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Exploring the drivers of invertebrate, soil, and plant health across urban greenspaces

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Abstract

The global population is becoming increasingly urban, with more than half of people now living in cities, a figure projected to reach 70% by 2050. In the UK, 84% of people are city dwellers. Greenspaces within cities and towns, including parks, gardens, woodlands and allotments, provide crucial ecosystem services (e.g. carbon storage, water regulation, and pollution capture) and support biodiversity. They minimise the impact of urban areas on the wider environment and help mitigate or adapt to challenges of city living and climate change (e.g. extreme heat and flooding).

However, as the urban population grows, there are competing demands on greenspaces for development or conversion. To inform decision-making and enhance greenspace networks, it is crucial to understand how different forms of greenspace affect soil quality, plant growth, and both above- and below-ground biodiversity. Here, I use a multidisciplinary approach, combining invertebrate sampling, soil analysis, and plant bioassays to examine interactions between greenspace type, invertebrate communities, soil, and plant health across urban landscapes, including woodlands, park grasslands, gardens, allotments, and small inner-city patches.

In Chapter 2, I demonstrate temporal patterns in below-ground invertebrate abundance across greenspace types, with highest populations in garden soils in spring, and in park grassland and woodland soils in late summer. Soil carbon did not vary across greenspace types, but heavy metal levels did, with lowest levels in woodland sites, which also showed lower pH. In Chapter 3, I show that soils from different greenspace types significantly influence plant health and physiological indicators. In Chapter 4, results suggest that higher inner-city greenspace heterogeneity, larger site area, and higher surrounding greyspace are negative drivers of invertebrate abundance, though vegetation effects vary by taxon. Collectively, these results highlight the complexity of greenspace transformation and the need for further work to understand ecosystem service provision and sustain urban greenspaces.

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Covid impact statement

I began my PhD in October 2020, during the height of the COVID-19 pandemic. The ongoing restrictions and disruptions posed significant challenges to my research in the early stages of the project. Due to national lockdowns and institutional closures, I experienced delays in accessing laboratory facilities, libraries, and essential research materials. Fieldwork for Chapter 2 was delayed due to loss of access to university buildings and resources. Access to laboratory equipment for preliminary testing of methods was delayed, resulting in delays of approximately 5 months to fieldwork. Chapter 2 garden sites recruitment was limited by university procedure limiting contact with members of the public, resulting in access to garden sites being volunteered from the members of the University of Sheffield Bioscience department. This caused a bias in sites to the west of the city centre.

With in-person activities restricted, much of my initial work transitioned to remote learning and research. This included attending virtual seminars, conducting literature reviews, and undertaking online training. While this provided an opportunity to deepen my theoretical understanding, the lack of physical presence delayed hands-on experimental work and direct collaboration with peers and supervisors.

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Chapter 1: Introduction Contextualizing Urban Greenspaces

Since 1500, the proportion of the UK living in cities has risen from 3% to 84% (Withington, 2017). Urbanisation, the process through which rural areas transform to urban ones, results in the growth and expansion of cities and towns. In the UK, urban areas generally expand into agricultural land that surround cities and towns (UKCEH, 2020). Land conversion poses challenges to ecosystem function, primarily through habitat fragmentation, degradation, and pollution. Cities are complex systems, characterised by human made impervious sealed surfaces (greyspace), areas of water (bluespace) and greenspaces (Suligowski et al., 2021). Density of urban greyspace varies across cities and towns, with higher proportions found within urban centres compared to suburban spaces (Nuissl et al, 2021).

As urban areas expand, they disrupt the greenspace in and around them. Urban sprawl and densification encroach into habitats and the plant and animal communities living within them often negatively affecting biodiversity (Theodorou, 2021; Concepción et al., 2015). Pollution, including air, water, and soil contaminants, exacerbates these issues, further degrading the quality of urban environments (Appleton et al., 2018; Miller et al., 2017; Rodríguez et al., 2016). Additionally, the conversion of greenspace or vegetated land to impervious surfaces, such as roads and buildings, diminishes soil health and increases runoff, contributing to both soil degradation, water pollution, and urban flooding (Martin et al., 2023; Miller et al., 2017). It is important to maintain ecosystem services in urban areas which help buffer the environmental stresses caused by rapid urbanisation.

The shifts in planning priorities in the past can be exemplified through greenspace evolution across UK cities. One of the key examples can be seen in the aftermath of the 18th century Industrial Revolution, where extreme, rapid urbanisation led to overcrowding and deprioritisation of greenspaces (Whitten et al., 2020; O'Reilly et al., 2019). This deficit later led to greenspace creation projects across the UK, such as Victoria Park (London), Birkenhead Park (Liverpool) and Peel Park (Salford) (GL Council, 2019; Denby et al., 2024; Hayes et al., 2023). Additionally, many existing greenspaces were adapted to meet public demand, such as Regent's Park in London, which saw transformation from royal land to a public park (Rabbits, 2014). This social desire for greenspace within planning continued into the early 20th century, with the Garden City movement creating spaces centred around greenspace land, and post-war reconstruction reshaping existing urban spaces (Vernet et al., 2017; Moerman et al., 2020).

However, in the more recent years, the protection of the greenbelt has increased pressure on existing urban greenspaces, with attempts to reduce urban sprawl compelling an increase in urban density (Boulton et al., 2018). Current UK policy - such as the National Planning Policy Framework (NPPF) and the Environment Act 2021 - highlights the importance of urban greenspaces, aiming to promote biodiversity net gain, enhance ecosystem services, and support sustainable urban development. These policies highlight the importance of urban greenspaces, with efforts to include new greenspaces within planning and increasing recognition of the environmental services they provide (Davies et al., 2017; Hislop et al., 2019).

Urban greenspaces are areas primarily covered with vegetation within cities and towns, including for example parks, gardens, allotments, green roofs, and other vegetated environments (Suligowski et al., 2021). Urban greenspaces vary significantly in size, ranging from expansive city parks that span several square kilometres to smaller community gardens, green roofs, or road verges integrated into the built environment (Gozalo et al., 2019). Despite their varying scales, all forms of urban greenspace contribute to the overall resilience and liveability of urbanenvironments and can provide essential habitats for biodiversity (MacGregor-Fors et al. 2016), particularly in the face of climate change and rapid urbanisation.

Greenspaces are integral to the urban ecosystem, providing numerous ecological, social, and health benefits (Semeraro et al., 2021). Parks provide vital recreational spaces, promote physical activity, and enhance mental well-being of surrounding populations (Rakhshandehroo et al., 2015; Rakhshandehroo et al., 2017). Woodlands serve as habitats for various vegetation species and act as flood control resources (Canedoli et al., 2018; Moffat et al., 2016). Allotments offer a space for urban horticulture, promoting food security, community engagement, and biodiversity through pollinator-friendly plants (Edmondson et al., 2020b; Griffiths-Lee et al., 2020). Gardens promote everyday human-environmental interactions, encouraging biodiverse, aesthetically pleasing vegetation. Street trees improve air quality, reduce noise pollution, and provide shade to cool urban areas (Semeraro et al., 2021). However, while urban greenspaces offer numerous ecological services, they may also present disservices such as maintenance costs, allergen production, and the potential for invasive species spread. Additionally, dedicating land to greenspace can pose an opportunity cost, particularly in densely populated cities where land is scarce, and housing demand is high. All greenspaces contribute to the surrounding urban area understanding how they contribute different levels of ecological services, disservices, and environmental benefits can help to better understand these urban systems.

The role of soils in urban greenspaces

Soils play a fundamental role in ecosystem functioning, acting as the foundation for a wide range of ecological processes (Coleman et al., 2017). They are critical for nutrient cycling, as they store, transform, and release essential nutrients (Coleman et al., 2017; Marschner et al., 2023).
Soils also regulate water availability by absorbing, filtering, and redistributing water, thereby influencing hydrological cycles and mitigating flooding and drought (Steffan et al., 2018; Singh et al., 2020). Moreover, soils are crucial for carbon storage, with twice as much carbon than is held in the atmosphere as CO_2, which assists regulating atmospheric CO_2 levels and mitigate climate change (Lal et al., 2015; Luo et al., 2017). Soil biodiversity, including microorganisms, fungi, and invertebrates, underpins these functions by driving decomposition, nutrient mineralization, and soil structure formation (Dobrovol'Skaya et al., 2017; Robert et al., 2021). Healthy soils, therefore, support ecosystem productivity, resilience, and stability, making them indispensable to both natural ecosystems and human well-being.

Urban soils serve a particularly distinct set of functions to urban areas and underpin their environmental quality. With increased greyspace and infrastructure, urban greenspaces play a key role in regulating water dynamics by absorbing and filtering water runoff and mitigating urban flooding (Bai et al., 2018; Ferreira et al., 2021; Rawlins et al., 2022). Additionally, urban soils act as a pollution sink, trapping and breaking down contaminants such as heavy metals, hydrocarbons, and other urban pollutants (Yang et al., 2015; Li et al., 2018). Urban soils support vegetation across urban areas, which are essential for providing ecosystem services, particularly urban cooling and biodiversity support (Guilland et al., 2018; Jamei et al., 2021). Soil organic matter and invertebrate activity contribute to carbon storage within cities, helping to offset greenhouse gas emissions - a role often underestimated due to assumptions that sealed surfaces and grey infrastructure dominate urban environments, reducing their contribution and limiting potential of carbon storage. (O'Riordan et al., 2021; Edmondson et al., 2015). Despite the soil constraints seen within urban development, such as increased contamination and compaction, urban soils remain vital for sustainability of urban greenspaces, contributing to the ecological service potential provided within these areas (Heyman et al., 2019; O'Riordan et al., 2021).

Indicators of soil health are used to assess the capacity of soils to perform ecological functions. Soil health indicators include several physical, chemical, and biological properties that reflect the overall quality and function of the soil (EA 2019). Physical indicators, such as soil structure, bulk density, and water holding capacity can reflect other characteristics, such as root penetration, soil aeration, and water infiltration, all of which are critical for plant development (Byrnes et al., 2018; Jabro et al., 2021; Lehmann et al., 2020). Chemical indicators, including pH, nutrient content, and exchange capacity, determine the availability of essential nutrients and influence microbial activity (van Es et al., 2019). Biological indicators, such as organic matter and soil invertebrate communities provide insights into the biological processes driving decomposition and nutrient cycling (Fierer et al., 2021; Lu et al., 2022; Ngatia et al., 2021). Together, these indicators offer insight into soil health, enabling understanding of effective management practices to improve soil quality. Understanding these indicators is vital for maintaining productive ecosystems and promoting long-term sustainability.

Within these indicators, the health of the soil biota community is a particularly key focus area, with less than 3% of soil studies investigating urban soils, and less than 3% of these articles focus on soil biota (Guilland et al., 2018). This underrepresentation may be attributed to several intersecting factors. Methodologically, studying soil biota presents significant challenges: organisms are small, diverse, and often require specialised techniques such as DNA metabarcoding or soil extraction methods for accurate identification and quantification. Additionally, soil invertebrates tend to be less visible and less charismatic compared to above-ground fauna like birds or pollinators, leading to limited public interest. Historically, ecological research has prioritised species with direct cultural, economic, or aesthetic value, which has skewed conservation and biodiversity efforts toward more conspicuous taxa. As a result, soil biodiversity - despite its foundational role in ecosystem functioning - has received comparatively little scientific attention, especially within complex urban environments.

Soil biota, which includes a diverse array of organisms such as bacteria, fungi, and arthropods, play a fundamental role in maintaining soil health and ecosystem function (Guilland et al., 2018; Geisen et al., 2019). These organisms drive critical processes, including the decomposition of organic matter and the soil structure formation. Microbial communities, particularly bacteria and fungi, are essential for breaking down organic material and releasing essential nutrients like nitrogen and phosphorus, making them available for plant uptake (O'Riordan et al., 2021). Earthworms and other macrofauna contribute to soil aeration and the creation of pore spaces, which improve water infiltration and root growth (Plaas et al., 2019). Additionally, the presence of diverse soil biota fosters soil stability, enhancing the soil's resistance to erosion. Soil biota respond quickly to changes in soil conditions such as pollution, moisture levels, and organic matter availability, allowing their communities to effectively represent soil health (Morris et al., 2024; Yang et al., 2018). Therefore, the abundance, diversity, and activity of soil organisms are crucial for sustaining soil fertility, structure, and resilience to environmental stresses, directly influencing the productivity and sustainability of ecosystems, however, at present there is a poor understanding of how these communities cope with the stress of living within the urban environment

The role of invertebrates in urban greenspaces

Invertebrates are the most numerous and diverse classes within the animal kingdom and make up 97% of known animal species, with insects making up 90% (Hickman, 2017; Roskov et al., 2020). They are vital components of all ecosystems, performing essential functions that sustain ecological processes (Eggleton, 2020). They play critical roles in pollination, decomposition, and the wider food web, contributing significantly to wider ecosystems (Eggleton et al., 2020). While invertebrates are critical to ecological function, certain species can also act as pests, spread disease, or disrupt native species dynamics when present in high numbers or as non-natives. Research suggests that invertebrate populations are undergoing an overall pattern of decline (though with variation in taxon sensitivity) (Wagner et al., 2020). With a loss of up to 5.2% per year of flying insect biomass, approximately 75% loss has been seen in the last

26 years (Habel et al., 2020; Wagner et al., 2020; Wagner et al., 2021). Within the UK specifically, 30% to 60% of species per order have declined in range, demonstrating the scale of

the problem (Wagner et al., 2020). Within the wider invertebrate community, soil invertebrates are essential to soil ecosystems, contributing to decomposition, nutrient cycling, and soil structure formation (Bray et al., 2019; Griffiths et al., 2021). Including groups such as earthworms, collembola, and mites, these organisms facilitate cycling of nutrients like nitrogen and phosphorus, which are critical for plant growth and overall ecosystem productivity. (Griffiths et al., 2021). Soil invertebrates enhance soil through their burrowing activities, while their organic byproducts help to stabilise soil aggregates (Le Bayon et al., 2021; Griffiths et al., 2021; Menta et al., 2020). However, research has focused on above-ground invertebrates, particularly pollinator species, overlooking less ‘attractive’ insect taxa and giving uncertainty in the effect on the entire invertebrate community (Noriega et al., 2018). As shifts in invertebrate diversity and abundance can signal broader ecological changes and degradation, monitoring and assessing the health of ecosystems is vital.

The relationship between invertebrates and urban greenspaces is critical for understanding the overall health and ecological functioning of these areas. By contributing to ecosystem processes, invertebrates can directly influence the capacity of soils to support ecosystem services (Bray et al., 2019; Griffiths et al., 2021). These interactions can create feedback loops, with processes such as soil compaction reducing invertebrate diversity, which in turn hinders invertebrate movement in soils and further degrades soil quality (Devigne et al., 2016; Vásquez et al., 2023). For example, in allotment gardens where intensive digging and chemical fertiliser use are common, microbial and invertebrate communities such as springtails and mites could be disrupted. This disturbance could reduce the breakdown of organic matter and slows nutrient cycling, leading to poorer soil fertility. In turn, this reduced nutrient availability limits plant growth, which further restricts organic inputs into the soil, compounding the decline in soil biological activity and overall greenspace productivity. Studying the complexities of these interactions provides insights into urban greenspaces, identifying management practices that support both healthy soils and diverse invertebrate communities.Identifying how invertebrate communities vary across different forms of greenspace would inform understanding of their role in greenspace health and function and inform management decisions to optimise greenspaces for both ecosystem service provision and biodiversity. As these sites become more valuable with continued urbanisation, enhancing the potential of these areas will provide an effective resource for biodiversity support into the future.

The role of vegetation cover in urban greenspaces

There are multiple key forms of greenspace within urban areas, designed and managed for different purposes and ecosystem services to urban communities. Some examples include allotment sites, which refers to a parcel of land divided into smaller plot, typically rented from a local authority or private owner, where individuals can grow their own fruits, vegetables, and flowers. A garden site is a parcel of land typically adjoining a residential property, in which plants such as flowers, vegetables, trees are cultivated for aesthetic, practical, or recreational value. A park refers to an area of public or publicly accessible land that is maintained for recreational use or leisure. These spaces typically feature multiple land covers, such as lawns, trees, and amenity grassland. Woodland sites refer to a parcel of land characterised by dense tree and shrub cover, typically features a variety of tree species, undergrowth, and ground vegetation. In addition to these greenspace types, there are many other urban greenspace habitats such as green roofs, road verges, and canal towpaths also contribute to urban biodiversity, often serving as ecological corridors or microhabitats.

This variety in greenspace type and land use has a significant effect on vegetation, influencing plant diversity, landscape composition, and subsequent ecosystem function (Paudel et al., 2023; Pataki et al., 2021). Land managed for agricultural or horticultural purposes, such as allotment land, can promote biodiversity through diverse cropping and inclusion of native vegetation (Royer et al., 2023; Schneider et al., 2022). Heavy use of fertilisers and pesticides in some sites, such as gardens, can degrade soil health and inhibit the growth of non-target vegetation (Cameron, 2023; Meftaul et al., 2020). Site size and isolation can lead to habitat fragmentation and soil compaction, reducing the ability of native plants to thrive (Karimi et al., 2021). Monocultures, such as those common in park grass environments, can result in the loss of native plant communities and disrupt natural successional processes (Paudel et al., 2023). Through these variations in greenspace type, the effect of urban greenspaces on maintenance of diverse plant communities can be observed.

The potential of vegetation to support biodiversity has led to the rise of numerous promoting schemes within urban areas. These schemes aim to enhance biodiversity by promoting diverse vegetation, with a focus on native ecosystems, habitat connectivity, and pollinator support. One approach is referred to as rewilding and involves the planting of native species to restore habitats (Lehmann et al., 2021; Webb et al., 2023). With a similar principle, urban greening initiatives aim to increase green space in cities, through approaches such as planting green roofs, vertical gardens, and establishing urban parks (Berthon et al., 2021). Pollinator-friendly planting schemes allow another, more publicly accessible approach, with schemes to encourage wildflower, native species, and nectar-rich species planting (Daniels et al., 2020; Majewska et al., 2020). These approaches are typically aimed towards pollinator species, such as bees and butterflies, but there is limited research to support the effectiveness on these taxa, or on the wider invertebrate community (Guilland et al., 2018). Additionally, public preference still trends towards less ‘wild looking’ areas, which can negatively impact native biodiversity value in greenspaces (Hoyle et al., 2017). For example, the planting of popular non-native ornamental species in gardens and parks can outcompete native flora, reducing habitat suitability for specialist invertebrate species and disrupting local ecological networks. Assessing the effect of vegetation structure and maintenance on their contributions to biodiversity enhancement can allow effective planning for ecosystem resilience.

The connection between invertebrate populations and their effect on vegetation within greenspaces remains under-researched. Invertebrates can drive vegetation health and growth in several ways, including contributions to soil health and nutrient cycling. Facilitation of nutrient decomposition and cycling of organic matter, such as the leaf litter layer within woodland greenspaces, allows essential nutrients to enter the soils and makes them available for plant uptake, promoting growth (Griffiths et al., 2021). Additionally, over 80% of plant species rely on invertebrates for pollination, without which reproduction and genetic diversity would be significantly limited (Guilland et al., 2018). The transfer of pollen among flowers enhances plant genetic diversity, thereby enhancing fruit and seed production whilst also benefiting invertebrate populations (Wei et al., 2021). Structure of site vegetation can also provide shelter to invertebrates alongside food sources, benefiting communities (Mata et al., 2021). Furthermore, some plant species can attract invertebrates, demonstrating the direct interactions between vegetation and invertebrate populations (Griffiths-Lee et al., 2020). The richness and duration of blossoming flowers presence in urban spaces worked to attract pollinators (Scriven et al., 2013; Daniels et al., 2020). The relationship between invertebrates and vegetation is complex and multifaceted, with plants providing resources for invertebrate survival, and invertebrates contributing to plant health and growth. Understanding how these relationships are influenced by the uniqueness of the urban environment will allow better insight for urban planning. By recognising these processes and their relevance to specific ecosystem processes, such as the importance of resources such as leaf litter and dead wood in supporting decomposer invertebrates that maintain soil health, city planners can design greenspaces that preserve microhabitats, helping to sustain vegetation health and biodiversity over time.

Thesis aims and objectives

The aim of this thesis was to assess how urban greenspace types across a range of scales influence the invertebrate biodiversity, soil characteristics and plant growth. I assessed invertebrate biodiversity using traditional insect sampling techniques, a bioassay to assess plant growth, and several soil assessment techniques to unravel the drivers influencing greenspace health and invertebrate community structure within these areas. My work was based upon three key investigations (Figure 1.1). Each chapter's focus is represented by coloured arrows, showing how these topics are interconnected across different scales and environments. Chapter 2 focuses on the relationships involving soil characteristics, urbanisation, and greenspace type, with a focus on below-ground invertebrate communities. Chapter 3 explores how greenspace characteristics like site area and vegetation within the site influence plant health. Chapter 4 investigates the invertebrate communities of greenspace sites and the interactions between urban greenspace characteristics and vegetation.

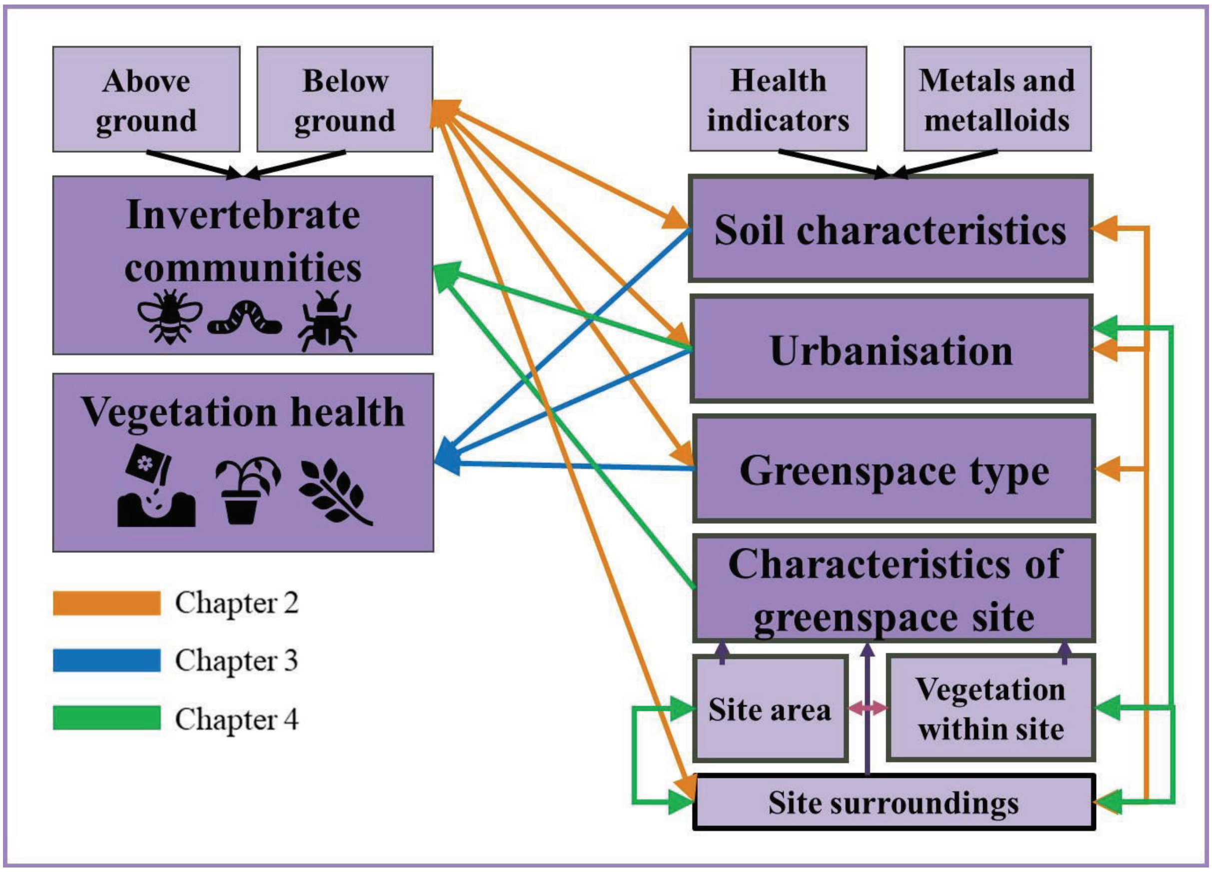


Figure 1. 1 The investigations this thesis has undertaken to further the understanding of factors influencing invertebrate abundance and plant health across a range of urban greenspaces

Chapter 2: Greenspace type and soil characteristics

Despite an increase in urban research, the wider effects of greenspace type on the surrounding ecosystem and ecosystem services, particularly soil characteristics, remain unclear. Many studies consider only one greenspace type (such as garden or allotment soils) or limited invertebrate taxa, such as pollinators. Studies directly comparing different greenspace type and linking the soil properties with invertebrate communities are lacking (Guilland et al., 2018). Secondly, the interactions between urban soils and urban soil invertebrates are scarce. Understanding the influence of greenspace type on this community and the vast ecological services they provide is important in supporting urban invertebrates.

In chapter 2 I assess soil characteristics and invertebrate communities in four key greenspace types (allotment, garden, park grass and woodland) across the urban landscape over a twelve- month period. The aim of this research was to test, (i) the effect of greenspace type on soil characteristics, including soil health indicators and the concentration of metals and metalloids within the soil and (ii) the effect of greenspace type on soil invertebrate communities.

Chapter 3: Greenspace type and plant growth

Plant growth is influenced by complex interactions with climate conditions, soil characteristics, relationships with pests and beneficial organisms, and land management. There is variation in nutrient availability and pollutants between rural sites (characterised by low population density, limited infrastructure, and greater proportions of natural or agricultural land)and urban sites, but we do not know how these variations might affect the potential of plants grown in different greenspace types. This vegetation is essential to deliver ecosystem services in these greenspaces.

In chapter 3, I used soils from four greenspace types (allotment, garden, park grass and woodland) to analyse the effect of greenspace type on plant growth and health across five different plant species. We compared five plants with the aims to (i) assess the effect that greenspace type has on plant health and (ii) understand the soil characteristics driving plant growth and health indicators.

Chapter 4: Invertebrates within inner-city urban greenspaces

The densification of urban areas has placed constraints on inner-city greenspaces, reducing their scale and connectivity. Despite the rapid expansion of urban areas and the increasing biodiversity-enhancing schemes, few studies have been conducted to assess the effect of greenspace type on invertebrates. The entirety of the invertebrate community is rarely assessed when examining the benefits of urban greenspaces, and particularly not with a focus on inner- city areas and small greenspace patches. Many studies select sections of the invertebrate community due to time restrictions, complex sampling schemes, and funding biases. This has led to research-based planting schemes focused on the needs and preferences of these species, rather than the entirety of the community. As the effect on only some species is known, and the schemes are designed without the needs of the entire invertebrate community in mind , it is likely that the effect on invertebrates will not be consistent across the community. Understanding the effect of vegetation and site connectivity on the entire invertebrate community is vital in effective urban planning.

In chapter 4, I used traditional insect sampling methods alongside vegetation assessments of small inner-city greenspace sites to analyse soils from four greenspace types (allotment, garden, park grass and woodland) to analyse (i) analyse the effect of site isolation and connectivity of inner-city greenspaces on invertebrate communities and (ii) assess the vegetation structure and compositions of inner-city greenspaces, and the impact on invertebrate communities.

Chapter 2: The influence of greenspace type on soil characteristics and invertebrate communities in UK urban greenspaces

Abstract

Urban soils are integral to the sustainability and functioning of urban ecosystems. Greenspaces, such as parks, gardens, woodlands and allotments, support a range of ecosystem services including wastewater regulation, carbon sequestration, and habitats and resources for biodiversity. Soils underpin the ecosystem services supported by urban greenspaces, but the quality of these soils will vary according to human management, use and the vegetation in different forms of greenspace. Here, we use positive and negative indicators of soil health including proportions of heavy metals and carbon stocks, alongside soil invertebrate communities, to explore the complex interactions between greenspace type and soil health. The findings show that the greenspace type has a significant effect on the proportion of some metals and heavy metals within the soil, but not on other soil health indicators. We observed temporal variation in soil invertebrate populations across four sampled greenspace types (allotment, garden, park grass and woodland), with consistent patterns across different taxa. Garden soils had highest invertebrate abundance in spring, while park grass and woodland soils had highest abundance in late summer. The impact of greenspace type on soil type and invertebrate communities demonstrates that the way we use land impacts the wider biodiversity of urban spaces and underscores the need for urban greenspace type to be integrated into future urban planning regulations.

Introduction

Urban greenspaces

The growing global population is becoming more urban, with more than 50% of people living in cities and towns. As a result of this urbanisation trend there are issues if both urban densification and urban sprawl globally. In the UK, where >80% of people live in urban areas there are growing pressures on the urban environment (ONS, 2017). With increased urbanisation, there is increased demand for land for development within cities and towns and therefore pressure to expand urban greyspace areas, alongside increasing the density of greyspace within existing urban spaces. The increased demand and construction of infrastructure has led to an increase in greyspace proportions across the UK (FAO, 2020; Robinson et al, 2022). As a result of this pressure and demand for land, urban greenspace land is continuously increasing in social and economic value.

The developmental pressure of urban land has led to de-prioritisation of greenspace preservation within urban spaces (Dallimer et al., 2011; ODPM, 2000). Urban greenspaces are a broad category and encompass a variety of different greenspace types, including parks, allotments, green roofs, gardens, green roofs, and more (WHO, 2016). Each of these greenspace uses provides ecological, social, and health benefits to urban environments, but the extent of these services may differ by greenspace type (Jabber et al., 2022: WHO, 2016). For example, parks and larger green spaces provide substantial areas for social interaction, exercise, and physical activity, but the lack of tree species may limit their heat mitigation potential (Dinda et al., 2021: Gomez et al., 2015). Allotments and community gardens could provide a pathway to enhanced food security, alongside fostering community engagement and education about sustainable growing practices but cropping practices may reduce their support of native vegetation biodiversity (Edmondson et al., 2020b; Kwartnik-Pruc et al., 2023). Woodland spaces and street trees can improve air quality and reduce the urban heat island effect, but have less opportunities for recreation (Pataki et al., 2021; Wolf et al., 2020). These diverse greenspace spaces contribute in multiple ways to the sustainability of urban areas, highlighting the need for urban planning that recognizes their unique contributions to the future urban landscape.

Though typically regarded as less economically valuable than developed urban land or commercial real estate, urban greenspaces provide numerous ecosystem services, such as flood prevention, urban heat mitigation, and sustainable food production (Hunter et al., 2019; Davies et al., 2011; Grafius et al., 2020; Jabbar et al., 2021; Semeraro et al. 2021). The pressure of increased densification within UK urban spaces has led to loss of greenspace in nearly 70% of studied cities within England (Dallimer et al., 2011). Additionally, some areas are not converted to greyspace, but to alternate forms of greenspace. For example, within the 20th century, 65% of allotment sites have been lost to land conversion within urban spaces, whilst further policy reforms favouring densification have led to other greenspace land loss across the UK (Dobson et al., 2020; Robinson et al, 2022; ODPM, 2000). As site loss and conversion is an issue, and many historically greenspace sites are no longer suitable, it is important to preserve and utilise sites to maximise services provided to the urban environment (Boulton et al., 2018; Dobson et al., 2020; Mell, 2018).

To understand the influence of greenspace type and cover on the sites which remain, it is necessary to understand the characteristics driving the ecosystem services that they provide.

Urban soils

Urban soils play a critical role in the ecological, hydrological, and atmospheric functions of urban environments, underpinning the sustainability and health of urban ecosystems (Sauerwein, 2011; Kumar, 2016) including providing a crucial role in supporting greenspaces and the services they provide. These soils provide critical ecosystem services such as carbon sequestration, stormwater management, and mitigation of the urban heat island effect within urban spaces (Scharenbroch et al., 2022; Kumar et al., Fokaides et al., 2016). Moreover, urban soils contribute to biodiversity by providing habitats for microorganisms, invertebrates, and plants, thus supporting urban wildlife and contributing to ecological resilience of the spaces (Szlavecz et al., 2018). They also play a significant role in biogeochemical cycles, influencing nutrient dynamics and pollutant degradation (Fokaides et al., 2016). Understanding the key effects of urban greenspaces on soils and invertebrate communities and effectively managing them is pivotal for improving urban resilience, enhancing ecosystem services, and fostering sustainable urban development in the face of rapid urbanisation and climate change.

The breadth of research into soils in urban spaces has expanded significantly over the past century, with growing recognition of the potential of urban soils. The focus of historic investigations, such as Purves (1972) and Chaney (1984), primarily centred on the influence of soil contamination, rather than soil health or carbon stocks, with documented issues surrounding industrial emissions and waste disposal. In the past, the prevalence of greyspace has led to an underestimation of the potential of soils in urban spaces due to the prevalence of greyspace (Norra et al., 2003; Davies et al., 2011). However, recent research has demonstrated that although urban soils are understudied, they do provide ecosystem services such as carbon storage

(O’Riordan et al., 2021; Canedoli et al., 2020; Edmondson et al., 2014a). Soil science has expanded beyond contamination concerns towards understanding urban soils as integral components of urban ecosystems (Kumar et al., 2016; Scharenbroch et al., 2018). This shift is driven by growing awareness in research and policy of soil biodiversity’s crucial role in urban ecosystem health, resilience, and climate regulation. Current research demonstrates the diversity and range of services across urban soils, examining their roles in supporting biodiversity, wastewater retention, and contribution to climate resources (O’Riordan et al., 2022: Sauerwein, 2011). This shift in research pathways demonstrates an increasingly interdisciplinary approach, integrating soil science with ecological and public health perspectives to address the complex challenges o urban development.

Soil health is a fundamental component of ecosystem sustainability, agricultural productivity, and environmental quality. In the context of urban ecology, ‘healthy soils’ refer to those capable of sustaining plant productivity, supporting belowground biodiversity, enhancing water and air quality, and contributing to human well-being (That et al., 2020). The importance of soil health extends far beyond agricultural landscapes; in urban areas, healthy soils are integral to ecosystem functioning, infrastructure resilience, and the delivery of key ecosystem services. A wide range of indicators are used to assess soil health, broadly grouped into biological, physical, and chemical categories. Biologically, a healthy soil hosts a rich community of microorganisms, fungi, and invertebrates. These organisms drive essential processes such as organic matter decomposition, nutrient cycling, and aggregate formation, and serve as bioindicators of pollution and degradation (Seaton et al., 2021; Furey et al., 2021). Microbial activity, enzyme presence, and invertebrate abundance all reflect the vitality and resilience of the soil system. Biotic diversity also supports aboveground vegetation by improving nutrient availability and facilitating symbiotic relationships, thereby reinforcing a positive feedback loop between soil and plant communities.

Physically, healthy soils exhibit good structure and low levels of compaction, commonly measured through bulk density (BD). Lower BD values typically reflect better porosity and aeration, which in turn support root penetration, water infiltration, and gas exchange. Soil structure is also closely tied to organic matter content, which binds soil particles together and improves resistance to erosion. Organic matter, particularly in the form of Soil Organic Carbon (SOC), plays a leading role in maintaining soil fertility, water-holding capacity, and long-term carbon sequestration (Shaxson et al., 2008). SOC levels are often higher in well-managed urban greenspaces, including allotments, where regular inputs of compost and manure are common (Edmondson et al., 2014a; Dobson et al., 2021).

Chemically, healthy soils maintain a balanced pH and adequate levels of essential nutrients such as nitrogen, phosphorus, and potassium. Excessive or insufficient nutrient concentrations can impair plant growth and microbial functioning. Moreover, in urban environments, the presence of heavy metals and other pollutants can compromise both ecological and human health. Monitoring these chemical parameters and comparing them with benchmark values such as Soil Guideline Values or Suitable 4 Use Levels (S4ULs), is therefore critical for assessing urban soil safety and suitability.

Urban soils also play a significant role in climate regulation through their capacity to store carbon. In addition to SOC, black carbon—formed from incomplete combustion of fossil fuels—can accumulate in urban soils, reflecting past land use and atmospheric deposition. These carbon pools, while sometimes overlooked, contribute to the long-term carbon balance of cities and represent an important component of climate mitigation strategies (Rate et al., 2022).

Fundamental to the maintenance of soil health is its relationship with vegetation. Vegetation affects soil through litter deposition, shading, moisture retention, and root exudation. These processes influence soil temperature, organic input, and microbial habitats, creating feedback loops that enhance or degrade soil quality depending on the type and stability of vegetation present (Furey et al., 2021; Nicolay et al., 2021). In turn, healthy soils provide optimal conditions for vegetation growth, highlighting the mutually reinforcing nature of this relationship. These dynamics are particularly pronounced in urban greenspaces, where vegetation structure varies widely—from closed canopy woodland to frequently mown park grass or tilled allotments.

While healthy soils generally support a broader range of plants and ecosystem services, some plant species have evolved to thrive in poor or nutrient-poor soils. Therefore, soil quality needs can vary substantially depending on the plant community and ecological context. Rather than targeting a single ideal soil type, urban greenspace management should embrace soil heterogeneity to support diverse plant and invertebrate communities. However, plant health is only one consideration in soil management, and to manage for the range of vegetation which may be present, it is important to understand which management practices promote for healthy soil. Ensuring soil health is thus essential only in agricultural environments, but in all habitats. Protecting soil health in urban areas will provide a more robust surface for these environments, but ensuring and protecting this health necessitates planning and soil management policies which prioritise long term potential.

Fundamental to effective soil health management is the relationship between soils and vegetation. Vegetation can affect soils through stabilising structure, changing organic matter input, and facilitating nutrient cycling processes. There are complex but clear interactions between these vegetation and soils, with one influencing and reinforcing the other's health and function (Edmondson et al., 2014a; Nicolay et al., 2021). Healthy soils provide an optimal environment for vegetation growth, allowing the increased production of vegetation. Vegetation is documented to benefit soil health through increased litter deposition, water filtration, and creating a greater range of habitats for subterranean communities (Furey et al., 2021).

Additionally, vegetation cover affects soil moisture and temperature through water runoff, storage, and filtration, alongside shading ground from sunlight (Nicolay et al., 2021; Lui et al., 2021). These differences are particularly key in urban spaces, which have distinct divides in greenspace types. This distinctiveness arises from the human influence within urban spaces, where greenspaces are intentionally designed for different purposes, from recreation and aesthetics to urban agriculture. Additionally, urban vegetation is frequently fragmented by infrastructure, intensively managed, and shaped by socio-economic decisions. A woodland urban greenspace patch has a very different vegetation structure to an amenity grassland park, or to an allotment, where vegetation is regularly removed. Because vegetation structure and site management are deeply interlinked, with management shaping what vegetation can persist, considering both together provides a more complete picture of how urban soils function. The effect of vegetation and management are therefore important factors to consider together when investigating wider effects on soils within greenspaces to protect the integrity of both aspects of the environment.

The management of land is intrinsically linked to the health of soils. Management practices vary across urban spaces, with the desired use of a site necessitating a particular vegetation structure or maintenance. For example, an area of park grass, used as a sports field, will be subject to regular mowing, while an allotment site will likely undergo regular tilling, artificial inputs (in the form of fertilisers and pesticides) and manure applications. These management practices influence not only the soil, but the wider site through the soil health. Differences in maintenance, such as mowing and fertilising schedules, can greatly affect the nutrient and organic matter deposition within the sites, thereby influencing the health of the site (Williams et al., 2020; Keck et al., 2020). Mowing can increase inorganic nitrogen concentrations within the soil, while mulching of waste can lead to higher nutrient input in soils (Berger et al., 2024; Ning et al., 2023). Allotment sites, which regularly see inputs of organic matter and manure, are of higher quality than commercial horticultural soils (Dobson et al., 2021). Edmondson et al (2014a) demonstrated that over three quarters of allotment holders added manure to sites, and 95% composted on site, leading to high levels of Soil Organic Carbon (SOC), Carbon: Nitrogen (C: N) ratios and Bulk Density (BD) of soils. Conversely, 42.8% of UK gardeners used pesticides, which can degrade soil health by reducing microbial and invertebrate diversity (HSE, 2019). The interplay between management thereby acts within a system, with healthier soils better situated to cope with pollution and depletion, while degraded soils from poor management practices necessitate greater inputs and interventions to restore them to their natural productivity. Understanding these interactions is important in effectively managing urban areas as it enables targeted soil and vegetation management strategies that optimise ecosystem services, reduce maintenance costs, and improve site-specific outcomes, whether that be flood mitigation, biodiversity support, or food production.

Invertebrates are critical to the maintenance of ecosystem functions and services. They provide a plethora of ecological services, both to the site they inhabit and the surrounding area (Eisenhauer et al., 2021). Organisms contribute to soil health through nutrient cycling, soil structure formation, and organic matter decomposition (Szlavecz et al., 2019). They are a fundamental part of wider food webs, providing food sources for many bird and mammal species (Noriega et al., 2018). Additionally, invertebrates play a role in pest control, with predatory species helping to regulate populations of agricultural and garden pests. Understanding and preserving invertebrate diversity is therefore essential for sustaining overall ecosystem health, resilience, and productivity and requires focused conservation efforts and sustainable management practices.

Particularly key in many systems are soil invertebrates, such as earthworms, Collembola, mites and nematodes (Noriega et al., 2018). These groups play specific and critical roles that support soil health and stability.. They play a key role in organic matter decomposition by breaking down complex materials into simpler forms. These simpler materials can then be further processed by microorganisms, which mineralise nutrients and release them into the wider soil environment, thereby increasing soil fertility. Continued interactions between soil invertebrates and microorganisms amplify nutrient availability and support a dynamic soil ecosystem (Lavelle et al., 2006). Other taxa, in particular earthworms, significantly influence soil structure. Often referred to as ecosystem engineers, through activities of burrowing and movement, they create tunnels through the soil, increasing aeration, water infiltration, and ease of root penetration (Blouin et al., 2013). Further to this, burrowing invertebrates such as ants and beetles also contribute to this soil structure formation, whilst additionally promoting the distribution of organic material throughout soils. Soil invertebrates are linked to the aggregate stability of soils, which improve the soil structure, making it more resistant to erosion and water damage (Lavelle et al., 2020). The presence of a diverse soil invertebrate communities is thus indicative of healthy soil systems, making their conservation crucial for sustainable soil management within all systems (Lavelle et al., 2020; Griffiths et al., 2021).

Though historically understudied, research into invertebrates in urban spaces has increased considerably in the 21st century, reflecting broader trends in urban ecology (Noriega et al., 2018). Initial research focused on pollinators, driven by interests in horticulturally and socially valuable species (Ayers et al., 2021) but as urbanisation accelerated, interest grew in other taxa, including ground and soil invertebrates (Noriega et al., 2018., Jones et al., 2013). Prior research has been focused on the presence and absence of macroinvertebrate soil species along with microorganism abundance (Smetak et al., 2007; Joimel et al., 2021). However, the wider community structure of invertebrates has not been a focus (Szlavecz et al., 2020). Recent publications have emphasised the importance of understanding soil communities to effectively understand these urban spaces (Szlavecz et al., 2020). This shift in focus underscores the growing desire to effectively understand and manage urban ecosystems and the key factors affecting these invertebrate communities.

Prior studies into soil invertebrates have focused on agricultural habitats or on a subset of taxa, making the overall effect of greenspace type on soil invertebrates difficult to define (Szlavecz et al., 2020). Key drivers of invertebrate communities in agricultural environments are likely key drivers in urban soils. Previous research has demonstrated that intensification of artificial inputs and soil disturbance can lead to reductions in the overall abundance of invertebrates and the diversity of soil invertebrate communities (Beaumelle et al., 2023). It is likely that disturbance and pollutants in urban soils will also damage the complexity of soil communities, while a site with less anthropogenic influence and inputs could allow a more complex community to develop. This is because disturbances such as soil compaction, chemical pollutants, and habitat disruption can reduce habitat quality and resource availability for sensitive soil organisms. Pollutants may be toxic to certain species, leading to declines in diversity, while physical disturbance can disrupt the intricate soil structure needed for diverse microhabitats. In contrast, sites with lower anthropogenic influence typically have more stable conditions and organic matter inputs, fostering a richer and more complex soil community that supports greater ecological functions. Numerous studies into patterns of habitat isolation on invertebrate species have been undertaken, though the effect of greenspace type has not been considered. These have shown that increased connectivity within sites can assist in the dispersion of invertebrates (Vergnes et al., 2012; Braaker et al., 2014; Wenzel et al., 2020). The effect of site fragmentation has been shown a barrier to species diversity and movement by Riutta (2016), with edge effects of sites reducing feeding activity by 17%. Only through understanding the unique structure of urban green spaces on invertebrate community size and structure can we fully understand the effect of greenspace planning and use on soil invertebrate health and community.

Though the impact of urban greenspace type on soil invertebrates is poorly studied, there is a greater depth of literature regarding the influence of vegetation. Habitat and landscape complexity are important drivers of invertebrate abundance and diversity in agricultural sites, with vegetation type, composition, and structure affecting soil invertebrate communities through habitat, and resource effects (Mancini et al., 2023). Diverse plant species on the surface allow resources for a wider range of invertebrate species, particularly those adapted to favour specific vegetation. Root depth affects soil chemistry and nutrient availability, which in turn affects the composition and activity of soil invertebrate communities (Marschner et al., 2023). Certain plant characteristics can also attract specific soil invertebrate taxa by providing suitable microhabitats for nesting and foraging, such as the positive effect of oak trees on ant species diversity (Bluhm et al., 2019; Gaytán et al., 2021). Additionally, vegetation cover influences soil microclimate, moderating temperature fluctuations and moisture levels that directly impact soil invertebrate physiology and activity (Stoutjesdijk et al., 2015). Understanding these complex relationships between vegetation and soil invertebrates is essential for ecosystem management and conservation efforts aimed at maintaining biodiversity and enhancing ecosystem services provided by soil communities. By recognising how vegetation types and management practices influence invertebrate habitats and resources, managers can tailor interventions - such as selection of appropriate plant species, minimising soil disturbance, and reducing chemical inputs - to support diverse and functional soil communities. Thi intervention ultimately enhances overall ecosystem resilience and health.

The effect of greenspace type on soil characteristics and invertebrates is key in understanding their ecosystem services, but the links are currently not fully understood. To achieve the full potential of these spaces, it is important to manage them in the most effective way. Being able to compare soil health across different greenspace types will allow greater understanding of the potential services these spaces could deliver and provide a foundation for future research links into wider ecosystem services, such as vegetation growth, carbon storage, and water runoff capture. Understanding these interactions, and planning for them in future research, will allow more detailed and more effective urban planning strategies.

Research focus

Urban spaces are extremely valuable, providing a range of ecosystem services such as carbon storage, water retention and a habitat for urban species. Urban soils and urban invertebrates assist in provision of many of these services, but the effect of greenspace type on the soil characteristics and invertebrate community composition is understudied. With this context, we proposed two key research questions for investigation:

- How does greenspace type affect soil characteristics?
- How does greenspace type affect soil invertebrate communities?

We hypothesize that greenspace type will significantly influence both soil physicochemical properties and invertebrate community composition, driven by variation in vegetation structure, management intensity, and historical land use. Actively managed greenspaces, such as allotments and gardens, are expected to exhibit higher soil fertility and elevated metalloid levels due to regular inputs of compost and fertilizers, whereas less-managed greenspaces like woodlands are likely to display greater variability in soil properties, reflecting limited human intervention and slower organic matter decomposition. Furthermore, we predict that soil invertebrate communities will be more abundant and diverse in gardens and allotments as a result of greater organic inputs, habitat heterogeneity, and plant diversity, while park grasslands will support lower invertebrate abundance and diversity due to soil compaction, uniform vegetation, and reduced resource availability.

Through a year-long soil sampling campaign, we attempted to comprehensively assess the soil health and invertebrate communities of greenspaces, and the effect that greenspace type has on these characteristics. We collected soil samples from four key greenspace types across the city of Sheffield (allotment, woodland, garden and park grassland) across the year to assess temporal variation in characteristics. We took soil samples to assess the metals and metalloids present within the soil, alongside carbon stocks, bulk density (BD), pH and vital nutrients within the soil. We extracted invertebrates from soils from each site to assess the temporal variation in community structure. This work has been used to assess the influence of greenspace type on invertebrate community and soil quality.

Materials and methods

Study system

Soils were sampled from greenspace sites across 2021-2022 in Sheffield, England (53.3811° N, 1.4701° W). Sheffield was selected as a representative mid-sized UK city due to its hih greenspace availability and history of greenspace research but is acknowledged that using one city as a representative presents difficulties due to differences in climate, land use history, management practices, and socio-economic factors. These differences mean results may vary in other urban contexts. Future studies comparing multiple cities would help determine the generalizability of these findings. We selected four greenspace types for sampling: allotment, garden, park grass and woodland.

They were defined as: allotment sites (a parcel of land divided into smaller plot, typically rented from a local authority or private owner, where individuals can grow their own fruits, vegetables, and flowers); garden site (a parcel of land typically adjoining a residential property, in which plants such as flowers, vegetables, trees are cultivated for aesthetic, practical, or recreational value are grown); park grass (a park refers to an area of public or publicly accessible land that is maintained for recreational use or leisure. These spaces typically feature multiple land covers, such as lawns, trees, and sports fields, with park grass hereby referring to a section of amenity grass); woodland sites (a parcel of land characterised by dense tree and shrub cover, typically features a variety of tree species, undergrowth, and ground vegetation. Here, a woodland additionally included over 40% canopy cover).

These greenspace types exemplify the variety of greenspaces present across the city, with each managed for different objectives and with differences in anthropogenic influence. They are common greenspace types within the UK. Gardens make up 38% of the city's greenspace (Edmondson et al., 2011). Allotments provide a key resource for urban agriculture. Parks are demonstrated to benefit mental and physical health, while trees contribute disproportionally to ecosystem services provided within urban areas. We sampled from 10 sites of each greenspace type, with samples taken from multiple plots (2-4 plots) at each allotment to reduce the effect of individual management. This gave a total of 52 sample sites across 40 greenspaces (Figure 2.1). The definition of "urban" is inherently ambiguous, and in this context, sites were included where urban pressures, such as habitat fragmentation, human activity, and adjacent land use, were present. To capture variation across the urban gradient, we identified the point of highest greyspace concentration in Sheffield city centre using the 'Calculate Density' tool in QGIS, . We then used a 'multi-ring buffer' to create a set of five 2km buffers radiating from this point. We identified park greenspaces and allotment sites within these buffers using OS MasterMap. We identified woodland sites using the National Tree Map, where we merged the crown of each tree point to identify areas of continuous woodland. Two sites in each greenspace type were then selected from each buffer using the 'random selection' tool. Although some selected sites are located outside the desired buffer zone, they were chosen due to the lack of accessible, comparable greenspaces within the buffer. All sites fall within the Sheffield council boundary. These sites share similar land-use histories, management practices, and urban influence, making them appropriate inclusions within this study. To additionally ensure that all sites were located within urban areas, the proportion of greenspace in a 1km buffer surrounding each site was calculated.

The proportion of greenspace was under 40% for all sites, and under 30% for 34 of the 40 sites, demonstrating the urban setting of the sites.

An area limit of <30,000m^2 was selected as this included the majority (80%) of all allotment sites. A lower boundary of 5,000m^2 was selected, as an allotment plot of this size would have approximately 20 plots (with average plot size of 250m^2, Edmondson 2020b), which was determined as the minimum to allow adequate recruitment of plot holders

Random point inside polygon' was used to generate a sampling point at each greenspace site. We took soil samples as close as possible from these generated points, whilst accounting for access (undergrowth density, allotment holder permission), minimising public disturbance (proximity to trails, park equipment, and structures), and successful sampling technique (soil depth, tree root depth). Within park grassland sites, we took soil samples from amenity grassland (mown, closely maintained grass patches) at least 20 metres from other land cover types to reduce influence of community mixing. This distance was selected as the largest distance possible whilst enabling replicability across all sampling. Within gardens, we sampled from a random point within the planted borders/flowerbed of the site, avoiding amenity grassland or food growing spaces. This was selected as it was the only land cover present across every garden sample site, making it the only environment possible to be repeatable across all garden sites.

Within allotments, we sampled from a random point from a section of the site actively (within six months from June 2021) used for growing food crops, whilst avoiding areas with freshly planted seeds and active crops to avoid damage. While we recognise that for some allotment sites, the entire site is not typically under active cultivation, Sheffield City Council states that at least 75% of an allotment plot should be used for cultivating fruit and vegetables. This approach was therefore intended to capture the majority of allotment plot use, alongside soil conditions representative of cultivated plots specifically, as these are likely to be most affected by management practices relevant to soil health.

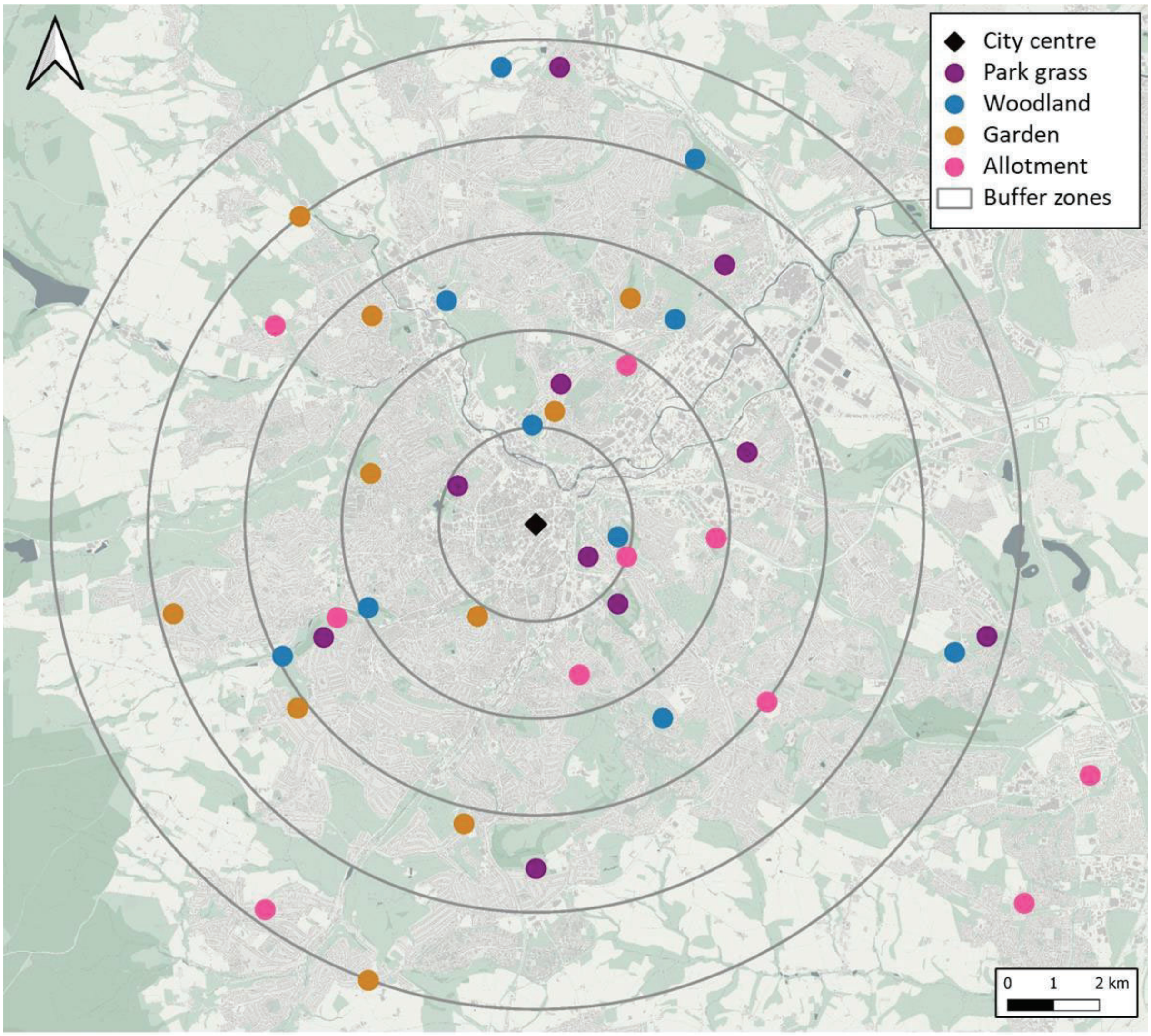


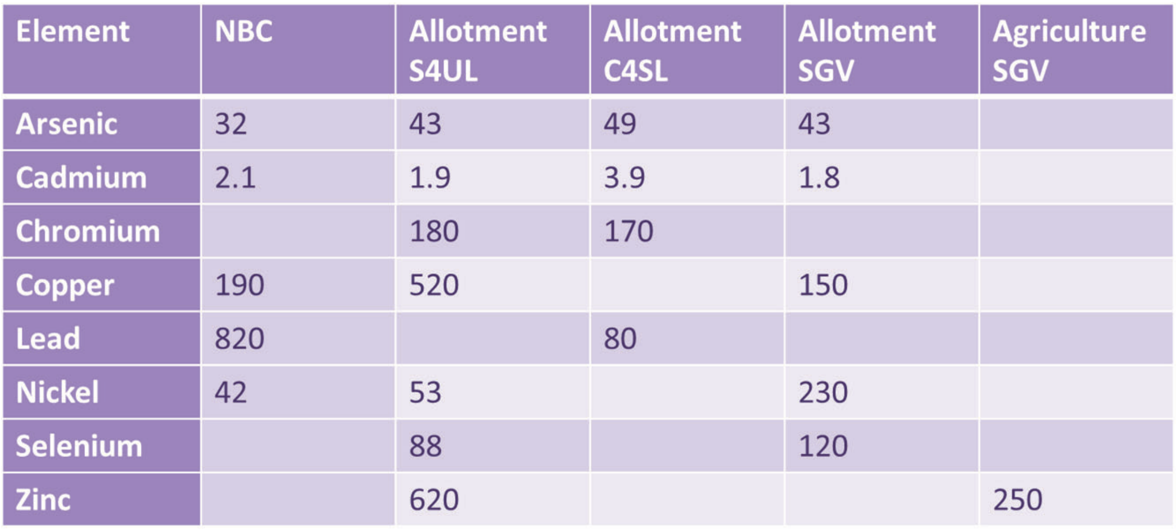
Figure 2.1. City level sampling strategy for study city Sheffield (purple dots; park grass sites, blue dots; woodland sites, yellow dots; garden sites, pink dots: allotment sites)

Six total sampling periods were undertaken between July 2021-June 2022. Sampling periods were six weeks apart during the summer (April-October) and three months apart during the winter (October-April). Each sampling period was a duration of 14 days, and each of the 40 greenspace sites was sampled within this time period to minimise the impact of weather and temporal change. All sampling except earthworm sampling was undertaken during this period. Sampling order between individual greenspaces was randomised to ensure no temporal effects within the sampling period.

Elemental composition of soils

At each sample site, four soil samples were taken using a specialist soil corer for bulk density (BD) sampling (Eijkelkamp, ring kit C53). Each soil core has a volume 100 ml (maximum volume deviation: 0.5%). The four samples were used to find an average BD for each site (Edmondson et al. 2011).

These four soil cores were then bulked, sieved (2 mm), and dried at 80°C for 24 hours. The samples were ball milled for five minutes to homogenise. Dried and milled samples were weighed to estimate BD. To determine metals and metalloids present, we processed soils with an Aqua Regia Digest. One gram of soil from each site (+/-2µm) was put into a glass tube, before 2ml of 67% HNO3 was added followed by 6 ml of HCl 37%. Temperature was slowly increased to 120°C for 2 hours. Tubes were left overnight to cool and complete the digestion. We then filtered the solution through 41 Whatman paper and added ultrapure water to make up to 50 ml with UP water. Samples were then analysed using Inductively Coupled Plasma Mass Spectrometry (ICP-MS) to quantify total soil content of Arsenic (As), Cadmium (Cd), Chromium (Cr), Copper (Cu), Nickel (Ni), Lead (Pb) and Zinc (Zn). Metal and metalloid values were compared against previous recorded values and recommended levels for human safety. These consisted of National Background Concentrations (NBCs), Suitable for Use Levels (S4UL), Category 4 Screening Levels (C4SL), and Soil Guideline Values (SGV's). These values are given by the British Geological Survey and can be used as a guide for normal levels of contaminants (NBC) in English soils (Table 1) and the levels beyond which there is a risk to human health (S4UL, C4UL, SGV).



Element	NBC	Allotment S4UL	Allotment C4SL	Allotment SGV	Agriculture SGV
Arsenic	32	43	49	43	
Cadmium	2.1	1.9	3.9	1.8	
Chromium		180	170		
Copper	190	520		150	
Lead	820		80		
Nickel	42	53		230	
Selenium		88		120	
Zinc		620			250

Table 1. Normal Background Concentrations (NBC's)(Johnson et al., 2012) and soil screening values (S4UL, C4SLs and SGVs)(Environment Agency, 2009) for the heavy metal and metalloids investigated. Values are expressed in mg kg−1 soil dry weight.

To demonstrate the overall composition of the site, a weighted index of metals and metalloids was created. This was performed following the method outlined in Wang (2017). This method integrates both the relative loading of each metal/metalloid (derived from the PCA) and its proportional concentration at each site to create a composite score that reflects overall contamination levels. Using a PCA of the metals and metalloids, the loading value of each element was used to calculate a relative weight (Table 1). The weighted index value of each site was then calculated using the formula:

$$\text{Index Value} = \text{Weight Value} \times \sum \left(\frac{n}{\sum n} \times 100 \right)$$

Where:

- ‘Weight Value’ represents the PCA loading value of each metal/metalloid (typically from PC1, which explains the highest variance),
- ‘n’ is the concentration of an individual metal/metalloid at a site,
- ‘∑n’ is the total concentration of all included metals/metalloids at that site.

The index value of each element was then summed to give one value for each site. This weighted index enables direct comparison across sites, accounting for both the importance (via PCA) and magnitude (via raw concentrations) of each element. The resulting values provide a more integrated assessment of site contamination than unweighted concentration measures alone.

Soil carbon analysis

Organic carbon values were found through chemical oxidation. Samples were weighed (<10 mg total weight and 200 µg C) and placed into Ag capsules. Each sample was wetted with 25 µl of water before the addition of 25 µl of 1 M HCl. The samples were covered with aluminium foil and left at room temperature for 1 hour. After cooling, an additional 50 µl of 1 M HCl was added, followed by a further 30 minutes of cooling. The samples were then dried in an oven at 60 °C. These steps were repeated until effervescence upon acid addition ceased, indicating all reactive carbonates had been removed.

For black carbon values we used thermal oxidation; samples were placed in a muffle furnace and oxidized at 375°C for 24 hours. The oxidized samples were then analysed for black carbon (BC) content using a CN analyser (Gustafsson et al., 2001).

Soil invertebrate sampling

Soil invertebrate sampling was conducted at six timepoints over the course of one year. We sampled 10 sites per greenspace type, including 2–4 allotment plots per site to minimise the influence of individual management practices. In total, invertebrates were sampled from the same 52 sites distributed across 40 greenspaces.

Four soil cores were taken at each site, giving a total volume of 400 cm^3. These four samples were combined and placed in a standard Tullgren funnel system (Burbank 12 funnel model) for a period of 96 hours. Invertebrate samples were processed through the funnel system into 70% ethanol solution, within which they were stored for identification.

We identified soil invertebrates to family level (Collembola) or order level (all other individuals). Identifying individuals to these levels was used due to the volume of individuals and documented issues with identifying mite species to a greater level. Collembola identification was prioritised due to their status as their family-level identification is ecologically meaningful, providing sufficient resolution to detect changes in soil quality and habitat conditions.

A diversity index was used to quantify biodiversity. The Simpson index was selected over the Shannon index due to the focus on dominant species and reduced sensitivity to rare species, making it more effective for assessing community dominance and evenness. As the focus here was on community structure, Simpson's index was determined to be more appropriate for this investigation.

Earthworm sampling

We sampled earthworm populations at two timepoints, October (11/10/2021-25/10/2021) and April (04/04/2022 – 17/04/2022) (Edwards et al., 1996**)**. We sampled earthworm populations at these two timepoints due to the complexity of sampling and due to their increased activity during these seasons, favourable soil conditions, and accessibility.

Following the method of Lawrence (2002) and a circular metal frame (10cm diameter) was driven 1cm into the ground. Commercial 'hot' mustard was mixed with water to achieve a concentration of 15 gl (13.5 g of mustard powder to 900 ml of water) (Valckx, 2011). The mixture was mixed and agitated on site. It was applied across 3 intervals (10 minutes apart, with 4.5g of mustard powder to 300ml of water for each application) for a total sampling time of 30 minutes. This was repeated in three locations at each site, 3m apart from one another in a triangle format. Emerging earthworms were retrieved during the 30 minutes following first expellant application, before we placed them into 80% ethanol. This ethanol was then replaced following by the end of the day to account for fluid loss from earthworm death.

We identified each adult earthworm individual to species level. Juvenile individuals were not identified due to documented issues of visually identifying species using visual keys (Pelosi et al., 2008).

Statistical analysis

Analyses were done in R version 4.12 (R Core Team, 2022).

For analysis of metals and metalloids, a generalised linear model (GLM) with a Gamma distribution and log link was used to analyse metal concentrations. This model was selected because the response variables were continuous, strictly positive, and exhibited right-skewed distributions. The Gamma family is appropriate for modelling such non-normal data where variance increases with the mean. This approach allows for interpretable, multiplicative effects of predictors and avoids the need for arbitrary data transformation. Model selection was performed using stepwise regression with the ‘both’ direction in R, enabling the inclusion or removal of predictors based on Akaike’s Information Criterion (AIC) to identify the most parsimonious model. Given that multiple metals and metalloids were analysed simultaneously, p-values were adjusted for multiple comparisons using the False Discovery Rate (FDR) method. This approach controls the expected proportion of false positives while maintaining statistical power, thereby providing a more robust inference framework compared to overly conservative methods such as Bonferroni correction.

Principal component analysis (PCA) was performed on the correlation matrix to provide an overall view of the relations among variables. For the metal and metalloid PCA, measurements of As, Ca, Cd, Cu, Fe, K, Mg, Ni, P, Pb, Se, Zn were used within the PCA. For soil health PCA, measurements of OC, BC, N, P, BD and pH were used. Variables were standardised (z-scores) prior to PCA to remove the influence of differing measurement scales. PCA based on the correlation matrix was selected because variables were measured in different units and exhibited differing variances, making the correlation-based approach more appropriate than covariance-based PCA. Following PCA, multivariate analysis of variance (MANOVA) was used to test whether principal component (PC) scores differed significantly across predefined groups (e.g., land-use types or sampling timepoints). MANOVA was chosen because it allows simultaneous testing of group differences across multiple dependent variables (PC scores), while accounting for correlations among PCs.

To assess the effects of sampling timepoint (October vs. April) and land-use type on earthworm abundance, a Poisson regression model with a log link function was fitted using the glm() function, appropriate for count data. Model assumptions were checked by examining residual plots and testing for overdispersion.

For soil invertebrate abundance, generalised linear models (GLMs) with a Poisson or negative binomial distribution (as required) were used to evaluate the effect of greenspace type across six timepoints. Results were expressed as incidence rate ratios (IRRs) with 95% confidence intervals to aid ecological interpretation. To examine differences in invertebrate community composition, an Analysis of Similarities (ANOSIM) was performed using the anosim() function in the vegan package. ANOSIM was used to test for differences in community structure between predefined groups (greenspace types and sampling seasons) based on Bray–Curtis dissimilarities.

All models included greenspace type and sampling timepoint as fixed effects, and significance was assessed at $\alpha = 0.05$. Where relevant, post hoc pairwise comparisons were performed using the emmeans package with p-values adjusted for multiple testing (Tukey method).

To assess spatial autocorrelation in soil and invertebrate variables, Moran’s I was calculated using the spatial weights matrix derived from the geographic coordinates of the sampling sites. This analysis was performed in R using the spdep package. Non-significant Moran’s I values ($p > 0.05$) indicated that residuals from the models were not spatially autocorrelated, confirming that spatial clustering did not bias the results.

Results

Soil characteristics

Metals and metalloids

There was a significant effect of greenspace type on four soil metal concentrations: Ca ($F(3, 36) = 15.8$, $p < 0.001$, $R^2 = 0.46$)(Figure 2.2 a), Cd ($F(3, 36) = 4.9$, $p = 0.014$, $R^2 = 0.25$)(Figure 2.2 b), Pb ($F(3, 36) = 6.9$, $p = 0.002$, $R^2 = 0.28$)(Figure 2.2 c), and Zn ($F(3, 36) = 12.3$, $p < 0.001$, $R^2 = 0.43$)(Figure 2.2 d). These results indicate that metal concentrations varied significantly with land use, with the largest effects observed for Ca. In contrast, no significant effect of greenspace type was observed for As ($F(3, 36) = 0.36$, $p = 0.740$), Cu ($F(3, 36) = 1.3$, $p = 0.290$), Fe ($F(3, 36) = 1.9$, $p = 0.170$), K ($F(3, 36) = 1.3$, $p = 0.240$), Mg ($F(3, 36) = 0.6$, $p = 0.600$), Ni ($F(3, 36) = 2.7$, $p = 0.056$), or Se ($F(3, 36) = 1.3$, $p = 0.300$). For these metals, median concentrations in urban greenspace soils were below the Normal Background Concentrations (NBC) reported for UK soils, consistent with previous studies for Cd, Cu, Fe, Ni, Se, and Zn.

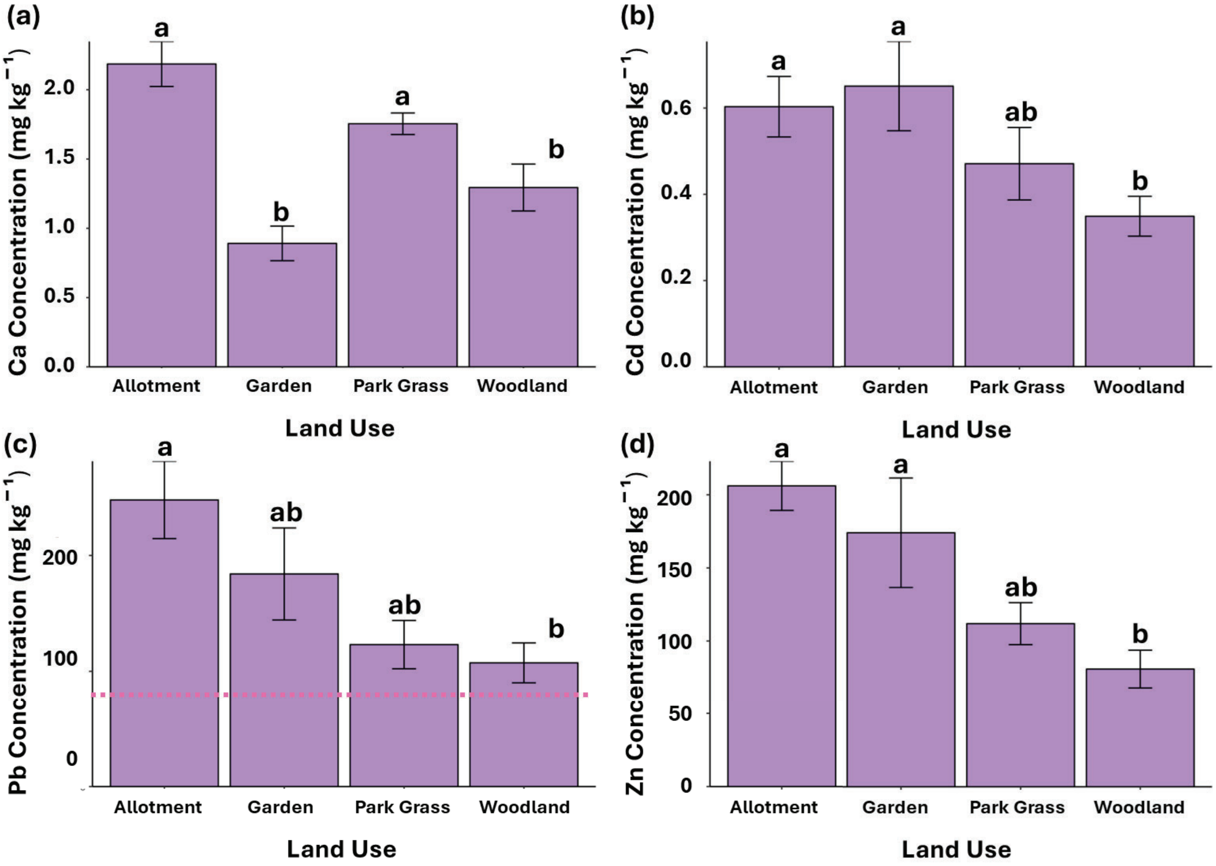


Figure 2.2. Values of (a) calcium, (b) cadmium, (c) lead and (d) zinc values found in allotment, woodland, garden, and park grass soils. Pink dashed lines represent the NBC for the element in urban soils.

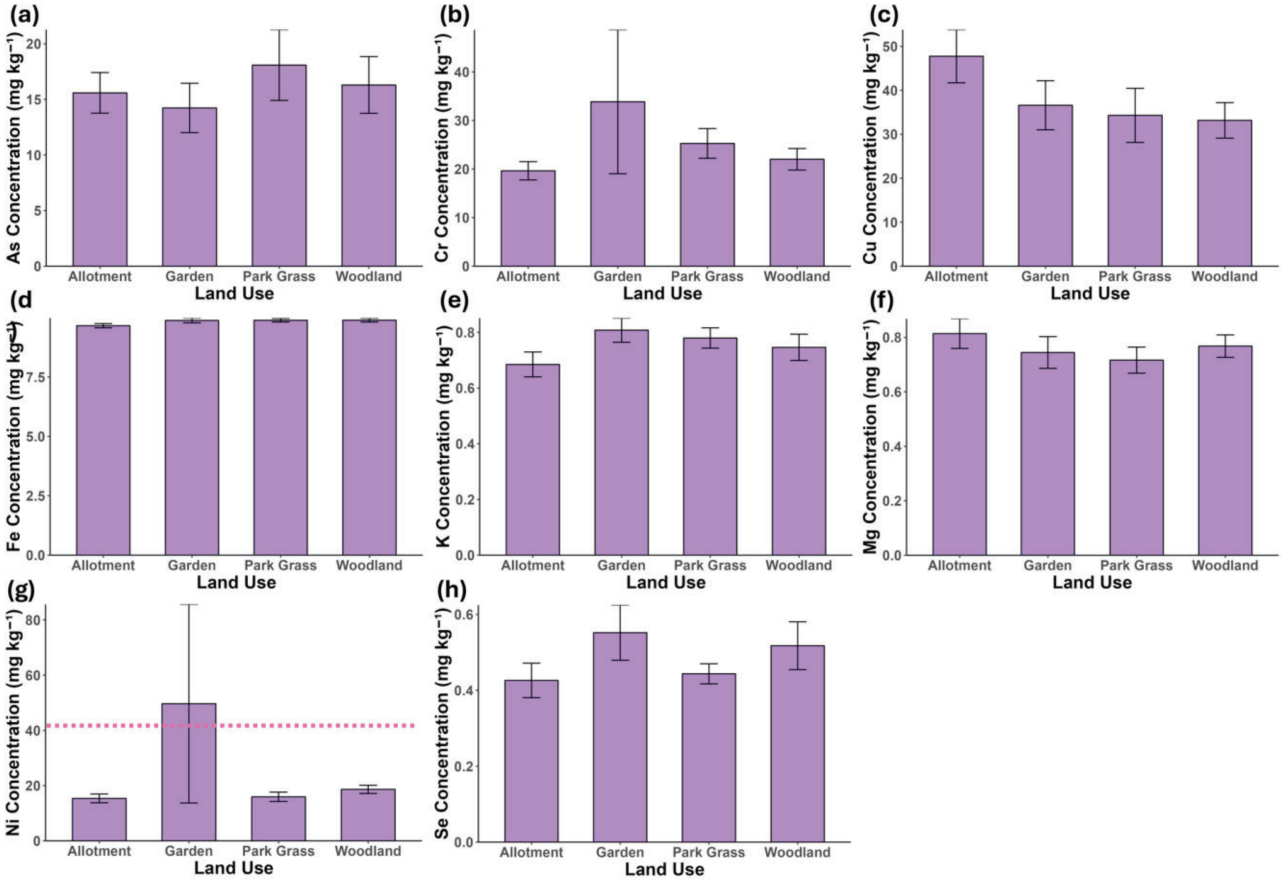


Figure 2.3. Values of (a) arsenic, (b) copper, (c) chromium, (d) iron, (e) potassium, (f) magnesium, (g) nickel and (h) selenium found in greenspace soils. Green box plots represent concentration of essential nutrients (mg kg-¹ soil dry weight), with error bars representing standard error. Pink dashed lines represent the NBC for the element in urban soils.

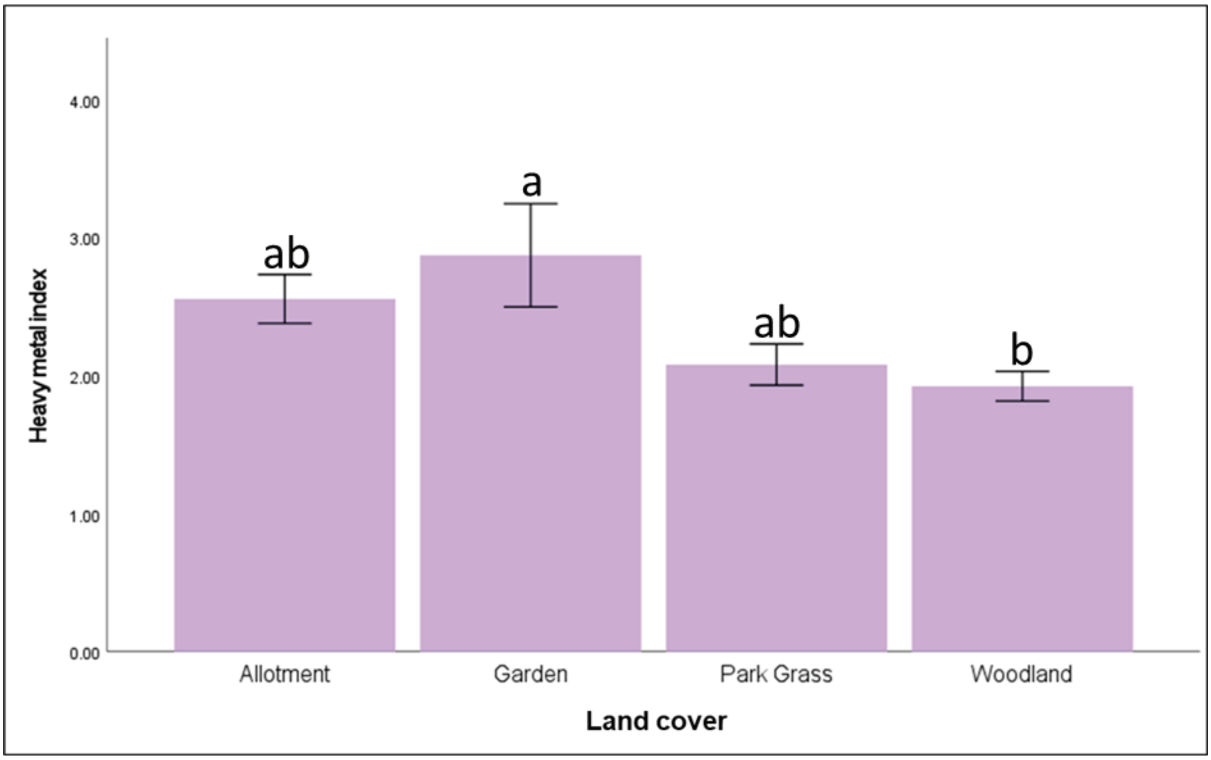


Figure 2.4. Heavy metal index of four greenspace land cover categories based on the variance in 11 metal contaminants.

Greenspace type significantly affected the weighted index of heavy metal proportions and contributions (Figure 2.4) (F(3,36)=3.70, p=0.02). Garden soils had significantly higher index values than woodland soils, indicating that there were higher overall levels of metals and metalloids within garden soils.

For the PCA regarding metal contaminant concentrations, four PCs were extracted from the PCA with eigenvalues >1, accounting for approximately 84.66% of the total variation (Figure 2.5). PC1 accounted for approximately 37.42% of the variance, with strong positive loadings (>0.8) with Cd, Cu and Pb, a moderate positive loading (>0.5) with As, Cr, Zn, Fe. PC2 accounted for 19.67% of the variation, in which Ca had a strong positive loading, while Zn and Mg had weak positive loading (<0.5). A moderate negative loading was observed in Cr and Ni, with weak negative loading observed in Fe.

This PCA revealed partial separation among greenspace types based on metal contaminant profiles. Allotment and Garden soils formed more distinct and compact clusters, suggesting consistent metal compositions. In contrast, Park Grass and Woodland soils showed greater within-group variability and overlapping distributions, particularly along PC2. These patterns suggest that while greenspace type influences metal composition, shared urban factors or environmental gradients may also contribute to similarities between site types.

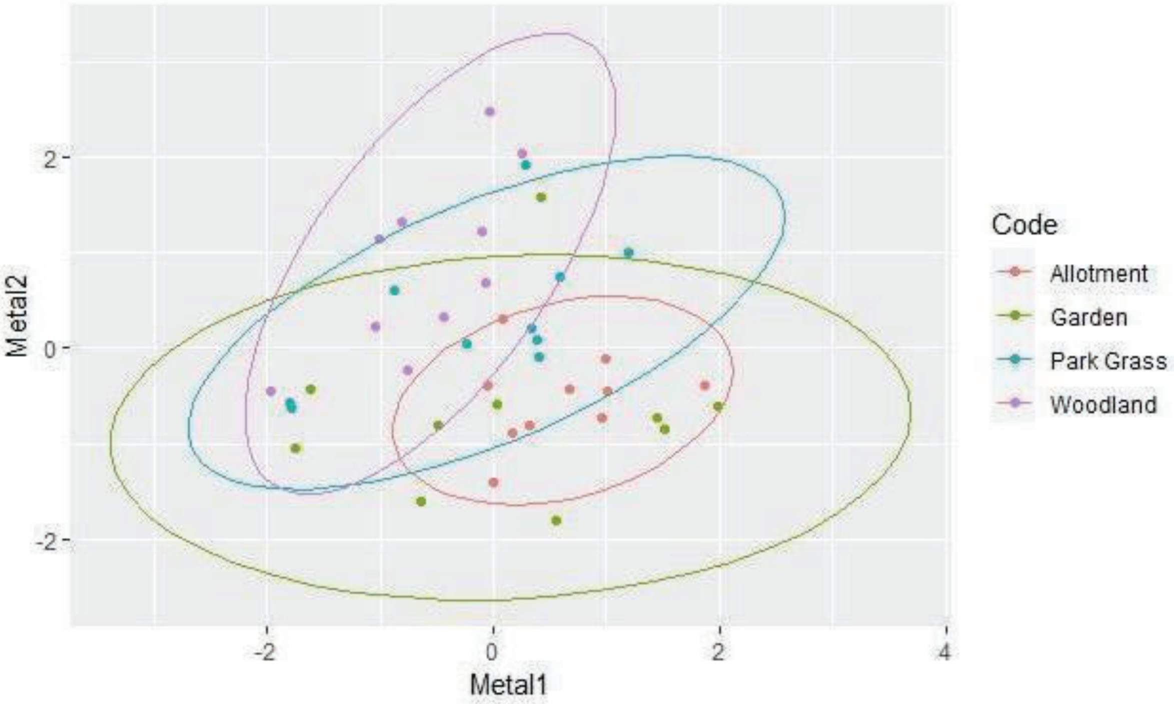


Figure 2.5. Principal Component Analysis (PCA) biplot based onr greenspace type based on the variance in 11 metal contaminants. The first two components explained 37.42% and 19.67% of the variances, respectively. The different colours denote the greenspace type category represented.

Soil health characteristics

There was a significant effect of greenspace type on two soil health characteristics; P (F(3,36)=8.65, $p<0.001$)(Figure 2.6 c) and pH (F(3,36)=12.94, $p<0.001$)(Figure 2. 6.c). Woodland soils had a lower pH and lower total P concentration (mg kg-1) than both allotment and garden soils.

There was no significant effect of greenspace type on any other soil health characteristics; N (F(3,36)=1.40, p=0.26)(Figure 2.6.b), TC (F(3,36)=0.38, p=0.77)(Figure 2.6.d), OC (F(3,36)=0.52,p=0.67)(Figure 2.6.e), BC (F(3,36)=2.53(Figure 2.6.f), p=0.07) and BD (F(3,36)=1.66, p=0.19)(Figure 2.6.a).

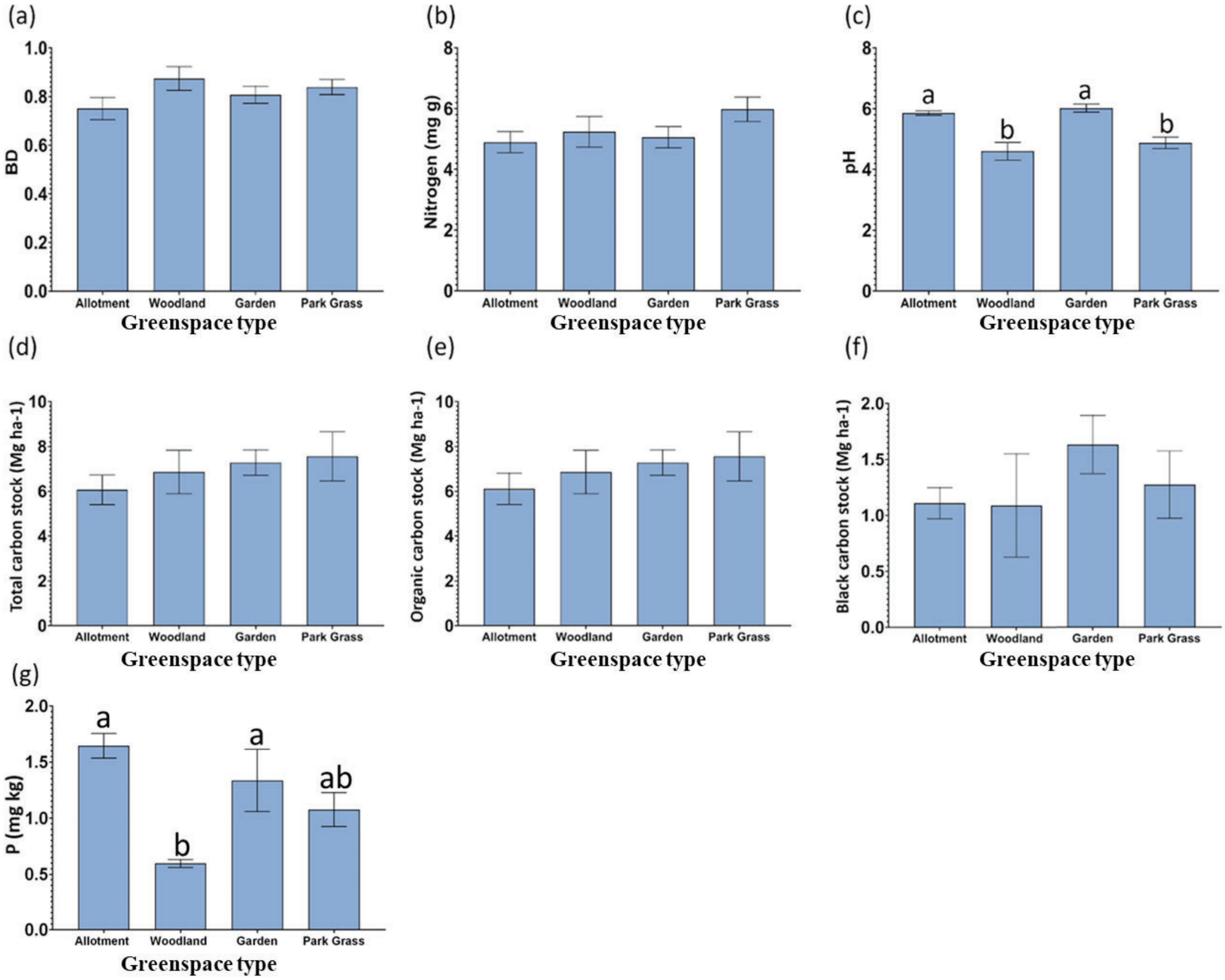


Figure 2.6. Values of (a) bulk density, (b) nitrogen, (c) pH, (d) total carbon stock, (e) organic carbon stock, (f) black carbon stock and (g) phosphorus found in allotment, woodland, garden, and park grass soils. Blue bars represent mean values with error bars representing standard error.

With eigenvalues >1, three PCs were extracted from the PCA regarding soil health characteristics, accounting for approximately 77.20% of the variation (Figure 2.7). PC1 accounted for approximately 34.24% of the variance, with strong positive correlations with N (0.80) and OC (0.93), moderate positive correlation with BC (0.77) and weak positive correlation with Se (0.43). PC2 accounted for 22.09% of the variation, in which pH had a strong positive loading (0.88), while P had moderate positive correlation 0.588) and Se had weak negative correlation (- 0.493).

This PCA reveals that different greenspace types are associated with distinct soil health profiles. Allotments and gardens tend to group together, indicating similar soil fertility likely driven by active management practices such as composting and soil amendments. In contrast, woodland soils show greater variation and a shift along the second principal component, suggesting differing chemical properties, particularly higher variability in pH and selenium levels. These patterns highlight how greenspace type influences both the consistency and composition of soil health characteristics, which may in turn impact ecosystem functioning. Together, PC1 and PC2 represent gradients of soil fertility and chemical composition. PC1 captures variation in organic nutrient content, while PC2 distinguishes soils based on pH and elemental composition. The combination of these axes enables comparison of greenspaces along multiple soil health dimensions, revealing, for example, that allotments exhibit nutrient-rich, chemically stable profiles, while woodland soils tend to be more variable and chemically distinct.

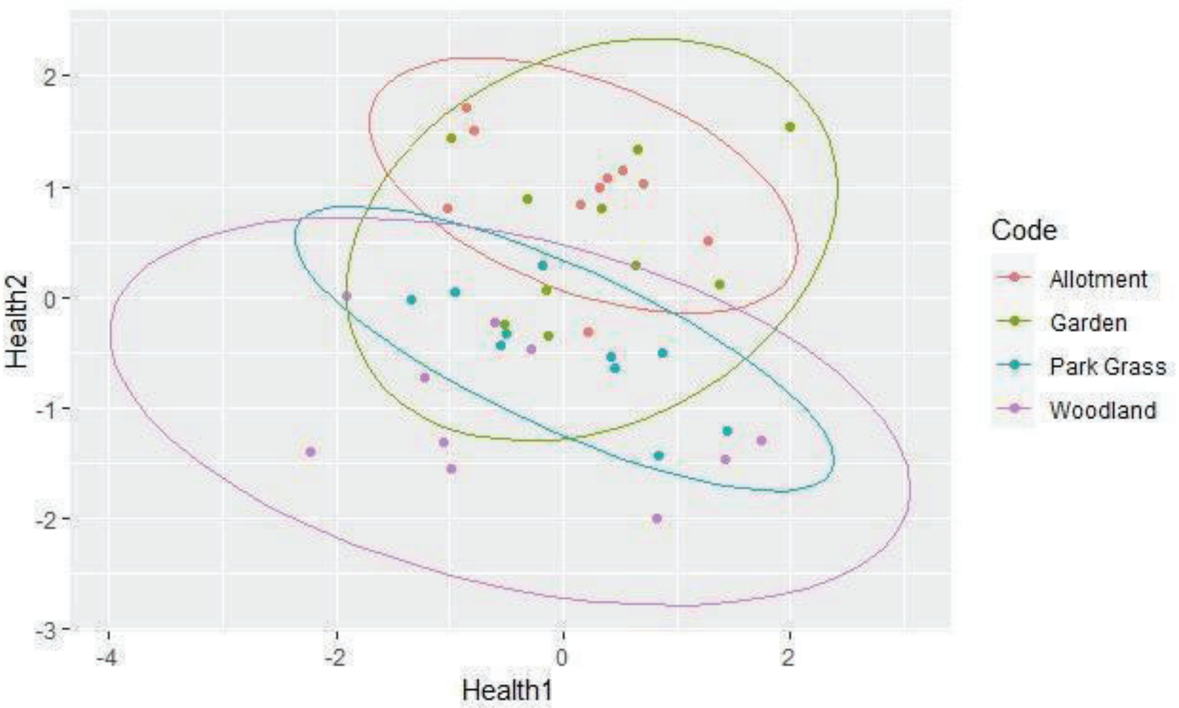


Figure 2.7. Principal Component Analysis (PCA)−biplot of four greenspace type categories based on the variance in 7 soil characteristics. The first two components explained 34.24% and 22.09% of the variances, respectively. The different colours denote the greenspace type category represented.

Invertebrate data

Earthworm data

A Poisson regression model was used to assess the effects of sampling timepoint (October vs. April) and land use type on total earthworm abundance (Figure 8.a). Earthworm counts were significantly higher in April compared to October. Specifically, samples collected in April had a 57% higher expected count of earthworms than those collected in October, as indicated by an incidence rate ratio (IRR) of 1.57 (95% confidence interval: 1.38 to 1.80, $p < 0.001$). Land use type also had a significant effect on earthworm abundance. Soils from garden areas showed a 59% higher expected earthworm count compared to allotment soils (IRR = 1.59, 95% CI: 1.26 to 2.04, $p < 0.001$). Park grasslands were associated with a 132% increase in expected counts (IRR = 2.32, 95% CI: 1.85 to 2.95, $p < 0.001$), while woodland soils had a 99% higher expected count than the reference land use (IRR = 1.99, 95% CI: 1.58 to 2.53, $p < 0.001$). The model intercept represents the estimated count in October for the reference land use category, corresponding to an IRR of 3.73 (95% CI: 2.76 to 4.98, $p < 0.001$). These results suggest that both sampling time and land use type significantly influence earthworm abundance..

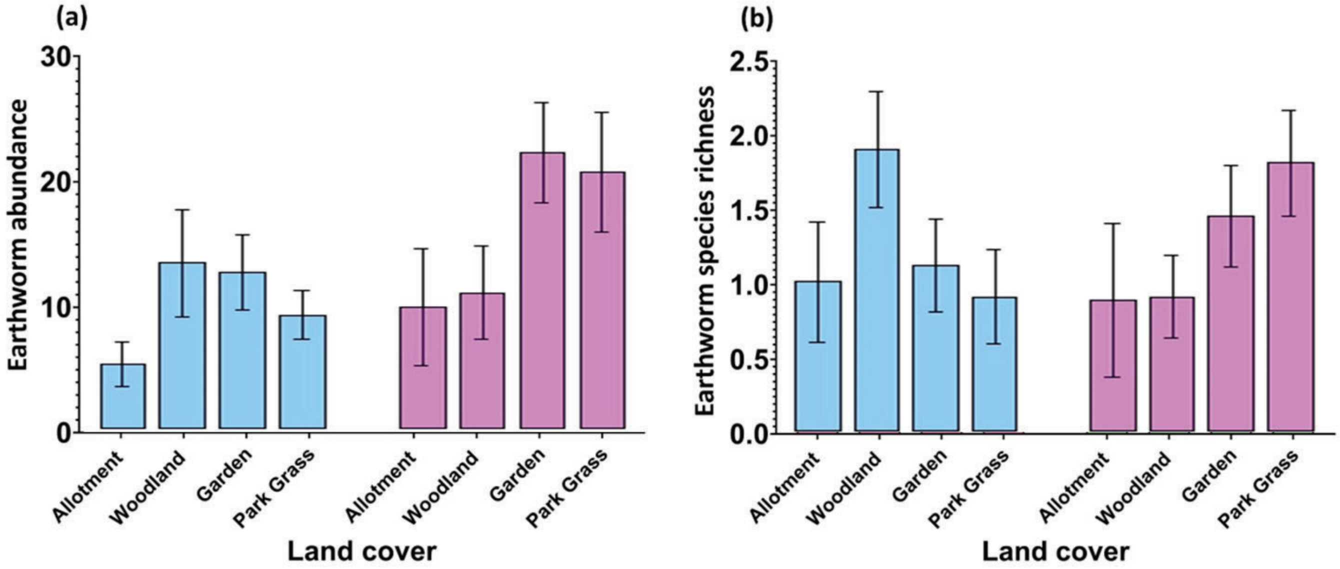


Figure 8. (a) Abundance and (b) species richness of earthworms sampled in April (blue) and October (pink) across allotment, woodland, garden and park grassland soils. Bars represent the mean total abundance of earthworms collected with error bars representing standard error

earthworm abundance did not significantly differ between April and October, with an estimated incidence rate ratio (IRR) of 1.04 (95% confidence interval: 0.69 to 1.59, $p = 0.84$), indicating virtually no change in expected counts. Garden soils were associated with a 46% higher expected count compared to allotment, but this was not statistically significant (IRR = 1.46, 95% CI: 0.74 to 3.06, $p = 0.29$). Similarly, park grasslands and woodlands had nonsignificant increases in expected earthworm counts relative to allotments, with IRRs of 1.36 (95% CI: 0.68 to 2.89, $p = 0.39$) and 1.44 (95% CI: 0.74 to 3.01, $p = 0.30$), respectively.

The model intercept, representing expected abundance in October within allotments, had an IRR of 0.88 (95% CI: 0.36 to 2.01, $p = 0.77$), suggesting no meaningful baseline deviation. Overall, the model indicates that neither sampling timepoint nor land use type significantly predicted earthworm abundance in this subset of the data. There were trends within the data in April, with woodland and park grassland demonstrating higher earthworm species richness than allotment and garden soils. There were also visual differences in earthworm abundance in October, with allotment showing lowest values and garden showing highest values, and in species richness, with garden showing highest values.

Soil invertebrate abundance

32,565 individuals were extracted from the soil samples and identified. Of these, 82.64% were mites (26,912 individuals), 15.25% (4,967) were Collembola, and the remaining 2.11% were predominantly beetle, worm and symphylan species.

Greenspace type significantly affected total invertebrate abundance across five of the six timepoints (Figure 2.9). In February, invertebrate abundance was significantly higher in garden soils (IRR = 1.53, 95% CI [1.39, 1.69], $p < 0.001$), while park grassland soils had significantly lower abundance (IRR = 0.58, 95% CI [0.51, 0.66], $p < 0.001$). In June, park grassland and woodland soils showed elevated abundances compared to the reference, with IRRs of 1.62 (95% CI [1.44, 1.83], $p < 0.001$) and 1.14 (95% CI [1.00, 1.29], $p = 0.041$), respectively. In July, park grassland soils had the highest invertebrate abundance, with an IRR of 6.25 (95% CI [5.57, 7.02], $p < 0.001$), while garden soils showed reduced abundance (IRR = 0.52, $p < 0.001$). August mirrored this pattern, with invertebrate counts highest in park grassland (IRR = 2.30, 95% CI [2.04, 2.58], $p < 0.001$), and lowest in gardens (IRR = 0.61, $p < 0.001$). Although October showed marginal effects (e.g., park grassland IRR = 2.03, $p < 0.001$), woodland and garden interactions were weaker. These results suggest strong seasonal shifts in invertebrate abundance, with gardens dominating in early timepoints (e.g., February), and park grassland sites supporting peak abundance in summer.

Greenspace type significantly affected mite abundance across all six timepoints (Figure 2.9). In February, mites were most abundant in garden soils, with significantly higher counts compared to the reference land use (IRR = 1.38, 95% CI [1.25, 1.53], $p < 0.001$). April showed a similar pattern, with consistently elevated abundance in garden soils (see Supplementary Table X). In July and August, mite abundance peaked in park grassland and woodland soils. Notably, in July, park grassland sites had a nearly sevenfold increase in mite abundance compared to the reference (IRR = 6.95, 95% CI [6.13, 7.88], $p < 0.001$), while woodland sites also showed significantly higher counts (IRR = 2.35, 95% CI [2.06, 2.68], $p < 0.001$). In August, the park grassland again supported the highest mite abundance (IRR = 2.42, 95% CI [2.13, 2.75], $p < 0.001$). Interaction effects indicate that the positive influence of garden soils diminished in later months (e.g., Garden × August: IRR = 0.57, $p < 0.001$), suggesting a temporal shift in habitat suitability for mites across greenspace types.

Greenspace type significantly affected total invertebrate abundance at three of the six timepoints: February, July, and October. In February, all greenspace types showed significantly higher invertebrate abundance compared to April, with garden soils exhibiting the strongest effect (IRR = 9.30, $p < 0.001$), followed by park grass (IRR = 6.05, $p < 0.001$) and woodland soils (IRR = 5.17, $p < 0.001$). In July, park grass soils supported the highest abundance, significantly greater than both garden (IRR = 0.43, $p < 0.001$) and woodland soils (IRR = 2.29, $p < 0.001$). In October, park grass again had significantly higher invertebrate abundance (IRR = 1.94, $p < 0.001$), while woodland soils had significantly lower abundance (IRR = 0.57, $p = 0.002$) compared to April. The remaining timepoints—August and June—did not show consistent or significant differences across greenspace types relative to the baseline.

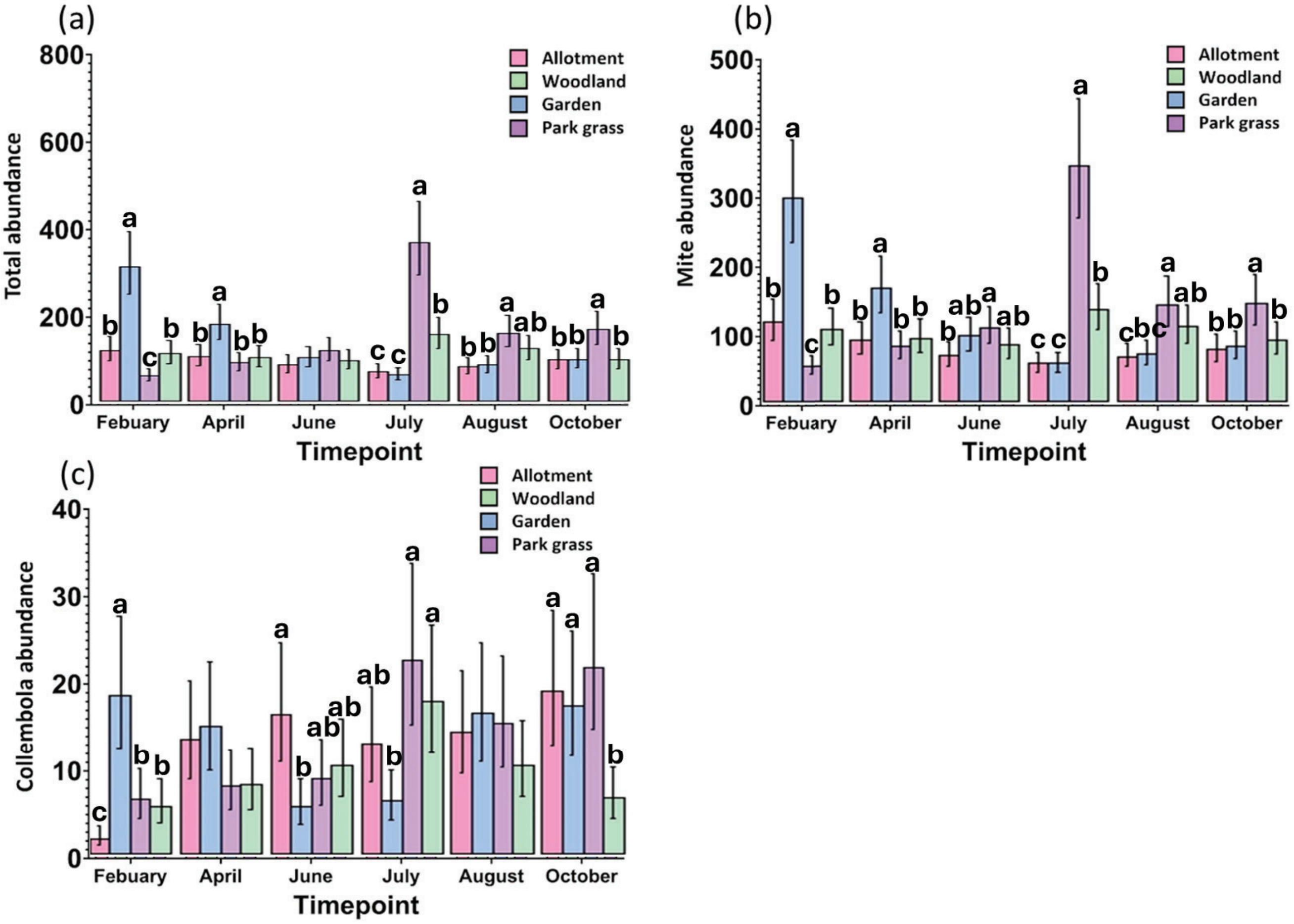


Figure 2.9. (a) Total abundance, (b) mite abundance and (c) Collembola abundance of soil invertebrates sampled across six sampled timepoints. Bars represent the mean species richness of soil invertebrates collected within allotment (pink), woodland (green), garden (blue) and park (purple)grassland soils, with error bars representing standard error.

Invertebrate diversity

Greenspace type had a significant effect on species richness for two timepoints, February and October. (Figure 2.10.a) In February, woodland and garden showed higher diversity, whereas in October, the lowest richness was found in woodland and garden. Greenspace type had a significant effect on Simpson’s diversity index of invertebrates for two timepoints, August and October (Figure 2.10.b). At both timepoints, allotment had a significantly higher index value than woodland.

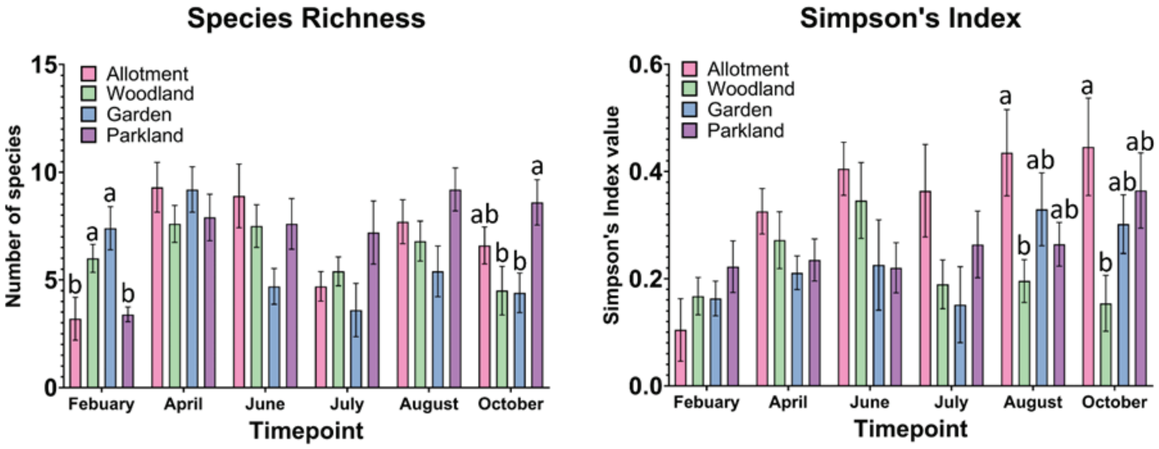


Figure 10. (a) Species richness and (b) Simpson’s Index of soil invertebrates sampled across six sampled timepoints. Bars represent the mean species richness of soil invertebrates collected within allotment (pink), woodland (green), garden (blue) and park (purple)grassland soils, with error bars representing standard error.

For five of the six timepoints, an Analysis of similarities (ANOSIM) was significant (February, April, June, July, August)(Figure 2.11). In February, the distance linking allotment to any other sample was lowest, indicating the least similarity between this invertebrate population and other greenspace types. In April, allotment and garden showed the overlap was lowest, indicating lower similarity between these sites and park grassland and woodland populations. In June, July and August, garden showed the lowest overlap with other greenspace uses, indicating that there was lower similarity between garden invertebrate populations and other greenspace invertebrate populations.

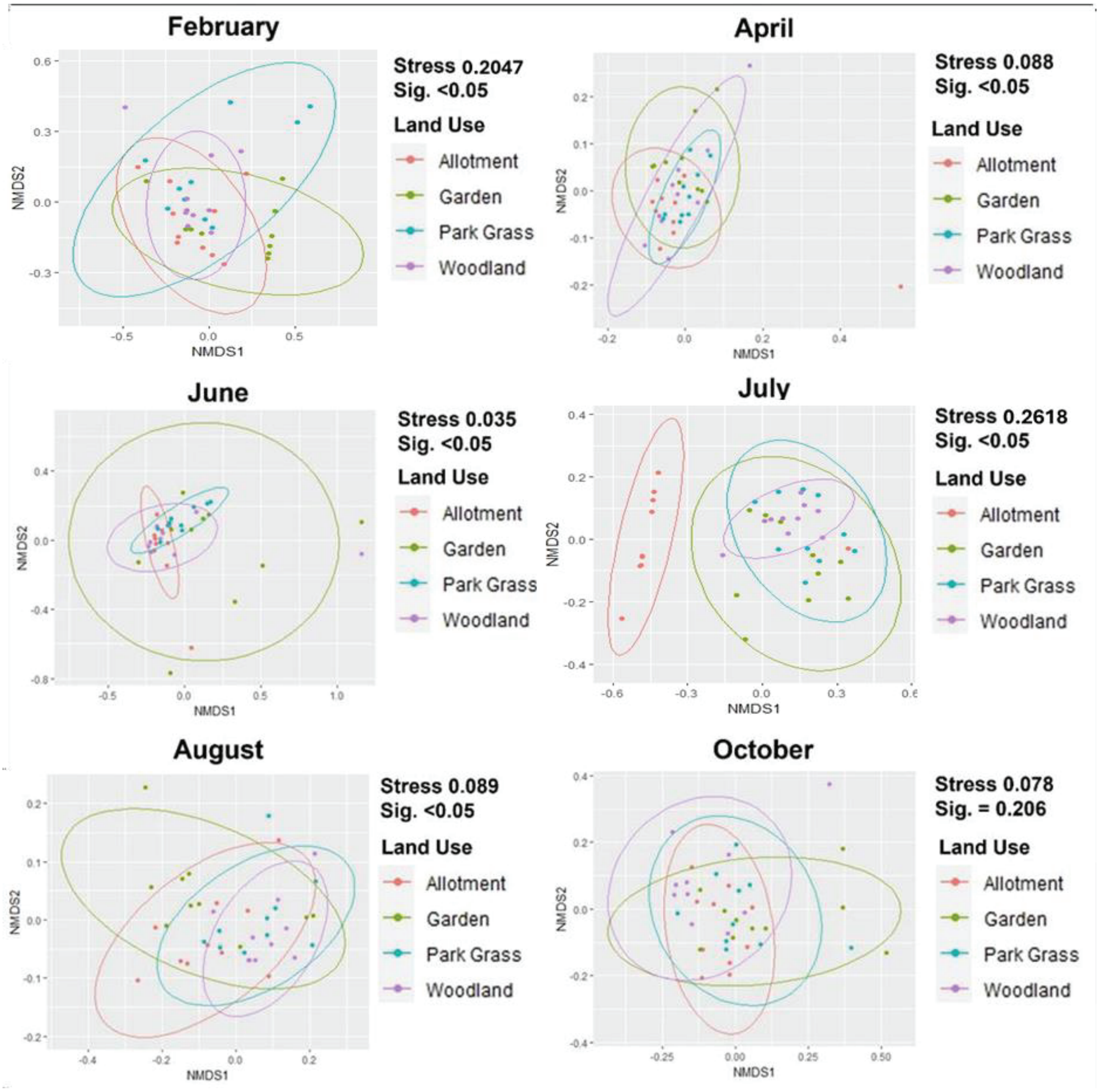


Figure 2.11. NMDS−biplot of invertebrate population for 6 timepoints. Samples represent four greenspace type categories based on the variance in6 population characteristics. The different colours denote the greenspace type category represented.

Greyspace surrounding sites

We compared the greyspace proportion surrounding of each greenspace type. Greenspace type was not significantly associated with greenspace proportions surrounding the site at 250m (F(3,36)=1.782, p=0.168), 500m (F(3,36)=0.297, p=0.827), 750m (F(3,36)=0.789, p=0.508) and the 1000m (F(3,36)=0.910, p=0.446). Greenspace type did have a significant association with greyspace proportion surrounding at 100m (F(3,36)=5.080, p<0.005), where garden sites showed a lower proportion of surrounding greenspace than other sampled greenspace types.

Discussion

Protection of urban greenspaces is now a key component of urban planning and development, but the long-term potential of greenspaces hinges on the protection of their soils. Here, our investigation into urban greenspace soils indicated significant differences in soil composition and soil invertebrate communities within urban greenspaces of different greenspace types. Woodland soils showed significantly lower concentrations of four metals (Ca, Cd, Pb, Zn) than other greenspace types, reflecting minimal anthropogenic disturbance. However, soil health indicators, such as pH and phosphorus levels did not significantly differ by greenspace type and were therefore not considered strong differentiators of soil health in this study. Garden soils had significantly higher invertebrate abundance in early spring (February), while park grass soils showed elevated abundance later in the year (July and October). These patterns suggest temporal shifts in habitat suitability across greenspace types. These results indicate that woodland soils appear to have better health than allotment and garden soils but that thisimproved soil condition does not consistently translate to higher invertebrate abundance or diversity across all time points. Future greenspace planning aimed at promoting healthy soils must also consider the specific needs of soil invertebrate communities, as improvements in soil conditions do not automatically translate to increased biodiversity. This requires integrated management strategies, such as reduced chemical inputs, increased habitat heterogeneity, and organic matter retention, that support both soil function and invertebrate diversity

Here, we provided evidence that although metal and metalloid elements were present in urban soils, they were within the documented range across UK urban soils. The UK government has published NBC and SGVs values for As, Cd, Cu, Ni and Pb (Table 2) (Ander et al., 2013;

Ferguson et al., 1993; Johnson et al., 2012). Values found were at or below levels recommended for human health for all elements except for Pb and As, which had a mean value above SGVs (Table 1). If values are below SGV's, growing edible plants is considered safe, although there is still potential for metal accumulation and risk to human health. However, as values of As and Pb were higher than this, there is potential risk to human health within these sites. Lead exposure can lead to cognitive development issues in children, while arsenic is a carcinogen, with long term exposure posing risks. Therefore, the soils across all greenspace types sampled across Sheffield within this study may pose a risk to human health. Continuous monitoring is recommended to ensure food safety from urban crops. In addition to this, although other heavy metal values were below SGV recommended levels, the focus on human health does not consider the implications on the wider ecosystem. Concentrations of metals within urban soils are consistently higher than in rural soils, where issues of pollutant effects on soil invertebrates have been recorded (Lui et al., 2023; Panico et al., 2022; Monchanin et al, 2021). There is risk that high concentrations of metals would be detrimental to invertebrate health, and thereby to wider soil health. These findings demonstrate that although urban contaminants were not at a dangerous level to human health within any greenspace type, there may still be implications on wider ecosystem health.

Higher levels of five metalloids were found within garden and allotment sites, demonstrating that land management within these greenspace types may be a driver of metal concentration. Here, the effect of greenspace type appeared to be metalloid dependent, with four (Ca, Cd, Pb, Zn) of the 12 investigated metals significantly higher in allotment and garden sites. These elements (Ca, Cd, Pb, Zn) are common within urban pollution sources (industrial effluents, building materials, vehicle emissions) and agricultural inputs (phosphate fertilizers, liming, pesticide residues). Allotment soils often see high pollutant levels, with informal waste disposal and historic land use leading to high contaminant levels (Papritz et al., 2009). In addition, regular inputs of artificial fertilisers and pesticides are seen in these spaces, which contain high levels of the metals and metalloids and are likely contributors to increased pollutant presence (Joimel et al., 2021; Voigt et al., 2015). Woodland greenspaces see lower levels of human intervention, with self-sustaining systems leading to lower levels of metals and metalloids within the soil. A similar difference in metal levels was seen in the weighted index, with garden soils showing higher metal levels than woodland soils. This indicates that despite having higher leaf surface areas that can trap pollutants, woodland soils still have lower heavy metal concentrations than more intensively managed garden and allotment soils due to anthropogenic inputs. Therefore, while woodland greenspaces benefit from their natural, self- sustaining ecosystems, the higher concentrations of metals and metalloids in more heavily managed greenspaces underscore the need for improved land management practices to mitigate pollution and enhance soil health in urban environments.

While fertilisers and pesticides are intended to enhance crop productivity, improper application or runoff can lead to soil contamination, affecting not only the health of crops but also the surrounding environment (Müller et al., 2020). Phosphorus and nitrogen concentrations did not vary significantly across greenspace types, suggesting relatively uniform nutrient conditions in these urban soils. However, allotments still showed higher levels of some metals and metalloids, indicating potential pollutant accumulation from historic or ongoing inputs. This contamination raises concerns for human health, particularly through the consumption of food crops grown in these soils. As urban greenspace conversion is considered, it is essential to weigh the ecological and public health risks associated with different land use histories and soil conditions (Edmondson et al., 2020b).

We did not demonstrate an effect of greenspace type on metals and metalloid concentrations for eight elements. The lack of pattern was seen in elements such as arsenic, copper, chromium, and iron. Though these metals and metalloids are commonly sourced from fertilisers and pesticides, they have a number of other sources in urban spaces (Tóth et al., 2016; Alahabad et al., 2017). Air quality is one source of metal pollutants, which can lead to higher rates of heavy metals and metalloids in soils due to the pollution capture and subsequent deposition of leaves (Masindi et al., 2018). The high levels of wastewater runoff within park soils and subsequent soil contamination may have raised metalloid levels within park grassland, thereby reducing the variation between greenspace type (Ma et al., 2016). Although there is variation in the sources of metals and metalloids and their subsequent levels within the soil, the variety in greenspace site urbanisation, surroundings, and management style may have obscured this effect within our research. The variation demonstrated here illustrates how greenspace type can, but does not always, relate to heavy metal levels within soils. Investigating these interactions further is important to fully understand the complex interactions which exist in urban greenspaces.

Soil health characteristics, such as pH, are also affected by greenspace type (Thomas, 1996). Disturbing management practices, including reduced tillage of soils, have been shown to drive acidification (Li et al., 2020). Land which undergoes regular tillage, such as allotments and gardens herbaceous borders, has been linked to increased soil pH levels (Li, 2020; Liu et al., 2021; Uwituze et al., 2022).Soil pH levels did not significantly differ by greenspace type, indicating that any soil amendments had limited or inconsistent influence on pH across the sampled sites, although there were visible differences in pH across greenspaces. s. The efforts of home-growers to counteract acidic soils could be the cause. Acidic soils can reduce nutrient transfer effectiveness and limit plant growth. To mitigate these effects, many allotment holders apply soil amendments such as lime, compost - often homemade green waste or manure-based composts - and alkaline fertilisers, aiming to raise soil pH and improve nutrient availability (Bosiacki et al., 2021; Blum et al., 2017). These amendments, which are not typically used in woodland or park grass management, likely contributed to the higher pH levels observed in allotment and garden soils in this study. While allotment beds are often covered with compost, their management frequently involves the removal of aboveground vegetation post-harvest, reducing the potential for in situ decomposition. In contrast, woodland and park grass soils receive continuous organic inputs through undisturbed litter decomposition (Dennis et al., 2020; Poeplau et al., 2017). Garden borders, often dominated by evergreen or ornamental species, generate less seasonal litter (Ignatieva, 2021). These patterns reflect how greenspace type and vegetation management influence soil conditions, suggesting that soils in woodlands and park grass areas might benefit from targeted interventions to enhance long-term vegetation health and sustainability.

Another indicator of soil health is the carbon stocks within the soil, which serve as a measure of the ecological functionality of various greenspaces. This study found an average of 7.47 kg m^2 TOC. The carbon stocks observed in this study fall within a similar range to those reported in past studies, underscoring the importance of urban greenspaces in carbon sequestration. The variations in TOC may be attributed to the diversity of greenspace types sampled, which included parks, gardens, and woodlands, each exhibiting distinct soil characteristics and carbon dynamics. Parks, which are typically dominated by managed grassland, often show lower carbon storage due to frequent mowing and reduced organic inputs. Gardens, by contrast, benefit from regular organic amendments such as compost and manure, boosting SOC levels. Woodlands tend to support higher below-ground biomass and leaf litter accumulation, promoting carbon input, though mixed-species stands may show greater variability in SOC than monocultures. Vegetation cover and land use can influence TOC stocks in urban soils. Previous research has demonstrated that residential gardens and open spaces had the highest carbon stock within them (Ziter and Turner, 2018). Residential garden soil has also been shown to have higher SOC concentration public greenspaces (Edmondson et al., 2014b). Edmondson et al. (2014c) also showed associations between SOC and tree species richness, with SOC levels reduced under mixed woodland. Overall tree cover has also been associated with higher C stock in urban spaces than grass (Livesley et al., 2016). It is important to note that this study only measured total soil organic carbon (TOC) and does not include above-ground carbon stored in vegetation, which also contributes substantially to total ecosystem carbon stock. Although variations in soil organic carbon levels were observed between allotment and garden soils, these differences were not statistically significant within this study, but the range of past research emphasises the importance of carbon in soils. This lack of statistical significance may be due to high within-group variability in management intensity, organic inputs, and vegetation cover across sites, which can obscure consistent patterns in SOC levels despite observable trends. It is also important to consider the sources and types of carbon pools within these soils, such as black carbon, which is produced from incomplete combustion of organic materials and often serves as an indicator of pollution (Lian et al., 2017). Understanding the dynamics of carbon stocks in urban greenspaces is essential for informing land management practices and promoting sustainable urban greenspaces. Soil carbon is a key component of ecosystem functioning, and effective management strategies that enhance carbon storage can therefore contribute to climate mitigation goals and overall soil health in urban environments.

Another key indicator of soil health is biodiversity present. In this study, earthworm abundance did not significantly differ between timepoints or greenspace types. This may reflect high natural variability or insufficient sample sizes to detect trend. . Our findings contrasted with some studies on earthworm population dynamics, with Ligrone et al (2024) demonstrating highest abundance in autumn. However, contrary to their reports, we saw no significant change in species diversity between time periods. One of the influencing factors could have been climatic variation during the sampling period. During the summer of 2022, the UK experienced a series of heatwaves, with the extreme weather creating abnormal conditions (Yule et al., 2023). As heat tolerance is species specific in earthworms, the reduced abundance in October 2022 could have been prompted by the temperature and drought (Singh et al., 2019; Davie et al., 2023). Despite potential implications of heatwave and drought, which may affect the generalizability of our results, this work opens new avenues for research into urban earthworms and their links to urban soil health by providing baseline data on species diversity and abundance in various urban environments. We highlight the potential role of earthworms in enhancing soil structure, nutrient cycling, and pollutant degradation in city soils, thereby encouraging further studies on their ecological functions and contributions to urban ecosystem services.

As extreme weather events become more frequent, understanding these dynamics will be increasingly relevant for urban communities and their resilience**.** Future research could involve monitoring these urban soil systems before, during, and after extreme weather events—tracking changes in earthworm populations, soil structure, moisture retention, and nutrient availability. This would allow assessment of the direct and indirect impacts on soil biota and ecosystem functioning, thereby informing adaptive urban land management strategies.

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Significant changes in invertebrate abundance were seen across the year, both temporally and between greenspace land uses . During spring timepoints, we found garden soils demonstrated the highest invertebrate abundance across all taxa. Vegetation as a driver of pollinator communities has been documented across multiple landscapes (Braschler et al., 2020; Kotze et al., 2022; Damptey et al., 2022). With high plant diversity and presence of flowering stems, gardens have demonstrated positive associations with invertebrate populations, particularly when compared to surrounding urban green spaces with lower vegetation complexity and areas where habitat heterogeneity is limited.(Palmersheim, et al., 2022). However, assessing drivers for replication in soil invertebrates are less clear. Our analysis suggests that the management of the site may influence temporal changes in abundance. As garden and allotment sites are intensively managed for consistent aesthetic value and productivity, the diversity of vegetation present is supportive of the invertebrate communities due to the diversity of organic matter (Philpott et al., 2020; Kwartnik-Pruc et al., 2023). This was demonstrated through levels of heavy metals found within this study, with highest levels of Ca, Cd, Pb and Zn found in allotment and garden soils when compared to woodland soils. The levels of heavy metals in soil affect soil invertebrates by reducing their abundance and impairing reproduction, ultimately leading to decreased biodiversity of soil invertebrates. This diversity could reduce vulnerability of populations over the hibernal season, allowing rapid population growth in the spring. The differences in invertebrate populations across greenspace types highlights the importance of these environments for supporting urban biodiversity.

The rise in invertebrate abundance in woodland and park grassland over the summer indicates that vegetation complexity is likely not the only key driver. Population change pattern was predominantly driven by abundance of soil mites. This could be due to increased disturbance in allotment and garden soils over these timepoints, reducing abundance relative to park grass and woodland soils (Crotty et al., 2021). Garden herbaceous borders and allotment sites are intensively managed and exposed to several anthropogenic actions over the summer months, including tilling, pesticides and fertilisers use (Tresch et al., 2018; Edmondson et al., 2014b). Woodland and park grassland sites are left relatively undisturbed throughout the year and see little below-ground disruption (Edmondson et al., 2014a). Allotment and garden soils, which are regularly disturbed, may leave invertebrate populations more vulnerable as result. For example, the summer temperatures could have compounded the effects of disturbance on population change – this investigation was partially undertaken during the 2022 heatwave, a summer of abnormally high temperatures in the UK, and thus the patterns seen may be emphasized by this extreme weather (Davie et al., 2023; Yule et al., 2023). The effect of extreme heat is accentuated by low vegetation cover, such as the exposed soils of garden flowerbeds and allotment beds. This can lead to increased mortality of soil invertebrates, resulting in lower population densities.

Replication of the investigation both within Sheffield and other urban spaces would allow potential influences to be confirmed, giving a comprehensive analysis into the factors affecting soil invertebrate abundance. Further experimental designs would include sampling across a range of urban species, including standardised soil and environmental measurements, and seasonal repetition to account for temporal variation. This would strengthen the robustness and reproducibility of the findings.

Significant differences in invertebrate species richness were observed in February and October, with woodland and garden soils supporting greater richness than park grasslands. These differences may reflect structural complexity and organic matter availability in these soils. Invertebrate species richness typically exhibits variation across the year, with cyclical changes in soil temperature, moisture and nutrient availability affecting some species more drastically (Wu et al., 2023). We observe these changes reflected through differences in community composition across garden soils across the sampling period, providing a likely contributor to observed temporal changes in species richness. Woodland showed greater similarity to park grassland and allotment soils, and community composition is likely not the key driver of temporal change. Leaf litter inputs in the latter half of the year could be related, with a high influx of organic matter providing a resource for decomposer species, permitting these groups to dominate, and reducing overall species diversity (Bedano et al., 2016; Griffiths et al., 2021). Patterns in diversity indices reflected this, with Simpson's index diversity lowest in woodlands, indicating that species evenness was low. These seasonal fluctuations in species richness we observed are crucial for maintaining soil health and ecosystem function, with different species contributing to various ecological processes throughout the year, and therefore understanding the drivers within urban greenspaces is key. While a baseline level of functional group persistence might be expected, our findings suggest that the relative abundance and activity of these groups vary significantly by season, potentially altering their functional contributions. Identifying which taxa are seasonally dominant or vulnerable can inform how ecosystem functions are maintained or disrupted over time in urban soils. We demonstrated the time periods with highest invertebrate abundance for different greenspace types, with woodland and garden soils showing highest species richness in the spring, but lowest in the summer, demonstrating the importance of timing when considering the invertebrate communities within these spaces. Ensuring that management strategies and further research is undertaken are undertaken across all timepoints will allow accurate insight into the true complexity of urban greenspace types.

Our findings suggest that while greenspace type can influence soil metal concentrations and invertebrate abundance at specific times of year, these effects are inconsistent and often species- or season-specific. Urban soil health assessments should therefore incorporate both chemical and biological indicators across seasons to fully understand ecosystem function.

Conclusion

This investigation shows that measures of soil health and the provision of ecosystem services differ significantly between various types of urban greenspaces. Our findings indicate that garden and allotment soils have higher heavy metal concentrations compared to woodland soils, which exhibited the lowest levels of contamination. Additionally, greenspace type significantly affected several characteristics of invertebrate communities, with higher invertebrate abundance in gardens in spring, and in park grass in summer. This suggests that the management of greenspace types can lead to significant differences in soil characteristics and biological communities. The results highlight the influence of greenspace type on soil and invertebrate characteristics, emphasizing the potential of these spaces to sustain biodiversity and provide essential ecosystem services. By understanding how changes in greenspace types can be supported, we can select for differences in ecosystem services when incorporating urban greenspace into urban planning. These insights can inform urban design and promote the development of sustainable and resilient urban greenspaces.

Chapter 3: The influence of greenspace type on plant growth and health characteristics

Abstract

Urban greenspaces are increasingly recognised for their ecosystem services, including nutrient cycling, habitat provision, and primary production, but how this varies across different forms of greenspace under different management is poorly understood. Understanding how greenspace type influences ecosystem services will allow these effects to be included into urban planning regimes, creating greenspace types based upon the service needs of the surrounding areas.

However, the effectiveness of this planning is limited by a lack of research into how soils in different greenspace types differ in their quality, and the implications of this variation in the health and productivity of the range of plants we might try to grow in these different greenspaces. Here, we conducted a bioassay to compare plant growth and health of five plant species (alder, bean, cabbage, radish, and wild carrot) within four key greenspace categories (allotment, garden, park grass and woodland greenspaces). Plant biomass was demonstrated to be significantly lower in parkland soils across all species, with the highest plant productivity seen in woodland and garden soil. Chlorophyll fluorescence and mycorrhizal colonisation was minimally impacted by land use. More research into the causes of this difference is needed to fully understand the patterns of plant growth in urban spaces.

Introduction

Urbanisation in the United Kingdom has continued to increase throughout the past century, driven by population growth, economic development, and legislation and patterns of land use across the nation. More than 84% of the UK population reside in urban areas in 2024, making it one of the most densely populated countries in Europe (Merkle et al., 2022; Withington, 2017). The scale of this change has placed pressure on urban land, increasing the demand for housing, infrastructure, and service development (Merkle et al., 2022). It has also placed significant stress on the environmental sustainability of urban land, resulting in conflict between environmental sustainability and urban development needs (Burchardt et al., 2020). By addressing these issues with a multi-disciplinary approach to balance the demands of the entire community into the future.

Urban greenspaces are areas of vegetated land within urban areas, typically designed for recreation and physical activity, aesthetic value, and ecological benefits (Suligowski et al., 2021). The term encompasses a wide range of types of different scales, including parks, residential and community gardens, woodlands, allotments, playing fields and undeveloped land (Brindley et al., 2019; Suligowski et al., 2021). These spaces provide significant socio-cultural benefits, with noted benefits on mental health, economic benefits, and recreational opportunities (Hunter et al., 2019; Mears et al., 2019;). In addition, they provide numerous environmental benefits to the surrounding urban area, such as air quality improvement, flood risk mitigation and urban cooling (Green et al., 2021; Kasim et al., 2019; Mansournia et al., 2019; Semeraro et al., 2021). As urban areas fragment natural habitats, these greenspaces provide habitat for urban species, both flora and fauna, mitigating habitat loss and giving rise to biodiversity within cities

(Castelli et al., 2021; MacGregor-Fors et al., 2016). As urbanisation continues within the UK understanding greenspaces and the services allows planning to be better informed.

The current priority for development and maintenance of urban greenspace is increasingly focused on sustainability, climate adaptation, and biodiversity. As cities attempt to mitigate increasingly evident challenges, such as climate change, urban greenspaces are being redefined as critical infrastructure for environmental sustainability (Chatzimentor, et al., 2020). Recent trends emphasise "green infrastructure," where greenspaces are integrated into the broader urban ecosystem (Mell, 2018; Jerome et al., 2019). These spaces may be designed specifically to provide ecosystem services such as water purification and flood management, such as Sustainable Drainage Systems, or SuDS (Melville-Shreeve et al., 2018). Another current focus is the concept of "rewilding", which aims to promote biodiversity by including less intensively managed spaces with low vegetation diversity and assist the reintroduction off native vegetation species (Pettorelli et al., 2019). Finally, urban regeneration projects often incorporate "pocket parks" and green corridors, smaller but strategically placed greenspaces that are woven into dense cityscapes, connecting people to nature in highly urbanised environments (Dong et al., 2023; Zhang et al., 2021). These innovative approaches, such as pocket parks and green corridors, can be effective, but their ecological impact is often limited by size, connectivity, and management. Isolation and small size availability may prevent species from utilising these spaces. Their success therefore depends on thoughtful design and the broader urban landscape context in which they are embedded. This shift in recognising the needs and complexity of urban greenspace environments reflects a broader movement towards creating greener, more liveable cities while addressing the pressing challenges of urban growth and climate adaptation.

New greenspace creation or modification involves planting of new species, which may or may not thrive due to the historic use of the site. Large expanses of amenity grass, often maintained as park grass or recreational spaces, provide limited ecosystem services and minimal biodiversity benefits compared to more complex green spaces (Muir et al., 2021; Norton et al., 2019). These areas are dominated by monocultures of grass species, which are frequently mowed, irrigated, and treated with fertilizers and pesticides. As a result, they offer few habitat opportunities for pollinators, birds, or other wildlife (Muit et al., 2021). . The simplified structure of urban grasslands also limits their ability to regulate local microclimates, reduce air pollution, or support water infiltration and flood mitigation (Mexia et al., 2018; Muit et al., 2021).

However, converting these grass-dominated areas into ecologically richer habitats requires careful planning, as soil conditions, compaction, and urban stressors may initially limit plant growth. Additionally, athough these spaces offer relatively low ecological ecosystem service potential, particularly in terms of biodiversity and habitat provision, they still provide valuable cultural and recreational services for local communities. As such, while they are often prioritised for ecological conversion, any changes must consider their social importance and the role they play in urban wellbeing. Thus, while land conversion offers potential for maximising ecosystem services, understanding the potential of different plant species within biodiverse spaces is necessary to maximize environmental benefits.

For these greenspaces to support the growth of diverse plant species in different greenspaces it is crucial that their soils can support healthy plant growth. Through processes such as evapotranspiration, healthy vegetation is crucial for regulating urban microclimates, with plants helping to mitigate the urban heat island effect by cooling surrounding air and providing shade (Francini et al., 2021; Skelhorn et al., 2018; Taher et al., 2019). Urban plants also play a key role in improving air quality by absorbing pollutants such as carbon dioxide, nitrogen oxides, and particulate matter, contributing to cleaner air and reducing public health risks (Barwise et al., 2020; Han et al., 2022). Plants can also enhance soil stability, reducing erosion and managing stormwater by promoting infiltration and reducing surface runoff, which helps prevent urban flooding (Ferrini et al., 2020; Francini et al., 2021). The vegetation in these spaces support biodiversity by providing habitat and food for various species, including pollinators, birds, and small mammals, contributing to urban ecosystems' resilience and stability (Filazzola et al., 2019; Marselle et al., 2021). However, not all vegetation contributes equally to these services. For example, native flowering species are highly beneficial for pollinators, while larger canopy trees like *Tilia cordata* are particularly effective at intercepting airborne pollutants. Conversely, non-native or invasive species can reduce habitat quality, alter soil chemistry, or outcompete native flora, ultimately reducing the overall ecological value of a greenspace. The composition, structure, and functional traits of plant communities therefore strongly influence the type and magnitude of ecosystem services delivered. Poor plant health, additionally, can lead to degraded greenspaces that fail to deliver these essential ecosystem services, negatively impacting both environmental quality and human well-being. Data and research on the factors driving the productivity and health of different plant species in different greenspaces will inform urban planning and greenspace management by identifying which greenspace types and vegetation structures most effectively support key ecosystem services. This evidence can be used by local authorities and landscape designers to prioritise species and management practices that optimise both ecological and social benefits.

The impact of management practices on green spaces significantly influences soil properties and ecosystem functioning. Differences in greenspace type and subsequent land management can influence plant productivity and health by affecting soil characteristics, including soil composition, structure, and ecological function. Urbanisation and the development of infrastructure often lead to soil compaction, sealing, and contamination, which degrade soil quality by reducing porosity and water infiltration (Corrêa et al., 2021; Li et al., 2018). Compacted soils are less able to support future vegetation growth, with limits on root penetration and water drainage, which increases surface runoff and increases the risk of flooding (Kumar et al., 2021). Additionally, differences in greenspace type can lead to accumulation of pollutants, such as heavy metals and metalloids, from pollutants, fertilisers, and pesticides, further compromising soil health (Vigevani et al., 2021; Vieira et al., 2018). These contaminants can disrupt nutrient cycling and reduce soil biodiversity, affecting the ability of soils to support plant life and provide ecosystem services like carbon sequestration (Vasenev, et al., 2018). Conversely, land management that prioritises the sustainability of greenspaces, such as the use of permeable surfaces, organic growing, and rewilding, can improve soil structure, enhance nutrient availability, and promote the recovery of healthy soil ecosystems (Andriuzzi et al., 2023; Tessler et al., 2018; Pettorelli et al., 2022). Land use plays a critical role in determining the ecological functions and sustainability of soils within urban greenspaces.

Soils also play a crucial role in determining plant health by influencing the availability of nutrients, water retention, root development, and overall ecosystem support. Key soil properties, such as texture, structure, pH, organic matter content, and microbial activity, directly impact plant growth (O'Riordan et al,2021). Soil pH affects nutrient solubility and availability; in acidic soils, nutrients may be less accessible, while toxic elements may inhibit root growth (Neina et al., 2019; Tresch et al., 2018). Alkaline soils can lead to deficiencies in micronutrients, causing reduced photosynthesis in leaves (Neina et al., 2019). Organic matter is vital for maintaining soil fertility, as it provides nutrients and supports the communities essential for nutrient cycling (Cotrufo et al., 2022; Lal et al., 2020; Oldfield et al., 2019). Soils with high organic content support beneficial microorganisms that form symbiotic relationships with plants, such as mycorrhizal fungi, which enhance nutrient uptake (Cotrufo et al., 2022; Frey et al., 2019). Thus, healthy soil ecosystems are essential for supporting robust plant growth and resilience against environmental stressors. Degraded soils, by contrast, can limit plant health, productivity, and the ability to withstand pathogens and climate fluctuations.

Urban areas can experience high levels of pollutant input into soil because of their proximity to anthropogenic activity. The metals and pollutants in soil can have detrimental effects on plant growth by disrupting physiological processes, reducing nutrient availability, and inducing toxic responses. Heavy metals and metalloids, such as lead (Pb), cadmium (Cd), and arsenic (As) can accumulate in soil due to industrial activities and runoff from urban wastewater (O’Riordan et al., 2021; Li et al., 2018). These metals can be toxic even at low concentrations, affecting root development, photosynthesis, and water uptake (Ahmad et al., 2021; Zwolak et al., 2018). High levels of Cd can interfere with plant uptake of essential nutrients, leading to nutrient imbalances and stunted growth within the plant. Additionally, metals and metalloids can interfere with hormone regulation within plants, further inhibiting key processes like cell division and elongation (Morkunas et al., 2018; Rashid et al., 2023). Soil inputs, such as hydrocarbons, pesticides, and artificial fertilisers, can also impair plant health by altering soil structure and microbial communities, thereby affecting nutrient cycling (Hartmann et al., 2023). Consequently, soil contamination by metals and pollutants poses a significant threat to plant health, agricultural productivity, and ecosystem stability.

An additional measure of plant health and success is the colonisation of roots by mycorrhizal fungi. Colonisation is an important indicator of soil health. Mycorrhizal fungi are essential for plant health and ecosystem functioning due to their symbiotic relationship with plant roots (Chen et al., 2018; Frey et al., 2019). These fungi extend the root system through a network of hyphae, significantly increasing the surface area available for nutrient and water absorption, particularly in nutrient-poor soils (Bennett et al., 2022; Chen et al., 2018). In addition to nutrient transfer, AMF contributes to soil health by improving soil structure through the production of glomalin, a glycoprotein that binds soil particles and enhances water retention. This partnership is key for the uptake of phosphorus, nitrogen, and other essential minerals that plants would otherwise struggle to access. In return, the fungi receive carbohydrates produced by the plant through photosynthesis (Bennett et al., 2022). Beyond nutrient acquisition, mycorrhizal fungi enhance plant resilience to environmental stresses, such as drought, soil pathogens, and heavy metal toxicity, contributing to overall plant health and productivity (Bennett et al., 2022; Wang et al., 2020). Moreover, these fungi play a critical role in soil structure by promoting aggregation, which improves water retention, aeration, and nutrient cycling (Posta et al., 2020). Their influence extends across ecosystems, supporting biodiversity and enhancing carbon sequestration, making them indispensable to both natural and managed landscapes. However, landscape scale variation in AMF colonisation remains poorly understood in urban ecosystems. Multiple soil characteristics, including compaction and pH, can influence the rate of colonisation within soil. Additionally, areas with lower AMF spore density in the soil will see lower colonisation rates. Understanding how greenspace type affects mycorrhizal colonisation allows insight into the productivity and health of plant species planted in different urban greenspaces.

To address the knowledge gap in our understanding of how greenspace type affects growth of plants we determined three research questions:

- How does greenspace type of soil influence plant growth in different plant species?
- How do soil physicochemical properties affect plant growth and health?
- How does greenspace type affect mycorrhizal colonisation rates?

We hypothesise that plant growth (biomass allocation to root, shoot, and edible portions) and health will be highest in woodland and garden soils and lowest in park grass soils, reflecting differences in soil quality and management intensity across greenspace types. We further predict that variation in soil physicochemical properties, such as metal concentrations, pH, carbon content, phosphorus, and bulk density, will significantly influence plant performance, with improved growth and health associated with lower contamination and higher organic matter content. Finally, we hypothesise that mycorrhizal colonisation rates will be greatest in woodland soils and will positively correlate with plant growth and health in mycorrhizal-dependent species (alder, bean, wildflower).

To assist in answering these questions, we undertook a bioassay experiment to test the effect of greenspace type on plant growth and health. We collected soils from four key greenspace types across the city of Sheffield (allotment, woodland, garden and parkland). and grew five plant species typical of each of these greenspace types (allotment, garden, park grass and woodland) within these soils, taking measurements of the biomass, mycorrhizal colonisation, and plant health characteristics of each plant. We hypothesized that urban allotment, garden, and woodland soils, typically receive regular organic inputs, see higher soil nutrient availability, and therefore greater plant growth and health within these soils. Conversely, park grass soils, which see high soil compaction and limited biodiversity, would see reduced plant growth and health within their soils.

Materials and methods

Study system

This study was carried out using soil samples from 40 greenspace sites collected in May 2022 in Sheffield, England (53.3811° N, 1.4701° W). These sites were the same sites used for analysis of greenspace type on soil health and invertebrate populations (Chapter 2). The experimental design included four greenspace use types: allotments (a parcel of land divided into smaller plots, which are rented by an individual for horticultural production of fruits and vegetables); gardens (a parcel of land adjoining a residential property in which plants are grown, typically for aesthetic and recreational value); park grass (a park refers to an area of publicly accessible land, often multifunctional and with multiple land covers. Within this, park grass refers to a section of amenity grass); woodland (a parcel of covered by trees and shrubs, typically with over 40% canopy cover).

Within the Sheffield unitary authority boundary, all sites of park grassland and allotment were identified from the data of OS Greenspaces layer in Mastermap (using QGIS version 3.28.3). Woodland areas were identified using the National Tree Map, with areas of tree canopy merged to give insight into areas with continuous urban canopy (Bluesky, 2011). Garden sites were selected using a call for volunteers through the University of Sheffield. Using histograms, box plots and bar charts which were generated in SPSS (version 27), the distribution of area for each land type was identified. Allotment sites demonstrated a smaller range in site area than woodland and park grass sites. An area limit of <30,000m² was therefore chosen as these included the majority (80%) of all allotment sites. A lower boundary of 5,000m² was selected, as an allotment plot of this size would have approximately 20 plots (with average plot size of 250m², Edmondson 2020b), which was determined as the minimum to allow adequate recruitment of plot holders. Within allotment sites, we checked that at least 80% of sites were undergoing active crop growing to make sure the study site properly exemplified the characteristics of allotments.

We selected 10 sites for each type of greenspace, with multiple (between 2-4) plots sampled within each allotment to minimise the influence of individual site management. In total, 52 sites were surveyed across 40 different greenspaces (Figure 2.1). To capture variation along the urban gradient, we included locations subject to urban pressures - such as habitat fragmentation, human presence, and nearby land use - acknowledging the inherently ambiguous nature of the term "urban." The centre of greyspace density was identified in Sheffield City Centre using the 'Calculate Density' tool in QGIS, and five concentric 2 km buffers were generated from this point using a 'Multi-Ring Buffer' approach.

Sites were selected along an urbanisation gradient, with five buffer zones of radius 2km radiating from the city centre. We selected two sites of each greenspace classification per buffer zone (Figure 2.1), giving a total of 40 sites in total. Within these sites, we generated a randomly placed sampling point using the 'random points' tool in QGIS and sampled as close to possible at this point.

Following these criteria, we selected each site using the 'random selection' tool in QGIS. If a site was closer than 2km from another already selected site, it was rejected, and the process was rerun.

This was to ensure sites were not concentrated across one side of the city. For garden sites, the constraint of only having access to university employee gardens meant that housing was biased towards the west of the city centre (where the university is located), but best efforts were made to ensure an even spread of sites within these constraints.

15 l of soil was sampled from each site. At woodland sites, the sampling area was first cleared of leaf litter or debris, while at park grass sites, a section of turf approximately 2cm depth was first removed from the soil surface (this was a condition of sampling permission in park sites). Within allotment plots, we sampled soils from a section of the site actively (within 6 months) used for growing but avoided areas with freshly planted seeds. Sheffield City council mandates that 75% of allotment land is used for cultivating fruit and vegetables, and so this was taken as the most common. Within garden sites, soils were taken from the herbaceous borders of the site. Within woodland soils, we avoided sampling at the base of trees to reduce damage and removed any tree debris from the sample. Within park grass sites, we removed the turf layer of the ground, took soil samples from beneath, replaced the hollow area with compost and replaced the turf layer. A 50x50cm quadrat was placed within the site. From this area, soil was taken to a depth of approximately 10cm.

In the lab, we sifted all samples through a 1cm riddle before taking a 30g sample, which was dried in a 80°C oven for 48 hours and weighed again to calculate the fresh to dry weight ratio of each soil. Differences in drainage and soil compaction across sites needed to be accounted for to ensure that each plant had an equivalent weight of soil. Fresh soil (equivalent of 280g of the site dry soil) was added to 10 cm round pots.

Experimental design

To represent species found in different greenspaces, five study species were chosen for this bioassay experiment. To represent allotment plots and the diversity of vegetation within them, three species of crop plant were selected - French beans (*Phaseolus vulgaris*), Cabbage (*Brassica oleracea*) and Radish *(Raphanus sativusres*) - presenting food crops which are common amongst UK allotment holders, and which were present at 94%, 81% and 43% of sampled allotment plots. In addition to this, one wildflower species was selected to represent a UK species common in wildflower meadows (*Daucus carota*). One tree species was used for insight into tree growth, alder (*Alnus glutinosa*) chosen due to its fast growth and ability to be grown from seed. Three of the plant species selected were mycorrhizal host plants (alder, bean, wildflower) to allow patterns of colonisation, a key factor in soil-plant nutrient transfer, to be examined.

Seeds were germinated in petri dishes within a laboratory environment on filter paper, placed in indirect sunlight and watered daily, before transferring to soil after seedlings had sprouted (14 days). Each greenspace type had ten examples from across the city, with between 2-4 allotment plots sampled from each allotment site, giving a total of 52 soil samples. Four replicates of each of the five plant species were grown in each of the soil samples (n = 4). This gave a total of 208 individuals of each species, and 1040 plants in total. Plants were grown outdoors without shelter, watered daily and with position rotated on a weekly basis to ensure equal access to sunlight/shade/shelter. Plants were grown for 16 weeks (Alder) and 10 weeks (all other crops). The growing period was from May-September 2022, and so encompassed the British summertime period including the 2022 United Kingdom heatwaves, with an average temperature of 15.7°C.

Plants were grouped into crops for this growth period to control for specific needs of different plant species.

Measures of plant health

To assess the effect of greenspace type on the different plant species we took multiple measures of plant health, including growth, flower production, and resistance to environmental stress.
Plant growth rate was measured and the investment into each section of the plant physiology: root, shoot, and edible portion (where relevant). Measurements of plant health were taken using a MultispeQ V 2.0 (Kuhlgert et al., 2016), using protocol Photosynthesis RIDES 2.0 Measurements using the MultispeQ were taken at the halfway point of growth (6 weeks after transfer into soil, 8 weeks for alder) and the day before harvesting for four of the plant species grown (alder, beans, cabbage, and radish). The shape of wildflower leaves meant that measurements could not be taken using the MultispeQ. This program took multiple measures to represent plant health and the plant conditions such as leaf temperature, angle, and relative humidity. From this data, two key indicators of plant health were selected. We first selected SPAD (Soil Plant Analysis Development), which is a measurement of the chlorophyll content of leaves. Chlorophyll concentration is often an indicator of leaf health, with lower SPAD values indicating potential nitrogen deficiency, lower photosynthetic capacity, and stress. The second measure taken was F_v/F_m (Fluorescent Variable/Fluorescence Maximum), which is a measure of the potential efficiency of the light-dependent reactions of photosynthesis. Lower values indicate stress responses or damage, such as nutrient deficiency, water stress, or temperature extremes.

Plant root and shoot material was harvested and any remaining soil was washed from the plant. The plant was then divided into subsections (root, shoot, and the edible portion for the three food crops) and we recorded the fresh biomass of each of these sections. An approximately 0.5 g subsample of the root section was taken and preserved in ethanol for mycorrhizal colonisation analysis. Samples were then oven dried at 70°C until reaching a constant weight (approximately 48 hours) and weighed The weight of the removed subsection was calculated and added to the total biomass.

Mycorrhizal colonisation of plant roots

Mycorrhizal colonisation of plant roots was examined in the three mycorrhizal plant species (alder, bean, and wildflower). Root samples (0.5g) were placed in a histology cassette. Samples were placed in 10% KOH, before heating in an 80°C oven for 1 hour. Samples were then removed from heat and washed in tap water three times. They were placed in ink and vinegar solution (5% Pelikan Brilliant Black, 5% acetic acid, 90% dH2O) for 30 minutes, then destained in 1% acetic acid for one hour (Vierheilig et al., 1998). Twenty cm long subsections of root were removed, mounted on slides alongside a small amount of PVLG- polyvinyl alcohol, lactic acid and glycerol.

Mycorrhizal colonisation was assessed using a compound microscope with a hairline graticule. The field of view was shifted to approximately 50 random locations across the root sample, where the intersections with arbuscules, vesicles, and hyphal colonisation are recorded at each point. This data was used to determine the percentage of the root colonised by each mycorrhizal structure. (Brundrett, 1994).

Measures of soil health

As soil samples were taken from the same greenspace sites investigated within Chapter 2, this data was available as an explanatory variable for this investigation. Measures of soil metals and metalloids, total carbon, organic carbon, pH, P, and bulk density were included in this analysis.

Data analysis

All statistical analyses were conducted using R version 4.12 (R Core Team, 2022). Data were first examined for normality and homogeneity of variance using Shapiro-Wilk and Levene's tests. Where necessary, data were transformed (log transformations) to meet the assumptions of parametric tests.

To investigate the effects of land use type on the biomass of each plant species and tissue type, linear mixed-effects models (LMMs) were fitted using the lme4 package (Bates et al., 2015). Land use type was included as a fixed factor, and site or block was included as a random effect to account for spatial replication and potential site-specific variability.

Estimated marginal means (EMMs) and their standard errors were computed from the fitted models using the emmeans package (Lenth, 202X). Pairwise comparisons between land use categories were performed with Tukey's Honestly Significant Difference (HSD) adjustment for multiple comparisons to control the family-wise error rate. Compact letter displays (CLDs) were generated to visually represent statistically significant differences among group means. Significance was determined at $\alpha = 0.05$.

Graphical visualizations, including bar charts of estimated means with standard error bars and significance letters, were produced using ggplot2 (Wickham, 2016).

Spatial autocorrelation of soil and invertebrate variables was examined using Moran’s I, based on the geographic coordinates of all sampling sites. Analyses were conducted in R with the spdep package. Moran’s I values were non-significant ($p > 0.05$), indicating no evidence of spatial clustering in model residuals and confirming that spatial dependence did not influence the results.

Results

Plant biomass

Greenspace type significantly affected total biomass for four species (Figure 3.2). Biomass was highest in woodland soils for wildflower F(3,36) = 5.089, p=0.005), in woodland and garden for cabbage (F(3,36) = 4.890, p=0.006), and in allotment and garden for bean (F(3,36 = 14.036, p<0.001). Biomass was lowest in allotment, garden, and woodland for alder (F(3,36) = 4.165, p=0.012). Biomass was lowest in park grass soils for all four species with significant differences but did not show significant differences for radish (F(3,36) = 0.679, p = 0.570). The largest difference for alder was between park grass and woodland (272% increase), for bean was between park grass and garden (237% increase), for cabbage was between park grass and woodland (135% increase) and for wildflower was between park grass and woodland (229% increase).A two-way ANOVA revealed significant main effects of plant species (F(4,180) = 9.47, p < 0.001) and land use (F(3,180) = 4.16, p = 0.007), as well as a significant plant species × land use interaction (F(12,180) = 3.90, p < 0.001). Post hoc Tukey-adjusted comparisons demonstrated that the effect of land use varied across species. For example, Bean exhibited significantly higher means in gardens compared to park grass (p < 0.001), and wildflower values were significantly higher in woodland than allotments (p = 0.0035). Conversely, cabbage and radish showed no significant differences among land use types. When comparing species within each land use, bean consistently showed higher values than cabbage and radish in allotments and gardens (p < 0.001), whereas in woodland, wildflower values were significantly greater than cabbage and radish (p ≤ 0.003). These results indicate that both species identity and land use independently and interactively influence the response variable, with beans and wildflowers displaying particularly strong habitat-dependent patterns.

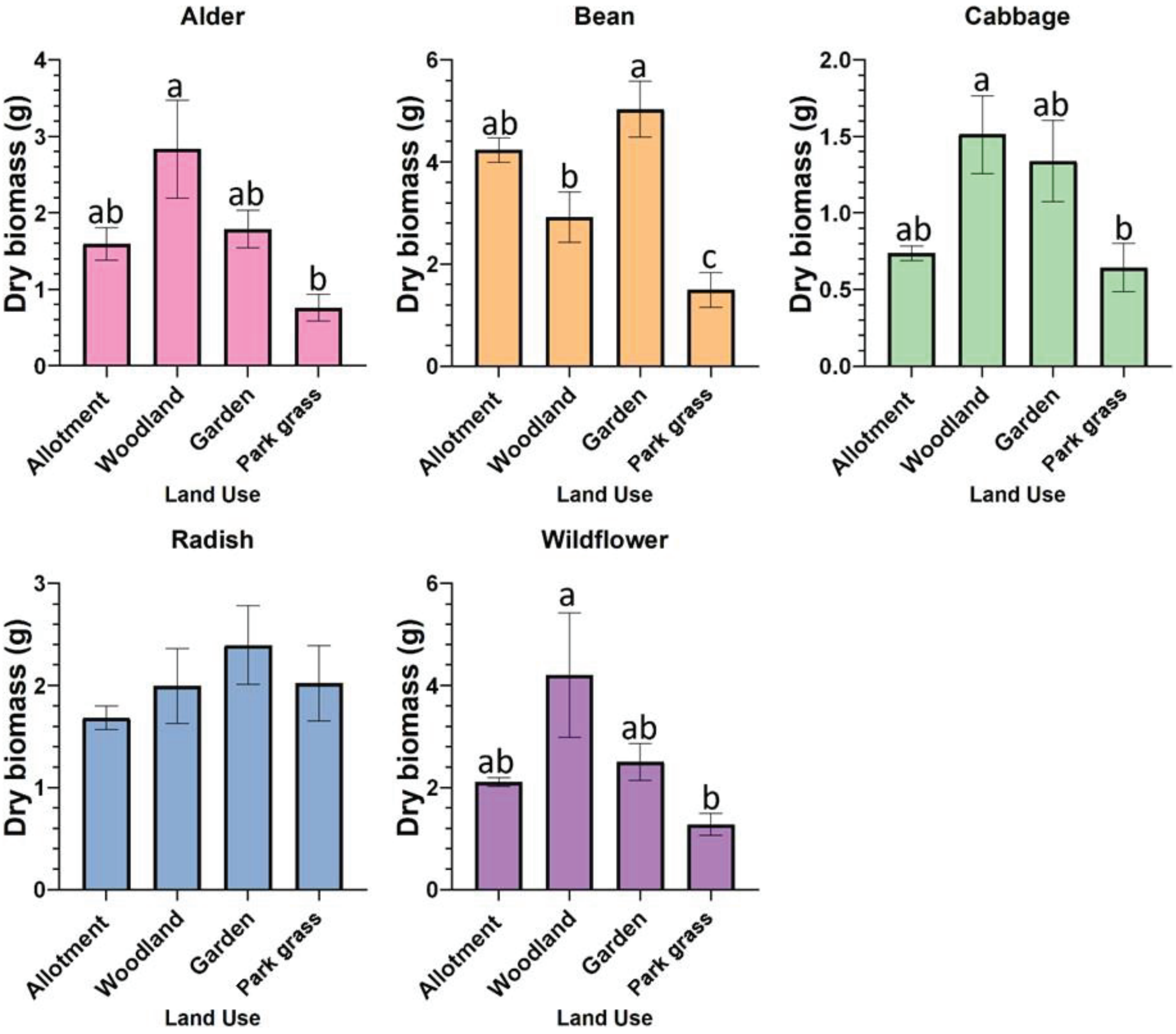


Figure 3.2. The total biomass of plants grown in allotment, garden, park grass, and woodland soils. The total biomass of alder, bean, cabbage, radish, and wildflower species is shown for each greenspace type

Above ground biomass in all species was significantly affected by soils from different greenspace types (Figure 3.3). Park grass produced significantly lower above-ground biomass for alder (F(3,36) = 3.178, p=0.036), bean (F(3,36) = 13.082, p<0.001), cabbage (F(3, 36) = 4.899, p=0.006), radish (F(3, 36) = 3.715, p=0.020) and wildflower (F(3, 36) = 6.779, p<0.001). Allotment only showed a significant difference in biomass production for bean plants, where it produced more than park grass. The highest biomass was seen in woodland (alder, cabbage, wildflower) and garden (bean, cabbage, radish, wildflower) soils. A Type III ANOVA revealed significant main effects of plant species (F(4,180) = 30.65, p < 0.001) and land use (F(3,180) = 4.44, p = 0.0049) on shoot biomass, as well as a significant plant species × land use interaction (F(12,180) = 5.49, p < 0.001). Post hoc Tukey-adjusted pairwise comparisons showed that the effect of land use on shoot biomass differed markedly among species. For example, in Alder, shoot biomass was significantly lower in Park Grass compared to Woodland (p = 0.003), while in Bean, shoot biomass was highest in Garden and Allotment sites and significantly greater than in Park Grass and Woodland (p < 0.01). Other species such as Cabbage, Radish, and Wildflower exhibited less pronounced differences across land uses. Additionally, within each land use, Bean consistently produced significantly greater shoot biomass than other species (p < 0.01), particularly in Allotment and Garden environments. These results indicate that species-specific responses to land use strongly influence shoot biomass production, emphasizing the importance of considering species identity and habitat context in ecological assessments.

Greenspace type had a significant effect on below-ground biomass for four of the five species (Figure 3.3). Alder (F(3,36) = 6.803, p<0.001); bean (F(3, 36) = 12.438, p<0.001); cabbage (F(3, 36) = 4.280, p=0.011); and wildflower (F(3, 36) = 4.642, p=0.008) showed lower biomass one or more greenspaces, but radish (F(3, 36) = 0.551, p=0.651) showed no significant effects. For alder, below-ground biomass was significantly higher in woodland soils than allotment, garden, and park grass soils. For bean plants, below ground biomass was significantly higher in garden soils than woodland soils, while woodland soils produced significantly higher below-ground biomass than park grass soils. For cabbage plants, plants grown in woodland soils had significantly higher below-ground biomass than plants grown in allotment and park grass soils. For the wildflower plants, below-ground biomass was significantly higher in woodland soils than park grass soils.

Analysis of variance revealed significant effects of both plant species (F(4,180) = 3.16, $p = 0.015$) and land use type (F(3,180) = 2.73, $p = 0.046$) on root biomass, as well as a significant interaction between plant species and land use (F(12,180) = 2.75, $p = 0.002$), indicating that the response of root biomass to land use varies among species. Post hoc Tukey-adjusted pairwise comparisons showed that for several species, notably Alder and Wildflower, root biomass was significantly greater in Woodland compared to other land uses, particularly Allotment and Garden areas. For Wildflower species, the Woodland land use had significantly higher root biomass compared to all other land uses ($p < 0.005$). In contrast, species such as Cabbage showed no significant differences in root biomass across land uses. Comparing species within land use types indicated that Wildflower consistently exhibited the highest root biomass in Woodland, significantly surpassing other species ($p < 0.001$). These results suggest that both species identity and land use interactively influence root biomass distribution, with natural woodland environments supporting greater belowground biomass for certain species.

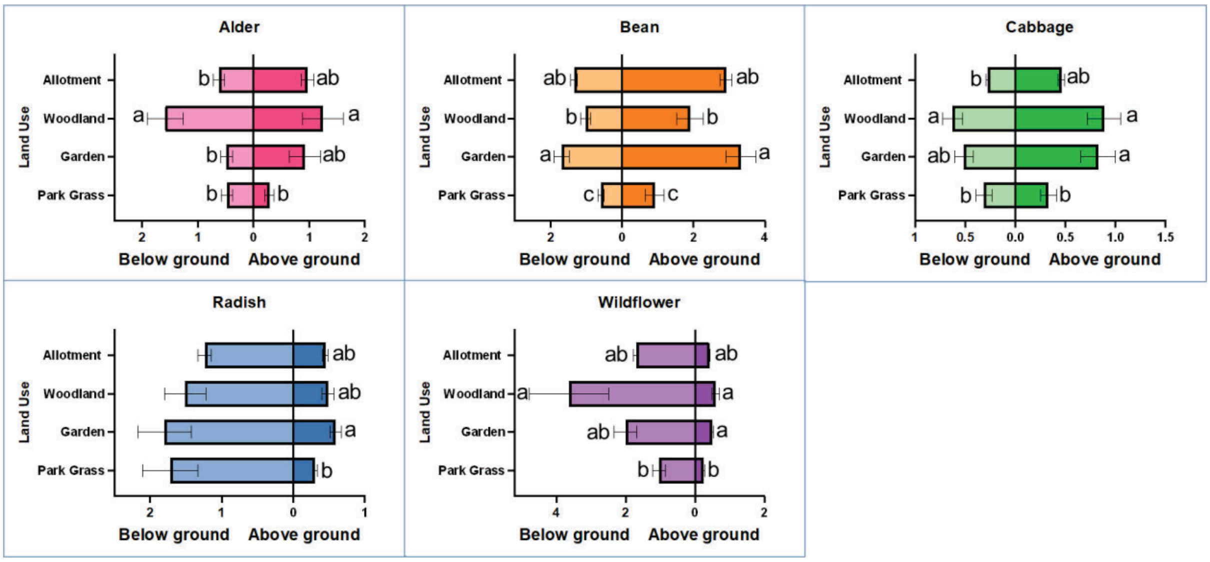


Figure 3.3. The biomass of plants grown in allotments, garden, parkland, and woodland soils. Above ground biomass is shown on the right side of each figure (darker colours), with the below ground biomass shown on the left side of each figure (lighter colours)

Greenspace type had a significant effect on edible biomass for all food crops produced (Figure 3.4); bean (F(3, 36) = 8.640, p<0.001), cabbage (F(3, 36) = 4.899, p=0.006), and radish (F(3, 36) = 3.444, p=0.027). For bean plants, park grassland produced significantly lower edible biomass than any other soil. For both cabbage and radish plants, edible biomass was significantly lower in park grass than garden soils, but cabbage also showed a significant difference between park grass and woodland soils. The analysis of variance (ANOVA) revealed that both plant species identity (F(1,72) = 7.39, p = 0.0082) and land use type (F(3,72) = 10.40, p < 0.001) significantly influenced harvest yield, while the interaction between plant species and land use showed a marginally non-significant trend (F(3,72) = 2.39, p = 0.075). Post hoc pairwise comparisons indicated that, for Bean, harvest was significantly higher in Garden (emmean = 1.79) and Allotment (1.40) compared to Park Grass (0.51), with the Garden vs Park Grass contrast showing the strongest difference (p < 0.0001). No significant differences in harvest across land uses were observed for Radish. When comparing species within each land use, Bean consistently exhibited significantly higher harvest than Radish in Allotment (p = 0.0082) and Garden (p = 0.0011) sites, whereas no significant differences were detected in Park Grass or Woodland. These results suggest that harvest outcomes are strongly dependent on both species and land use context, with Beans generally outperforming Radishes in managed environments such as Gardens and Allotments.

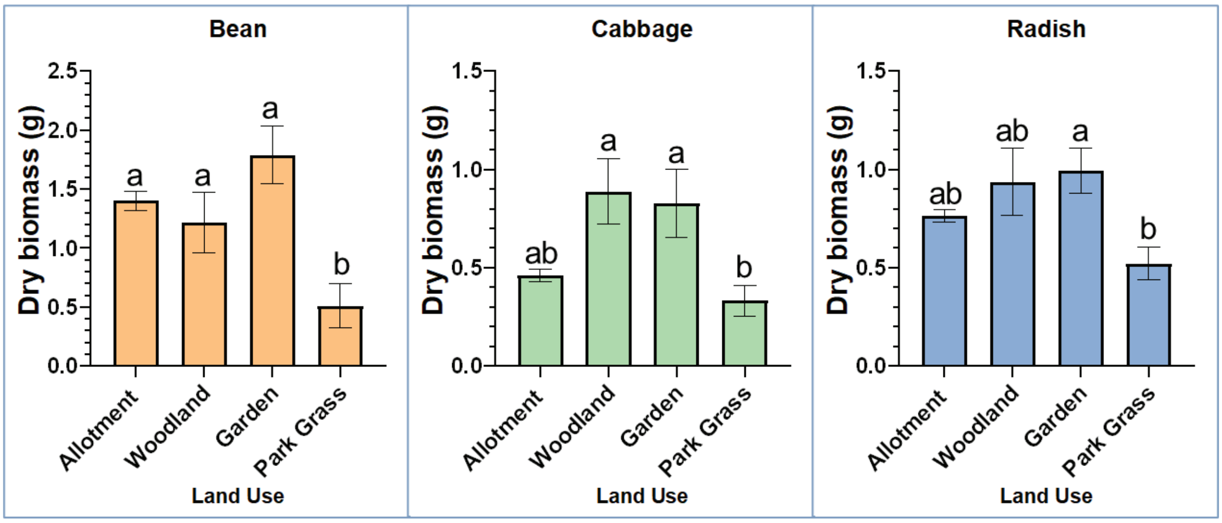


Figure.3.4 The total biomass of edible sections for food crops grown in allotment, garden, parkland, and woodland soils.

Greenspace type showed no significant effect on chlorophyll content (SPAD) of the four species (Figure 3.5.a) alder (F(3, 36) = 2.639, p=0.064), bean (F(3, 36) = 2.176, p=0.108), cabbage (F(3, 36) = 0.725, p=0.544); radish (F(3, 36) = 1.739, p=0.176). A two-way ANOVA assessing the effects of plant species, land use, and their interaction on SPAD values revealed highly significant main effects of plant species ($F_{3,144}$ = 146.69, p < 0.001) and land use ($F_{3,144}$ = 4.06, p = 0.008), while the plant species x land use interaction was not statistically significant ($F_{9,144}$ = 1.67, p =

0.10).

The efficiency of photosystem II in leaves (F_v/F_m) was affected by greenspace type for two species, bean(F(3, 36) = 4.059, p=0.014) and cabbage(F(3, 36) = 4.059, p=0.006) (Figure 3.5.b), but there was no effect in alder (F(3, 36) = 0.353) or radish plants (F(3, 36) = 0.925, p=0.438). The ANOVA results showed that species identity had a highly significant effect on FVFM (F = 189.15, $p < 0.001$), while land use type alone did not (F = 0.039, p = 0.99). However, the interaction between species and land use was significant (F = 2.34, p = 0.017), indicating that the effect of land use on FVFM varies among species. Post hoc comparisons revealed significant differences in FVFM among species within each land use, with Alder consistently having the highest and Radish the lowest FVFM values. Differences among land use types within species were generally not significant except for some contrasts in Bean and Cabbage, particularly involving Park Grass. Overall, species identity is the primary driver of FVFM variation, modulated to a lesser extent by land use depending on species.

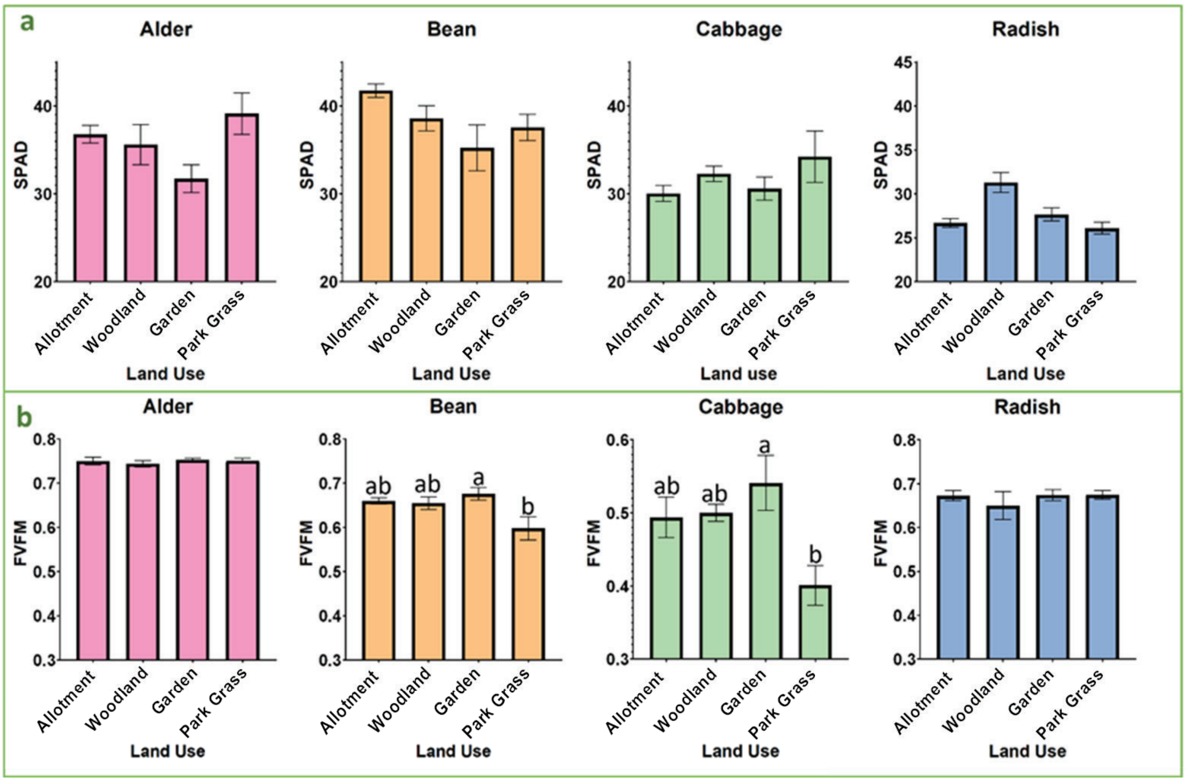


Figure 3.5. Measures of leaf health for plants grown in allotment, garden, parkland, and woodland soils. Graphs demonstrate (a) SPAD measures of leaves and (b) F_v/F_m *measures of leaves.*

<u>Mycorrhizal fungi colonisation</u>

There was no effect of greenspace type on hyphal colonisation (Figure 3.6.a) for alder (F(3,36) = 1.699, p = 0.184), bean (F(3,36) = 1.273, p = 0.298) and wildflower (F(3,36) = 1.089, p = 0.336). Analysis of hyphal colonisation revealed a significant effect of plant species (F(2,108) = 43.03, $p < 0.001$), while land use and the interaction between plant species and land use were not significant (p = 0.91 and p = 0.31, respectively). Post hoc pairwise comparisons showed no significant differences between land use types within any species after Tukey adjustment. However, comparisons across species within each land use type revealed that Alder consistently exhibited significantly lower HC values than both Bean and Wildflower species across all land uses (all $p < 0.001$). Differences between Bean and Wildflower were not statistically significant in most land uses. These results suggest that variation in HC is primarily driven by species identity rather than land use, with Alder differing markedly from the other species regardless of habitat.

Greenspace type significantly affected vesicle colonisation in wildflower plants (Figure 3.6.b) (F(3,36) = 3.587, p = 0.023). There was no effect of greenspace type for alder (F(3,36) = 0.979, p = 0.413) or bean (F(3,36) = 1.144, p = 0.344) plants.

The analysis of variance (Type III) revealed a significant main effect of plant species on the variable VC (F(2,108) = 10.72, p < 0.001), while land use alone did not significantly affect VC (F(3,108) = 0.075, p = 0.973). The interaction between plant species and land use approached significance (F(6,108) = 2.02, p = 0.069), indicating potential species-specific responses to land use. Post hoc pairwise comparisons showed no significant differences in VC among land use categories within each species after Tukey adjustment. However, when comparing species within each land use category, Bean consistently exhibited significantly higher VC values than Alder across all land uses (p < 0.001), and Wildflower generally had intermediate values. Notably, Bean's VC was also significantly higher than Wildflower in Park Grass and Woodland environments (p = 0.016 and p = 0.0005, respectively). These results suggest that species identity is a stronger determinant of VC variation than land use type, although subtle species-by-land-use interactions may exist.

There was no effect of greenspace type seen on arbuscule colonisation of alder (F(3,36) = 0.302, p = 0.824), bean (F(3,36) = 1.763 p = 0.172), and wildflower (F(3,36) = 2.625, p = 0.065). The analysis of variance revealed a significant main effect of plant species on the response variable AC

($F(2,108) = 13.91$, $p < 0.001$), while land use alone had no significant effect ($F(3,108) = 0.003$, $p = 0.9998$). The interaction between Species and Land Use approached significance ($F(6,108) = 2.11$, $p = 0.058$), suggesting potential species-specific responses to land use types. Post hoc pairwise comparisons showed no significant differences among land use levels within Alder species. However, for Bean, the Garden land use had significantly lower AC values compared to Park Grass ($p = 0.007$), and a trend towards lower AC than Allotment and Woodland, though not statistically significant after adjustment. Wildflower species did not show significant pairwise differences across land uses. When comparing species within each land use, Alder consistently had significantly lower AC values than both Bean and Wildflower across all land use categories ($p < 0.05$), with Bean generally exhibiting the highest means. These results indicate that species identity influences AC, and effect of land use is species-dependent, with Beans showing sensitivity to land use types, particularly lower AC in Garden areas.

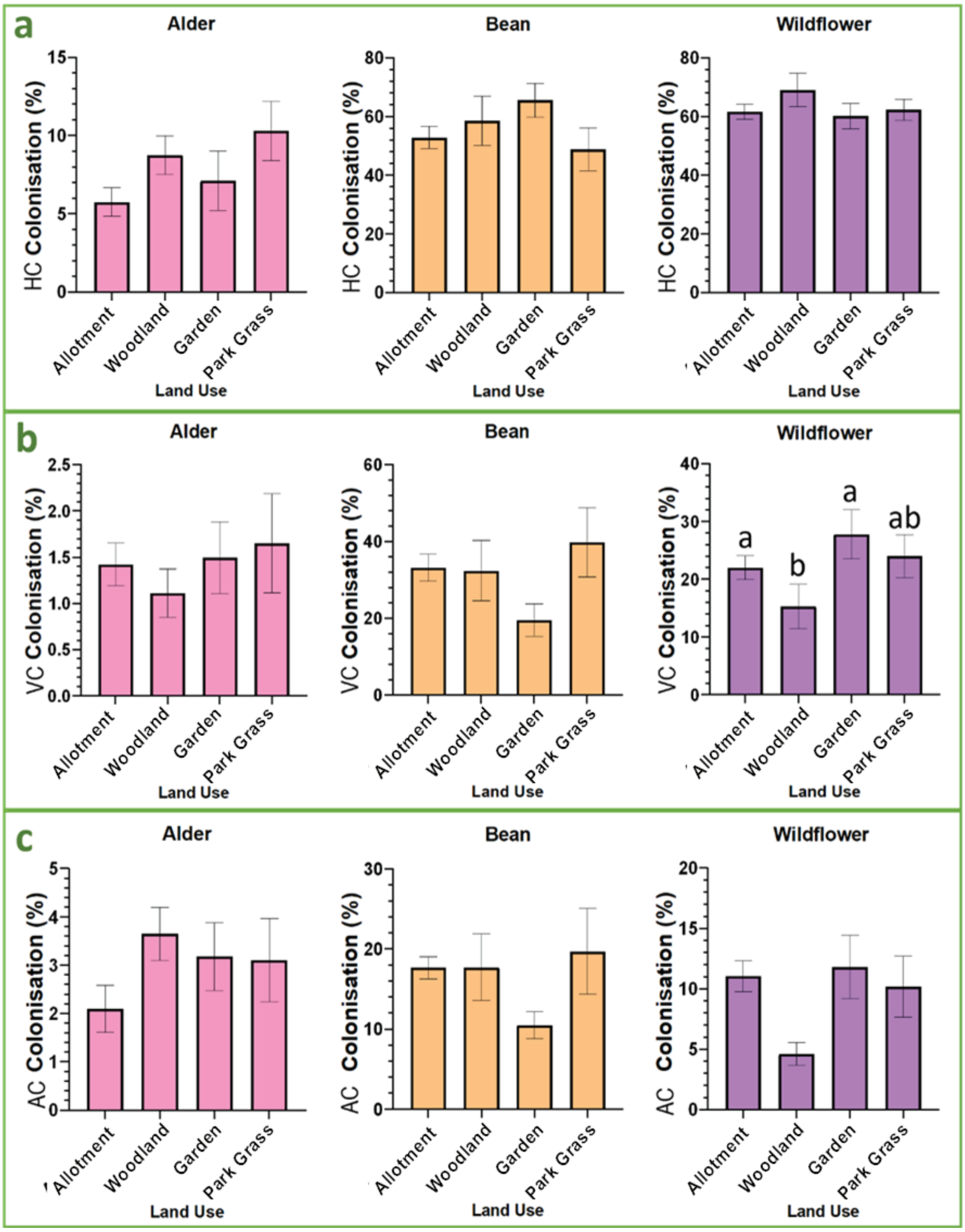


Figure 3.6. Mycorrhizal fungi colonisation of roots for plants grown in allotment, garden, parkland, and woodland soils. Figures show (a) hyphal colonisation percentage of plant roots, (b) vesicule colonisation percentage of plant roots, and (c)) arbuscle colonisation percentage of plant roots.

Influence of soil characteristics on plant health indicators

Alder

The analysis examined the effects of various metal concentrations and soil parameters on different physiological responses of Alder. Most predictors showed no statistically significant influence after adjusting for multiple comparisons (FDR), as indicated by high adjusted p-values and lack of significance. Notably, Iron had a borderline significant positive effect on total and root responses before adjustment, but these did not remain significant after correction. Aluminium showed a near-significant negative effect on root response and a positive effect on hydraulic conductivity (HC), but again these did not meet the adjusted significance threshold. Overall, no predictors were significantly associated with Alder's physiological traits when controlling for false discovery rate, suggesting limited or subtle effects in this dataset.

	Alder Soil health indicators							
	Total biomass	Above-ground biomass	Below-ground biomass	SPAD	FVFM	HC	AC	VC
N	F(1, 38) = 0.05, p=0.81	F(1, 38) = 0.00, p=0.98	F(1, 38) = 0.19, p=0.66	F(1, 38) = 0.81, p=0.37	F(1, 38) = 0.20, p=0.66	F(1, 38) = 0.44, p=0.51	F(1, 38) = 0.01, p=0.94	F(1, 38) = 0.74, p=0.39
C:N	F(1, 38) = 1.76, p=0.18	F(1, 38) = 0.36, p=0.55	F(1, 38) = 3.82, p=0.05	F(1, 38) = 0.05, p=0.83	F(1, 38) = 0.00, p=0.98	F(1, 38) = 3.55, p=0.06	F(1, 38) = 0.15, p=0.70	F(1, 38) = 1.34, p=0.25
TC	F(1, 38) = 0.05, p=0.81	F(1, 38) = 0.11, p=0.74	F(1, 38) = 0.01, p=0.94	F(1, 38) = 0.90, p=0.34	F(1, 38) = 0.03, p=0.87	F(1, 38) = 5.32, p=0.02	F(1, 38) = 0.99, p=0.32	F(1, 38) = 2.84, p=0.09
BC	F(1, 38) = 1.25, p=0.26	F(1, 38) = 0.25, p=0.62	F(1, 38) = 2.69, p=0.10	F(1, 38) = 1.00, p=0.32	F(1, 38) = 0.37, p=0.55	F(1, 38) = 4.48, p=0.03	F(1, 38) = 1.17, p=0.28	F(1, 38) = 4.58, p=0.03
OC	F(1, 38) = 0.35, p=0.55	F(1, 38) = 0.12, p=0.73	F(1, 38) = 0.60, p=0.44	F(1, 38) = 0.46, p=0.50	F(1, 38) = 0.31, p=0.58	F(1, 38) = 3.46, p=0.06	F(1, 38) = 0.35, p=0.56	F(1, 38) = 2.70, p=0.10
Iron	F(1, 38) = 3.05, p=0.08	F(1, 38) = 1.55, p=0.21	F(1, 38) = 4.05, p=0.04	F(1, 38) = 3.28, p=0.07	F(1, 38) = 0.78, p=0.38	F(1, 38) = 0.80, p=0.37	F(1, 38) = 0.27, p=0.61	F(1, 38) = 0.97, p=0.33
Zinc	F(1, 38) = 0.03, p=0.86	F(1, 38) = 0.87, p=0.35	F(1, 38) = 1.99, p=0.16	F(1, 38) = 1.58, p=0.21	F(1, 38) = 0.13, p=0.72	F(1, 38) = 3.02, p=0.08	F(1, 38) = 0.14, p=0.71	F(1, 38) = 1.73, p=0.19
Lead	F(1, 38) = 0.06, p=0.80	F(1, 38) = 0.25, p=0.62	F(1, 38) = 1.13, p=0.29	F(1, 38) = 0.43, p=0.51	F(1, 38) = 0.20, p=0.66	F(1, 38) = 4.02, p=0.04	F(1, 38) = 0.67, p=0.41	F(1, 38) = 2.52, p=0.11
Arsenic	F(1, 38) = 0.43, p=0.51	F(1, 38) = 1.20, p=0.27	F(1, 38) = 0.00, p=0.95	F(1, 38) = 0.15, p=0.70	F(1, 38) = 5.51, p=0.02	F(1, 38) = 0.25, p=0.62	F(1, 38) = 0.72, p=0.40	F(1, 38) = 0.04, p=0.85
Aluminum	F(1, 38) = 0.05, p=0.05	F(1, 38) = 3.18, p=0.07	F(1, 38) = 2.95, p=0.09	F(1, 38) = 0.66, p=0.42	F(1, 38) = 2.52, p=0.11	F(1, 38) = 0.83, p=0.36	F(1, 38) = 0.02, p=0.90	F(1, 38) = 0.20, p=0.65
Copper	F(1, 38) = 0.57, p=0.45	F(1, 38) = 0.16, p=0.69	F(1, 38) = 1.04, p=0.31	F(1, 38) = 0.09, p=0.77	F(1, 38) = 3.00, p=0.08	F(1, 38) = 3.48, p=0.06	F(1, 38) = 1.51, p=0.22	F(1, 38) = 2.63, p=0.11
Chronium	F(1, 38) = 1.55, p=0.21	F(1, 38) = 2.18, p=0.14	F(1, 38) = 0.61, p=0.43	F(1, 38) = 0.36, p=0.55	F(1, 38) = 1.13, p=0.29	F(1, 38) = 6.86, p=0.01	F(1, 38) = 5.04, p=0.02	F(1, 38) = 3.60, p=0.06
Cadmium	F(1, 38) = 0.03, p=0.87	F(1, 38) = 0.24, p=0.62	F(1, 38) = 0.78, p=0.38	F(1, 38) = 0.26, p=0.61	F(1, 38) = 0.02, p=0.89	F(1, 38) = 2.25, p=0.13	F(1, 38) = 1.10, p=0.29	F(1, 38) = 1.72, p=0.19
Nickel	F(1, 38) = 0.68, p=0.41	F(1, 38) = 0.68, p=0.41	F(1, 38) = 0.47, p=0.49	F(1, 38) = 0.35, p=0.55	F(1, 38) = 0.43, p=0.51	F(1, 38) = 7.25, p=0.01	F(1, 38) = 5.91, p=0.02	F(1, 38) = 4.38, p=0.04
Calcium	F(1, 38) = 0.03, p=0.85	F(1, 38) = 1.65, p=0.20	F(1, 38) = 3.46, p=0.06	F(1, 38) = 4.55, p=0.03	F(1, 38) = 2.41, p=0.12	F(1, 38) = 1.12, p=0.29	F(1, 38) = 0.22, p=0.64	F(1, 38) = 0.81, p=0.37
Potassium	F(1, 38) = 1.45, p=0.23	F(1, 38) = 1.47, p=0.23	F(1, 38) = 0.96, p=0.33	F(1, 38) = 0.01, p=0.91	F(1, 38) = 0.07, p=0.80	F(1, 38) = 0.01, p=0.93	F(1, 38) = 0.19, p=0.67	F(1, 38) = 0.10, p=0.76
Magnesium	F(1, 38) = 0.51, p=0.47	F(1, 38) = 1.16, p=0.28	F(1, 38) = 0.04, p=0.85	F(1, 38) = 1.41, p=0.24	F(1, 38) = 0.23, p=0.64	F(1, 38) = 0.21, p=0.65	F(1, 38) = 0.01, p=0.92	F(1, 38) = 0.45, p=0.50
Phosphorus	F(1, 38) = 0.01, p=0.91	F(1, 38) = 0.40, p=0.53	F(1, 38) = 0.88, p=0.35	F(1, 38) = 1.85, p=0.17	F(1, 38) = 0.61, p=0.43	F(1, 38) = 4.19, p=0.04	F(1, 38) = 0.20, p=0.65	F(1, 38) = 4.10, p=0.04
Selenium	F(1, 38) = 0.40, p=0.53	F(1, 38) = 1.43, p=0.23	F(1, 38) = 0.01, p=0.92	F(1, 38) = 1.80, p=0.18	F(1, 38) = 10.38, p=0.00	F(1, 38) = 0.19, p=0.67	F(1, 38) = 0.00, p=0.96	F(1, 38) = 0.56, p=0.46

Table 1: The effect of soil characteristics on indicators of plant health, including biomass of the plant, chlorophyll and photochemical efficiency, and mycorrhizal colonisation percentage of roots. Colours of cells indicate positive response (green), negative response (red) and non-significant effects (white) tested using linear models

Beans

There were positive effects of zinc, aluminium, calcium, and phosphorus on all biomass measures. There were positive effects of lead and cadmium levels on total, above ground, and below-ground biomass, but no significant effect on harvest biomass. There were positive effects of TC and BC on above-ground and harvest biomass. SPAD showed a significant direction of response with BC, while F_v/F_m showed significant relationships with N, TC, OC, and calcium. Arsenic, magnesium and selenium had positive effects on hyphal colonisation, arbuscule colonisation and vesicule colonisation.

	Bean Soil health indicator								
	Total biomass	**Above-ground biomass**	**Below-ground biomass**	**Harvest biomass**	**SPAD**	**FVFM**	**HC**	**AC**	**VC**
N	F(1, 38) = 0.01, p=0.90	F(1, 38) = 0.14, p=0.71	F(1, 38) = 0.23, p=0.63	F(1, 38) = 0.65, p=0.42	F(1, 38) = 1.06, p=0.30	F(1, 38) = 6.23 p=0.01	F(1, 38) = 0.06, p=0.80	F(1, 38) = 0.06, p=0.80	F(1, 38) = 0.23, p=0.63
C:N	F(1, 38) = 2.87, p=0.09	F(1, 38) = 2.60, p=0.11	F(1, 38) = 2.48, p=0.12	F(1, 38) = 1.51, p=0.22	F(1, 38) = 3.17, p=0.08	F(1, 38) = 0.35, p=0.55	F(1, 38) = 0.65, p=0.42	F(1, 38) = 1.45, p=0.23	F(1, 38) = 1.95, p=0.16
TC	F(1, 38) = 3.30, p=0.07	F(1, 38) = 4.16, p=0.04	F(1, 38) = 1.08, p=0.30	F(1, 38) = 5.06, p=0.02	F(1, 38) = 3.55, p=0.06	F(1, 38) = 4.13, p=0.04	F(1, 38) = 0.29, p=0.59	F(1, 38) = 0.54, p=0.46	F(1, 38) = 0.86, p=0.35
BC	F(1, 38) = 2.59, p=0.11	F(1, 38) = 4.50, p=0.03	F(1, 38) = 0.08, p=0.77	F(1, 38) = 5.40, p=0.02	F(1, 38) = 5.01, p=0.03	F(1, 38) = 0.10, p=0.76	F(1, 38) = 0.18, p=0.67	F(1, 38) = 0.23, p=0.63	F(1, 38) = 0.90, p=0.34
OC	F(1, 38) = 1.04, p=0.31	F(1, 38) = 1.45, p=0.23	F(1, 38) = 0.20, p=0.66	F(1, 38) = 1.78, p=0.18	F(1, 38) = 3.43, p=0.06	F(1, 38) = 4.39, p=0.04	F(1, 38) = 0.53, p=0.47	F(1, 38) = 0.87, p=0.35	F(1, 38) = 1.53, p=0.22
Iron	F(1, 38) = 0.03, p=0.87	F(1, 38) = 0.02, p=0.90	F(1, 38) = 0.04, p=0.84	F(1, 38) = 0.04, p=0.84	F(1, 38) = 0.43, p=0.51	F(1, 38) = 1.00, p=0.32	F(1, 38) = 0.08, p=0.77	F(1, 38) = 0.16, p=0.69	F(1, 38) = 0.25, p=0.62
Zinc	F(1, 38) = 20.46, p=0.00	F(1, 38) = 17.15, p=0.00	F(1, 38) = 19.09, p=0.00	F(1, 38) = 8.06, p=0.00	F(1, 38) = 0.62, p=0.43	F(1, 38) = 2.14, p=0.14	F(1, 38) = 0.12, p=0.73	F(1, 38) = 0.18, p=0.67	F(1, 38) = 0.13, p=0.71
Lead	F(1, 38) = 5.73, p=0.02	F(1, 38) = 4.66, p=0.03	F(1, 38) = 6.44, p=0.01	F(1, 38) = 1.28, p=0.26	F(1, 38) = 0.06, p=0.81	F(1, 38) = 0.33, p=0.57	F(1, 38) = 2.11, p=0.15	F(1, 38) = 2.51, p=0.11	F(1, 38) = 2.81, p=0.09
Arsenic	F(1, 38) = 0.87, p=0.35	F(1, 38) = 1.08, p=0.30	F(1, 38) = 0.28, p=0.60	F(1, 38) = 1.35, p=0.24	F(1, 38) = 1.16, p=0.28	F(1, 38) = 1.56, p=0.21	F(1, 38) = 5.46, p=0.02	F(1, 38) = 5.75, p=0.02	F(1, 38) = 6.13, p=0.01
Aluminum	F(1, 38) = 5.97, p=0.01	F(1, 38) = 4.96, p=0.03	F(1, 38) = 6.06, p=0.01	F(1, 38) = 4.69, p=0.03	F(1, 38) = 0.14, p=0.71	F(1, 38) = 2.15, p=0.14	F(1, 38) = 1.08, p=0.30	F(1, 38) = 0.45, p=0.50	F(1, 38) = 0.11, p=0.74
Copper	F(1, 38) = 1.04, p=0.31	F(1, 38) = 0.93, p=0.34	F(1, 38) = 1.07, p=0.30	F(1, 38) = 0.18, p=0.67	F(1, 38) = 0.02, p=0.89	F(1, 38) = 0.06, p=0.81	F(1, 38) = 2.58, p=0.11	F(1, 38) = 2.46, p=0.12	F(1, 38) = 2.12, p=0.15
Chronium	F(1, 38) = 0.74, p=0.39	F(1, 38) = 0.74, p=0.39	F(1, 38) = 0.53, p=0.47	F(1, 38) = 0.26, p=0.61	F(1, 38) = 0.00, p=0.98	F(1, 38) = 0.80, p=0.37	F(1, 38) = 1.56, p=0.21	F(1, 38) = 2.28, p=0.13	F(1, 38) = 1.89, p=0.17
Cadmium	F(1, 38) = 6.63, p=0.01	F(1, 38) = 5.66, p=0.02	F(1, 38) = 6.64, p=0.01	F(1, 38) = 2.95, p=0.09	F(1, 38) = 0.06, p=0.80	F(1, 38) = 1.48, p=0.22	F(1, 38) = 1.25, p=0.26	F(1, 38) = 2.51, p=0.11	F(1, 38) = ,2.60 p=0.11
Nickel	F(1, 38) = 0.23, p=0.63	F(1, 38) = 0.18, p=0.67	F(1, 38) = 0.29, p=0.59	F(1, 38) = 0.00, p=0.96	F(1, 38) = 0.16, p=0.69	F(1, 38) = 0.40, p=0.52	F(1, 38) = 1.04, p=0.31	F(1, 38) = 1.34, p=0.25	F(1, 38) = 1.03, p=0.31
Calcium	F(1, 38) = 31.48, p=0.00	F(1, 38) = 34.85, p=0.00	F(1, 38) = 14.54, p=0.00	F(1, 38) = 23.12, p=0.00	F(1, 38) = 0.19, p=0.66	F(1, 38) = 4.20, p=0.04	F(1, 38) = 0.15, p=0.70	F(1, 38) = 0.03, p=0.87	F(1, 38) = 0.00, p=0.96
Potassium	F(1, 38) = 0.11, p=0.74	F(1, 38) = 0.24, p=0.63	F(1, 38) = 0.00, p=0.96	F(1, 38) = 0.49, p=0.49	F(1, 38) = 0.41, p=0.52	F(1, 38) = 0.01, p=0.94	F(1, 38) = 1.41, p=0.24	F(1, 38) = 3.92, p=0.05	F(1, 38) = 3.53, p=0.06
Magnesium	F(1, 38) = 1.14, p=0.29	F(1, 38) = 1.71, p=0.19	F(1, 38) = 0.12, p=0.72	F(1, 38) = 2.32, p=0.13	F(1, 38) = 0.84, p=0.36	F(1, 38) = 0.14, p=0.71	F(1, 38) = 5.27, p=0.02	F(1, 38) = 11.07, p=0.00	F(1, 38) = 7.59, p=0.01
Phosphorus	F(1, 38) = 10.27, p=0.00	F(1, 38) = 9.44, p=0.00	F(1, 38) = 8.66, p=0.00	F(1, 38) = 3.96, p=0.05	F(1, 38) = 1.29, p=0.26	F(1, 38) = 0.61, p=0.44	F(1, 38) = 0.93, p=0.33	F(1, 38) = 1.21, p=0.27	F(1, 38) = 2.08, p=0.15
Selenium	F(1, 38) = 2.69, p=0.10	F(1, 38) = 3.32, p=0.07	F(1, 38) = 0.96, p=0.33	F(1, 38) = 3.03, p=0.08	F(1, 38) = 1.46, p=0.23	F(1, 38) = 0.73, p=0.39	F(1, 38) = 4.66, p=0.03	F(1, 38) = 5.51, p=0.02	F(1, 38) = 5.79, p=0.02

Table 2. The effect of soil characteristics on indicators of bean plant health, including biomass of the plant, chlorophyll and photochemical efficiency, and mycorrhizal colonisation percentage of roots. Colours of cells indicate positive response (green), negative response (red) and non-significant effects (white) tested using linear models

Cabbage

There were positive effects of copper and cadmium levels in soil on SPAD, and of TC and calcium on F_v/F_m .

	Cabbage Soil health indicator					
	Total biomass	**Above-ground biomass**	**Below-ground biomass**	**Harvest biomass**	**SPAD**	**FVFM**
N	F(1, 38) = 0.81, p=0.37	F(1, 38) = 1.05, p=0.31	F(1, 38) = 0.41, p=0.52	F(1, 38) = 1.05, p=0.31	F(1, 38) = 0.48, p=0.49	F(1, 38) = 0.01, p=0.92
C:N	F(1, 38) = 0.10, p=0.75	F(1, 38) = 0.15, p=0.91	F(1, 38) = 0.98, p=0.32	F(1, 38) = 0.15, p=0.91	F(1, 38) = 3.10, p=0.08	F(1, 38) = 2.48, p=0.12
TC	F(1, 38) = 1.33, p=0.25	F(1, 38) = 2.38, p=0.12	F(1, 38) = 0.22, p=0.64	F(1, 38) = 2.38, p=0.12	F(1, 38) = 0.60, p=0.44	F(1, 38) = 2.21, p=0.14
BC	F(1, 38) = 0.53, p=0.47	F(1, 38) = 1.01, p=0.31	F(1, 38) = 0.05, p=0.81	F(1, 38) = 1.01, p=0.31	F(1, 38) = 0.40, p=0.53	F(1, 38) = 3.87, p=0.05
OC	F(1, 38) = 0.30, p=0.59	F(1, 38) = 0.80, p=0.37	F(1, 38) = 0.00, p=0.97	F(1, 38) = 0.80, p=0.37	F(1, 38) = 0.47, p=0.49	F(1, 38) = 0.53, p=0.47
Aluminum	F(1, 38) = 1.83, p=0.28	F(1, 38) = 1.98, p=0.16	F(1, 38) = 1.38, p=0.24	F(1, 38) = 1.98, p=0.16	F(1, 38) = 2.07, p=0.15	F(1, 38) = 1.21, p=0.27
Arsenic	F(1, 38) = 0.05, p=0.82	F(1, 38) = 0.36, p=0.55	F(1, 38) = 0.14, p=0.71	F(1, 38) = 0.36, p=0.55	F(1, 38) = 2.04, p=0.15	F(1, 38) = 1.17, p=0.28
Cadmium	F(1, 38) = 0.01, p=0.93	F(1, 38) = 0.13, p=0.71	F(1, 38) = 0.13, p=0.72	F(1, 38) = 0.13, p=0.71	F(1, 38) = 4.50, p=0.03	F(1, 38) = 0.00, p=0.96
Calcium	F(1, 38) = 0.32, p=0.57	F(1, 38) = 0.40, p=0.52	F(1, 38) = 0.17, p=0.68	F(1, 38) = 0.40, p=0.52	F(1, 38) = 0.10, p=0.75	F(1, 38) = 4.24, p=0.04
Chronium	F(1, 38) = 0.18, p=0.67	F(1, 38) = 0.16, p=0.69	F(1, 38) = 0.19, p=0.66	F(1, 38) = 0.16, p=0.69	F(1, 38) = 0.01, p=0.92	F(1, 38) = 1.66, p=0.20
Copper	F(1, 38) = 0.02, p=0.89	F(1, 38) = 0.25, p=0.62	F(1, 38) = 0.19, p=0.66	F(1, 38) = 0.25, p=0.62	F(1, 38) = 7.04, p=0.01	F(1, 38) = 0.85, p=0.36
Iron	F(1, 38) = 0.09, p=0.76	F(1, 38) = 0.03, p=0.86	F(1, 38) = 0.23, p=0.63	F(1, 38) = 0.03, p=0.86	F(1, 38) = 2.43, p=0.12	F(1, 38) = 2.39, p=0.12
Lead	F(1, 38) = 0.02, p=0.88	F(1, 38) = 0.10, p=0.75	F(1, 38) = 0.79, p=0.75	F(1, 38) = 0.10, p=0.75	F(1, 38) = 3.73, p=0.05	F(1, 38) = 0.16, p=0.69
Magnesium	F(1, 38) = 0.67, p=0.41	F(1, 38) = 0.18, p=0.67	F(1, 38) = 1.98, p=0.16	F(1, 38) = 0.18, p=0.67	F(1, 38) = 1.33, p=0.25	F(1, 38) = 0.04, p=0.85
Nickel	F(1, 38) = 0.08, p=0.77	F(1, 38) = 0.10, p=0.75	F(1, 38) = 0.05, p=0.82	F(1, 38) = 0.10, p=0.75	F(1, 38) = 0.44, p=0.51	F(1, 38) = 2.47, p=0.12
Phosphorus	F(1, 38) = 0.01, p=0.92	F(1, 38) = 0.12, p=0.73	F(1, 38) = 0.09, p=0.76	F(1, 38) = 0.12, p=0.73	F(1, 38) = 1.57, p=0.21	F(1, 38) = 0.11, p=0.74
Potassium	F(1, 38) = 1.97, p=0.16	F(1, 38) = 1.34, p=0.25	F(1, 38) = 2.91, p=0.09	F(1, 38) = 1.34, p=0.25	F(1, 38) = 0.84, p=0.36	F(1, 38) = 0.54, p=0.46
Selenium	F(1, 38) = 0.05, p=0.82	F(1, 38) = 0.40, p=0.53	F(1, 38) = 0.19, p=0.67	F(1, 38) = 0.40, p=0.53	F(1, 38) = 1.14, p=0.29	F(1, 38) = 0.00, p=0.96
Zinc	F(1, 38) = 1.25, p=0.26	F(1, 38) = 2.12, p=0.15	F(1, 38) = 0.28, p=0.60	F(1, 38) = 2.12, p=0.15	F(1, 38) = 1.50, p=0.22	F(1, 38) = 0.02, p=0.88

Table 3: The effect of soil characteristics on indicators of cabbage plant health, including biomass of the plant, chlorophyll and photochemical efficiency, and mycorrhizal colonisation

percentage of roots. Colours of cells indicate positive response (green), negative response (red) and non-significant effects (white) tested using linear models

Radish

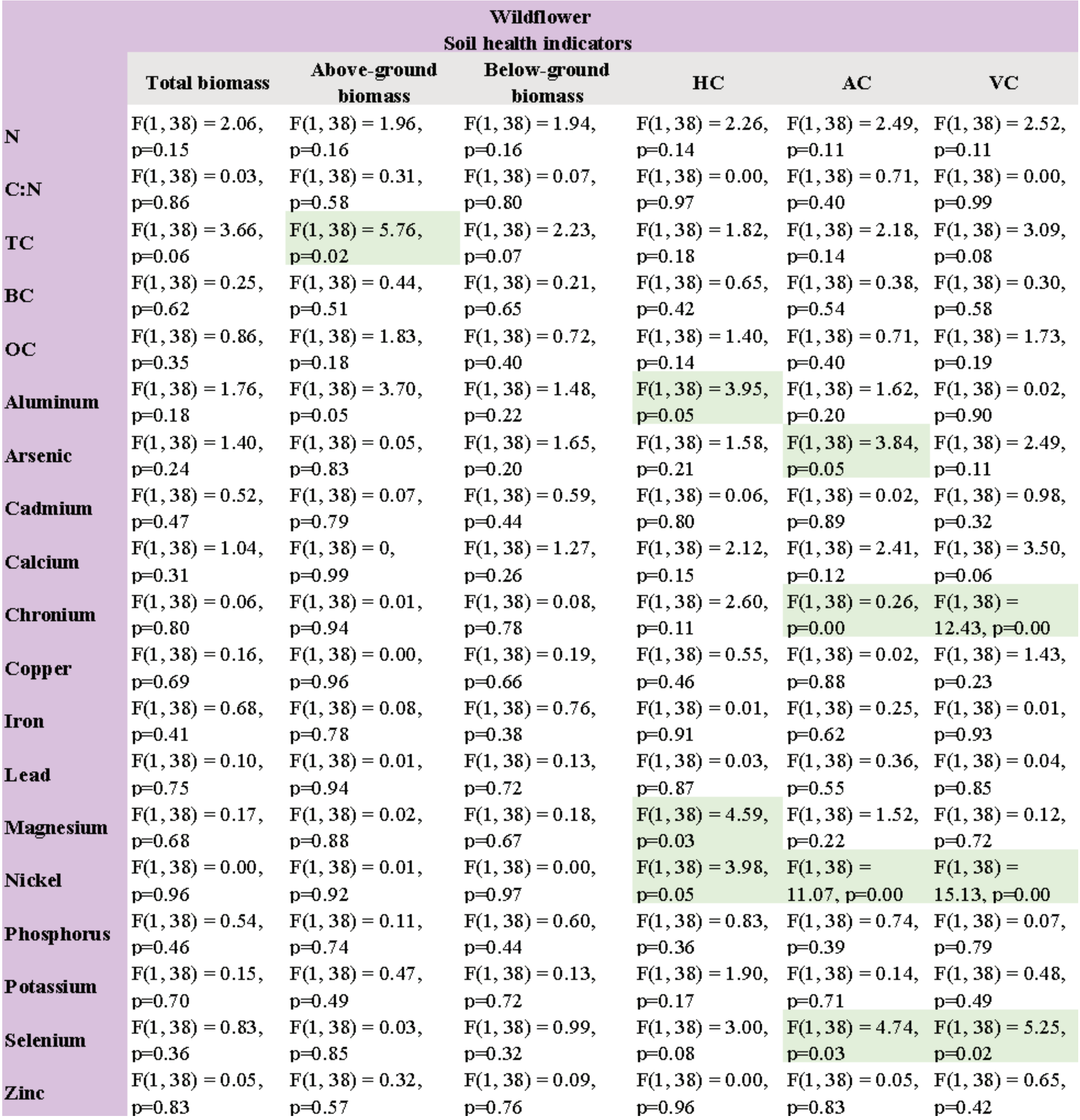
There were positive effects of zinc, aluminium, calcium, and phosphorus on all biomass measures. There were positive effects of lead and cadmium levels on total, above ground, and below-ground biomass, but no significant effect on harvest biomass. There were positive effects of TC and BC on above-ground and harvest biomass. SPAD showed a significant relationship with BC, while F_v/F_m showed significant relationships with N, TC, OC, and calcium. Arsenic, magnesium and selenium had positive effects on hyphal colonisation, arbuscule colonisation and vesicule colonisation.

	Radish Soil health indicators					
	Total biomass	**Above-ground biomass**	**Below-ground biomass**	**Harvest biomass**	**SPAD**	**FVFM**
N	F(1, 38) = 1.69, p=0.19	F(1, 38) = 0.72, p=0.40	F(1, 38) = 1.43, p=0.23	F(1, 38) = 2.06, p=0.15	F(1, 38) = 0.34, p=0.56	F(1, 38) = 0.66, p=0.42
C:N	F(1, 38) = 1.93, p=0.16	F(1, 38) = 0.68, p=0.41	F(1, 38) = 1.67, p=0.20	F(1, 38) = 0.01, p=0.92	F(1, 38) = 4.20, p=0.04	F(1, 38) = 1.12, p=0.29
TC	F(1, 38) = 4.03, p=0.04	F(1, 38) = 3.18, p=0.07	F(1, 38) = 2.95, p=0.09	F(1, 38) = 4.52, p=0.03	F(1, 38) = 5.63, p=0.02	F(1, 38) = 2.46, p=0.12
BC	F(1, 38) = 0.47, p=0.49	F(1, 38) = 4.14, p=0.04	F(1, 38) = 0.09, p=0.76	F(1, 38) = 2.75, p=0.10	F(1, 38) = 6.31, p=0.01	F(1, 38) = 5.77, p=0.02
OC	F(1, 38) = 2.66, p=0.10	F(1, 38) = 1.67, p=0.20	F(1, 38) = 2.07, p=0.15	F(1, 38) = 1.54, p=0.22	F(1, 38) = 3.67, p=0.06	F(1, 38) = 2.41, p=0.12
Aluminum	F(1, 38) = 0.72, p=0.40	F(1, 38) = 5.32, p=0.02	F(1, 38) = 0.17, p=0.68	F(1, 38) = 5.48, p=0.02	F(1, 38) = 1.52, p=0.22	F(1, 38) = 0.30, p=0.58
Arsenic	F(1, 38) = 0.14, p=0.71	F(1, 38) = 0.02, p=0.90	F(1, 38) = 0.18, p=0.67	F(1, 38) = 0.02, p=0.88	F(1, 38) = 0.12, p=0.73	F(1, 38) = 0.07, p=0.80
Cadmium	F(1, 38) = 0.02, p=0.90	F(1, 38) = 1.88, p=0.17	F(1, 38) = 0.19, p=0.66	F(1, 38) = 0.60, p=0.44	F(1, 38) = 2.45, p=0.12	F(1, 38) = 1.74, p=0.19
Calcium	F(1, 38) = 0.00, p=0.96	F(1, 38) = 6.28, p=0.01	F(1, 38) = 0.33, p=0.57	F(1, 38) = 3.63, p=0.06	F(1, 38) = 1.55, p=0.21	F(1, 38) = 0.80, p=0.37
Chronium	F(1, 38) = 0.75, p=0.39	F(1, 38) = 0.15, p=0.70	F(1, 38) = 0.70, p=0.40	F(1, 38) = 0.53, p=0.47	F(1, 38) = 4.57, p=0.03	F(1, 38) = 2.34, p=0.13
Copper	F(1, 38) = 0.85, p=0.36	F(1, 38) = 1.01, p=0.31	F(1, 38) = 1.48, p=0.22	F(1, 38) = 0.31, p=0.58	F(1, 38) = 3.43, p=0.06	F(1, 38) = 2.23, p=0.14
Iron	F(1, 38) = 2.06, p=0.15	F(1, 38) = 0.74, p=0.39	F(1, 38) = 1.77, p=0.18	F(1, 38) = 0.35, p=0.55	F(1, 38) = 0.75, p=0.39	F(1, 38) = 0.69, p=0.41
Lead	F(1, 38) = 0.33, p=0.56	F(1, 38) = 1.74, p=0.19	F(1, 38) = 0.83, p=0.36	F(1, 38) = 0.28, p=0.60	F(1, 38) = 3.44, p=0.06	F(1, 38) = 1.84, p=0.17
Magnesium	F(1, 38) = 0.07, p=0.80	F(1, 38) = 0.02, p=0.88	F(1, 38) = 0.09, p=0.76	F(1, 38) = 0.18, p=0.67	F(1, 38) = 0.10, p=0.75	F(1, 38) = 0.0, 33p=0.57
Nickel	F(1, 38) = 0.66, p=0.41	F(1, 38) = 0.07, p=0.79	F(1, 38) = 0.66, p=0.42	F(1, 38) = 0.34, p=0.56	F(1, 38) = 4.90, p=0.03	F(1, 38) = 3.14, p=0.08
Phosphorus	F(1, 38) = 0.23, p=0.63	F(1, 38) = 3.62, p=0.06	F(1, 38) = 0.85, p=0.36	F(1, 38) = 1.01, p=0.31	F(1, 38) = 4.10, p=0.04	F(1, 38) = 4.33, p=0.04
Potassium	F(1, 38) = 1.30, p=0.25	F(1, 38) = 0.40, p=0.52	F(1, 38) = 1.15, p=0.28	F(1, 38) = 0.10, p=0.75	F(1, 38) = 0.09, p=0.77	F(1, 38) = 0.28, p=0.60
Selenium	F(1, 38) = 0.01, p=0.91	F(1, 38) = 0.00, p=0.95	F(1, 38) = 0.01, p=0.92	F(1, 38) = 0.18, p=0.67	F(1, 38) = 0.46, p=0.50	F(1, 38) = 0.75, p=0.39
Zinc	F(1, 38) = 0.14, p=0.71	F(1, 38) = 3.59, p=0.06	F(1, 38) = 0.65, p=0.42	F(1, 38) = 0.66, p=0.42	F(1, 38) = 3.53, p=0.06	F(1, 38) = 1.57, p=0.21

Table 4: The effect of soil characteristics on indicators of radish plant health, including biomass of the plant, chlorophyll and photochemical efficiency, and mycorrhizal colonisation percentage of roots. Colours of cells indicate positive response (green), negative response (red) and non-significant effects (white) tested using linear models

Wildflower

There were positive effects of TC on above-ground biomass. There were positive effects of nickel on hyphal colonisation, arbuscule colonisation and vesicule colonisation, and positive effects of chromium and selenium on arbuscule colonisation and vesicule colonisation. There were positive effects of aluminium and magnesium on hyphal colonisation, and of arsenic on arbuscule colonisation.



	Wildflower Soil health indicators					
	Total biomass	**Above-ground biomass**	**Below-ground biomass**	**HC**	**AC**	**VC**
N	F(1, 38) = 2.06, p=0.15	F(1, 38) = 1.96, p=0.16	F(1, 38) = 1.94, p=0.16	F(1, 38) = 2.26, p=0.14	F(1, 38) = 2.49, p=0.11	F(1, 38) = 2.52, p=0.11
C:N	F(1, 38) = 0.03, p=0.86	F(1, 38) = 0.31, p=0.58	F(1, 38) = 0.07, p=0.80	F(1, 38) = 0.00, p=0.97	F(1, 38) = 0.71, p=0.40	F(1, 38) = 0.00, p=0.99
TC	F(1, 38) = 3.66, p=0.06	F(1, 38) = 5.76, p=0.02	F(1, 38) = 2.23, p=0.07	F(1, 38) = 1.82, p=0.18	F(1, 38) = 2.18, p=0.14	F(1, 38) = 3.09, p=0.08
BC	F(1, 38) = 0.25, p=0.62	F(1, 38) = 0.44, p=0.51	F(1, 38) = 0.21, p=0.65	F(1, 38) = 0.65, p=0.42	F(1, 38) = 0.38, p=0.54	F(1, 38) = 0.30, p=0.58
OC	F(1, 38) = 0.86, p=0.35	F(1, 38) = 1.83, p=0.18	F(1, 38) = 0.72, p=0.40	F(1, 38) = 1.40, p=0.14	F(1, 38) = 0.71, p=0.40	F(1, 38) = 1.73, p=0.19
Aluminum	F(1, 38) = 1.76, p=0.18	F(1, 38) = 3.70, p=0.05	F(1, 38) = 1.48, p=0.22	F(1, 38) = 3.95, p=0.05	F(1, 38) = 1.62, p=0.20	F(1, 38) = 0.02, p=0.90
Arsenic	F(1, 38) = 1.40, p=0.24	F(1, 38) = 0.05, p=0.83	F(1, 38) = 1.65, p=0.20	F(1, 38) = 1.58, p=0.21	F(1, 38) = 3.84, p=0.05	F(1, 38) = 2.49, p=0.11
Cadmium	F(1, 38) = 0.52, p=0.47	F(1, 38) = 0.07, p=0.79	F(1, 38) = 0.59, p=0.44	F(1, 38) = 0.06, p=0.80	F(1, 38) = 0.02, p=0.89	F(1, 38) = 0.98, p=0.32
Calcium	F(1, 38) = 1.04, p=0.31	F(1, 38) = 0, p=0.99	F(1, 38) = 1.27, p=0.26	F(1, 38) = 2.12, p=0.15	F(1, 38) = 2.41, p=0.12	F(1, 38) = 3.50, p=0.06
Chronium	F(1, 38) = 0.06, p=0.80	F(1, 38) = 0.01, p=0.94	F(1, 38) = 0.08, p=0.78	F(1, 38) = 2.60, p=0.11	F(1, 38) = 0.26, p=0.00	F(1, 38) = 12.43, p=0.00
Copper	F(1, 38) = 0.16, p=0.69	F(1, 38) = 0.00, p=0.96	F(1, 38) = 0.19, p=0.66	F(1, 38) = 0.55, p=0.46	F(1, 38) = 0.02, p=0.88	F(1, 38) = 1.43, p=0.23
Iron	F(1, 38) = 0.68, p=0.41	F(1, 38) = 0.08, p=0.78	F(1, 38) = 0.76, p=0.38	F(1, 38) = 0.01, p=0.91	F(1, 38) = 0.25, p=0.62	F(1, 38) = 0.01, p=0.93
Lead	F(1, 38) = 0.10, p=0.75	F(1, 38) = 0.01, p=0.94	F(1, 38) = 0.13, p=0.72	F(1, 38) = 0.03, p=0.87	F(1, 38) = 0.36, p=0.55	F(1, 38) = 0.04, p=0.85
Magnesium	F(1, 38) = 0.17, p=0.68	F(1, 38) = 0.02, p=0.88	F(1, 38) = 0.18, p=0.67	F(1, 38) = 4.59, p=0.03	F(1, 38) = 1.52, p=0.22	F(1, 38) = 0.12, p=0.72
Nickel	F(1, 38) = 0.00, p=0.96	F(1, 38) = 0.01, p=0.92	F(1, 38) = 0.00, p=0.97	F(1, 38) = 3.98, p=0.05	F(1, 38) = 11.07, p=0.00	F(1, 38) = 15.13, p=0.00
Phosphorus	F(1, 38) = 0.54, p=0.46	F(1, 38) = 0.11, p=0.74	F(1, 38) = 0.60, p=0.44	F(1, 38) = 0.83, p=0.36	F(1, 38) = 0.74, p=0.39	F(1, 38) = 0.07, p=0.79
Potassium	F(1, 38) = 0.15, p=0.70	F(1, 38) = 0.47, p=0.49	F(1, 38) = 0.13, p=0.72	F(1, 38) = 1.90, p=0.17	F(1, 38) = 0.14, p=0.71	F(1, 38) = 0.48, p=0.49
Selenium	F(1, 38) = 0.83, p=0.36	F(1, 38) = 0.03, p=0.85	F(1, 38) = 0.99, p=0.32	F(1, 38) = 3.00, p=0.08	F(1, 38) = 4.74, p=0.03	F(1, 38) = 5.25, p=0.02
Zinc	F(1, 38) = 0.05, p=0.83	F(1, 38) = 0.32, p=0.57	F(1, 38) = 0.09, p=0.76	F(1, 38) = 0.00, p=0.96	F(1, 38) = 0.05, p=0.83	F(1, 38) = 0.65, p=0.42

Table 5: The effect of soil characteristics on indicators of wildflower plant health, including biomass of the plant, chlorophyll and photochemical efficiency, and mycorrhizal colonisation percentage of roots. Colours of cells indicate positive response (green), negative response (red) and non-significant effects (white) tested using linear models

Discussion

Effective and sustainable greenspace planning is integral to future UK urban planning, but it is vital that the implications and effectiveness of change in greenspace type are understood. Allowing for continued urbanisation whilst maximising ecosystem benefits of greenspaces has been a concern following historic policy changes to promote greenspaces (Boulton et al., 2018; Tallon, 2020; Chan et al., 2022). Here, we present an example of the effects of greenspace type on the health and growth of various plant species, demonstrating the potential of different greenspaces to support vegetation. Our findings highlight the suitability of parklands for promoting plant growth, while also revealing notable similarities in plant health between those grown in garden and woodland soils.

A key consideration in urban planning is vegetation growth and health within urban greenspaces. Our investigation demonstrated significant differences in plant growth within soils from different urban greenspace types, with park grass soils generating lower biomass than other greenspace types. This was true for both total and above ground biomass, while below ground biomass showed the same for three study species. Park grass and garden harvest biomass was lower for all food crops. Furthermore, measurements of photosynthetic efficiency also indicate that plants in park grass soils were the most stressed. Park grass soils are managed as amenity grassland, and therefore often receive regular inputs, such as fertilisers or herbicides (Aronson et al., 2017; Garbuzov, et al., 2015). The alternate management techniques of woodland sites may encourage conditions that promote nutrient availability, soil structure, and biological activity. Woodland spaces see limited management, centred around the safety of public spaces, instead of aesthetic purposes (Van Der Jagt et al., 2019; Meyer et al., 2020). This may promote growth of both tree and understory species. In woodland soils, the accumulation of organic matter in the form of leaf litter and fallen branches enriches the soil through provision of vital nutrients (Meyer et al., 2020). This influx of organic matter can also improve the structure of the soil, increasing the texture, water infiltration, and aeration, crucial to soil invertebrate communities and plant development. Through the soil 'sponge effect', higher density of organic matter, tree roots, and reduced disturbance allow woodland soils to hold more water (Nisbet and Thomas, 2006, Nisbet et al., 2011). In a similar way, garden and allotment soils receive high levels of organic inputs in the form of compost, mulch, and fertilisers, but garden soils they see little removal of these nutrients through cropping, whilst allotment crops are regularly removed, reducing the organic component of the site (Tresch et al., 2018; Tresch et al., 2019). However, herbicide and pesticide applications are common in gardens and allotments, as is the use of synthetic fertiliser, which can disrupt soil health and microbial communities, reducing plant health and growth, Management of these soil inputs and direct human influence on urban greenspaces could be key in successful plant growth in urban greenspaces.

While variation in plant growth can often be attributed to abundance of beneficial soil microbes such as AMF, our research indicated that arbuscular mycorrhizal fungi colonisation showed no significant effect of greenspace type on colonisation among three plant species, with wildflower being the only species to demonstrate lower colonisation rates, seen in woodland areas. This reduced colonisation in woodland soils may be due to higher organic matter content or competition from other microbial communities that suppress AMF activity, as well as potential differences in soil pH, moisture, or light availability that can influence AMF-host interactions.

Although there was some variation in the most productive greenspace soils across species and plant sections, greater plant biomass was most consistently found in woodland and garden soils. Though our investigation was novel in the comparison of soils from greenspace types and their effects on plant growth, research in adjacent fields has demonstrated that the way small scale gardeners manage their plots is an important driver of soil properties (Tresch et al., 2018; Tresch et al., 2019; Schram-Bijkerk et al., 2018). One driver of mycorrhizal colonisation is the increased biological complexity of both greenspace types. Woodland soils see larger vegetation structure and diversity, leading to deeper root penetration and water infiltration potential (Roman et al., 2018; Roman et al., 2021; Morgenroth et al., 2016). Additionally, within woodland soils, organic matter such as leaf litter is left to decompose naturally, whereas in park grass soils, grass clippings are often moved or removed. In gardens and allotments, organic matter such as compost or manure is frequently added to improve soil fertility, although they can also see the removal of organic matter such as plant debris to maintain a neat appearance. This process of organic matter deposition and decomposition may impact the structure of the soil surface, reducing plant health in park grass soils.

We demonstrated that heavy metal levels and soil health measures were lowest in woodland soils, and highest in allotment and garden soils, whilst park grass soils were not significantly different to other greenspace types for all heavy metals besides Ca (Chapter 2). The lower levels of heavy metals in woodland soils could be a driver of increased plant biomass. High concentrations of heavy metals within soil can inhibit root growth, reduce chlorophyll content, and decrease photosynthetic efficiency of plants (Asati et al., 2016). Some metals, such as cadmium, interfere with metabolic pathways involved in respiration and photosynthesis (Dobrikova et al., 2019; Ivanov et al., 2020). Zinc contaminated soils can also impair reproductive development of plants

(Noulas et al., 2018), which would be reflected in this experiment through lower harvest biomass, which we found in all three food crop species. Reduced biomass is also common in plants exposed to high levels of lead (Zulfiqar et al., 2019). The levels of heavy metals seen across greenspace types is associated with plant biomass in woodland soils, but does not entirely explain trends, with lower plant biomass in park grass soils, which did not display the highest levels of heavy metals in the soil. In Chapter 2, we showed lower pH and P levels within woodland soils. Acidic soils (typically pH below 5.5) can limit availability of essential nutrients like phosphorus to plants, causing nutrient deficiency that impair growth (Li et al., 2019; Goulding, 2016). Acidic soils can also increase the solubility of toxic metals (Kicińska et al., 2022). With lower pH in park grass and woodland sites, but lower levels of heavy metals in woodland sites, the interaction between these soil characteristics may have led to reduced plant biomass in park grass soils. The combination of a low soil pH and increased levels of heavy metals such as zinc and lead, plant growth could be inhibited.

Numerous plant health indicators and soil micronutrients did not have a clear difference between greenspace type and were not associated with plant growth. Numerous studies assessing urban soils have demonstrated high nutrient content within them. Vauramo and Setala, 2010, indicated that managed gardens demonstrated higher levels of organic N than surrounding agricultural fields, while Edmondson et al., 2014a showed that allotment soils had higher organic N levels than rural agricultural land. Our study demonstrated limited effects of soil nutrient levels on plant growth, while the consistent significance we did find within our study species was centred around bean species, widely recognised to be sensitive to changes in soil condition (Meng et al., 2021). Zinc is a crucial micronutrient for plant development (Hacisalihoglu et al., 2020; Castro- Guerrero et al., 2016; Vassilev et al., 2011). It performs an essential role in physiological processes in plants, such as protein synthesis and hormone regulation, contributing to healthy root development and chlorophyll formation (Barman et al., 2018; Cakmak et al., 2019). The differences in plant growth between different greenspace types may reflect changes in nutrient availability, demonstrated in bean growth, but which were not as clear in other, less sensitive plant species. This reveals that soil may be a driver of plant growth, highlighting that even within urban greenspaces, where soils are often assumed to be homogenised or degraded, differences in soil properties can lead to measurable variation in plant performance. This indicates the need for a more comprehensive understanding of drivers across greenspace uses. It is also likely that the positive effects on plant health across woodland and garden greenspace types is due to a complex matrix of factors, and interactive effects of key drivers. Soil composition and microbial communities can significantly influence microclimate conditions and moisture levels, which enhance nutrient cycling processes (Frindte et al., 2019; Borowik et al., 2016). The role of organic matter - whether sourced from decomposing leaf debris in woodlands or from compost in gardens - interacts with these factors to further improve soil fertility and structure (Gerke, 2022; Mayer et al., 2021). Additionally, management practices, such as mulching and pruning, contribute to these interactions by affecting how soil and organic matter dynamics influence plant health. This complex interplay among environmental conditions and management strategies suggests that multiple factors collaboratively shape the growth of plants, highlighting the need for a more nuanced understanding of how these interactions function across different greenspace types. Such understanding could inform more targeted urban land management practices, including plant selection, optimising irrigation or composting strategies, and designing multifunctional greenspaces that support both biodiversity and human use.

Conclusion

In conclusion, this research underscores the intricate interplay of factors that influence plant growth across various urban greenspace types. Our findings reveal significant differences in plant health indicators based on greenspace type and management practices, with woodland soils consistently supporting higher biomass across all species studied. The reduced management of these sites, such as lower compaction, limited removal of plant debris and leaf litter, and minimal introduction of artificial pesticides or herbicides. Conversely, park grass soils exhibited reduced biomass and heightened levels of plant stress, likely attributable to a combination of factors including less diverse plant communities, shallow root systems, and limited organic matter decomposition. As we move forward, it is essential to investigate the long-term ecological impacts of herbicide use and other management practices in urban greenspaces, with a focus on promoting ecosystem health and resilience. A deeper understanding of these dynamics will facilitate the development of strategies that enhance the sustainability and productivity of urban environments, ultimately contributing to the well-being of both plants and the communities they serve.

Chapter 4: Negative effects of landscape complexity on invertebrate communities in inner- city urban greenspaces

Abstract

Invertebrate diversity is declining globally, with the effect of urbanisation and land use change identified as key drivers. While inner-city urban greenspaces as a resource for invertebrates is increasingly appreciated, the features supporting many invertebrate taxa remain poorly understood. We assessed the impact of inner-city urban greenspace surroundings and land cover on both above and below ground invertebrate populations. There were consistent negative effects of greater greenspace area and the proportion of greyspace surrounding greenspaces on invertebrate communities, suggesting that smaller, less isolated sites can benefit invertebrate communities in inner-city greenspaces. The effect of land cover type (tree, grass, flowering border, and shrub) within the greenspace differed among taxa, demonstrating the complexity and habitat preferences of different aspects of the invertebrate community. My results suggest that land cover and connectivity of small scale, inner-city sites are relevant in supporting invertebrate communities, and managing these sites as a biological resource will support the ecosystem services they provide.

Introduction

Driven by population growth and increasing urbanisation, the UK is experiencing pressure to modify and expand urban landscapes to suit the changing needs of the UK population (Theodorou et al., 2022). An increased requirement for urban land is leading to an increase in greyspace proportions and density within cities (Robinson et al., 2022: Imbrenda et al., 2022). Government policies and market demand have prioritised the development of new residential areas, often within existing urban boundaries, to limit urban sprawl and make efficient use of available land (Nam, 2021; Bishop et al., 2020). This conflicts with the protection of existing urban greenspaces, which see removal, reduction, or further fragmentation (Dobson et al., 2020; Colding et al., 2020; Kondo et al, 2018).

The drive for densification has resulted in changes to existing urban greenspaces. Here, an urban greenspace refers to an area within an urban environment which is primarily covered with vegetation, such as parks, gardens, grass verges and SuDS systems. Urban greenspaces provide numerous ecological, social, and health benefits, making them critical components of sustainable cities (Kondo et al., 2018: WHO., 2017). Ecologically, they serve as vital refuges for biodiversity, supporting a wide range of species, including pollinators, birds, and invertebrates, which contribute to essential ecosystem services such as pollination, pest control, and nutrient cycling (Castelli et al., 2021). Greenspaces also play a key role in mitigating urban environmental challenges by improving air quality, reducing the urban heat island effect, and managing stormwater (Kondo et al., 2018). Socially, they offer recreational opportunities, enhance mental and physical well-being, and promote community cohesion (WHO, 2017). Additionally, greenspaces have been shown to increase property values and contribute to the aesthetic quality of urban environments, making cities more liveable (Chen et al., 2022: Trojanek, et al., 2018). As cities densify, balancing the need for sufficient housing with the preservation of natural habitats becomes a key challenge for planners, as maintaining and expanding urban greenspaces is crucial for preserving these ecological functions and improving the quality of life for residents.

The term ‘inner-city greenspaces’ refers to small-scale, often fragmented, green areas integrated within densely built urban environments. There are a variety of site ages, with both historic examples and an increasing trend to retrofit small greenspaces into city centres (Buckland et al., 2023; Dromgold et al., 2020). These include environments such as roadside verges (narrow belts of predominantly grassy vegetation along streets), pocket parks (small areas of public green space, often within previously underutilised land), Sustainable Drainage Systems (SuDS) (areas of permeable ground with vegetation, designed to capture water runoff), and green roofs (vegetation installed on rooftops to provide greenery)(Speak et al., 2013; Berthon et al., 2021). While limited in size, a growing body of evidence highlights that such spaces can offer meaningful ecological and social value (Hunter et al., 2019; Baldock et al., 2015). Even small greenspaces have been shown to support considerable biodiversity, particularly for invertebrates and pollinators, due to floral diversity and reduced human disturbance (Baldock et al., 2015; Hall et al., 2017). They also contribute to urban climate regulation, stormwater management, and air quality improvement, especially when designed as part of multifunctional infrastructure (Speak et al., 2013; Giese et al., 2020). Importantly, these greenspaces offer accessible natural experiences to local residents - an increasingly important function in densely populated areas - linked to mental well-being and social cohesion (Hunter et al., 2019; WHO, 2017). Despite their small size and fragmentation, they have the potential to enhance ecosystem services and serve as stepping-stones for urban biodiversity

(Roman et al., 2018). However, this variation in site size and isolation means that the ecological impact of individual sites may vary, raising questions regarding the consistency and scalability of their contributions (Suligowski et al., 2021; Vanbergen et al., 2017). Nonetheless, their incorporation within inner-city areas is a critical strategy to mitigate the effects of environmental change and biodiversity loss into the future.

Due to the density of urban areas, the surroundings of inner-city greenspaces are typically composed of denser greyspace and infrastructure than suburban spaces (Buckland et al., 2023). Spaces are often bordered by multi-story buildings instead of smaller residences, which create a larger vertical barrier between greenspaces (Dromgold, et al., 2020). These high structural barriers can disrupt ecological connectivity by limiting the movement of birds, insects, and other mobile species between greenspaces (Von Thadden et al., 2021). Additionally, impermeable services of the roads, pavements, and drainage systems are denser within inner-city urban spaces, leading to increased water runoff and pollution within these spaces (Radford et al., 2013). Proximity to heavier traffic, industrial fumes and anthropogenic structures can cause microclimatic variation, with increased heat and exhaust fumes emitted from these sources (Jaung et al., 2020). Furthermore, construction of larger building complexes and their subsequent foundations, alongside the below-ground infrastructure necessary to support dense urban spaces, has potential to affect soil characteristics and below-ground connectivity (Dromgold et al., 2020).

Management of inner-city greenspaces has shifted over time to reflect changes in urban planning. Historically, many greenspaces were constructed as an effort towards urban beautification and human health during previous periods of rapid industrialisation (Reeves., 2000). Verges were created alongside roads for safety and visual appeal, while pocket parks emerged in response to the social demand for recreational land in dense urban centres (Phillips et al., 2021). In recent decades, management has shifted towards environmental benefits and potential of these spaces, with losses in biodiversity and the climate crisis brought to public awareness (Derkzen at al., 2017). Recent research underscores the urgency of optimising these habitats, including the formation of SuDS systems and sustainable planting. The UK is witnessing a decline in wildlife populations, as seen in the 19% reduction in UK wildlife abundance since 1970 (Joint Nature Conservation Committee., 2023). Rewilding projects are working to reverse trends by restoring greenspaces with the focus of specific species habitats, such as water voles in London (Hill et al., 2022: Thomas, 2022). Additionally, small scale rewilding schemes, such as ‘no-mow’ schemes, have been incorporated into planning across the UK, with reduction in management such as mowing and chemical treatment designed to allow native plant growth (Wynne-Jones et al., 2020). The new focus on biodiversity, ecosystem services, and environmental resilience marks a shift from traditional urban planning practices, but it is vital to understand the wider ecological implications as the transition continues.

Urban greening centres on enhancing biodiversity, improving environmental quality and community wellbeing. Projects typically include the creation and revitalisation of small areas of greenspace, including small parks, green corridors, grass verges and pathways, and wildflower meadows (Berthon et al., 2021). However, projects can be hindered by the breadth of research available to support planning. Many projects focus on the creation of wildlife corridors, a practice demonstrated to be beneficial to invertebrates in a wide variety of spaces, and to urban mammals (Zellmer, 2022). However, the issue of invertebrate movement across urban spaces has historically focused on pollinator species, leaving a gap in knowledge regarding the benefits to the wider invertebrate community (Kirk et al., 2023). Through not supporting the invertebrate diversity in its entirety, the benefits to pollinators may be undermined by the detriment to the other invertebrates they interact with. Additionally, many projects incorporate the use of planted vegetation species to provide flowering resources, such as wildflower meadows, with the understanding that all flowering resources will be beneficial to pollinators (Berthon et al., 2021). However, these projects can be undertaken without a full understanding of the species which would most benefit key pollinator groups, which specific pollinators they would benefit, , and the ability of those pollinators to travel to and interact with the resources. Planning in both current and future projects are hindered by a lack of specific research into urban environments and the ways that small greenspaces can benefit the invertebrate community in its entirety. Including findings from research into invertebrate communities and urban planning can lead to more robust urban greenspaces which provide a wide range of benefits to the surrounding community.

Invertebrates, which make up over 95% of animal diversity, play vital roles in ecological processes, such as nutrient cycling, soil aeration, and decomposition (Eisenhauer et al., 2021). The variety of species fulfil a plethora of environmental services and perform numerous ecological functions. Urban invertebrates play a critical role in maintaining the health and functionality of vulnerable ecosystems (Jones et al., 2013). Many invertebrates have a high reproductive rate and so can adjust quickly to urban areas, including species such as the common earthworm **(*Lumbricus terrestris*),** which reproduces rapidly and can re-establish populations after soil disruptions. This adaptability helps maintain their role in the wider ecosystem even in anthropogenic environments. Many species assist in the breakdown of organic matter, enhancing soil fertility and promoting plant productivity (Eisenhauer et al., 2023). Other invertebrates are key pollinators, supporting plant biodiversity and ecosystem resilience. Invertebrates serve as essential prey within the wider food web, underpinning food security for numerous vertebrate species. Furthermore, some invertebrates, including Collembola species and certain earthworm species, like *Aporrectodea caliginosa*, can act as bioindicators, allowing insight into environmental change, ecosystem health, and pollution (Prather et al., 2013; Eisenhauer et al., 2021; Eisenhauer et al., 2023). Prior studies into invertebrate interactions, such as pollination networks and predator-prey dynamics, also demonstrate how these relationships can affect ecosystem functioning in urban spaces—for example, by supporting plant reproduction, regulating pest populations, and maintaining soil processes (Rocha et al., 2020). Through advancing our understanding of the invertebrate communities present within environments, it is possible to better inform planning strategies and environmental management practices.

Research has typically focused on known service providers, such as beetle and Collembola species (McNaughton et al., 2023; Qiao et al., 2022). The composition of the entire invertebrate community, including both above and below ground invertebrates, has not been widely researched (Baldock et al., 2015: Braman et al., 2015). Rapid urban expansion, along with habitat fragmentation, pesticide application and climate change have caused drastic reduction in global invertebrate abundance (Forister et al., 2019). The unique characteristics of urban spaces, such as air, light and soil pollution, heat island effects, and density of impermeable surfaces present a unique environment, and requiring investigation to fully understand their intricacies (Helden et al., 2018). Although these factors present unique challenges within each city, they are widely prevalent across urban areas globally, underscoring the importance of both site-specific and comparative research. Vegetation characteristics and composition directly influence invertebrate abundance and species richness by shaping the availability of critical resources such as food, habitat, and microclimates. The structural complexity of vegetation, including factors such as plant height, leaf density, and canopy cover, creates various niches that invertebrates utilise for shelter and reproduction (Kotze et al., 2022). Plants with diverse morphological traits offer a wider range of physical environments, supporting higher invertebrate diversity. Additionally, plant composition affects resource availability, as different species produce varying types and quantities of resources for invertebrates, such as foliage, pollen, or organic matter, which sustain different taxa (Luiza-Silvia et al., 2020).

Vegetation also regulates microclimatic conditions, such as temperature and humidity, which are crucial for invertebrate physiology and survival. Therefore, areas with diverse plant species and complex vegetation structure tend to support greater invertebrate abundance and species richness due to the range of ecological opportunities they provide. According to metapopulation and landscape ecology theory, these patches function as interconnected habitats where the balance between connectivity and isolation determines species movement and gene flow. Connectivity through corridors facilitates dispersal and recolonisation, which enhances population stability and diversity. Conversely, isolated patches may suffer from local extinctions due to limited immigration. Edge effects further modify these dynamics by altering habitat conditions at patch boundaries, influencing species interactions and microclimates. Together, these processes shape the structure and resilience of urban invertebrate communities.

Characteristics of greenspaces, including habitat connectivity, fragmentation, and site area, are critical for invertebrate abundance and species richness in urban ecosystems. Habitat connectivity allows for the movement and dispersal of invertebrate populations between greenspaces, facilitating gene flow and reducing the risk of local extinctions (Gibb et al., 2002). Fragmentation, on the other hand, can isolate populations. Isolated sites are less likely to receive species from surrounding areas, limiting gene flow and species dispersal, which can reduce both species richness and population stability (Piano et al., 2020). For example, urban parks that are connected by green corridors enable populations to migrate and access different habitats, enhancing biodiversity (Kotze et al., 2022) Conversely, fragmented landscapes with limited connectivity can hinder movement, resulting in reduced species richness (Christie et al., 2010). Edge effects, which occur at the boundary between different habitat types, can also either positively or negatively impact invertebrate populations. Increased edge exposure may lead to higher predation rates, more frequent disturbances, and changes in microclimatic conditions, potentially reducing invertebrate diversity. However, edges can also create habitat variety, attracting species that thrive in transitional zones. Therefore, understanding the dynamics of sites and their surroundings is essential for effective planning aimed at conserving invertebrate diversity and promoting resilient urban ecosystems.

The complexity of vegetation structure can promote higher species richness by offering niches that support both specialist and generalist invertebrate species (Ellis et al., 2021; Ozaki et al., 2022). In contrast, simplified vegetation composition may limit resource availability and habitat options, leading to reduced invertebrate diversity and abundance. For example, a park with a mix of grassy areas, flower beds, and scattered trees creates a mosaic of habitats that cater to different invertebrate species (Savage et al, 2015). Additionally, the presence of dead wood or leaf litter within these habitats provides crucial resources for detritivores and decomposers, enhancing nutrient cycling. However, there are also documented issues with site heterogeneity, particularly in small sites (Deak et al., 2021). Different types of vegetation or habitats may provide conflicting services, and the limit of site area could reduce the use of the site overall (Tamme et al., 2010). For instance, Collembola species may be better suited to wooded areas, with regular organic matter inputs available. A greenspace which was entirely tree cover would not further fragment or reduce the size of tree habitat within the site. By examining invertebrate populations, we can develop strategies to enhance habitat connectivity, mitigate the effects of fragmentation, and promote sustainable planning.

Here, using Sheffield as a case-study, we assessed the invertebrate communities within inner-city greenspaces. We investigated the effects that land cover,its heterogeneity within greenspaces, and the proportion of greyspace surrounding these greenspaces have on these communities . We undertook a multifaceted invertebrate sampling scheme to examine the invertebrate communities present and the relationships with site size, surroundings, and vegetation. This work has been used to assess drivers of invertebrate community size and structure within inner-city greenspaces.

We began this research with three key research questions:

- How does greenspace size influence the abundance and diversity of invertebrate communities in inner-city greenspaces?
- How does the vegetation composition and land cover within greenspaces affect

invertebrate communities?

- How does the proportion of surrounding greyspace (site isolation) influence invertebrate abundance and diversity?

We tested the hypothesis that greenspace size and then heterogeneity of land covers within the greenspace would benefit invertebrate communities by increasing resources and mitigating the negative effects of site isolation. We expected that site isolation would have a negative effect on abundance and diversity, while that the presence of vegetation known to attract invertebrate species, such as flowering stems, would increase the abundance and diversity of invertebrates within the site. We also expected that the heterogeneity of vegetation of the site would have a positive effect on above-ground invertebrates, but a negative effect on below-ground invertebrates due to ease of dispersal, alongside differences in taxa and functional groups.

Methodology

Study system

For this study, Sheffield was selected as a focal city. The city is a good representation of UK city due to the size and greenspace proportion and has a wide background of urban greenspace research for reference (Ellis et al., 2023; Grafius et al., 2019).

To identify the city centre, an urbanisation gradient analysis of the OS MasterMap data was performed. An urbanisation gradient analysis based on OS MasterMap data offers detailed and extensive spatial information to objectively identify the city centre by measuring urban intensity. However, this method may miss socio-economic factors and might not always reflect recent urban developments A central point within the city centre was found using the 'Heatmap plugin' to assess impervious surface density and find the area surrounded by the highest proportion of greyspace. This point was taken as the centre of urban density and the city centre of Sheffield. From this point, a circular buffer of 2 km in diameter was then created. As the study focus was on inner- city urban spaces, this buffer size was selected due to the decrease in urbanisation and the increase in larger greenspace and residential areas outside of this radius. Within this buffer radius, Google Satellite imagery XYZ tiles were then used within QGIS to map each greenspace patch over 1 m^2 within the city centre. The entirety of the 2km buffer was later assessed in person to verify the accuracy of this satellite imagery. Any additional sites which were not originally captured, such as newly constructed SuDS systems, were mapped within the 2 km city centre and added to the assessment layer. The characteristics of the sites were measured, including the area of each of the sites and the distance to other greenspace patches. To ensure each site could be considered as independent, we merged any greenspaces with less than 6 m between one another. This 6 m distance was selected as it matched the size of a standard two- lane road, the most common size within the study area. The range of areas within all greenspace sites were then assessed and were found to be normally distributed. Sites with an area between the 25th and the 75th percentile (100 m^2-492 m^2) were selected to represent the inner-city urban greenspaces present. This gave a total of 60 appropriate sites within the study area, of which 57 were sampled from successfully, and 3 of which were excluded throughout sampling due to human interference and disturbance (Figure 4.1). The smallest site was 105.97m^2 and the largest site was 489.62m^2. The mean site area used for the study was 260.54m^2 (N=57, SD = 113.31).


● City Centre
🟪 Greenspace sites
☐ 1000m buffer
0
250
500 m

Figure 4.1. Sampling sites for inner city greenspace sites within study city Sheffield. Pink polygons show the outlines of each sample site. The centre of the city is shown in black. The circular border represents the 1km radius surrounding the city centre, using which the inner city was identified and sites were selected.

Surrounding landscape characteristics

The greyspace surrounding each site was used to investigate the effects of site surroundings and the influence on greenspace. For each site, the surroundings were captured using the 'buffer' tool in QGIS at size 25m, 50m, 75m, and 100m. Within each of these buffers, the greyspace present was mapped using the OS MasterMap layer and the previously mapped greenspaces. OS MasterMap was preferred over traditional land cover maps due to observed inaccuracies with land-cover maps for the study site and the scale of data available. The proportion of greyspace surrounding each site at different radii was then calculated. There was no significant difference between greyspace at 25m (mean = 73.149, SD = 13.573), 50m (mean = 70.603, SD = 12.489), 75m (mean = 68.160, SD = 12.004) and 100m (mean = 67.293, SD = 11.729).

Vegetation surveys

A detailed vegetation survey was undertaken at each site. The perimeter of the site was mapped using GIS, verified with in-person measurements and GPS coordinates.

Tree species were identified, with the diameter at breast height (dbh) and height of tree measured through trigonometry. Shrub and flowering stem species was identified where possible, with recognition of the number of non-native species present across the city planting schemes. The number of flowering stems was counted where present. Management practices (e.g. recent mowing, tilling of soil, and trimming of shrubs) was noted. Condition of grass (e.g. no-mow sites) was recorded.

Land cover was then mapped at each site. Five key land cover categories were observed, with the following classifications used to outline each land cover type observed at the site.

Tree; any area with, or under the canopy of, a tree with dbh >10cm

Grass; any area with mown or unmown grass

Shrub; any area with shrub or woody stems

Border: any area covered with planted beds, predominantly flowerbeds

Grey: any area covered with impervious cover, predominantly paving or tarma

For each of these land covers, the area and shape of cover was measured and calculated. This was used to calculate the proportional cover within each site and contributing to the heterogeneity indexes of the site. The heterogeneity of sites was assessed using both a Shannon's evenness index (HGI Shannon) and a Simpson index (HGI Simpson). A value for each site was calculated using the following formulas:

Shannon's index Simpson's Reciprocal Index

$$H' = -\sum_{i=1}^{S} p_i \ln(p_i)$$

$$D = \sum_{i=1}^{S} \frac{n_i(n_i - 1)}{N(N - 1)}$$

H is the Shannon's Diversity Index D is Simpson's Diversity Index.

S is the total number of land cover types within the study n_i is the area of land cover *i* within the site

N is the total area of all land covers at the site pi is the relative abundance of each land cover ln is the natural logarithm.

In addition to this, more detailed site characteristics were measured. The diameter at breast height (dbh) and height were used to calculate biomass for each tree present within the site. The species of shrub and flowering stems present was also recorded. The number of flowering stems was also recorded for each site.

Invertebrate sampling

We sampled invertebrates over a two-week period (09/08/2023-25/08/2023). This sampling period was chosen to capture invertebrate activity during peak late summer phenology, ensuring representation of species most active during this seasonal window. We assessed invertebrate populations using a combination of invertebrate monitoring methods to minimise the potential biases and limitations of each method, with the intention that a multi-faceted sampling approach would allow samples to accurately represent the invertebrate communities at each site. Five invertebrate sampling techniques were selected to capture the full range of invertebrate diversity across the sites. These consisted of three techniques to sample flying invertebrates (pan traps, sticky traps and sweep net sampling), one method to capture ground invertebrates (pitfall traps) and one method to assess soil invertebrates (Tullgren funnel extraction). The 60 sites were separated into 8 groups, with an even spread across the city urban space. Sites within the same block were visited and had traps placed on the same day to minimise the effect of weather variation on trap placement.

Pan traps: Traps were placed with signage on-site informing passersby about the research and requesting that traps not be disturbed. Two pan trap sets were placed randomly within each study site. Where possible, traps were partially concealed within vegetation or shielded using mesh covers to deter interference by birds or small mammals. Each set of traps contained three pan traps (volume 150ml), with one blue, one yellow, and one white trap, as is standard practice for reducing invertebrate preference effects. Each pan was filled 100ml of soapy water and secured on site for 7 days. Traps were checked every 3 to 4 days to ensure liquid had not evaporated or become saturated, with liquid replaced if necessary. During each visit, traps were also checked for signs of disturbance (e.g., displacement, damage), and any compromised data points were recorded and excluded from analysis. Overall, incidents of interference were minimal and did not significantly impact the results.

Sweep net sampling: Ten-minute transact sampling with a sweep net was carried out at each site. Transacts were carried out using a W shaped pathway due to irregular shape of many sites, making a central path difficult to consistently perform. All insects observed within this ten- minute period were captured in a sweep net, euthanised on site, preserved in ethanol and later identified to taxon level. Further 10-minute focused sampling was undertaken in visibly populated areas of the site which were not on the transact path to fully assess the site biodiversity. All insects were identified to the lowest taxonomic rank possible through visual identification.

Sticky trap sampling: One sticky trap was placed in the approximate centre of each site. Each sticky trap was a standard 10cm x 25cm in size, and both sides were uncovered. The traps were attached to a metal pole driven into the ground, placing them at a height of approximately 90cm from the ground to the top of the trap. This trap was left in place for one week, with each site checked at 3-4 days and the trap replaced if it were full or close to full. Insects were recorded to the lowest taxonomic level (any Hymenoptera, Coleoptera and Lepidoptera species) or counted (all other species).

Tullgren funnels: Soil samples were taken using a Eijkelkamp closed ring soil corer (model C53) using a closed sample ring holder. Each soil core was a volume 100 ml (maximum volume deviation: 0,5%), with a diameter of 53mm and a length of 5mm, taken at a depth of 5cm-10cm. Two soil samples were taken approximately 1m from one another from the centre of each land cover category (tree, grass, shrub, border) present within the site. The two samples were combined and placed in a standard Tullgren funnel system for a period of 96 hours. Invertebrate samples were processed through the funnel system into 70% ethanol solution, within which they were stored for identification. Species were identified to the lowest possible taxonomic level (Collembola) and to the class (all other individuals).

Pitfall traps: Each trap (volume 150 ml) was placed in the same spot soil cores were taken from. These traps were transparent to minimise the effects of trap colour on insect preferences. Traps were placed level to the ground to prevent an additional barrier to capture. We filled the pitfall traps with 100ml of soapy water and secured them on site for 7 days. Traps were checked on the 3rd or 4th day to ensure they had an appropriate level of water within them and replaced if necessary.

Statistical modelling

All analyses were done in R version 4.12 (R Core Team, 2022).

To test the effect of land cover on invertebrate populations, we modelled the effect of the key site characteristics described previously (site area, site heterogeneity, land cover types, and greyspace surrounding the site) on above-ground and below-ground invertebrate abundance and invertebrate diversity. Covariates included different land cover categories (tree, grass, shrub, border, and grey), area of the site, biomass of trees within the site, and proportion of greyspace (multiple models tested substituting the four buffer sizes) surrounding the site. We used negative binomial regression models due to overdispersion in the count response variabls, which violated the equidispersion assumption of Poisson regression. To identify key predictors, stepwise model selection was performed in both directions (forward and backward), using p-values to retain statistically significant terms. Predictor variables were mean centereed to reduce multicollinearity, especially in models including interaction terms. For the land cover data, which exhibited high collinearity (e.g., Grey cover area and Shrub area VIF > 200), Principal Component Analysis (PCA) was used to extract orthogonal components. The first six PCs (explaining >95% of variance) were used as predictors in an alternative model. Results from both models were reported as incidence rate ratios (IRRs), and approximate original variable effects were recovered from the PCA-based model using loadings and coefficient reconstruction. Invertebrate measures were then divided to explore effects on functional invertebrate groups (pollinator and decomposers), and four key taxa of invertebrates found across the sites (bee, wasp, fly and Collembola).

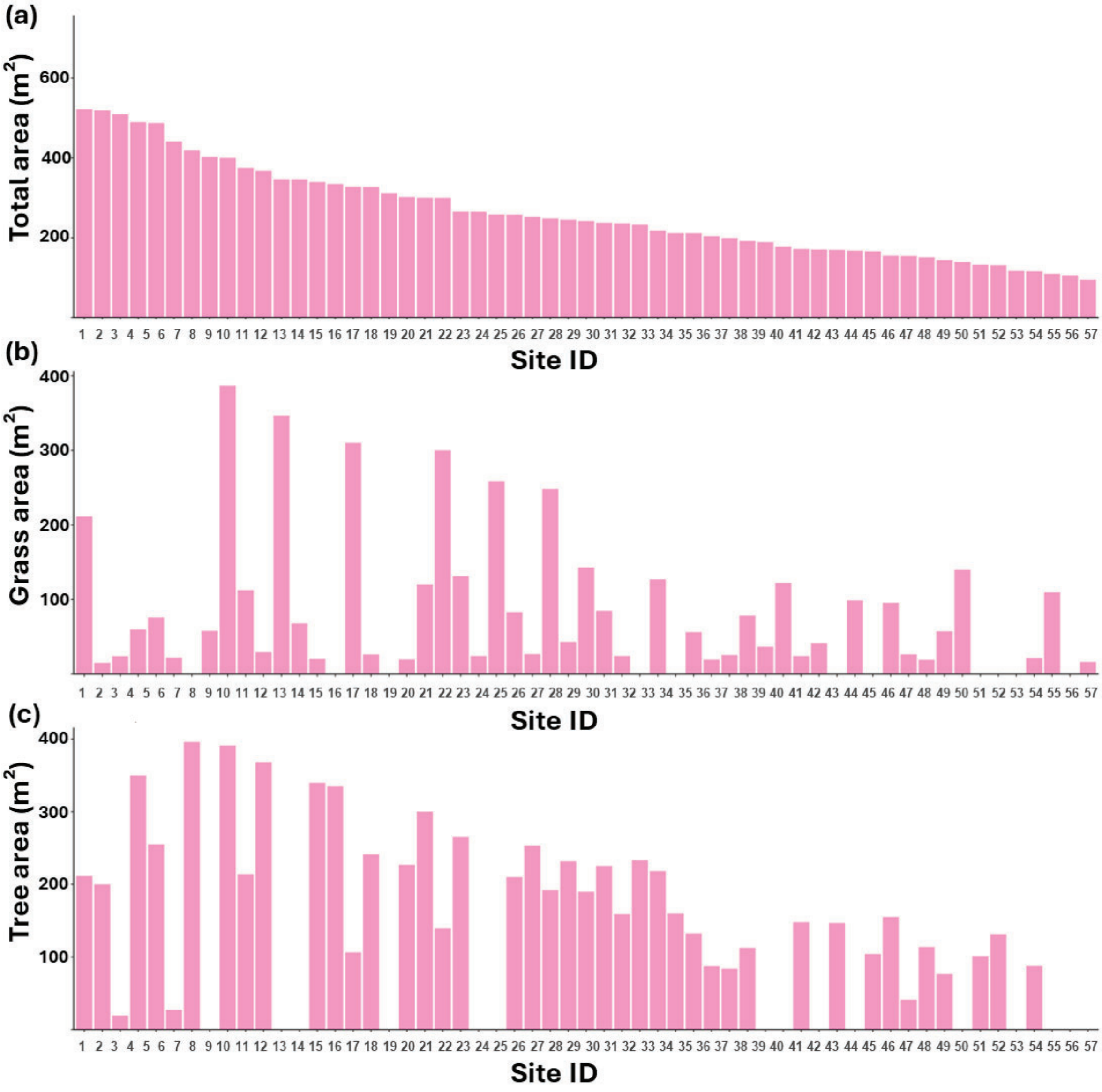
Following this, to test the effect of land heterogeneity on invertebrate populations, we calculated a Simpson’s and Shannon’s index approximate for the number of land cover categories and their proportional areas on the invertebrate abundance and invertebrate diversity. Glm models included multiple covariates, including one of the different land cover heterogeneity indexes (models run to compare both Shannon and Simpson index), area of the site, biomass of trees within the site, and proportion of greyspace (multiple models tested substituting the four buffer sizes) surrounding the site. Invertebrate measures were then divided to explore effects on functional invertebrate groups (pollinator and decomposers), and four key taxa of invertebrates found across the sites (bee, wasp, fly and Collembola).

Results

Greenspace land characteristics

Site land cover composition

The median tree cover across sites was 132m² (mean = 138 m²), amenity grass cover was 33m² (mean = 77m²), border was 0m² (mean = 23m²), shrub cover was 0m² (mean = 54m²) and greyspace cover was 0m² (mean = 19.71m²). Tree cover showed the highest relative proportion of land cover, covering over 50% of land at 28 sites. Tree cover was present at 41 sites, grass cover present at 45 sites, border cover present at 16 sites, shrub cover present at 24 sites, and grey cover present at 13 sites (Figure 4.2)


(a)
Total area (m^2)
600
400
200
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26 27 28 29 30 31 32 33 34 35 36 37 38 39 40 41 42 43 44 45 46 47 48 49 50 51 52 53 54 55 56 57
Site ID
(b)
Grass area (m^2)
400
300
200
100
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26 27 28 29 30 31 32 33 34 35 36 37 38 39 40 41 42 43 44 45 46 47 48 49 50 51 52 53 54 55 56 57
Site ID
(c)
Tree area (m^2)
400
300
200
100
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26 27 28 29 30 31 32 33 34 35 36 37 38 39 40 41 42 43 44 45 46 47 48 49 50 51 52 53 54 55 56 57
Site ID

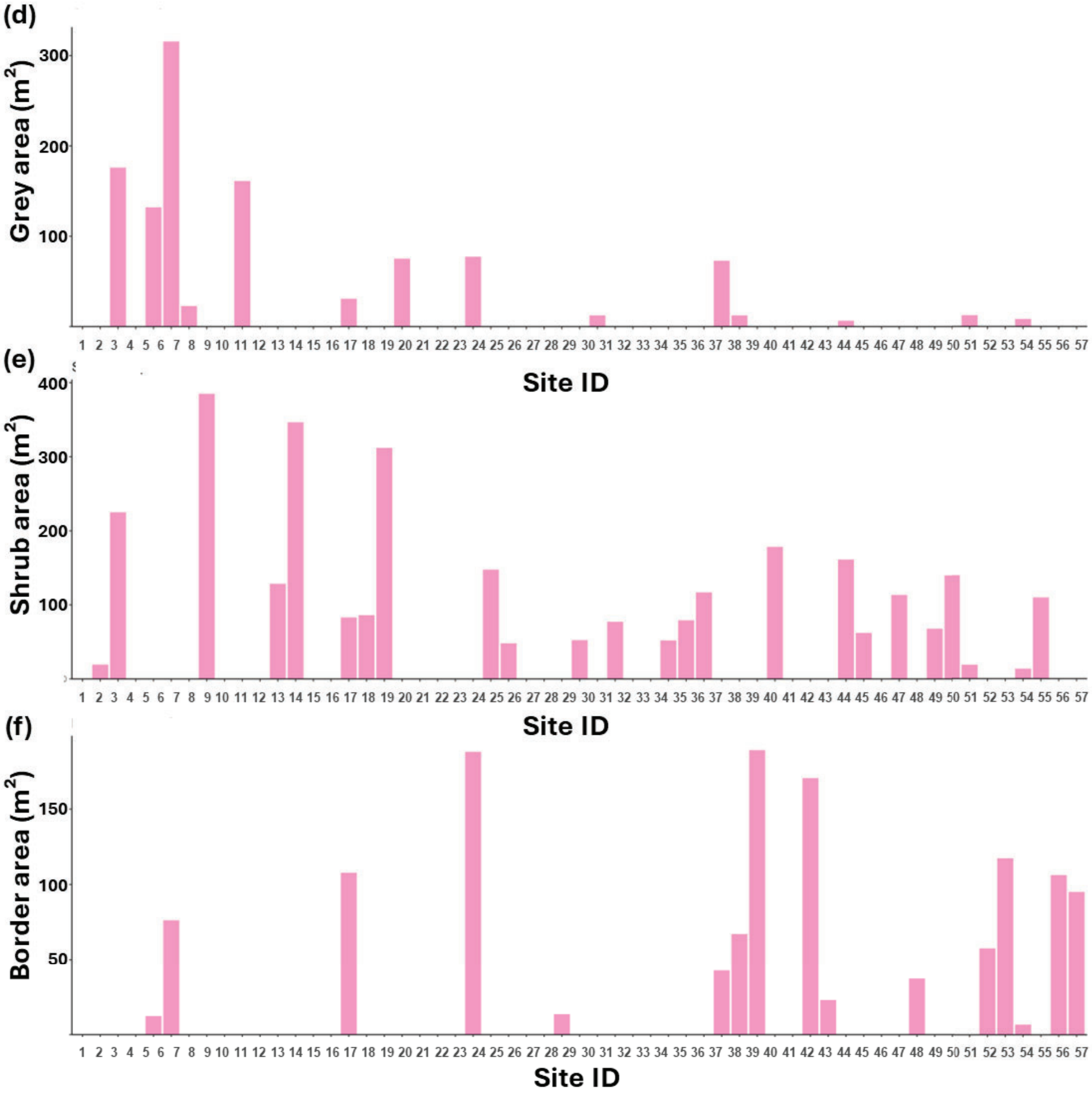


Figure 4.2. *Relative proportion of land cover types across 57 greenspace sites. Each panel shows thearea (m²) of a different land cover type: (a) total site area, (b) grass, (c) tree, (d) greyspace, (e) shrub, and (f) border*

The mean Shannon's value of site vegetation richness and evenness was 0.58 (SE = 0.04), while the average Simpson's value was 1.78 (SE = 0.08). There was a significant positive effect of amenity grass area on the subsequent Shannon (F (1, 55) = 7.194, p = 0.010) and Simpson's evenness value (F (1,55) = 5.485, p = 0.023). There was no significant relationship between the area of any other land cover and the measures of heterogeneity.

Above-ground invertebrates

A negative binomial regression model was used to assess the effect of vegetation heterogeneity and centred site characteristics on above-ground invertebrate abundance (Table 4.1). The model revealed a significant main effect of biomass (IRR = 0.996, SE = 0.001, $p < 0.001$), indicating that as biomass increases, above-ground invertebrate abundance slightly decreases. The interaction between biomass and Surrounding greyspace proportion was also significant (IRR = 1.000, SE < 0.001, $p = 0.012$), suggesting that in sites with greater buffering, the negative effect of biomass is moderated. Surrounding greyspace proportion alone did not have a significant effect (IRR = 0.996, $p = 0.590$). These findings suggest that although higher vegetation biomass may reduce above-ground invertebrate abundance, this effect is context-dependent and can be offset by the composition of the surrounding landscape.

Table 4.1. Generalised linear models (glm) fitted with negative binomial distribution for the effect of centred site characteristics and land cover on total above-ground invertebrate abundance. Includes testing of site area, proportion of greyspace surrounding the site (25m buffer greyspace%), land cover heterogeneity, and biomass of trees within the site.

Term	Estimate (β)	Std. Error	IRR (e^β)	IRR Lower 95% CI	IRR Upper 95% CI	p-value
(Intercept)	5.617	0.097	275.093	228.754	334.791	<0.001
Biomass	-0.004	0.001	0.996	0.994	0.998	<0.001
Surrounding greyspace proportion	-0.004	0.008	0.996	0.981	1.011	0.590
Biomass x Surrounding greyspace proportion	0.000	0.000	1.000	1.000	1.000	0.012

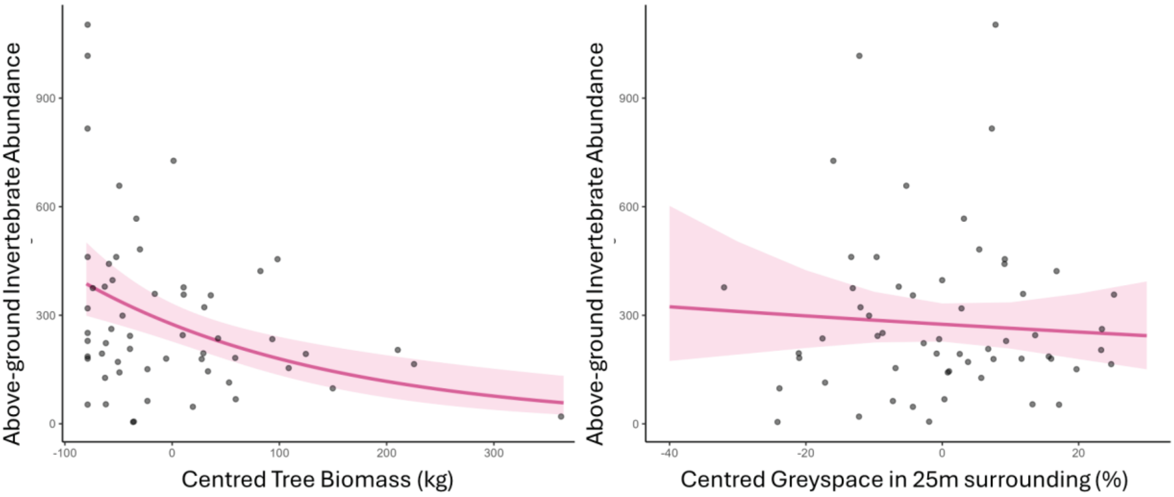


Figure 4.3. *Predicted above-ground invertebrate abundance in response to tree biomas and surrounding greyspace proportion.*

A negative binomial regression model was used to examine the effects of vegetation and landscape heterogeneity on above-ground species richness (Table 4.2). The model identified a significant interaction between area and surrounding greyspace proportion (IRR = 1.000, p = 0.001), indicating that above-ground species richness is influenced by the combined effect of site area and surrounding landscape configuration. However, neither area (IRR = 1.000, p = 0.594) nor surrounding greyspace proportion (IRR = 1.000, p = 0.874) individually showed significant effects on above-ground richness. These findings suggest that while single predictors alone may not be influential, their interactive effects can shape above-ground species richness patterns.

Table 4.2. Generalised linear model (glm) fitted with a negative binomial distribution for the effect of site characteristics and land cover on above-ground species richness. Includes site area and surrounding greyspace proportion (25 m).

Term	Estimate (β)	Std. Error	IRR (e^β)	95% CI (Lower)	95% CI (Upper)	p-value
(Intercept)	2.790	0.076	16.200	14.000	18.900	<0.001
Area (c)	0.000	0.001	1.000	0.999	1.000	0.594
Buffer (25 m) (c)	-0.001	0.006	0.999	0.988	1.010	0.874
Area × Buffer (25 m)	0.000	0.000	1.000	1.000	1.000	0.001

Observations: 57; Nagelkerke R² = 0.252

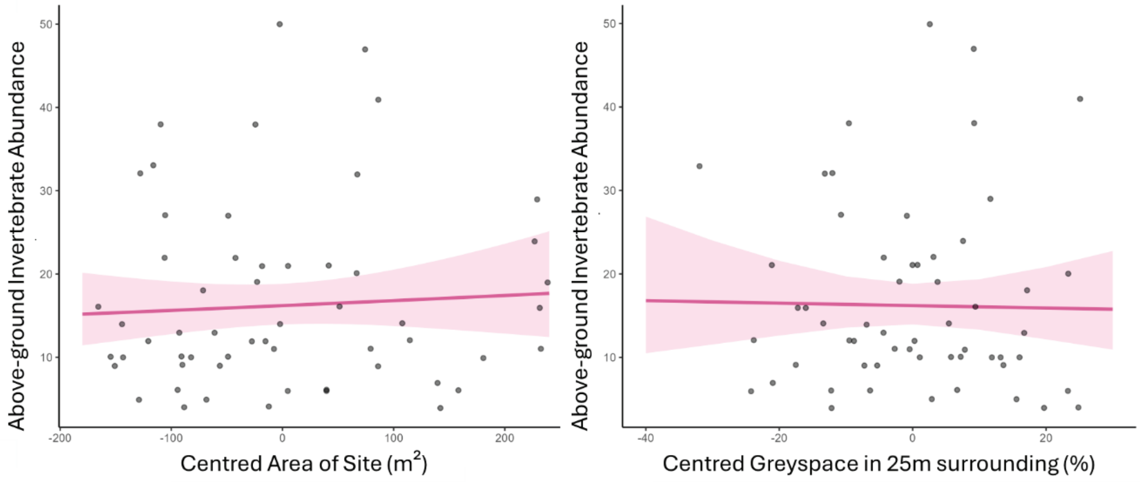


Figure 4.4. *Predicted above-ground species richness in response to site area and surrounding greyspace proportion.*

A negative binomial regression model was used to evaluate the effects of vegetation structural on above-ground invertebrate abundance (Table 4.3). The model revealed that PC1 (representing increasing vegetation density and structure) was positively associated with abundance (IRR = 1.267, SE = 0.064, p < 0.001), suggesting that more structurally complex vegetation supports greater above-ground invertebrate activity. PC4 also had a significant positive effect (IRR = 1.539, p < 0.001), indicating benefits of associated structural traits. Conversely, PC5 showed a negative effect on abundance (IRR = 0.667, p = 0.004), while PC6 had a strong positive effect (IRR = 3.298, p < 0.001), highlighting the importance of specific vertical or compositional elements of vegetation.

Table 4.3. Negative binomial generalized linear model (GLM) results showing the effect of principal components derived from site characteristics and land cover variables on total above-ground invertebrate abundance. Principal components were generated from original predictors including site area, proportion of greyspace (25m buffer), land cover types (Tree, Grass, Shrub, Grey, Border), and tree biomass to reduce multicollinearity.

Term	Estimate (β)	Std. Error	IRR (e^β)	IRR Lower 95% CI	IRR Upper 95% CI	p-value
(Intercept)	5.023	0.100	151.831	125.299	185.871	<0.001
PC1	0.237	0.064	1.267	1.110	1.450	<0.001
PC4	0.431	0.099	1.539	1.261	1.858	<0.001
PC5	-0.406	0.139	0.667	0.512	0.863	0.004
PC6	1.193	0.270	3.298	1.993	5.386	<0.001

Table 4.4 presents the estimated effects of individual environmental variables on above-ground invertebrate abundance, expressed as incidence rate ratios (IRRs). Site area, biomass, and grass cover

were negatively associated with invertebrate abundance (IRRs between 0.88 and 0.90), indicating modest reductions in abundance with increasing values. Tree cover also showed a slight negative effect (IRR = 0.96), while shrub and grey space had minimal influence (IRR = 0.99). Border and buffer variables showed no effect (IRR = 1.00), suggesting they do not significantly impact invertebrate counts.

Table 4.4. Approximate effects of original environmental variables on above-ground invertebrate abundance, based on loadings from principal component axes used in the negative binomial GLM. Effects are expressed as estimated coefficients and incidence rate ratios (IRR), indicating the direction and magnitude of influence for each variable.

Variable	Approximate Effect	IRR
Area	-0.11	0.90
Biomass	-0.10	0.90
Tree	-0.04	0.96
Shrub	-0.01	0.99
Grass	-0.12	0.88
Grey	-0.01	0.99
Border	0.00	1.00
Buffer (25m)	0.00	1.00

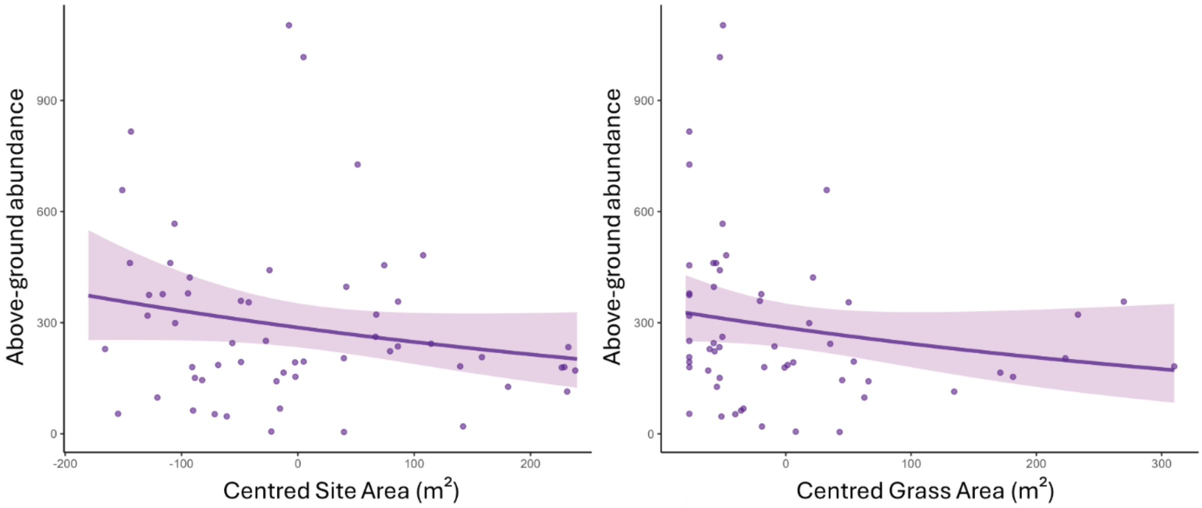


Figure 4.5. *Predicted above-ground invertebrate abundance in response to site area and grass area.*

A negative binomial regression model was used to examine the effect of vegetation structural characteristics (PC1–PC5) on above-ground species richness (Table 4.5). Among the predictors, PC1 had a significant positive effect (IRR = 1.170, p = 0.023), suggesting that structural features captured by PC1 are associated with higher above-ground species richness. PC2, PC3, PC4, and PC5 showed weak, non-significant associations with species richness (IRRs between 0.96 and 1.08).

These findings indicate that PC1, which represents sites with lower area, biomass, grass cover, and shrub cover but higher border vegetation, is positively related to above-ground species richness, while other vegetation structure components did not significantly influence richness (Table 4.6)

Table 4.5. Negative binomial generalized linear model (GLM) results showing the effect of principal components derived from site characteristics and land cover variables on above-ground species richness. Principal components were generated from original predictors including site area, proportion

of greyspace (25 m buffer), land cover types (Tree, Grass, Shrub, Grey, Border), and tree biomass to reduce multicollinearity.

Term	Estimate (β)	Std. Error	IRR (e^β)	IRR Lower 95% CI	IRR Upper 95% CI	p-value
(Intercept)	5.638	–	281.150	231.800	345.500	<0.001
PC1	0.157	–	1.170	1.010	1.370	0.023
PC2	0.039	–	1.040	0.890	1.220	0.602
PC3	-0.041	–	0.960	0.810	1.180	0.687
PC4	0.077	–	1.080	0.890	1.300	0.444
PC5	0.067	–	1.070	0.870	1.300	0.551

Table 4.6. Approximate effects of original environmental variables on above-ground species richness, based on loadings from principal component axes used in the negative binomial GLM.

Variable	Approximate Effect	IRR
Area	-0.090	0.910
Biomass	-0.100	0.910
Tree	-0.040	0.960
Grass	-0.130	0.870
Shrub	0.000	1.000
Grey	-0.030	0.970
Border	0.020	1.020
Buffer (25m)	0.000	1.000

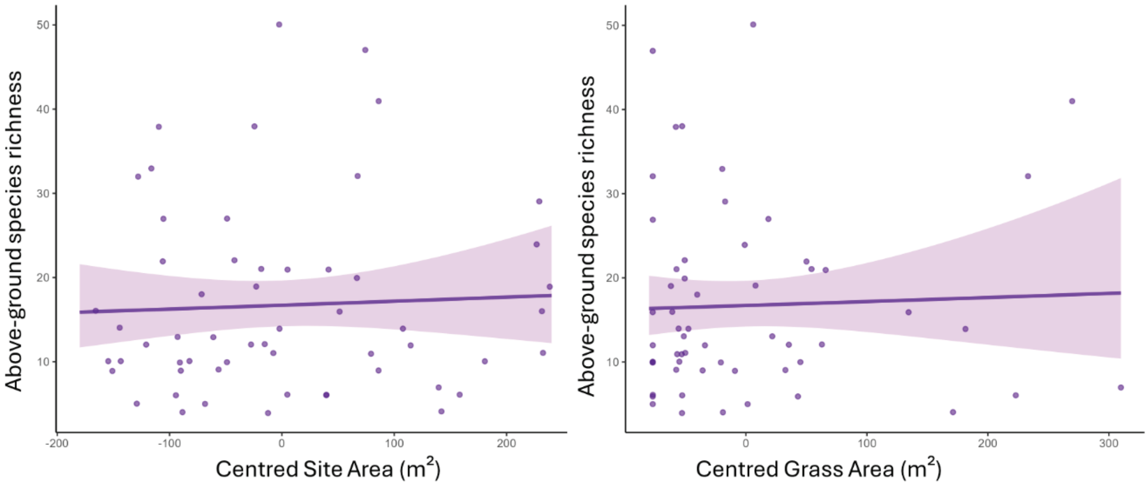


Figure 4.6. *Predicted above-ground species richness in response to site area and grass area.*

Pollinators

A negative binomial regression model was used to examine the effects of vegetation and landscape heterogeneity on pollinator abundance (Table 4.7). The model did not identify any significant main effects of area (IRR = 1.000, p = 0.700), biomass (IRR = 0.999, p = 0.140), Surrounding greyspace proportion (IRR = 1.007, p = 0.263), or habitat heterogeneity (heterogeneity index (Shannon)) (IRR = 1.005, p = 0.987). However, the interaction between area and surrounding greyspace proportion was significant (IRR = 1.000, p = 0.007), indicating that pollinator abundance is influenced by the spatial configuration of site size and its surrounding buffer. The interaction between surrounding greyspace proportion and heterogeneity index (Shannon) was not statistically significant (p = 0.122), though it showed a slight positive trend. These results suggest that while individual predictors had minimal impact, the combined effect of site area and landscape buffering plays a role in shaping pollinator community patterns.

Table 4.7. Generalised linear models (glm) fitted with negative binomial distribution for the effect of site characteristics and land cover on total pollinator abundance. Includes site area, proportion of greyspace (25m buffer), land cover heterogeneity, and biomass of trees.

Term	Estimate (β)	Std. Error	IRR (e^β)	IRR Lower 95% CI	IRR Upper 95% CI	p-value
(Intercept)	2.184	0.080	8.884	7.602	10.414	<0.001
Area	0.000	0.001	1.000	0.999	1.002	0.700
Surrounding greyspace proportion	0.007	0.006	1.007	0.995	1.019	0.263
Biomass	-0.001	0.001	0.999	0.997	1.001	0.140
Heterogeneity index (Shannon)	0.005	0.285	1.005	0.582	1.730	0.987

Area x Surrounding greyspace proportion	0.000	0.000	1.000	1.000	1.000	0.007
Surrounding greyspace proportion x Heterogeneity index (Shannon)	0.035	0.023	1.036	0.990	1.084	0.122

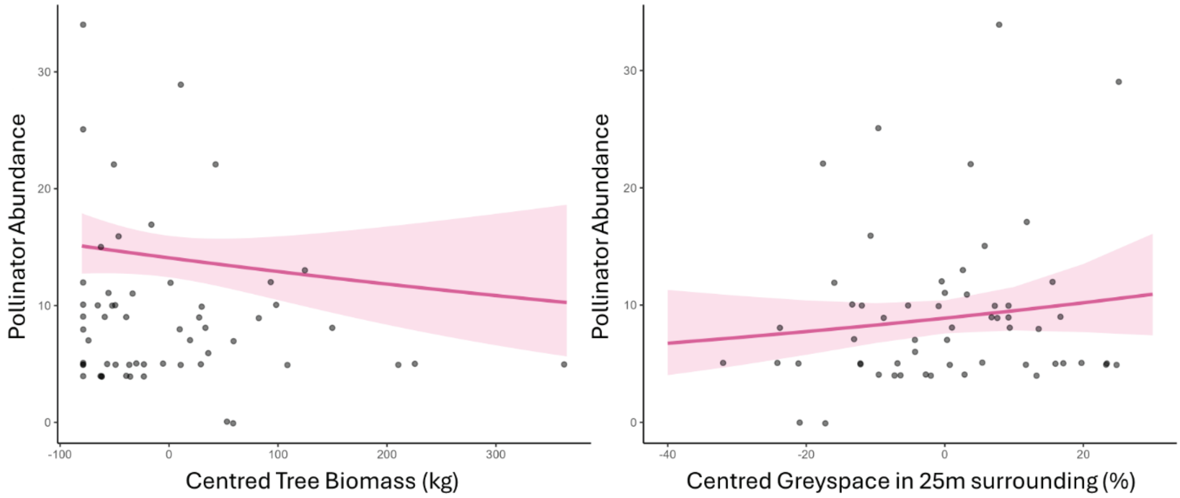


Figure 4.7. *Predicted pollinator abundance in response to tree biomass and surrounding greyspace proportion.*

A negative binomial regression model was used to examine the effects of vegetation and landscape heterogeneity on pollinator species richness (Table 4.8). The model identified a significant interaction between area and surrounding greyspace proportion (IRR = 1.000, p = 0.004), suggesting that pollinator richness is influenced by the combined effects of site area and surrounding landscape configuration. However, neither area (IRR = 1.000, p = 0.560) nor surrounding greyspace proportion (IRR = 1.000, p = 0.641) individually showed significant effects. These results indicate that interactive landscape-scale processes may play a stronger role in shaping pollinator richness than individual site characteristics.

Table 4.8. Generalised linear model (glm) fitted with a negative binomial distribution for the effect of site characteristics and land cover on pollinator species richness. Includes site area and surrounding greyspace proportion (25 m).

Term	Estimate (β)	Std. Error	IRR (e^β)	95% CI (Lower)	95% CI (Upper)	p-value
(Intercept)	2.190	0.075	8.920	7.720	10.300	<0.001
Area (c)	0.000	0.001	1.000	0.999	1.000	0.560
Buffer (25 m) (c)	-0.003	0.006	0.997	0.986	1.010	0.641
Area × Buffer (25 m)	0.000	0.000	1.000	1.000	1.000	0.004

Observations: 57; Nagelkerke R² = 0.214

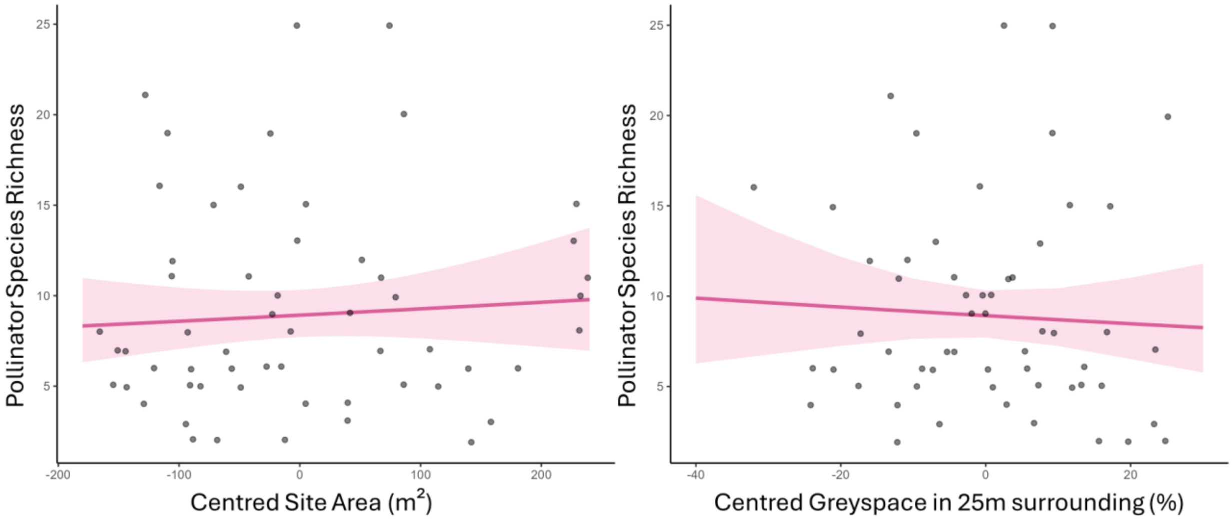


Figure 4.8. *Predicted pollinator species richness in response to site area and surrounding greyspace proportion.*

A negative binomial regression model was used to assess the effect of vegetation structural characteristics (PC2, PC3, and PC4) on pollinator abundance (Table 4.9). Among the tested predictors, PC4 had a significant positive effect (IRR = 1.478, SE = 0.134, p = 0.004), indicating that structural attributes represented by this component - potentially related to vertical layering or species richness - are associated with increased pollinator abundance. PC2 and PC3 showed weak positive associations (IRR = 1.139 and 1.141, respectively), though neither reached statistical significance (p = 0.091 and p = 0.135, respectively). These results suggest that specific elements of vegetation structure can promote pollinator presence, particularly those captured by PC4.

Table 4.9. Negative binomial generalized linear model (GLM) results showing the effect of principal components derived from site characteristics and land cover variables on total pollinator abundance. Principal components were generated from original predictors including site area, proportion of greyspace (25m buffer), land cover types (Tree, Grass, Shrub, Grey, Border), and tree biomass to reduce multicollinearity.

Term	Estimate (β)	Std. Error	IRR (e^β)	IRR Lower 95% CI	IRR Upper 95% CI	p-value
(Intercept)	2.173	0.119	8.784	6.883	10.969	<0.001
PC2	0.130	0.077	1.139	0.977	1.324	0.091
PC3	0.132	0.088	1.141	0.959	1.358	0.135
PC4	0.390	0.134	1.478	1.152	1.956	0.004

Table 4.10 presents the estimated effects of environmental variables on pollinator abundance, expressed as incidence rate ratios (IRRs). Grass cover, border vegetation, tree cover, and site area were all negatively associated with abundance (IRRs between 0.85 and 0.93), suggesting modest declines as

these variables increase. Biomass showed a minimal negative effect (IRR = 0.97). In contrast, shrub cover, grey space, and the buffer zone had positive associations (IRRs from 1.10 to 1.17), indicating slight increases in invertebrate abundance with higher values of these variables.

Table 4.10 Approximate effects of original environmental variables on pollinator invertebrate abundance, based on loadings from principal component axes used in the negative binomial GLM. Effects are expressed as estimated coefficients and incidence rate ratios (IRR), indicating the direction and magnitude of influence for each variable.

Variable	Approximate Effect	IRR
Area	-0.07	0.93
Biomass	-0.03	0.97
Tree	-0.09	0.91
Shrub	0.09	1.10
Grass	-0.13	0.88
Grey	0.10	1.11
Border	-0.16	0.85
Buffer (25m)	0.16	1.17

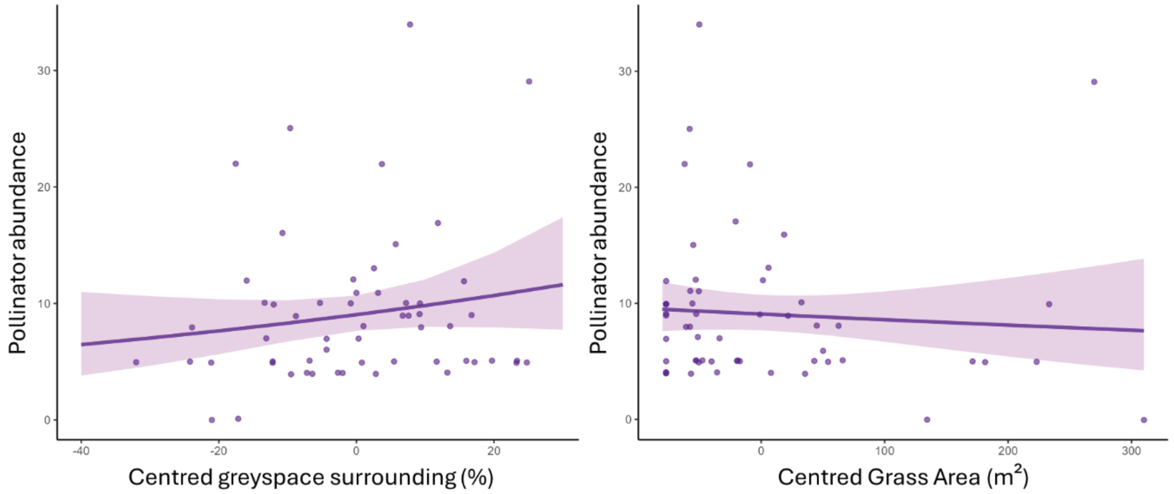


Figure 4.9. *Predicted pollinator abundance in response to surrounding greyspace proportion and grass area.*

A negative binomial regression model was used to evaluate the relationship between vegetation structure (PC1–PC5) and pollinator species richness (Table 4.11). The model revealed a significant positive association with PC5 (IRR = 1.30, 95% CI = 1.07–1.57, p = 0.004), indicating that vegetation characteristics represented by PC5 were linked to higher soil fauna richness. In contrast, PC1, PC2, PC3, and PC4 were not significant predictors (p > 0.32).

PC5 was primarily characterized by high shrub cover and buffer vegetation, coupled with low grass cover and border vegetation (Table 4.12), suggesting that pollinator species richness is positively influenced by shrub-dominated and well-buffered sites. Approximate effects of the original variables indicated that shrub cover (IRR = 1.12) and buffer vegetation (IRR = 1.17) were positively related to soil fauna richness, while grass cover (IRR = 0.84) and tree cover (IRR = 0.94) were associated with reductions in richness.

Table 4.11. Negative binomial generalized linear model (GLM) results showing the effect of principal components derived from vegetation and site characteristics on pollinator species richness.

Term	Estimate (β)	IRR (e^β)	95% CI (Low)	95% CI (High)	p-value
(Intercept)	2.179	8.83	7.53	10.39	<0.001
PC1	0.039	1.04	0.92	1.17	0.545
PC2	0.058	1.06	0.94	1.21	0.323
PC3	0.049	1.05	0.93	1.21	0.450
PC4	0.010	1.01	0.87	1.18	0.889
PC5	0.262	1.30	1.07	1.57	0.004

Table 4.12. Approximate effects of original vegetation and site variables (derived from PCA loadings) on pollinator species richness.

Variable	Approximate Effect	IRR
Area	0.00	1.00
Biomass	-0.01	0.99
Tree	-0.06	0.94
Grass	-0.17	0.84
Shrub	0.12	1.12
Grey	0.03	1.03
Border	-0.06	0.94
Buffer (25m)	0.15	1.17

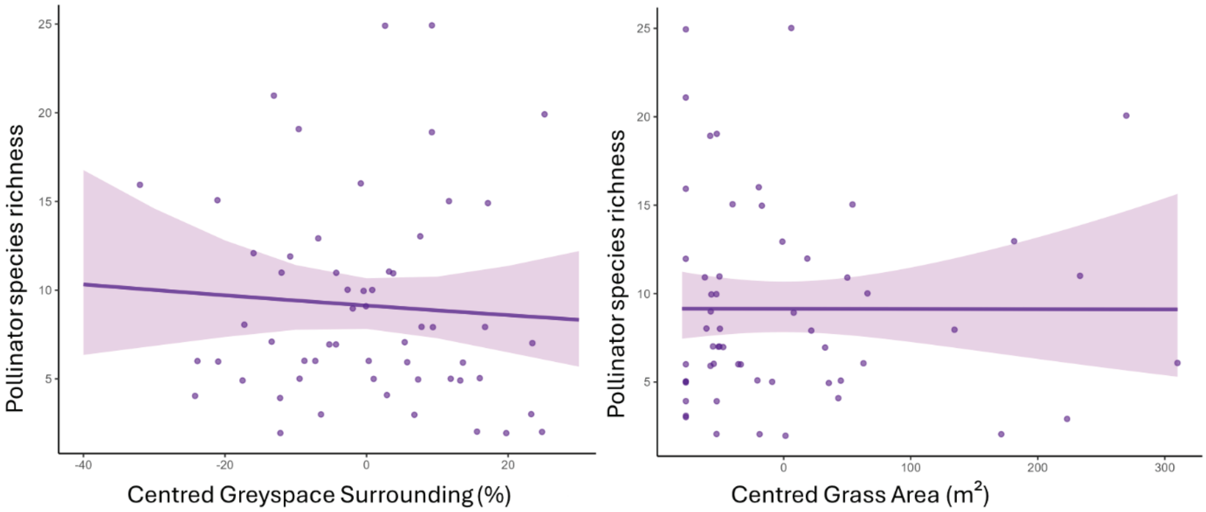


Figure 4.10. *Predicted pollinator species richness in response to surrounding greyspace proportion and grass area.*

Wasp

A negative binomial regression model was used to assess the influence of vegetation heterogeneity and landscape factors on wasp abundance (Table 4.13). None of the main effects - area (IRR = 1.000, p = 0.736), biomass (IRR = 0.999, p = 0.445), heterogeneity index (Shannon) (IRR = 1.025, p = 0.935), or surrounding greyspace proportion (IRR = 1.010, p = 0.139) - were statistically significant. Among the interaction terms, only the interaction between area and surrounding greyspace proportion reached significance (IRR = 1.000, p = 0.016), suggesting that wasp abundance is influenced by the spatial relationship between site size and surrounding land cover. Other interactions, including biomass × heterogeneity index (Shannon) and heterogeneity index (Shannon) × surrounding greyspace proportion, were not significant. These findings indicate that wasp abundance may respond more to combined landscape-scale features than to any single site variable alone.

Table 4.13. Generalised linear models (glm) fitted with negative binomial distribution for the effect of centred site characteristics and land cover on wasp abundance. Includes testing of site area, proportion of greyspace surrounding the site (25m buffer greyspace%), land cover heterogeneity, and biomass of trees within the site.

Term	Estimate	Std. Error	IRR	IRR Low	IRR High	p-value
(Intercept)	1.589	0.090	4.901	4.109	5.834	<0.001
Area	0.000	0.001	1.000	0.999	1.002	0.736
Biomass	-0.001	0.001	0.999	0.997	1.001	0.445
Heterogeneity index (Shannon)	0.025	0.305	1.025	0.568	1.850	0.935
Surrounding greyspace proportion	0.010	0.006	1.010	0.997	1.023	0.139

Area x Biomass	0.000	0.000	1.000	1.000	1.000	0.955
Area x Heterogeneity index (Shannon)	0.002	0.003	1.002	0.995	1.009	0.546
Area x Surrounding greyspace proportion	0.000	0.000	1.000	1.000	1.000	0.016
Biomass x Heterogeneity index (Shannon)	-0.004	0.005	0.996	0.986	1.006	0.488
Biomass x Surrounding greyspace proportion	0.000	0.000	1.000	1.000	1.000	0.864
Heterogeneity index (Shannon) x Surrounding greyspace proportion	-0.005	0.023	0.995	0.949	1.043	0.823

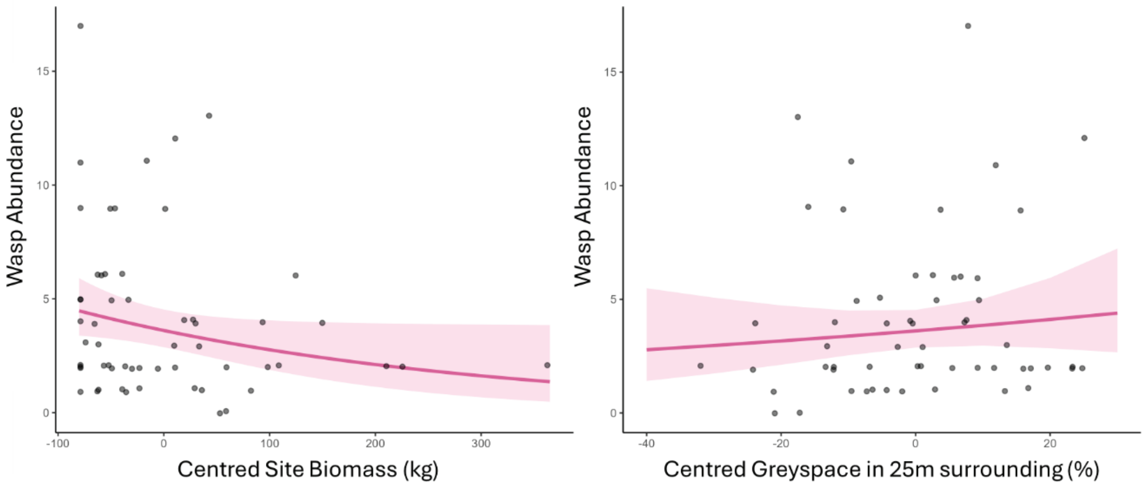


Figure 4.11. *Predicted wasp abundance in response to site biomass and surrounding greyspace proportion.*

A negative binomial regression model was used to assess the effect of vegetation structural characteristics on pollinator abundance (Table 4.14). The intercept is highly significant, representing the baseline wasp abundance. Among the predictors, PC4 is statistically significant (p = 0.004), with an IRR of 1.478, indicating a 47.8% increase in wasp abundance per unit increase in PC4. PC2 and PC3 show positive but non-significant trends (p = 0.091 and 0.135, respectively), suggesting possible associations worth exploring further, though not conclusive at the 0.05 level.

Table 4.15 summarises the approximate effects of original environmental variables on above-ground invertebrate abundance using coefficients and incidence rate ratios (IRR) derived from the negative binomial GLM. Positive coefficients and IRR values greater than 1 (e.g., shrub cover [IRR = 1.13] and Buffer (25m) [IRR = 1.17]) indicate variables associated with increased invertebrate abundance. Conversely, negative coefficients and IRR values below 1 (e.g., grass cover [IRR = 0.82] and border [IRR = 0.83]) reflect variables linked to reduced abundance. Most vegetation variables (area, biomass, tree, grass) showed negative effects, whereas grey cover had a slight positive influence (IRR = 1.09), suggesting complex habitat relationships influencing invertebrate populations.

Table 4.14. Negative binomial generalized linear model (GLM) results showing the effect of principal components derived from site characteristics and land cover variables on total wasp invertebrate abundance. Principal components were generated from original predictors including site area, proportion of greyspace (25m buffer), land cover types (Tree, Grass, Shrub, Grey, Border), and tree biomass to reduce multicollinearity.

Term	Estimate (β)	Std. Error	IRR (e^β)	IRR Lower 95% CI	IRR Upper 95% CI	p-value
(Intercept)	2.173	0.119	8.784	6.883	10.969	<0.001
PC2	0.130	0.077	1.139	0.977	1.324	0.091
PC3	0.132	0.088	1.141	0.959	1.358	0.135
PC4	0.390	0.134	1.478	1.152	1.956	0.004

Table 4.15. Approximate effects of original environmental variables on above-ground invertebrate abundance, based on loadings from principal component axes used in the negative binomial GLM. Effects are expressed as estimated coefficients and incidence rate ratios (IRR), indicating the direction and magnitude of influence for each variable.

Variable	Approximate Effect	IRR
Area	-0.07	0.93
Biomass	-0.05	0.96
Tree	-0.09	0.92
Shrub	0.13	1.13
Grass	-0.20	0.82
Grey	0.08	1.09
Border	-0.18	0.83
Buffer (25m)	0.16	1.17

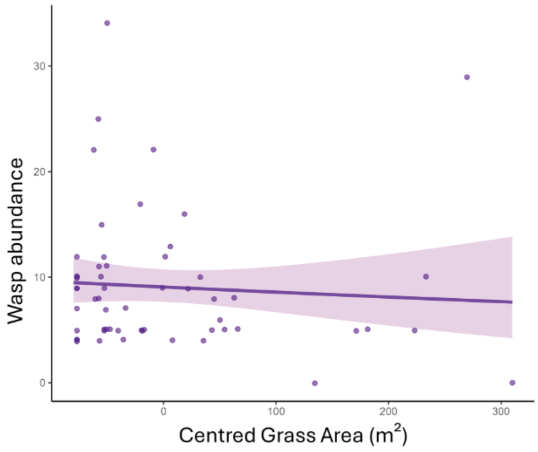


Figure 4.12. *Predicted wasp abundance in response to grass area..*

Bee

A negative binomial regression model assessed the effects of area, surrounding greyspace proportion, and their interaction on bee abundance (Table 4.16). The interaction between area and surrounding greyspace proportion was significant (IRR = 1.000, p = 0.004), indicating a positive combined effect on bee abundance in buffered landscapes. Surrounding greyspace proportion alone showed a positive trend (IRR = 1.010, p = 0.090), while area alone was not significant (IRR = 1.000, p = 0.856). These results suggest local landscape context modifies the influence of site area on bee abundance.

Term	Estimate (β)	Std. Error	IRR (e^β)	IRR Lower 95% CI	IRR Upper 95% CI	p-value
(Intercept)	1.578	0.080	4.847	4.138	5.668	<0.001
Area	0.000	0.001	1.000	0.999	1.002	0.856
Surrounding greyspace proportion	0.010	0.006	1.010	0.998	1.022	0.090
Area x Surrounding greyspace proportion	0.000	0.000	1.000	1.000	1.000	0.004

Table 4.16. Generalised linear models (glm) fitted with negative binomial distribution for the effect of centred site characteristics and land cover on bee abundance. Includes testing of site area, proportion of greyspace surrounding the site (25m buffer greyspace%), land cover heterogeneity, and biomass of trees within the site.

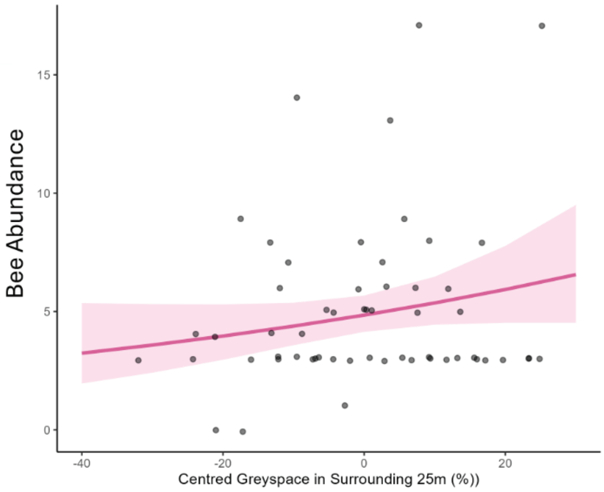


Figure 4.13. *Predicted bee abundance in response to surrounding greyspace proportion.*

A negative binomial regression model was used to assess the effect of vegetation structural characteristics on pollinator abundance (Table 4.17). Neither PC2 (IRR = 1.169, p = 0.121) nor PC4 (IRR = 1.280, p = 0.144) were statistically significant predictors of bee abundance, though both showed positive trends.

Table 4.18 presents the estimated effects of environmental variables on bee abundance derived from principal component loadings used in the negative binomial GLM. The results indicate that shrub cover (IRR = 1.05), grey infrastructure (IRR = 1.13), and buffer zones (IRR = 1.16) positively influenced bee abundance, suggesting these features may provide beneficial foraging or nesting opportunities. Conversely, variables such as area (IRR = 0.94), tree cover (IRR = 0.92), grass cover (IRR = 0.94), and border presence (IRR = 0.86) had negative effects, implying reduced suitability or resource availability for bees in these contexts. Biomass showed only a minimal negative effect (IRR = 0.98). Overall, the findings highlight the importance of structural and landscape features in shaping bee populations within urban greenspaces.

Table 4.17. Negative binomial generalized linear model (GLM) results showing the effect of principal components derived from site characteristics and land cover variables on total bee abundance. Principal components were generated from original predictors including site area, proportion of greyspace (25m buffer), land cover types (Tree, Grass, Shrub, Grey, Border), and tree biomass to reduce multicollinearity.

Term	Estimate (β)	Std. Error	IRR (e^β)	IRR Lower 95% CI	IRR Upper 95% CI	p-value
(Intercept)	1.556	0.157	4.739	3.420	6.343	<0.001
PC2	0.156	0.101	1.169	0.953	1.419	0.121
PC4	0.247	0.169	1.280	0.939	1.830	0.144

Table 4.18. Approximate effects of original environmental variables on bee abundance, based on loadings from principal component axes used in the negative binomial GLM. Effects are expressed as estimated coefficients and incidence rate ratios (IRR), indicating the direction and magnitude of influence for each variable.

Area	-0.07	0.94
Biomass	-0.02	0.98
Tree	-0.08	0.92
Shrub	0.05	1.05
Grass	-0.06	0.94
Grey	0.12	1.13
Border	-0.15	0.86

Buffer (25m)	0.15	1.16

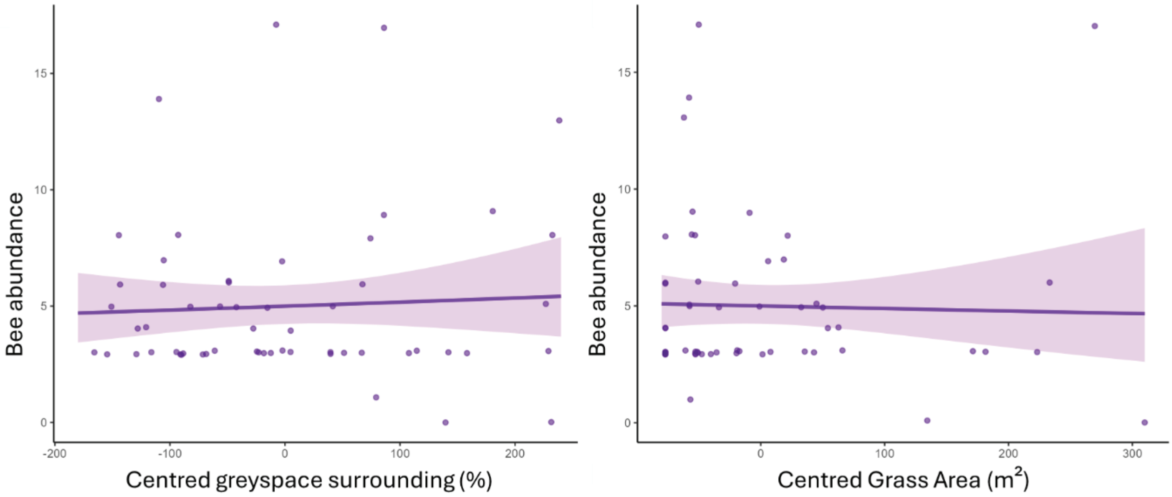


Figure 4.14. *Predicted bee abundance in response to surrounding greyspace proportion and grass area,*

Fly

Biomass was a significant negative predictor of fly abundance (Table 4.19) (IRR = 0.995, p < 0.001). Surrounding greyspace proportion was not significant (p = 0.536). The interaction between biomass and surrounding greyspace proportion was significant (IRR = 1.000, p = 0.005), indicating the effect of biomass varies with surrounding greyspace proportion levels.

Table 4.19. Generalised linear models (glm) fitted with negative binomial distribution for the effect of centred site characteristics and land cover on fly abundance. Includes testing of site area, proportion of greyspace surrounding the site (25m buffer greyspace%), land cover heterogeneity, and biomass of trees within the site.

Term	Estimate (β)	Std. Error	IRR (e^β)	IRR Lower 95% CI	IRR Upper 95% CI	p-value
(Intercept)	5.586	0.097	266.534	221.758	324.169	<0.001
Biomass	-0.005	0.001	0.995	0.993	0.998	<0.001
Surrounding greyspace proportion	-0.005	0.007	0.995	0.981	1.010	0.536
Biomass x Surrounding greyspace proportion	0.000	0.000	1.000	1.000	1.000	0.005

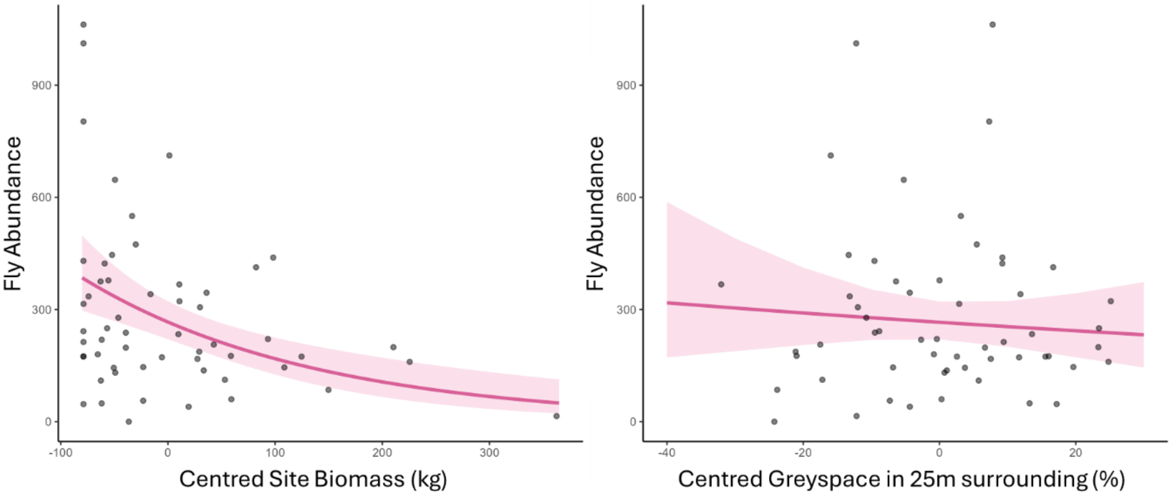


Figure 4.15. *Predicted fly abundance in response to site biomass and surrounding greyspace proportion*

A separate negative binomial regression model was used to examine the influence of site-level environmental variation, represented by the first principal component (PC1), on total invertebrate count (Table 4.20). The model identified a significant positive effect of PC1 (IRR = 1.19, SE = 0.09, p = 0.026), indicating that sites with higher PC1 scores - potentially reflecting gradients in environmental structure - were associated with increased invertebrate abundance. Although the model explained a smaller proportion of variance (Nagelkerke R^2 = 0.096), it highlights that broader environmental gradients captured by PC1 can play a role in shaping invertebrate communities.

Table 4.21 summarises the approximate effects of environmental variables on above-ground invertebrate abundance. Negative coefficients for area (-0.11; IRR = 0.89), biomass (-0.11; IRR = 0.90), grass (-0.13; IRR = 0.88), and tree cover (-0.04; IRR = 0.96) indicate a general negative influence on invertebrate abundance, while shrub cover (-0.01; IRR = 0.99) and grey surface (-0.02; IRR = 0.98) showed negligible effects. Border and buffer variables exhibited no measurable influence (coefficients = 0.00; IRR = 1.00).

Table 4.20. Negative binomial generalized linear model (GLM) results showing the effect of principal components derived from site characteristics and land cover variables on total fly abundance. Principal components were generated from original predictors including site area, proportion of greyspace (25m buffer), land cover types (Tree, Grass, Shrub, Grey, Border), and tree biomass to reduce multicollinearity.

Term	Estimate (β)	Std. Error	IRR (e^β)	IRR Lower 95% CI	IRR Upper 95% CI	p-value	Term
(Intercept)	1.556	0.157	4.739	3.420		6.343	<0.001
PC2	0.156	0.101	1.169	0.953		1.419	0.121
PC4	0.247	0.169	1.280	0.939		1.830	0.144

Table 4.21. Approximate effects of original environmental variables on above-ground invertebrate abundance, based on loadings from principal component axes used in the negative binomial GLM. Effects are expressed as estimated coefficients and incidence rate ratios (IRR), indicating the direction and magnitude of influence for each variable.

Variable	Approximate Effect	IRR
Area	-0.11	0.89
Biomass	-0.11	0.90
Tree	-0.04	0.96
Shrub	-0.01	0.99
Grass	-0.13	0.88
Grey	-0.02	0.98
Border	0.00	1.00
Buffer (25m)	0.00	1.00

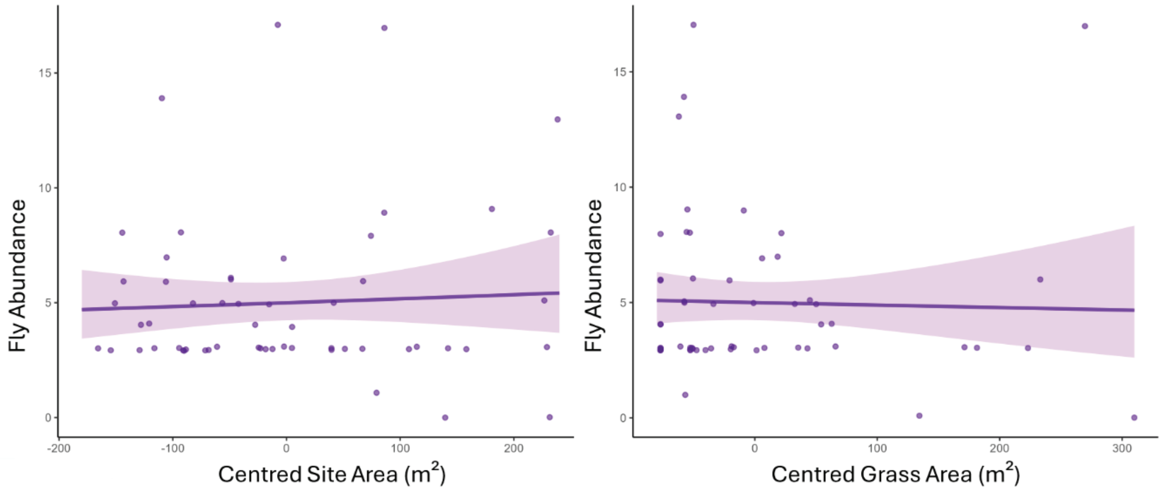


Figure 4.16. *Predicted fly abundance in response to site area and grass area..*

Below-ground invertebrates

A negative binomial regression model evaluated the effects of area, biomass, habitat heterogeneity (HGI Shannon), and their interactions on below-ground invertebrate abundance (Table 4.22). Significant predictors included site heterogeneity (HGI Shannon) which showed a negative effect (IRR = 0.365, p = 0.012), indicating that higher habitat heterogeneity reduces below-ground invertebrate abundance. The interaction between area and HGI Shannon was significant and positive (IRR = 1.010, p = 0.017), suggesting the negative effect of habitat diversity weakens with increasing site area. Biomass and surrounding greyspace proportion had a significant negative interactive effect (IRR = 0.9999, p = 0.039). The three-way interaction among area, biomass, and HGI Shannon was significant (IRR = 1.000, p = 0.030), indicating complex combined influences on abundance. The main effect of area was marginally significant and negative (IRR = 0.998, p = 0.052).

Table 4.22. Generalised linear models (glm) fitted with negative binomial distribution for the effect of centred site characteristics and land cover on total below-ground invertebrate abundance. Includes testing of site area, proportion of greyspace surrounding the site (25m buffer greyspace%), land cover heterogeneity, and biomass of trees within the site.

Term	Estimate (β)	Std. Error	IRR (e^β)	IRR Lower 95% CI	IRR Upper 95% CI	p-value
(Intercept)	3.799	0.098	44.646	37.008	54.457	<0.001
Area	-0.002	0.001	0.998	0.997	1.000	0.052
Biomass	0.002	0.001	1.002	0.999	1.004	0.255
Heterogeneity index (Shannon)	-1.007	0.399	0.365	0.162	0.844	0.012
Surrounding greyspace proportion	-0.005	0.006	0.995	0.983	1.008	0.429
Area x Biomass	-0.00001	0.00001	1.000	1.000	1.000	0.326
Area x Heterogeneity index (Shannon)	0.010	0.004	1.010	1.001	1.019	0.017
Biomass x Heterogeneity index (Shannon)	-0.011	0.006	0.989	0.977	1.002	0.076
Biomass x Surrounding greyspace proportion	-0.00015	0.00007	0.9999	0.9997	1.000	0.039
Area x Biomass x Heterogeneity index (Shannon)	0.00016	0.00007	1.0002	1.0000	1.0003	0.030

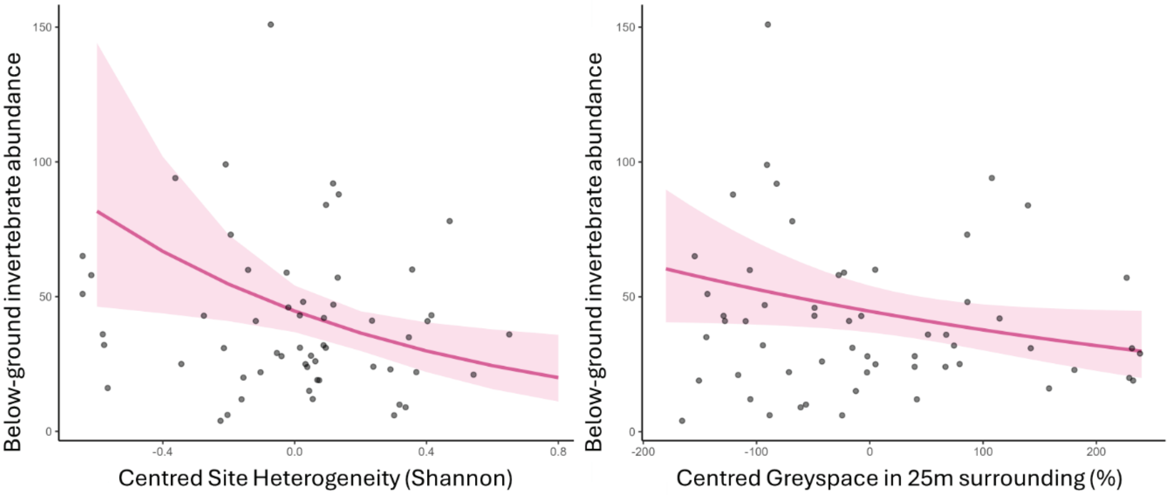


Figure 4.9. *Predicted below-ground abundance in response to site heterogeneity and surrounding greyspace proportion.*

A negative binomial regression model was used to examine the effects of vegetation and habitat heterogeneity on below-ground species richness (Table 4.23). The model revealed a significant negative effect of habitat heterogeneity (Shannon) (IRR = 0.650, p = 0.030), indicating that below-ground species richness decreased with increasing habitat heterogeneity. Area (IRR = 1.000, p = 0.919) and biomass (IRR = 1.000, p = 0.203) showed no significant effects, and the interaction between area and biomass was also non-significant (p = 0.104). These findings suggest that below-ground communities may be more sensitive to habitat composition than to site area or vegetation biomass.

Table 4.23. Generalised linear model (glm) fitted with a negative binomial distribution for the effect of site characteristics and habitat heterogeneity on below-ground species richness. Includes site area, habitat heterogeneity (Shannon), and biomass.

Term	Estimate (β)	Std. Error	IRR (e^β)	95% CI (Lower)	95% CI (Upper)	p-value
(Intercept)	1.610	0.064	4.980	4.390	5.630	<0.001
Area (c)	0.000	0.001	1.000	0.999	1.000	0.919
Heterogeneity (Shannon)	-0.434	0.200	0.648	0.439	0.960	0.030
Biomass (c)	-0.001	0.001	0.999	0.997	1.000	0.203
Area × Biomass	0.000	0.000	1.000	1.000	1.000	0.104

Observations: 57; Nagelkerke $R^2 = 0.296$

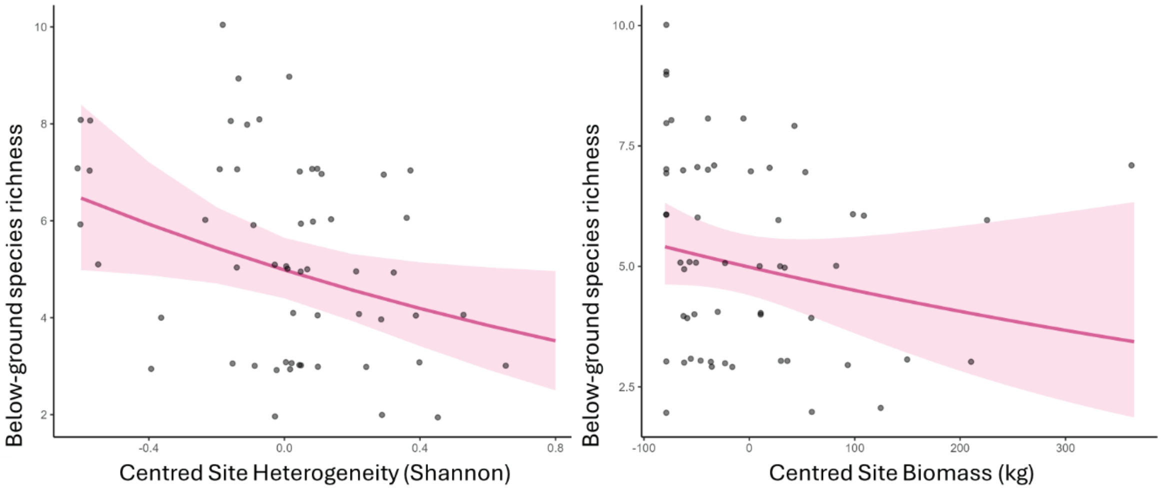


Figure 4.24. *Predicted below-ground species richness in response to site heterogeneity and site biomass.*

A negative binomial regression model assessed the effects of principal components (PC2 to PC6) on invertebrate abundance (Table 4.24). Significant negative effects on abundance were found for PC2 (IRR = 0.764, p = 0.002), PC3 (IRR = 0.720, p = 0.002), PC4 (IRR = 0.654, p < 0.001), and PC5 (IRR = 0.529, p < 0.001), indicating these environmental gradients negatively influence invertebrate abundance. Conversely, PC6 had a significant positive effect (IRR = 2.023, p = 0.037).

Table 4.25 shows that below-ground invertebrate abundance is most positively associated with border vegetation (IRR = 1.17) and grass cover (IRR = 1.02), while grey cover (-0.10, IRR = 0.90) and buffer proximity (-0.10, IRR = 0.91) exhibit negative effects. Other variables display minimal influence, indicating a relatively weak overall relationship between environmental factors and below-ground invertebrate abundance.

Table 4.24. Negative binomial generalized linear model (GLM) results showing the effect of principal components derived from site characteristics and land cover variables on total below-ground invertebrate abundance. Principal components were generated from original predictors including site area, proportion of greyspace (25m buffer), land cover types (Tree, Grass, Shrub, Grey, Border), and tree biomass to reduce multicollinearity.

Term	Estimate (β)	Std. Error	IRR (e^β)	IRR Lower 95% CI	IRR Upper 95% CI	p-value
(Intercept)	3.778	0.115	43.750	35.009	55.010	<0.001
PC2	-0.269	0.085	0.764	0.642	0.914	0.002
PC3	-0.328	0.105	0.720	0.586	0.881	0.002
PC4	-0.424	0.104	0.654	0.535	0.794	<0.001
PC5	-0.636	0.167	0.529	0.380	0.729	<0.001
PC6	0.705	0.338	2.023	1.045	3.991	0.037

Table 4.25. Approximate effects of original environmental variables on below-ground invertebrate abundance, based on loadings from principal component axes used in the negative binomial GLM. Effects are expressed as estimated coefficients and incidence rate ratios (IRR), indicating the direction and magnitude of influence for each variable.

Variable	Approximate Effect	IRR
Area	-0.01	0.99
Biomass	0.01	1.01
Tree	-0.02	0.98
Shrub	0.00	1.00
Grass	0.02	1.02
Grey	-0.10	0.90
Border	0.16	1.17
Buffer (25m)	-0.10	0.91

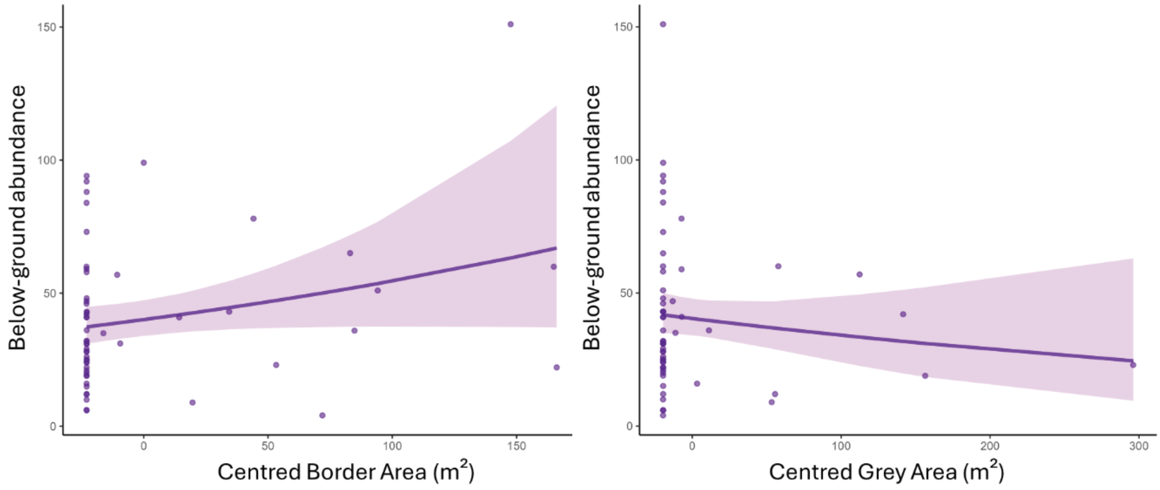


Figure 4.25. *Predicted below-ground invertebrate abundance in response to border area and grey area.*

A negative binomial regression model was used to examine the relationship between vegetation structure (PC1–PC5) and below-ground species richness (Table 4.26). None of the principal components were statistically significant predictors (all $p > 0.10$), indicating that vegetation structure was not strongly associated with below-ground species richness.

Approximate effects of the original variables (Table 4.27) indicated weak, non-significant trends. Border vegetation showed a small positive effect (IRR = 1.08), while buffer vegetation (IRR = 0.91) and area (IRR = 0.93) showed slight negative trends. However, these associations were weak and not statistically significant.

Table 4.26. Negative binomial GLM results showing the effect of principal components on below-ground species richness.

Term	Estimate (β)	IRR (e^β)	95% CI (Low)	95% CI (High)	p-value
(Intercept)	3.68	39.83	33.93	47.12	<0.001
PC1	0.05	1.05	0.94	1.18	0.365
PC2	0.08	1.08	0.96	1.21	0.245
PC3	-0.03	0.97	0.84	1.14	0.669
PC4	0.01	1.01	0.84	1.20	0.944
PC5	-0.14	0.87	0.73	1.02	0.107

Table 4.27. Approximate effects of original vegetation and site variables (derived from PCA loadings) on below-ground species richness.

Variable	Approximate Effect	IRR
Area	-0.07	0.93
Biomass	-0.01	0.99
Tree	-0.05	0.95
Grass	0.06	1.06
Shrub	-0.03	0.97
Grey	-0.03	0.97
Border	0.08	1.08
Buffer (25m)	-0.10	0.91

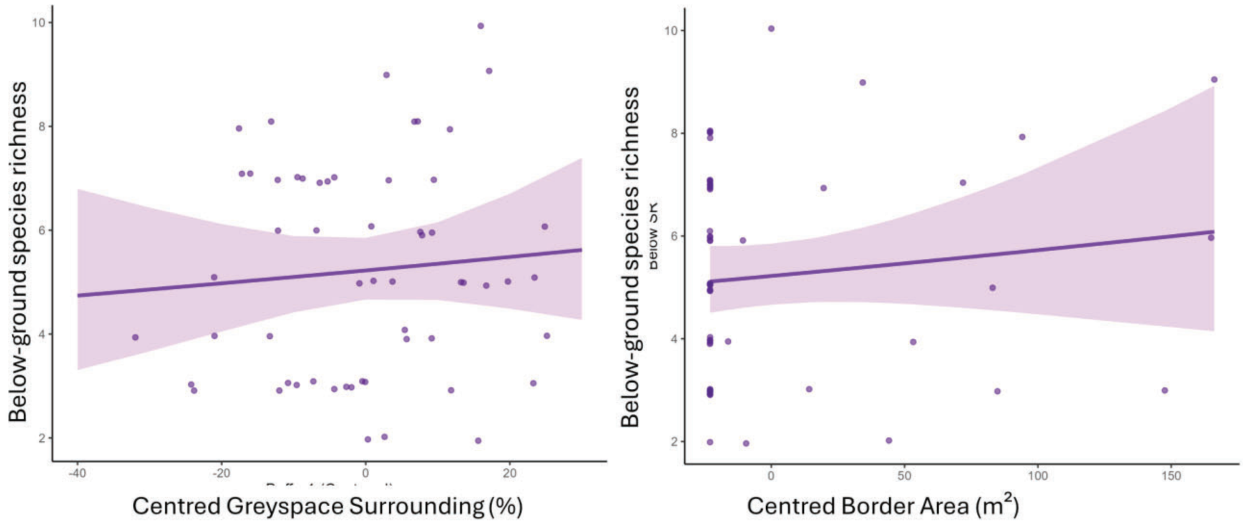


Figure 4.26. *Predicted below-ground species richness in response to surrounding greyspace proportion and border area.*

Decomposer

A negative binomial regression evaluated the effects of area, biomass, habitat heterogeneity (HGI Shannon), buffer, and their interactions on decomposer abundance (Table 4.28). The model showed a significant positive effect of habitat heterogeneity (IRR = 2.132, p = 0.037), significant interaction effects between area and buffer (IRR = 1.000, p = 0.035) and between biomass and habitat heterogeneity (IRR = 1.028, p = 0.005). These suggest that habitat heterogeneity strongly enhances abundance, while combined effects of area & buffer and biomass & habitat heterogeneity also influence the response.

Table 4.28. Generalised linear models (glm) fitted with negative binomial distribution for the effect of centred site characteristics and land cover on total Decomposer abundance. Includes testing of site area, proportion of greyspace surrounding the site (25m buffer greyspace%), land cover heterogeneity, and biomass of trees within the site.

Term	Estimate (β)	Std. Error	IRR (e^β)	IRR Lower 95% CI	IRR Upper 95% CI	p-value
(Intercept)	1.526	0.103	4.601	3.726	5.589	<0.001
Area	0.000	0.001	1.000	0.999	1.002	0.829
Biomass	-0.003	0.002	0.997	0.994	1.000	0.095
Heterogeneity index (Shannon)	0.757	0.362	2.132	1.037	4.305	0.037
Surrounding greyspace proportion	0.002	0.006	1.002	0.991	1.014	0.728
Area x Surrounding greyspace proportion	0.00013	0.00006	1.000	1.000	1.000	0.035
Biomass x Heterogeneity index (Shannon)	0.028	0.010	1.028	1.008	1.049	0.005

Heterogeneity index (Shannon) x Surrounding greyspace proportion	0.039	0.024	1.040	0.992	1.091	0.108

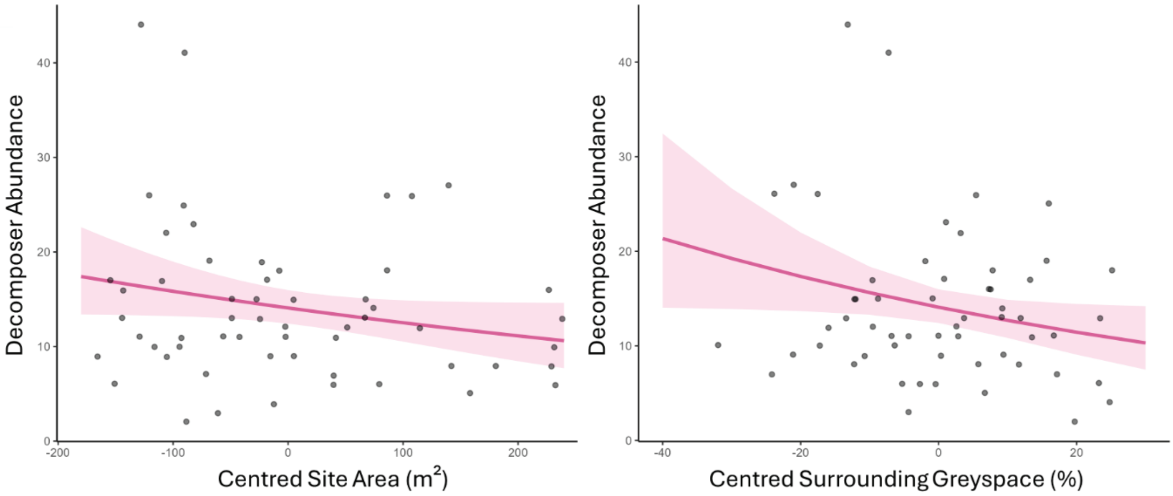


Figure 4.27. *Predicted decomposer abundance in response to site area and surrounding greyspace proportion.*

A negative binomial regression model was used to assess the effects of vegetation and landscape heterogeneity on decomposer species richness (Table 4.29). The model revealed a significant positive effect of habitat heterogeneity (HGI Shannon) (IRR = 2.130, p = 0.037), indicating that higher habitat heterogeneity is associated with increased decomposer richness. Additionally, the interaction between biomass and habitat heterogeneity was significant (IRR = 1.030, p = 0.005), suggesting that habitat heterogeneity modifies the effect of biomass on decomposer communities. The interaction between area and surrounding greyspace proportion was also significant (IRR = 1.000, p = 0.035), though its effect size was small. Other predictors, including area, biomass, surrounding greyspace proportion, and the interaction between heterogeneity and surrounding greyspace proportion, were not statistically significant. Overall, these findings highlight the importance of habitat heterogeneity and interactive effects in shaping decomposer richness.

Table 4.29. Generalised linear model (glm) fitted with a negative binomial distribution for the effect of site characteristics and land cover on decomposer species richness. Includes site area, biomass, habitat heterogeneity (HGI Shannon), and surrounding greyspace proportion (25 m).

Term	Estimate (β)	Std. Error	IRR (e^β)	95% CI (Lower)	95% CI (Upper)	p-value
(Intercept)	1.530	0.103	4.600	3.730	5.590	<0.001
Area (c)	0.000	0.001	1.000	0.999	1.000	0.829
Biomass (c)	-0.003	0.002	0.997	0.994	1.000	0.095
Heterogeneity index (HGI Shannon)	0.757	0.362	2.130	1.040	4.300	0.037
Buffer (25 m) (c)	0.002	0.006	1.000	0.991	1.010	0.728
Area × Buffer (25 m)	0.000	0.000	1.000	1.000	1.000	0.035
Biomass × Heterogeneity (HGI)	0.028	0.010	1.030	1.010	1.050	0.005
Heterogeneity × Buffer (25 m)	0.039	0.025	1.040	0.992	1.090	0.108

Observations: 36; Nagelkerke $R^2 = 0.422$

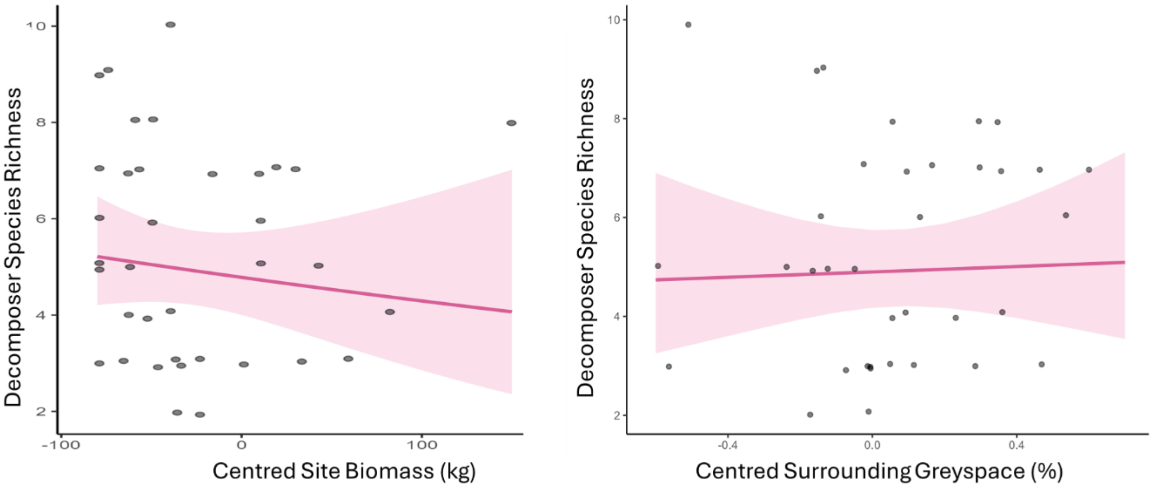


Figure 4.28. *Predicted decomposer abundance in response to site biomass and surrounding greyspace proportion.*

A negative binomial regression model was used to assess the effect of vegetation structural characteristics on pollinator abundance (Table 4.30). PC4 had a statistically significant positive effect (IRR = 1.478, 95% CI x 1.152–1.956, p = 0.004), indicating that an increase in PC4 is associated with about a 48% increase in the expected count or rate of the outcome. PC2 and PC3 had positive but non-significant effects (IRR ≈ 1.14; p = 0.091 and 0.135, respectively).

Table 4.31 shows that decomposer invertebrate abundance is most negatively associated with buffer distance (IRR = 0.88) and grey cover (IRR = 0.92), with weaker negative effects from area, biomass, and vegetation variables. Border presence was the only variable with a positive influence (IRR = 1.06), suggesting limited but localized environmental drivers.

Table 4.30. Negative binomial generalized linear model (GLM) results showing the effect of principal components derived from site characteristics and land cover variables on total decomposer abundance.

Principal components were generated from original predictors including site area, proportion of greyspace (25m buffer), land cover types (Tree, Grass, Shrub, Grey, Border), and tree biomass to reduce multicollinearity.

Term	Estimate (β)	Std. Error	IRR (e^β)	IRR Lower 95% CI	IRR Upper 95% CI	p-value
(Intercept)	2.173	0.119	8.784	6.883	10.969	<0.001
PC2	0.130	0.077	1.139	0.977	1.324	0.091
PC3	0.132	0.088	1.141	0.959	1.358	0.135
PC4	0.390	0.134	1.478	1.152	1.956	0.004

Table 4.31. Approximate effects of original environmental variables on decomposer invertebrate abundance, based on loadings from principal component axes used in the negative binomial GLM. Effects are expressed as estimated coefficients and incidence rate ratios (IRR), indicating the direction and magnitude of influence for each variable.

Variable	Approximate Effect	IRR
Area	-0.06	0.94
Biomass	-0.03	0.97
Tree	-0.01	0.99
Shrub	-0.01	0.99
Grass	-0.01	0.99
Grey	-0.08	0.92
Border	0.06	1.06
Buffer (25m)	-0.13	0.88

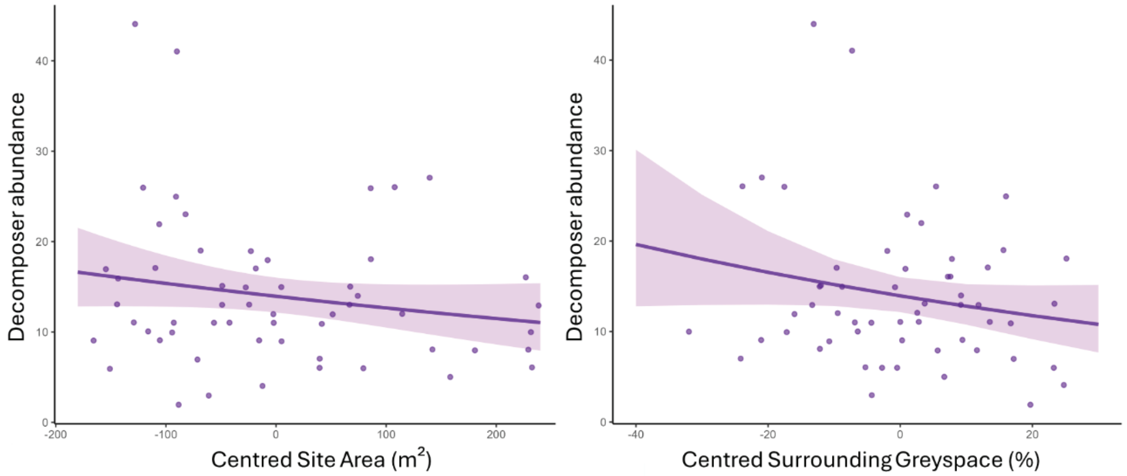


Figure 4.29. *Predicted pollinator abundance in response to surrounding greyspace proportion and surrounding greyspace proportion.*

A negative binomial regression model was fitted to assess the effect of vegetation structure (PC1–PC5) on decomposer species richness (Table 4.32). None of the principal components were statistically significant predictors (all p > 0.11), indicating no strong association between vegetation structure and species richness for this group.

Approximate effects of the original variables (Table 4.33) suggest weak trends, with border vegetation showing a slight positive association (IRR = 1.03) and buffer vegetation and grey cover showing small negative associations (IRR = 0.88 and IRR = 0.94, respectively). However, these effects were not statistically significant, indicating limited evidence for vegetation effects on species richness for this group.

Table 4.32. Negative binomial generalized linear model (GLM) results showing the effect of principal components derived from site characteristics and land cover variables on total decomposer species

richness. Principal components were generated from original predictors including site area, proportion of greyspace (25m buffer), land cover types (Tree, Grass, Shrub, Grey, Border), and tree biomass to reduce multicollinearity.

Term	Estimate (β)	IRR (e^β)	95% CI (Low)	95% CI (High)	p-value
(Intercept)	2.62	13.78	12.09	15.75	<0.001
PC1	0.06	1.06	0.97	1.16	0.200
PC2	0.06	1.06	0.96	1.17	0.245
PC3	-0.07	0.93	0.82	1.06	0.244
PC4	0.08	1.08	0.94	1.24	0.238
PC5	-0.10	0.90	0.78	1.02	0.119

Table 4.33. Approximate effects of original environmental variables on decomposer species richness, based on loadings from principal component axes used in the negative binomial GLM. Effects are expressed as estimated coefficients and incidence rate ratios (IRR), indicating the direction and magnitude of influence for each variable.

Variable	Approximate Effect	IRR
Area	-0.08	0.92
Biomass	-0.04	0.96
Tree	-0.02	0.98
Grass	0.01	1.01
Shrub	-0.02	0.98
Grey	-0.06	0.94
Border	0.03	1.03

Buffer (25m)	-0.13	0.88

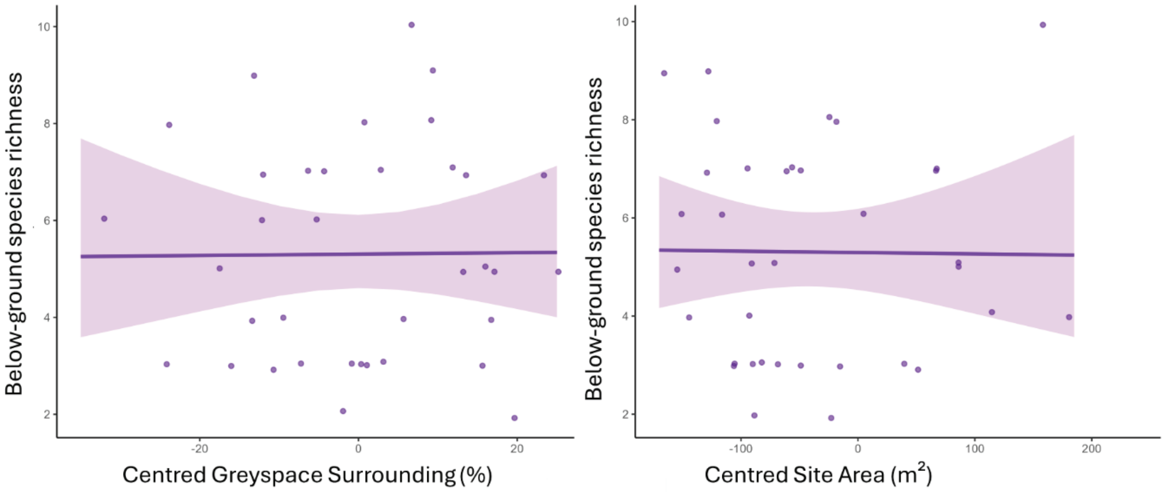


Figure 4.30. *Predicted decomposer species richness in response to surrounding greyspace proportion and site area.*

Collembola

A negative binomial regression model was used to assess the effect of vegetation heterogeneity (Table 4.34). The intercept is highly significant, indicating the baseline Collembola abundance. None of the predictors (Biomass, Surrounding greyspace proportion, or their interaction) were significant (all p > 0.14). IRRs near 1 suggest negligible effects.

Table 4.34. Generalised linear models (glm) fitted with negative binomial distribution for the effect of centred site characteristics and land cover on Collembola abundance. Includes testing of site area, proportion of greyspace surrounding the site (25m buffer greyspace%), land cover heterogeneity, and biomass of trees within the site.

Term	Estimate (β)	Std. Error	IRR (e^β)	IRR Lower 95% CI	IRR Upper 95% CI	p-value
(Intercept)	2.330	0.080	10.310	8.860	12.030	<0.001
Biomass	-0.001	0.001	0.999	0.997	1.001	0.293
Surrounding greyspace proportion	-0.004	0.006	0.996	0.984	1.008	0.531
Biomass x Surrounding greyspace proportion	-0.0001	0.0001	1.000	1.000	1.000	0.149

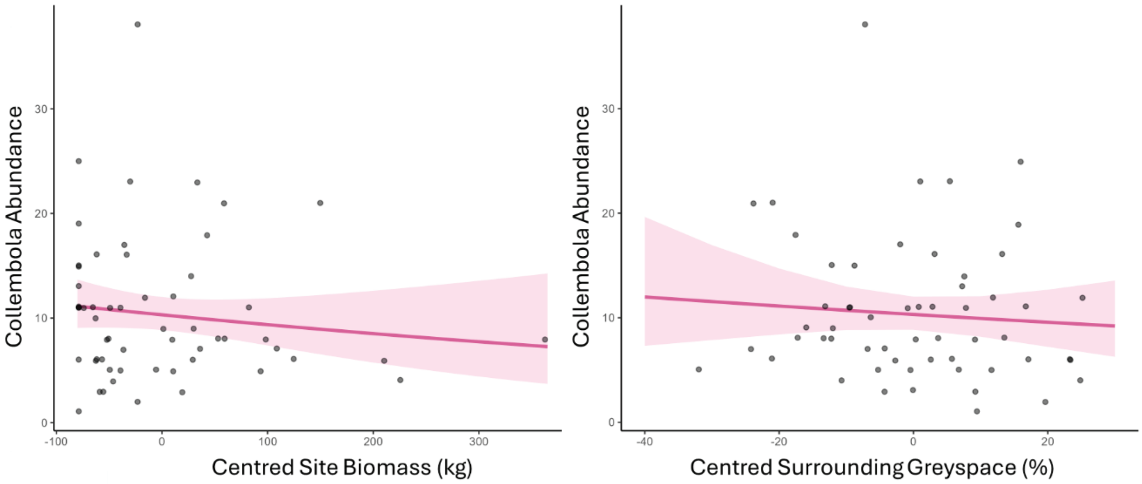


Figure 4.31. *Predicted Collembola abundance in response to site biomass and surrounding greyspace proportion.*

A negative binomial regression model was fitted to assess the effect of vegetation structure (PC1–PC5) on Collembola Abundance. The intercept is highly significant (Table 4.35). PC4 showed a non-significant negative effect (IRR = 0.790, p = 0.140), suggesting a potential but unsupported decrease in Collembola abundance with PC4.

Table 4.36 shows that above-ground invertebrate abundance is most negatively associated with grey cover (IRR = 0.90) and buffer distance (IRR = 0.92), while border presence exerts a modest positive effect (IRR = 1.15). Other variables display minimal influence, suggesting that habitat structure and proximity play more substantial roles than vegetation characteristics in shaping invertebrate abundance.

4.35 Negative binomial generalized linear model (GLM) results showing the effect of principal components derived from site characteristics and land cover variables on total collembola abundance. Principal components were generated from original predictors including site area, proportion of greyspace (25m buffer), land cover types (Tree, Grass, Shrub, Grey, Border), and tree biomass to reduce multicollinearity.

Term	Estimate (β)	Std. Error	IRR (e^β)	IRR Lower 95% CI	IRR Upper 95% CI	p-value
(Intercept)	1.630	0.160	5.100	3.670	6.850	<0.001
PC4	-0.240	0.160	0.790	0.570	1.070	0.140

Table 4.36. Approximate effects of original environmental variables on above-ground invertebrate abundance, based on loadings from principal component axes used in the negative binomial GLM. Effects are expressed as estimated coefficients and incidence rate ratios (IRR), indicating the direction and magnitude of influence for each variable.

Variable	Approximate Effect	IRR
Area	-0.01	0.99
Biomass	0.01	1.01
Tree	-0.02	0.98
Shrub	0.00	1.00
Grass	0.01	1.01
Grey	-0.10	0.90
Border	0.14	1.15
Buffer (25m)	-0.09	0.92

Overall Patterns

Invertebrate abundance responses to site-level and landscape-scale variables varied markedly between above- and below-ground groups. Biomass emerged as the most consistent predictor, with higher vegetation biomass significantly reducing overall above-ground invertebrate abundance (IRR = 0.90, $p < 0.01$) and fly abundance (IRR = 0.995, $p < 0.01$), and exerting a marginal negative effect on Collembola (IRR = 0.997, $p < 0.10$). This suggests that dense vegetation may impede movement or reduce foraging efficiency for some invertebrate groups, while also altering microclimatic conditions that influence soil-dwelling taxa. Habitat heterogeneity (HGI Shannon) showed contrasting effects: it significantly reduced total below-ground abundance (IRR = 0.365, $p < 0.05$) yet strongly increased decomposer abundance (IRR = 2.132, $p < 0.05$), indicating that structurally complex habitats favour specialized soil functional groups but may suppress broader soil invertebrate assemblages. By contrast, site area and the 25 m buffer had limited or non-significant effects across all functional groups, highlighting that local vegetation structure and habitat characteristics play a more prominent role than broader landscape context in shaping invertebrate communities within urban greenspaces.

Species richness showed fewer significant relationships compared to abundance, indicating weaker environmental filtering for richness-based metrics. For above-ground taxa, no significant predictors were identified, with site area, biomass, and buffer effects all non-significant across pollinators and broader above-ground assemblages. In contrast, below-ground richness was more responsive to habitat heterogeneity: HGI Shannon significantly reduced total below-ground richness (IRR = 0.65, $p < 0.05$) but increased decomposer richness (IRR = 2.13, $p < 0.05$), suggesting that structurally complex habitats selectively promote functionally specialized soil taxa while limiting broader richness. Site area, biomass, and buffer effects on below-ground richness were non-significant, reinforcing the conclusion that fine-scale vegetation structure rather than landscape context drives below-ground richness patterns in urban greenspaces.

	Abundance							
	Above-ground	Pollinator	Wasp	Bee	Fly	Below-ground	Decomposer	Collembola
Area of site	0.90 (ns)	1.00 (ns)	1.00 (ns)	1.00 (ns)	–	0.998 (.)	1.00 (ns)	–
Biomass	0.90 (**)	0.999 (ns)	0.999 (ns)	–	0.995 (**)	1.002 (ns)	0.997 (.)	0.999 (ns)
HGI Shannon	–	1.005 (ns)	1.025 (ns)	–	–	0.365 (*)	2.132 (*)	–
Buffer (25m)	1.00 (ns)	–	1.010 (ns)	–	–	–	1.002 (ns)	–

	Species richness			
	Above-ground	Pollinator	Below-ground	Decomposer
Area of site	0.91 (ns)	1.00 (ns)	1.00 (ns)	1.00 (ns)
Biomass	0.91 (ns)	0.99 (ns)	1.00 (ns)	1.00 (ns)
HGI Shannon	–	–	0.65 (*)	2.13 (*)
Buffer (25m)	1.00 (ns)	1.00 (ns)	–	1.00 (ns)

Figure 4.31. *Landscape drivers of insect communities in inner-city greenspaces The effect of area of greenspace, the proportion of greyspace surrounding the site (25m buffer greyspace%), the land cover heterogeneity of the site and the biomass of trees within the site. Colours of cells indicate positive or negative response (or non-significant) effects tested using best fit GLM.*

When examining the approximate effect of land cover and land characteristics on invertebrate populations, abundance patterns were primarily influenced by vegetation biomass, with higher biomass associated with significant reductions in both overall above-ground abundance (IRR = 0.90, $p < 0.01$) and fly abundance (IRR = 0.90, $p < 0.01$). This suggests that dense vegetation may create physical or microclimatic barriers that reduce accessibility or foraging efficiency for certain above-ground invertebrates. Other structural variables, including tree, shrub, grass, and border area, showed no significant effects across any above-ground or below-ground groups, indicating that fine-scale compositional differences alone did not drive strong abundance changes. Below-ground abundance was largely unresponsive to all predictors, although decomposers and Collembola showed weak, non-significant trends in relation to grey area and buffer, implying a minimal role of immediate landscape context in shaping soil invertebrate abundance.

Species richness displayed similarly weak responses, with no significant predictors identified for above-ground taxa. Below-ground richness also showed no significant effects of biomass, vegetation composition, or buffer variables, although decomposers exhibited a slight but non-significant positive association with border area. These findings indicate that species richness is less sensitive than abundance to vegetation structure and landscape composition in urban greenspaces, suggesting that richness-based metrics may be buffered against fine-scale environmental variation, while abundance metrics more readily capture the effects of vegetation density and habitat complexity. Together, these results highlight that local vegetation biomass is the most consistent driver of invertebrate abundance, whereas richness is comparatively stable across urban habitat gradients.

	Abundance							
	Above-ground	Pollinator	Wasp	Bee	Fly	Below-ground	Decomposer	Collembola
Area of site	0.90 (ns)	0.93 (ns)	0.93 (ns)	0.94 (ns)	0.89 (ns)	0.99 (ns)	0.94 (ns)	0.99 (ns)
Biomass	0.90 (**)	0.97 (ns)	0.96 (ns)	0.98 (ns)	0.90 (**)	1.01 (ns)	0.97 (ns)	1.01 (ns)
Tree area	0.96 (ns)	0.91 (ns)	0.92 (ns)	0.92 (ns)	0.96 (ns)	0.98 (ns)	0.99 (ns)	0.98 (ns)
Shrub area	0.99 (ns)	1.10 (ns)	1.13 (ns)	1.05 (ns)	0.99 (ns)	1.00 (ns)	0.99 (ns)	1.00 (ns)
Grass area	0.88 (ns)	0.88 (ns)	0.82 (ns)	0.94 (ns)	0.88 (ns)	1.02 (ns)	0.99 (ns)	1.01 (ns)
Grey area	0.99 (ns)	1.11 (ns)	1.09 (ns)	1.13 (ns)	0.98 (ns)	0.90 (ns)	0.92 (ns)	0.90 (ns)
Border area	1.00 (ns)	0.85 (ns)	0.83 (ns)	0.86 (ns)	1.00 (ns)	1.17 (ns)	1.06 (ns)	1.15 (ns)
Buffer (25m)	1.00 (ns)	1.17 (ns)	1.17 (ns)	1.16 (ns)	1.00 (ns)	0.91 (ns)	0.88 (ns)	0.92 (ns)

	Species richness			
	Above-ground	Pollinator	Below-ground	Decomposer
Area of site	0.91 (ns)	1.00 (ns)	0.93 (ns)	0.92 (ns)
Biomass	0.91 (ns)	0.99 (ns)	0.99 (ns)	0.96 (ns)
Tree area	0.96 (ns)	0.94 (ns)	0.95 (ns)	0.98 (ns)
Shrub area	1.00 (ns)	1.12 (ns)	0.97 (ns)	0.98 (ns)
Grass area	0.87 (ns)	0.84 (ns)	1.06 (ns)	1.01 (ns)
Grey area	0.97 (ns)	1.03 (ns)	0.97 (ns)	0.94 (ns)
Border area	1.02 (ns)	0.94 (ns)	1.08 (ns)	1.03 (ns)
Buffer (25m)	1.00 (ns)	1.17 (ns)	0.91 (ns)	0.88 (ns)

Figure 4.32: *Landscape drivers of insect communities in inner-city greenspaces. The effect of area of greenspace, the proportion of greyspace surrounding the site (25m buffer greyspace%), the land cover of the site (Tree, Grass, Shrub, Grey, and Border area), and the biomass of trees within the site. Colours of cells indicate positive or negative response (or non-significant) effects tested using best fit GLM.*

Discussion

Our investigation indicates that increasing the area of planted sites had a negative effect on invertebrate abundance, while the composition and management of site vegetation showed mixed influences. The evidence highlights divergent habitat preferences between above- and below-ground invertebrates, with a notable negative effect of habitat heterogeneity. This suggests that the addition of small, diverse planting zones within inner-city greenspaces may inadvertently reduce invertebrate abundance, particularly where surrounding greyspace limits connectivity. Here, we evaluate how vegetation structure, habitat complexity, and surrounding urban context shape invertebrate communities, drawing out patterns in abundance and species richness across strata and functional groups. This analysis provides insight into the mechanisms driving invertebrate diversity in fragmented inner-city greenspaces and informs future approaches to habitat management.

Invertebrate species richness

Invertebrate species richness across all sites exhibited relatively weak associations with local habitat composition and complexity. This finding aligns with previous studies showing that urban invertebrate communities are often dominated by generalist species, capable of exploiting a wide range of resources, which thrive in disturbed and simplified environments (Langellotto et al., 2020; New et al., 2015). The high proportion of impervious greyspace within our study area likely exacerbates this trend, constraining colonisation by specialist taxa and homogenising local assemblages (McKinney, 2008). Greenspaces in inner-city settings are often highly isolated, with reduced plant diversity and microhabitat availability that filter out specialists reliant on continuous habitat networks (Aronson et al., 2017). As a result, the species detected likely reflect an assemblage skewed toward urban-tolerant generalists adapted to variable or degraded conditions (Filazzola et al., 2019).

Despite these constraints, patterns emerged across strata: below-ground richness exhibited negative associations with grass area, while above-ground richness declined in relation to planted border areas. Additionally, heterogeneity indices were negatively linked to below-ground richness but positively to above-ground richness, suggesting strata-specific responses to local vegetation configuration. These outcomes have important implications for urban planting schemes that prioritise floral display and ornamental diversity as a means of enhancing urban biodiversity (Hoyle et al., 2017). While such schemes often aim to sustain pollinator populations through staggered bloom periods and high floral resource provision, our findings indicate they may have unintended consequences for overall invertebrate richness, particularly among below-ground communities dependent on stable soil conditions and organic continuity (Lehmann et al., 2020).

Habitat heterogeneity impacts and mechanisms

The push toward biodiversity-promoting greenspace management has increasingly encouraged greater habitat complexity and plant species diversity, exemplified by high-profile initiatives such as those at Kew Gardens, which advocate the introduction of varied vegetation layers to maximise ecological benefits (Kew, 2023). While such approaches may deliver benefits in some contexts, they are often criticised for applying generalised principles without considering the distinct constraints of small, isolated urban greenspaces (Gaston et al., 2005; Ellis et al., 2020). Traditional ecological theory holds that heterogeneous habitats increase niche availability, promote functional complementarity, and enhance community stability (Dylewski et al., 2019; Mata et al., 2017). However, our results indicate that within small, inner-city sites, habitat heterogeneity negatively affects invertebrate abundance both above- and below-ground, challenging these assumptions.

One plausible explanation for this discrepancy lies in the disruption of habitat continuity within fragmented greenspaces. In small sites, increasing structural diversity may introduce sharp transitions in soil properties, vegetation cover, and microclimatic conditions (Tamme et al., 2010). For below-ground fauna, which are limited by soil structure and moisture availability, such abrupt variability may hinder dispersal and resource access, reducing the suitability of these sites for specialist taxa (Braschler et al., 2020; Fan et al., 2023). Above-ground, increased heterogeneity may amplify edge effects, exposing invertebrates to predators and fluctuating abiotic conditions (Wilmer et al., 2022). These combined pressures likely restrict the persistence of sensitive species, resulting in communities dominated by tolerant generalists. Furthermore, land cover heterogeneity may increase predator activity by expanding edge habitats and diversifying hunting opportunities for predatory birds, mammals, and even larger arthropods (Evans, 2020; Christopher, 2020). For prey species already stressed by habitat fragmentation and limited refugia, this elevated predation pressure may further depress abundance. This aligns with meta-analyses showing that structural complexity in small urban greenspaces can heighten predator-prey interactions, undermining biodiversity objectives (Tylianakis et al., 2008).

It is also important to consider scale: while heterogeneity is often beneficial in larger peri-urban or rural habitats where resource continuity is maintained, its effects in small urban patches appear fundamentally different. Microclimatic gradients—such as soil temperature fluctuations, surface moisture retention, and shading patterns—may vary substantially over only a few metres in fragmented greenspaces, creating unsuitable microhabitats within otherwise resource-rich sites (Edmondson et al., 2016). For below-ground invertebrates reliant on detrital pathways and microbial activity, such instability may reduce soil food web functioning (Lehmann et al., 2020).

Consequently, heterogeneity in urban greenspaces may act not as a driver of stability, but as a source of ecological disturbance that interacts with isolation and greyspace pressures to suppress abundance across both strata.

Above-ground vs below-ground drivers and contrasts

Our results reveal pronounced differences between above- and below-ground invertebrate responses to habitat features, underscoring the distinct ecological filters operating across strata. Above-ground abundance was most strongly associated with shrub cover, while below-ground abundance responded to a broader suite of variables, including tree cover, grass area, and planted borders. These findings reflect the contrasting ecological requirements of surface-active and soil-dwelling invertebrates and highlight the importance of microhabitat stability, vegetation architecture, and resource continuity.

For above-ground invertebrates, shrub cover appears to provide crucial microclimatic buffering and structural refuge. Shrubs create shaded, humid understories that reduce exposure to desiccation and temperature extremes while simultaneously supplying foraging opportunities through floral resources and associated arthropod prey (Forio et al., 2020; Kotze et al., 2022). This layered vegetation may also reduce predation risk by offering concealed foraging pathways and shelter. The lack of a similar relationship with tree cover likely stems from the dominance of urban tree species with sparse understories, which provide limited shelter or floral resources at ground level (Jensen et al., 2022). Trees may also reduce light penetration, suppressing undergrowth and diminishing habitat complexity where canopy cover is dense (Francini et al., 2021).

By contrast, below-ground invertebrates were influenced by a more complex set of habitat variables. Tree cover enhances soil quality by contributing organic matter through litterfall and root turnover, improving soil structure and nutrient cycling (Ding et al., 2021). Grass-dominated areas, however, tended to reduce below-ground richness and abundance, likely due to their shallow root systems, compaction-prone soils, and low litter production (Jacobs et al., 2022). Similarly, planted borders - often heavily managed and frequently disturbed - may disrupt soil faunal communities by altering moisture regimes and organic matter inputs (Marshall et al., 2023). These patterns support the view that soil invertebrates, with their limited dispersal capacity and reliance on stable detrital pathways, are more sensitive to local habitat stability than above-ground taxa (Braschler et al., 2020; Seaton et al., 2021).

Moreover, while above-ground invertebrates can exploit adjacent habitats through flight or surface movement, below-ground fauna remain confined by soil properties and hydrological barriers. Urban below-ground environments are often highly fragmented by buried infrastructure and impermeable surfaces, creating "islands" of soil habitat with limited connectivity (Blanchart et al., 2018). Consequently, below-ground communities may experience a "funnel effect," where isolation reduces gene flow and recolonisation potential, leading to depauperate assemblages even within otherwise suitable patches (Fan et al., 2023).

These divergent drivers reinforce that above- and below-ground invertebrates cannot be managed through identical interventions. Strategies effective for above-ground taxa, such as increasing shrub density to enhance microclimatic refuge, may offer limited benefits below-ground if soil continuity, moisture stability, and organic inputs are not simultaneously addressed. The distinct responses of these strata underscore the need for integrated greenspace planning that explicitly considers vertical habitat partitioning and the limited mobility of soil fauna relative to their above-ground counterparts.

Functional group responses

Functional group analyses revealed contrasting responses between pollinators and decomposers, highlighting how specific ecological roles shape sensitivity to urban habitat features. While overall pollinator abundance showed negative correlations with tree, shrub, and border areas, individual taxa within this group exhibited divergent patterns. Bees, for instance, displayed positive associations with tree cover, likely due to the nesting opportunities offered by cavities and sheltered microclimates, even in the absence of abundant floral resources (Prendergast et al., 2022). Tree bumblebee species (*Bombus hypnorum*) are well documented to utilise arboreal nesting sites, and similar cavity-nesting behaviour has been observed in other bee taxa, suggesting that structural features rather than floral density may drive tree associations (Ayers & Rehan, 2021).

Conversely, the overall negative relationship between pollinator abundance and shrubs or borders may reflect limitations in floral availability and accessibility. Dense shrub canopies reduce understorey light penetration, constraining flower production, while ornamental flowerbeds—though visually attractive—are often managed intensively, with frequent planting and soil disturbance reducing habitat stability (Hoyle et al., 2017). Moreover, these beds often contain non-native species with limited nectar or pollen value for local pollinators (Harrison & Winfree, 2015). Such findings suggest that increasing "attractive" plantings in urban greenspaces may not translate into meaningful resource gains for pollinator communities and, in some cases, may even impose ecological costs through habitat disruption (Ellis et al., 2020).

Wasp and fly abundance patterns further illustrate functional differentiation within pollinators. Wasps, many of which are predatory or parasitoid, rely on prey availability rather than floral resources and showed more complex relationships with vegetation heterogeneity (Wenzel et al., 2020). This highlights the indirect role of vegetation in structuring prey-rich habitats suitable for these taxa. Syrphid flies, in contrast, exhibited weaker associations with specific habitat features, consistent with their generalist foraging strategies and capacity to exploit diverse organic substrates across urban environments (Orford et al., 2015).

Decomposer responses were similarly shaped by resource availability but showed distinct habitat linkages. Decomposer abundance was negatively correlated with grass cover, reflecting the limited organic inputs, compacted soils, and homogenous root systems typical of these habitats (Dennis et al., 2020). Grass-dominated areas often lack the structural diversity needed to maintain soil moisture or support detrital pathways crucial for decomposer communities (Jacobs et al., 2022). In contrast, shrubs and planted flowerbeds positively influenced decomposer abundance, likely due to higher leaf litter deposition, increased soil shading, and more stable microclimatic conditions.

Key soil taxa such as Collembola demonstrated strong associations with organically enriched habitats, particularly planted borders receiving frequent organic inputs (Chauvat et al., 2021). These groups benefit from high-quality detrital material and stable soil environments, highlighting how management practices such as mulching or organic amendments can directly enhance decomposer communities (Kotze et al., 2022). However, intensive disturbance within these same borders - through frequent replanting or soil turnover - may offset some of these gains, illustrating the fine balance between organic enrichment and habitat stability in supporting soil fauna (Marshall et al., 2023).

Together, these functional group findings underscore that while general patterns of abundance reflect broad habitat filtering, finer-scale management interventions need to be tailored to ecological roles. Pollinator-friendly planting schemes must move beyond ornamental floral diversity toward structurally and temporally continuous resources, while decomposer-focused interventions should prioritise soil stability and organic enrichment over superficial increases in vegetation complexity.

Greyspace, isolation, and planning implications

The negative influence of surrounding greyspace on invertebrate abundance across both strata highlights the profound effect of urban matrix context on greenspace biodiversity. High levels of impervious surface reduce habitat connectivity, restrict dispersal, and isolate populations, creating ecological "islands" within the urban landscape (Braaker et al., 2014; Svenningsen et al., 2022). For above-ground invertebrates, this isolation diminishes access to external floral and nesting resources, while intensifying exposure to heat islands and microclimatic extremes (Aronson et al., 2017). Pollinator taxa capable of longer-distance foraging may partially offset these effects, but for many smaller or less mobile species, such as solitary bees, dispersal barriers imposed by surrounding greyspace reduce both local abundance and potential recolonisation following disturbance (Harrison & Winfree, 2015).

Below-ground invertebrates face even greater constraints. Impermeable surfaces disrupt soil hydrology, compact substrate layers, and prevent organic matter exchange across sites, further fragmenting an already patchy soil habitat (Braschler et al., 2020; Kotze et al., 2022). Urban infrastructure—such as buried utilities—compounds these barriers, restricting below-ground dispersal and limiting recolonisation pathways (Blanchart et al., 2018). These factors contribute to what has been described as a "funnel effect," where soil fauna in isolated patches experience population decline and local extirpation due to an absence of viable recolonisation routes (Fan et al., 2023).

Our fine-scale buffer analysis (25–100 m radii) reinforces that local-scale urban context exerts significant ecological influence. Unlike broader GIS-based landcover assessments, which often overlook small greenspaces (Stehman et al., 2019), this approach captures the immediate effects of adjacent greyspace, demonstrating that urban invertebrates are highly sensitive to proximal landscape structure. This is particularly critical for soil-dwelling communities, whose dispersal limitations mean even minor increases in surrounding impermeable surface can dramatically reduce connectivity and community resilience.

These findings align with calls for urban biodiversity planning to move beyond patch-level habitat enhancement toward matrix-level interventions that mitigate isolation (Braaker et al., 2014; Norton et al., 2019). Potential strategies include creating continuous "green corridors" linking fragmented greenspaces, implementing vegetated verges and street plantings to serve as dispersal stepping stones, and reducing impervious surface cover around ecologically valuable sites. Additionally, soil-specific interventions—such as connected rain gardens or permeable substrates—could restore hydrological continuity and enhance below-ground habitat connectivity, directly addressing the dispersal bottlenecks faced by soil fauna (Karimi et al., 2021).

Urban planning must also account for the compounded impacts of greyspace and vegetation heterogeneity observed in our study. The interaction between local habitat instability and broader-scale isolation suggests that even well-intentioned planting schemes may fail if implemented in highly fragmented, grey-dominated contexts. Instead, management should prioritise spatially cohesive networks of simpler but well-connected habitat patches to support colonisation, reduce ecological filtering, and buffer against stochastic population losses.

Implications

Our findings highlight the intricate interplay between vegetation structure, habitat heterogeneity, and surrounding urban context in shaping invertebrate communities within small inner-city greenspaces. While biodiversity-promoting strategies often focus on increasing habitat complexity and planting diversity, our results demonstrate that in isolated, highly urbanised environments, such interventions can have counterintuitive effects. Increased vegetation heterogeneity was associated with reduced invertebrate abundance across both above- and below-ground strata, likely driven by microclimatic instability, disrupted habitat continuity, and elevated predation pressure. These effects were particularly pronounced for below-ground taxa, whose dispersal is severely limited by soil fragmentation and greyspace barriers.

Species richness patterns further emphasise the dominance of urban-adapted generalists within these sites. The weak associations between richness and local vegetation composition, alongside consistent negative effects of surrounding greyspace, suggest that specialist taxa are excluded by the dual pressures of isolation and habitat instability. The filtering of invertebrate assemblages toward generalist-dominated communities in such contexts is well documented (McKinney, 2008; Aronson et al., 2017) and underscores the need for targeted, context-sensitive management that recognises the constraints imposed by urban form and scale.

Functional group responses reinforce this perspective. Pollinators exhibited divergent taxon-specific patterns, with cavity-nesting bees benefiting from tree-associated nesting resources, while wasps and flies responded more strongly to prey or organic matter availability. Decomposer abundance was closely linked to habitats enriched in organic inputs, such as shrub zones and planted borders, but remained sensitive to soil disturbance and homogenised grass cover. These findings illustrate that functional groups cannot be addressed uniformly; instead, interventions must align with ecological roles, combining floral resource continuity for pollinators with organic enrichment and soil stability measures for decomposers (Kotze et al., 2022; Chauvat et al., 2021).

Greyspace emerged as a pervasive constraint, exerting negative effects on both abundance and richness. Its influence at even the smallest buffer scales suggests that patch-level improvements will be insufficient without simultaneous efforts to alleviate surrounding isolation. Strategies that integrate greenspace enhancement with matrix-level planning—such as establishing linear corridors, permeable surfaces, and connected soil systems—are critical to restoring habitat continuity (Braaker et al., 2014; Karimi et al., 2021).

In light of these findings, we recommend that urban biodiversity planning shift from solely "in-patch" habitat manipulation toward an integrated approach that prioritises connectivity, stability, and simplicity. Rather than pursuing increasingly heterogeneous vegetation compositions within small, isolated sites, managers should focus on Creating networks of simpler, well-connected patches that support colonisation and reduce edge effects, whilst also enhancing soil habitat continuity, including organic amendments and moisture-retaining vegetation, to sustain below-ground fauna.

Connectivity of sites should be encouraged by embedding greenspace within permeable, vegetated matrices that facilitate dispersal. By aligning habitat design with the spatial and ecological constraints of inner-city greenspaces, it is possible to foster resilient invertebrate communities despite the challenges posed by urban fragmentation. Future research should build on this work by incorporating long-term monitoring and experimental manipulations of connectivity to disentangle the relative contributions of patch-level habitat structure and matrix permeability to urban invertebrate dynamics.

Chapter 5: Discussion

Within this thesis, I have investigated the characteristics of different forms of urban greenspace within Sheffield, using a multidisciplinary approach. I assessed the influence that land cover, site isolation and size, and greenspace type have on soil characteristics, plant growth, and invertebrate community structure. Chapter 2 evaluates the effects that different greenspace types have on soil characteristics and soil invertebrate communities, identifying differences between four greenspace types. It highlights the importance of greenspace type on invertebrate populations across a temporal pattern, with higher below-ground invertebrate abundance in garden soils in spring sampling timepoints, and higher below-ground abundance in summer in urban park grass soils. Woodland soils demonstrated lowest levels of heavy metals (Tresch et al., 2018; Edmondson et al., 2012). In Chapter 3, I examine the growth of five crop species within soils from four different greenspace types. The results indicate that park grass soils are least able to support plant growth, while woodland and garden soils supported the highest growth, demonstrating the influence of soil characteristics and greenspace type on plant performance, indicating that those beneath park grassland are of the poorest quality. In Chapter 4, I investigated the characteristics, including patch size, surrounding landscape, and vegetation structure, of inner-city greenspace sites and their associated invertebrate communities. The findings reveal that site homogeneity positively impacts invertebrate diversity, while site isolation and site area have negative effects, with vegetation exhibiting significant variability among different invertebrate species (Braaker et al., 2014; Sánchez-Galindo et al., 2021).

Despite increasing research interest in urban areas, important gaps still exist in our comprehension of the complex interactions and ecological dynamics present in urban settings. In particular, there remains a limited integration of soil ecological processes, vegetation structure, and invertebrate community dynamics across temporal and spatial scales. Many studies focus on single taxa or on isolated ecosystem services, without considering how these are interconnected. Furthermore, investigations often neglect seasonal variation or microhabitat-level drivers of soil biodiversity, limiting their capacity to inform urban management decisions (Huang et al., 2022; Farley & Fitter, 1999). Decisions about different forms of greenspace therefore continue to be made with incomplete evidence regarding their ability to support ecosystem services, particularly in relation to plant productivity, soil quality, and biodiversity support. This thesis helps to address this gap by combining soil chemistry, plant growth, and both temporal and spatial invertebrate sampling to offer a more integrated perspective.

Patterns in Invertebrate Diversity

The question of how invertebrate abundance changes temporally below ground is a key focus in understanding ecosystem dynamics and the health of urban environments. Below-ground invertebrates are central to nutrient cycling, organic matter decomposition, and soil structure maintenance, making them vital indicators of soil ecosystem functioning (Edmondson et al., 2012; Tresch et al., 2018). Their populations are highly sensitive to microclimatic shifts, resource inputs, and disturbance regimes, all of which are strongly modified in urban areas. We observed significant shifts in soil invertebrate populations over the course of the year, highlighting the importance of temporal changes within different greenspace types and the impact of seasonal variations on soil communities (Chapter 2). We observed that soils had a greater density of invertebrates beneath garden borders in February and April sampling timepoints, and beneath park grass and woodland soils in the July and August sampling timepoint (Chapter 2). These peaks coincide with periods of vegetation phenology (spring emergence and late-summer biomass accumulation), which likely influence litter deposition and root exudation, both of which are key resources for soil invertebrates (Huang et al., 2022; Farley & Fitter, 1999).

As discussed in Chapter 2, this pattern in invertebrate abundance across greenspace type was not associated with measured soil characteristics, such as levels of heavy metals found in woodland soils (Ulgiati et al., 2011; Tresch et al., 2018), but did show trends within greenspace types. This disconnect suggests that organic inputs and vegetation structure, rather than soil chemistry alone, may be primary drivers of invertebrate distribution, consistent with evidence that soil fauna are more responsive to plant functional traits and litter quality than bulk soil parameters (Sánchez-Galindo et al., 2021). However, these temporal results contrast with Chapter 4, where summer sampling of inner-city greenspaces showed strong associations between invertebrate abundance and border and grass areas, but with patterns modified by urban context factors (e.g., isolation, greyspace). This suggests that in highly urbanised settings, landscape context may override greenspace-type effects observed in Chapter 2. Additionally, Chapter 4 revealed a negative effect of tree area on below-ground invertebrate populations, a relationship also noted during spring timepoints in Chapter 2 but absent in summer — further demonstrating seasonality's role in moderating invertebrate-vegetation interactions.

This inconsistency between chapters underscores the influence of both habitat quality and scale. Urban greenspaces differ not only in size and vegetation but also in their surrounding matrix. As demonstrated in Chapter 4, increased site heterogeneity negatively affected invertebrate abundance, possibly due to disruption of microhabitat stability. Previous research has shown that microdetrivore communities prefer stable, resource-consistent microhabitats and can decline when exposed to highly variable vegetation and litter environments (Sánchez-Galindo et al., 2021). Furthermore, some decomposers exhibit litter specificity, requiring certain leaf chemistries to support their feeding and reinforcing the strong role of vegetation composition. These findings suggest that while homogeneous park grass may support larger but less diverse invertebrate populations due to stable litter inputs and minimal disturbance, structurally complex or frequently disturbed sites like gardens and allotments may fragment soil communities and inhibit biomass accumulation.

Urban Context Effects

The observed differences in invertebrate populations across urbanised sites can be attributed to the varying impacts of urbanisation and site scale. Urbanisation alters both the abiotic and biotic environment: impervious surfaces exacerbate the urban heat island effect, intensify runoff and soil compaction, and reduce infiltration rates, all of which constrain soil faunal activity (Smith et al., 2011; Rajagopal et al., 2013). Moreover, particulate and atmospheric pollutants can accumulate in soils, reducing organic matter inputs and altering microbial communities, indirectly affecting detritivore populations (Tresch et al., 2018). The urban heat island effect can create a warmer microclimate in these densely populated areas, potentially increasing thermal stability and facilitating higher reproductive success for invertebrates (Rajagopal et al., 2013). However, this warming also accelerates organic matter turnover, potentially depleting detrital food sources faster than they are replenished, particularly in nutrient-poor soils. Soil moisture deficits caused by higher evapotranspiration rates in built-up areas may further limit faunal survival, as many soil invertebrates are sensitive to moisture levels. These opposing pressures may also help to explain why in Chapter 4, invertebrate abundance in smaller, isolated greenspaces was often low despite their vegetative cover.

Additionally, increased pollution is often seen in inner-city greenspace soils, which could be a driver of below-ground abundance that was not assessed within Chapter 4. Heavy metals in particular can act as selective pressures on soil communities, reducing overall diversity but allowing metal-tolerant taxa to proliferate (Tresch et al., 2018). Woodland soils in Chapter 2 demonstrated the lowest heavy metal levels, suggesting that their reduced below-ground invertebrate abundance patterns are linked more closely to vegetation structure than soil toxicity. A future investigation into contamination data within inner-city greenspaces of Sheffield, to correspond with the data in Chapter 4, would allow this relationship to be explored more thoroughly.

Another potential driver of discrepancy could be differences in sampling timescale. Chapter 4 examined only one timepoint, limiting our understanding of temporal variation in invertebrate communities, whereas Chapter 2 explicitly captured seasonal shifts. Seasonal drivers, including temperature, precipitation, and organic matter variation, are known to strongly affect soil invertebrate communities (Huang et al., 2022; Farley & Fitter, 1999). Without multi-season sampling in Chapter 4, short-term patterns driven by summer microclimatic stability may obscure broader annual dynamics. Indeed, summer peaks in invertebrate activity observed in Chapter 2 are consistent with urban warming amplifying invertebrate reproductive rates during this season.

Greenspace Isolation and Connectivity

The novelty of investigating inner-city urban greenspaces, which are often small and surrounded by greyspace, lies in the uniqueness of their highly anthropogenic surroundings and the historical oversight within ecological studies. These environments, characterised by their small size and fragmentation, present severe barriers to dispersal and resource exchange. For taxa such as earthworms or other low-dispersal invertebrates, many of these greenspaces function as ecological islands, resulting in limited recolonisation potential following local disturbance events (Braaker et al., 2014). This isolation likely amplifies the negative effects of heterogeneity on below-ground abundance observed in Chapter 4, as microhabitat specialists cannot easily migrate between sites to track suitable conditions.

Our findings indicate that vegetation composition within these urban patches can be strongly influenced by their surrounding greyspace. While Chapter 2 found no significant differences in the amount of greyspace surrounding sampled sites, this was due to site selection criteria controlling for isolation. By contrast, Chapter 4 revealed a clear negative relationship between surrounding greyspace and soil invertebrate abundance, reinforcing the importance of landscape context in modulating greenspace quality. This underscores that habitat patches embedded in grey-dominated landscapes suffer reduced faunal movement and diminished cross-site connectivity. This implies that urban greenspace management must adopt a multi-scalar lens: local habitat quality alone cannot offset the isolation-driven constraints imposed by surrounding impermeable surfaces. Strategic placement of vegetated corridors, stepping-stone patches, and linear green infrastructure could alleviate dispersal limitations and improve community stability across the urban matrix.

Vegetation Planting and 'Wildlife-Friendly' Schemes

Current schemes to support ecosystem services and biodiversity within urban greenspaces typically focus on park and garden management for pollinator support or tree planting for canopy cover. These interventions have strong above-ground benefits but may conflict with below-ground stability if they increase habitat heterogeneity or disturbance frequency. Our plant bioassay (Chapter 3) further demonstrates that soil quality differences strongly influence vegetation success, implying that planting initiatives need to consider soil conditions to maximise ecological returns.

The lower growth observed in park grass soils aligns with their higher compaction and nutrient limitations, indicating that simply adding vegetation diversity without improving soil health may be insufficient (Tresch et al., 2018). Woodland and garden soils, with their richer organic matter and lower contamination, better support plant growth and below-ground fauna. This reinforces the idea that soil remediation and organic inputs should accompany planting efforts to foster resilient urban ecosystems (Tresch et al., 2018). This finding highlights a critical oversight in many urban greening projects: planting schemes that neglect underlying soil constraints risk poor establishment, reduced survival rates, and diminished ecological function. Our bioassay results (Chapter 3) clearly demonstrate that soils beneath park grasslands — despite their prevalence in urban areas - are poorly suited to support plant growth without remediation. This suggests that without targeted soil interventions, such as decompaction, organic matter amendments, or targeted microbial inoculation, the benefits of tree planting or pollinator-friendly schemes within these spaces may be severely constrained. Furthermore, failure to address soil limitations may lead to a mismatch between planted vegetation and the site's capacity to sustain it, resulting in increased maintenance costs, higher plant mortality, and reduced delivery of associated ecosystem services such as carbon sequestration, shading, and below-ground biodiversity support. Integrating soil restoration into urban greening strategies is therefore essential to ensure both the immediate success of plantings and the long-term resilience of urban ecosystems.

Limitations and Future Directions

While this thesis has expanded understanding of urban greenspace ecology, limitations remain. Firstly, the focus on a singular study city and the exclusion of temporal variation in the inner-city sampling (Chapter 4) constrain generalisability. Secondly, more comprehensive soil chemical profiling, including heavy metals and organic matter fractions, would refine understanding of soil–fauna relationships. Thirdly, more detailed vegetation surveys quantifying functional traits and litter inputs would elucidate plant–soil fauna linkages. Finally, experimental manipulations altering heterogeneity and connectivity in controlled urban settings could test causality.

Future work should also explore microbial communities as mediators between vegetation and soil fauna and incorporate functional trait approaches to assess urban ecosystem service potential. Given the predicted increases in urbanisation and climate change, understanding how to design and manage urban greenspaces for multifunctional biodiversity support remains a key priority.

Conclusion

This thesis has demonstrated the importance of greenspace type, site context, and temporal variation in shaping soil quality, plant performance, and invertebrate community structure within Sheffield's urban landscape. By integrating soil chemistry, plant bioassays, and both spatial and temporal sampling of invertebrates, this research provides a more holistic understanding of how urban ecological systems function and interact.

Our findings reveal that below-ground invertebrate abundance is strongly influenced by seasonal dynamics and vegetation-driven resource inputs rather than soil chemistry alone, while inner-city sites are further constrained by their isolation, size, and surrounding greyspace. This highlights the need for multi-scalar management approaches that consider both local habitat quality and broader landscape connectivity. Soil quality emerged as a critical but often overlooked factor: poor plant growth in compacted, nutrient-poor park grass soils illustrates how inadequate soils limit the success of planting schemes and undermine efforts to promote biodiversity.

These results underscore that enhancing urban biodiversity and ecosystem services requires coordinated interventions addressing vegetation, soil health, and spatial configuration. Initiatives such as organic soil amendments, targeted planting schemes matched to soil conditions, and the establishment of green corridors could mitigate the effects of isolation and improve habitat suitability for low-dispersal taxa. Furthermore, integrating temporal sampling and functional trait approaches would strengthen our ability to predict and manage ecosystem responses to seasonal shifts and urban stressors.

Ultimately, urban greenspaces hold significant potential to support biodiversity and deliver ecosystem services but realising this potential demands evidence-based management that accounts for the interactions among soil processes, vegetation structure, and invertebrate communities. By adopting an integrated framework and addressing key limitations in current practice, we can design resilient urban landscapes that sustain both ecological function and human well-being in the face of continued urbanisation and climate change.

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