven urban and suburban areas across the UK with a mean size of 190 $m^2$ and Loram et al., (2008) who reported a mean of 197.5 $m^2$ across five cities. The City of Salford's urban gardens were found to be smaller than previous literature has reported and have more impermeable land.

Table 5.2 shows a mean 47 per cent of garden landcover is impermeable, equating to 61 $m^2$ in the average 139 $m^2$ garden. Loram et al., (2008) reported on average only 28 per cent of land cover was impermeable (combining patio and path) equating to 55.7 $m^2$ of a 197.5 $m^2$ garden. Figure 5.1 shows that gardens in the City of Salford have a general configuration of approximately 40 per cent lawn, 40 per cent pave, and 20 per cent vegetation cover. This garden land cover composition has not previously been published for urban gardens in the UK, with previous studies only including lawn (Gaston et al., 2005) or the mean area ($m^2$) of land cover types (Loram et al., 2008). Graph E on Figure 4.1 highlights the Cosmopolitans Supergroup were the outlier, with a configuration of approximately 64 per cent pave, 24 per cent lawn, and 12 per cent vegetation cover. Previously Gaston et al., (2005) reported an estimate of 60 per cent lawn cover in gardens across Sheffield. This estimate is significantly larger than the one reported in this thesis. It may show a trend towards increased paving in contemporary gardens. Gaston et al. (2005) was approximately 10 years prior to this thesis, since then paving over gardens, particularly front gardens has increased in popularity (Kelly, 2018; Perry & Nawaz, 2008). However, the sample size in Gaston et al., (2005) was 250 houses compared to the 6881 in this thesis suggesting it may have not been a representative sample of houses at that time. To support this, Loram et al. (2008) also estimated mown grass cover as 43 per cent in their study of 267 gardens across five cities. Comparing tree cover to previous research shows similar results. Davies et al. (2009) reported 11 per cent tree cover in their national study of gardens, 11 per cent tree cover was also the overall mean when combining all gardens surveyed in this thesis (Figure 4.1).

The data in Figure 4.1 also highlight the extensive cover of paving in urban gardens, an under-researched area of urban planning (Scott et al., 2014). Whilst the impact of impermeable surfaces was not an aim of this thesis, the data contribute to the evidence base needed to expose the extent of the problem; therefore, it is briefly discussed. This thesis has found gardens in the City of Salford are comprised of approximately 40 per cent paving, with pave occurring in 6880 of 6881 gardens sampled (≥ 5 per cent land cover). The collective size of gardens is substantial (Loram et al., 2007; Kelly 2018), and their ecological importance for controlling surface water runoff and is vital (Kelly et al., 2018). The Royal Horticultural Society (2006) reported nearly half of all gardens in North-East England and almost one third in South-West and Eastern England had at least 75 per cent paving in their front garden. By covering gardens in impermeable paving, rainwater can no longer infiltrate into the ground causing increased runoff, increased pollutant loading, and increased peak flows which can all increase flood risk (Willems et al., 2012; Kelly et al., 2018). Localised flooding is of particular importance in the City of Salford with its flat topography and two rivers, the Irwell and Glaze Brook (Salford City Council, 2008).

Modelling of gardens in the City of Manchester has shown a potential 6 per cent decrease in rainfall absorption due to, in part, the increase in impermeable surfaces (Cavan et al., 2018). Cavan et al. (2018) reported only 50 per cent of total garden area was greenspace in the City of Manchester and highlighted the substantial effect this has on rainfall absorption and surface temperature. This thesis found a similar land cover split, with a median of 50 per cent permeable and 50 per cent non-permeable landcover, likely resulting in similar environmental consequences (Cavan et al., 2018) in addition to increased flood risk (Kelly, 2016; Perry & Nawaz, 2008; Scott, Shannon, Hardman, & Miller, 2014).

Another type of impermeable surface found in gardens is artificial lawn, which has increased in popularity in recent years (Francis 2018; Brooks & Francis, 2019). But, as with paving, this impermeable surface has negative environmental impacts such as increased rainfall runoff and decreased infiltration, as well as decreased carbon sequestration (Francis, 2018). In the UK a petition to the government to ban the sale of artificial grass was signed by over 11,600 people in 2020 due to the potential damaging environmental impacts (Simpson & Francis, 2021). It was not possible from the methodology used in the thesis to detect types of paving or to detect artificial lawns, however, it was not an aim of the research to do so. This thesis does provide an accurate estimate for the proportion of paving in urban gardens which was a gap in the literature and important for the evidence base on ecosystem services of greenspaces in urban areas.

Garden size was found to significantly affect content in the City of Salford's urban gardens. Increasing garden size had a positive linear correlation with feature abundance (trees, shrubs, plants, lawn, & pave), $r_s = 0.59$ p = ≤0.05 (Figure 5.3). Median values for each feature abundance category were: 1 = 42 m², 2 = 106 m², 3 = 142 m², 4 = 179 m², 5 = 254 m². A similar relationship was found by Loram et al. (2008) in Belfast ($r_s$ = 0.66, df = 53, p = <0.0001), Cardiff ($r_s$ = 0.61, df = 52, p = 0.0001) and Leicester ($r_s$ = 0.73, df = 47, p = 0.0001) with increasing land use richness. Positive linear correlations were also found in the City of Salford gardens between increasing garden size, shrubs ($r_s$ = 0.45 p = ≤0.05) and green cover ($r_s$ = 0.54 p = ≤0.05). It is logical to expect larger gardens to have a greater feature diversity and a greater cover of vegetation, these were also shown by Loram et al. (2008). Shrubs, however, had not previously been recorded in the literature, but have, by the data presented in this thesis, now been shown to follow the same relationship. Whilst the correlations here are not strongly positive, they are significant and illustrate the environmental importance of a larger garden.

This thesis reports urban gardens in the City of Salford had a mean percentage vegetation land cover of 16 per cent, 13 per cent trees and shrubs and 3 per cent plants (Figure 5.1). Trees occurred in 22 per cent of gardens, shrubs in 50 per cent, and plants in 23 per cent (Table 5.3). Previous studies have used count data to record vegetation rather than landcover. Gaston et al. (2005) found 48 per cent of gardens in Sheffield had at least one tree over 3m tall. Davies et al. (2009) reported a similar figure of 54 per cent of gardens having at least one tree over 3m, extrapolating from their data gives a mean tree land cover of 11 per cent. Comparing the City of Salford to previous research shows a far lower percentage occurrence of trees, 22 per cent in Salford compared to 48 per cent (Gaston et al., 2005) and 54 per cent nationally (Davies et al., 2009); and trees in these two publications had a minimum height of 3m whereas there was no minimum in this thesis. These data suggest the City of Salford has far fewer trees within domestic gardens than other urban areas. Table 5.4 scales up these findings to conclude an approximate total of 84,133 trees in the City of Salford, equating to 8 per cent of the total number of trees in Greater Manchester. These data suggest the City of Salford's urban garden trees could contribute far more to Greater Manchester's tree total, which is vital to increase reach the carbon neural goal of 2038 (City of Trees, n.d.).

As stated above and throughout Chapter Two, it is the quality of a greenspace which has great bearing on its health and wellbeing benefits. The amount of vegetation is an important factor in this quality, with rich environments being shown to reduce stress (Hartig et al.,

2003) and mental fatigue (Moran, 2019). A more direct link was reported by Chalmin-Pui et al. (2020) who found landscaping previously baron front gardens with ornamental plants resulted in decreased perceived stress and a reduction in chronic stress after 3-months (n = 38). Further, in a national study (n = 6015) the same researchers found increasing vegetation cover by 10 per cent increased residents self-reported wellbeing score by 26 per cent (Chalmin-Pui et al., 2021). The 16 per cent vegetation found in the City of Salford, compared to 44 per cent pave and 40 per cent lawn, highlights the lack of feature variation in Salford's gardens. This thesis filled a gap in the knowledge by reporting both count and percentage land cover data for trees and shrubs, resulting in rich composition data used to draw radar graphs in Chapter Four and Five, and occurrence data which exposed the comparative lack of trees in household gardens compared to other urban areas in the UK.

### 7.3.2 Urban form

Urban form, the physical characteristics that make up built-up areas (Williams, 2014), has been shown to contribute to the heterogeneity of urban household gardens (Dewaelheyns, Rogge, & Gulinck, 2014; Gaston et al., 2005; Loram, Tratalos, Warren, & Gaston, 2007; Loram, Warren, & Gaston, 2008). In the UK 4600 hectares of garden land was converted to other uses between 2013–2017 (Brindley et al., 2018). There has been a call for a deeper understanding of the relationship between urban form and land cover (Conway & Hackworth, 2007), through an approach which captures the fine scale datasets needed in rich urban environments (Grafius et al., 2016). Such data would increase the evidence base and give policymakers and stakeholders a more compelling argument for keeping household gardens in newer developments. Therefore, this thesis set out to test three urban form characteristics, house type, density, and age, against their garden size and content.

In the UK, the most common urban house types are terraced, semi-detached, or detached, hence these categories are often used in urban form research (Loram et al., 2008; Cavan et al., 2018; Loram et al., 2007; Tratalos et al., 2007), although Baker and Smith (2019) split terraced housing into modern and Victorian this approach was not taken in this thesis as age of development is discussed separately below. Data from this thesis has shown median garden size (combined front and back) more than doubles from terraced 72 $m^2$ to semi-detached 156 $m^2$, with a further increase in size in detached houses 202 $m^2$ (Table 5.5).

Garden size data in this thesis had a non-normal distribution, therefore median is the correct central tendency. However, for comparison to the literature mean sizes were also included: terraced 83 $m^2$, semi-detached 170 $m^2$, and detached 257 $m^2$ (Table 5.5). Loram et al. (2008) reported mean garden size across five cities in the UK as larger than those in the City of Salford, terraced 94 $m^2$, semi-detached 189, and detached 315 $m^2$, although the data follow a similar pattern. Detached housing was least common in the City of Salford, contributing only 8 per cent to the proportion of properties surveyed, similar results were found in Belfast, Leicester, Oxford, and Cardiff where detached housing ranged from 11 – 13 per cent of houses (Loram et al., 2008). Semi-detached housing was the most common in the City of Salford at 48 per cent of houses surveyed (Table 5.6). Previous literature has shown detached and semi-detached housing have greater carbon sequestration (Tratalos et al., 2007) than terraced housing, therefore cities which have high proportions of this house type, like the City of Salford's 44 per cent, are limiting their potential ecosystem services.

House type has also been shown to affect the content within urban household gardens in one study by Loram et al. (2008). In their study, Loram et al. (2008) used 267 in-field surveys across five UK cities, measuring garden size and recording land cover it was reported land cover did vary between terraced, semi-detached, and detached houses. Thirteen features were recorded (see Table 2.2 for a full list), including patio, path, and mown grass which were also recorded in this thesis but omitting any mention of vegetation. When pooling the data from all five cities it was found terraced housing had significantly lower land-use richness than the other two property types (two-way ANOVA $f = 23.7$, $df = 2,240$, $p = <0.001$) (Loram et al., 2008). Data from this thesis for gardens in the City of Salford shows a similar result, feature abundance decreased with house type from a median of 4 in detached, to 3 in semi-detached, and 2 in terraced (Table 5.7). The number of shrubs decreased between house types, as expected due to the decrease in garden size and the relationship explored above between size and content. Interestingly, median percentage green cover also significantly decreased from detached 67.5 per cent, to semi-detached 60 per cent, to terraced 30 per cent (Table 5.7). Converting the data to a percentage in Loram et al. (2008) gives a mean land cover of 34 per cent for mown grass, not including vegetation as this thesis did, but giving a very similar figure for permeable land within terraced gardens. Baker and Smith (2019) also reported the same 34 per cent green land cover (grass and shrubs) for Victorian terraced houses. The evidence suggests, therefore, that terraced housing is the most limiting for ecosystem services such as the percolation of rainwater.

In their urban form study, Baker and Smith (2019) distinguished between Victorian and non-Victorian terraced housing and found green cover to differ significantly between the two. Yet, the era of housing development is a relatively unresearched variant when investigating household gardens and their content. This thesis used the 6881 the City of Salford urban houses surveyed and five development ages: Historic (Pre 1914), Interwar (1918 – 1939), Postwar (1945 - 1959), Sixites/Seventies (1960 - 1979), and Modern (1980 – current) to begin to fill this research gap. Garden size was found to have a weak positive relationship with development age ($r_s = 0.38$ $p = \leq 0.05$). Median garden size increased from Historic to Postwar housing, 45 m², 144 m², 158 m², but decreased and levelled off in developments since the Sixties/Seventies at 133 m² and 138 m² for Modern (Figure 5.7), and 129 m² from the 2018 household sample. There were significant but weak positive correlations between age and green cover ($r_s = 0.42$ $p = \leq 0.05$), and feature abundance ($r_s = 0.31$ $p \leq 0.05$). Green cover was lowest in Historic housing at 31 per cent and highest in Modern at 61 per cent, but the eras in between did not show a clear linear relationship. Similarly, mean feature abundance was lowest in Historic at 2.27 and Interwar at 2.71, then highest in Modern at 3.00, but Interwar and Sixties/Seventies were not linear.

Whilst the exploration into development era and garden size and content (Section 5.4) did show some significant and interesting trends, further research is needed. Due to the sample size, it was not possible to analyse development era and house type independently, which would be the next step to further the investigation into these relationships. However, the research does suggest garden size has levelled off in housing developments since the Sixties/Seventies. Therefore, this thesis does contribute evidence to show garden size is smaller in newer housing developments, the second part of research question two presented in Section 1.4. Further, percentage garden cover ranged from 9.25 to 21.07 per cent total land cover in districts across Greater Manchester, illustrating collective garden extent does vary significantly, the first part of research question two.

## 7.4 Valuing private urban gardens and the impacts on residents

Research presented in Chapter Six of this thesis was conducted to answer research question three: how do the benefits within urban household gardens compare to those in public UGI, and are these benefits equally accessible (Section 1.4). This was explored by monetising the benefits of household gardens. In addition, it contributed substantially to the Urban Pioneer initiative (Figure 7.2), ensuring the inclusion of gardens as an important UGI in national environmental policy.

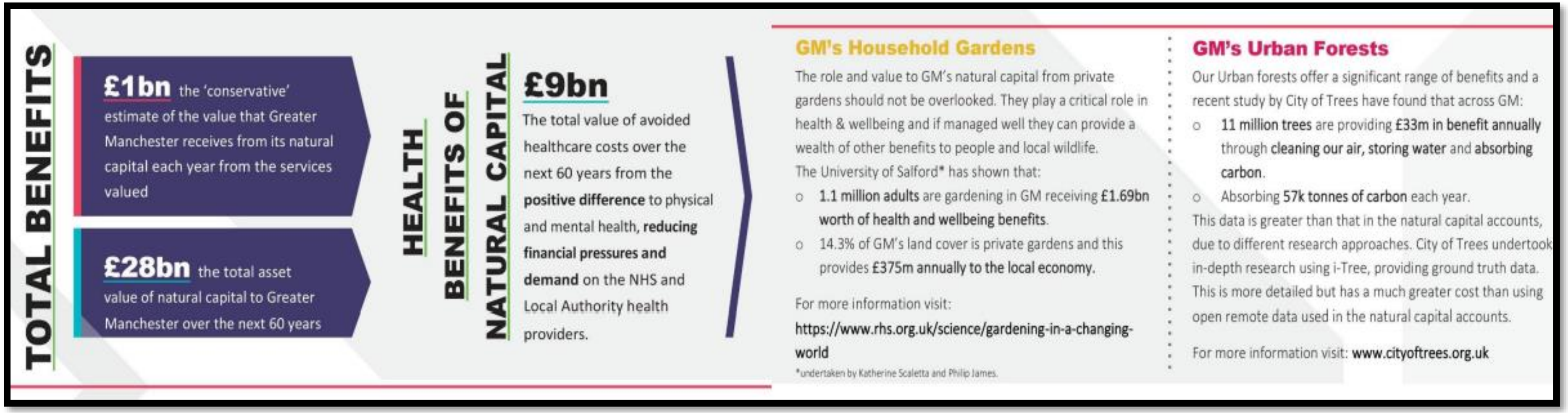



Figure 7.2: Greater Manchester's Household Gardens. Taken from The Natural Capital Approach in Greater Manchester (Urban Pioneer, 2019, p. 2).

Defra established The Urban Pioneer in Greater Manchester in 2018 to test deliverables of the 25-YEP in an urban setting. The Urban Pioneer set out to explore the links between the environment, society, and economy, focusing on improving the natural environment through improved decision making to support the health and wellbeing of GM's residents (Urban Pioneer, 2018). But, as with much of UK environmental policy, as discussed in Section 2.2, household gardens did not feature within the early strategic plan. This gap was highlighted, and part of the research presented in this thesis was conducted to fill this gap, providing data on the value and spatial extent of gardens in GM. Figure 7.2 is an extract from The Natural Capital Approach in GM and reports early estimates from this thesis monetising household gardens. These early estimates, along with data on garden extent in GM, provided an evidence base for the potential of urban gardens and their benefits to residents. As previously stated, the use and having a view of a garden is worth the equivalent of approximately 8 – 25 per cent of GM's health and social care budget (Section 6.2), illustrating how utilising gardens could save a significant amount of public funding.

Nature-based activities are increasingly gaining momentum as a cost-effective, easy, low-risk preventative and therapeutic intervention (van den Berg, 2017). Attempting to calculate a social return on investment (SROI) is frequently used when justifying numerous health and wellbeing measures or by stakeholders making recommendations (Bosco, Schneider, & Broome, 2019). For example, the Green Gym project estimate a return of £2.55 for every £1 (Yerrelll, 2008); health walks in Glasgow gave a return of £8 for every £1 invested (Carrick, 2013); and for a food growing community garden in London a return on investment of £3 for every £1 invested (Schoen, Caputo, & Blythe, 2020). But, there is a lack of SROI estimates within the literature in private UGI as private spaces exclude the social aspect needed for such investments. A major component of SROI valuations is the social aspect (Moron & Klimowicz, 2021) which gardens and gardening to not provide, at least not to the level of public UGI interventions. Therefore, SROI is not the most suitable valuing tool for gardens.

Data in this thesis shows investing in household gardens to be a very cost-effective way to improve residents' health and wellbeing. In Section 6.3 it is calculated that the average two adult household receives £612 of benefits but spends only £385 on gardening goods and services annually. Therefore, for every £1 spent by residents on their gardens the NHS is saving £1.60 in health and wellbeing benefits. Whilst this is a simplistic estimate it is interesting and highlights a valid resource which would be highly cost-effective for the NHS to exploit. Whilst the monetary estimates used in this thesis are secondary and taken from a publication over 10 years ago, the estimates still provide a valid baseline for the potential of household gardens. By using the estimates published by Mourato et al. (2010) it allows for a unique direct comparison between gardens and public greenspaces. For example, in the City of Salford the use of a garden is valued, most conservatively, at £21.4million and the use of an urban greenspace as £15.9million (Table 6.1). Despite the proportion of the adult population frequenting urban greenspaces (57 per cent) being higher than that which garden (50.2 per cent) the value of gardens is approximately 25 per cent greater.

Residents age has consistently been a key factor in their likelihood to garden as well gardening duration. Since the 1990s (Bhatti & Church, 2004) and shown more recently 2015 – 2019 (Figure 2.7) those over 65 years have had the highest participation rates in gardening when compared to other age groups. A similar trend is shown in garden access, with 65 years and over having the highest proportion of residents with access to a private garden at 86 per cent compared to 16–24 years the lowest at 70 per cent (Figure 2.11). But post COVID-19 pandemic almost three million people took up gardening as a new hobby in 2020, with nearly half under 45-years-old (Horticulture Week, 2020), suggesting new data due to be published in July 2023 by the People and Nature survey may show a spike in gardening in this age bracket.

Constrained City Dwellers and Multicultural Metropolitans were exposed as the two Supergroups most likely to experience environmental injustice due to their smaller, less diverse gardens (Section 4.5). The summary of these Supergroups in Table 4.1 highlights these both have high unemployment rates. Therefore, both the Supergroup household sample data (Chapter Four) and the combination of two national datasets (Section 2.7) show similar trends of environmental injustice. Figure 2.16 shows group E had the second smallest proportion of residents who consider their garden as important at 41 per cent, the smallest being group D at 38 per cent. For comparison group A reported 63 per cent of residents view their garden as important. Further, only 12 per cent of group E consider their garden as too small (Figure 2.17). These data suggest this socio-economic group is disinterested in their garden, yet they may be the group most in need of the health and wellbeing benefits highlighted in Section 2.6. A study in the Netherlands found unemployed persons to have a higher prevalence of psychological disorders (18.3 per cent vs. 5.4 per cent) and respiratory diseases (11.7 per cent vs. 6.5 per cent) compared to those in employment (Yildiz, Schuring, Knoef, & Burdorf, 2020). Plus, the COVID-19 pandemic has reportedly exasperated the injustices in healthcare (Public Health England, 2020). Therefore, the data suggest changing the attitudes of unemployed people towards their gardens may have the biggest effect on the general populations' health and wellbeing.

In conclusion, garden use was valued at £21.4million compared to £15.9million in the City of Salford annually (Table 6.1). In addition to the benefits of physically using a garden, the view of a garden can also have wellbeing benefits (Section 2.6) and was valued at £17million annually in the City of Salford (Table 6.1). The MENE and Taking Part datasets (Section 2.7) have shown gardens are not equally accessible to all, with gardening being more popular in residents over retirement age who are of White British ethnicity and within the least disadvantaged groups of society. These national data suggest a similar relationship to that presented from the household sample of the City of Salford, where it was found Multicultural Metropolitans and Constrained City Dwellers were evidenced to have smaller, less diverse gardens.

# Chapter Eight: Conclusions

## 8.1 Originality and contributions to knowledge

This thesis set out with the overall aim to undertake a critical exploration and analysis of the value of private gardens, with a key focus on the City of Salford, evaluating the size and content of urban household gardens to determine factors which contribute to the heterogeneity of the resource and their associated benefits. Resulting in four research objectives, from which three research questions (Section 1.4) were formed to be investigated in a deductive approach to the contribution of knowledge. Below (Table 8.1) is a brief overview of how the research presented has furthered the understanding of environmental injustice within private UGI in the field of urban ecology.

Table 8.1: A brief outline of the unique contributions to knowledge presented in this thesis.

| Section | Headline | Previous literature |
| --- | --- | --- |
| Chapters Four and Five | In total 6881 gardens were surveyed. | Sample sizes of similar studies: 250 (Gaston et al., 2005) and 267 (Loram, Warren & Gaston, 2008).<br><br>Those with a more large-scale approach have not recorded the level of detail in this thesis (Loram et al., 2007; Tratalos et al., 2007; Baker & Smith, 2019). |
| | Garden feature abundance is a key element of its quality and is different between OAC Supergroups. | Not previously reported.<br><br>Greenspace diversity was shown to be important in: Apparicio et al., 2016; Baker & Smith, 2019; Birch et al., (2020); Cameron et al., 2012; and Dillen et al., 2012. |
| Chapter Four | Multicultural Metropolitans and Constrained City Dwellers are likely to benefit least from their gardens. | Not previously reported. |

| | | |
|---|---|---|
| Section 5.2 | Collective garden cover ranges from 9.25-21.07 per cent in districts of GM. | Not previously reported. |
| **Section** | **Headline** | **Previous literature** |
| Section 5.3 | Salford's gardens are smaller than other urban regions in the UK. | Gaston et al., (2005) mean size 151 m². Loram et al., (2008) mean size 198 m². Davies et al., (2009) mean size 190 m². |
| | Salford's gardens have a larger cover of pave than previously recorded (mean 61 m²). | Not previously recorded for Salford. |
| | Salford's gardens have a smaller cover of lawn than previously recorded (mean 56 m²). | Not previously recorded for Salford. |
| | Salford's gardens have an approximately even split of lawn and pave, each covering 40 per cent of total land cover. | Loram et al., (2008) found considerably larger lawn cover 43 per cent than pave 28 per cent. Lawn cover was 60 per cent total land cover in Sheffield (Gaston et al., 2005). |
| | A typical Salford garden's landcover is: 44 per cent pave, 40 per cent lawn, 13 per cent trees & shrubs, and 3 per cent plants. | 11 per cent tree cover (Davies et al., 2009). 60 per cent lawn (Gaston et al., 2005) |
| Section 5.4 | Garden size has decreased in developments since the 1960s. | Garden size was smaller in Victorian terraced compared to modern terraced, detached, and semi-detached housing (Baker & Smith, 2019). |
| Section 6.2 | The use of a garden is valued at £21.4million compared to the use of a public UGI at £15.9million in the City of Salford. | Not previously reported. |
| | The view of a garden is valued at £17million in the City of Salford and | Valuations not previously reported. |

| | | |
|---|---|---|
| | £174million in Greater Manchester. | Health and wellbeing benefits of a greenspace view found in Chalmin-Pui et al., 2021; Hartig et al., (2003); Pretty et al., (2005). |

## 8.2 Limitations of research

As an important part of any research is reflection, the following section considers the limitations of this thesis and finally any recommendations for further research (Section 8.3). Firstly, there is a temporal limitation on the findings of this thesis. The household garden sample was conducted mainly throughout 2018, for consistency the monetary valuations were also calculated for the same year. This can be viewed as a limitation as the data are five years old. But the household garden sample was a lengthy data collection process to undertake and is still relevant for as long as the same OAC classifications hold. The OAC data used in this thesis was from 2011 as the study was undertaken in 2018, new OAC data were published in 2021 with the next 10-year census. Further, the conclusions drawn regarding the typology of different Supergroups gardens and the injustice exposed through this are still relevant if the socio-economic categories remain the same. As an extension of this thesis, a comparison between the OAC classifications in 2011 and 2021 would be interesting to test the level of variation in 10 years. For example, in the City of Salford the percentage of private renters rose from 19 per cent in 2011 to 27 per cent in 2021; and unemployment rose by nearly 3 per cent (ONS, 2023). An analysis into how these data have changed the configuration of the City of Salford's OAC Supergroups would be a good next step from this thesis.

The OAC is a three-tier classification, Supergroup, Group, and Subgroup. Data for the household sample were collected at Group level initially to gather socio-economic information at a smaller scale (as shown in Section 3.6). But the data size of a number of Groups was too small for analysis, as shown in Table 3.5 several Groups were only represented in the data by one OA sample. Apparent from the literature was the need for a large sample size to represent the heterogeneity within gardens, thus the larger Supergroup tier was chosen for analysis. Although this method did give a more accurate representation of the City of Salford's residents, there was still a disparity in the number of samples taken. For example, Cosmopolitans had 2 OAs sampled and Hard-Pressed Living had 18 (Table 3.5). The stratified sampling methodology did give an accurate representation of the City of

Salford, but if the data were to be scaled for another city more sampling within Supergroups with less OAs sampled would strengthen the findings.

To conduct the household garden sample, Google Earth was chosen as it provided a freely accessible database with a resolution lower than most other landcover data (Malarvizhi et al., 2016). From the data it was possible to record individual land cover which were split into four categories: trees and shrubs, plants, lawn, and pave. Count data for the number of trees and shrubs were also recorded. Data were collected when canopy cover would be highest to make recording most accurate, as outlined in Section 3.7. However, distinguishing between individual canopies was a limitation of this method. Where possible images were manipulated to differentiate between individual tree and shrub canopies. Comparison to previous research shows the data reported in this thesis may have underestimated the number of trees. Table 5.5 reports trees in 22 per cent of gardens, whilst Gaston et al. (2005) reported trees in 48 per cent of gardens in Sheffield and Davies et al. (2009) in 54 per cent. A further limitation was the inability to identify artificial grass, which may have been recorded as permeable grass by the researcher. Addressing these two limitations would require in-field surveys, which would have significantly reduced the number of gardens sampled overall. As a main aim of this thesis was to gather a large sample size, 6881 gardens sampled in total, these limitations were accepted as a small draw-back of the method chosen.

## 8.3 Future research

The two most at risk Supergroups for environmental injustice within gardens, Multicultural Metropolitans and Constrained City Dwellers, were exposed from this thesis. Next, a qualitative investigation of residents in these Supergroups could help to establish how to best advise stakeholders to increase nature engagement amongst these members of the community.

A next step on from this thesis could be a comparison of OAC classifications from 2011 (the data used in this thesis) and 2021 (the most research census). In the City of Salford, ONS (2023) reported an 8 per cent increase in the number of private renters and a rise of 3 per cent in unemployment in the 10 years between the two data. A comparison of how the demographics have changed within Supergroups and whether the configuration of

Supergroups in the City of Salford have changed would be a good next step to test how well the results in this thesis would withstand over time.

# References:


Adams, D., Tiesdell, S. (2012). Chapter 4: Real estate values and the state. In Taylor & Francis Ltd (Ed.), *Shaping Places: Urban Planning, Design and Development,* (pg. 64).

Agyeman, J. (2013). *Just Sustainability: Planning, Policy and Practice*. In Zed Books Ltd.

Agyeman. J. (2020). Local environment and the UN sustainable development goals. *Local Environment, 5*, 336-337.

Ainamani, H. E., Gumisiriza, N., Bamwerinde, W. M., Rukundo, G. Z. (2022). Gardening activity and its relationship to mental health: Understudied and untapped in low-and middle income counties. *Preventative Medicine Reports, 29*, Article 101946.

All-Party Parliamentary Group. (2022). Overcoming health inequalities in 'left behind' neighbourhoods. Retrieved 05/06/2023, from https://www.appg-leftbehindneighbourhoods.org.uk/publication/overcoming-health-inequalities-in-left-behind-neighbourhoods/

Allen, J., Balfour, R. (2014). Natural solutions for tackling health inequalities. Institute of Health Equity.

Ambrose, G., Das, K., Fan, Y., Ramaswami, A. (2020). Is gardening associated with greater happiness of urban residents? A multi-activity, dynamic assessment in the Twin-Cities region, USA. *Landscape and Urban Planning, 198*, Article 103776.

Andersson, M., A. (2016). Health returns to education by family socioeconomic origins 1980-2000: testing the importance of gender, cohort, and age. *SSM Population Health, 2*, 549-560.

Angle, J. (1983). The surplus theory of social stratification and the size distribution of personal wealth. *Social Forces, 65(2), 293-326.*

Armson, D., Stringer, P., Ennos, A. R. (2013). The effect of street trees and amenity grass on urban surface water runoff in Manchester, UK. *Urban Forestry & Urban Greening, 12*(3), 282-286.

Arshad, H. S. H., Routray, J. K. (2018). From socioeconomic disparity to environmental injustice: the relationship between housing unit density and community green space in a medium city in Pakistan. *The International Journal of Justice and Sustainability,* 23(5), 536-548.

Aschner, M., Paoliello, M. M. B., Tsatsakis, A., Bowman, A. B., Dorea, J. G., Hartung, T., Domingo, J. L., Barbosa, S. (2021). Social injustice in environmental health: A call for fortitude. *Environmental Research, 194*, Article 110675.

Astell-Burt, T., Feng, X., Mavoa, S., Badland, H. M., Giles-Corti, B. (2014). Do low-income neighbourhoods have the least greenspace? A cross-sectional study of Australia's most populous cities. *BMC Public Health, 14*(292), 1-11.

Apparicio, P., Pham, T. T. H., Seguin, A. M., Dube, J. (2016). Spatial distribution of vegetation in and around city block on the Island of Montreal: A double environmental equity? *Applied Geography, 76*, 128-136.

Bagstad, K. J., Johnson, G. W., Voigt, B. Villa, F. (2013). Spatial dynamics of ecosystem service flows: a comprehensive approach to quantifying actual services. *Ecosystem Services, 4,* 117-125.

Baker, F., Smith, C. L., Cavan, G. (2018). A combined approach to classifying land surface cover of urban domestic gardens using citizen science data and high resolution image analysis. *Remote Sensing, 10*, 1-23.

Baker, F., Smith, C. (2019). A GIS and object based image analysis approach to mapping the greenspace composition of domestic gardens in Leicester, UK. *Landscape and Urban Planning,* 183, 133-146.

Barbosa, O., Tratalos, J. A., Armsworth, P. R., Davies, R. G., Fuller, R. A., Johnson, P., Gaston, K.J. (2007). Who benefits from access to greenspace? A case study from Sheffield, UK. *Landscape and Urban Planning, 83*, 187–195.

Barger, M. M., Perez, T., Canelas, D. A., Linnenbrink-Garcia, L. (2018). Constructivism and personal epistemology development in undergraduate chemistry students. *Learning and Individual Differences,* 63, 89-101.

Barthel, S., Folke, C., Colding, J. (2010). Social-ecological memory in urban gardens – retaining the capacity for management of ecosystem services. *Global Environmental Change, 20*, 255-265.

Baro, F., Chaparro, L., Gomez-Baggethun, E., Langmeyer, J., Nowak, D., J., Terradas, J. Contribution of ecosystem services to air quality and climate change mitigation policies: the case of urban forests in Barcelona, Spain. *Ambio, 43, 466-479.*

Basch, C. E. (2011). Asthma and the achievement gap among minority youth. *Journal of School Health,* 81, 606-613.

Barton, C., Wilson, W., Booth, L. (2022). Tackling the under-supply of housing in England. Retrieved 10/04/2023, from https://commonslibrary.parliament.uk/research-briefings/cbp-7671/

BBC. (2016). NHS spending devolved in Greater Manchester. Retrieved 13/10/2020, from https://www.bbc.co.uk/news/uk-england-manchester-35933922

BBC (2020). Sheffield trees: Council tried to 'avoid scrutiny' over scheme. Retrieved 18/10/2021, from https://www.bbc.co.uk/news/uk-england-south-yorkshire-54532066

BBC. (2021). What is Black Lives Matter and what are the aims? Retrieved 20/07/2022, from https://www.bbc.co.uk/news/explainers-53337780

Bhatti, M., Church, A. (2004). Home, the culture of nature and meanings of gardens in late modernity. *Housing Studies, 19*(1), 37-51.

Belcher, R. N., Chrisholm, R. A. (2018). Tropical vegetation and residential property value: a hendoic pricing analysis in Singapore. *Ecological Economics, 149*, 149-159.

Bell, S., White, M., Griffiths, A., Darlow, A., Taylor, T., … Lovell, R. (2020). Spending time in the garden is positively associated with health and wellbeing: Results from a national survey in England. *Landscape and Urban Planning, 200*, Article 103836.

Burt, K. G., Mayer, G., Paul, R. (2021). A systematic, mixed studies review of the outcomes of community garden participation related to food justice. *Local Environment,* 26(1), 17-42.

Bertram, C., Rehdanz, K. (2015). The role of urban greenspace for human well-being. *Ecological Economics*, *120*, 139-152.

Bibby, P., Henneberry, J., Halleux, J., M. (2018). Under the radar? Soft residential densification in England, 2001-2011. *Environment and Planning B: Urban Analytics and City Science, 47(1), 102-118.*

Bigirimana, J., Bogaert, J., De Canniere, C., Bigendako, M., J., Parmentier, I. (2012). Domestic garden plant diversity in Bujumbura, Burundi: Role of the socio-economical status of the neighbourhood and alien species invasion risk. *Landscape and Urban Planning, 107*, 118-126.

Bijker, R. A., Sijtsma, F. J. (2017). A portfolio of natural places: Using a participatory GIS tool to compare the appreciation and use of greenspaces inside and outside urban areas by urban residents. *Landscape and Urban Planning*, *158*, 155-165.

Birch, J., Rishbeth, C., Payne, R. S. (2020). Nature doesn't judge you – how urban nature supports young people's mental health and wellbeing in a diverse UK city. *Health & Place,* 62, Article 102296.

Bisgrove, R., Hadley, P. (2002). Gardening in the global greenhouse: the impacts of climate change on gardens in the UK. Technical Report. UK climate impacts programme, Oxford, UK.

Bonham, C., Williams, S., Grimstead, I., Ricketts, M. (2019). Greenspaces in residential gardens. Data Science for Public Good. Data Science Campus. Retrieved, 02/08/19 from https://datasciencecampus.ons.gov.uk/projects/green-spaces-in-residential-gardens/

Brander, L. M., Koetse, M. J. (2011). The value of urban open space: Meta-analyses of contingent valuation and hedonic pricing results. *Journal of Environmental Management, 92,* 2763-2773.

Brazier, J., Roberts, J. and Deverill, M. (2002) The estimation of a preference-based measure of health from the SF-36. *Journal of Health Economics, 21*, 271–292.

Britannica. (n.d.). Cities & Towns: Salford. Retrieved 10/6/21, from https://www.britannica.com/place/Salford-England

Brindley, P., Jorgensen, A., Maheswaran, R. (2018). Domestic gardens and self-reported health: a national population study. *International Journal of Health Geographics,* 17(31), 1-11.

Brown, S., Lombard, J., Wang, K., … Szapocznik, J. (2018). Neighbourhood greenness and chronic health conditions in medicare beneficiaries. *American Journal of Preventive Medicine, 51*(1), 78-89.

Brookes, A., Francis, R. A. (2019). Artificial lawn people. Environment and Planning E: Nature and Space, *2(3),* 548-564.

Boone, C. G., Buckley, G. L., Grove, J. M., Sister, C. (2009). Parks and people: an environmental justice inquiry in Baltimore, Maryland. *People, Place, Religion, 99*(4), 767-787.

Bosco, A., Schneider, J., Broome, E. (2019). The social value of the arts for care home residents in England: A social return on investment (SROI) analysis of the Imagine Arts Programme. *Maturitas,* 124, 15-24.

Calderon-Algelich, A., Benetti, S., Anguelovski, I., Connolly, J. J. T., Langemeyer, J.,M Baro, F. (2021. Tracing and building up environmental justice considerations in the urban ecosystem service literature: A systematic review. *Landscape and Urban Planning,* 214, Article 104130.

Cambridge Dictionary. (n.d.). Meaning of socioeconomic in English. Retrieved 03/04/2020, from https://dictionary.cambridge.org/dictionary/english/socioeconomic

Cameron, R. W. F., Blanusa, T., Taylor, J. E., Salisbury, A., Halstead, A. J., Henricot, B., Thompson, K. (2012). The urban garden – Its contribution to urban green infrastructure. *Urban Forestry & Urban Greening*, *11*, 129-137.

Carey, G. W., Harvey, D. (1975). Social justice and the city. *Geographical Review,* 65(3), 421.

Carrick, K. (2013). Glasgow Health Walks Social Return on Investment Analysis Alloa: Paths for All. Retrieved 02/02/2023, from http://www.pathsforall.org.uk/pfa/health-walks/our-research.html

Carrus, G., Scopelliti, M., Lafortezza, R., Colangelo, G., Ferrini, F., … (2015). Go greener, feel better? The positive effects of biodiversity on the well-being of individuals visiting urban and peri-urban green areas. *Landscape and Urban Planning, 134, 221-228.*

Cavan, G., Smith, C., Tzoulas, K., Baker, F., Barlow, D., Hall, J., Reece, P., Stringer, P., Butlin, T. (2018). My Backyard. An Action Plan to increase greenspace and enhance wildlife in domestic gardens across Manchester.

Centres for Disease Control and Prevention. (2011). Vital signs: asthma prevalence, disease characteristics and self-management education: United States 2001-2009. Retrieved 26/03/2020, from https://www.ncbi.nlm.nih.gov/pubmed/21544044

Camps-Calvet, M., Langmeyer, J., Calvet-Mir, L., Gomez-Baggethun, E. (2016). Ecosystem services provided by urban gardens in Barcelona, Spain: Insights for policy and planning. *Environmental Science & Policy, 62, 14-23.*

Certoma, C., Martellozzo, F. (2019). Cultivating urban justice? A spatial exploration of urban gardening crossing spatial and environmental injustice conditions. *Applied Geography,* 106, 60-70.

Chalmin-Pui, L. S., Griffiths, A., Roe, J. J., Cameron, R. W. (2019). Bringing fronts back: A research agenda to investigate the health and well-being impacts of front gardens. *Challenges, 10(2),* https://doi.org/10.3390/challe10020037

Chalmin-Pui, L. S., Griffiths, A., Roe, J., Heaton, T., Cameron, R. (2021). Why garden? – Attitudes and the perceived health benefits of home gardening. *Cities,* 112, Article 103118.

Chalmin-Pui, L. S., Roe, J., Griffiths, A., Smyth, N., Clayden A., Cameron R. (2021). 'It made me feel brighter in myself' – The health and well-being impacts of a residential front garden horticultural intervention. *Landscape and Urban Planning, 205, Article 103958.*

Chamberlain, D. E., Henry, D. A. W., Reynolds, C., Caprio, E., Amar, A. (2019). The relationship between wealth and biodiversity: A test of the Luxury Effect on bird species richness in the developing world. *Global Change Biology, 25*(9), 3045-3055.

Chaparro, M. P., Benzeval, M., Richardson, E., Mitchell, R. (2018). Neighbourhgood deprivation and biomarkers of health in Britain: the mediating role of the physical environment. *BMC Public Health,* 18, 801.

Chen, E., Hanson, M. D., Paterson, I. Q., Griffin, M. J., Walker, H. A., Miller, G. E. (2006). Socioeconomic status and inflammatory processes in childhood asthma: the role of psychological stress. *Journal of Allergy and Clinical Immunology,* 117, 1014-1020.

Chen, T. K., Horsdal, H. T., Samuelson, K., Closter, A. M., Davies, M., Barthel, S., Pederson, C. B., Prishchepov, A. V., Sabel, C. E. (2023). Higher depression risks in medium- than in high – density urban form across Denmark. *Science Advances,* 9(21), doi: 10.1126/sciadv.ad3760

Chiesura, A. (2004). The role of urban parks for the sustainable city. *Landscape and Urban Planning*, *68*(1), 129-138.

Christensen, S., Malberg Dyg, P., Allenbergm K. (2018). Urban community gardening, social capital, and 'integration' – a mixed method exploration of urban 'integration-gardening' in Copenhagen, Denmark. *Local Environment,* 24(3), 231-248.

Church, A., Fish, R., Haines-Young, R., Mourato, S., Tratlos, J. (2014). UK national ecosystem assessment follow-on. Work package report 5: Cultural ecosystem services and indicators. Retrieved, 24/02/22, from http://uknea.unep-wcmc.org/Resources/tabid/82/Default.aspx

Church, A., Burgess, J., Ravenscroft, N. (2011). Chapter 16: Cultural Services. UK National Ecosystem Assessment.

Cilliers, S., Cilliers, J., Lubbe, R., Siebert, S. (2013). Ecosystem services of urban greenspaces in African countries-perspectives and challenges. *Urban Ecosystems, 16(4),* 681-702.

Cilliers, S., Siebert, S., Davoren, E., Lubbe, R. (2012). Chapter 10: Social aspects of urban ecology in developing countries, with an emphasis on urban domestic gardens. In Ritcher, M. & Weiland, U (Eds.), *Applied Urban Ecology: A Global Framework*, pg. 123-133.

City of Trees. (2018). All our trees: Greater Manchester's Tree and Woodland Strategy. Retrieved 05/11/2020, from https://www.cityoftrees.org.uk/allourtrees

City of Trees. (n.d.). About City of Trees. Retrieved 02/02/2022, from https://www.cityoftrees.org.uk/about-city-trees

City of Trees. (n.d.) Climate emergency. Retrieved 16/11/2022, from https://www.cityoftrees.org.uk/climate-emergency

Citypopulation (2020). Major agglomerations of the world. Retrieved 26/08/2020, from https://www.citypopulation.de/en/world/agglomerations/

Clark, H. (2021). Examining the end of the furlough scheme. Retrieved 20/06/2022, from https://commonslibrary.parliament.uk/examining-the-end-of-the-furlough-scheme/

Clayton, S. (2007). Domesticated nature: Motivations for gardening and perceptions of environmental impact. *Journal of Environmental Psychology*, *27*, 215-224.

Cumbers, A., Shaw, D., Crossan, J., McMaster, R. (2017). The work of community gardens: Reclaiming place for community in the city. *Work, Employment and Society, 32*(1), 133-149.

Cook, P. A., Howarth, M., Wheater, C. P. (2019). Biodiversity and Health in the face of climate change: Implications for public health. Springer Link. Pg. 262

Colding, J. (2007). 'Ecological land-use complementation' for building resilience in urban ecosystems. *Landscape and Urban Planning, 81*, 46-55.

Conway, T., Hackworth, J. Urban pattern and land cover variation in the Greater Toronto area. *The Canadian Geographer, 51*(1), 43-57.

Corley, J., Okley, A, J., Taylor, M. A., Page, D., Welstead, M., … Russ, C. S. (2021). Home garden use during Covid-19: Associations with physical and mental wellbeing in older adults. *Journal of Environmental Psychology,* 73, Article 101545.

Corporate Finance Institute. (2021). Hedonic Pricing. Retrieved 15/06/2022, from https://corporatefinanceinstitute.com/resources/knowledge/valuation/hedonic-pricing/

Costanza, R. (2008). Ecosystem services: multiple classification systems are needed. *Biological Conservation,* 141, 350–352.

Cowell, R. (2018). Can Brexit bring a better way to protect the environment? Brexit & Environment. Retrieved 02/08/19, from https://www.brexitenvironment.co.uk/2018/09/11/can-brexit-bring-better-way-protect-environment/

Cox, T. C. D., Shanahan, F. D., Hudson, L. H., Plummer, E. K., Siriwardena, M. G., … Gaston, J. K. (2017). Doses of neighbourhood nature: the benefits for mental health of living with nature. *BioScience,* 67(2), 147-155.

Da Silva, I. M., Fernandes, C. O., Castiglione, B., Costa, L. (2016). Characteristics and motivations of potential users of urban allotment gardens: The case of Vila Nova de Gaia municipal network of urban allotment gardens. *Urban Forestry and Urban Greening, 20*, 56-64.

Davies, Z. G., Fuller, R. A., Loram, A., Irvine, K. N., Sims, V., Gaston, K. J. (2009). A national scale inventory of resource provision for biodiversity within domestic gardens. *Biological Conservation*, *142*(4), 761-771.

Daniels, G. D., Kirkpatrick, J. B. (2006). Comparing the characteristics of front and back domestic gardens in Hobart, Tasmania, Australia. *Landscape and Urban Planning, 78*, 344-352.

Daniels, G. D., Kirkpatrick, J. B. (2006). Does variation in garden characteristics influence the conservation of birds in suburbia? *Biological Conservation, 133*, 326-355.

Dayson, C., Bashir, H., Bennett, E., Sanderson E. (2016). The Rotherham social prescribing service for people with long-term health conditions. Annual evaluation report. Sheffield Hallam University Centre for regional, social and economic research.

Dearden, E. K., Lloyd, C. D., Catney, G. (2019). A spatial analysis of health status in Britain, 1991 -2011. *Social Science & Medicine,* 220, 340-352.

Dennis, M., Barlow, D., Cavan, G., Cook, P. A., Gilchrist, A., Handley, J., James, P., Thompson J., Tzoulas, K., Wheater, P., Lindley, S. (2018). Mapping urban green infrastructure: A novel landscape-based approach to incorporating land use and land cover in the mapping of human-dominated systems. *Land, 7,* 1-25.

Dennis, M., Scaletta, K. L., James, P. (2019). Evaluating urban environmental and ecological landscape characteristics as a function of land-sharing-sparing, urbanity, and scale. *PLOS One, 14*(7), 1-24.

Denscombe, M. (2014). *Good Research Guide: For small-scale social research projects.* 5th ed. Retrieved from, https://ebookcentral-proquest-com.salford.idm.oclc.org/lib/salford/detail.action?docID=1910221#

Department for Digital, Culture, Media & Sport (DCMS). (2017). Ad-hoc statistical analysis:2017/18 Quarter 3. Adult Participation in gardening in the North West, 2015/16. Retrieved, 01/11/17, from, https://www.gov.uk/government/statistical-data-sets/ad-hoc-statistical-analysis-201718-quarter-3#november-2017---adult-16-participation-in-gardening-in-the-north-west-201516

Department for Digital, Culture, Media & Sport (DCMS). (2021). Taking Part Survey: Update to the Taking Part Survey. Retrieved 08/05/2023, from https://www.gov.uk/guidance/taking-part-survey

Department of Food and Rural Affairs (DEFRA). (2018). A Green Future: Our 25 Year Plan to Improve the Environment. Retrieved, 01/09/2019, from https://www.gov.uk/government/publications/25-year-environment-plan

Department of Food and Rural Affairs (DEFRA). (2019). 25 Year Environment Plan Progress Report. January 2018-March 2019. Retrieved, 01/09/2019, from https://www.gov.uk/government/publications/25-year-environment-plan-progress-reports

Department of Food and Rural Affairs (DEFRA). (2021). 25 year environment plan annual progress report. April 2020-March 2021. Retrieved 11/04/2023, from https://www.gov.uk/government/publications/25-year-environment-plan-progress-reports

Department of Food and Rural Affairs (DEFRA). (2022). 25 year environment plan annual progress report. April 2021-March 2022. Retrieved 11/04/2023, from https://www.gov.uk/government/publications/25-year-environment-plan-progress-reports

Dewaelheyns, V., Rogge, E., Gulinck, H. (2014). Putting domestic gardens on the agenda using empirical spatial data: The case of Flanders. *Applied Geography, 50*, 132-143.

Douglas, O., Lennon, M., Scott, M. (2017). Greenspace benefits for health and wellbeing: A life-course approach for urban planning, design, and management. *Cities, 66*, 53-62.

Douglas, O., Russell, P., Scott, M. (2019). Positive perceptions of green and open space as predictors of neighbourhood quality of life: Implications for urban planning across the city region. Journal of Environmental Planning and Management, 62(4), 626–646.

Dillen, S. M. V., Vries, S. D., Groenewegen, P. P., Spreeuwenberg, P. (2012). Greenspace in urban neighbourhoods and residents' health: adding quality to quantity. *Epidemiology & Community Health, 66*(6), 1-5.

Druckman, A., Sinclair, P., Jackson, T. (2008). A Geographically and socio-economically disaggregated local household consumption model for the UK. *Journal of Cleaner Production,* 16, 870-880.

Drummond, M., O'Brien, B., Stoddart, G. and Torrance, G. (1997) Methods for the economic evaluation of health care programmes (2nd ed). Oxford: Oxford University press.

Dunnett, N., Qasim, M. (2000). Perceived benefits to human well-being of urban gardens. *HortTechnology, 10*(1), *40-45.*

Dzhambov, M. A., Lercher, P., Browning H. E. M. M., Stoyanov, D., Petrova, N., Novakov, S., Dimitrova, D. D. (2021). Does greenery experienced indoors and outdoors provide an escape and support mental health during the Covid-19 quarantine? *Environmental Research,* 196, Article 110420.

Edina Digimap. (n.d.). Ordnance Survey. Retrieved, 15/12/2020, from Digimap (edina.ac.uk)

Ellis, J. B., D'Arcy, B. J., Chatfield, P. R. (2007). Sustainable Urban-Drainage Systems and Catchment Planning. *Water and Environment Journal, 16*(4), 286-291.

Environment Agency. (2008). Permeable surfacing of front gardens: guidance. Retrieved, 05/09/2019, from https://www.gov.uk/government/publications/permeable-surfacing-of-front-gardens-guidance

Environment Agency. (2020). Review of the impact of the Urban Pioneer. Retrieved 12/01/22, from https://democracy.greatermanchester-ca.gov.uk

Essex Planning Officers Association. (2018). The Essex design guide: garden size. Retrieved 13/04/2023, from https://www.essexdesignguide.co.uk/design-details/architectural-details/garden-size/

*European Environment Agency. (2021). Urban Green Infrastructure. Retrieved 15/03/2023, from* https://www.eea.europa.eu/data-and-maps/dashboards/urban-green-infrastructure-2018

*Eskandari, S., Sarab, S. A. M. (2022). Mapping land cover and forest density in Zagros forests of Khuzestan province in Iran: A study based on Sentinel-2, Google Earth and field data. Ecological Informatics, 70, Article: 101727.*

Evening Standard. (2022). Comment: millennials are buying fewer homes – we wouldn't have to care if the alternatives were better. Retrieved 13/09/22, from https://www.standard.co.uk/homesandproperty/property-news/millennials-buying-homes-property-affordable-housing-policy-house-prices-b992410.html

Experian Mosaic. (n.d.). Mosaic United Kingdom: The consumer classification of the United Kingdom. Retrieved 08/02/2020, from https://www.experian.co.uk/assets/business-strategies/brochures/Mosaic%2520UK%25202009%2520brochure%5B1%5D.pdf

Exposito, A., Espinosa, M., Villa-Damas, A. (2021). Valuing visitor willingness to pay for urban green space conservation: case of Maria Luisa Park in Seville, Spain. *Journal of Urban Planning and Development, 147(3), doi:10.1061*

Fancourt, D., Baxter, L. (2020). Differential partipation in community cultural activites amongst those with poor mental health: Analyses of the UK Taking Part Survey. *Social Science & Medicine, Article 113221.*

Ferrari, E., Dankowska, A., Dragon, A., Hassee, A., Kronenberg, J., Hasse, D. (2023). Towards an integrated garden. Gardeners of all types, unite! *Urban Forestry & Urban Greening, 81, Article 127857.*

Fields in Trust. (2018). Revaluing Parks and Green Spaces. Retrieved 08/02/2020, from http://www.fieldsintrust.org/News/research--new-research-shows-uk-parks-and-green-spaces-generate-over-£34-billion

Financial Times. (2022). Why are there so few houses for sale in the UK? Retrieved 13/09/22, from https://www.ft.com/content/3d6c7112-c527-4a62-a964-db6b3d39c886

Fincher, R., Iveson, K. (2012). Justice and injustice in the city. *Geographical Research, 50(*3), 231-241.

Fischer, B., Telser, H., Zweifel, P., Wyl, V., Beck, K., Weber, A. (2023). The value of QALY towards the end of life and its determinants: Experimental evidence. *Social Science & medicine, 326,* Article 115909.

Fixton, A., Barrett, S. (2022). Challenged and approaches to green social prescribing during and in the aftermath of COVID-19: A qualitative study. *Frontiers in Psychology, 13,* doi: 10.3389/fpsyg.2022.861107

Fjaestad, S. L., Mackelprang, J. L., Sugiyama, T., Chandrabose, M., Owen, N., Turrell, G., Kingsley, J. (2023). Associations of time spent gardening with mental wellbeing and life satisfaction in mid-to-late adulthood. *Journal of Environmental Psychology,* doi: https://doi.org/10.1016/ j.jenvp.2023.101993.

Fong, E. (1994). Residential proximity among racial groups in US and Canadian neighbourhoods. *Urban Affairs Review,* 30(2), 285-297.

Foody, G. M. (2008). Harshness in image classification accuracy assessment. *International Journal of Remote Sensing, 29*, 3137-3158.

Francis, R. A. (2018). Artificial lawns: Environmental and societal considerations of an ecological simulacrum. Urban Forestry & Urban Greening, 30, 152-156.

Fuller, R. A., Irvine, K. N., Davies, Z. G., Armsworth, P. R., Gaston, K. J. (2012) Interactions between people and birds in urban landscapes. Urban bird ecology and conservation. *Studies in avian ecology, 45*, 249-266.

Fuller, R. A., Irvine, K. N., Devine-Wright, P., Warren P. H., Gaston, J. K. (2007). Psychological benefits of greenspace increase with biodiversity. *Biology Letters,* 3(4), 390-394.

Gale, C. G., Singleton, A. D., Bates, A. G., Longley, P. A. (2016). Creating the 2011 area classification for output areas (2011 OAC). *Journal of Spatial Information Science,* 12, 1-27.

Garcia-Antunez, O., Lindgaard, J., Lampinen, J., Olafsson, A. S. (2023). Gardening for wildlife: A mixed-methods exploration of the factors underlying engagement in wildlife-friendly gardening. *People & Nature, 00, 1-18, doi: 10.1002/pan3.10450*

Gaston, K. J., Warren, P. H., Thompson, K., Smith, R. S. (2005). Urban domestic gardens: the extent of the resource and its associated features. *Biodiversity and Conservation*, *14*, 3327-3349.

GHIA. (n.d.). Interactive map of results. Retrieved 23/01/2020, from http://maps.humanities.manchester.ac.uk/ghia-web/extract/extract.html

Gibson, K., Pollard, T. M., Moffatt, S. (2021). Social prescribing and classed inequality: A journey of upward health mobility? *Social Science & Medicine,* 280, Article 114037.

Giezen, M., Pellerey, V. (2021). Renaturing the city: Factors contributing to upscaling green schoolyards in Amsterdam and The Hague. *Urban Forestry & Urban Greening, 63,* Article 127190.

Gill, S. E., Handley, J. F., Ennos, A. R., Pauleit, S. (2007). Adapting cities for climate change: the role of green infrastructure. *Built Environment, 33*, 115-133.

Gill, S. E., Handley, J. F., Ennos, A. R., Pauleit, S., Theuray, N., Lindley, S. J. (2008). Characterising the urban environment of UK cities and towns: A template for landscape planning. *Landscape and Urban Planning, 87*, 210-222.

Gomez-Baggethun, E., Barton, D., N. (2013). Classifying and valuing ecosystem services for urban planning. *Ecological Economics,* 86, 235-245.

Gonzalez, M. T., Hartig, T., Patil, G., Martinsen, E. W., Kirkevold, M. (2010). Therapeutic horticulture in clinical depression: A prospective study of active components. *Journal of Advanced Nursing, 66*(9), 2002-2013.

Guerry, A. D., Polasky, S., Lubchenco, J., Chaplain-Kramer, R., Daily, G. C., … Vira, B. (2015). Natural capital and ecosystem services informing decisions: From promise to practice. *Proceedings of the national academy of sciences of United States of America.* 112(24), 7348-7355.

Gustafson, P. (2001). Meanings of place: Everyday experiences and theoretical conceptualizations. *Journal of Environmental Psychology, 21*, 5-16.

Goddard, M. A., Dougill, A. J., Benton, T. G. (2013). Why garden for wildlife? Social and ecological drivers, motivations and barriers for biodiversity management in residential landscapes. *Ecological Economics, 86*, 258-273.

Government Commercial Function. (2020). The Social Value Model. Retrieved 22/21/22, from https://assets.publishing.service.gov.uk

Gov.uk. (2021). Writing about ethnicity. Retrieved 23/06/2023, from https://www.ethnicity-facts-figures.service.gov.uk/style-guide/writing-about-ethnicity

Grafius, D. R., Corstanje, R., Warren, P. E., Evans, K. L., Hancock, S., Harris, J. A. (2016). The impact of land use/land cover scale on modelling urban ecosystem services. *Landscape Ecology, 31,* 1509-1522.

Grahn, P., Stigsdotter, U. (2003). Landscape planning & stress. *Urban Forestry & Urban Greening, 2*(1), 1-18.

Greater Manchester Combined Authority. (2016a). The Greater Manchester Population Health Plan. Retrieved 13/10/2020, from https://www.gmhsc.org.uk/wp-content/uploads/2018/05/Population-Health-Plan-2017-2021.pdf

Greater Manchester Combined Authority. (2016b). Greater Manchester Mental Health and Wellbeing Strategy. Retrieved 13/10/2020, from https://www.gmhsc.org.uk/wp-content/uploads/2018/04/GM-Mental-Health-Summary-Strategy.pdf

Greater Manchester Combined Authority. (2019). 5-year Environment Plan for Greater Manchester 2019-2014. Retrieved 12/01/22, from https://www.greatermanchester-ca.gov.uk/what-we-do/environment/five-year-environment-plan/

Greater Manchester Combined Authority. (2019b). Greater Manchester Housing Strategy 2019-2024. Retrieved 27/01/22, from from https://www.greatermanchester-ca.gov.uk

Greater Manchester Combine Authority. (2020). Greater Manchester leaders unveil vision for post-Covid economy. Retrieved 22/01/22, from https://democracy.greatermanchester-ca.gov.uk/documents/s19940/09%20Green%20Spaces%20Fund%20final.pdf

Greater Manchester Combine Authority. (2022). GM Green City Region Partnership: Greater Manchester Green Spaces Fund. Retrieved 06/01/2023, from https://www.democracy.greatermanchester-ca.gov.uk

Greater Manchester Poverty Action. Index of Multiple Deprivation. Retrieved 10/01/2023, from https://www.gmpovertyaction.org/pm2022-imd/

Grove, J. M., Troy, A. R., O'Neil-Dunne, J. P. M., Burch, W. R., Cadenasso, Jr. M. L., Pickett, S. T. A. (2006). Characterization of households and its implications for the vegetations of urban ecosystems. *Ecosystems,* 9, 578-597.

Gurgatz, B. M., Carvalho-Oliveira, D., Joucoski, E., Antoniaconi, G., Saldiva, P. H. N., Reis, R. A. (2016). Atmospheric metal pollutants and environmental injustice: A methodological approach to environmental risk analysis using fuzzy logic and tree bark. *Ecological Indicators,* 71, 428-437.

Haase, D., Janicke, C., Wellmann, T. (2019). Front and back yard analysis with subpixel vegetation fractions from earth observation data in a city. *Landscape and Urban Planning*, 182, 44-54.

Haase, D. (2019). The rural-to-urban gradient and ecosystem services. *Atlas of Ecosystem Services,* 141-146.

Haller, T., Crole-Rees, A., Dumondel, M. (2013). Attitudes towards growing food in cities: the case of Lausanne, Switzerland. *Journal of Socio-Economics, 6*, 258-273.

Handley, J., Carter, J. (2006). Adaptation strategies for climate change in the urban environment. National Steering Group. University of Manchester. ESPRC Report GR/S 19233/01.

Harris, K., M. (2019). Mapping inequality: Childhood asthma and environmental injustice, a case study of St Louis. Missouri. *Social Science & Medicine,* 230, 91-110.

Hartig, T., Evans, G. W., Jamner, L. D., Davis, D. S., Garling, T. (2003). Tracking restoration in natural and urban field settings. *Journal of Environmental Psychology*, *23*, 109-123.

Haughton, G., Hunter, C. (2003). Sustainable Cities. London: Taylor & Francis.

He, B., Zhu, J. (2018). Constructing community gardens? Residents' attitudes and behaviour towards edible landscapes in emerging urban communities of China. *Urban Forestry & Urban Greening, 34*, 154-165.

Hernandez, M., Collins, T. W., Grineski, S. E. (2015). Immigration, mobility, and environmental injustice: A comparative study of Hispanic people's residential decision making and exposure to hazardous air pollutants in Greater Houston, Texas. *Geoforum,* 60, 83-94.

Hessayon, D., G., Hessayon, J., P. (1973). *The Garden Book of Europe*. Elm Tree Books, London.

Hinds, J., Camic, P. M. (2013). Gardening as a mental health intervention: A review. *Mental health Review Journal, 18*(4), 214-225.

Hines, A., B. (2011). Asthma: a health disparity among African American children. The impact and implications for paediatric nurses. *Journal of Paediatric Nursing, 26, 25-33.*

HM Revenue & Customs (HMRC). (2023). CG64360 – Private residence relief: garden and grounds: definitions. Retrieved 21/02/23, from https://www.gov.uk/hmrc-internal-manuals/capital-gains-manual/cg64360

Hope, D., Gries C., Zhu, W., Fagan W. F., Redman, C. L., Grimm, N. B., Nelson, A. L., Martin, C., Kinzig, A. (2003). Socioeconomics drive urban plant diversity. *PNAS*, *100*(15), 8788-8792.

Horticultural Trades Association. (2018). *The economic impact of ornamental horticulture and landscaping in the UK.* Oxford: Oxford Economics.

Horticultural Trades Association. (n.d.) Garden industry welcomes Defra 25-Year Environment plan. Retrieved, 13/01/2020, from https://hta.org.uk/news/hta-welcomes-defra-25-year-environment-plan.html

Horticulture Week. (2020). Covid-19 has transformed the gardening industry – so what are the new trends? Retrieved 22/01/22, from https://www.hortweek.com/covid-19-transformed-gardening-industry-%E2%80%93-so-new-trends/retail/article/1701037

Houlden, V., Weich, S., Jarvis, S. (2017). A cross-sectional analysis of green space prevalence and mental wellbeing in England. *BMC Public Health*, 17, Article 460.

Houlden, V., Albuquerque, J. P., Weich, S., Jarvis, S. (2019). A spatial analysis of proximate greenspace and mental wellbeing in London. *Applied Geography, 109*, Article 102036.

Howarth, M., Lawler, C., de Silva, A. (2021). Creating a transformative space for change: A qualitative evaluation of the RHS Wellbeing Programme for people with long term conditions. *Health and Place,* 71, Article 102654.

Howarth, M., Rogers, M., Withnell, N., McQuarrie, C. (2019). Growing spaces: An evaluation of mental health recovery programme using mixed methods. *Journal of Research in Nursing, 23(6), 476-489.*

Huang, C., Yang, J., Lu, H., Huang, H., Yu, L. (2017). Greenspaces as an Indicator of Urban Health: Evaluating its Changes in 28 Mega-Cities. *Remote Sensing*, 9, Article 1266.

Iliyasu, R., Etikan, I. (2021). Comparison of quote sampling and stratified random sampling. International *Journal of Biometrics,* 10, 24-27.

Independent. (2021). How much have Britons spent on their gardens in the past year? Retrieved 08/05/2023, from https://www.independent.co.uk/life-style/britons-gardening-lockdown-uk-b1811706.html

Jackson, J. (2017). Growing the community – a case study of community gardens in Lincoln's Abbey Ward. *Renewable Agriculture and Food Systems, ISSN 1742-1705.*

James, P., Banay, R. F., Hart, J. E., Laden, F. (2015). A review of the health benefits of greenness. *Current Epidemiology Reports, 2,* 131-142.

Jerrim, J. (2020). Measuring socio-economic background using administrative data. What is the best proxy available? *Quantitative Social Science,* 20-29.

Jim, C. Y., Chen, W. Y. (2006). Impacts of urban environmental elements on residential housing prices in Guangzhou (China). *Landscape and Urban Planning, 78,* 422-434.

Jones, A., Hillsdon, M., Coombes, E. (2009). Greenspace access, use, and physical activity: Understanding the effects of area of deprivation. *Preventative Medicine, 49(6),* 500-505.

Jonassen, D. H. (1991). Objectivism versus constructivism: do we need a new philosophical paradigm? Educational Technology Research and Development, 39(3), 5-14.

Jones, D. N., Reynolds, S. J. (2008). Feeding birds in our towns and cities: A global research opportunity. *Journal of Avian Biology, 39*(3), 2-14.

Joye, Y., Dewitte, S. (2018). Nature's broken path to restoration. A critical look at attention restoration theory. *Journal of Environmental Psychology, 59*, 1-8.

Kelly, D. A. (2018). Impact of paved front gardens on current and future urban flooding. *Journal of Flood Risk Management, doi: 10.1111/jfr3.12231.*

Kelly-Reif, K., Wing, S. (2016). Urban-rural exploitation: An underappreciated dimension of environmental injustice. *Journal of Rural Studies,* 47, 350-358.

Kaplan, S. (1995). The Restorative Benefits of Nature: Toward an Integrative Network. *Journal of Environmental Psychology*, *15*, 169-182.

Kaplan, R. (2001). The nature of the view from home: psychological benefits. *Environmental Behaviour, 33*, 507-542.

Kazmierczak, A., Cavan, G. (2011). Surface water flooding risk to urban communities: Analysis of vulnerability, hazard and exposure. *Landscape and Urban Planning,* 103, 185-197.

Kelly, D. A. (2016). Impact of paved front gardens on current and future urban flooding. *Journal of Flood Risk Management,* 11(5), 434-443.

Kendal, D., Williams, N. S. G., Williams, K. J. H. (2012). Drivers of diversity and tree cover in gardens, parks and streetscapes in an Australian city. *Urban Forestry & Urban Greening, 11*, 257-265.

Kettle, P. (2012). Motivations for investing in allotment gardening in Dublin: A sociological analysis. *American Journal of Sociology,* 22(2), 30-63.

Kim, K. H., Park, S. A. (2018). Horticultural therapy programme for middle-aged women's depression, anxiety and self-identity. *Complementary Therapies in Medicine, 39*, 154-159.

Kimpton, A. (2017). A spatial analytical approach for classifying greenspace and comparing greenspace social equity. *Applied Geography, 82*, 129-142.

Kingsfund. (2020). The HNS budget and how it has changed. Retrieved 13/10/2020, from https://www.kingsfund.org.uk/projects/nhs-in-a-nutshell/nhs-budget

Kingsley, J., Diekmann, L., Egerer, M,. H., Lin, B. B., Ossola, A., March, P. (2022). Experiences of gardening during the early stages of the Covid-19 pandemic. *Health and Place,* 76, Article 102854.

Kinzig, A. P., Warren, P. Martin, C., Hope, D. (2005). The effects of human socioecological status and cultural characteristics on urban patterns of biodiversity. *Ecological Society, 10(1), 1-13.*

Kirkpatrick, J. B., Daniels, G. D., Zagorski, T. (2007). Explaining variation in front gardens between suburbs of Hobart, Tasmania, Australia. *Landscape and Urban Planning, 79*, 314-322.

Kirkpatrick, J., B., Davison, A. (2018). Home-grown: Gardens, practices and motivations in urban domestic vegetable production. *Landscape and Urban Planning,* 170, 24-33.

Koh, S. Y. (2019). Inequality, International Encyclopedia of Human Geography (Second Edition), 269-277.

Kortright, R., Wakefield, S. (2011). Edible backyards: a qualitive study of household food growing and its contributions to food security. *Agriculture and Human Values, 28*, 39-53.

Kos, K., Bujoczek, M., Bujoczek, L. (2021). How do the features of individual gardens affect bird diversity in rural-suburban areas? *Urban Forestry & Urban Greening, 58,* Article 126962.

Kostourou, F. (2021). Housing growth: impacts on density, space consumption and urban morphology. *Buildings and Cities*, 2(1), 55–78, doi: https://doi.org/10.5334/bc.75

Kumagai, Y., Yamada, Y. (2008). Greenspace relations with residential values in downtown Tokyo – implications for urban biodiversity conservation. *Local Environment, 13(2),* 141-157.

Kuras, E. R., Warren P. S., Zinda, J. A., Aronson, M. F. J., Cilliers, S., … Winkler, R. (2020). Urban socioeconomic inequality and biodiversity often converge, but not always: A global meta-analysis. *Landscape and Urban Planning,* 198, 103799.

Kwiatlowski, L., Gaston, K. J., Beck, H., Brett, H., Batty, M., …. Harris, J. A. (2014). Chapter 10: Urban. In National Ecosystem Assessment technical report.

Larkham, P. J. (2006). The study of urban form in Great Britain. *Urban Morphology,* 10(2), 117-141.

Larson, K. L., Cook, E., Strawhacker, C., Hall, S. J. (2010). The influence of diverse values, ecological structure, and geographic context on residents' multifaceted landscaping decisions. *Human Ecology, 38*, 747-761.

Larson, E. K., Perrings, C. (2013). The value of water-related amenities in an arid city: The case of Phoenix Metropolitan Area. *Landscape and Urban Planning, 109,* 45-55.

Latinopoulos, D., Mallios, Z., Latinopoulos, P. (2016). Valuing the benefits of an urban park project: A contingent valuation study in Thessaloniki, Greece. *Land Use Policy, 55, 130-141.*

Leaders. (2019). DIY-loving Brits spend £22bn on home improvements. Retrieved 14/10/2020, from https://www.leaders.co.uk/advice/brits-spend-22bn-on-home-improvements

Leong, M., Dunn, R. R., Trautwein, M. D. (2018). Biodiversity and socioeconomics in the city: a review of the luxury effect. *Biology Letter, 14*, 1-6.

Leopold, L., Leopold, T. (2018). Education and health across lives and cohorts: a study of cumulative (dis)advantage and its rising importance in Germany. *Journal of Health and Social Behaviour,* 59(1), 94-112.

Lerman, S. B., Warren, P. S. (2011). The conservation value of residential yards: linking birds and people. *Ecological Applications,* 21, 1327-1339.

Leve, M., Baudry, E., Bessa-Gomes, C. (2019). Domestic gardens as favourable pollinator habitats in impervious landscape. *Science of the Total Environment,* 10, 420-430.

Lepczyk, C. A., Warren, P. S., Mertig, A. G. (2012). Who feeds the birds? A comparison across regions. Urban bird ecology and conservation. *Studies in Avian Biology, 45*, 267-282.

Li, V. O., Han, Y., Lam, J. C., Zhu, Y., Bacon-Shone, J. (2018). Air pollution and environmental injustice: Are the socially deprived exposed to more PM2.5 pollution in Hong Kong? *Environmental Science and Policy,* 80, 53-61.

Li, S., Jiang, Y., Ke, S., Nie, K., Wu, C. (2021). Understanding the effects of influential factors on housing prices by combining extreme gradient boosting and a hedonic price model (XGBoost-HPM). *Land,* 10(5), 533.

Lin, B. B., Fuller, R. A. (2013). Sharing or sparing? How should we grow the world's cities? *Journal of Applied Ecology,* 50(5), 1161-1168.

Lin, B. B., Gaston, K. J., Fuller, R. A., Wu, D., Bush, R., Shanahan, D. F. (2017). How green is your garden? Urban form and socio-demographic factors influence yard vegetation, visitation, and ecosystem service benefits. *Landscape and Urban Planning, 157*, 239-246.

Liotta, C., Kervinio, Y., Levrel, H., Tardieu, L. (2020). Planning for environmental justice – reducing well-being inequalities through urban greening. *Environmental Science and Policy,* 112, 47-60.

Liu, Y., Zhao, B., Cheng, Y., Zhao, T., Zhang, A., Cheng, S., Zhang, J. (2023). Does the quality of street greenspace matter? Examining the associations between multiple greenspace exposures and chronic health conditions of urban residents in a rapidly urbanising Chinese city. *Environmental Research, 222. 115344.*

Locke, D. H., Avoloio, M., Trammell, T. L. E., Chowdhury, R. R., Grove, J. M., Rogan, J., … Wheeler, M. M. (2018). A multi-city comparison of front and backyard differences in plant species diversity and nitrogen cycling in residential landscapes. *Landscape and Urban Planning, 178, 102-111.*

Loram, A., Tratalos, J., Warren, P. H., Gaston, K. J. (2007). Urban Domestic Gardens: the extent and structure of the resource in five major cities. *Landscape Ecology*, *22*, 601-615.

Loram, A., Thompson, K., Warren, P. H., & Gaston, K. J. (2008). Urban domestic gardens (XII): The richness and composition of the flora in five cities. *Journal of Vegetation Science, 19,* 321–330.

Loram, A., Warren, P. H., Gaston, K. J. (2008). Urban Domestic Gardens: The Characteristics of Gardens in Five Cities. *Environmental Management, 42*, 361-376.

Loss, S. R., Ruiz, M. O., Brawn, J, D. (2009). Relationships between avian diversity neighbourhood age, income, and environmental characteristics of an urban landscape. *Biological Conservation,* 142, 2578-2585.

Luck, G. W., Smallbone, L., O'Brian, R. (2009). Socioeconomics and vegetation change in urban ecosystems: patterns in space and time. *Ecosystems, 12*, 604-620.

Lynch, M., Spencer, L. H., Edwards, R. T. (2020). A systematic review exploring the economic valuation of accessing and using green and blue spaces to improve public health. *Environmental Research and Public Health,* 17(11), doi:10.3390

Maantay, J. (2007). Asthmas and air pollution in the Bronx: Methodological and data considerations in using GIS for environmental justice and health research. *Health and Place,* 13, 32-56.

Maas, J., Verheji, R. A., de Vries, S., Spreeuwenberg, P., Schellevis, F. G., Groenewegen, R., A. (2009). Morbidity is related to a green living environment. *Journal of Epidemiological Community Health, 63*, 967-973.

Maas, J., Verheji, R. A., Groenewegen, P. P., de Vries, S., Spreeuwenberg, P. (2006). Greenspace, urbanity, and health: How strong is the relation? *Journal of Epidemiology & Community Health*, *60*, 587-592.

Maat, K., De Vries, P. (2006). The influence of the residential environment on greenspace travel: Testing the compensation hypothesis. *Environment and Planning A: Economy and Space, 38(*11), 2111-2127.

Manchester History. (n.d.). A brief history of Salford. Retrieved, 07/01/2020, from [https://manchesterhistory.net/manchester/outside/SALFORD/salfordhistory.html](https://manchesterhistory.net/manchester/outside/SALFORD/salfordhistory.html)

Malarvizhi, K., Vasantha Kumar, S., Porchelvan, P. (2016). Use of high resolution Google Earth imagery in land use map preparation for urban related applications. *Procedia Technology, 24*, 1835-1842.

Manchester City Council. (2019). Indices of Deprivation 2019. Retrieved 07/06/2021, from https://manchester.gov.uk/download/downloads/id/27587

Martin, C., A. (2003). Residential landscaping in Phoenix, Arizona, US: practices and preferences relative to covenants, codes and restrictions. *Journal of Arboriculture, 29*(1), 9-17.

Maslow, A. H. (1943). A theory of human motivation. *Phycological Review, 50(4),* 370-396.

Mason. D. (2017). Top 10 factors that affect your property's popularity. Retrieved 14/01/2022, https://www.foxtons.co.uk/discover/2017/08/top-10-factors-that-affect-your-propertyas-popularity.html

Massey, D., S. (1996). The age of extremes: concentrated affluence and poverty in the twenty first century. *Demography,* 33, 395-412.

Mathieu, R., Freeman, C., Aryal, J. (2007). Mapping private gardens in urban areas using object orientated techniques and very high-resolution satellite imagery. *Landscape and Urban Planning*, *81*, 179-192.

Martin, S., Claxton, K., Lomas, J., Longo, F. (2023). The impact of different types of NHS expenditure on health: Marginal cost per QALY estimates for England for 2016/17. *Healthy Policy,* 132, Article 104800.

Mason, H., Jones-Lee, M., Donaldson, C. (2009). Modelling the monetary value of a QALY: A new approach based on UK data. *Health Economics*,*18*, 933–950.

Mavoa, S., Kooshari, M. J., Badland, H. M., Davern, M., Feng, X., Astell-Burt, T., Giles-Corti, B. (2015). Area-level disparities of public open space: a geographic information systems analysis in metropolitan Melbourne. *Urban Policy and Research, 33*(3), 306-323.

Maxwell, S., Lovell, R. (2017). Evidence statement on the links between natural environment and human health.

Mayor of London. (2010). Housing Design Quality and Standards. Retrieved 17/04/2023, from https://www.london.gov.uk/programmes-strategies/housing-and-land/improving-quality/housing-design-and-quality

Mayorga, I., Bichier, P., Philpott, S. M. (2020). Local and landscape drivers of bird abundance, species richness, and trait composition in urban agroecosystems. *Urban Ecosystems, 23, 495-505.*

McClintock, N., Simpson, M. (2014). A survey of urban agriculture organizations and businesses in the US and Canada: Preliminary results.

McVey, D., Nash, R., Stansbie, P. (2018). The motivations and experiences of community garden participants in Edinburgh, Scotland. *Regional Studies, Regional Science, 5*(1), 40-56.

Mead, B. R., Davies, J. A. C., Falagan, N., Kourmpetli, S., Liu, L., Hardman, C. A. (2021). Growing your own in time of crisis: the role of home food growing in perceived food insecurity and well-being during the early COVID-19 lockdown. *Emerald Open Research, 3*(7).

Mears, M., Bindley, P., Maheswaran, R., Jorgensen, A. (2019). Understanding the socio-economic equity of publicly accessible greenspace distribution: The example of Sheffield, UK. *Geoforum, 103*, 126-137.

Mee, K. J., Instone, L., Williams, M., Palmer, J., Vaughan, N. (2013). Renting over troubled waters: an urban political ecology of rental housing. *Geographical Research,* 52(4), 349-459.

Mell, I. C., Can you tell a green field from a cold steel rail? Examining the 'green' of Green Infrastructure development. *Local Environment,* 18(2), 152-166.

Melles, S, J. (2015). Urban bird diversity as an indicator of human social diversity and economic inequality in Vancouver, British Columbia. *Urban Habitats,* 3, 25-48.

Methorst, J., Bonn, A., Marselle, M., Bohning-Gaese, K., Rehdanz, K. (2021). Species richness is positively related to human health – A study for Germany. *Landscape and Urban Planning, 211, Article 104084.*

Ministry of Housing, Communities & Local Government. (2020). White Paper: Planning for the Future.

Mitchell R., Popham, F. (2008). Effect of exposure to natural environment on health inequalities: an observational population study. *The Lancet, 372*(9650), 1655-1660.

Ministry of Housing, Communities & Local Government. (2019). English indices of deprivation 2019. Retrieved 02/11/2020, from https://www.gov.uk/government/statistics/english-indices-of-deprivation-2019

Ministry of Housing, Communities & Local Government. (2021). National Design Guide. Retrieved: 25/01/23, from https://www.gov.uk/government/publications/national-design-guide

Moran, D. (2019). Back to nature? Attention restoration theory and the restorative effects of nature contact in prison. *Health & Place, 57*, 35-43.

Morancho, A. B. (2003). A hedonic valuation of urban green areas. *Landscape and Urban Planning,* 66(1), 35-41.

Moron, D., Klimowicz, M. (2021). Using the social return on investment (SROI) as a measure of the effectiveness of social innovation projects implemented under public policies. *Social Enterprise Journal, 1750-8614.*

Moon, K., Blackman D. (2014). A guide to understanding social science research for natural scientists. *Conservation Biology,* 28(5), 1167-1177.

Moonie, S., Sterling, D. A., Figgs, L. W., Castro, M. (2008). The relationship between school absence, academic performance, and asthma status. *Journal of School Health, 78, 140-148.*

Moss, E. D., Evans, D. M., Atkins, J. P. (2021). Investigating the impacts of climate change on ecosystem services in UK ago-ecosystems: An application of the DSIR framework. *Land-use Policy, Article* 105394.

Mourato, S., Atkinson, G., Collins, M., Gibbons, S., MacKerron, G., Resende, G. (2010). UK National Ecosystem Assessment Economic Analysis of Cultural Services.

Murkin, K., Shiode, N. Shiode, S., Kidd, D. (2023). Biodiversity and the recreational value of green infrastructure in England. *Sustainability,* 15, doi: 10.3990/su15042915

Naderifar, M., Goli, H., Ghaljaie, F. (2017). Snowball sampling: A purposeful method of sampling in qualitative research. *Strides of Development of Medical Education, 14(3),* doi: 10.5812/sdme.67670.

National Gardening Association. (2009). The impact of home and community gardening in America. South Burlington: National Gardening Association.

National Health Service. (2019). Putting Health into Practice: Principles 1-3 Plan, Assess and Involve. Retrieved 27/10/19, from https://www.england.nhs.uk/publication/putting-health-into-place-principles-1-3-plan-assess-and-involve/

National Health Service. (2017). Health survey for England 2016: adult overweight and obesity. Retrieved 04/02/2020, from http://healthsurvey.hscic.gov.uk/media/63745/HSE2016-Adult-obe.pdf

National Health Service. (n.d.). Business definitions. Retrieved 23/01/2020, from https://www.datadictionary.nhs.uk/data_dictionary/nhs_business_definitions/o/output_area_de.asp?shownav=1?query=%22output+area%22&rank=100&shownav=1

Natural Capital Committee. (n.d.). Natural Capital. Retrieved 09/04/18, from http://www.naturalcapitalcommittee.org/natural-capital/

Natural England. (2022). Monitor of Engagement with the Natural Environment (MENE). https://www.gov.uk/government/collections/monitor-of-engagement-with-the-natural-environment-survey-purpose-and-results

Nature Greater Manchester. (2021). Urban Pioneer. Retrieved 17/02/2021, from https://naturegreatermanchester.co.uk/project/urban-pioneer/

National Health Service Digital. (n.d.). Health Survey for England. Retrieved 21/01/2020, from https://digital.nhs.uk/data-and-information/publications/statistical/health-survey-for-england

National Health Service Digital. (2016). National Child Measurement Programme: England 2015/16 school year. Retrieved 04/02/2020, from https://webarchive.nationalarchives.gov.uk/20180328135000/http://digital.nhs.uk/catalogue/PUB22269

National Institute for Health and Care Excellence. (n.d.). Glossary. Retrieved 13/04/2023, from https://www.nice.org.uk/glossary?letter=q

Nature Greater Manchester. (n.d.). Who sits on the Greater Manchester Natural Capital Group? Retrieved 14/01/2020, from https://naturegreatermanchester.co.uk/our-work/sits-greater-manchester-natural-capital-group

Nazir, N. N. M., Othman, N., Nawawi, A. H. (2015). Role of green infrastructure in determining house value in Labuan using hedonic pricing model. *Procedia – Social and Behavioural Sciences,* 170, 484-493.

Nicholas, S. O., Giang, A. T., Yap, P. L. K. (2019). The effectiveness of horticultural therapy on older adults: A systematic review. *Journal of American Medical Directors Association*, in press.

Nur-Shafiza, M. I., Syamsul-Herman, M. A., Wan-Norhidayah, W. M. (2023). Contingent valuation method for urban green space: Case of Burkit Kiara, Kuala Lumpa, Malaysia. *Journal of Tropical Forest Science,* 35(1), 20-27.

Nwokah, N. G., Briggs, A., Kiabel, B. D. (2009). Philosophical foundations and research relevance: issues for marketing information research. Journal of Information Science, 29(6), 445-452.

Office for National Statistics(a). (n.d.). England: Detailed information on the administrative structure within England. Retrieved 21/01/2020, from https://www.ons.gov.uk/methodology/geography/ukgeographies/administrativegeography/england

Office for National Statistics(b). (n.d.) The National Statistics Socio-economic classification (NS-SEC). Retrieved 21/01/2020, from https://www.ons.gov.uk/methodology/classificationsandstandards/otherclassifications/thenationalstatisticssocioeconomicclassificationnssecrebasedonsoc2010

Office of National Statistics(c). (n.d.). O – U. 2001 census glossary terms beginning O – U. Retrieved 21/01/2020, from https://www.ons.gov.uk/census/2001censusandearlier/glossary/ou

Office for National Statistics. (2011). 2011 Census: population and household estimates for the United Kingdom. Retrieved 01/01/2020, from https://www.ons.gov.uk/peoplepopulationandcommunity/populationandmigration/populationestimates/datasets/2011censuspopulationandhouseholdestimatesfortheunitedkingdom

Office for National Statistics. (2012). Ethnicity and national identity in England and Wales: 2011. Retrieved 03/02/2020, from https://www.ons.gov.uk/peoplepopulationandcommunity/culturalidentity/ethnicity/articles/ethnicityandnationalidentityinenglandandwales/2012-12-11#changing-picture-of-ethnicity-over-time

Office for National Statistics. (2013). General health in England and Wales: 2001 comparison with 2011. Retrieved 01/02/2020, from https://www.ons.gov.uk/peoplepopulationandcommunity/healthandsocialcare/healthandwellbeing/articles/generalhealthinenglandandwales/2013-01-30

Office for National Statistics(b). (2013). What does the 2011 census tell us about older people. Retrieved .0/02/2020, from https://www.ons.gov.uk/peoplepopulationandcommunity/birthsdeathsandmarriages/ageing/articles/whatdoesthe2011censustellusaboutolderpeople/2013-09-06

Office for National Statistics. (2015). Pen Portraits for the 2011 Area Classification for Output Areas. Retrieved 01/01/2020, from https://www.ons.gov.uk/methodology/geography/geographicalproducts/areaclassifications/2011areaclassifications/penportraitsandradialplots

Office for National Statistics(b). (2015). Population estimates for the UK, England and Wales, Scotland and Northern Ireland: mid-2019. Retrieved 26/08/2020, from https://www.ons.gov.uk/peoplepopulationandcommunity/populationandmigration/populationestimates/bulletins/annualmidyearpopulationestimates/latest

Office of National Statistics. (2017). Overview of the UK population: July 2017. Retrieved 01/02/2020, from https://www.ons.gov.uk/peoplepopulationandcommunity/populationandmigration/populationestimates/articles/overviewoftheukpopulation/july2017

Office for National Statistics (2019). Families and households in the UK: 2019. Retrieved 14/10/2020, from https://www.ons.gov.uk/peoplepopulationandcommunity/birthsdeathsandmarriages/families/bulletins/familiesandhouseholds/2019

Office for National Statistics (2019b). Population estimates for the UK, England and Wales, Scotland, and Norther Ireland: mid-2018. Retrieved 20/02/2021, from Population estimates for the UK, England and Wales, Scotland and Northern Ireland - Office for National Statistics (ons.gov.uk)

Office for National Statistics. (2020). Family spending in the UK: April 2018 to March 2019. Retrieved 14/10/2020, from https://www.ons.gov.uk/peoplepopulationandcommunity/personalandhouseholdfinances/expenditure/bulletins/familyspendingintheuk/april2018tomarch2019

Office for National Statistics. (2020b). One in eight British households has no garden. Retrieved 22/01/22, from https://www.ons.gov.uk/economy/environmentalaccounts/articles/oneineightbritishhouseholdshasnogarden/2020-05-14

Office for National Statistics. (2021a). Census 2021 geographies. Retrieved 13/04/2023, from https://www.ons.gov.uk/methodology/geography/ukgeographies/censusgeographies/census2021geographies

Office for National Statistics. (2021b). Understanding towns in England and Wales: house price analysis. Retrieved 25/04/2023 from https://www.ons.gov.uk/peoplepopulationandcommunity/housing/articles/understandingtownsinenglandandwaleshousepriceanalysis/2021-10-18

Office for National Statistics. (2022). The People and Nature Survey for England: Year 2 Annual Report – data and publications (April 2021 – March 2022) main findings. Retrieved 08/05/2023, from https://www.gov.uk/government/statistics/the-people-and-nature-survey-for-england-year-2-annual-report-data-and-publications-april-2021-march-2022-official-statistics-main-findings/the-people-and-nature-survey-for-england-year-2-annual-report-data-and-publications-april-2021-march-2022-official-statistics-main-findings#what-is-the-people-and-nature-survey

Office for National Statistics. (2023). How life has changed in Salford: Census 2021. Retrieved 09/02/2023, from https://www.ons.gov.uk/visualisations/censusareachanges/E08000006/

Office of the Deputy Prime Minister (2002). English Housing Survey (London).

Office of the Deputy Prime Minister (2003). Housing in England 2002/3. Retrieved 10/6/21, from Housing in England 2002/3, Office of the Deputy Prime Minister - Publication Index | NBS (thenbs.com)

Oliveira, J., Roca, Z., Leitao, N. (2010). Territorial identity and development: from topophilia and terraphilia. *Land Use Policy,* 27(3), 801-814.

Olsen, J. R., Caryl, F. M., McCrorie, P., Mitchell, R. (2022). Socioeconomic inequality in Scottish children's exposure to and use of natural space and private gardens, measured by GPS. *Landscape and Urban Planning, 223, 104425.*

Ordnance Survey. (2017). OS MasterMap Greenspace: Product Guide. Retrieved 08/02/2020, from https://www.ordnancesurvey.co.uk/documents/osmm-greenspace-layer-product-guide.pdf

Osborne, P. E., Alvares-Sanches, T. (2019). Quantifying how landscape composition and configuration affect urban land surface temperature using machine learning and neutral landscapes. *Computers, Environment and Urban Systems, 76, 80-90.*

Owens, D. E. (1983). The Manchester Ship Canal. Manchester University Press. Pg. 120.

Padulles Cubino, J., Kirkpatrick, J. B., Subiros, J. V. (2016). Do water requirements of Mediterranean gardens relate to socio-economic and demographic factors? *Urban Water Journal, 14*(4), 401-408.

Panagiotakopulu, E. (2001). New record for ancient pests: Archaeoentomology in Egypt. *Journal of Archaeological Science,28(11),* 1235-1246.

Parker, Y., Yom-Tov, Y., Alon-Mozes, T., Barnea A. (2014). The effect of plant richness and urban garden structure on bird species, diversity and community structure. *Landscape and Urban Planning, 122, 186-195.*

Patton, M. Q. (2002). Qualitative research & evaluation methods. Sage, Thousand Oaks, California.

Penninx, B. WJH., Milaneschi, Y., Lamers, F., Vogelzangs, N. (2013). Understanding the somatic consequences of depression: biological mechanisms and the role of depression symptom profile. *BMC Medicine, 11*, 129.

Perino, G., Andrews, B., Kontoleon, A., Bateman, I. (2014). The value of urban green space in Britain: A methodological framework for spatially referenced benefit transfer. *Environmental Resource Economics,* 57, 251-272.

Perry, T., Nawaz, R. (2008). An investigation into the extent and impacts of hard surfacing of domestic gardens in an area of Leeds, United Kingdom. *Landscape and Urban Planning, 86*, 1-13.

Petts, J. (2005). Chapter 9: The economics of urban and peri-urban agriculture. *Continuous productive urban landscapes: Designing urban agriculture for sustainable cities. Viljoen, A., Bohn, K., Howe, J. (Eds) Oxford: Architectural Press.*

Philips, C. V., Sexton, K. (1999). Science and policy implications of defining environmental justice. *Journal of Exposure Analysis and Environmental Epidemiology, 9, 9-17.*

Pratt, D. D. (1998). Five perspectives on teaching in adult and higher education. Krieger Publishing Company, Malabar, Florida.

Pretty, J., Peacock, J., Sellens, M., Griffin, M. (2005). The mental and physical health outcomes of green exercise. *International Journal of Environmental Health Research, 15*(5), 319-337.

Proctor, J. D. (1998). The social construction of nature: relativist accusations, pragmatist and critical realist responses. *Annals of the Association of American Geographers*, 88,352–376.

Property Reporter. (2020). British gardens reaping the benefits of life under lockdown. Retrieved 20/06/2020, from https://www.propertyreporter.co.uk/at-home/british-gardens-reaping-the-benefits-of-life-under-lockdown.html

Pocock & Shaw: Residential sales, lettings & management. (2021). Retrieved 15/06/2022, from https://www.pocock.co.uk/news/garden-space-sees-average-property-value-grow-by-11k.html

Poulsen, M. N., Hulland, K. R. S., Gulas, C. A., Pham, G., Dalglish, S. L., Wilkinson, R. K., Winch, P. J. (2014). Growing an urban oasis: a qualitative study of the perceived benefits of community gardening in Baltimore, Maryland. *The Journal of Culture and Agriculture, 36(2), 69-82.*

Pourias, J., Aubry, C., Duchemin, E. (2015). Is food a motivation for urban gardeners? Multifunctionality and the relative importance of the food function in urban collective gardens of Paris and Montreal. *Journal of Agriculture, Food, and Human Values Society, 33*(2), 257-273.

Poortinga, W., Bird, N., Hallingberg, B., Phillips, R., Williams, D. (2021). The role of perceived public and private greenspace in subjective health and wellbeing during and after the first peak of Covid-19 outbreak. *Landscape and Urban Planning,* 211, 104092.

Public Health England. (2019). National child measurement programme. Retrieved 21/01/2020, from https://www.gov.uk/government/collections/national-child-measurement-programme

Public Health England. (n.d.). Data: Physical Activity. Retrieved 01/04/2018, from https://fingertips.phe.org.uk/profile/physical-activity/data#page/4/gid/1938132899/pat/15/par/E92000001/ati/6/are/E12000002/iid/848/age/168/sex/4

Public Health England. (2020). Beyond the data: Understanding the impact of COVD-19 on BAME groups. Retrieved 23/06/2020, from https://www.gov.uk/government/publications/covid-19-understanding-the-impact-on-bame-communities

Public Health England. (2020b). Improving access to greenspace: A new review for 2020. Retrieved 07/02/22, from https://www.artshealthresources.org.uk/docs/improving-access-to-greenspace-a-new-review-for-2020/

Pulighe, G., Baiocchi, V., Lupia, F. (2016). Horizontal accuracy assessment of very high resolution Google Earth images in the city of Rome, Italy. *International Journal of Digital Earth, 9*(4), 342-362.

Radford, K., G., James P. (2013). Changes in the value of ecosystem services along a rural-urban gradient: A case study of Greater Manchester, UK. *Landscape and Urban Planning*, 109, 117-127.

Rahman, A., Aggarwal, S., P., Netzband, M., Fazal, S. (2011). Monitoring urban sprawl using remote sensing and GIS techniques of a fast growing urban centre, India. *Journal of Selected Topics in Applied Earth Observations and Remote Sensing, 4*(1), 56-64.

Ramsden, S. (2020). 'It's one of the few things that…pulls us together when the outside world is really tough.' Exploring the outcomes of a charity led community garden in a disadvantaged English city. *Local Environment,*26(2), 283-296.

RAND Health Care. (n.d.). 36-Item short form survey (SF-36). Retrieved 23/01/2020, from https://www.rand.org/health-care/surveys_tools/mos/36-item-short-form.html

Rauhut, D. (2018). A Rawls-Sen approach to spatial injustice. *Social Science Spectrum, 4*(3), 109-122.

Raymond, C. M., Diduck, I. P., Buijs, A., Boerchers, M., Moquin, R. (2018). Exploring the co-benefits (and costs) of home gardening for biodiversity conservation. *Local Environment, 24(3),* 258-273.

Richards, D. R., Thompson, B. S. (2019). Urban ecosystems: A new frontier for payments for ecosystem services. *People & Nature,* 1(2), 249-261.

Richardson, T. (2012). Meet Britain's most influential gardeners. Retrieved 21/01/2020, from https://www.telegraph.co.uk/gardening/8993038/Meet-Britains-most-influential-gardeners.html

Riley, G. (n.d.). Public goods and market failure. Retrieved 15/01/20202, from https://www.tutor2u.net/economics/reference/public-goods

Rishbeth, C., Ganji, F., Vodicka, G. (2017). Etnographic understandings of ethnically diverse neighbourhoods to inform urban design practice. *Local Environment,* 23(1), *36-53.*

Robb, G. N., McDonald, R. A., Chamberlain, D. E., Bearhop, S. (2008). Food for thought: supplementary feeding as a driver of ecological changes in avian populations. *Frontiers in Ecology and the Environment, 6*, 476-484.

Robbins, P. (2007). *Lawn people: How grasses, weeds, and chemicals make us who we are.* Temple University Press.

Robert Holmes & Company. (2018). What is the value of public parks? Retrieved 08/02/2020, from https://robertholmes.co.uk/news/value-public-parks/

Rother District Council. (2017). Development and site allocations local plan. Retrieved 13/04/2023, from https://www.rother.gov.uk/planning-and-building-control/planning-policy/local-plan/about-the-local-plan-2/

Rothwell, J. J., Dise, N. B., Taylor., K. G., Allott, T. E. H., Schofield, P., Davies, H., Neal, C. (2010). A spatial and seasonal assessment of river water chemistry across North West England. *Science of the Total Environment,* 408, 841-855.

Royal Horticultural Society. (n.d.). About RHS Garden Bridgewater. Retrieved 02/02/2022, from https://www.rhs.org.uk/gardens/bridgewater/about-bridgewater

Royal Horticultural Society(b). (n.d.) Britain in Bloom. Retrieved 09/01/2023, from https://www.rhs.org.uk/get-involved/britain-in-bloom

Royal Horticultural Society. (2006). Gardening matters. Front gardens: are we parking on our gardens? Do driveways cause flooding? Royal Horticultural Society.

Royal Horticultural Society. (2015). Why we all need greening grey Britain: Royal Horticultural Society, London, UK. Retrieved 14/04/2023, from https://www.rhs.org.uk/get-involved/greening-great-britain

Russell, R., Guerry, A. D., Balvanera, P., Gould, R. K., Basurto, X. … Tam, J. (2013). Humans and nature: how knowing and experiencing nature affect well-being. *Annual Review of Environment and Resources, 38, 473-502.*

Salford City Council. (2008). Housing Market Renewal Fund. Retrieved 07/06/2021, from https://www.bing.com/search?q=Salford+City+Council.+(2008).+Housing+Market+Renewal+Fund.&cvid=6b7dd29f752a4cb08d4702a3b2307089&aqs=edge..69i57.846j0j1&pglt=43&FORM=ANNTA1&PC=U531

Salford City Council. (2008). Strategic flood risk assessment for greater Manchester. Chapter Six: Salford City Council.

Salford City Council. (2020). Salford's housing strategy. Our home, our city. 2020 – 2025. Retrieved 27/07/2021, from https://www.salford.gov.uk/housing/strategies-policies-and-partnerships/strategies/

Salford City Council. (2023). Salford Local Plan: Chapter 11 Housing. Retrieved 17/04/2023, from https://www.salford.gov.uk/planning-building-and-regeneration/planning-policies/local-planning-policy/salfords-development-plan/salford-local-plan/

Salford City Council. (n.d.). Revised Draft Local Plan, Chapter 12: Housing. Retrieved 25/01/2023, from https://www.salford.gov.uk/planning-building-and-regeneration/planning-policies/local-planning-policy/salfords-development-plan/salford-local-plan/revised-draft-local-plan-chapters/12-housing/.

Salford City Council. (n.d.). Peel Park: Heritage. Retrieved 25/03/2023, from https://www.salford.gov.uk/parks-and-open-spaces/salford-parks/peel-park-salford/heritage/

Salix Homes. (2021). Contractor announced for innovative 'Neighbourhood' housing development in Salford. Retrieved 11/01/2023, https://www.salixhomes.co.uk/news/contractor-announced-for-innovative-neighbourhood-housing-development-in-salford/

Sams, C. (2020). COVID-19 has transformed the gardening industry – so what are the new trends? Horticulture Week, published 25/11/2020, retrieved 12/12/2021, from https://www.hortweek.com/covid-19-transformed-gardening-industry-%E2%80%93-so-new-trends/retail/article/1701037

Samus, A., Freeman, C., Dickinson, K. J. M., van Heezik, Y. (2022). Relationships between nature connectedness, biodiversity of private gardens, and mental well-being during the Covid-19 lockdown. *Urban Forestry & Urban Greening,* 69, 127519.

Sandquist, K., Frank, G., Sandquist, J. (2004). Urbanisation and incidence of psychosis and depression: Follow-up study of 4.4million women and men in Sweden. *British Journal of Psychiatry, 184*, 293-298.

Savas, E. S. (2000). *Privatisation and public-private partnerships. London: C Q Press.*

Scott, AQ., Shannon, P., Hardman, M., Miller, D. (2014). Evaluating the cumulative impact problem in spatial planning: a case study of wind turbines in Aberdeenshire, UK. *The Town Planning Review,* 85(4), doi:10.3828/tpr.2014.27.

Schoen, V., Caputo, S., Blythe, C., (2020). Valuing physical and social output: a rapid assessment of a London community garden. *Sustainability,* 12(13), https://doi.org/10.3390/su12135452

Sempik, J., Aldridge, J., Becker, S. (2003), Social and Therapeutic Horticulture: Evidence and Messages from Research, Thrive in association with the Centre for Child and Family Research, Loughborough University, Reading.

Shanahan, D. F., Lin, B. B., Gaston, K. J., Bush, R., Fuller, R. A. (2014). Socio-economic inequalities in access to nature on public and private lands: A case study from Brisbane, Australia. *Landscape and Urban Planning, 130*, 14-23.

Shen, J., Li, S., Wang H., Wu, S., Liang, Z., Zhang, Y., Wei, F., Li S., Ma, L., Wang, Y., Liu, L., Zhang, Y. (2023). Understanding the spatial relationships and drivers of ecosystem service supply-demand mismatched towards spatially-targeted management of social-ecological system. *Journal of Cleaner Production, Article 136882.*

Shokry, G., Connolly, J. J. T., Anguelovski, I. (2020). Understanding climate gentrification and shifting landscapes of protection and vulnerability in green resilient Philadelphia. *Urban Climate,* 31, Article 100539.

Siftova, J. (2021). Shaping the urban home garden: Socio-ecological forces in the management of private green spaces. *Land Use Policy,* 111, Article 105784.

Simpson, T. J., Francis, R. A. (2021). Artificial lawns exhibit increased runoff and decreased water retention compared to living lawns following controlled rainfall experiments. Urban Forestry & Urban Greening, *63, Article* 127232.

Sinha, A., Elliott, D., Adebowale-Schwarte, M., Gavron, N. (2021). How social value can help London build back better. A Covid and post Covid approach. London Sustainable Development Commission.

Sirgy, M., J., Cornwell, T. (2002). How neighbourhood features affect quality of life. *Social Indicators Research, 59*(1), 79-114.

Smith, C., Dawson, D., Archer, J., Davies, M., Frith, M., Hughes, E. and Massini, P., 2011. From green to grey; observed changes in garden vegetation structure in London, 1998-2008, London Wildlife Trust, Greenspace Information for Greater London, and Greater London Authority.

Soga, M., Gaston, K., Yamaura, Y. (2017). Gardening is beneficial for health: A meta-analysis. *Preventative Medicine Reports, 5*, 92-99.

Sohn, W., Kim, H. W., Kim, J., Li, M. (2020). The capitalized amenity of green infrastructure in single-family housing values: An application of the spatial hedonic pricing method. *Urban Forestry & Urban Greening,* 49, Article 126643.

Staats, H., Gatersleben, B., Hartig, T. (1997). Change in mood as function of environmental design: arousel and pleasure on a simulated forest hike. *Journal of Environmental Psychology,* 17, 283-300.

Statista., (2015). Number of gardeners in the USA. Retrieved 11/12/15, from https://www.statista.com/statistics/227419/number-of-gardeners-usa/

Statistics Bureau, Ministry of Internal Affairs and Communications. (2011). Retrieved on 11/12/15, from https://www.e-stat.go.jp/en/stat-search/files?page=1

Stehman, S. V., Wickham, J. D. (2011). Pixels, blocks of pixels, and polygons: choosing a spatial unit for thematic accuracy assessment. *Remote Sensing of Environment, 115*, 3044-3055.

Stormberg, P. M., Ohrner, E., Brockwell, E., Liu, Z. (2021). Valuing urban green amenities with an inequality lens. *Ecological Economics*, 107067.

Spencer, M. (2017). 'Healthier Fleetwood': Creating healthier communities via improved social networking in a disadvantaged area of the UK. *The British Journal of Diabetes, 17*(3), 1-4.

Spencerm A., Rivero-Arias, O., Wong, R., Tsuchita, A., Bleichrodt, H., Edwards, R., T., Norman, R., Lloyd, A., Clarke, P. (2022). The QALY at 50: One story many voices. *Social Science & Medicine, 296, Article 114653.*

Sugg, H. E. (2020). Special Envoy for girls education. Retrieved 18/10/2021, from https://sustainabledevelopment.un.org/memberstates/unitedkingdom

Tallis, H., Guerry, A., Daily, C. D., (2013). Ecosystem Services. Encucloedia of Biodiversity (Second Edition) Stanford University Press.

Tennessen, C., M., Cimprich, B. (1995). Views to nature: effects on attention. *Journal of Environmental Psychology, 15*, 77-85.

The Natural Capital Approach in Greater Manchester 2019. Retrieved on 05/09/2019, from https://naturegreatermanchester.co.uk/wp-content/uploads/2019/03/GM-Natural-Capital-Accounts-Summary-March-2019_Digital.pdf.

The Property Landlord. Tenants and gardens: who is responsible for gardens? Retrieved 13/09/22, from https://www.thepropertylandlord.co.uk/tenants-gardens-who-responsible-for-maintenance.html

Tilling, C., Krol, M., Tsuchiya, A., Brazier, J., van Excel, J., Brouwer, W. (2009). Measuring the value of life: Exploring a new method for deriving monetary value of a QUALY. Health Economics and Decision Science, University of Sheffield Discussion Paper. Retrieved 23/01/2020, from https://www.sheffield.ac.uk/scharr/sections/heds/discussion-papers/914-1.291844

Time. (2018). #MeToo and Time's Up founders explain the difference between the 2 movements – and how they're alike. Retrieved 20/07/2022, from https://time.com/5189945/whats-the-difference-between-the-metoo-and-times-up-movements/

Time. (2021). The racial reckoning went global last year. Here's how activists in 8 countries are fighting for justice. Retrieved 20/07/2022, from https://time.com/6046299/fighting-injustice-world/

Tinson A., Clair, A. Better Housing is crucial for our health and the COVID-19 recovery. The Health Foundation. Retrieved 10/05/2023, from https://www.health.org.uk/

Toures, A. C., Prevot, A. C., Nadot, S. (2018). Small but powerful: The importance of French community gardens for residents. *Landscape and Urban Planning, 180*, 5-14.

Tratalos, J., Fuller, R. A., Warren, P. H., Davies, R. G., Gaston, K. J. (2007). Urban form, biodiversity potential and ecosystem services. *Landscape and Urban Planning,* 83, 308-317.

Twedt, E., Rainey, R. M., Proffitt, D. R. (2016). Designed natural spaces: Informal gardens are perceived to be more restorative than formal gardens. Frontiers in Psychology, https://doi.org/10.3389/fpsyg.2016.00088

Ugolini, F., Massetti, L., Calaza-Martinez, P., Carinanos, P., Dobbs, C., Ostoic, S. K., Simonetti, M. (2020). Effects of the COVID-19 pandemic on the use and perceptions of urban green space: and international exploratory study. *Urban Forestry & Urban Greening, Article* 56.

Ulrich, R., S. (1984). View through a window may influence recovery from surgery. *Science, 224*(4647), 420-421.

Ulrich, R. S., Simons, R. F., Losito, B. D. (1991). Stress Recovery During Exposure to Natural and Urban Environments. *Journal of Environmental Psychology*, *11*, 201-230.

United Kingdom National Ecosystem Assessment. (n.d.). Ecosystem services. Retrieved 01/10/2021, from http://uknea.unep-wcmc.org/EcosystemAssessmentConcepts/EcosystemServices/tabid/103/Default.aspx

United Nations. (2014). World Urbanization Prospects: The 2014 Revision, Highlights, from https://www.un.org/en/development/desa/publications/2014-revision-world-urbanization-prospects.html

United Nations. (n.d.). Goal 11: Make cities and human settlements inclusive, safe, resilient and sustainable. Retrieved 01/05/2021, from Goal 11 | Department of Economic and Social Affairs (un.org)

United States Environmental Protection Agency. (n.d.). Environmental Justice. Retrieved 09/03/2020, from https://www.epa.gov/environmentaljustice

University of Salford. (n.d.). Heritage. Retrieved 13/09/2022, from https://www.salford.ac.uk/corporate-information/heritage

Urban Pioneer Committee. (2017). Urban Pioneer: Strategic Plan – Greater Manchester. Retrieved, 05/09/2019, from https://www.greatermanchester-ca.gov.uk/media/1291/urban-pioneer-strategic-plan-jan-2018.pdf

Urban Pioneer (2019). Urban Pioneer Natural Capital Accounts. Retrieved, 05/09/2019, from https://gmgreencity.com/resource-library/

Van de Berg, A. E., Mass, J., Verheij, R. A., Groenewegen, P. P. (2010). Greenspace as a buffer between stressful life events and health. *Social Science & Medicine, 70*(8), 1203-1210.

Van den Berg, E. A. (2017). From green space to green prescriptions: challenges and opportunities for research and practice. *Frontiers in Psychology,* 8, https://doi.org/10.3389/fpsyg.2017.00268

Van de Voorde, T., Vlaeminck, J., Canters, F. (2008). Comparing different approaches for mapping urban vegetation cover from Landsat EMT+ data: a case study on Brussels. *Sensors,* 8, 3880-3902.

Van Heezik, Y., Freeman, C., Porter, S., Dickinson, K. J. M. (2013). Garden size, householder knowledge, and socio-economic status influence plant and bird diversity at the scale of individual gardens. *Ecosystems, 16*, 1442-1454.

Van Heezik, Y., Freeman., C., Davidson, K., Lewis, B. (2020). Uptake and engagement of activities to promote nature species in private gardens. *Environmental Management, 66,* 42-55. https://doi.org/10.1007/s00267-020-01294-5

Vaz, E., Anthony, A., McHenry, M. (2017). The geography of environmental injustice. *Habitat International,* 59, 118-125.

Venkatachalam, L. (2004). The contingent valuation method: a review. *Environmental Impact Assessment Review,* 24, 89-124.

Vina, A., Gitelson, A. A., Nguy-Robertson, A. L., Peng, Y. (2011). Comparison of different vegetation indices for the remote assessment of green leaf area index of crops. *Remote Sensing of Environment, 115*, 3468-3478.

Wall, T. (2020). Salford beats Brighton and Bristol title of ‘greenest place to live’. Retrieved 10/01/2023, from https://www.theguardian.com/environment/2020/mar/14/salford-greenest-city-bristol-brighton-england-wales

Walford, N., Armitage, R. (2020). Residential neighbourhood classification: An environmentally enhanced approach. *Applied Geography,* 121, *Article* 102219.

Warhurst, J. R., Parks, K. E., McCulloch, L., Hudson, M. D. (2014). Front gardens to car parks: Changes in garden permeability and effects on flood regulation. *Science of the Total Environment,* 485-486, 329-339.

Waylen, K. A., Fischer, A, McGowan, P. J. K., Thirgood, S. J., Milner-Gulland, E. J. (2010). Effect of local cultural context on the success of community-based conservation interventions. *Conservation Biology*, 24, 1119–1129.

Wellmann, T., Schug, F., Haase, D., Pflugmacher, D., Linden, S. (2020). Green growth? On the relation between population density, land use and vegetation cover fractions in a city using a 30-years Landsat time series. *Landscape and Urban Planning,* 202, Article 103857.

Whitford, V., Ennos, A. R., Handley, J. F. (2001). ‘City form and natural process’ – indicators for the ecological performance of urban areas and their application to Merseyside, UK. *Landscape and Urban Planning*, *57*, 91-103.

White, M. P., Alcock, A., Wheeler, B. W., Depledge, M. H. (2013). Would you be happier living in a greener urban area? A fixed-effects analysis of panel data. *Psychological Science, 24*(6), 920-928.

Whitehand, J. W. R., Morton N. J., Carr, C. M. H. (1999). Urban morphogenesis at the microscale: how houses change. *Environment and Planning B: Urban Analytics and City Science,* 26(4), doi.org/10.1068/b260503

Wildlife and Countryside Link. (2019). The 25 year environment plan: One year on. Retrieved: 27/10/19, from https://www.wcl.org.uk/25-year-plan-for-nature-group.asp

Wilkerson, M. L., Mitchell, M. G. E., Shanahan, D., Wilson, K. A., Ives, C. D., Lovelock, C. E., Rhodes, L. R. (2018). The role of socio-economic factors in planning and managing urban ecosystem services. *Ecosystem Services, 31*, 102-110.

Wilkinson, R. G., Pickett, K. (2009). The Spirit Level: Why more equal societies almost always do better. Alan Lane.

Willems, P., Arnbjerg-Nielson, K., Olsson, J., Hguyen, V., T., V. (2012). Climate change impact assessment on urban rainfall extremes and urban drainage: methods and shortcomings. Atmospheric Research, 103, 106-118.

Williams, K., Burton, E., Jenks, M. (2000). Achieving sustainable urban form. E & FN Spon, London.

Williams, K. (2014). Urban form and infrastructure: a morphological review: *Future cities working paper.* London: Stationary Office.

Williams, K., Burton, E., Jenks, M. (2000). Achieving Sustainable Urban Form. London: Spon Press.

Willis, K. (2005). Chapter 3: Measuring health impacts in economic terms. Economic benefits of accessible greenspaces for physical and mental health: scoping study. CJC Consulting.

Wilkshurst, J., MacKay, A. (1999). The Fundamentals of Advertising. 2nd Edition, London.

Wise, P. (2014). Grow your own: The potential value and impacts of residential and community food gardening. The Australia Institute, Policy Brief 59.

Wolch, J., Wilson, J. P., Fehrenbach, J., (2005). Parks and park funding in Los Angeles: an equity-mapping analysis. *Urban Geography, 26*,4–35.

World Forum on Natural Capital. (n.d.). What is natural capital? Retrieved 16/11/2021, from https://naturalcapitalforum.com/about/

World Health Organisation. (2004). The global burden of disease report: 2004 update. Retrieved on 05/08/2019, from https://www.who.int/healthinfo/global_burden_disease/GBD_report_2004update_full.pdf?ua=1

World Health Organisation. (2019). Coronavirus disease 2019 (Covid-19): Situation report – 89 (19 April 2020). Retrieved 13/07/2020, from, Coronavirus disease 2019 (COVID-19): Situation Report - 89 (18 April 2020) - World | ReliefWeb

World Health Organisation. (n.d.). Global Health Observatory (GHO) data: Urban population growth. Retrieved 29/01/2020, from https://www.who.int/gho/urban_health/situation_trends/urban_population_growth_text/en/

World Wildlife Fund. (2018). Our reaction to the 25 year environment plan. Retrieved: 27/10/19, from https://www.wwf.org.uk/updates/our-reaction-25-year-environment-plan

Wood, C. J., Pretty, J., Griffin, M. (2016). A case-control study of the health and well-being benefits of allotment gardening. *Journal of Public Health, 38*(3), 336-344.

Wright, C., Booth, I., Buckler, J., Cameron, N., Cole, T., Healy, M., Hulse, J., Preece, M., Reilly, J., Williams, A. (2002). Growth reference charts for use in the United Kingdom. *Archive of Disease in Childhood, 86*(1), 11-14.

Wustemann, H., Kalisch, D., Kolbe, J. (2017). Access to urban greenspace and environmental inequalities in Germany. *Landscape and Urban Planning, 164*, 124-131.

Xioa, Q., McPherson, E. G. (2002). Rainfall intercept by Santa Monica's municipal urban forest. *Urban Ecosystems, 6*, 291-302.

Xue, J., Su, B. (2017). Significant remote sensing vegetation indices: a review of developments and applications. *Journal of Sensors, 2017*, 1-17.

Yazar, M., York, A. (2023). Nature-based solutions through collective actions for spatial justice in urban green commons. *Environmental Science and Policy,* 145, 228-237.

Yerrell P (2008). National evaluation of BTCV's Green Gym. Oxford: School of Health and Social Care, Oxford Brookes University.

Yildiz, B., Schuring, M., Knoef, M. G., Burdorf, A. (2020). Chronic disease and multimorbidity among unemployed and employed persons in the Netherlands: a register-based cross-sectional study. *British Medical Journal,* doi:10.1136/ bmjopen-2019-035037

Young, C., Frey, D., Moretti, M., Bauer, N. (2019). Research Note: Garden-owner reported habitat heterogeneity predicts plant species richness in urban gardens. *Landscape and Urban Planning, 185*, 222-227.