University of Staffordshire

Of urban mice and men: How habitat structure, grassland management, and landscape configuration shape urban small mammal communities in Stoke-on-Trent, UK.

Baylee P. Jennings

A thesis submitted in partial fulfilment of the requirements for the degree of Doctor of Philosophy.

Department of Health, Education, Policing and Sciences, University of Staffordshire.

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I declare that the work in this thesis was carried out in accordance with the regulations of the University of Staffordshire and is original except where indicated by specific reference in the text. No part of the thesis has been submitted as part of any other academic award. The thesis has not been presented to any other education institution in the United Kingdom or overseas. Any views expressed in the thesis are those of the author and in no way represent those of the University.

Signed Baylee Phillip Jennings **Date** 07/09/2026

Abstract

Urban green spaces are increasingly expected to support biodiversity, yet they are often treated as ecologically equivalent despite variation in habitat structure, management and landscape context. Small mammals remain underrepresented in multiscale research within post-industrial cities. This thesis investigated how patch-scale habitat conditions and landscape configuration shape small mammal communities across Stoke-on-Trent and Newcastle-under-Lyme, Staffordshire, United Kingdom, and whether these relationships could support spatially explicit assessment of management scenarios.

Repeated Longworth trapping was undertaken across 21 green space patches over five survey periods, generating 7,200 nominal trap nights, equivalent to 6,966 adjusted trap nights after accounting for unavailable traps, and 765 captures. Responses were assessed using species richness, catch per unit effort (CPUE), Simpson's reciprocal diversity and species-specific CPUE for wood mouse (*Apodemus sylvaticus*), field vole (*Microtus agrestis*), bank vole (*Myodes glareolus*) and common shrew (*Sorex araneus*). GIS quantified habitat composition, woody linear vegetation, paths, watercourses, patch area and grassland management within patches, together with green space amount, green space count, nearest green space distance, hedgerows, watercourses and roads at 250 m and 500 m extents. Patch- and landscape-scale relationships were modelled separately before supported predictors were integrated into a spatial CPUE model evaluated using site-blocked cross-validation.

Urban green spaces were not ecologically equivalent. Woody linear density was the most consistent positive within-patch predictor, while path density and management intensity were negatively associated with CPUE. Patch area had limited explanatory value, and focal species showed contrasting responses to grassland, woody structure, watercourses and disturbance. At the landscape scale, subdivision and isolation were the most consistent negative predictors, while road density was negatively associated with CPUE. The combined model retained a positive association with woody linear density and negative associations with management intensity, green patch count and road density. Site-blocked cross-validation produced a Pearson correlation of 0.719 between observed and predicted CPUE. Across 598 environmentally comparable unsurveyed patches, mean predicted CPUE declined from 9.83 under low mowing to 6.85 under moderate mowing and 4.77 under intensive mowing. Urban habitat value therefore depends on internal structural quality, grassland management and matrix permeability rather than area or designation alone.

“Due to the current workload, the light at the end of the tunnel has officially been switched off”

Anonymous

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Chapter 1 Introduction and Literature Review

1.1 Introduction

Biodiversity encompasses variation within genes, among species and across ecosystems, together with the ecological interactions and functions arising from that variation (Rawat & Agarwal, 2015; Stange et al., 2020). Its decline therefore represents more than the loss of individual species, as changes in biological communities can alter resource use, food-web structure and the capacity of ecosystems to respond to environmental change (Elmqvist et al., 2013; Isbell et al., 2023). Land-use change remains one of the major drivers of these losses because it alters both the amount of habitat available and the condition and spatial arrangement of what remains (Cornford et al., 2023; Isbell et al., 2023). Within the United Kingdom (UK), prolonged conversion of land for agriculture, settlement and infrastructure has left many habitats as reduced or modified components of intensively used landscapes (Lawton et al., 2010; State of Nature Partnership, 2023; Isbell et al., 2023).

Urbanisation represents a concentrated form of this landscape modification, in which natural and semi-natural habitats are converted into impervious, sealed surfaces. Expansion of buildings, roads and other sealed infrastructure removes vegetation and subdivides remaining habitat, but urban landscapes are not uniformly devoid of ecological value. Parks, brownfield land, nature reserves, watercourses and remnant semi-natural habitats can retain substantial biodiversity within developed areas (Savard et al., 2000; Elmqvist et al., 2013; Lepczyk et al., 2017). Their ecological value cannot be inferred from their mapped extent alone. Urban green spaces vary internally in vegetation composition and structure, while their position within the surrounding

built matrix determines the degree to which they remain separated from, or structurally linked with, other habitats.

The management of associated vegetation within these green space patches adds another source of variation to the available habitats, often preventing natural succession in favour of aesthetics and recreation. In this thesis, management refers to repeated intervention in vegetation, including mowing and the maintenance or clearance of vegetated features. Mowing frequency can alter vegetation height, structure and plant composition without changing the area mapped as green space (Norton et al., 2019; Watson et al., 2020). Two parks of similar size and broad land cover composition can therefore provide very different ecological conditions. Frequently mown grassland may retain short, simplified vegetation, while less frequently managed areas can support taller grass and greater structural variation. Management must therefore be considered alongside habitat composition and spatial configuration when assessing the ecological value of urban green spaces (Lepczyk et al., 2017; Wood et al., 2018).

The surrounding landscape creates a related problem at a broader spatial scale, as urban green space patches are often isolated or disconnected. A green space containing suitable habitat may still be separated from other habitat by roads, sealed surfaces or breaks in vegetation, while linear vegetated features may retain some structural continuity through developed land (McDonald & St. Clair, 2004; Ascensão et al., 2016; Grilo et al., 2018). Urban ecological responses may therefore arise from conditions operating at more than one scale: the habitat and management occurring within an individual green space, and the composition and configuration of the matrix

surrounding it. Treating either mapped green space area or total surrounding green cover as a complete measure of habitat availability risks overlooking this variation.

Research on urban biodiversity has expanded substantially since the early twenty-first century, alongside growing interest in conservation, ecosystem services and the planning and management of green infrastructure (Elmqvist et al., 2013; Lepczyk et al., 2017; Zhao et al., 2022). The evidence base nonetheless remains uneven. Reviews of urban fauna research show a strong emphasis on birds and insects, frequent reliance on richness or diversity as community responses, and limited integration of local habitat conditions with wider landscape structure (Zhao et al., 2022; Rega-Brodsky et al., 2022; Wang et al., 2025). This leaves an important gap in understanding how less mobile terrestrial fauna respond to the combined effects of habitat structure, vegetation management and landscape configuration.

Small mammals provide a useful assemblage through which to examine these ecological relationships within highly modified environments. Rodents and shrews occupy important positions within terrestrial food webs and respond strongly to vegetation structure and ground-level cover (Pearce & Venier, 2005; Rowe & Terry, 2014; Torre & Balčiauskas, 2024). Their movement through urban landscapes can also be affected by breaks in vegetation and exposed or modified surfaces (McDonald & St. Clair, 2004; Ascensão et al., 2016; Grilo et al., 2018). Species within the assemblage differ in their use of grassland, woody vegetation, wetter habitats and more modified environments, meaning that community-level responses can conceal substantial ecological variation among taxa (Santini et al., 2019; Tranquillo et al., 2024). These characteristics make small mammals well-suited to examining whether apparently

similar urban green spaces differ in ecological value according to their internal structure and management, and whether those relationships change according to the landscape in which each patch is embedded.

This thesis addresses that problem by examining urban green spaces as heterogeneous habitat patches embedded within an equally heterogeneous urban matrix. The literature review first establishes the ecological basis for species–habitat relationships and disturbance before developing the landscape concepts used throughout the thesis, including patches, edges, fragmentation, isolation and connectivity. Island biogeography, metapopulation theory and the Habitat Amount Hypothesis are then considered in relation to urban landscapes, followed by the ecology of small terrestrial mammals and the use of Geographic Information Systems (GIS) and spatial modelling to quantify habitat and landscape relationships. These themes provide the basis for the research questions and hypotheses examining how habitat structure, vegetation management and wider landscape configuration combine to shape small mammal communities within urban green spaces.

1.2 Ecological systems, biodiversity and species-habitat relationships

The ecosystem concept provides a foundation for understanding how biological communities are organised. Tansley (1935) defined ecosystems as the interaction between organisms and their physical environment, recognising that biotic communities cannot be separated from the soils, climate, water and other abiotic conditions within which they occur. Ecosystems are dynamic: energy and materials move through food webs, organisms modify their surroundings, populations fluctuate and communities reorganise in response to environmental change. The persistence of

these processes contributes to ecological stability and the capacity of ecosystems to withstand disturbance (Elmqvist et al., 2013; Newman et al., 2024).

Biodiversity operates within this organisation at several interconnected levels. Genetic variation can influence the capacity of populations to respond to disease and environmental change, while species diversity determines the range of ecological strategies and interactions present within a community (Stange et al., 2020; Isbell et al., 2023). Variation among habitats and ecosystems produces further spatial differences in resources, environmental conditions and biological assemblages.

Spatial heterogeneity adds further variation to ecological systems, with differences in soil, hydrology, vegetation and disturbance creating a mosaic of environmental conditions. The habitat heterogeneity hypothesis predicts that structurally varied environments can support more species by providing a greater range of resources and microhabitats (Tews et al., 2004; Betts et al., 2014). Structural complexity does not benefit all species equally, however. Open-habitat specialists may decline as woody vegetation develops, while woodland-associated species may be disadvantaged where vegetation is repeatedly maintained at an early successional stage (Uchida & Ushimaru, 2014; Fuentes-Montemayor et al., 2020). Habitat heterogeneity is therefore ecologically valuable because it provides complementary conditions.

Habitats are commonly defined through the resources and environmental conditions that allow a species to survive and reproduce in a specific area (Hall et al., 1997; Dennis et al., 2003; Gaillard et al., 2010). The concept is species-centred, as the same mapped land cover can provide suitable habitat for one organism while offering little ecological value to another (Zeller et al., 2012; Betts et al., 2014). Species-habitat

relationships emerge from interactions between organism traits and environmental conditions, with body size, diet, physiology, reproductive strategy and dispersal capacity influencing how species use their surroundings (Betts et al., 2014; Santini et al., 2019). Mapped habitat classes therefore provide only a broad description of ecological conditions and cannot be assumed to represent equivalent habitat quality across taxa.

At fine spatial scales, microhabitat structure can directly influence survival and habitat use (Jorgensen, 2004). Vegetation height and density affect concealment from predators, leaf litter can provide nesting material and invertebrate prey, and moisture conditions can influence both food availability and thermal stress (Dickman & Doncaster, 1987; Tattersall et al., 2002; Fuentes-Montemayor et al., 2020). Disturbance is also fundamental to ecosystem organisation. Natural processes such as flooding, fire and herbivory can remove biomass, alter competitive relationships and initiate successional change, with their ecological effects depending on intensity, frequency, duration and spatial extent (Tews et al., 2004; Newman et al., 2024; Hu & Lima, 2024).

The biological responses used to evaluate these processes represent different aspects of community structure. Species richness describes the number of species present but does not account for their relative representation, while abundance reflects the number of individuals recorded (Hillebrand et al., 2008; Morris et al., 2014). In capture-based studies, abundance is more commonly expressed as relative abundance or activity because not every individual present will be detected (Efford, 2004). Diversity indices combine information on richness and relative abundance and may therefore

respond differently to environmental change. This thesis uses species richness, catch per unit effort (CPUE) and Simpson's reciprocal diversity as complementary community responses, alongside species-specific analyses where sufficient capture data were available.

1.3 Landscape ecology: patches, edges, corridors and matrices

Landscape ecology developed from the recognition that ecological processes are influenced by the amount or type of habitat present and by how that habitat is distributed across space (Turner, 1989; Wiens, 1989). This introduces a spatial component to habitat ecology, separating landscape composition, which describes the amount and type of land cover present, from landscape configuration, which describes its spatial arrangement. Patch size, shape, isolation and connectivity can therefore influence ecological responses even where the overall amount of habitat remains similar (Taylor et al., 1993; Fahrig, 2003; Jackson & Fahrig, 2012). The ecological importance of configuration is also dependent on the organism being considered. A landscape divided into several patches may remain readily traversable for a mobile species but function as a highly fragmented environment for a species that requires continuous ground cover or is reluctant to cross exposed areas (Wiens, 1989; Taylor et al., 1993). Landscape structure must therefore be considered from the scale at which organisms experience it.

The patch-corridor-matrix framework provides a useful basis for describing these spatial relationships. A patch represents an area that differs from its surroundings in a way that is ecologically meaningful, while edges form the transitions between adjacent habitat or land cover types (Murcia, 1995; Fahrig, 2003). Corridors describe linear

habitat features that may connect otherwise separated patches, and the matrix represents the wider landscape within which these components occur (Tischendorf & Fahrig, 2000; Baguette et al., 2013). These categories provide a convenient spatial framework but are not fixed ecological properties. Their meaning changes according to species and scale.

Vegetation classified as a discrete patch at one spatial resolution may contain several contrasting habitats at a finer resolution, while land classified as matrix may still contain resources or cover that can be used by some organisms. The framework is therefore most useful when its spatial categories are related to the ecological processes being investigated. Patch area has received specific attention because larger habitat patches are generally expected to contain more resources, support larger populations and encompass a wider range of environmental conditions (MacArthur & Wilson, 1967; Fahrig, 2003). Larger populations may also be less susceptible to stochastic local extinction, which may explain why patch area has frequently been associated with species richness and persistence.

Patch geometry adds further complexity because increasing area does not necessarily increase the amount of habitat protected from surrounding influences at the same rate. Patton (1975) drew attention to the relationship between patch shape and the amount of habitat edge created, while subsequent work established that conditions at habitat boundaries can differ substantially from those occurring further within a patch (Murcia, 1995). Compact patches generally contain a greater proportion of interior habitat, whereas narrow, elongated or irregular patches expose more of their area to adjoining land cover. These edge environments may differ in temperature, humidity,

vegetation structure, disturbance, predation and competitive interactions, although the direction and strength of such effects depend on the adjoining habitats and species involved (Murcia, 1995; Betts et al., 2014). Edges should therefore not automatically be treated as ecologically negative. Transitional habitats can provide additional resources for some species, while abrupt boundaries can reduce habitat suitability for others.

The spatial arrangement of patches becomes increasingly important once ecological processes extending beyond individual patch boundaries are considered. Landscape connectivity describes the extent to which the landscape facilitates movement between areas of suitable habitat (LaPoint et al., 2015; Liccari et al., 2022). Tischendorf and Fahrig (2000) highlighted the need to separate structural connectivity, describing the physical continuity and arrangement of habitat, from functional connectivity, describing the actual response of organisms to that structure. A vegetated feature may appear to link two habitat patches spatially, but this does not establish that individuals move through it. In contrast, an apparent gap in mapped habitat does not necessarily prevent movement if the intervening land cover remains sufficiently permeable to the species concerned (Baguette et al., 2013; Laliberté & St-Laurent, 2020).

Corridors provide one possible means through which structural continuity can be maintained. Linear vegetated features have long been recognised for their potential to connect larger habitat areas, although their ecological importance extends beyond movement alone (Lawton et al., 2010). Tattersall et al. (2002), for example, found that field boundaries supported substantial small mammal use, while work by Atkins et al. (2018) recorded small mammal activity more frequently in urban hedges than in non-

hedged linear boundaries. Such findings show that linear features can provide feeding opportunities, refuge and habitat in their own right as well as forming potential connections between larger patches. Their value cannot therefore be reduced simply to whether a line exists between two mapped areas.

The condition of a corridor is equally important; features such as hedgerow width, vegetation density, continuity and management can substantially alter ecological value (Hinsley & Bellamy, 2000; Atkins et al., 2018), while breaks created by roads or intensive management may reduce the continuity suggested by spatial mapping. Galantinho et al. (2020) similarly showed that transport infrastructure can have contrasting effects within the same landscape: roads can reduce connectivity while associated vegetated margins may still provide usable habitat. The matrix further modifies these relationships because the land separating habitat patches is rarely ecologically uniform. Traditional patch-based representations can imply a relatively simple division between habitat and non-habitat, but urban landscapes contain a heterogeneous mixture of surfaces with very different ecological properties (Lepczyk et al., 2017; Rupprecht et al., 2015; Van Helden et al., 2020). Zeller et al. (2012) demonstrated the wider importance of this issue in landscape modelling by showing that resistance varies according to both landscape characteristics and the biology of the organism being considered. Isolation is therefore not solely a function of straight-line distance between patches.

The importance of these landscape properties is ultimately dependent on spatial scale. Wiens (1989) argued that ecological patterns cannot be separated from the scale at which they are observed, and subsequent landscape studies have shown that

the apparent strength of habitat and configuration relationships can change according to the area over which predictors are measured (Jackson & Fahrig, 2012; Miguet et al., 2016). A relatively small analytical extent may characterise conditions immediately surrounding a focal patch, while a larger extent incorporates additional habitat, barriers and landscape features. Different species may also respond most strongly at different extents because their resource requirements and movement capacities differ.

1.3.1 Island biogeography and habitat islands

The theory of island biogeography developed from attempts to explain why islands of different sizes and degrees of isolation support different numbers of species. MacArthur and Wilson (1967) proposed that species richness reflects a dynamic equilibrium between immigration and extinction. Immigration was expected to decline as isolation increased because fewer dispersing individuals would reach distant islands, while extinction was expected to decline with increasing island area because larger islands could support larger populations and a greater range of ecological conditions. The theory therefore established area and isolation as two major spatial controls on community composition and provided a basis for predicting how species richness might change where habitat becomes spatially restricted.

Experimental work by Simberloff and Wilson (1969, 1970) provided an early test of these ideas by removing arthropod assemblages from mangrove islands and recording subsequent recolonisation. Species returned following defaunation, while the rate and sequence of recolonisation differed among islands and were influenced by their geographical separation from source populations. The experiments were influential because they demonstrated that community composition could change through

repeated immigration and local loss over relatively short periods, supporting the dynamic view of species richness proposed by MacArthur and Wilson (1967). They also helped move ecological thinking away from treating the species assemblage of an isolated habitat as a fixed property of that habitat.

The importance of island biogeography has since been extended beyond oceanic islands. Diamond (1975) applied its principles to conservation planning, arguing that protected areas surrounded by unsuitable land could experience processes comparable with those affecting true islands. Habitat loss could reduce the area available to populations while increasing the separation between remaining habitat fragments. Extinction risk would then rise where populations became smaller and immigration declined. Newmark (1987) provided empirical support for aspects of this argument in western North American protected areas, where mammal extinctions were greater in smaller reserves and patterns of species loss were associated with the degree of isolation. Area and isolation therefore became central concepts in the study of fragmented terrestrial landscapes as well as geographically isolated islands.

Applying island biogeography to habitat fragments also exposed limits in the original analogy. An ocean provides a relatively clear boundary between terrestrial islands, while terrestrial habitat patches are embedded within landscapes containing varying degrees of habitat suitability and permeability.

Urban green spaces brought these limitations into sharper focus because they can function as discrete habitat patches while remaining embedded within the highly heterogeneous landscape. Davis and Glick (1978) were among the earlier authors to apply island-biogeographic ideas to urban habitat, recognising that development can

separate populations and restrict the amount of habitat available within individual patches. The island analogy is useful at this level because increasing urban development can reduce opportunities for colonisation and increase the likelihood that local populations become isolated.

Island biogeography remains important to urban landscape ecology because it establishes clear expectations for the effects of habitat area and isolation, and its limits are equally useful. The theory therefore provides the starting point for examining spatially separated populations, while questions concerning local extinction, recolonisation and exchange among populations require a more explicit treatment of dispersal. Metapopulation theory developed directly around those processes.

1.3.2 Metapopulations, metacommunities and recolonisation

Metapopulation theory takes local extinction, recolonisation and dispersal as central processes in the persistence of spatially divided populations. Hanski (1998) described metapopulations as sets of local populations occupying separate habitat patches and connected through movement between them. Individual populations can disappear following demographic fluctuations, disturbance or deterioration in local habitat conditions, yet extinction from one patch does not necessarily represent loss from the wider landscape. Vacant habitat may subsequently be recolonised, and immigration into occupied patches can reduce extinction risk through the rescue effect. Persistence is therefore a property of both local population dynamics and exchange among populations across the wider network.

Patch area and isolation remain important within this framework because they can affect different stages of population turnover. Small populations are generally more

vulnerable to stochastic extinction, while increasing isolation can reduce opportunities for immigration and recolonisation (Hanski, 1998; Palmeirim et al., 2020). Metapopulation theory therefore develops the spatial expectations established by island biogeography by linking patch arrangement directly to population persistence and turnover.

Community composition introduces an additional level of complexity because species sharing the same landscape can differ in environmental requirements, dispersal ability and sensitivity to local extinction. Metacommunity theory considers communities distributed among habitat patches and connected through the dispersal of multiple species, with local composition emerging from the combined influence of environmental filtering, species interactions and movement among sites (Hagen et al., 2012; Gallo et al., 2017). Species sorting can produce different assemblages where environmental conditions vary among patches, while dispersal determines which members of the wider species pool are able to reach those conditions. Community structure across a habitat network can therefore reflect both differences among local environments and variation in the ability of species to move among them.

Gallo et al. (2017) provide direct urban evidence for these processes from mammal communities sampled across city parks, cemeteries, golf courses and natural areas in greater Chicago, USA. Community composition differed markedly among green space types, even though α-diversity was broadly similar across several of them, and green space type strongly influenced estimated colonisation and persistence rates. Most species also showed low probabilities of colonising city parks even where sites appeared available. Their results demonstrate that apparently comparable

components of an urban green space network can contribute differently to mammal persistence and community turnover, giving empirical support to the use of metacommunity approaches in urban landscapes.

The analyses in this thesis examine spatial variation in small mammal communities across urban green spaces and their surrounding landscapes. Colonisation, local extinction, gene flow and individual dispersal were not measured directly, so metapopulation and metacommunity theory provide expectations for interpreting spatial patterns without extending inference beyond the data collected. Habitat loss and fragmentation then become important because they can alter the number, size and spatial arrangement of habitat patches across which these population and community processes operate.

1.3.3 Habitat loss, fragmentation and the habitat amount hypothesis

Habitat loss and habitat fragmentation are closely linked but describe different landscape changes. Habitat loss reduces the total amount of suitable habitat, while fragmentation divides remaining habitat into smaller or more isolated components, often increasing edge exposure (Fahrig, 2003; Fischer & Lindenmayer, 2007). Their effects are difficult to separate because fragmentation frequently occurs alongside habitat removal. Fahrig (2003) argued that some effects attributed to fragmentation may reflect the amount of habitat lost, although the configuration of remaining habitat can still influence ecological processes where spatial separation affects access or persistence.

The Habitat Amount Hypothesis developed from this argument and proposes that species richness within a focal area is strongly influenced by the total amount of

suitable habitat in the surrounding landscape (Fahrig, 2013). Under this hypothesis, patch size and isolation should add relatively little explanatory information once surrounding habitat amount has been considered. The prediction therefore shifts attention from the focal patch alone to the quantity of suitable habitat available across the wider landscape.

The relative importance of habitat amount and spatial configuration remains debated, with evidence that patch size, isolation and fragmentation can retain ecological effects after habitat amount has been considered (Haddad et al., 2017; Fletcher et al., 2018; Rybicki et al., 2019). The number of mapped habitat patches is also not inherently a measure of either habitat availability or fragmentation: a greater count may reflect habitat creation or the subdivision of formerly continuous habitat (Liu et al., 2016; Fahrig, 2003; Fahrig, 2013). Measures of patch number, nearest-patch distance, roads and linear vegetation therefore represent different components of landscape structure and must be interpreted according to the ecological process each is intended to describe (Tischendorf & Fahrig, 2000; Zeller et al., 2012; Laliberté & St-Laurent, 2020).

Table 1.1 summarises these concepts, their operational representation within the thesis and the limitations attached to their interpretation. The empirical chapters apply them sequentially by assessing internal patch conditions and surrounding landscape structure separately before integrating the supported relationships.

Table 1.1 Principal ecological concepts underpinning the thesis and their operational interpretation.

Concept	Ecological Expectation	Relevance to Small Mammals	Thesis Representation	Interpretive Limitation
Heterogeneity & Filtering	Resources/structure permit persistence; unsuitable conditions exclude species.	Species differ in dependence on grass, woody cover, moisture, refuge.	Habitat proportions, woody linear density, path density, management intensity.	Complexity is not universally beneficial; responses are species-specific.
Island Biogeography	Larger, less isolated patches support more species/lower extinction risk.	Green spaces may operate partly as habitat islands within built land.	Patch area and nearest-green space distance.	The urban matrix is not a uniform "sea"; patch quality varies.
Metapopulation Theory	Dispersal/colonisation link populations across patches.	Isolation/matrix resistance affect local persistence and composition.	Patch isolation, green space subdivision, connective features.	Movement, colonisation, and turnover were not measured directly.
Habitat Amount Hypothesis	Richness depends more on surrounding habitat than focal patch size.	Mammals use resources distributed across neighbouring spaces.	Matrix green proportion (250m & 500m).	Mapped green cover combines varied habitat suitability.
Fragmentation & Resistance	Subdivision, isolation, and resistant covers reduce habitat access.	Roads/exposed surfaces constrain ground-level movement.	Green-patch count, nearest-patch distance, road density.	Associations indicate potential resistance, not observed movement.
Structural Connectivity	Continuous vegetation/linear features increase landscape permeability.	Hedgerows, scrub, watercourses provide refuge/movement.	Within-patch/matrix woody linear density, watercourse density.	Structural continuity does not prove functional connectivity.
Disturbance & Management	Uniform disturbance simplifies habitat/arrests succession.	Mowing reduces sward height, litter, food, and concealment.	Three-level management gradient and path density.	Management intensity is an applied proxy, not an exhaustive intervention measure.
Predictive Modelling	Sampled relationships support prediction at equivalent sites.	GIS allows consistent measurement across the urban network.	Integrated CPUE model, validation, range screening.	Prediction is associative; requires care beyond training domain.

1.4 Urbanisation as an ecological filter

Urbanisation alters ecological systems through land conversion, surface sealing, infrastructure development, pollution, artificial lighting and repeated human disturbance (Grimm et al., 2008; Pickett et al., 2011; Elmqvist et al., 2013). The intensity and distribution of these pressures vary substantially within and between cities because urban development reflects differences in built density, historical land use, vegetation retention and management (Grimm et al., 2008; Pickett et al., 2011). Urbanisation is therefore better understood as a collection of interacting environmental pressures operating across several spatial scales than as a single uniform gradient.

Changes in habitat availability form one component of that process; buildings, roads, and hardstanding replace vegetation and soils, while remaining habitats become embedded within landscapes with varying levels of development and disturbance. Urbanisation can also modify resource availability without removing habitat directly. Native vegetation may decline, while refuse, supplementary feeding, ornamental planting and irrigated vegetation create resources that some species can exploit (McKinney, 2008; Santini et al., 2019; Tranquillo et al., 2024). Species able to use these resources or tolerate repeated disturbance can remain abundant within developed areas, producing urban communities that differ in composition from those occurring in less modified landscapes.

Environmental filtering describes the process by which environmental conditions permit some species to persist while excluding others (McKinney, 2008; Santini et al., 2019). In cities, filtering includes human-generated pressures such as sealed surfaces,

altered temperatures, disturbance and vegetation management (Björklund et al., 2010; Ehlers Smith et al., 2018; Fung et al., 2024). Species able to exploit a broad range of resources or tolerate frequent disturbance may be favoured, while species dependent on vegetation structures or environmental conditions can decline. This can contribute to biotic homogenisation, although the outcome varies among cities because regional species pools, historical land use and local environmental conditions also influence which species occur (McKinney, 2008; Aronson et al., 2014; Santini et al., 2019; Kondratyeva et al., 2020; Wang et al., 2025).

Small mammal studies provide clear evidence that these responses vary among species. Łopucki et al. (2013) compared 15 urban habitat patches in Lublin, Poland, with 15 patches in the surrounding agricultural landscape and recorded lower species richness and diversity along the urbanisation gradient. The abundance of species associated with urban environments increased, with *Apodemus agrarius* showing the strongest urban association. Isolation of urban green patches also influenced community composition, while the management differences examined in that study produced no significant effect. High numbers of individuals could therefore occur within urban habitats even where the assemblage had become less diverse.

Klimant et al. (2017) also recorded declines in small mammal richness, diversity and relative abundance along an urban-rural gradient in Nitra, Slovakia. Seven species showed different associations with zones across the gradient, and relationships with local habitat structure were also identified. Urban position explained part of the variation in the assemblage, while environmental conditions within individual sites remained influential. The results show why broad urbanisation gradients can identify

community change without identifying all of the ecological processes responsible for it.

Evidence from Larson and Sander (2024) demonstrates that responses to urban development are not consistently negative. Small mammal diversity increased with the cover of some human-modified land covers in their study, including impervious surfaces and turf grass. Taller vegetation remained alongside streams and managed parkland at many sites, with prairie-associated species occurring more frequently where taller vegetation bordered mown areas and other species responding to canopy and shrub cover. The result differs from the declines reported by Łopucki et al. (2013) and Klimant et al. (2017), showing that the ecological effect associated with an urban land cover measure can depend strongly on the habitat retained within the surveyed area.

Variation among mammal species is also evident from broader comparative research. Santini et al. (2019), analysing 190 mammal species recorded in urban settlements, found associations between urban occurrence and life-history traits, although the relationships differed among taxonomic groups. Larger litter size was associated with urban occurrence across the dataset, while associations with brain size varied among groups. Tranquillo et al. (2024) also identified substantial variation among chipmunks, arboreal squirrels and flying squirrels in behavioural, physiological and morphological responses to urbanisation. Urban adapters, exploiters and avoiders provide useful broad categories (McKinney, 2008), yet considerable ecological variation remains within them.

Long-established cities also contain land uses that weaken any simple centre-to-edge ecological gradient. Brownfield sites, railway land, vacant plots and unmanaged vegetation can retain or develop structurally complex habitats within densely built areas, while green spaces farther from urban centres may consist largely of intensively maintained vegetation (Rupprecht et al., 2015). Differences among the studies of Łopucki et al. (2013), Klimant et al. (2017) and Larson and Sander (2024) are consistent with this heterogeneity. Urbanisation establishes the wider environmental context, while the ecological response recorded within a green space also depends on the habitat and management conditions present there. Examining urban green spaces as heterogeneous ecological units is therefore necessary before their position within the wider landscape can be interpreted.

1.5 Urban green spaces as heterogeneous ecological networks

Urban green space encompasses formal and informal vegetated land with widely differing ecological characteristics, including parks, playing fields, cemeteries, allotments, woodland, scrub, brownfield land and riparian habitats (Rupprecht et al., 2015; Taylor & Hochuli, 2017; Wood et al., 2018). Grouping these environments under a single land-use category can conceal substantial differences in vegetation composition and structure. Two green spaces with similar mapped areas may contain very different combinations of grassland, scrub, woodland and wetland habitat, creating different opportunities for feeding, refuge and reproduction (Beninde et al., 2015; Wood et al., 2018; Wang et al., 2025).

Informal green spaces demonstrate how strongly ecological value can diverge from land-use designation. Previously developed sites can pass through stages of

succession after disturbance ceases, producing combinations of grassland, tall herbs, scrub and developing woody vegetation (Rupprecht et al., 2015). Small mammal assemblages have been shown to change during such successional development as vegetation cover and resource availability alter through time (Churchfield et al., 1997; Jasiulionis et al., 2011; Balčiauskas et al., 2017). Formal green spaces can follow very different trajectories where frequent maintenance retains large areas of short vegetation. Ecological value therefore depends on the conditions developing within the mapped boundary as well as its area or administrative designation.

Green spaces also operate as components of a wider urban system. Gomes et al. (2011) found that characteristics of the surrounding matrix explained substantial variation in small mammal responses across their 15 urban habitat patches, including negative associations between impervious cover and species richness and *A. sylvaticus* abundance. Angold et al. (2006) similarly identified relationships between landscape structure and mammal persistence in Birmingham. These studies show that internal habitat conditions and wider landscape context can influence the same assemblage at different spatial scales.

Stoke-on-Trent provides a useful example of such heterogeneity. Historical development, land reclamation and the persistence of canals, railway infrastructure and remnant vegetation have produced a mixture of formal green spaces, regenerated land and linear green and blue habitats within the built environment (Stoke-on-Trent City Council, 2013, 2026; Staffordshire Wildlife Trust, 2024). Staffordshire Wildlife Trust (2024) identifies rivers and canals as important

components of the city's ecological network, while habitat condition and continuity vary along these features. Similar variation occurs within individual green spaces through differences in vegetation maintenance. Understanding the ecological value of the urban network therefore requires attention to management as a process capable of changing habitat structure even when mapped green space area remains stable.

1.6 Vegetation management as an urban disturbance regime

Vegetation management is a major source of ecological change within urban green spaces because repeated cutting, pruning, clearance and access maintenance alter vegetation structure through time (Aronson et al., 2017; Aguilera et al., 2019; Watson et al., 2020). Grasslands may be maintained as short amenity turf, woody vegetation can be repeatedly cut for access or safety, and vegetation surrounding paths can be removed or suppressed. These interventions influence vegetation height, density, composition and successional development, creating differences among green spaces that remain invisible in simple measurements of land cover area.

Mowing of grass provides some of the clearest evidence for these effects. Chollet et al. (2018) found that reduced mowing frequency increased plant taxonomic, functional and phylogenetic diversity in urban lawns, while Sehrt et al. (2020) recorded rapid increases in plant species richness after mowing was reduced. Associated faunal communities can respond to the structural and trophic changes that follow. Wintergerst et al. (2021) recorded higher insect abundance and diversity where urban lawns were only partly mown, complementing evidence of reduced plant and invertebrate diversity under frequent cutting (Garbuzov et al., 2015).

Changes in vegetation height, seed availability, litter and invertebrate abundance create several pathways through which mowing can influence small mammals. Repeated mowing can therefore alter physical cover and trophic resources simultaneously. Community responses are unlikely to be uniform where species differ in their dependence on grassland structure, woody cover or invertebrate availability.

Management intensity also interacts with ecological succession, as evidenced by Churchfield et al. (1997), who found changes in small mammal community structure and habitat use across grasslands of different successional ages, while Jasiulionis et al. (2011) and Balčiauskas et al. (2017) recorded changes during woodland establishment and meadow-forest succession. Early and intermediate stages can contain dense grass, tall herbs and developing scrub, while later stages become increasingly dominated by woody vegetation. Species associated with dense herbaceous cover can therefore respond differently from taxa associated with woodland or scrub as succession develops (Uchida & Ushimaru, 2014; Fuentes-Montemayor et al., 2020). Management that maintains a mixture of structural states across a site may support a wider range of habitat requirements than management that maintains uniform vegetation structure (Ofori et al., 2018; Norton et al., 2019; Hu & Lima, 2024).

Woody vegetation is also sensitive to management intensity, as hedgerows, treelines and scrub margins can provide feeding habitat, refuge and dense ground-level cover, while heavy cutting or removal of basal vegetation can substantially alter these

functions without removing the mapped feature itself (Hinsley & Bellamy, 2000; Tattersall et al., 2002).

Access management creates another source of repeated disturbance; formal paths replace vegetation with sealed or compacted surfaces, while informal routes can cause trampling and vegetation loss. Increasing path density can reduce vegetation continuity and expose more of a site to regular human activity (Tikka et al., 2001; Van Helden et al., 2020; Hu & Lima, 2024). Small terrestrial mammals may experience reduced cover and increased exposure where vegetation is repeatedly disturbed. Similar management occurs beyond formal green space boundaries through verge cutting and vegetation clearance along transport and watercourse corridors, linking local habitat condition with the quality of vegetation extending through the wider urban landscape (O'Sullivan et al., 2017; Galantinho et al., 2020).

Small mammal studies have established strong associations with vegetation composition and structural habitat features, while direct assessments of management intensity remain less common (Tranquillo et al., 2024; Dondina et al., 2025). The unresolved question is therefore how repeated vegetation management operates alongside internal habitat structure and the surrounding landscape in shaping urban small mammal communities. That question provides the ecological basis for examining mowing intensity alongside patch-scale habitat characteristics in the analyses developed later in the thesis.

1.7 Small mammals and the ecological function in urban green spaces

Small mammals provide a useful group for examining these relationships because small non-volant rodents and shrews respond to fine-scale vegetation conditions while also being affected by the arrangement of habitat beyond individual patches (Dickman & Doncaster, 1987; Pearce & Venier, 2005; Rowe & Terry, 2014). Within this thesis, the term small mammals refers specifically to the non-volant terrestrial rodents and shrews sampled using Longworth traps.

Trophic and non-trophic interactions connect individual organisms to broader community processes (Hagen et al., 2012; Isbell et al., 2023). Small mammals occupy several positions within terrestrial food webs, consuming plant material and invertebrates while also providing prey for avian and mammalian predators (Jones & Safi, 2011; Mendoza & Araújo, 2019; Torre & Balčiauskas, 2024). Changes in their abundance or community composition can therefore have effects extending beyond the small mammal assemblage itself.

Species grouped as small mammals differ substantially in their ecological requirements. *Microtus* voles are closely associated with dense herbaceous vegetation, *M. glareolus* commonly uses woody and scrub habitat, *A. sylvaticus* occupies a broader range of vegetation types, and *S. araneus* is strongly influenced by moisture, litter and invertebrate prey availability (Dickman & Doncaster, 1987; Fuentes-Montemayor et al., 2020; Dondina et al., 2025). Community-level measures can therefore conceal contrasting species responses. A site dominated by an adaptable species may produce a high capture rate without supporting the same range or balance of taxa as a structurally varied site. Richness, relative abundance, diversity

and species-specific responses are therefore more informative when considered together than any single measure alone (Pearce & Venier, 2005; Rowe & Terry, 2014; Morris et al., 2014).

Urban studies support this species-specific view, Dickman and Doncaster (1987) showed that small mammal species differed in their use of patchy urban habitats, while Baker et al. (2003) found that distributions within an urban area were associated with local habitat characteristics beyond the presence of green space itself. Parsons et al. (2018) recorded greater mammalian abundance, richness and diversity in moderately developed landscapes than in the least developed areas they surveyed, Ofori et al. (2018) showed that urban green areas could retain native small mammal biodiversity, and Rimbach et al. (2025) found little evidence of changes in small mammal community composition, diversity or evenness along an urbanisation gradient in Münster. The key element here is the question of how small mammals truly perceive and use the urban landscape, as suggestions from the above literature differ considerably from moderate disturbance intensity, to having little effect on the community structure at all.

The landscape concepts developed in Section 1.2 are also directly relevant to small mammals. McDonald and St. Clair (2004) found that artificial and natural barriers altered small mammal movement, Ascensão et al. (2016) detected road-barrier effects through genetic patterns, and Grilo et al. (2018) showed that road responses differed with species movement traits and specialisation. Galantinho et al. (2020) found that roads reduced overall landscape connectivity for *A. sylvaticus*, while vegetated road margins could still contribute to connectivity.

Small mammals have also been used as indicators of environmental change because changes in habitat structure can be reflected in abundance, distribution and community composition (Pearce & Venier, 2005; Rowe & Terry, 2014). Viteri and Hadly (2022), for example, found declines in richness and evenness with increasing human modification and differences between modern and historical assemblages. Their use as indicators still requires caution. Capture rates vary with season, weather, behaviour and trappability, populations can fluctuate independently of habitat change, and adaptable generalists may remain common after other species decline (Otto & Roloff, 2011; Harkins et al., 2019). Small mammal responses are therefore evidence of ecological pattern, not a direct measure of environmental quality in isolation.

Despite a long history of urban small mammal research, the group remains underrepresented within the wider urban biodiversity literature. Wang et al. (2025) identified a strong taxonomic bias towards birds and invertebrates in studies of terrestrial fauna within urban green spaces. Existing small mammal research has established that local habitat structure, urbanisation and landscape configuration can all influence communities (Dickman & Doncaster, 1987; Baker et al., 2003; Atkins et al., 2018; Larson & Sander, 2024; Dondina et al., 2025). Less well developed is research that examines vegetation management and internal habitat structure alongside the surrounding landscape for whole small mammal communities, especially within medium-sized UK cities. That gap defines the biological focus of this thesis.

1.8 Geographic Information Systems and spatial modelling in landscape ecology

The landscape concepts established above require measurements that can be applied consistently beyond individual survey locations. GIS can provide this capacity

by linking field observations with mapped environmental data and quantifying patch and landscape characteristics across large areas (Guisan & Zimmermann, 2000; Elith & Leathwick, 2009). Their ecological value lies in making spatial relationships measurable and repeatable.

Landscape metrics illustrate this problem; measures such as habitat amount, nearest-patch distance, road density and linear-feature density can represent aspects of habitat availability, isolation, disturbance or potential connectivity (Tischendorf & Fahrig, 2000; Zeller et al., 2012). The same metric can have different biological meanings among species, and the extent over which it is calculated can alter the relationship detected (Wiens, 1989; Jackson & Fahrig, 2012; Miguet et al., 2016). GIS, therefore, allows patch characteristics and surrounding landscape conditions to be examined separately and at multiple spatial extents, but buffer distances remain analytical scales unless independent evidence shows they correspond to movement or home-range distances.

Spatial modelling extends this framework by relating biological observations to environmental gradients and applying the fitted relationships beyond surveyed locations (Guisan & Zimmermann, 2000; Elith & Leathwick, 2009). This is useful where ecological surveys cover only a fraction of the landscape relevant to planning or conservation. Prediction, however, places stronger demands on a model than explaining the data from which it was developed. Models can reproduce sampled observations well while performing less effectively at new locations, especially where repeated observations from the same sites occur in both model fitting and evaluation. Validation that withholds complete sites provides a more realistic test of transfer to

unsurveyed patches because the withheld location contributes no observations to model fitting (Araújo et al., 2019).

Transferability also depends on the environmental conditions represented by the survey data. A statistical model can return predictions outside the range used for calibration, but the existence of a numerical prediction does not demonstrate ecological support for extrapolation. Predictions become more difficult to justify as combinations of habitat and landscape conditions depart from those represented by the sampled sites (Araújo et al., 2019; Franklin, 2023). Spatial prediction should therefore be accompanied by explicit assessment of the environmental domain within which the fitted relationships are being applied.

Mapped environmental data introduce a further limitation because broad land cover classes simplify habitat structure. A mapped grassland class cannot show all variation in vegetation height or management, and a mapped linear feature may not describe gaps, basal vegetation or condition. Similar limitations apply to road data, which describe the distribution of infrastructure without directly measuring traffic, mortality or behavioural avoidance. Structural datasets can therefore represent potential habitat pattern and potential connectivity. They cannot demonstrate how animals move through those features (Tischendorf & Fahrig, 2000; Zeller et al., 2012; Laliberté & St-Laurent, 2020). Increasing spatial coverage usually involves some loss of ecological detail, making field observations important for interpreting what the mapped variables represent.

For this thesis, GIS provides the link between the field survey and the wider urban green space network. Patch-scale measurements describe the surveyed green spaces,

matrix variables quantify the surrounding landscape using consistent rules, and the supported relationships can then be tested across unsurveyed patches that remain environmentally comparable to the survey data. The resulting predictions are intended for relative comparison and management scenario assessment, not as direct estimates of small mammal population size. This allows spatial modelling to support urban habitat planning while retaining clear limits on what can be inferred from mapped data alone.

1.9 Biodiversity policy, development and nature recovery in England

The applied relevance of this research has increased through recent environmental policy in England, UK. The Environment Act 2021 introduced statutory Biodiversity Net Gain (BNG) and Local Nature Recovery Strategies (LNRS). Subject to exemptions, planning permissions are required to deliver at least a 10% increase in biodiversity value relative to the pre-development baseline using the statutory biodiversity metric (Environment Act, 2021; UK Government, 2024). LNRS provide a spatial framework for identifying local priorities and opportunities for habitat creation and improvement across England (Department for Environment, Food and Rural Affairs, 2023).

These mechanisms sit within a broader shift from protecting individual sites towards restoring connected ecological networks. The Lawton Review summarised this through the need for wildlife sites to be better, bigger, more abundant and joined, recognising that protected areas cannot be considered independently of the landscapes around them (Lawton et al., 2010). Nature Recovery Networks and LNRS translate that principle into spatial planning by identifying existing habitat, restoration

opportunities and locations where intervention could strengthen ecological networks (Department for Environment, Food and Rural Affairs, 2023).

BNG provides a measurable mechanism for compensating for habitat loss and creating additional biodiversity value. Its statutory metric incorporates habitat type, distinctiveness, condition and strategic significance, which moves assessment beyond simple area replacement (Environment Act, 2021; UK Government, 2024). Habitat metrics nevertheless remain proxies for ecological function. They do not directly measure species-specific habitat use, community composition or functional connectivity, and newly created habitat may take years to develop the structure of established vegetation (Fuentes-Montemayor et al., 2020; UK Government, 2024). Evidence on how organisms respond to habitat structure and management can therefore complement metric-based assessment, where retention, enhancement and long-term management decisions determine whether habitat remains usable.

LNRS provide the wider spatial context by identifying where habitat creation or improvement may contribute most effectively to nature recovery. Their usefulness depends on ecological evidence capable of distinguishing mapped opportunity from demonstrated biological value. The contribution of this thesis is therefore to add organism-based evidence concerning how the condition and spatial setting of urban green spaces influence their ecological use.

1.10 Research gap and conceptual synthesis

The literature reviewed in this chapter establishes three linked components of urban habitat value: conditions within individual green spaces, recurrent vegetation management and the structure of the surrounding landscape. Each has an established

ecological basis, but they are often studied separately. Patch-scale studies can overlook the landscape surrounding the focal site, while landscape analyses can reduce internal habitat condition to broad mapped classes. Urban green spaces can therefore appear equivalent in spatial datasets despite differing substantially in the conditions experienced by organisms.

For small mammals, the evidence base shows that urban populations respond to local habitat structure, disturbance, isolation and surrounding land cover, but responses differ among species and community metrics (Dickman & Doncaster, 1987; Baker et al., 2003; Atkins et al., 2018; Larson & Sander, 2024; Dondina et al., 2025). The remaining gap is how internal habitat structure and management combine with wider landscape configuration to shape whole-community responses, and whether those relationships remain informative when considered together.

This thesis addresses this gap sequentially. Patch-scale analyses test how habitat composition, vegetation structure and grassland management are associated with community and species responses. Matrix analyses then test whether surrounding habitat amount, configuration, isolation and landscape structure provide additional information beyond the focal patch. Supported predictors from both spatial levels are subsequently combined to examine whether their relationships persist when considered together. GIS-based prediction then tests whether the integrated relationships can be transferred to environmentally represented unsurveyed green spaces and used to compare alternative management scenarios.

The central research problem is whether small mammal communities within urban green spaces are better explained by considering local habitat and management

together with the surrounding landscape, and whether the resulting evidence can support spatially explicit urban habitat management without extending inference beyond the environmental conditions represented by the survey data. Figure 1.1 links this conceptual framework to the research questions and hypotheses that follow.

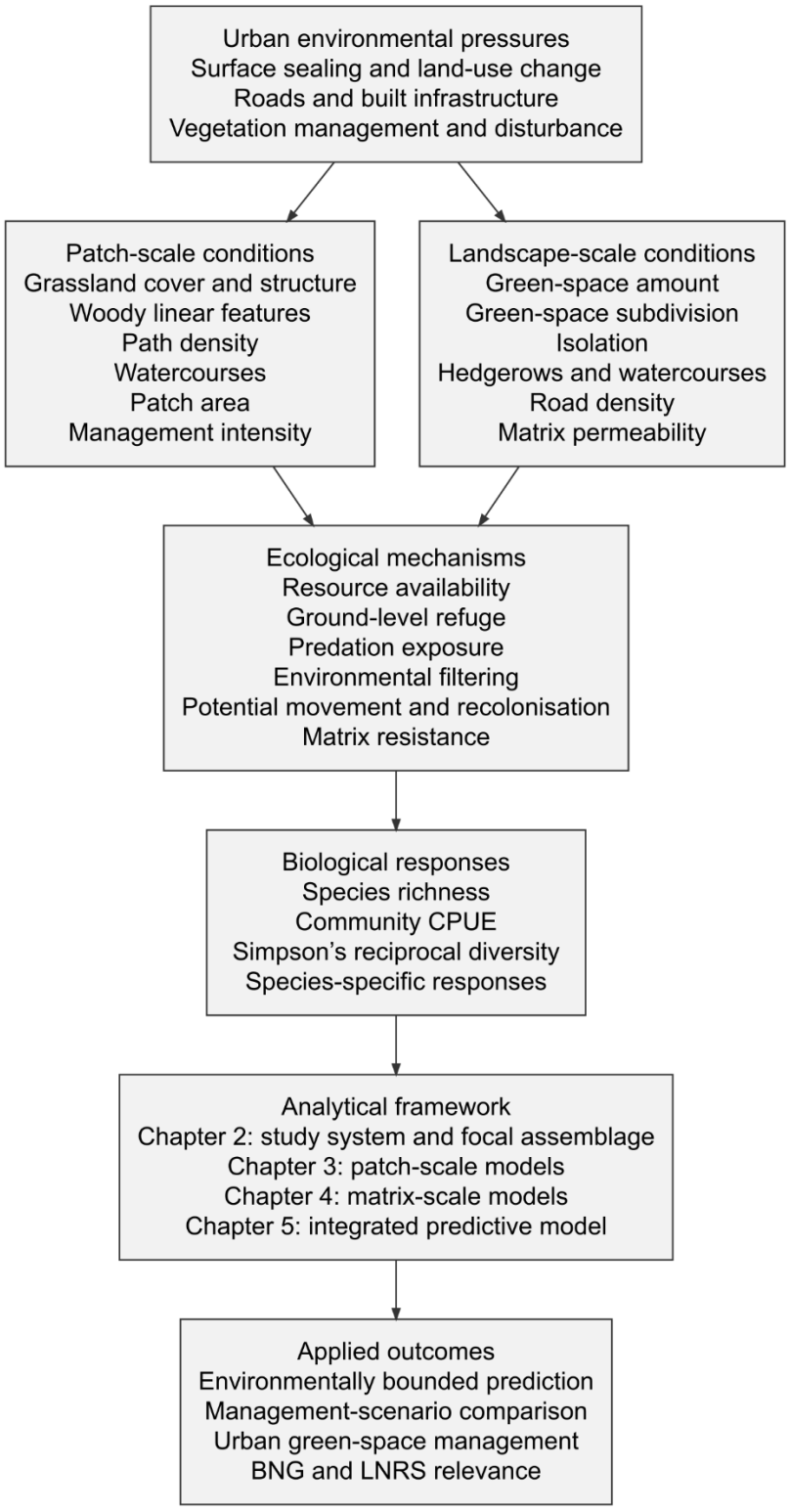


Figure 1.1 Conceptual and analytical framework of the thesis. Urban environmental pressures influence internal patch conditions and surrounding landscape configuration, which operate through ecological mechanisms including resource availability, refuge, disturbance, environmental filtering and matrix resistance. These processes are evaluated through community and species responses before being integrated into spatial prediction and management-scenario assessment.

1.11 Research aim, objectives, questions and hypotheses

1.11.1 Overarching aim

The overarching aim of this thesis is to determine how local habitat structure and management interact with wider landscape configuration to shape small mammal communities within urban green spaces, and to translate these relationships into a spatially explicit evidence base for urban habitat management and nature-recovery planning.

1.11.2 Research objectives

1. To characterise the urban landscape of Stoke-on-Trent and Newcastle-under-Lyme and establish the locally occurring and appropriate focal small mammal assemblage.

2. To determine how patch-scale habitat composition, vegetation structure and grass management (mowing) intensity influence small mammal species richness, CPUE, Simpson's reciprocal diversity and focal-species responses.

3. To assess how surrounding matrix composition, green space subdivision, isolation, connective linear features and road density influence small mammal communities at 250 m and 500 m spatial extents.

4. To integrate supported patch- and matrix-scale predictors into a spatially explicit CPUE model, validate its performance and compare predicted CPUE under low, moderate and intensive grassland-management scenarios across environmentally comparable unsurveyed green spaces.

1.11.3 Research questions

1. Which characteristics of individual urban green space patches explain variation in small mammal community and species responses?

2. How does the composition, configuration and potential permeability of the surrounding urban matrix influence small mammal communities within those patches?

3. Do local patch conditions and wider landscape structure retain complementary effects when considered within an integrated model?

4. Can these relationships support environmentally bounded spatial prediction and comparison of grassland-management scenarios across an urban green space network?

1.11.4 Hypotheses

H1: Patch-scale habitat structure, habitat composition and management intensity influence small mammal richness, Catch Per Unit Effort (CPUE) and community diversity within urban green spaces. Richness and CPUE are expected to be higher where dense ground-level vegetation and woody linear structure provide refuge, and lower where management and path density increase disturbance and exposure.

H2: Focal species differ in their responses to patch- and landscape-scale habitat features according to ecological specialisation, specifically their dependence on grassland, woody cover, and connected habitat. Community-level responses are therefore expected to conceal contrasting species-specific associations.

H3: Wider landscape structure contributes explanatory information beyond conditions within the focal patch. Small mammal community responses are expected to decline with increasing isolation, green space subdivision and road density, while greater habitat amount and structurally continuous green or blue linear features are expected to support higher CPUE and community diversity.

H4: Patch-scale habitat structure, grassland management and landscape configuration retain complementary effects within a combined predictive model. Predicted CPUE is expected to decline from low- to moderate- and intensive-mowing scenarios, but the magnitude of management effects will vary among patches according to their structural condition and surrounding matrix.

1.11.5 Thesis structure

Chapter 2 establishes the study area, urban green space network and baseline small mammal assemblage. It explains why Stoke-on-Trent and Newcastle-under-Lyme provide an appropriate post-industrial case study, reviews the local species records and describes how pilot work informed the final focus and survey method. Chapter 3 tests patch-scale effects of habitat composition, vegetation structure, paths and grassland management on community and focal-species responses. Chapter 4 extends the analysis into the surrounding matrix, examining habitat amount, fragmentation, isolation, roads, hedgerows and watercourses at 250 m and 500 m. Chapter 5 integrates supported predictors from both scales, validates a combined CPUE model and applies it to environmentally comparable unsurveyed patches under alternative mowing scenarios. Chapter 6 synthesises the findings, evaluates the thesis-wide hypotheses and considers the implications for urban green space management, BNG, LNRS and future research.

Chapter 2 Study Area and Baseline Small Mammal Assemblage

Chapter 2 establishes the ecological and spatial framework for interpreting the small mammal communities investigated in this thesis. By situating Stoke-on-Trent within its broader geographic and post-industrial context, the chapter defines the unique 'polycentric' urban matrix that underpins the study area. The composition and structural configuration of the landscape are quantified using high-resolution land cover data, providing a baseline for identifying available habitat types. This environmental context is then integrated with a review of the local small mammal assemblage, which justifies the selection of focal species based on their ecological requirements and sensitivity to urban stressors. Ultimately, this chapter provides the theoretical and methodological bridge between the general principles of urban ecology established in Chapter 1, the patch-scale and landscape-scale community and species analyses presented in Chapters 3 and 4, and the integrated predictive framework developed in Chapter 5.

2.1 Introduction

The United Kingdom (UK) has experienced substantial biodiversity loss due to long-term habitat destruction and fragmentation, with urbanisation identified as a key driver of these changes (Boakes et al., 2024; Mainwaring et al., 2024). Urban expansion and associated land-use change have driven the conversion of natural and semi-natural habitats into sealed surfaces and intensively managed environments, reducing both habitat availability and connectivity (State of Nature Partnership, 2023; Mainwaring et al., 2024). As a result, urban landscapes are increasingly characterised by fragmented habitat patches embedded within a human-dominated matrix, where

ecological processes are constrained, and species persistence depends on local habitat conditions and landscape structure (Fung et al., 2024; Mainwaring et al., 2024; Cooke et al., 2026).

Urban green spaces have the potential to mitigate some of these impacts by providing refugia for wildlife within otherwise inhospitable environments (Jorgensen & Gobster, 2010; Concepción et al., 2015; Dover, 2015; McDonnell & Hahs, 2008; Mainwaring et al., 2024). However, these spaces are highly variable in their composition, structure, and management, resulting in heterogeneous ecological conditions across the urban landscape (Concepción et al., 2015; Aronson et al., 2017; Wood et al., 2018). Differences in habitat structure, quality, patch size, and connectivity are therefore likely to influence the distribution and abundance of species, especially for taxa with limited dispersal ability, such as small mammals (Concepción et al., 2015; Parsons et al., 2018; Fung et al., 2024).

Small mammals represent an ecologically important group within urban ecosystems, functioning as key components of food webs and having been used as indicators of habitat and environmental condition (Dickman & Doncaster, 1987; Łopucki et al., 2013; Klimant et al., 2017). Their responses should not, however, be assumed to represent all components of biodiversity. Despite their ecological importance, relatively little is known about how small mammal communities respond to variation in urban habitat structure and management. Stoke-on-Trent provides a suitable case study for examining how landscape structure, management and spatial configuration influence small mammal communities.

2.2 Background of the study area: Stoke-on-Trent

Stoke-on-Trent is a post-industrial city located in the West Midlands of England, UK (Figure 2.1). It forms a unitary authority within the county of Staffordshire and is directly adjacent to the borough of Newcastle-under-Lyme, which is included within the study area due to its spatial continuity and shared urban landscape. Stoke-on-Trent and Newcastle-under-Lyme cover approximately 304.5 km^2 together and support a human population of approximately 390,000 individuals (Office for National Statistics, 2022).



Figure 2.1 Location of the study area (Stoke-on-Trent) within the United Kingdom. The red indicator identifies the North Staffordshire conurbation, situated in the West Midlands, England.

The study area is characterised by its historical development as an amalgamation of six towns: Stoke, Hanley, Burslem, Tunstall, Longton, and Fenton, which were formally unified into a single city in 1910. The city was widely known as “The Potteries”, reflecting its role as the centre of the UK ceramics industry for over two centuries (Leach, 2024). This industrial legacy has left a distinct imprint on the urban fabric, including the persistence of bottle kilns, canals, wharves, and disused rail infrastructure (Tandy, 1975). The spatial extent of the study area used for subsequent survey and analysis is shown in Figure 2.2.

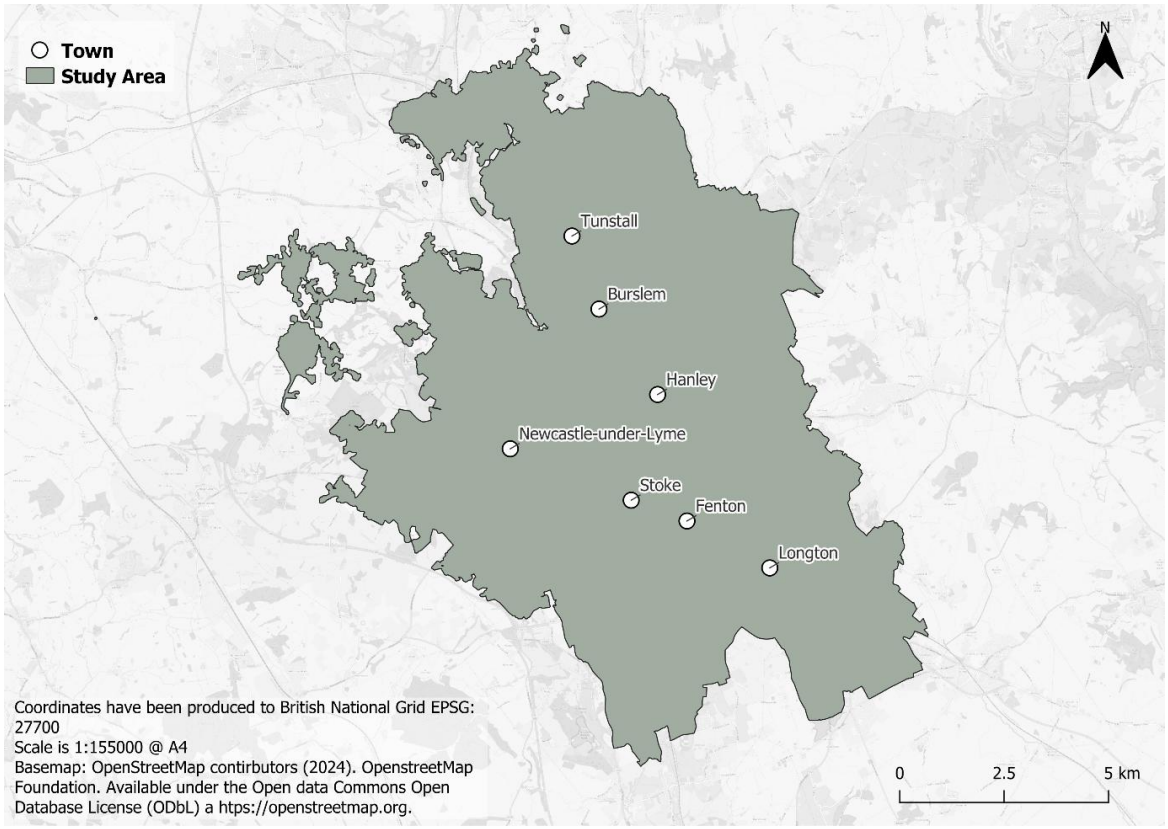


Figure 2.2 Spatial extent of the study area, comprising the urban conurbation of Stoke-on-Trent and Newcastle-under-Lyme. The map identifies the polycentric urban matrix of the region, including the six historical town centres of Stoke-on-Trent (Tunstall, Burslem, Hanley, Stoke, Fenton, and Longton) and the neighbouring centre of Newcastle-under-Lyme. Base mapping and town centre locations are derived from Ordnance Survey data © Crown copyright (2024).

In addition to ceramics, the region historically supported extensive coal mining and steel production. The decline of these industries during the early twentieth century

left large areas of derelict, contaminated land. By 1967, Stoke-on-Trent contained the greatest extent of derelict land of any district in the UK (Tandy, 1975). Subsequent regeneration and land reclamation initiatives have transformed sections of this post-industrial landscape, converting former industrial sites into residential areas, commercial developments, and urban green spaces. As a result of these regeneration efforts, Stoke-on-Trent is now considered one of the UK's greener cities, with approximately 49% of its land area designated as open green space (Bixby et al., 2015; Stoke-on-Trent City Council, 2013, 2026).

These processes have resulted in a highly heterogeneous urban landscape; a mosaic where habitat patches are non-uniformly distributed in terms of their size, quality, and composition (Bixby et al., 2015; Stoke-on-Trent City Council, 2013, 2026). The resultant green spaces are therefore characterised by a mosaic of remnant habitats, managed green spaces, and redeveloped land, creating substantial variation in habitat structure and quality across the study area. In this context, habitat structure and quality refer to the vegetation's three-dimensional physical architecture. Remnant habitats are patches of the original semi-natural vegetation (e.g. ancient woodland or heathland) that persisted throughout the urbanisation process, as opposed to "reclaimed" or "created" spaces.

This spatial configuration is characteristic of habitat loss and fragmentation, through which continuous or formerly connected habitats are reduced and subdivided within a modified matrix (Fahrig, 2003; Fischer & Lindenmayer, 2007). These processes can produce smaller patches, greater isolation and increased exposure to habitat boundaries. Edge effects describe changes in environmental conditions, population

responses or community structure associated with transitions between adjacent habitats, but their magnitude and direction depend on the contrast between land cover types and the requirements of the species concerned (Murcia, 1995; Ries et al., 2010). Abrupt transitions between vegetation, amenity grassland, paths, roads and sealed surfaces are therefore relevant to the ecological interpretation of urban green spaces. However, external boundary geometry alone cannot establish that an edge effect has occurred, and edge influence was not measured directly in this thesis.

Comparable post-industrial urban areas across the UK, including Sheffield, Manchester and Nottingham, contain patchworks of remnant habitat, reclaimed land and regenerating vegetation produced through repeated cycles of industrial expansion and decline (Breuste et al., 2008; Jorgensen & Keenan, 2012). In Stoke-on-Trent, the resulting network of green and blue features provides visible structural continuity in some parts of the city, although physical adjacency does not necessarily translate into realised animal movement or functional connectivity. Ecological responses may therefore depend on both the condition of individual green spaces and the composition, configuration and resistance of the surrounding urban matrix (Cushman & McGarigal, 2002; Bennett & Mulongoy, 2006).

The strategic importance of this urban system is further highlighted by its role in emerging nature recovery frameworks. According to the Staffordshire Wildlife Trust (2024), the city’s network of rivers and canals represents a 'signature landscape' of fundamental importance to the natural environment. These features are designated as a backbone for the local Nature Recovery Network (NRN) and the forthcoming Local Nature Recovery Strategy (LNRS) (Davies, 2021; Stoke-on-Trent City Council,

2025), positioning Stoke-on-Trent as a critical site for examining how development and nature can be aligned within a biodiversity crisis context. These frameworks aim to mitigate the impacts of urbanisation by enhancing functional connectivity across the fragmented landscape (Environment Act, 2021; Stoke-on-Trent City Council, 2025). The intersection of this industrial legacy and the city's polycentric development, comprising six distinct towns, has created a fragmented but 'green' urban fabric. This includes successional habitats in which vegetation communities continue to develop following disturbance (Prach & Walker, 2011; Tischew et al., 2014), including within heavily urbanised parts of the city. This complicates a simple core-to-periphery gradient and instead produces a spatially variable mosaic of built-up land, managed green spaces and regenerating habitats (Luck & Wu, 2002; Alberti, 2005).

By offering a diverse range of successional stages and habitat configurations within a compressed spatial extent (Stoke-on-Trent City Council, 2013, 2026), Stoke-on-Trent provides an appropriate context for examining how small mammal communities respond to differing grassland management intensities. The city therefore functions as a case-study system in which the influence of internal patch structure, grassland management and wider matrix configuration can be evaluated using community and species capture responses. This provides an empirical basis for considering how urban development and nature-recovery objectives may be aligned within a medium-sized UK city.

2.3 Spatial framework for landscape-scale modelling

While the primary objective of this thesis was to investigate the factors influencing small mammal communities within surveyed urban green spaces, the wider aim was to develop an ecological framework capable of extending these findings across the broader urban landscape. Therefore, the study area was considered as an interconnected network of habitat patches embedded within a heterogeneous urban matrix.

A unified spatial framework of urban green spaces was developed using the study area boundary described previously in combination with Ordnance Survey Open Greenspace data and supplementary green space datasets obtained from Stoke-on-Trent City Council and Natural England (Ordnance Survey, 2026a; Ordnance Survey, 2026b). Individual green space polygons were merged, cleaned and clipped to the final study area extent to produce a standardised dataset of green space patches in QGIS v.3.28 using the British National Grid coordinate system (EPSG:27700). This dataset represented all potential habitat patches within the study area along with those selected for ecological survey and formed the spatial framework used throughout subsequent analyses. The resulting network comprised 1,280 individual green space polygons varying substantially in area, habitat composition and surrounding landscape context.

Twenty-one of the 1,280 green space polygons were selected for repeated small mammal surveys using the stratified random sampling approach described in Chapter 3, with the remaining polygons retained for later spatial analysis and prediction. The remaining 1,259 unsurveyed polygons were retained as the prediction

frame. A *patch* represents the mapped extent of an individual green space, although these areas frequently contained mixtures of grassland, woody vegetation, water, paths and other built or ecological features. Habitat composition and structural characteristics were quantified separately within each patch.

Habitat composition was derived from the UK Centre for Ecology & Hydrology (UKCEH) Land Cover Map 2021 (Marston et al., 2022). The dataset provides a nationally consistent land cover classification that could be applied using the same definitions across all surveyed green spaces and the wider network of unsurveyed sites. Its use therefore allowed habitat variables to be extracted reproducibly across the complete spatial framework and avoided reliance on field-derived classifications that could not feasibly be replicated across the full urban green space network. This standardisation reduced the fine-scale ecological detail available from field surveys, but ensured that the habitat variables used in subsequent spatial analyses were directly comparable among surveyed and unsurveyed green spaces. Additional measures of habitat structure and landscape configuration were calculated from mapped linear features, roads, watercourses and the distribution of surrounding green space (UK Centre for Ecology & Hydrology, 2000; Broughton et al., 2024; Ordnance Survey, 2024; Environment Agency, 2025). Variables required for the later spatial analyses were also extracted across the unsurveyed green spaces where the corresponding spatial data were available.

The same spatial framework allowed environmental variables measured at the 21 surveyed patches to be calculated across the wider green space network and later used for spatial prediction. Relationships between small mammal CPUE and the

environmental predictors identified at surveyed sites could then be applied to unsurveyed patches with comparable environmental conditions. Combining field observations with spatial environmental datasets is increasingly used in landscape ecology to extend ecological inference beyond sampled locations (Wade et al., 2015; Kirk et al., 2023; Cork et al., 2026). In this study, predictions were confined to mapped urban green spaces and did not include residential areas, commercial development or major transport infrastructure. Figure 2.3 shows the 21 surveyed patches within the wider network of green spaces used in the subsequent patch, landscape, and predictive analyses.

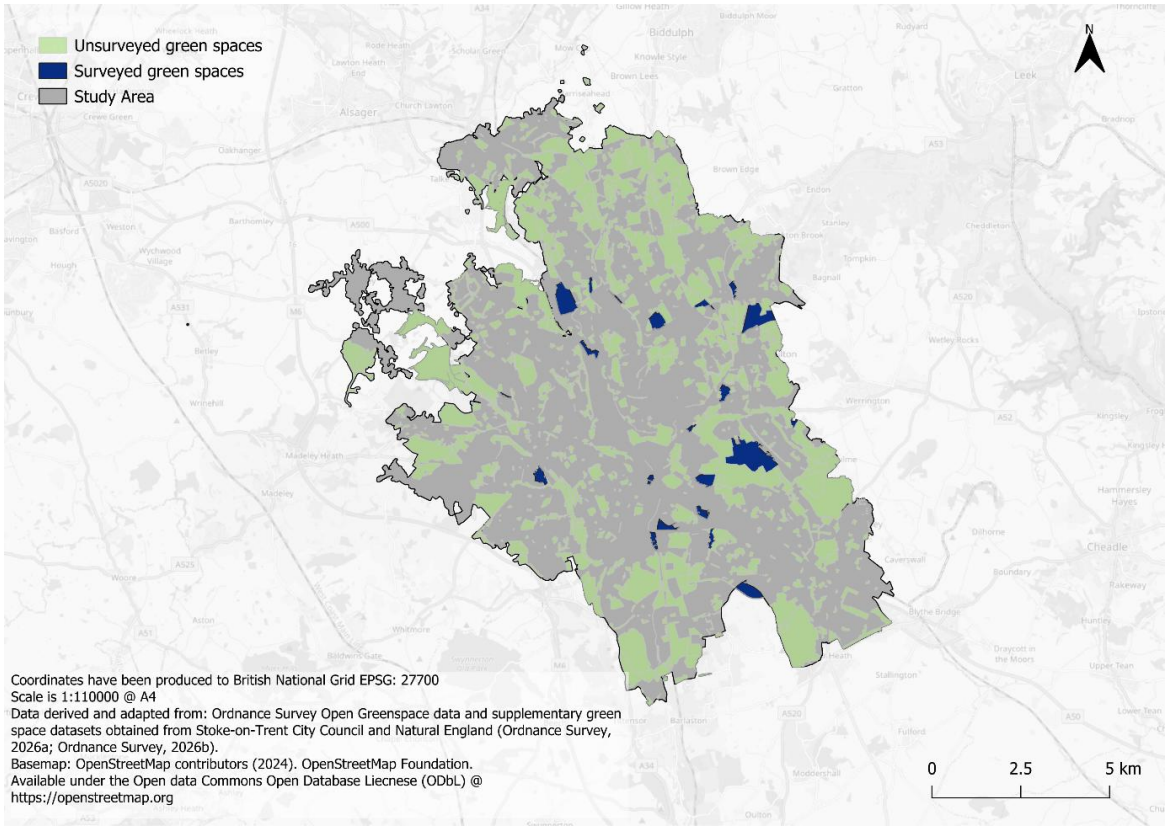


Figure 2.3 Unified urban green space network within the Stoke-on-Trent and Newcastle-under-Lyme study area. The complete network of mapped green space polygons formed the common spatial framework used throughout this thesis. Twenty-one surveyed green space patches (dark blue) were selected for repeated small mammal surveys, while the remaining mapped green space patches (light green) formed the candidate prediction framework for subsequent spatial modelling.

2.3.1 Development of the final analytical framework

The pilot phase informed the selection of the field survey method and the development of the spatial and analytical framework used in the subsequent chapters. Initial consideration was given to variables describing green space area, habitat composition, management, internal structure, external geometry and connectivity. Variables were retained when they represented a probable ecological process, could be measured consistently across surveyed and unsurveyed green spaces, and could be interpreted in relation to the habitat requirements of the focal small mammal assemblage. This approach is consistent with modern landscape ecology guidance that emphasises the selection of organism-relevant variables and cautions against interpreting spatial metrics independently of scale, landscape definition, and ecological mechanisms (Jackson & Fahrig, 2012; Miguet et al., 2016; Wang et al., 2025).

The final framework therefore separated conditions operating within each mapped green space boundary from those operating in the surrounding matrix. Chapter 3 evaluates patch-scale habitat composition, internal structural features, grassland management intensity, path density, watercourse density and patch area. Chapter 4 evaluates surrounding habitat amount, green space subdivision, isolation, roads, hedgerows and watercourses within 250 m and 500 m matrix rings. This structure prevents local habitat condition and wider landscape context from being treated as interchangeable representations of connectivity and allows their respective associations with small mammal responses to be evaluated before supported predictors are integrated in Chapter 5.

Shape index (Patton, 1975), was considered during the early development of the spatial framework but was not retained in the revised analyses. Perimeter-area shape metrics can be informative where a mapped polygon represents a relatively homogeneous habitat and external geometry provides a defensible proxy for boundary exposure or remaining interior habitat (McGarigal & Marks, 1995). The green space polygons used in this thesis differed as they were heterogeneous green spaces containing variable mixtures of grassland, woodland, scrub, water, paths, hedgerows, and internal habitat boundaries. The mapped boundary of a green space did not always correspond to a single ecological edge, as it could follow roads, ownership limits or the source mapping itself (Dennis et al., 2019). Shape index could therefore assign similar values to sites with very different internal habitat arrangements, including patches with internal subdivision despite having relatively simple outer boundaries (Grafius et al., 2018).

Shape index was not retained because the mapped geometry of the green spaces could not be linked consistently to one ecological process across the focal species. Recent synthesis of urban-fauna studies found no consistent directional relationship for shape index or edge density, while habitat quality, vegetation cover, isolation, connectivity and surrounding matrix conditions showed clearer associations (Wang et al., 2025). The final framework therefore focused on variables with clearer ecological meaning: habitat proportions described broad resource availability; woody linear, path and watercourse densities described internal structure; management intensity represented mowing disturbance (Watson et al., 2020); patch area represented spatial extent (Beninde et al., 2015); and green-patch count, nearest-green space distance, hedgerow density and road density described subdivision,

isolation, permeability and resistance within the surrounding landscape. Edge ecology remains relevant to urban fragmentation, but edge effects were not tested directly.

2.4 Landscape and habitat context of the study area

The city's green and blue infrastructure includes public parks, amenity grasslands, cemeteries, nature reserves and allotments, alongside canal corridors and railway verges (Staffordshire Wildlife Trust, 2024). These habitats occur within a fragmented urban matrix and vary in size, structure and ecological value, with some forming relatively isolated patches and others connected through linear vegetation or waterways (Concepción et al., 2015; Liordos et al., 2021; Fung et al., 2024).

Past industrial land use has also influenced the present ecological character of many green spaces. Redevelopment of previously contaminated land has produced novel soil profiles that affect vegetation composition and succession, contributing to variation from intensively managed vegetation through to areas undergoing passive regeneration (Stoke-on-Trent City Council, 2013, 2025, 2026). Land cover classification therefore describes the physical surface present, while habitat value depends on whether those conditions meet the requirements of a particular species (Nedd et al., 2021; Osman et al., 2022).

The study area covers 132.79 km^2 (132,791,100 m^2) based on the UKCEH Land Cover Map 2021 (LCM2021) (Marston et al., 2022). Suburban land was the dominant classification (47.40%), followed by improved grassland (23.02%), urban land (16.69%), broadleaved woodland (8.03%) and arable and horticultural land (3.47%);

all remaining LCM2021 classes each accounted for less than 1% of the classified area (Figure 2.4).

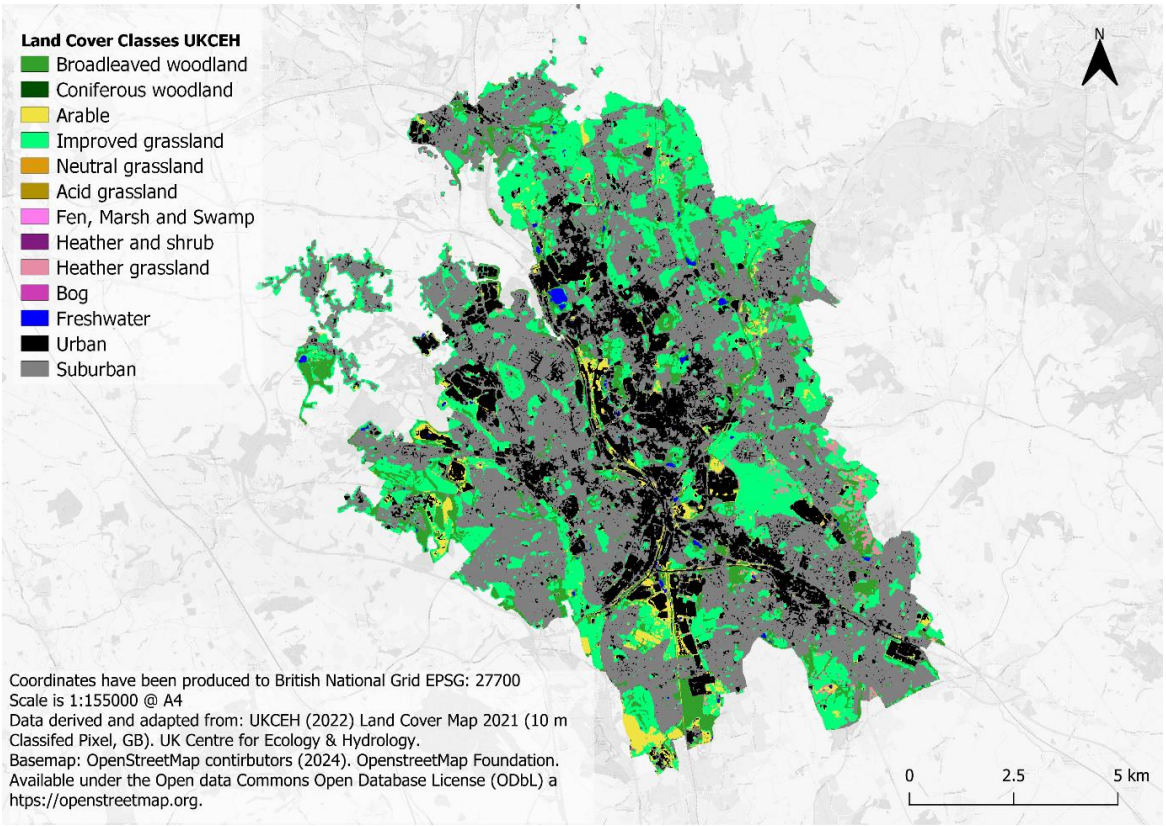


Figure 2.4 Broad Land cover classification of the study area. Data derived and adapted from: Marston et al. (2022) Land Cover Map 2021 (CBF 10 m Classified Pixel, GB). UK Centre for Ecology & Hydrology. Base mapping is derived from Ordnance Survey data © Crown copyright (2024).

Table 2.1 presents the land cover classes by percentage cover in reference to the study area extent.

Table 2.1 Proportional composition of land cover classes within the study area based on UKCEH Land Cover Map 2021.

LCM2021 Class	Land cover Class	Pixels	Area (km^2)	Area (%)
1	Broadleaved woodland	106,665	10.67	8.03
2	Coniferous woodland	23	<0.01	<0.01
3	Arable and horticulture	46,099	4.61	3.47
4	Improved grassland	305,677	30.57	23.02

LCM2021 Class	Land cover Class	Pixels	Area (km^2)	Area (%)
5	Neutral grassland	5	<0.01	<0.01
7	Acid grassland	228	0.02	0.02
8	Fen, marsh and swamp	161	0.02	0.01
9	Heather	103	0.01	0.01
10	Heather grassland	12,063	1.21	0.91
11	Bog	2	<0.01	<0.01
14	Freshwater	5,878	0.59	0.44
20	Urban	221,643	22.16	16.69
21	Suburban	629,364	62.94	47.40
Total	**Classified Area**	**1,327,911**	**132.79**	**100.00**

While land cover data provides a quantitative baseline, the ecological value of these classes is often determined by their planning status and conservation priority (Gaston et al., 2005; Angold et al., 2006). In practice, it is the designation and management of sites within these classes that determines their ecological value, with vegetation structure, disturbance regime, and resource availability shaping habitat suitability for wildlife (Gaston et al., 2005; Angold et al., 2006; Macgregor et al., 2022).

Within Stoke-on-Trent, this management context is shaped by the Local Plan, the city's primary strategic framework for development (Environment Act, 2021; Stoke-on-Trent City Council, 2013, 2025, 2026). The draft plan defines green and blue infrastructure largely in terms of public amenity, recreation, and associated environmental benefits. However, responses by the Staffordshire Wildlife Trust (2024) highlight that this emphasis may overlook the city's underlying ecological value, which is concentrated within post-industrial land, canal corridors, and remnant habitat patches that have emerged through passive regeneration. Brownfield sites are

indicated to support structurally diverse vegetation assemblages that contribute disproportionately to urban biodiversity, despite often being undervalued in planning contexts (Cox & Rodway-Dyer, 2023; Macgregor et al., 2022). However, as these habitats frequently fall outside formal green space designations, they are subject to uneven protection and management, increasing their vulnerability within the wider urban matrix (Staffordshire Wildlife Trust, 2024).

Responsibility for green space management within the study area is divided between local authorities, private landowners, transport bodies and other stakeholders, meaning that sites with similar broad habitat types are not necessarily managed in the same way. Differences in mowing, vegetation clearance, planting, access and the extent to which areas are left to regenerate can produce marked variation in vegetation structure, disturbance and habitat suitability between neighbouring green spaces (Lawton et al., 2010; State of Nature Partnership, 2023). Within Stoke-on-Trent, this variation occurs alongside the existing fragmentation of the urban landscape, with land cover classifications often grouping green spaces that differ in their management and ecological condition.

The Trent and Mersey Canal runs through much of the urban area alongside a network of smaller watercourses and drainage systems, adding linear green and blue features to the terrestrial habitat patches described above (Stoke-on-Trent City Council, 2013; Davies, 2021; Staffordshire Wildlife Trust, 2024; Stoke-on-Trent City Council, 2025, 2026). Riparian vegetation along these features can differ from adjoining managed grassland and built land, with banks supporting combinations of tall herbs, scrub, trees and ground-level cover. Railway networks have further ecological relevance due

to the vegetation running alongside them, which is often less simplified than roadside vegetation (Tikka et al., 2001; Penone et al., 2012). Where this vegetation remains continuous, canal and watercourse margins may provide small mammals with feeding areas, refuge, and routes through sections of the urban matrix where surrounding land uses offer little cover (Gurnell et al., 2019). Figure 2.5 shows the main water course networks throughout Stoke-on-Trent and the railway networks, which are often considered to be important features for the mobility of urban fauna.

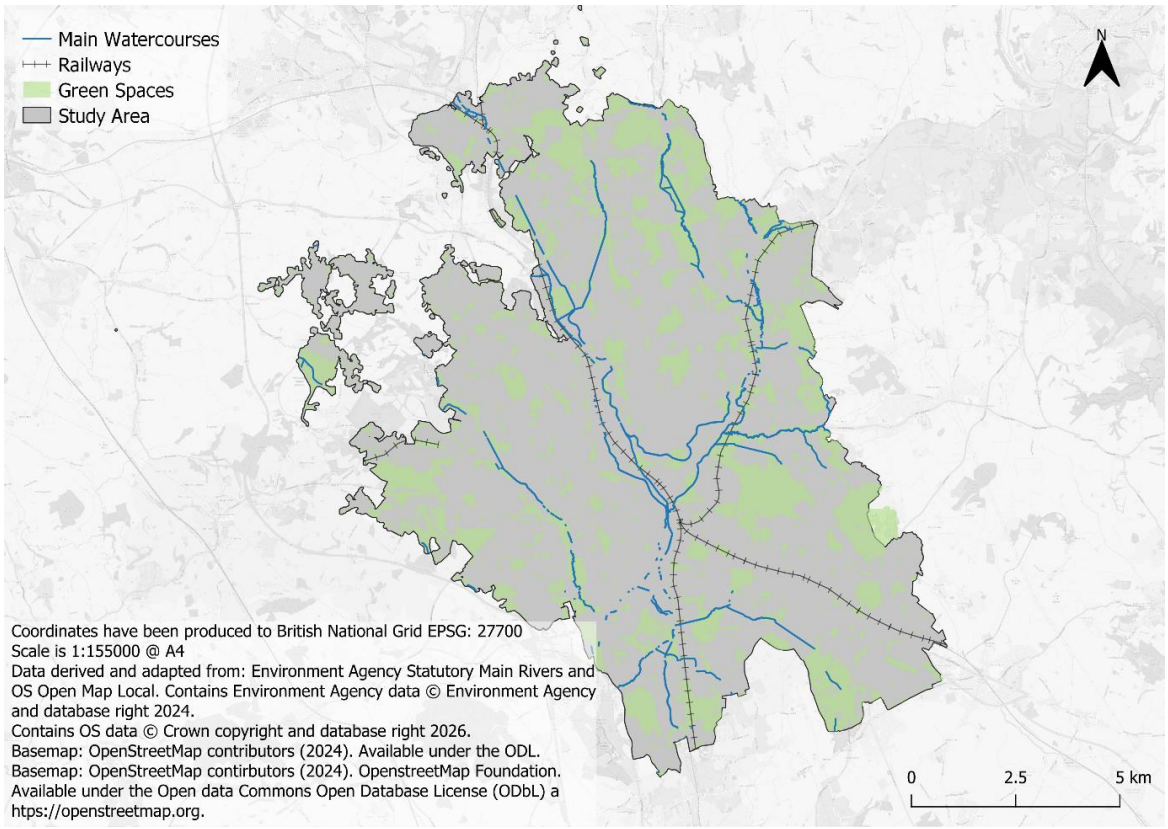


Figure 2.5 Major watercourses and railway infrastructure within Stoke-on-Trent, illustrating the distribution of linear features that may facilitate movement for ground-dwelling species. These 'signature' corridors include canal systems, railway verges and riparian zones (Staffordshire Wildlife Trust, 2024). Railway feature locations are derived from Ordnance Survey data © Crown copyright (2024).

While vegetated railway verges, canal margins and towpaths may provide cover and potential movement routes for ground-dwelling species, large areas of sealed surface and high-traffic road infrastructure, most notably the A50 and A500 (D-Road)

corridors, are expected to form high-resistance components of the matrix and substantial potential barriers to dispersal (Figure 2.6). Habitat connectivity within the city is therefore context-dependent and cannot be inferred solely from proximity or patch size (Tikka et al., 2001; Penone et al., 2012). The mapped arrangement of these features suggests greater north-south structural continuity along major hydrological and rail corridors, whereas east-to-west potential connectivity appears more constrained by the limited availability of parallel vegetated cover and the concentration of major road infrastructure.

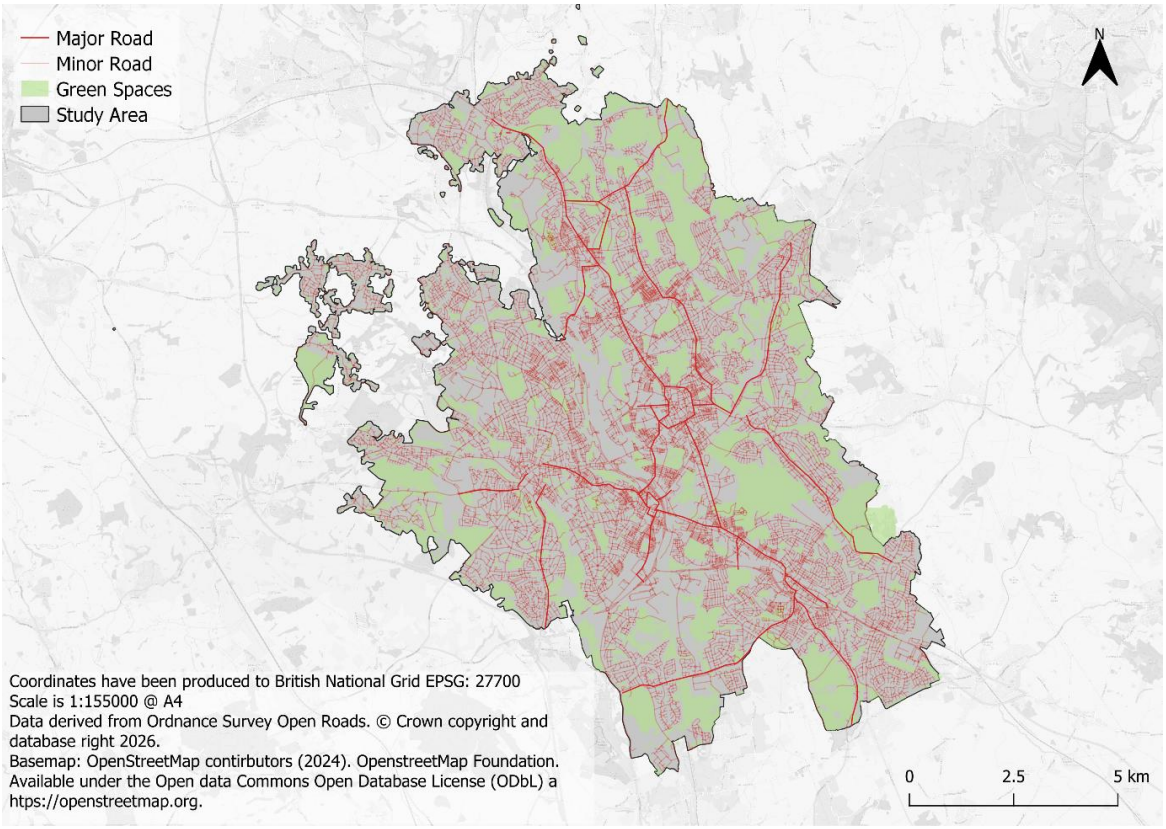


Figure 2.6 Road infrastructure in Stoke-on-Trent, illustrating the distribution of potential barriers to small mammal movement. Major transport corridors, the A50 and A500 (D-road), represent substantial barriers to dispersal, while the dense minor road network contributes to cumulative fragmentation across the urban matrix. Base mapping and road infrastructure are derived from Ordnance Survey data © Crown copyright (2024).

Fragmentation within the study area, therefore, varies with the amount and continuity of vegetation between green spaces, the presence of linear features, and the intensity

of surrounding development. Some patches are separated by roads and sealed surfaces, while others occur alongside canals, watercourses, railway margins or adjoining green spaces that may retain some continuity of ground-level cover (Staffordshire Wildlife Trust, 2024; Stoke-on-Trent City Council, 2013, 2026). These differences are relevant to small terrestrial mammals because many species depend on vegetated cover for movement and refuge, although roads and exposed gaps do not affect all species to the same degree (Baker et al., 2003; Galantinho et al., 2020). The surrounding urban matrix can consequently vary from areas containing several possible routes between patches to sections where movement is more restricted by built infrastructure and breaks in vegetation.

The landscape analyses assess this variation using structural measures of connectivity and resistance around the surveyed green spaces. These variables indicate the arrangement of potentially connective or restrictive features within the matrix and are not treated as direct measurements of dispersal or population exchange.

2.5 Urban grassland dominance and management context

Urban green spaces across the UK are frequently dominated by grassland habitats, such as amenity grassland associated with parks, recreational areas, and roadside verges (Tikka et al., 2001; Ascensão et al., 2012; Galantinho et al., 2020; Phillips, 2021). National statistics indicate that grassland can comprise up to one-third of the total area within many towns and cities, with approximately two-thirds of this resource classified as closely mown amenity grassland (Office for National Statistics, 2026). In contrast, semi-natural grasslands represent a minute proportion of the

urban landscape, accounting for approximately 1.2% of the total urban area. This reflects the functional role of urban green spaces, in which large areas are managed for recreation and aesthetics, resulting in widespread homogenisation of vegetation structure across urban environments (Aronson et al., 2017; Gallo et al., 2017; Liordos et al., 2021).

Around 49% of Stoke-on-Trent is classified as green space, while improved grassland accounts for approximately 23% of the study area (Figure 2.3). Local green space strategies distinguish intensively managed amenity grassland from less frequently managed and semi-natural habitats, including post-industrial sites undergoing passive regeneration (Macgregor et al., 2022; Cox & Rodway-Dyer, 2023). As a result, areas falling within similar broad land cover classes can differ considerably in vegetation structure and habitat condition, reflecting both current management and the land-use history of the city.

Within grass-dominated green spaces, management intensity provides a measure of recurrent human disturbance through its effects on vegetation height, litter accumulation and ground-level cover (Aronson et al., 2017; Darkhani et al., 2019; Winkler et al., 2024; Lisec et al., 2025). Frequent mowing maintains short swards and limits the development of litter, dead vegetation and moss at the soil surface (O'Sullivan et al., 2017; Winkler et al., 2024; Lisec et al., 2025). These features are important for small mammals because dense ground vegetation and litter can provide nesting material, thermal cover and concealed movement routes that reduce exposure to predators (Deeming, 2023). Much of Stoke-on-Trent's amenity grassland is maintained through repeated cutting, meaning that areas mapped purely as green

space can differ in their value to species requiring taller vegetation, basal cover or continuous refuge. Less frequently managed grassland, specifically where scrub or woody vegetation is also present, can retain more of this structure, although the response depends on the habitat requirements of individual species (Dehling et al., 2021; Havrilla & Villarreal, 2024).

Where grass-dominated patches undergo passive or low-intervention regeneration, changes in vegetation are not restricted to increasing sward height, as hedgerows, scrub-lined boundaries and spontaneous treelines can develop and alter the structure of the patch over time (Prach & Walker, 2011; Tischew et al., 2014; Havrilla & Villarreal, 2024). The development of woody vegetation increases vertical and ground-level cover and provides additional forage, concealment and sheltered routes for movement, contrasting with frequently mown grassland where vegetation remains short and exposed (Zhang et al., 2017; Osman et al., 2022; Deeming, 2023; Lisec et al., 2025). This creates variation within grass-dominated green spaces that may appear similar under broad land cover classifications but differ in the amount and arrangement of vegetation available to small mammals. Hedgerows, scrub and treelines may also persist around the margins of managed grassland where the sward itself provides little cover, creating localised areas of shelter and vegetation continuity within otherwise open patches.

The effects of grassland management are unlikely to be the same across the small mammal assemblage, given the differences in habitat use and tolerance of disturbance between species (Churchfield, 1982; Neet & Hausser, 1990; Hare, 2005; Buesching et al., 2008; Haapakoski & Ylönen, 2010; Wilson et al., 2016). Management

can alter vegetation height, ground cover, litter, scrub development and the availability of refuge within the same broad habitat type, and these differences can influence which species are able to persist within regularly disturbed green spaces (Milberg et al., 2017; Parsons et al., 2018; Winkler et al., 2024). Intensively managed grassland may favour species capable of using open and frequently disturbed habitats while reducing suitability for species associated with denser vegetation and greater ground-level cover (Berger et al., 2024; Dondina et al., 2025; Lisec et al., 2025). Management intensity was included within the research as grassland mowing to examine these differences directly across the surveyed green spaces, alongside measures of habitat composition and vegetation structure.

Fragmentation adds another influence because the condition of an individual green space does not determine how easily small mammals can move through the surrounding urban landscape. Roadside verges, hedgerows, scrub margins and other linear vegetation can extend cover beyond individual patches, although their usefulness depends on their continuity and management (Breuste et al., 2008; Darkhani et al., 2019; Ryszkowski et al., 2019; Liordos et al., 2021). A vegetated boundary that appears continuous within mapped data may contain frequently cut sections, gaps or exposed ground that alter its use by small mammals, while unmanaged sections may retain denser cover and refuge (O'Sullivan et al., 2017; Aronson et al., 2017; Darkhani et al., 2019). Connectivity within the study area cannot be inferred from the presence of a linear feature alone, because its structure and management determine the conditions encountered between habitat patches.

This provides a structural framework for interpreting variation in the biological records presented in the following section. Adaptable generalists may persist across a broader range of management conditions because of greater dietary and behavioural flexibility (Callaghan et al., 2019; Dehling et al., 2021; Rimbach et al., 2025), whereas species with stronger requirements for dense grass, litter or woody cover are expected to occur more consistently where those structures are retained. By quantifying management, internal habitat structure and surrounding matrix conditions in the subsequent chapters, the study moves beyond broad land cover descriptions to evaluate their associations with small mammal community responses and potential landscape permeability.

2.6 Small mammal records in Stoke-on-Trent

Existing knowledge of small mammal presence within Stoke-on-Trent and the adjoining urban area of Newcastle-under-Lyme is derived primarily from biological records held by the National Biodiversity Network (NBN) and Staffordshire Ecological Record (SER). These datasets comprise presence-only records collected through a mixture of structured surveys and opportunistic recording over extended time periods. While such records do not provide information on abundance or population trends, they offer important contextual insight into the species known to occur within the study area (Isaac & Pocock, 2015; Van Eupen et al., 2021; Turner et al., 2023). Biological records that were obtained from the NBN Atlas comprise multiple datasets from a range of providers (Appendix A).

To ensure alignment between the contextual species pool and the field methodology employed in this study, the scope of species considered was deliberately restricted to

small terrestrial mammals that could be reliably detected using Longworth live traps. This approach necessitated the exclusion of larger-bodied mammals (e.g. hedgehog *Erinaceus europaeus* and brown rat *Rattus norvegicus*), arboreal species, and aquatic taxa, as these groups require alternative survey methods and would not be consistently comparable and detectable within a standardised trapping framework.

By constraining the species pool in this way, the study focuses on a functionally comparable group of ground-dwelling small mammals, primarily rodents and terrestrial shrews, for which detection probability and survey effort can be standardised across sites (Flowerdew et al., 2004; Gurnell et al., 2019). This filtering process ensures that subsequent analyses are methodologically coherent and that patterns observed in the field can be directly interpreted in relation to the species pool identified from existing biological records (Isaac & Pocock, 2015; Van Eupen et al., 2021; Pocock et al., 2023).

The filtered records indicate a typical assemblage of lowland UK small mammals, consistent with expectations for urban and peri-urban landscapes in central England (Atkins et al., 2018; Mammal Society, 2024). Species records (n = 358) include wood mouse (*Apodemus sylvaticus* (Linnaeus, 1758)), harvest mouse (*Micromys minutus* (Pallas, 1771)), house mouse (*Mus musculus* Linnaeus, 1758), bank vole (*Myodes glareolus* (Schreber, 1780)), field vole (*Microtus agrestis* (Linnaeus, 1761)), common shrew (*Sorex araneus* Linnaeus, 1758) and pygmy shrew (*Sorex minutus* Linnaeus, 1766). These taxa are characteristic of landscapes containing managed green spaces, linear vegetation and remnant semi-natural habitats (Gurnell et al., 2008; Mammal Society, 2024). *Microtus agrestis* and *Apodemus sylvaticus* were the most

frequently recorded taxa, although record frequency reflects differences in recorder effort, habitat accessibility and species detectability as well as underlying occurrence.

Records span multiple decades, with a pronounced temporal bias toward more recent years; over 50% of the total dataset (n=358) has been submitted since 2021. This represents a substantial increase from the records available during the early development of the thesis in 2021. Spatially, records are highly aggregated, with significant concentrations in accessible public green spaces and established nature reserves. Notable hotspots include Berry Hill Fields, Apedale Community Country Park, and the Trent and Mersey Canal corridor, which function as popular walking routes for biological recorders. A summary of species records derived from the filtered dataset is presented in Table 2.2 with the full data set in Appendix A.

Table 2.2 Summary of species records for small mammal taxa in Stoke-on-Trent and Newcastle-under-Lyme, according to the National Biodiversity Network (NBN) and Staffordshire Ecological Record (SER).

Species	Scientific Name	N	First Record	Last Record	Primary Data Source
Field Vole	*Microtus agrestis*	108	1966	2025	National Mammal Atlas Project
Wood Mouse	*Apodemus sylvaticus*	105	1960	2025	National Mammal Atlas Project
Common Shrew	*Sorex araneus*	62	1964	2025	National Mammal Atlas Project
Bank Vole	*Myodes glareolus*	41	1972	2025	National Mammal Atlas Project
Harvest Mouse	*Micromys minutus*	21	1736*	2024	National Mammal Atlas Project
Pygmy Shrew	*Sorex minutus*	14	1960	2025	SER Species-based Surveys

Species	Scientific Name	N	First Record	Last Record	Primary Data Source
House Mouse	*Mus musculus*	6	1983	2023	SER Species-based Surveys
Shrew sp.	Sorex	1	1982	1982	Stoke-on-Trent Env. Survey

*The 1736 date is reproduced from the NBN Atlas source record listed in Appendix A.

In contrast, large proportions of the private urban matrix, industrial estates and less-accessible brownfield sites contain few or no records (Figure 2.7), which shows a strong spatial bias associated with recorder effort and site accessibility. However, the records identify frequently reported taxa associated with the grassland and woody habitat mosaic present across the study area. Improved grassland (23.02%) and broadleaved woodland (8.03%) form the principal mapped vegetated land cover classes within a predominantly suburban (47.40%) and urban (16.69%) matrix (Figure 2.4). The observed pattern is therefore consistent with grassland structure, woody vegetation and linear habitat features contributing to the available habitat network, but the presence-only data cannot establish habitat preference or quantify the effects of those variables (Dickman & Doncaster, 1987; Flowerdew et al., 2004; Parsons et al., 2018; Atkins et al., 2018).

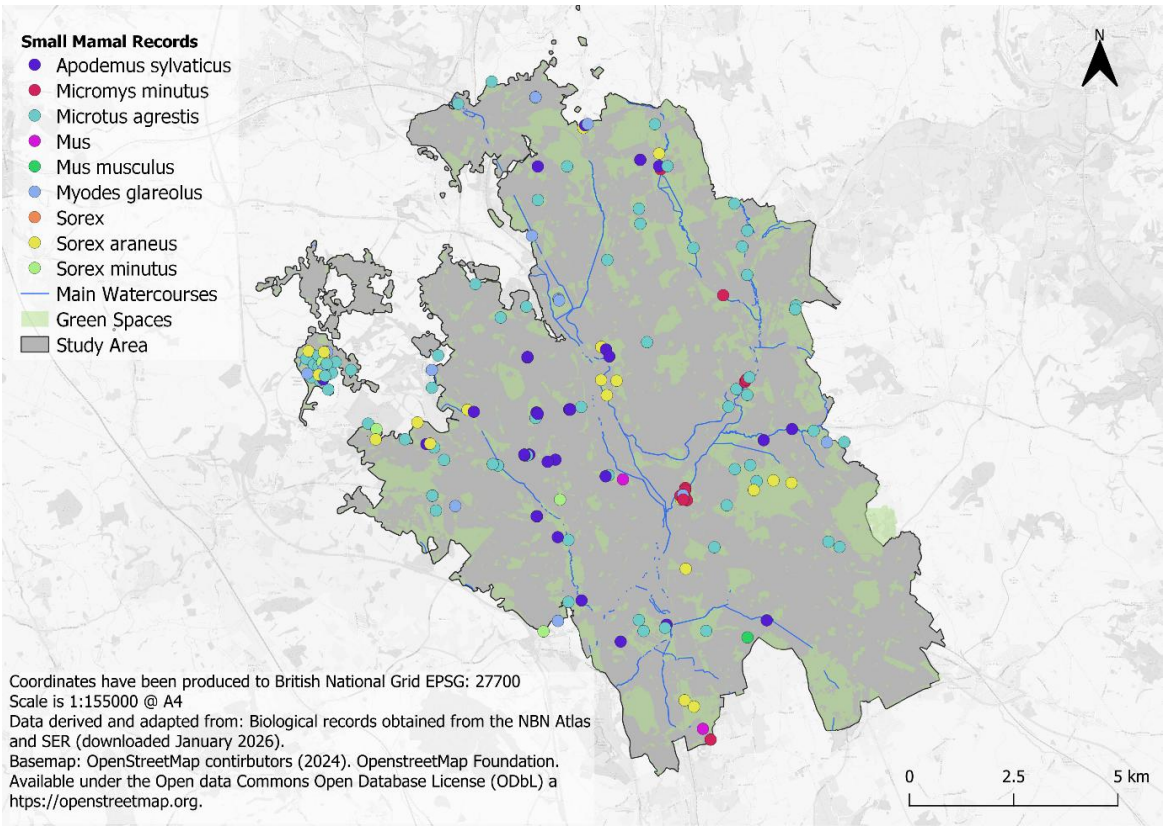


Figure 2.7 Small mammal records obtained from the National Biodiversity Network (NBN) Atlas and Staffordshire Ecological Record (SER), comprising multiple datasets from a range of providers. Base mapping is derived from Ordnance Survey data © Crown copyright (2024).

These data establish the small mammal species known to occur within the study area but cannot be used to estimate abundance, habitat quality or population change, as recording effort is uneven and the absence of a record does not demonstrate species absence (Mathews et al., 2018; Van Eupen et al., 2021; Coomber et al., 2021). Their use within this study is to define the regional species pool against which the field surveys can be compared and to identify areas where existing biological recording is limited. The spatial and temporal gaps within the dataset also support the need for standardised field surveys, where consistent survey effort can be used to examine variation in small mammal richness and relative capture activity in relation to habitat structure, management and the surrounding landscape.

2.7 Ecological characteristics of the local small mammal community

The small mammal communities present within Stoke-on-Trent and Newcastle-under-Lyme occur within a post-industrial landscape where green spaces are fragmented by surrounding development and grassland forms a large proportion of the remaining vegetated habitat. Much of this grassland is subject to regular management, which contrasts with the wider range of vegetation structures that would occur across less intensively managed lowland habitats in the UK (Norton et al., 2019; Staffordshire Wildlife Trust, 2024; Stoke-on-Trent City Council, 2025). Frequent mowing reduces vegetation height, litter accumulation and the development of ground-level cover, all of which influence the availability of shelter and resources for small terrestrial mammals (Aronson et al., 2017; Milberg et al., 2017; Darkhani et al., 2019). Green spaces classified under the same broad habitat type can consequently provide different conditions depending on their management, internal vegetation structure and position within the surrounding urban landscape. Patch area, habitat composition, woody and ground-level structure, grassland management and the configuration of the surrounding matrix were included within the research to examine how these differences were associated with small mammal communities.

The assemblage expected within Stoke-on-Trent reflects species capable of occupying fragmented urban and peri-urban habitats, although their ability to persist within them differs according to habitat requirements and tolerance of disturbance (Dickman & Doncaster, 1987; Gurnell et al., 2008; Łopucki et al., 2013; Rimbach et al., 2025). *A. sylvaticus* occurs across a broad range of vegetated habitats and is able

to persist in disturbed environments, including urban green spaces and vegetated components of the surrounding matrix. This species is often referred to as a generalist species due to its widespread occurrence and adaptability to environments that are markedly different from natural landscapes (Dickman & Doncaster, 1987; Wilson et al., 2016; Nardotto, 2023). European studies of fragmented and urban landscapes frequently record *Apodemus* species as a large component of small mammal captures, reflecting their broad habitat use and ability to exploit resources across different vegetation types, along with tolerance to a wide range of conditions (Łopucki et al., 2013; Rimbach et al., 2025; Baláž et al., 2025).

The vole species present in the data have a different set of habitat requirements; *M. glareolus* is associated with woodland edges, hedgerows, scrub and habitats containing developed understorey vegetation, where cover and food resources occur across both ground and shrub layers (Gurnell et al., 2008; Haapakoski & Ylönen, 2010; Mathews et al., 2018). Unlike more ground-dependent voles, it climbs readily and uses scrub vegetation when foraging for fruits, seeds and invertebrates, while leaf litter and dense woody cover provide additional feeding and refuge opportunities. Loss or simplification of the understorey can alter the suitability of otherwise wooded or scrub-dominated habitats (Gurnell et al., 2008; Haapakoski & Ylönen, 2010; Mathews et al., 2018). *M. agrestis* is more closely associated with dense grassland vegetation, where litter, uncut swards and basal cover provide nesting and movement habitat (Tattersall et al., 2002; Buesching et al., 2008; Gurnell et al., 2008; Haapakoski & Ylönen, 2010). Vegetation height alone does not describe these conditions, as tall grass with little basal density or accumulated litter may still provide poor cover. *M. agrestis* constructs nests and tunnel systems within dense vegetation close to ground

level, making the retention of an established sward and litter layer relevant to habitat use (Tattersall et al., 2000; Gurnell et al., 2008; Sundell et al., 2012). Frequent mowing can remove these features, while less frequently managed grassland allows them to develop over time. The contrasting requirements *of M. glareolus* and *M. agrestis* also illustrate why broad classifications such as woodland or grassland provide limited information without considering the structure present within them.

Insectivorous species such as *S. araneus* represent a different component of the small mammal assemblage because their habitat use is closely linked to the availability of invertebrate prey and the microhabitat conditions that support it. Shrews (*Sorex*) have high metabolic demands and must forage frequently, making the availability of prey within litter, vegetation and the upper soil layers an important influence on their distribution (Churchfield, 1982; Neet & Hausser, 1990; Schmidt et al., 2009; Sundell et al., 2012; Dokulilová et al., 2023). Dense ground vegetation and accumulated litter can support invertebrate communities while also providing cover and more stable conditions close to the soil surface, whereas repeated mowing removes vegetation and litter, increases exposure and alters the conditions in which prey species occur. These effects can make intensively managed grassland less suitable for shrews even where the broad habitat classification remains unchanged, although responses will also depend on factors such as soil condition, moisture and the distribution of prey within individual patches. Soil moisture can influence both shrew activity and invertebrate availability, but it was not measured during this study because conditions can change over short spatial and temporal scales and would have required repeated measurements across each site to describe them adequately (Schmidt et al., 2009; Sundell et al., 2012). Vegetation structure and management

were measured across all surveyed sites, allowing differences in ground cover, disturbance and habitat development to be examined alongside the occurrence and capture rates of *S. araneus*.

The interactions between species may also influence the composition of small mammal communities within individual patches, meaning that habitat characteristics alone are unlikely to explain all of the variation between sites. This is particularly relevant to *A. sylvaticus*, which was the dominant species recorded during this study and has a broad ecological niche that allows it to persist across woodland, grassland, scrub and heavily modified habitats (Dickman & Doncaster, 1987; William Schweiger et al., 2000; Probst & Probst, 2023). Where *A. sylvaticus* occurs alongside bank voles (*M. glareolus*), there is considerable potential for overlap in food and space, although the two species can reduce direct competition through differences in microhabitat use and activity patterns (Canova, 1993). More recent behavioural evidence supports this relationship, with Probst and Probst (2023) showing that increasing numbers of *Apodemus* mice were associated with greater diurnal activity in bank voles. This response did not require frequent aggressive encounters and showed that the presence of a dominant competitor alone can be sufficient to alter how another species uses otherwise suitable habitat.

This becomes especially relevant within urban landscapes, where suitable habitat is already restricted and species are repeatedly brought together within relatively small and isolated green spaces. Bouilloud et al. (2024) found clear differences in the persistence of small mammal species along an urbanisation gradient, with the relative abundance of *A. sylvaticus* increasing with urbanisation while *M. glareolus*

declined and *S. araneus* was largely associated with less urbanised sites. This supports the apparent high recordings of *A. sylvaticus* within the current study and suggests that its dominance may reflect both its ability to colonise disturbed habitats and its capacity to persist once other small mammal species become restricted. Fuentes-Montemayor et al. (2020) also found that *A. sylvaticus*, *M. glareolus* and *M. agrestis* were all capable of colonising newly created woodland, but the factors determining their abundance differed between species. Therefore, two patches with apparently similar habitat characteristics may support different communities depending on which species reach them, their abundance after colonisation, and the subsequent interactions that occur.

Shrews add another dimension to these relationships because they occupy a substantially different trophic niche from the predominantly granivorous and herbivorous rodents. *S. araneus* depend heavily on invertebrate prey and have high energetic requirements, resulting in frequent activity throughout both day and night. Lindsø et al. (2025) demonstrated this directly within a community containing *A. sylvaticus*, *M. glareolus* and *S. araneus*. Wood mice remained strongly nocturnal, whereas common shrews were cathemeral and bank vole activity changed between seasons. These differences provide a mechanism through which species occupying the same physical area can reduce the level of direct interaction between them. Shrews may therefore be capable of using patches already occupied by relatively high numbers of rodents because they are not entirely dependent on the same food resources or periods of activity. However, they are not independent of the wider community. Bujnoch et al. (2024) found that shrew abundance tended to be negatively associated with bank vole abundance, and that community composition

can influence population patterns even between species with relatively different feeding ecologies.

There are similar behavioural differences apparent amongst the vole species themselves. Sundell et al. (2012) found that bank voles and field voles were largely concentrated within their preferred woodland and grassland habitats when population densities were low but expanded into less preferred habitats as numbers increased. At higher densities, the two vole species also showed spatial avoidance of one another, whereas common shrews remained comparatively generalist in their habitat use. This is important when considering colonisation of fragmented urban green spaces because the presence of suitable vegetation does not necessarily mean that all species capable of using that habitat will occur together. A grassland patch may provide suitable conditions for *M. agrestis*, for example, but its use may also depend on population density, surrounding dispersal opportunities and the abundance of other small mammals already occupying the site. Further urban research still indicates that habitat composition remains the primary driver of small mammal communities rather than competitive exclusion alone (Dondina et al., 2025). However, the evidence suggests that species interactions can modify how suitable habitat is actually used once colonisation has occurred. Competition and behavioural avoidance were not measured during the field surveys in this study, and the capture data cannot demonstrate that either process occurred. However, these interactions provide an additional biological mechanism that may contribute to the differences in assemblage composition between otherwise similar urban green spaces.

Small terrestrial mammals generally operate across more restricted spatial ranges than birds and other mobile urban fauna, making the structure of the surrounding landscape increasingly relevant where suitable habitat is divided into separated patches (Savard et al., 2000; Concepción et al., 2015; Kondratyeva et al., 2020). Many species rely on vegetation at or close to ground level for shelter and movement, while roads, sealed surfaces and exposed gaps can increase resistance and leave individuals more vulnerable when moving between habitat areas (Baker et al., 2003; Richardson, 2010; Esperandio et al., 2019; Galantinho et al., 2020). Hedgerows, scrub margins, watercourses and other vegetated linear features may reduce these breaks by retaining cover through parts of the urban matrix, although their value will depend on the species attempting to use them and the condition of the vegetation itself. A landscape that remains permeable to *A. sylvaticus*, for example, may still restrict species with stronger requirements for dense grassland or continuous ground cover, as dispersal ability, habitat requirements and tolerance of exposed or disturbed environments differ across the assemblage (Neet & Hausser, 1990; Tattersall et al., 2000; Haapakoski & Ylönen, 2010; Galantinho et al., 2020). These differences provide the basis for examining community and species-level capture responses in the following chapters.

While this broader assemblage provides important ecological context, not all species are equally suitable for quantitative analysis within fragmented urban systems. Differences in detectability, capture probability, and methodological constraints associated with standard survey techniques (e.g. Longworth trapping) limit the extent to which some species can be reliably recorded. For example, species such as *M. minutus* are likely to be underrepresented due to their arboreal tendencies and low

body mass, while others may occur too infrequently within the study area to support robust statistical analysis (Brouard et al., 2015; Kettel et al., 2016). Additional species within the dataset further illustrate these limitations. *S. minutus*, although present within the regional species pool, is also typically associated with highly structured habitats and exhibits low detectability in live-trapping surveys due to its small body size and high metabolic demands (Churchfield, 1982; Sparti & Genoud, 1989; Shonfield et al., 2013). Furthermore, the physiological sensitivity of this species presents significant ethical and practical challenges; the high risk of trap-induced mortality necessitates specific modifications to trapping methods, such as reduced check intervals, which are often unfeasible in large-scale urban studies. Therefore, capture rates for *S. minutus* are typically low and inconsistent, especially in fragmented environments where suitable microhabitats are spatially restricted.

M. musculus was also excluded from the focal assemblage because its ecology is closely associated with human habitation and built structures, where food, shelter and nesting opportunities are often provided directly by human activity rather than semi-natural vegetation (Berry, 1981; Phifer-Rixey & Nachman, 2015). Its occurrence within biological records is more likely to represent commensal populations occupying buildings and other anthropogenic environments than responses to grassland management, woody structure or the configuration of surrounding green space. Including *M. musculus* alongside species whose distributions are more closely associated with vegetated habitats could obscure ecological relationships between habitat structure and the broader small mammal community (Weissbrod et al., 2014; Küçüktaş & Guenther, 2022). The study is focused on small mammals using

urban green spaces and the surrounding habitat network, so *M. musculus* was not retained within the focal species analyses.

Preliminary fieldwork and pilot trapping across a subset of green spaces reflected the pattern suggested by the biological records, with captures dominated by a small number of species. *A. sylvaticus, M. agrestis, M. glareolus and S. araneus* were recorded frequently enough to support further analysis, while the remaining species were either absent from the pilot surveys or encountered too infrequently to be analysed reliably. Although the NBN and SER records indicate a broader small mammal assemblage across Stoke-on-Trent and Newcastle-under-Lyme, the field surveys showed that only a smaller group was consistently encountered within the urban green spaces sampled, and it was these species that formed the basis of the later community and species-level analyses.

2.8 Pilot Study and trapping observations

The pilot phase was undertaken across a subset of urban green spaces within the study area to assess the suitability of standardised trapping methods and inform the wider survey design, while also allowing the practical limitations of working across sites with differing habitat structure, management and levels of public use to be identified before full deployment. The surveys were led by the author, whose professional background as an ecologist included experience in mammal survey, handling and field identification, supported by formal tracking and trapping certifications and engagement with the Mammal Society during the early development of the research. Preliminary research findings were also presented to peer specialists, providing an opportunity for the developing methodology and early

observations to be discussed within the wider research community, while consultation with national experts during the pilot phase helped align the sampling approach with established recording standards and accepted field practice for small mammals in urban environments. The trials were subsequently used to refine trap placement, bait selection and deployment strategy, alongside practical considerations including site access, habitat variability, public disturbance, trap security and differences in management and disturbance regimes between individual green spaces.

Before selecting the final methodology, non-invasive detection methods, including footprint tracking tunnels, hair traps and remote camera traps, were evaluated. While these methods can provide useful occurrence data, they were not suitable as the principal method for this thesis because the study required repeatable measures of focal-species identity, species richness, relative capture response and community diversity. These responses could not be obtained consistently from presence-only biological records, footprint tunnels or incidental detections alone (Sutherland, 2006; Gurnell & Flowerdew, 2019). Footprint-based methods were also limited by the difficulty of reliably differentiating mouse and vole prints to species level under field conditions. Longworth live-trapping was therefore selected because it allowed direct confirmation of species identity and the calculation of standardised capture-based metrics across repeated surveys.

Due to the relatively small size and irregular configuration of many urban green spaces, traps were deployed along transects rather than fixed grids (Sutherland, 2006). Initial trials with grid layouts showed that potential trap positions frequently

fell within unsuitable or exposed areas where traps would have been vulnerable to disturbance or susceptible to theft. A transect-based approach therefore enabled consistent sampling of the vegetated habitats within the green space patches relevant to the study, including grassland, hedgerows, scrub margins and woodland-edge features, while maintaining comparable effort where grid-based designs were not feasible (Pearson and Ruggiero, 2003). The design was not intended to sample every land cover class in proportion to its total availability. Chapter 3 reports both the wider composition of each green space patch and the habitats incorporated within the trapping layout. To ensure animal welfare during the pilot and main survey, all trapping followed established best-practice guidance and operated under the conditions of the Natural England General Licence GL01 for the trapping of shrews (see Appendix C). This included the provision of specialised bedding and high-calorie bait, including greenbottle pupae (*Lucilia sericata*), to meet the high metabolic demands of insectivorous species.

Pilot observations indicated that traps positioned within dense ground-level vegetation, hedgerow bases, scrub margins and grassland-woody transitions were regularly used by small mammals, whereas open and intensively managed grassland frequently provided fewer suitable trap positions. However, placement of transects within these habitats produced negligible differences when transect locations were chosen based on favourable-looking habitats or randomised across suitable vegetated areas. These observations informed the inclusion of structurally suitable vegetated habitat within the main transect layouts but were not treated as a formal analysis of edge effects or microhabitat preference (Tattersall et al., 2002; Gelling et al., 2007b; Schlinkert et al., 2016). Within the selected trapping habitat, transects

were randomly dispersed through the available vegetation and were not concentrated in locations considered most favourable for small mammals. Unsuitable areas were excluded from the sampling frame before randomisation, consistent with established ecological sampling (Greenwood & Robinson, 2006) and with British small mammal studies in which Longworth trapping locations were randomly selected from habitats meeting predefined survey criteria (Mathews et al., 2006). Longworth trapping was restricted to ground-level vegetated habitat, recognising that traps could not be distributed indiscriminately across sealed surfaces, paths, buildings, open water or other unsuitable areas (Flowerdew et al., 2004; Gurnell & Flowerdew, 2019).

Grassland formed the main habitat of interest because grassland management intensity was a central component of the research, but other areas providing ground-level structure were retained within the trapping area where they occurred. The habitats incorporated into the trapping layout varied between patches according to those present, with no requirement for each site to contain the same combination of grassland, woodland, scrub or boundary vegetation. This retained the differences in habitat composition and structure among the urban green spaces and reflected the habitat requirements of the focal small mammals, which use vegetation to differing degrees for shelter, foraging and movement (Coppeto et al., 2006; Stradiotto et al., 2009). Where a habitat was absent from a patch, it could not form part of that site's trapping layout. The habitat of each transect was recorded during the surveys, allowing the distribution of trapping effort to be examined alongside site-level habitat composition and management.

Transect arrangements can provide greater sampling efficiency for studies of community composition and relative abundance, whereas grid arrangements provide greater spatial resolution for estimating population density, home ranges and spatial dispersion (Pearson & Ruggiero, 2003; Machtinger & Williams, 2020). The pilot and main surveys were therefore designed to compare relative capture responses among heterogeneous green space patches and were not intended to estimate absolute population density.

Development of the survey method also established the biological response variables used in the final analyses. Presence-only records were retained as contextual evidence of the regional species pool, while repeated live-trapping data were used to calculate species richness, CPUE and Simpson's reciprocal diversity. CPUE was interpreted as a standardised measure of relative capture response and not as a complete census or direct estimate of population size. These three metrics described different properties of the assemblage: the number of focal species detected, total capture response after adjustment for sampling effort, and the balance of captures among species.

2.9 Chapter summary, species focus and research direction

The preceding assessment has established the ecological and spatial context of the Stoke-on-Trent and Newcastle-under-Lyme study area and identified the small mammal assemblage that could reasonably be expected within its urban green space network. The study area contains a heterogeneous mixture of managed grasslands, remnant and regenerating woody vegetation, riparian corridors, railway margins and other green spaces embedded within a fragmented post-industrial matrix. Although

mapped green space is relatively extensive, individual patches differ substantially in vegetation structure, management intensity, disturbance and surrounding landscape context. Therefore, their value to small mammals cannot be inferred from green space designation or area alone.

Regional records from the NBN and SER provided an initial indication of the small mammal species recorded within the study area. However, the records were unevenly distributed and had been collected using a range of different methods, meaning they could not be used to compare community composition or relative capture rates between individual green spaces. Pilot fieldwork was therefore used to bridge the gap between these regional records and the main site surveys, while also helping to refine the survey approach before the main fieldwork began.

The integration of regional records, known species ecology and pilot trapping resulted in the selection of four focal taxa: *A. sylvaticus*, *M. agrestis*, *M. glareolus* and *S. araneus*. These species were detected sufficiently frequently to support quantitative analysis, could be identified reliably using the selected live-trapping method and represented contrasting associations with urban habitat structure. *A. sylvaticus* is a relatively adaptable species capable of using a range of vegetated habitats, although its use of urban green spaces may remain dependent on woody cover and sheltered movement routes (Dickman & Doncaster, 1987; Atkins et al., 2018). *M. agrestis* is more strongly associated with grass-dominated habitats providing sufficiently tall and dense ground-level vegetation (Dickman & Doncaster, 1987). *M. glareolus* is largely associated with woody vegetation, scrub, litter and dense basal cover, while *S. araneus* depends on structurally complex and often moist vegetation supporting

concealment and invertebrate prey (Dickman & Doncaster, 1987). The focal assemblage therefore provides a means of examining both community-wide responses and contrasting species-specific associations, rather than whether all small mammals respond uniformly to urbanisation.

Pilot trapping also confirmed that suitable sampling habitat was concentrated within vegetated portions of the mapped green space patches rather than across the whole extent of each green space. Traps were therefore used to sample ground-level vegetated habitats, including grassland and associated woody or structurally complex features, while sealed surfaces, buildings, open water and exposed hardstanding fell outside the intended sampling domain. This distinction defines the scope of the subsequent analyses: the thesis evaluates variation among urban green spaces and their surrounding landscapes, rather than estimating small mammal CPUE across every land use within the city. The pilot work and spatial review refined the original candidate framework into the variables carried forward through Chapters 3-5. The mapped green spaces were treated as heterogeneous survey and prediction units containing internal habitat variation, not as homogeneous habitat islands whose ecological value could be inferred from external geometry (Angold et al., 2006). Shape index was considered but excluded because its perimeter-area relationship could not distinguish cartographic boundary complexity from the amount, quality or arrangement of habitat within each green space. The final design instead separated biologically interpretable patch-scale measures from the composition and configuration of the surrounding matrix.

The ecological characteristics of the focal species produce several expectations for the empirical chapters. Within individual green space patches, vegetation structure and management are expected to alter the availability of refuge, feeding habitat and continuous ground cover (Dickman & Doncaster, 1987). Woody linear features may provide local habitat and sheltered movement routes, whereas frequent mowing, dense path networks and repeated disturbance may reduce the value of otherwise extensive green space (Atkins et al., 2018; Watson et al., 2020). Patch area may increase the potential amount of habitat available, although its effect is expected to depend on the composition and structural condition of the vegetation within it (Dickman, 1987; Angold et al., 2006; Łopucki & Kitowski, 2017).

At the wider landscape scale, the surrounding matrix was expected to influence whether suitable patches remain accessible and capable of supporting a broader small mammal community. Greater isolation, subdivision of green space and road density may increase resistance and constrain movement, while hedgerows, watercourses and other vegetated linear features may indicate greater potential landscape permeability (de Redon et al., 2015; Galantinho et al., 2020). These spatial variables represent structural conditions consistent with movement and connectivity processes; they do not directly demonstrate animal movement, dispersal or gene flow.

This chapter therefore addresses the first thesis objective established in Chapter 1 by characterising the urban green space network and identifying the locally occurring and appropriate focal assemblage. It also provides the study-system-specific basis for the patch-scale, species-specific and landscape-scale expectations represented

by H1-H3, together with the combined predictive framework examined under H4 presented in Chapter 1.

Chapter 3 first examines how habitat composition, vegetation structure, paths, patch area and grassland management influence community metrics and focal-species responses within surveyed green spaces. Chapter 4 then evaluates whether the composition and configuration of the surrounding matrix provide additional explanation for community richness, CPUE and Simpson's reciprocal diversity at 250 m and 500 m spatial extents. Chapter 5 integrates supported patch- and matrix-scale predictors into a validated model of community CPUE and compares predictions under low, moderate and intensive grassland-management scenarios.

Chapter 3 Effects of Individual Green Space Patch Characteristics on Small Mammal Communities

3.1 Introduction

Chapter 2 establishes the broader ecological context of Stoke-on-Trent and Newcastle-under-Lyme as a fragmented urban landscape characterised by patch heterogeneity in habitat composition, vegetation structure, and management intensity. Within this polycentric urban matrix, green spaces occur as isolated or semi-connected habitat patches embedded within a highly modified environment dominated by sealed surfaces, transport infrastructure, residential development, and reclaimed industrial land (Tandy, 1975; Stoke-on-Trent City Council, 2013; Stoke-on-Trent City Council, 2025). Although approximately 49% of the study area is classified as green space, the ecological quality and functional suitability of these spaces vary considerably according to vegetation structure, disturbance regime, and habitat configuration.

Commonly accepted urban ecological theory predicts that species persistence within fragmented systems is influenced not only by habitat amount, but also by habitat quality, patch structure, and the degree of disturbance operating within individual patches (Fahrig, 2003; Fischer & Lindenmayer, 2007; Concepción et al., 2015). For less mobile taxa such as small mammals, these factors may be especially important because movement through the urban matrix is constrained by exposure risk, habitat discontinuity, and the availability of structurally suitable refuge habitat (Dickman & Doncaster, 1987; Łopucki et al., 2013; Parsons et al., 2018). Therefore, urban green spaces cannot be treated as ecologically equivalent units solely on the

basis of land cover classification, formal designation or total area. The internal structural composition and management of individual patches may fundamentally determine their suitability for wildlife.

In urban environments, grassland habitats are often subject to intensive management regimes aimed at recreation, aesthetics, and public amenity provision (Aronson et al., 2017; Gallo et al., 2017). Frequent mowing can substantially alter vegetation architecture by reducing sward height, removing litter accumulation, suppressing successional development, and limiting the formation of dense ground-level cover (O'Sullivan et al., 2017; Winkler et al., 2024). These changes may directly affect small mammals by reducing nesting opportunities, increasing predation risk, and disrupting movement pathways across the patch interior (Deeming, 2023). Equally, patches containing unmanaged grassland, scrub margins, hedgerows, and treelines may provide structurally complex refuge habitat capable of supporting greater abundance and diversity.

Structural habitat complexity is frequently associated with higher biodiversity in urban environments because a greater range of vegetation structures provides more opportunities for shelter, feeding and movement (Tews et al., 2004; Beninde et al., 2015). In fragmented urban habitats, woody linear features such as hedgerows, scrub edges and treelines may be especially important for small mammals by providing cover, food resources and routes between different areas of habitat (Gurnell et al., 2019; Osman et al., 2022). In contrast, features associated with greater human disturbance, such as footpaths, can reduce habitat quality through trampling, soil compaction and increased activity within the patch (Tikka et al., 2001; Van Helden et

al., 2020). These smaller-scale habitat features may therefore be just as important as the overall size of the green space in determining how suitable it is for small mammals.

Despite increasing recognition of the importance of urban green infrastructure, relatively few studies have examined how patch-scale habitat characteristics interact with vegetation management intensity to shape small mammal communities in medium-sized UK cities. Existing urban mammal research has frequently focused on broad urban-rural gradients or landscape-scale occupancy patterns, with comparatively less attention given to how fine-scale structural variation within individual urban green spaces influences abundance, diversity, and species composition (Łopucki et al., 2013; Liordos et al., 2021; Fung et al., 2024). Furthermore, urban grasslands are often treated as homogeneous habitat categories despite substantial differences in mowing intensity, vegetation structure, and successional complexity.

Stoke-on-Trent has a long history of industrial development and land-use change, leaving a mixture of amenity grasslands, remnant habitats, canal corridors, railway verges and naturally regenerating brownfield sites across the city (Tandy, 1975; Staffordshire Wildlife Trust, 2024; Stoke-on-Trent City Council, 2025). These habitats vary in their structure and management, providing an opportunity to assess how conditions within urban green spaces affect small mammal communities. This chapter investigates 21 urban green spaces across Stoke-on-Trent and Newcastle-under-Lyme to assess how habitat composition, habitat structure and mowing intensity affect small mammal communities. The analyses consider grassland cover,

woody linear density, path density, watercourse density and patch size, and their relationships with species richness, catch per unit effort (CPUE), community evenness and the responses of the focal species.

By integrating repeated live-trapping surveys with GIS-derived habitat metrics and mixed-effects modelling, this chapter aims to identify the key environmental drivers shaping small mammal assemblages within fragmented urban green spaces. In doing so, the chapter provides the empirical patch-scale foundation required for the subsequent landscape-scale connectivity analyses presented in Chapter 4.

Research question: Which characteristics of individual urban green space patches explain variation in small mammal community and species responses?

It was hypothesised that habitat quality and structural complexity exert stronger influences on these communities than patch area alone. It was predicted that:

1. Intensive grassland management and high path density negatively impact communities by reducing vegetation structure and increasing disturbance.
2. Structurally complex features, such as hedgerows and scrub margins, positively affect CPUE and richness by increasing refuge availability.
3. Species-specific responses vary by ecological specialisation. Under this final prediction, it was expected that structurally dependent species (*Myodes glareolus* and *Microtus agrestis*) would show stronger positive associations with woody vegetation and less frequently mown grassland than the generalist *Apodemus sylvaticus*.

3.2 Methods

3.2.1 Study area

Stoke-on-Trent and the adjacent borough of Newcastle-under-Lyme, located in the West Midlands of England, United Kingdom (UK), were selected as the study area due to their highly heterogeneous and fragmented post-industrial urban landscape (Chapter 2.2). The conurbation covers approximately 304.5 km^2 and supports a population of approximately 390,000 people (Office for National Statistics, 2022). The landscape is characterised by a mosaic of urban green spaces embedded within a heavily modified matrix of sealed surfaces, transport infrastructure, residential development, and reclaimed industrial land (Bixby et al., 2015; Stoke-on-Trent City Council, 2013, 2026). This spatial configuration has resulted in substantial variation in habitat composition, vegetation structure, patch connectivity, and management intensity across the urban environment.

Approximately 49% of the study area is classified as open green space, comprising a mixture of amenity grassland, semi-natural habitats, post-industrial successional vegetation, canal corridors, railway verges, and remnant woodland patches (Bixby et al., 2015; Stoke-on-Trent City Council, 2013, 2026). As outlined in Chapter 2, this diversity of habitat conditions and management regimes provides an appropriate urban system for investigating how patch-scale habitat characteristics and management intensity influence small mammal community structure within fragmented urban landscapes.

3.2.2 Study area extent

To establish a consistent spatial extent for patch selection and subsequent habitat analysis, a uniform study area layer was generated in QGIS (v.3.28) by merging the

administrative boundaries of Stoke-on-Trent and Newcastle-under-Lyme (Office for National Statistics, 2026). To ensure the research focused on the urban and peri-urban matrix, the combined vector layer was clipped using the Ordnance Survey (OS) Built-Up Area (BUA) data (Ordnance Survey, 2026a), excluding the predominantly agricultural regions of western Newcastle-under-Lyme. Green space data were then integrated using the OS Open Greenspace layer (Ordnance Survey, 2026b), local authority green space data obtained from Stoke-on-Trent City Council (Stoke-on-Trent City Council, 2021), and regional habitat assets identified through the Natural England Green Space dataset (Natural England, 2025). The resulting boundaries were dissolved to form the spatial framework used throughout the research.

Urban green spaces within Stoke-on-Trent and Newcastle-under-Lyme were identified and mapped in QGIS (v.3.28). The combined spatial layer included formal and informal green space polygons across the urban matrix, including public parks, amenity grasslands, semi-natural habitats, green corridors, and post-industrial land. The green space datasets were clipped to the study area extent and merged to form a unified green space layer.

Candidate green space patches were restricted to a 5 km radius surrounding the central urban extent of Stoke-on-Trent. This buffer was selected to capture the primary urban matrix while retaining variation in habitat composition, distance to surrounding green space, and grass management intensity across the landscape. Retained patches were classified according to grassland management intensity using Stoke-on-Trent City Council management guidance (Stoke-on-Trent City Council, 2024), publicly available management schedules, preliminary field assessments, and

preliminary meetings with Stoke-on-Trent City Council (see Appendices B and D). Management intensity was used as a surrogate measure of anthropogenic disturbance through mowing within each patch.

Three management categories were established, forming a gradient of grass mowing intensity (Table 3.1). Intensively managed patches (Intensity = 3) were characterised by frequent mowing and vegetation maintenance associated with amenity use and formal recreation. Moderately managed patches (Intensity = 2) were subject to intermittent or partial vegetation management, typically retaining areas of taller grassland, scrub, or unmanaged margins. Low-intensity patches (Intensity = 1) comprised habitats subject to minimal or no active vegetation management, including post-industrial land, unmanaged corridors, and naturally regenerating habitats.

Table 3.1 Classification of grassland management intensity across surveyed urban green spaces. Data derived and adapted from Stoke-on-Trent City Council, 2024.

Management Intensity	Category Name	Mowing Frequency	Typical Site Typology	Vegetation Characteristics
3	High	Every 2-4 weeks (High frequency)	Formal Parks (e.g. National Lottery Funded), Residential Estates, Playgrounds.	Uniformly short turf; high "aesthetic" maintenance; removal of arisings; minimal structural complexity.
2	Moderate	Every 6-10 weeks or "as required"	Non-naturalised open land, closed churchyards, green corridors.	Intermittent management; mosaic of short grass and taller margins; maintenance often restricted to fire breaks/paths.
1	Low	Twice annually or not at all	Countryside sites, Local Nature Reserves (LNRs), Post-industrial/Derelict land.	Naturally regenerating vegetation; high structural complexity; presence of successional scrub; "rewilded" characteristics.

Green spaces managed exclusively for organised sport, including dedicated football pitches, were excluded because their open layout and intensive public use would have made traps conspicuous and vulnerable to disturbance or theft. School grounds and inaccessible private land were also excluded from the candidate-site pool. Remaining candidate patches were inspected to confirm that they contained accessible vegetated habitat suitable for repeated live trapping and that traps could be deployed and checked safely throughout all five survey sessions. Following stratification, seven patches were randomly selected from each management category using the “Random Extract Within Subsets” tool in QGIS, resulting in 21 surveyed urban green space patches distributed across the study area (Figure 3.1). This stratified-random sampling design ensured representation across the urban management gradient, with surveyed patches distributed across the study area.

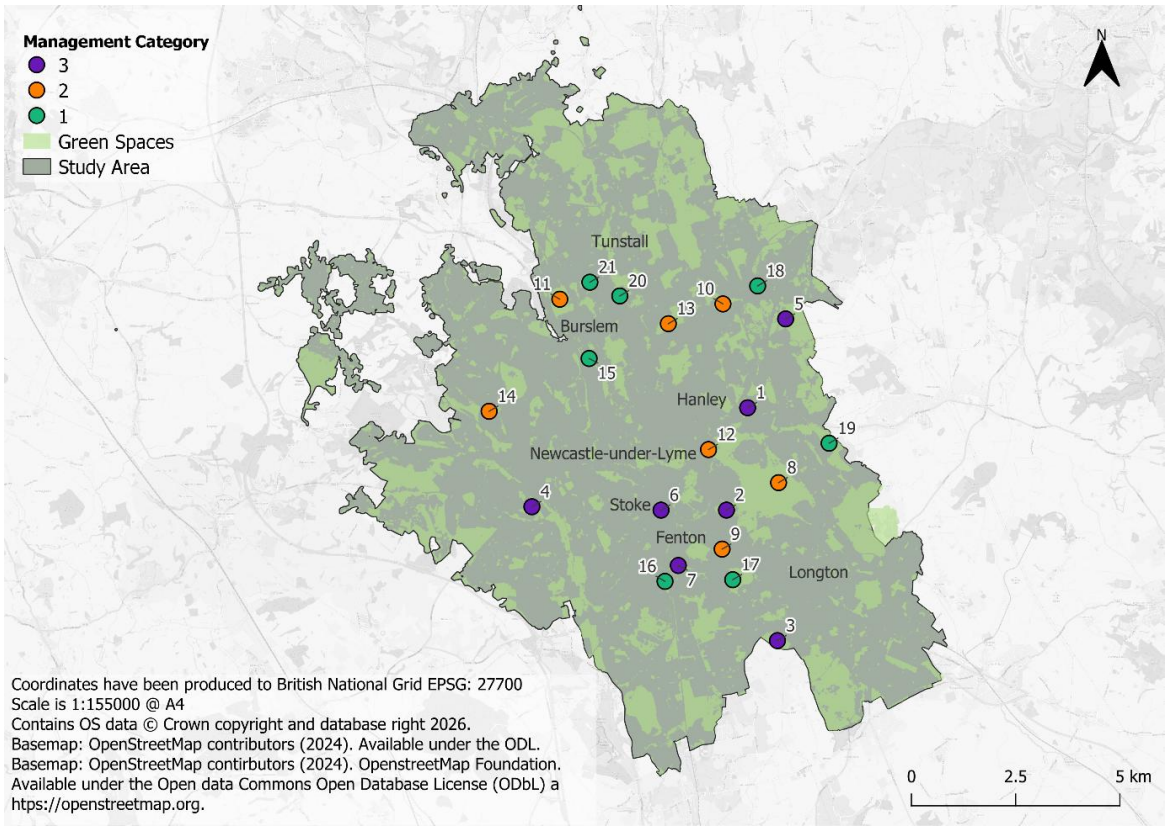


Figure 3.1 Spatial distribution of the 21 surveyed urban green space patches across Stoke-on-Trent and Newcastle-under-Lyme. Green spaces within the study area extent are shown in light green. Surveyed sites are categorised according to grassland management intensity (Green = 1, Orange = 2, Purple = 3).

Final boundaries for the 21 surveyed patches were established during patch selection and preliminary field validation. Patch polygons represented the accessible, functional extent of each surveyed green space patch and were used for all subsequent spatial extractions. Patch area (m^2) was calculated directly from each polygon. Because patch area varied substantially and was strongly right-skewed, area was log-transformed prior to statistical modelling to improve linearity and reduce leverage from exceptionally large patches.

3.2.3 GIS-derived variables

The spatial datasets used throughout the analysis included the UKCEH Land Cover Map 2021 (10 m Classified Pixels, GB) (Marston et al., 2022) and Ordnance Survey Open Greenspace data (Ordnance Survey, 2026b). All spatial analyses were conducted in QGIS (v.3.28) using the British National Grid coordinate system (EPSG:27700). Habitat cover was calculated from the UKCEH Land Cover Map 2021, while linear features were mapped using aerial imagery and information collected during field surveys. All variables were calculated within the final boundary of each surveyed patch.

Patch-scale habitat composition was calculated using proportional habitat cover (%) derived from the UKCEH Land Cover Map 2021 (10 m Classified Pixels, GB) (LCM2021) raster dataset (Marston et al., 2022). Prior to extraction, the national raster dataset was clipped to the Stoke-on-Trent and Newcastle-under-Lyme study area boundary to improve processing efficiency and ensure spatial consistency across the analysis.

Habitat extraction for the 21 surveyed patches was conducted using the QGIS Zonal Histogram tool. Final patch polygons were used as the extraction boundaries, allowing raster cells within each patch to be counted directly. The Zonal Histogram output recorded the number of 10 m × 10 m pixels assigned to each LCM2021 land cover class within each patch. As each raster pixel represented 100 m^2, pixel counts were converted to habitat area by multiplying the number of pixels in each class by 100.

Habitat area was therefore calculated as:

$$Habitat\ area\ (m^2)\ =\ pixel\ count\ \times\ 100$$

Habitat area was converted to hectares by dividing by 10,000, equivalent to multiplying the pixel count by 0.01. Percentage cover was then calculated by dividing the pixel count for each land cover class, or grouped habitat category, by the total number of classified pixels within the patch and multiplying by 100:

$$Habitat\ cover\ (\%) = \left(\frac{habitat\ class\ pixel\ count}{total\ classified\ patch\ pixel\ count}\right) X\ 100$$

For statistical modelling, habitat composition variables were retained as proportional cover values ranging from 0 to 1, calculated as the habitat class pixel count divided by the total classified pixel count within each patch. Expressing habitat composition as proportional cover enabled direct comparisons among surveyed sites despite substantial variation in patch size (1.4-79.6 ha).

The habitats identified within Stoke-on-Trent and Newcastle-under-Lyme are described in Chapter 2, Table 2.1. Following review of the LCM2021 class structure, no raster-derived scrub or dense structural cover variable was retained. LCM2021 does not provide a discrete scrub class, and the classes initially used to approximate dense structural vegetation corresponded to Heather and Heather grassland rather than scrub (Marston et al., 2022). These classes were also extremely sparse within the surveyed patches and were therefore unsuitable for estimating a general patch-scale effect. They were removed from the final model.

The final retained habitat composition variables were grassland cover and tree cover. Freshwater cover was calculated from LCM2021 but was not retained in the final model because water area within patches was limited and was considered less

biologically informative than watercourse density for the retained species, which better represented linear riparian features and potential movement corridors (Anthony et al., 1987; Davies et al., 2011). Sealed surface cover was calculated but was not retained in the final model to avoid compositional multicollinearity among proportional land cover variables. Path density was subsequently retained in favour of sealed cover, which provided a more direct within-patch measure of anthropogenic disturbance.

Table 3.2 describes how UKCEH LCM2021 land cover classes were grouped to produce patch-scale habitat composition variables for analysis within urban green space patches. Habitat classes were converted to proportional cover within each patch and reviewed for ecological relevance, interpretability and suitability for inclusion in the final models.

Table 3.2 Reclassification and Treatment of UKCEH Land Cover Map (LCM2021) Classes

LCM2021 Class(es)	Original Habitat Classification	Derived Variable	Final Treatment	Ecological & Statistical Justification
1, 2	Broadleaved woodland; Coniferous woodland	Tree Proportion	Retained	Measures large-scale, patch-level wooded habitat structure and canopy cover.
3, 4, 5, 6, 7	Arable and horticulture; Improved, neutral, calcareous, and acid grassland	Grass Proportion	Retained	Represents open foraging habitat; directly tests the grassland management intensity hypothesis.
9, 10	Heather; Heather grassland	N/A	Omitted	Structurally distinct from scrub and extremely rare within the dataset (near-zero variance; present at only one site).
14	Freshwater	Water Proportion	Omitted	Standing water area was negligible; linear Watercourse density was retained as a more ecologically relevant corridor variable.
20, 21	Urban; Suburban	Sealed Proportion	Omitted (Baseline)	Removed from the model formula to avoid compositional multicollinearity among proportional land cover predictors. Serves as the background urban baseline.

Structural variables were measured by manually digitising linear features in QGIS (v.3.28) following field surveys. Linear features were mapped using high-resolution aerial imagery and field observations within the boundary of each surveyed patch. Features included hedgerows, scrub margins, treelines, paths, and watercourses.

Linear tree features, hedgerows, and scrub margins were combined into a single "woody linear" variable representing structurally complex refuge habitat and

potential movement corridors for small mammals, with the prefix 'L' (Atkins et al., 2018; Galantinho et al., 2020). Total feature length (m) within each patch was calculated using the QGIS geometry calculator and standardised by patch area to provide a density measure (m ha^{-1}):

$$Woody\ Linear\ Density = \frac{LTree + LHedge + LScrub\ (m)}{Site\ Area\ (ha)}$$

Watercourses and paths were retained as separate structural variables. All linear variables were expressed as density measures (m ha^{-1}) relative to patch area to account for variation in patch size and allow comparison among surveyed patches. When multiple features of the same class occurred within a patch, their total length was summed before calculating density:

$$Linear\ Feature = \frac{Length\ of\ feature(s)\ (m)}{Site\ Area\ (ha)}$$

3.2.4 Field survey methods

Live-trapping surveys were conducted across the 21 surveyed patches using a repeated sampling design. Each patch was surveyed during five separate sessions between May and October (2021-2022), encompassing the primary active period for small mammals in temperate environments (Gurnell & Flowerdew, 1982; Gurnell et al., 2019). Each survey session comprised three consecutive trap nights, with successive sessions at each patch separated by three to four weeks. All trapping, handling and species identification were undertaken by the author, whose relevant training and experience are described in Section 2.8, in accordance with institutional

ethical approval and the conditions of Natural England GL01 for the trapping of shrews (Appendix C).

Small mammal trapping was conducted using standard Longworth live traps (Longworth Scientific Instrument Co., Abingdon, UK). Each trap comprised an entry tunnel containing a step-on wire trip-door mechanism and a detachable nest box. Nest boxes were filled with dry hay bedding to reduce thermal stress during confinement (Gurnell et al., 2019). Traps were baited with fresh carrot, apple, mixed grain, seed and nuts, and greenbottle fly pupae (*Lucilia sericata*) to provide food for captured rodents and insectivorous shrews.

To account for variation in patch size among survey sites, which ranged from 1.4 to 79.6 ha, sampling effort was scaled using broad categories based on the final field-validated patch area. Ten traps were deployed at patches up to 5 ha, 20 traps at patches larger than 5 ha and up to 10 ha, 30 traps at patches larger than 10 ha and up to 20 ha, and 40 traps at patches larger than 20 ha and up to 40 ha. The single patch larger than 40 ha, which covered 79.67 ha, received 60 traps. Trap numbers were therefore determined by patch area, whereas trap positions were restricted to suitable and accessible vegetated habitat within each patch. The maximum allocation of 60 traps balanced spatial coverage with field practicality, animal welfare, equipment availability and handling constraints.

Trap placement was designed to sample the focal small mammal community within accessible vegetated habitat across urban green spaces differing in grassland management intensity. The design was not intended to estimate population density within individual habitat types or to sample every mapped land cover class in

proportion to its availability within each patch. Suitable trapping habitat was classified into three broad placement contexts: grassland, including mown, rough and rank grassland; woodland; and woody-edge vegetation, including scrub margins, woodland edges, hedgerow bases, woody boundary features and successional woody vegetation. These contexts represented the ground-level vegetation considered relevant to the focal taxa (Fuentes-Montemayor et al., 2020; Dondina et al., 2025). Grassland was incorporated into the trapping layout at all 21 patches, while woodland and woody-edge vegetation were incorporated where present and accessible.

The placement contexts described the immediate habitat surrounding the traps and were distinct from the patch-wide habitat proportions derived through GIS. They were not always spatially discrete, and individual transects could extend across adjoining contexts, for example from grassland into a woody edge. Trap effort was not allocated equally among habitat contexts or in direct proportion to their area within each patch. Instead, transect locations were independently re-randomised during each of the five survey sessions within the suitable and accessible portions of the habitat contexts present. This represented a constrained stratified-random design rather than unrestricted random sampling across the entire patch (Greenwood & Robinson, 2006). Re-randomising locations between sessions increased cumulative spatial coverage and reduced dependence on a single set of trap locations, while retaining the same placement rules during every repeat survey.

Once positioned, traps remained fixed for the three consecutive nights of that survey session and were placed at 10 m intervals. Traps were arranged in transects

containing either five or ten traps. In narrow, irregularly shaped or obstructed patches, transects were segmented, cornered or arranged in a zig-zag formation while retaining the 10 m inter-trap spacing. Where a transect was interrupted by an unsuitable feature, it was terminated and continued within the next area of suitable vegetation. Sealed surfaces, paths, buildings, bare ground, hardstanding, open water and unsafe or inaccessible areas were excluded from the sampling frame.

The resulting CPUE and community-diversity measures were interpreted as standardised indices of focal-species capture activity within the accessible vegetated habitat of each patch, rather than area-weighted estimates of abundance across the entire patch. Patch-wide habitat composition was quantified independently through GIS and included in the statistical analysis to distinguish the wider composition of each green space from the immediate habitat contexts sampled by the trapping layout.

Table 3.3 summarises trapping effort, patch-level habitat composition and the broad habitat contexts sampled by the trapping layout. Grassland was incorporated at all 21 sites, while woodland and/or woody-edge habitat was incorporated at 19 sites. These categories identify the habitats crossed by one or more transects rather than the number or proportion of individual traps placed within each habitat. Trap positions were selected using the stratified-random procedure described above within the suitable vegetated habitats present in each patch.

Table 3.3 Longworth trapping effort and patch-level habitats across surveyed urban green space patches. Habitat percentages describe the wider composition of each surveyed patch, while “Habitats Within Transect(s)” records the broad habitats incorporated into the Longworth trapping layout.

Site ID	Management	Area ha	Transect clusters	Traps per repeat	Nominal trap nights	Adjusted trap nights	Grass %	Tree %	Sealed %	Water %	Habitats Within Transect(s)
1	3	9.05	2	20	300.0	293.5	60.9	14.9	23.9	0.3	Grassland/Woody Edge
2	3	14.44	3	30	450.0	438.5	52.2	35.1	12.7	0.0	Grassland/Woody Edge
3	3	19.11	3	30	450.0	442.0	42.3	33.3	17.1	7.3	Grassland/Woody Edge
4	3	10.67	3	30	450.0	441.5	45.2	32.1	22.7	0.0	Grassland/Woodland/Woody Edge
5	3	38.81	4	40	600.0	587.0	59.3	19.8	20.8	0.1	Grassland/Woody Edge
6	3	2.86	1	10	150.0	127.0	68.1	4.9	27.0	0.0	Grassland/Woody Edge
7	3	8.77	2	20	300.0	282.0	65.9	32.9	1.2	0.0	Grassland/Woodland
8	2	79.67	6	60	900.0	894.5	93.5	2.9	3.4	0.2	Grassland/Woody Edge
9	2	6.95	2	20	300.0	284.0	31	59.6	9.4	0.0	Grassland/Woodland/Woody Edge
10	2	5.88	2	20	300.0	280.5	9.1	33.5	11.9	45.5	Grassland/Woodland/Woody Edge
11	2	36.03	4	40	600.0	583.0	27.7	14.4	19.5	38.4	Grassland/Woodland/Woody Edge
12	2	2.13	2	10	150.0	144.0	52.7	23.8	23.5	0.0	Grassland/Woody Edge
13	2	15.99	3	30	450.0	448.0	20.6	71.2	8.2	0.0	Grassland/Woodland/Woody Edge
14	2	1.42	1	10	150.0	142.0	79.0	0.0	21	0.0	Grassland
15	1	8.00	2	20	300.0	291.5	75.8	22.0	2.0	0.2	Grassland/Woody Edge
16	1	5.88	2	20	300.0	291.0	51.3	21.2	3.2	24.3	Grassland/Woody Edge
17	1	5.84	2	20	300.0	297.0	24.8	75.0	0.0	0.2	Grassland/Woodland/Woody Edge
18	1	5.31	2	20	300.0	299.0	16.4	83.5	0.0	0.1	Grassland/Woodland/Woody Edge
19	1	3.07	1	10	150.0	147.0	98.9	0.0	1.1	0.0	Grassland
20	1	1.92	1	10	150.0	122.0	63.6	23.0	13.4	0.0	Grassland/Woody Edge
21	1	3.99	1	10	150.0	131.0	37.6	52.1	10.3	0.0	Grassland/Woody Edge

All capture and handling were carried out in accordance with standard small mammal welfare guidelines and institutional ethical approvals (Gurnell & Flowerdew, 1982; Gurnell et al., 2019). Traps were set for three consecutive nights and checked twice daily, at 07:00 h and 19:00 h, ensuring animals were not confined for more than 12 hours. Captured individuals were transferred to handling bags, identified to species level and marked using a unique hair-clipping method at each transect to allow short-term capture-recapture tracking over consecutive nights. All individuals were released immediately at their point of capture.

Because sampling effort varied from ten to sixty traps across the patch-size gradient, raw capture numbers could not be compared directly among surveyed patches. Capture data were therefore converted into a CPUE index expressed as the Trap Night Index (TNI), following the adjusted trap-night approach of Nelson and Clark (1973).

Nominal trap nights per survey session were calculated as the total number of active traps multiplied by the number of trapping nights. To account for sprung, closed or otherwise non-functional traps, adjusted trap nights were calculated by subtracting 0.5 trap nights for every trap found closed without a capture. The standardised metric was therefore calculated as:

$$TNI = \left(\frac{Total\ Captures}{Adjusted\ Trap\ Nights}\right) X\ 100$$

This standardisation expressed relative capture response as captures per 100 adjusted trap nights, reducing variation associated with unequal trapping effort among patches. Species-specific CPUE was calculated separately for each Site ID × Repeat observation using captures of the relevant focal species as the numerator and

adjusted trap nights for that observation as the denominator. Focal-community CPUE used the combined captures of the four focal species as the numerator and the same adjusted trap-night denominator. These 105 patch-repeat values supplied the response data used in the subsequent models. Simpson's reciprocal diversity was calculated from species-specific focal CPUE values within each patch-repeat observation. Unweighted means ± SE were used only for descriptive reporting.

Across the 21 surveyed patches and five repeated surveys, the verified trapping allocation produced 7,200 nominal trap nights. Following adjustment for traps found closed, sprung or otherwise unavailable without a capture, the effective sampling effort was 6,966 adjusted trap nights. These figures were recalculated directly from the original patch and survey-level trapping records and are reported separately throughout the revised thesis.

3.2.5 Statistical analysis

All statistical analyses were conducted in R (v.4.6.0) using the packages glmmTMB (Brooks et al., 2017), DHARMa (Hartig, 2022), performance (Lüdecke et al., 2021), car (Fox & Weisberg, 2019), ggeffects (Lüdecke, 2018), and broom.mixed (Bolker & Robinson, 2022). Analyses were undertaken at the patch-repeat level, with each surveyed green space patch contributing repeated observations across five survey sessions. Site identity was included as a random effect to account for non-independence among repeated surveys from the same patch.

3.2.5.1 Response variables and model formulation

Species richness was calculated as the number of focal small mammal species recorded within each patch-repeat survey. For each focal species, presence within a

repeat was defined as one or more captures during that survey session. Richness therefore represented the number of focal species detected during each repeat, rather than the cumulative number recorded across all five surveys.

Total community capture activity was expressed as total CPUE (CPUEtotal). Species-level CPUE was calculated using adjusted trap nights and expressed as captures per 100 adjusted trap nights. CPUEtotal was then calculated for each patch-repeat by summing species-level CPUE values across all focal species recorded during that survey session:

$$CPUEtotal = \sum_{i=1}^{S} CPUEi$$

where CPUE_i is the CPUE of focal species i, and S is the number of focal species included in the analysis.

Species-specific CPUE was calculated for each Site ID × Repeat observation using the number of captures of that focal species as the numerator and adjusted trap nights for that observation as the denominator. Focal-community CPUE used the combined captures of the four focal species as its numerator and the same adjusted trap-night denominator. This produced 105 patch-repeat observations, which formed the response data used in Chapters 3-5.

Community diversity, incorporating both the richness and relative distribution of focal-species capture activity, was measured using Simpson's reciprocal diversity index. The proportional contribution of each focal species to total CPUE within each patch-repeat survey was calculated as:

$$pi = \frac{CPUEi}{CPUEtotal}$$

where $p_i$is the proportional CPUE contribution of species i, $CPUE_i$ is the CPUE of species i within that site-repeat survey, and $CPUE_{total}$ is the summed CPUE of all focal species recorded within that same survey session.

Simpson's dominance index (D) was then calculated as:

$$D = \sum_{i=1}^{S} p_i^2$$

and Simpson's reciprocal diversity index as:

$$\frac{1}{D} = \frac{1}{\sum_{i=1}^{S} p_i^2}$$

Higher values of $\frac{1}{D}$ indicate greater community diversity and a more even distribution of capture activity among focal species, whereas a value of 1 indicates dominance by a single species. Patch-repeat surveys with no focal-species captures, for which p_i and $\frac{1}{D}$ could not be calculated, were assigned a Simpson's reciprocal diversity value of 0.

Patch-scale predictor variables included log-transformed patch area, proportional grassland cover, proportional tree cover, woody linear feature density, watercourse density, path density, grassland management intensity, and the interaction between grassland cover and management intensity. These variables represented patch size,

habitat composition, structural complexity, potential refuge corridors, local disturbance, and grassland management intensity.

Continuous predictor variables were standardised using z-score transformation prior to modelling to improve convergence and allow comparison of relative effect sizes among predictors (Schielzeth, 2010; Kuhn & Johnson, 2013). Management intensity was modelled as an ordered numeric gradient representing increasing grassland management intensity and was also standardised prior to analysis.

Multicollinearity among retained predictors was assessed using Spearman's rank correlations and variance inflation factors (VIFs). Predictor pairs with an absolute correlation coefficient of $|\rho| \geq 0.70$ were reviewed for redundancy. VIF values greater than ten were treated as indicating problematic multicollinearity, while values between five and ten were considered moderate and assessed in relation to the ecological distinctness of the predictors.

Generalised Linear Mixed Models (GLMMs) were fitted using the glmmTMB package. The same predictor structure was applied across the community-level models, including log-transformed patch area, grassland cover, tree cover, woody linear density, watercourse density, path density, management intensity, and the interaction between grassland cover and management intensity. Site ID was included as a random intercept to account for repeated survey observations within each patch.

The subsequent GLMMs were formulated as:

$$Response \sim logSiteArea + GrassProportion + TreeProportion +$$

$$WoodyLinearDensity + WatercourseDensity + PathDensity + Management +$$

$$GrassProportion{:}\,Management + (1|SiteID)$$

Species richness was modelled using a Poisson error distribution with a log-link function. Total CPUE was modelled using a Tweedie distribution with a log-link function due to right-skew and overdispersion (Dunn & Smyth, 2005). Simpson's reciprocal diversity index was treated as a continuous response and modelled using Gaussian errors based on data distribution checks.

An interaction between grassland cover and management intensity was included to test whether the effect of grassland cover varied with mowing intensity. Model formulations and full outputs are provided in Appendix E.

Model assumptions and fit were evaluated using simulated residual diagnostics generated with the DHARMa package (Hartig, 2022). Residual simulations were used to assess dispersion, residual uniformity, and zero inflation across all fitted models. Predictor collinearity was further evaluated using the performance package (Lüdecke et al., 2021). Model outputs were visualised using ggeffects and ggplot2 to generate predicted response curves and standardised coefficient plots (Lüdecke, 2018).

3.3 Results

3.3.1 Capture summary and focal species

Across 6,966 adjusted trap nights, 765 captures were recorded giving an overall CPUE of 10.98 captures per 100 adjusted trap nights. The captures represented seven

different species across the 21 urban green space patches: 476 *Apodemus sylvaticus*, 137 *Microtus agrestis*, 56 *Mus musculus*, 55 *Myodes glareolus*, 35 *Sorex araneus*, 4 *Neomys fodiens*, and 2 *Apodemus flavicollis* (Melchior, 1834) captures.

To assess the impact of grass management and patch characteristics, the analysis focused on four focal species: *M. agrestis*, *M. glareolus*, *A. sylvaticus*, and *S. araneus*, as described in Chapter 2. These species provided sufficient capture frequencies for statistical comparison. Three species were excluded from further analysis: *N. fodiens* and *A. flavicollis* due to low capture rates, and *M. musculus* to maintain a focus on species indicative of natural ecosystem functioning (Bouilloud et al., 2024).

The four focal species accounted for 703 captures (91.89% of all captures). Their pooled CPUE across the complete survey was 10.09 captures per 100 adjusted trap nights (703/6,966 × 100). For the statistical analyses, CPUE was calculated separately for each of the 105 patch-repeat observations. The unweighted mean of these observation-level focal-community CPUE values was 11.79 ± 0.77. This differs from the pooled CPUE because each patch-repeat observation contributes equally to the mean irrespective of its trapping effort. The 105 individual patch-repeat CPUE values supplied the response data used in Chapters 3-5.

Table 3.4 Summary of captures and patch-repeat CPUE across 21 urban green spaces surveyed on five occasions. Proportion of all captures was calculated using all 765 captures. Mean patch-repeat CPUE values are unweighted means ± SE across the 105 Site ID × Repeat observations. M1, M2 and M3 represent low, moderate and intensive management sites, respectively. 'Mean PR' represents mean patch-repeat CPUE. Non-focal species were excluded from the CPUE response used in Chapters 3-5.

Species	Total Capture	Capture Proportion (%)	Mean PR CPUE ± SE	M1 CPUE ± SE	M2 CPUE ± SE	M3 CPUE ± SE
A. sylvaticus	**476**	**62.22**	**8.17 ± 0.62**	**12.87 ± 1.27**	**7.75 ± 0.77**	**3.91 ± 0.28**
M. agrestis	**137**	**17.91**	**2.29 ± 0.21**	**2.92 ± 0.43**	**2.40 ± 0.41**	**1.53 ± 0.20**
M. glareolus	**55**	**7.19**	**0.81 ± 0.21**	**0.49 ± 0.26**	**0.96 ± 0.46**	**1.05 ± 0.35**
S. araneus	**35**	**4.58**	**0.52 ± 0.08**	**0.69 ± 0.21**	**0.45 ± 0.12**	**0.34 ± 0.12**
M. musculus	56	7.32	-	-	-	-
N. fodiens	4	0.52	-	-	-	-
A. flavicollis	2	0.26	-	-	-	-
Total	765	100.0	11.79 ± 0.77	16.97 ± 1.25	11.55 ± 1.34	6.83 ± 0.60

3.3.2 Overview of habitat and management drivers

Environmental variables tested showed different relationships with the community metrics and the four focal species. At the community level, woody linear density was positively associated with species richness, CPUE and Simpson's reciprocal diversity, while CPUE declined with increasing path density, patch size and management intensity. Significant predictors also differed between the focal species, with woody linear density, grassland cover and watercourse density showing positive relationships, and path density, management intensity and patch size showing negative relationships where effects were detected. Table 3.5 summarises the direction of these relationships, with the full model outputs and effect sizes presented in the following sections.

Table 3.5 Synthesis of significant environmental drivers across community metrics and focal species. Plus (+) and minus (-) signs indicate the direction of significant relationships; blanks indicate non-significance; R indicates Richness, 1/D indicates Simpson's reciprocal diversity.

Variable	**R**	**CPUE**	**1/D**	***A. sylvaticus***	***M. agrestis***	***M. glareolus***	***S. araneus***
Woody linear density	+	+	+	+		+	
Management intensity		-		-			-
Path density		-		-	-		-
Grassland cover		+		+	+		
Patch size		-		-	-		
Watercourse density		+			+		
Tree cover							
Grassland cover × management intensity						-	

3.3.3 Community-level responses to patch characteristics and management

Woody linear density emerged as the most consistent positive predictor across the community models, showing significant positive relationships with species richness, total CPUE, and Simpson's reciprocal diversity. Path density showed the opposite pattern, with a significant negative relationship with CPUE and marginal negative relationships with species richness and Simpson's reciprocal diversity.

CPUE was the community metric most strongly associated with patch-scale habitat and management variables. Capture rates declined with increasing patch size, path density, and management intensity, but increased with grassland cover, woody linear density, and watercourse density. Grassland cover was not significantly associated with richness or Simpson's reciprocal diversity, although the grassland cover x

management intensity interaction was non-significant for both CPUE and richness, indicating a tendency for the positive effect of grassland cover to weaken under increasing management intensity. Tree cover was not significantly associated with any community-level response. The effects of patch-scale environmental variables on species richness, CPUE, and Simpson's reciprocal diversity are summarised in Figure 3.2.

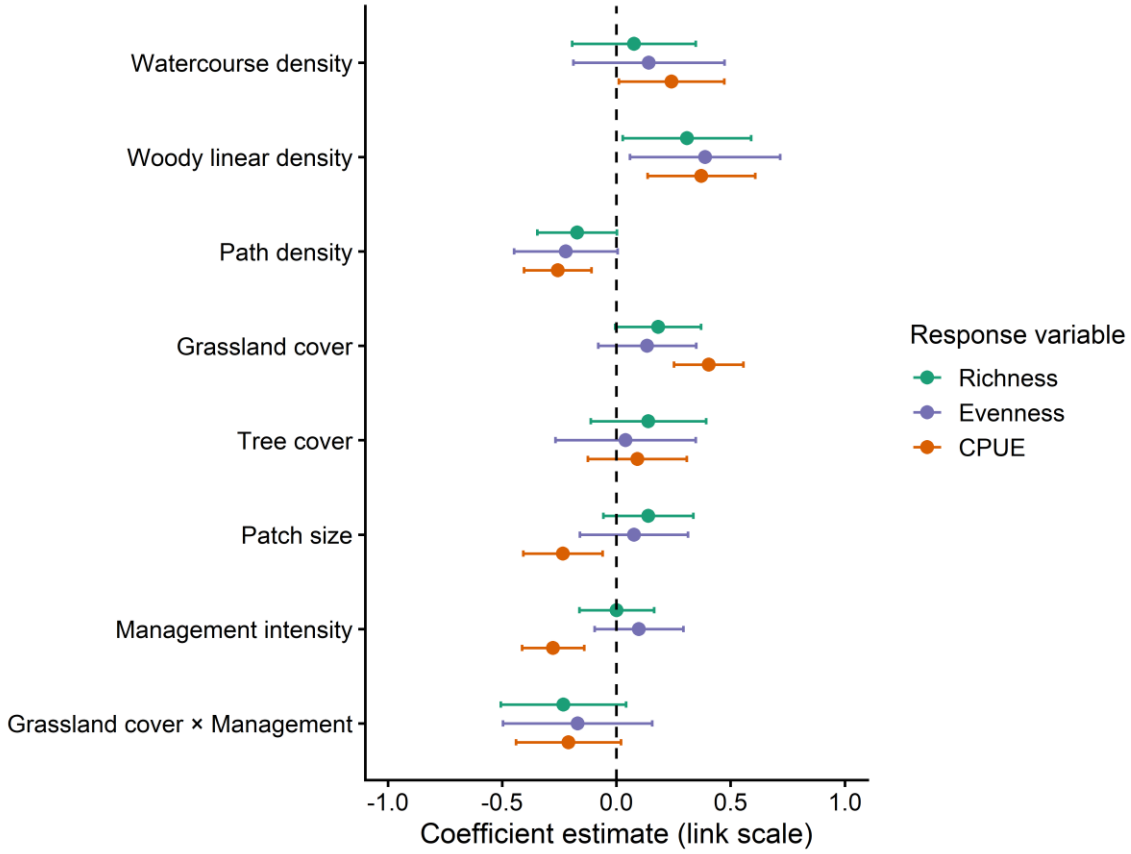


Figure 3.2 Standardised fixed-effect coefficient estimates (± 95% confidence intervals) from mixed-effects models examining environmental drivers of small mammal species richness, catch per unit effort (CPUE), and Simpson's reciprocal diversity across urban green spaces. All predictors were standardised (mean = 0, SD = 1); coefficients therefore represent the effect of a one standard deviation increase in each predictor on the response variable. Points indicate model estimates on the link scale (log link for richness and CPUE; identity link for Simpson's reciprocal diversity), with horizontal bars representing 95% confidence intervals. The dashed vertical line indicates no effect (β = 0). The grass × management term represents the interaction between grass cover and grass management intensity, describing how management modifies the effect of grass-dominated habitat on community metrics.

Woody linear density was the only predictor positively associated with all three community metrics. Species richness increased significantly with increasing woody linear density (β = 0.308 ± 0.144 SE; χ^2 = 4.61, p = 0.032), alongside CPUE (β = 0.372 ± 0.120 SE; χ^2 = 9.53, p = 0.002) and Simpson's reciprocal diversity (β = 0.388 ± 0.168 SE; χ^2 = 5.35, p = 0.021). Predicted responses indicated simultaneous increases in richness, CPUE, and Simpson's reciprocal diversity across the woody linear density gradient (Figure 3.3).

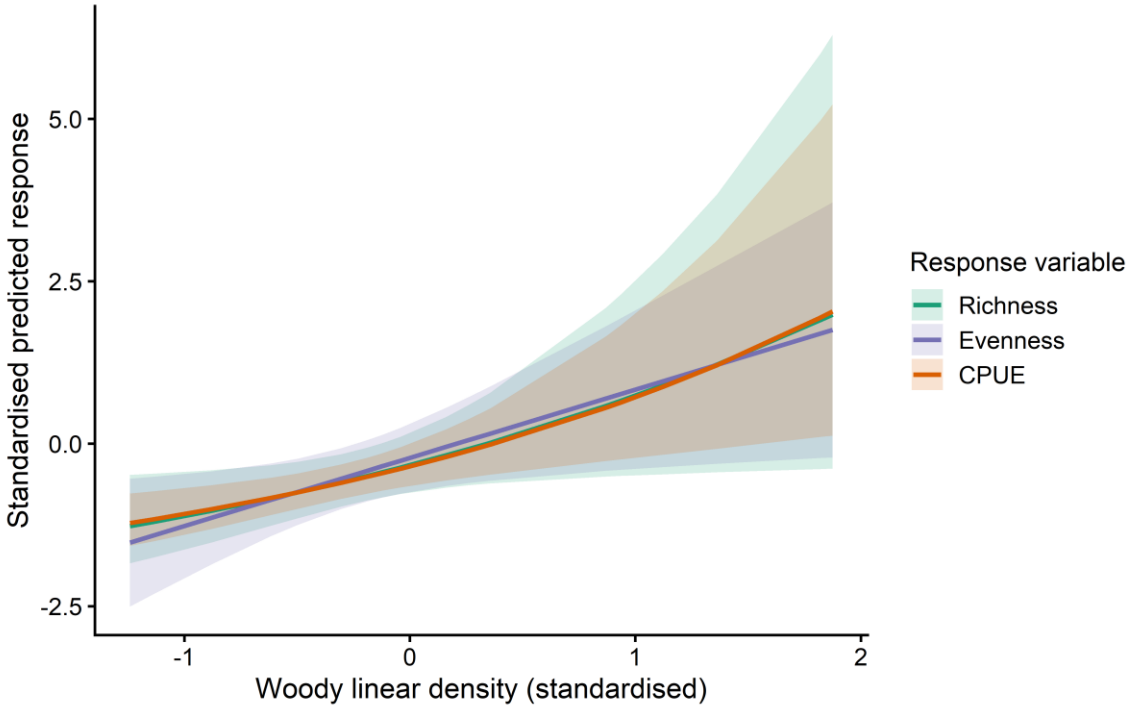


Figure 3.3 Predicted responses of species richness, catch per unit effort (CPUE), and Simpson's reciprocal diversity (1/D) across the observed gradient of woody linear density. Lines represent model-predicted responses from mixed-effects models with all other predictors held at their mean values. Shaded areas represent 95% confidence intervals. Increasing woody linear density was associated with positive responses across all three community metrics.

Path density showed the opposite pattern. CPUE declined significantly with increasing path density (β = -0.257 ± 0.075 SE; χ^2 = 11.64, p < 0.001), while species richness (β = -0.172 ± 0.089 SE; χ^2 = 3.75, p = 0.053) and Simpson's reciprocal diversity (β = -0.222 ± 0.116 SE; χ^2 = 3.68, p = 0.055) showed weaker negative

relationships. These results indicate that patches with greater woody linear structure and lower internal path density supported more favourable community responses among the focal species.

Management intensity, measured as mowing frequency, was negatively associated with CPUE ($\beta = -0.278 \pm 0.069$ SE; $\chi^2 = 19.28$, $p < 0.001$), indicating lower capture rates in more intensively managed grassland patches. In contrast, management intensity was not significantly associated with species richness ($\chi^2 = 0.11$, $p = 0.738$) or Simpson's reciprocal diversity ($\chi^2 = 0.69$, $p = 0.407$). Site-level distributions of Richness, CPUE and Simpson's reciprocal diversity across the three management categories are shown in Figure 3.4.

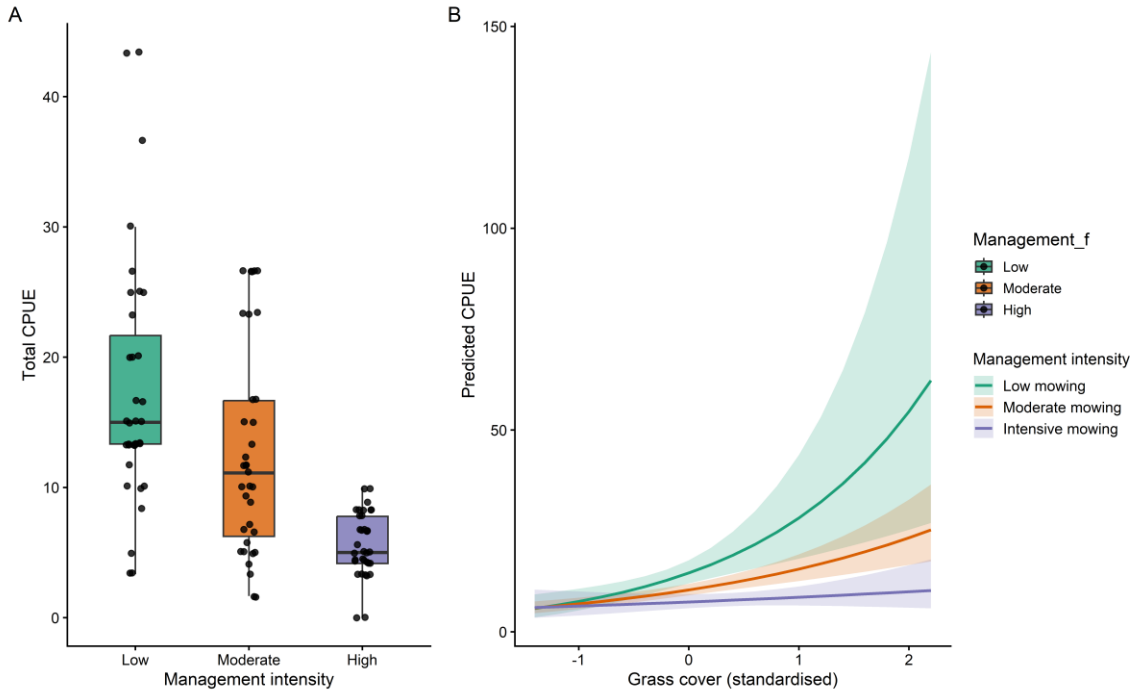


Figure 3.4 Effect of grassland management intensity and grass cover on small mammal CPUE. (A) Boxplots show observed total CPUE across low, moderate and high management categories, with points representing individual patch-repeat surveys. (B) Model-predicted CPUE across standardised grass cover under each management regime, with shaded areas showing 95% confidence intervals. CPUE declined with increasing management intensity, while the positive relationship with grass cover tended to weaken as management intensity increased.

Patch size was negatively associated with CPUE (β = -0.235 ± 0.089 SE; χ^2 = 7.02, p = 0.008), indicating higher capture rates in smaller patches and potentially reflecting density-compensation or resource concentration in restricted spaces. Patch size was not significantly associated with species richness (χ^2 = 1.91, p = 0.167) or Simpson's reciprocal diversity (χ^2 = 0.40, p = 0.527).

Grassland cover was positively associated with CPUE (β = 0.404 ± 0.078 SE; χ^2 = 24.36, p < 0.001) but was not significantly associated with species richness (χ^2 = 2.31, p = 0.129) or Simpson's reciprocal diversity (χ^2 = 0.97, p = 0.326). The grassland cover × management intensity interaction was weak for CPUE (β = -0.210 ± 0.117 SE; χ^2 = 3.21, p = 0.073) and species richness (β = -0.232 ± 0.140 SE; χ^2 = 2.75, p = 0.097), suggesting that the positive effect of grassland cover tended to weaken under increasing management intensity. No interaction effect was detected for Simpson's reciprocal diversity (χ^2 = 1.05, p = 0.307).

Finally, watercourse density was positively associated with CPUE (β = 0.241 ± 0.118 SE; χ^2 = 4.17, p = 0.041) but showed no significant relationship with species richness (χ^2 = 0.31, p = 0.580) or Simpson's reciprocal diversity (χ^2 = 0.70, p = 0.402). Tree cover was not significantly associated with any community-level response variable, with no detectable effect on species richness (χ^2 = 1.18, p = 0.278), CPUE (χ^2 = 0.68, p = 0.408), or Simpson's reciprocal diversity (χ^2 = 0.07, p = 0.798).

3.3.4 Patch characteristics and species-specific responses

To evaluate taxon-specific responses to patch-scale environmental characteristics and management, separate mixed-effects models were fitted for *A. sylvaticus*, *M.*

agrestis, *M. glareolus* and *S. araneus*. Standardised fixed-effect coefficient estimates from the four models are summarised in Figure 3.5.

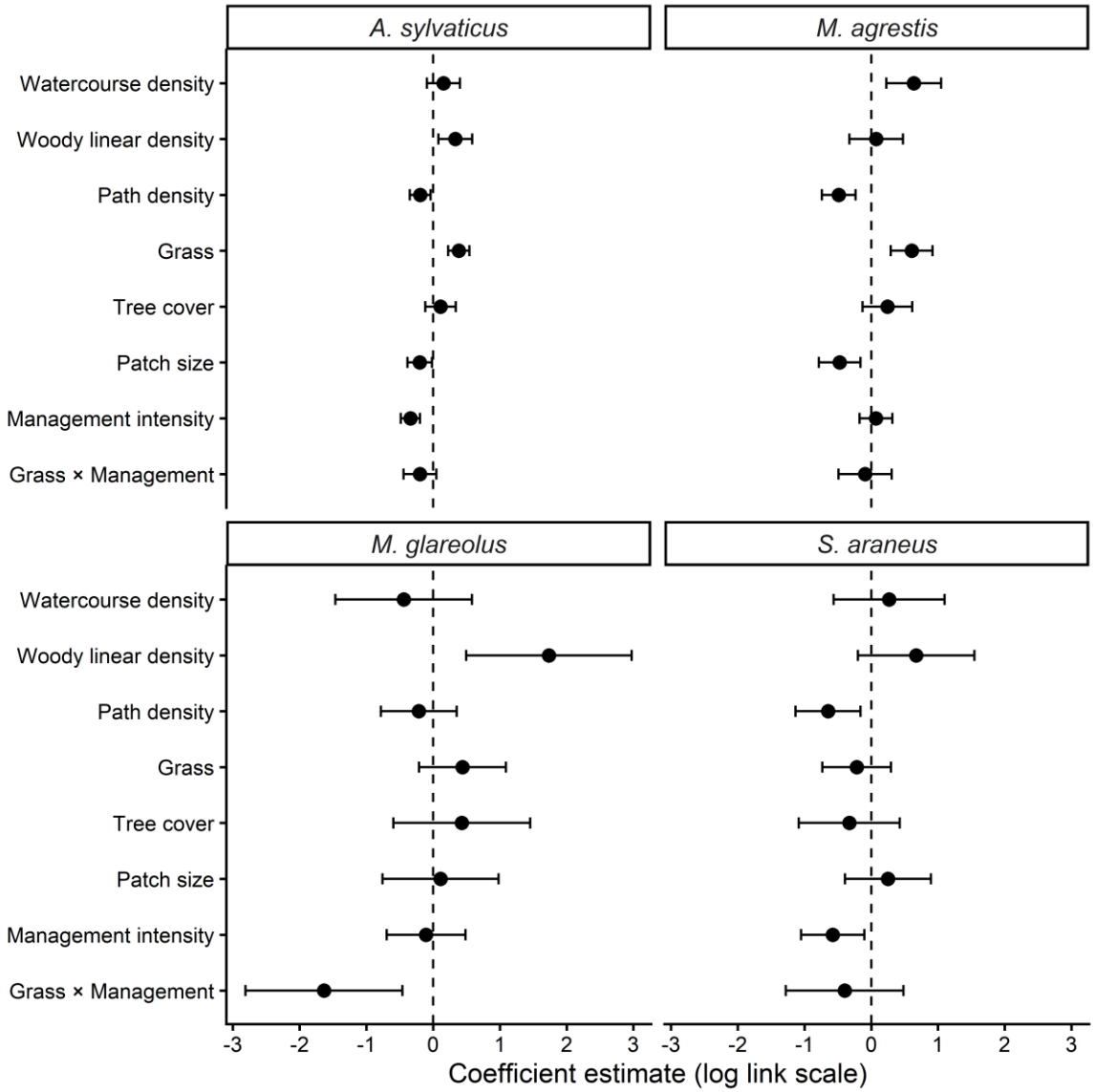


Figure 3.5 Species-specific standardised coefficient estimates (± 95% confidence intervals) from generalised linear mixed models examining environmental predictors of capture rates for four focal small mammal species across urban green spaces. Points represent fixed-effect estimates on the log link scale, with horizontal bars indicating 95% confidence intervals. The dashed vertical line denotes no effect (β = 0). Positive coefficients indicate increased predicted capture rates with increasing predictor values, whereas negative coefficients indicate reduced capture rates.

3.3.4.1 Common environmental drivers

Path density was the most consistent negative predictor across the species-level models. CPUE declined significantly with increasing path density for *A. sylvaticus* (χ^2

= 5.95, p = 0.015), *M. agrestis* (χ^2 = 14.31, p < 0.001) and *S. araneus* (χ^2 = 6.77, p = 0.009), whereas no significant relationship was detected for *M. glareolus* (χ^2 = 0.55, p = 0.460). In contrast, woody linear density was significantly associated with CPUE for *A. sylvaticus* (χ^2 = 6.66, p = 0.010) and *M. glareolus* (χ^2 = 7.54, p = 0.006), but not for *M. agrestis* or *S. araneus*. Tree cover was not a significant predictor for any focal species, indicating that the distribution of woody linear features was more informative than the overall proportion of tree cover within a patch.

3.3.4.2 Species-specific habitat responses

Beyond these shared responses, the focal species differed in the patch characteristics associated with their CPUE. For *A. sylvaticus*, significant effects were also detected for patch size (χ^2 = 4.48, p = 0.034), grassland cover (χ^2 = 20.59, p < 0.001) and management intensity (χ^2 = 24.60, p < 0.001). This species, therefore, showed the broadest response to the measured patch-scale predictors. For *M. agrestis*, significant associations were detected for patch size (χ^2 = 8.93, p = 0.003), grassland cover (χ^2 = 14.75, p < 0.001) and watercourse density (χ^2 = 9.27, p = 0.002), in addition to the negative effect of path density reported above.

The response of *M. glareolus* was more restricted. In addition to its association with woody linear density, CPUE was significantly influenced by the grassland cover × management intensity interaction (χ^2 = 7.45, p = 0.006), indicating that the relationship with grassland cover varied with mowing intensity. For *S. araneus*, management intensity was the only significant predictor additional to path density (χ^2 = 5.21, p = 0.022). Overall, path density represented the most widely shared negative influence, while associations with grassland cover, patch size, watercourses, woody

structure and management differed among species. Full significant and non-significant model outputs are provided in Appendix E.

3.3.5 Summary of observed effects

Across the surveyed urban green spaces, small mammal responses were shaped more consistently by habitat structure and disturbance than by patch area alone. Woody linear density was positively associated with community responses and with CPUE for *A. sylvaticus* and *M. glareolus*, emphasising the value of hedgerows, scrub margins and treelines within urban green space patches. In contrast, path density produced the most consistent negative response, reducing community-level CPUE and the CPUE of three of the four focal species.

Responses to grassland cover, watercourse density and management intensity were more taxon-specific. Management intensity reduced community-level CPUE and was also associated with responses in *A. sylvaticus* and *S. araneus*, while grassland-related effects differed among the focal species. The collective results show that urban green spaces were not ecologically equivalent: their value depended on interactions among habitat composition, linear vegetation, disturbance, and management. These patch-scale findings provide the basis for examining whether surrounding landscape structure further influenced small mammal communities in Chapter 4.

3.4 Discussion

The research conducted for this chapter was designed to provide evidence of how small mammal communities respond to varying levels of grassland management intensity through mowing, and their associations with within-patch habitat structure and designation within urban green spaces. Chapter 3 has supported H1 from the overall hypotheses developed in Chapter 1.11.4, that 'Patch-scale habitat structure, habitat composition and management intensity influence small mammal richness, Catch Per Unit Effort (CPUE) and community diversity within urban green spaces. Richness and CPUE are expected to be higher where dense ground-level vegetation and woody linear structure provide refuge, and lower where management and path density increase disturbance and exposure.' It has also provided evidence for H2 in the overall hypotheses, where it was predicted that 'Focal species differ in their responses to patch and landscape scale habitat features according to ecological specialisation, specifically their dependence on grassland, woody cover, and connected habitat. Community-level responses are therefore expected to conceal contrasting species-specific associations.'

Although the species-specific analyses focused on four focal taxa, these represented 91.89% of all captures rather than the complete assemblage recorded during the study. *M. musculus* accounted for a further 7.32% of captures but was excluded from the focal models to retain the study focus on species associated with more natural urban habitats (Berry, 1981; Phifer-Rixey & Nachman, 2015; Sked et al., 2022), while *N. fodiens* and *A. flavicollis* were recorded too infrequently for meaningful statistical analysis. Their occurrence, however, indicates that the surveyed green spaces supported a broader assemblage than the species-specific models represented, and

the ecological responses discussed below therefore pertain specifically to the four focal species for which sufficient data were available.

The results for the focal species broadly supported these predictions but also showed that patch-scale effects varied across community measures and species. Woody linear density was the only predictor positively associated with species richness, total CPUE and Simpson's reciprocal diversity, while tree cover had no significant effect on any of these community measures. Grassland cover and watercourse density increased total CPUE, while path density, management intensity and patch area reduced total CPUE (Figures 3.2 - 3.4).

3.4.1 Habitat structure shaped small mammal communities within urban green spaces

Of the patch-scale variables examined in this study, woody linear density provided the clearest evidence that the structure and arrangement of vegetation within urban green spaces were more consistently associated with small mammal communities than the amount of any single broad habitat type or the management of grassland. Woody linear density was the only predictor positively associated with all three community measures, increasing species richness ($\beta = 0.308$, $p = 0.032$), CPUE ($\beta = 0.372$, $p = 0.002$) and Simpson's reciprocal diversity ($\beta = 0.388$, $p = 0.021$). This pattern was not repeated by grassland cover, which increased CPUE but not richness or Simpson's reciprocal diversity, by watercourse density, which was also associated only with CPUE at the community level, or by tree cover, which had no significant effect on any of the three measures. Management intensity reduced CPUE but did not alter richness or Simpson's reciprocal diversity. The importance of woody linear density, therefore, lies in the consistency of the relationship across different properties of the community.

Green spaces containing more hedgerows, treelines and scrub margins supported higher capture rates, more species and a more even distribution of captures among those species, whereas broad measures of grassland and tree cover did not produce the same community-wide response.

This result is consistent with the habitat heterogeneity argument developed in Chapter 1, in which the ecological value of habitat was considered to depend on the resources and structural conditions available to the organism rather than on the mapped habitat class alone. For a small ground-dwelling mammal, a hedgerow occupies little area when viewed at the scale of an entire green space, but at the scale experienced by a small mammal it can contain dense basal vegetation, litter, woody cover, feeding opportunities and sheltered edges within the same narrow feature (Hinsley & Bellamy, 2000; Tattersall et al., 2002; Atkins et al., 2018). This is especially relevant in urban green spaces because extensive grassland or tree cover does not necessarily provide equal conditions at ground level. A large area of tree cover can consist of mature trees over short amenity grass, while a much smaller scrub margin or hedge may provide continuous cover from the ground upwards. Likewise, a grass-dominated green space may contain a large area of apparently suitable habitat yet provide relatively few protected routes between feeding and refuge areas if the grassland is divided by paths or surrounded by exposed ground. The results therefore support the ecological expectation that habitat amount and habitat structure are not interchangeable, and that the value of urban vegetation to small mammals depends partly on how that vegetation is arranged within the patch (Tews et al., 2004; Garden et al., 2007; Betts et al., 2014).

This is not an unusual result in small mammal ecology and closely follows some of the earliest work on small mammals in British cities. Dickman and Doncaster (1987) examined the same four focal species used here and found that their densities in Oxford were more strongly related to vegetation characteristics than to measures of urban disturbance. *A. sylvaticus* was widespread across the city but reached its highest densities in scrub, *M. glareolus* reached its highest abundance in woodland, while *M. agrestis* and *S. araneus* were most abundant in long grass and scrub and were largely absent from short grass and intensively cultivated habitats. More importantly, vegetation density at ground-level heights was a stronger predictor of small mammal density than broader measures of habitat modification, and Dickman and Doncaster (1987) argued that small mammals responded to urbanisation largely through the changes it produced in patch vegetation. This is closely matched by the results of this study, where broad tree cover was uninformative, but a measure describing dense linear vegetation at a scale directly relevant to small mammals was associated with all three community responses.

The agreement is stronger still when the local evidence from Stoke-on-Trent and Newcastle-under-Lyme is considered. Atkins et al. (2018) recorded small mammal activity in 63% of urban hedges surveyed locally, compared with only 10% of linear boundaries where hedgerows were absent, as outlined in Chapter 1. The Chapter 3 results extend this beyond the use of individual hedge features. Increasing woody linear density was associated not only with a higher total capture rate but also with greater richness and Simpson's reciprocal diversity across entire green space patches. This matters because the community response cannot be explained only by greater activity of the dominant *A. sylvaticus*: *M. glareolus* also responded positively to woody

linear density, while richness and Simpson's reciprocal diversity increased across the assemblage. British farmland studies provide additional support for the value of hedgerows and field boundaries, although the effects are not universal across species (Tattersall et al., 2002; Gelling et al., 2007b). Tattersall et al. (2002), for example, found field boundaries to be the most species-rich habitat surveyed and recorded greater *M. glareolus* abundance in linear hedgerows than in non-linear woodland, but found no effect of linearity on *A. sylvaticus*, *M. agrestis* or *S. araneus*. The broader community response found here, therefore, does not indicate a simple preference for linear habitat; rather, it suggests that the structure supplied by hedgerows, treelines and scrub margins was valuable within these urban patches.

The difference between woody linear density and proportional tree cover is also important within this study, as the proportion of tree cover had little effect on either of the community metrics or species recorded. Woody linear density, however, increased *M. glareolus* CPUE ($\beta = 1.736$, $p = 0.006$), while proportional tree cover showed no significant relationship. This is consistent with the established association of *M. glareolus* with woodland, scrub and dense understorey vegetation rather than canopy cover alone (Dickman & Doncaster, 1987; Buesching et al., 2008; Bush et al., 2012; Fuentes-Montemayor et al., 2020). Buesching et al. (2008), for example, found that both wood mice and bank voles altered their use of woodland according to understorey conditions, while Bush et al. (2012) linked recovery of wood-mouse and bank-vole communities to the recovery of vegetation density and complexity following reduced browsing pressure. The absence of a tree-cover response here should therefore not be interpreted as evidence that woody habitat was unimportant. However, it could suggest that the form in which woody vegetation occurred was more closely

associated with small mammal responses than its total area. Canopy cover alone cannot differentiate between woodland with dense ground vegetation and isolated mature trees standing above closely managed grass, while hedgerows and scrub margins inherently describe vegetation occurring much closer to the ground.

The negative effect of path density reflects the opposing side of the same structural pattern, as exposed or paved surfaces can interrupt continuous ground-level habitat, in contrast to woody linear features that provide cover and can maintain connectivity (Tattersall et al., 2002; Gelling et al., 2007b; Ascensão et al., 2016; Galantinho et al., 2020). Total CPUE declined as path density increased, and the same negative relationship was observed in three of the four focal species: *A. sylvaticus, M. agrestis,* and *S. araneus*. The result does not demonstrate that paths directly excluded species, but it does show that patches containing more internal paths supported lower small mammal capture rates after the other patch variables were accounted for. This is coherent with the effects of breaking up continuous ground-level vegetation into areas separated by exposed, compacted or repeatedly disturbed surfaces.

Chapter 1 identified vegetation loss, trampling, and repeated breaks in cover as processes that can increase exposure within otherwise suitable habitat (Tikka et al., 2001; Van Helden et al., 2020; Hu & Lima, 2024). Direct evidence from *A. sylvaticus* elsewhere supports this mechanism: Galantinho et al. (2020) found that a paved road reduced fine-scale connectivity for wood mice, while vegetated verges promoted movement and vegetation cutting reduced that benefit. Urban footpaths are clearly not equal to roads, and movement was not measured here, but the comparison shows that

relatively narrow changes between vegetated and exposed surfaces can affect how small mammals use space, on a finer scale within green space patches.

The contrast between woody linear density and path density is therefore ecologically coherent, as one describes an increase in continuous, structurally complex vegetation, while the other reflects an increase in internal patch structure likely to interrupt or disturb ground-level cover. This perception is especially relevant to the post-industrial urban landscape described in Chapter 2, where green spaces occur among roads, built land, canals, railway corridors and regenerating vegetation (Threlfall et al., 2017; Staffordshire Wildlife Trust, 2024; Stoke-on-Trent City Council, 2025). Within urban areas, linear vegetation can represent far more ecological value than the small proportion of patch area it occupies. Studies of Birmingham, UK, have likewise emphasised the potential importance of urban greenways and chains of habitat running through heavily modified landscapes, although Angold et al. (2006) also stressed that their value depends on the species involved and whether they function as habitat, movement routes, or both.

3.4.2 Grassland cover and mowing intensity produced contrasting responses

Grassland cover and management intensity produced a different pattern from woody linear density because their effects were concentrated on capture rate rather than extending across all three community measures. Total CPUE increased with grassland cover but declined with increasing management intensity, while neither predictor significantly altered species richness or Simpson's reciprocal diversity. The focal species also separated these responses further: grassland cover increased CPUE in *A. sylvaticus* and *M. agrestis*, management intensity reduced CPUE in *A. sylvaticus* and

S. araneus, and *M. glareolus* showed a significant interaction between grassland cover and management intensity. The ecological value of grassland, therefore, depended on more than its proportional area within a patch, while the effect of mowing was not shared uniformly by the assemblage.

The response of *M. agrestis* provides the clearest example as its CPUE increased with both grassland cover and watercourse density, but declined with path density and patch size. The positive watercourse coefficient ($\beta = 0.638$, $p = 0.002$) was slightly greater than that for grassland cover ($\beta = 0.606$, $p < 0.001$), despite the established association of field voles with dense herbaceous grassland (Tattersall et al., 2000). This does not imply that water by itself was more important than grassland, as watercourse density was measured by the extent of the linear feature relative to the vegetation growing alongside it, and urban watercourses frequently retain rough grass, tall herbs, and other vegetation along their banks (McIntyre, 2010; Davies et al., 2011; Staffordshire Wildlife Trust, 2024). Such margins could therefore combine the dense ground cover associated with *Microtus* with continuity through otherwise managed parts of the green space. Dickman and Doncaster (1987) also found field voles concentrated in long grass and scrub, while British set-aside studies have linked *M. agrestis* abundance to grass and litter within the sward (Tattersall et al., 2000).

The urban comparison provided by Larson and Sander (2024) is especially relevant because high small mammal diversity in some of their human-modified sites occurred in mown parks with streams and drainage channels bordered by tall vegetation. Their results suggest that managed grassland does not necessarily have uniformly low value where dense herbaceous structure is allowed to grow more naturally alongside

streams or other edges. This provides useful context for this study, in which grassland management reduced total CPUE, but watercourse density remained positively associated with community CPUE and, more significantly, with the UK's most common small mammal species, *M. agrestis*. The value of a managed green space, therefore, appears to depend partly on which structural features remain within it, rather than mowing alone, such as patterns which are also displayed by woody linear features in the preceding section of this discussion.

This also prevents the management result from being reduced to a simple difference between mown and unmown grassland. Increasing management intensity clearly reduced total CPUE and the CPUE of *A. sylvaticus* and *S. araneus*, but *M. agrestis* showed no significant direct response to management intensity despite responding strongly to grassland and watercourse density. Tattersall et al. (2000) likewise found field vole abundance to be related to grass and litter cover, while recommending periodic mowing as compatible with suitable set-aside management. This suggests that the conditions produced by management can be more important than the mere occurrence of cutting. For *S. araneus*, evidence from Danish meadows provides a closer parallel: Schmidt et al. (2009) found that high grazing intensity significantly reduced *S. araneus* abundance, whereas low-intensity grazing did not differ from ungrazed areas.

M. glareolus produced a different response because neither grassland cover nor management intensity alone had a significant effect on its CPUE, while the interaction between the two was significant. This indicates that the relationship between grassland cover and bank vole capture rate may have changed with management

intensity, rather than either variable having a consistent independent effect. This is important when considering the strong positive response of *M. glareolus* to woody linear density discussed in the previous section. *M. gloareolus* individuals are commonly associated with woodland, scrub and dense ground vegetation, and their use of more open habitat is likely to depend on the cover available within and around it (Dickman & Doncaster, 1987; Buesching et al., 2008; Fuentes-Montemayor et al., 2020). Grassland area alone may therefore provide a poor indication of its value to this species where mowing alters vegetation structure and where access to woody refuge varies between patches. The interaction also differs from that of *M. agrestis*, for which grassland cover itself had a strong positive effect, whereas management intensity did not. These contrasting responses support the expectation that the effect of grassland management depends on the ecological requirements of the species using it rather than producing a uniform response across the assemblage. Frequent mowing may also limit the development of the dense woody and ground-level structure associated with *M. glareolus*, reducing the value of grassland-dominated patches for this species.

The stronger response of CPUE than richness or Simpson's reciprocal diversity also suggests that grassland cover and mowing intensity affected the level of small mammal use within patches more clearly than the composition of the assemblage itself. Increased grassland cover was associated with more captures, while increasing mowing intensity was associated with fewer, but neither relationship was accompanied by a significant change in the number of species recorded or their relative Simpson's reciprocal diversity. Therefore, greater capture activity within a habitat does not necessarily indicate a more diverse community, especially when

changes are driven by common species that respond strongly to local habitat conditions.

3.4.3 Patch area was associated with lower capture rates but not greater community richness

Patch area produced a different result from that expected from simple species–area relationships because increasing patch area was negatively associated with total CPUE ($\beta = -0.235$, $p = 0.008$), with the same negative relationship also recorded for *A. sylvaticus* and *M. agrestis*. However, patch area had no significant relationship with species richness or Simpson's reciprocal diversity. The results therefore provide no evidence that the larger urban green spaces surveyed supported richer or more even small mammal communities, while capture rates were actually lower as mapped patch area increased. This contrasts with the expectation arising from island biogeography that larger habitat patches should generally support more species through greater habitat availability, larger populations and reduced extinction risk (MacArthur & Wilson, 1967), an expectation outlined in Chapter 1 for urban green spaces considered as habitat islands.

Applying species–area expectations to urban green spaces depends partly on how closely mapped patch extent corresponds to the habitat available to the organisms being studied. Urban green spaces can exhibit substantial internal variation in vegetation structure and management, so an increase in total area does not necessarily increase suitable ground-level habitat (Olff & Ritchie, 2002; Gomes et al., 2011; Larson & Sander, 2024). A large park may contain extensive short-mown grass, paths and exposed ground, while a smaller patch may contain a greater concentration of rough grassland, scrub margins and dense vegetation (Angold et al., 2006; Threlfall

et al., 2017; Dondina et al., 2025). Fragmentation can complicate this further because the proportion and character of edge habitat also changes with patch configuration, exposing small mammals to different combinations of cover, disturbance and surrounding land use rather than simply more or less habitat (Fischer & Lindenmayer, 2007). In this context, mapped area may describe the physical extent of the green space reasonably well while saying much less about the amount and arrangement of habitat usable by small mammals within it.

The sampling design may also be relevant to the relationships observed with patch area. Traps were positioned within suitable vegetated habitats rather than distributed across all land cover types in proportion to their occurrence within each mapped patch. CPUE, therefore, reflects capture rates within the habitats incorporated into the trapping design rather than population size across the whole green space. Live-trapping indices cannot be translated directly into population density without accounting for the effective area sampled and animal movements (Efford, 2004), so the lower CPUE recorded in larger patches does not imply that they contained fewer small mammals overall. However, CPUE does serve as an index of relative capture activity within the sampled vegetated habitat. Bowers and Matter (1997) found decreasing density with increasing patch area in seven of 32 mammal species and showed that density-area relationships varied with spatial scale, movement and habitat selection. A similar process may have contributed to the pattern recorded here if suitable vegetated habitat and small mammal activity were more concentrated within the sampled areas in smaller patches, while larger patches provided a greater area over which animals and resources could be distributed. The observed area effect could be linked to lower capture rates within sampled habitat as the mapped patch

area increased, without implying that smaller green spaces supported larger populations or were inherently higher-quality habitat, a limitation acknowledged by the trapping design and available traps.

The species responses strengthen this explanation because the negative relationship was shared by *A. sylvaticus* and *M. agrestis* despite their different habitat associations (Dickman & Doncaster, 1987; Fuentes-Montemayor et al., 2020). *A. sylvaticus* responded positively to woody linear density and grassland cover, while *M. agrestis* responded positively to grassland cover and watercourse density as outlined in previous sections. Their shared negative response to the patch area, therefore, occurred alongside quite different positive associations with habitat structure. This makes it difficult to explain the area effect using a single preferred habitat type and could suggest that patch extent alone contains relatively little information about the distribution of the specific conditions used by either species where urban fragmentation is concerned (Angold et al., 2006; Threlfall et al., 2017; Dondina et al., 2025). This is consistent with the wider pattern emerging from Chapter 3, where variables describing vegetation structure, management, and internal disturbance yielded clearer ecological relationships than patch area alone.

Viewed within the broader framework developed in Chapter 1, the patch-area result fits an urban system in which habitat quality and matrix permeability vary independently of mapped green space extent. Urban green spaces are internally heterogeneous, while the surrounding matrix contains both resistant and permeable elements, meaning that patches of similar size can contain very different amounts and arrangements of usable habitat (Angold et al., 2006; Gomes et al., 2011; Lepczyk et al.,

2017). Mapped patch area therefore provided a comparatively coarse measure of habitat value for the small mammal communities surveyed.

3.4.4 Species-specific responses explain differences among community measures

The species-specific models provide stronger support for H2 than would be apparent from the community models alone, because the four focal species responded to different combinations of patch characteristics rather than showing a uniform response to habitat composition, structure and management. Path density produced the most widely shared negative relationship, reducing CPUE in *A. sylvaticus, M. agrestis and S. araneus*, but the positive habitat associations were less consistent across species. *M. agrestis* responded to grassland cover and watercourse density, *M. glareolus* to woody linear density and the interaction between grassland cover and management intensity, and *S. araneus* responded negatively to management intensity in addition to path density. These responses have been considered individually in the preceding sections, but together they show that no single patch characteristic accounted for the distribution of capture rates across all four species. Instead, the community-level patterns emerged from species responding to different elements of the same urban green spaces, consistent with the species-specific ecological associations predicted under H2.

A. sylvaticus is especially informative in this respect because it accounted for 62.22% of all captures yet also showed the broadest response to the measured patch variables, indicating significant relationships with more predictors than any other focal species. *A. sylvaticus* is widely regarded as an ecological generalist and occurs across a broad range of rural and urban habitats (Dickman & Doncaster, 1987; Baker et al.,

2003), but these results show that widespread occurrence did not equate to weak environmental responsiveness. The dominance of *A. sylvaticus* may partly reflect its ability to use several habitat types within urban landscapes, while its variation in CPUE shows that the quality and arrangement of those habitats still influenced its occurrence within individual patches. This is important for interpreting generalism in urban systems: tolerance of a broad range of environments does not mean that local management, vegetation structure and disturbance cease to affect the species using them.

The contrasting responses of the remaining focal species also help explain the differences between the community-level and species-level models. Similar variation among species has been reported elsewhere. Fuentes-Montemayor et al. (2020) found that *M. agrestis*, *A. sylvaticus* and *M. glareolus* responded to different local and landscape attributes within the same study system. Sundell et al. (2012) also showed that habitat use by *M. agrestis*, *M. glareolus* and *S. araneus* changed with population density and the presence of other small mammals. Dondina et al. (2025) found comparable species-level variation across urban green spaces, where habitat composition influenced species differently within the same network. These results show that the community patterns were built from several species-specific responses operating together. Interspecific interactions may have contributed additional variation. Bujnoch et al. (2024) and Probst & Probst (2023) both found evidence that *M. glareolus* population dynamics or activity patterns were associated with other small mammals, including *Apodemus*. Competition and behavioural displacement were not measured here, so their influence cannot be separated from the habitat and management relationships identified by the models.

The three metrics describe different properties of the assemblage. Species richness records the number of species present, CPUE measures capture rate, and Simpson's reciprocal diversity incorporates both richness and the distribution of captures among species (Hillebrand et al., 2008; Morris et al., 2014). Similar differences among community measures have been recorded in small mammal succession studies, where abundance and diversity changed differently as assemblage structure changed through time (Jasiulionis et al., 2011; Balčiauskas et al., 2017). In this study, *A. sylvaticus* was recorded to have dominated captures, but the analyses did not test how much each focal species contributed to the significant total-CPUE relationships. The defensible conclusion is therefore that patch-scale conditions were associated with capture rate across a wider range of predictors than with richness or Simpson's reciprocal diversity.

3.4.5 Management implications and limitations

The interpretation of the Chapter 3 findings is bounded by several features of the survey design. Five repeated surveys across 21 green space patches produced 6,966 adjusted trap nights, providing substantial repeated sampling, but the independent spatial sample remained 21 patches. CPUE standardised capture rates to account for differences in trapping effort, but it did not separately account for species-specific detection probability. Capture probability in small mammals can vary with ranging behaviour, weather, resource distribution, neophobia and species-specific responses to traps (Flowerdew et al., 2004; Otto & Roloff, 2011; Harkins et al., 2019; Stryjek et al., 2019). The repeated survey design and adjustment of trap nights reduced several sources of variation, but lower CPUE cannot be assigned uniquely to lower local numbers or altered activity. The habitat variables also differed in the detail they

represented. Woody linear density measured the extent of hedgerows, treelines and scrub margins but did not describe their width, gaps, floristic composition, basal vegetation or management history. Management intensity was based on grouped mowing frequency and did not measure cutting height, biomass removal, litter depth or the proportion of each patch left uncut. These limitations restrict interpretation to the habitat and management characteristics measured and prevent the individual mechanisms underlying the observed relationships from being separated. Future work could address these constraints through greater spatial replication, finer field measurements of vegetation structure and grassland management, and longer-term individual marking and capture–recapture approaches capable of explicitly estimating detection probability.

The Chapter 3 results do, however, support several practical management measures for within green space patches. Woody linear density was the only predictor positively associated with species richness, CPUE and Simpson's reciprocal diversity, and it was also positively associated with CPUE in *A. sylvaticus* and *M. glareolus*. Retaining hedgerows, treelines and scrub margins should therefore be a priority within urban green spaces, with restoration or new planting used where these features are sparse (Tattersall et al., 2002; Gelling et al., 2007b; Atkins et al., 2018). Grassland management also requires attention in certain aspects of this study. As total CPUE declined when management intensity increased, and the same negative relationship was observed for *A. sylvaticus* and *S. araneus*, the latter of which was recorded least frequently. Reduced mowing in selected areas would be consistent with these results and with evidence that lower mowing frequencies can increase vegetation and invertebrate diversity in urban grasslands (Garbuzov et al., 2015; Chollet et al., 2018;

Sehrt et al., 2020; Wintergerst et al., 2021), with rotational mowing often touted as a practical solution for increasing diversity in modified landscapes. Watercourse density was positively associated with total CPUE and with *M. agrestis*, supporting the retention of vegetated watercourse margins where these occur. Path density was negatively associated with total CPUE and with three of the four focal species. Avoiding unnecessary increases in path density through suitable habitat would therefore reduce another feature associated with lower small mammal capture rates.

3.4.6 Conclusions

Chapter 3 answers the patch-scale research question by showing that variation in small mammal communities within urban green spaces was associated more consistently with vegetation structure, management and internal disturbance than with patch area alone. H1 was therefore supported in predicting that patch-scale habitat structure, composition and management would be associated with small mammal community responses, although the individual relationships were not shared across all three community measures.

The species-specific analyses also supported the broader prediction under H2 that community-level responses conceal contrasting associations among focal taxa. These differences support H2 at the level of species-specific ecological responses, although the more specific expectation that *M. glareolus* and *M. agrestis* would show stronger positive responses to woody vegetation and lower mowing than the generalist *A. sylvaticus* was only partly realised. The focal species responded differently, but not according to a single specialist-to-generalist gradient.

Chapter 4 The Influence of Landscape Matrix Structure on Small Mammal Composition Within Habitat Patches

4.1 Introduction

Urban green spaces do not function as isolated ecological units; they are embedded within surrounding matrices of sealed surfaces, roads, gardens, amenity grassland, watercourses, hedgerows, remnant habitats and other semi-natural features (Angold et al., 2006; Beninde et al., 2015; Lepczyk et al., 2017). As a result, the ecological value of an individual patch may depend not only on its internal habitat quality, but also on the composition, configuration and permeability of the wider landscape. This is central to modern landscape ecology theory, where species responses are shaped by habitat amount, fragmentation, isolation and connectivity operating across multiple spatial scales (Wiens, 1989; Fahrig, 2003; Fischer and Lindenmayer, 2007; Jackson and Fahrig, 2012; Miguet et al., 2016).

Chapter 2 established Stoke-on-Trent and Newcastle-under-Lyme as a fragmented post-industrial urban landscape in which green and blue infrastructure occurs as a heterogeneous network of parks, amenity grasslands, remnant habitats, railway verges, canal corridors, riparian features and regenerating brownfield land (Tandy, 1975; Staffordshire Wildlife Trust, 2024). The study area contains a high proportion of mapped green space, but its distribution, habitat quality, management intensity and structural connectivity vary considerably across the urban landscape (Chapter 2.2, 2.4). Chapter 3 examined the internal characteristics of surveyed green space patches and found that small mammal community responses were associated more consistently with habitat structure, path density and mowing frequency than with

patch area. Patch area showed no significant relationship with species richness or Simpson's reciprocal diversity, providing no support for the positive species–area relationship predicted by Island Biogeography Theory (MacArthur & Wilson, 1967). Similar importance of vegetation structure, management and linear habitat features has been reported for small mammals across urban and agricultural landscapes (Tattersall et al., 2002; Gelling et al., 2007b; Fischer et al., 2011; Atkins et al., 2018; Dondina et al., 2025). Woody linear density was the only predictor positively associated with richness, CPUE and Simpson's reciprocal diversity. Path density and management intensity were negatively associated with CPUE (Chapter 3, Figure 3.2).

However, patch-scale habitat quality represents only one component of small mammal persistence in urban landscapes. Small mammals moving beyond a green space boundary encounter roads, sealed surfaces, intensively managed vegetation and breaks in ground-level cover that can restrict movement between otherwise suitable habitats (McDonald & St. Clair, 2004; Ascensão et al., 2016; Grilo et al., 2018; Galantinho et al., 2020). This is important for relatively low-mobility ground-dwelling taxa because habitat occurring nearby may still be difficult to reach if the intervening matrix contains barriers or exposed land. A short geographical separation between two green spaces can therefore represent a substantial ecological barrier, while continuous vegetation can maintain permeability across a greater distance (LaPoint et al., 2015; Galantinho et al., 2020). Restricted movement can also limit immigration into otherwise suitable patches and reduce opportunities for recolonisation following local population loss (Hanski, 1998; Baguette et al., 2013). Hedgerows, watercourses and vegetated boundaries may reduce some of these

constraints by maintaining cover between separated habitat patches (Tattersall et al., 2002; Atkins et al., 2018; Galantinho et al., 2020).

Several urban small mammal studies have shown that community responses are influenced by landscape context as well as local habitat conditions. For example, Klimant et al. (2017) used a multiscale approach in Nitra, Slovakia, and found that small mammal diversity, richness and relative abundance declined with increasing built-up density along an urban-rural gradient. Other recent work has also shown that mammal assemblages in urban environments are filtered by habitat availability, disturbance, connectivity and species-specific tolerance of urban land uses (Fung et al., 2024; Larson and Sander, 2024; Dondina et al., 2025). These findings reinforce the need to assess urban green spaces as parts of wider habitat networks rather than as isolated patches.

The surrounding landscape matrix may influence small mammal communities through several related mechanisms. Fragmentation and isolation can reduce effective habitat availability by subdividing green space into smaller, more disconnected patches. A high number of separate green space polygons within a buffer may therefore indicate a fragmented network rather than a coherent area of usable habitat (Fahrig, 2003; Betts et al., 2014). Additionally, linear features such as hedgerows and watercourses may increase matrix permeability by providing sheltered movement routes, refuge habitat and foraging edges across otherwise exposed or modified land (Atkins et al., 2018; Gurnell et al., 2019; Galantinho et al., 2020). Roads may also reduce connectivity by creating barrier effects, increasing disturbance and interrupting vegetated routes between patches (de Redon et al.,

2015; Galantinho et al., 2020; Ouédraogo et al., 2020). Collectively, these processes mean that surrounding green cover alone may be insufficient to explain community responses; the arrangement and structural quality of that cover may be more important.

This chapter extends the patch-scale analysis in Chapter 3 by examining whether the external landscape matrix surrounding surveyed green space patches influences the small mammal communities recorded within those patches. Matrix predictors were calculated at two spatial scales: 250 m, representing the immediate matrix surrounding each surveyed patch, and 500 m, representing the broader local landscape composition (Bowman et al., 2002; Fischer et al., 2011; Dorigo et al., 2021), to test whether community responses differed between the immediate surrounding matrix and the broader local landscape. These distances were selected as predefined scales, akin to those in the broader literature, rather than as direct estimates of species-specific home-range or dispersal distances.

Comparable extents have been used in previous small mammal landscape studies; Fischer et al. (2011) assessed small mammal abundance, richness, diversity and community composition using landscape variables measured within 100 m, 250 m and 500 m radii around focal fields, while Dorigo et al. (2021) analysed small mammal responses to landscape composition and configuration within both 250 m and 500 m buffers around habitat patches. Bowman et al. (2002) also evaluated small mammal abundance in relation to landscape context at several local extents, including 250 m and 500 m. The two-scale design used here therefore provided an ecologically relevant comparison between immediate matrix conditions and broader

landscape-scale structure while avoiding unsupported selection of a single presumed scale of effect (Bowman et al., 2002; Fischer et al., 2011; Dorigo et al., 2021). The chapter focuses on three community-level responses: species richness, total CPUE and Simpson's reciprocal diversity. These represent complementary aspects of the assemblage: observed richness, standardised capture activity and community diversity.

In response to the landscape-scale questions outlined in Chapter 1, this chapter uses GIS-derived matrix variables to test whether small mammal community responses within surveyed patches were associated with the composition, configuration and potential permeability of the surrounding urban matrix. Matrix habitat amount was represented by the proportion of non-sealed green matrix. Green patch count was used to describe the number and subdivision of neighbouring mapped green space units, and nearest green patch distance measured structural isolation (Fahrig, 2003; Fischer & Lindenmayer, 2007; Pardini et al., 2010; Püttker et al., 2020). Hedgerow and watercourse density represented potential linear structural connectivity (Maisonneuve & Rioux, 2001; Michel et al., 2006; Sullivan et al., 2012; Galantinho et al., 2020), while road density represented barrier and disturbance pressure (de Redon et al., 2015; Ascensão et al., 2016; Galantinho et al., 2020). These variables therefore describe separate components of the surrounding matrix without treating any single metric as a complete measure of landscape fragmentation or functional connectivity.

This chapter directly tests H3 of the thesis, which predicts that wider landscape structure influences small mammal communities in addition to the patch-scale

habitat and management relationships examined in Chapter 3. H3 predicts lower community responses with increasing isolation, green space subdivision and road density, with higher responses associated with greater surrounding habitat amount and structurally continuous green and blue linear features. Chapter 4 evaluates these landscape relationships independently at 250 m and 500 m. Their combined influence with the patch-scale predictors is tested subsequently in Chapter 5.

Research question: How does the composition, configuration and potential permeability of the surrounding urban matrix influence small mammal richness, CPUE and Simpson's reciprocal diversity within surveyed green space patches?

It was hypothesised that surrounding matrix composition, configuration, isolation and linear structure would influence small mammal community responses. Specifically, it was predicted that:

1. Surrounding green space configuration and isolation would influence community responses, with greater nearest patch distance expected to reduce richness, CPUE and Simpson's reciprocal diversity. Green patch count was included as a measure of surrounding green space subdivision and configuration.

2. Greater matrix green proportion would be associated with higher community responses, testing whether the amount of surrounding non-sealed habitat influenced the assemblage.

3. Greater hedgerow and watercourse density would be associated with higher community responses as measures of potential linear structural connectivity.

4. Greater road density would be associated with lower community responses, reflecting increased barrier and disturbance pressure within the surrounding matrix.

These relationships were tested at 250 m and 500 m to examine whether matrix effects differed between the immediate surrounding landscape and the broader local matrix.

4.2 Methods

4.2.1 Analytical approach and spatial framework

This chapter extends the patch-scale analyses presented in Chapter 3 by testing whether the external landscape matrix surrounding surveyed urban green space patches influenced small mammal community responses. To isolate landscape-scale effects, no within-patch habitat, structural or management predictors were included in the Chapter 4 models.

Matrix predictors were calculated around the same 21 surveyed green space patches used in Chapter 3. The final surveyed site boundary layer generated during the Chapter 3 GIS workflow was used as the focal patch layer. Two landscape distances were used: 250 m, representing the immediate surrounding matrix, and 500 m, representing the broader local landscape (Bowman et al., 2002; Fischer et al., 2011; Dorigo et al., 2021). These distances were selected as predefined local landscape extents to test whether matrix effects differed between immediate and broader contexts, consistent with the landscape ecology principle that species-environment relationships are scale dependent (Wiens, 1989; Fahrig, 2003; Jackson and Fahrig, 2012; Miguet et al., 2016).

All spatial analyses were undertaken in QGIS v.3.28 using the British National Grid coordinate system (EPSG:27700). Concentric 250 m and 500 m buffers were generated around each surveyed site boundary using the Buffer tool. The focal surveyed patch polygon was removed from its own buffer using the Difference tool, producing annular matrix-ring polygons. This restricted the Chapter 4 GIS extraction to the landscape outside each surveyed patch. Each matrix ring retained the focal Site ID, allowing the resulting GIS predictors to be joined to the Chapter 3 site-repeat trapping dataset.

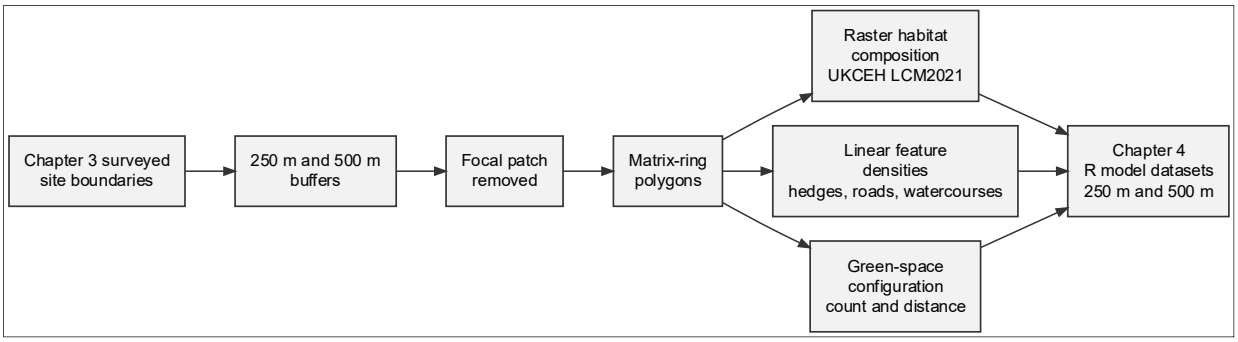


Figure 4.1 GIS workflow for Chapter 4 matrix predictor extraction. Final surveyed site boundaries from Chapter 3 were buffered at 250 m and 500 m in QGIS. The focal surveyed patch was removed from each buffer to create annular matrix-ring polygons representing only the surrounding landscape. Raster-derived habitat composition, linear feature densities and green space configuration variables were extracted from these matrix rings and joined by Site ID to the Chapter 3 site-repeat small mammal dataset for statistical modelling in R (v.4.6.0).

4.2.2 Small mammal response variables

The small mammal response variables used in Chapter 4 were the same response variables constructed for the Chapter 3 patch-scale analyses: focal species richness, total CPUE and Simpson's reciprocal diversity. These were derived from the Longworth trapping dataset described in Chapter 3 and were not recalculated using a different method for the landscape-scale analysis.

As in Chapter 3, response variables were calculated at the Site ID × Repeat level. Species richness represented the number of focal small mammal species recorded

within each site-repeat survey. Total relative community capture was expressed as total Catch Per Unit Effort (CPUEtotal), calculated by summing species-level CPUE values across the focal taxa within each site-repeat survey. Simpson's reciprocal diversity was calculated from the proportional CPUE contribution of each focal species within the same site-repeat survey.

These biological response variables were then joined to the GIS-derived matrix predictors by Site ID in R (v.4.6.0). Matrix predictors were calculated once for each surveyed patch at each buffer scale, while the biological response variables retained the repeated survey structure described in Chapter 3. Chapter 3 examined predictors measured within each surveyed patch. Chapter 4 examined predictors derived from the surrounding 250 m and 500 m annular matrix rings.

4.2.3 GIS-derived matrix predictors

Six matrix predictors were retained to represent complementary components of landscape composition, configuration and permeability. Matrix green proportion measured the amount of broadly non-sealed habitat surrounding each surveyed patch, reflecting evidence that small mammal communities respond to surrounding land cover composition and habitat availability as well as local patch conditions (Bowman et al., 2002; Fischer et al., 2011; Dorigo et al., 2021). Green patch count represented the subdivision of surrounding green space, while nearest green patch distance provided an index of structural isolation; both patch configuration and inter-patch separation can affect habitat accessibility, colonisation and small mammal community composition within fragmented landscapes (Fahrig, 2003; Fischer and Lindenmayer, 2007; Pardini et al., 2010; Püttker et al., 2020).

Matrix hedge density and watercourse density were included as measures of potential linear structural connectivity because hedgerows, vegetated boundaries and riparian habitats can provide refuge, foraging habitat and covered movement routes for small mammals through modified landscapes, building on within patch structural refuge identified in Chapter 3 (Maisonneuve and Rioux, 2001; Michel et al., 2006; Sullivan et al., 2012; Galantinho et al., 2020). Matrix road density represented barrier and disturbance pressure, as roads can restrict small mammal movement and reduce functional connectivity, although vegetated road verges may partly moderate these effects (de Redon et al., 2015; Ascensão et al., 2016; Galantinho et al., 2020). These represented matrix composition, green space configuration, isolation, structural connectivity and barrier density.

The matrix predictors (Table 4.1) were designed to describe broad properties of the external landscape and did not reproduce the detailed habitat-specific measurements used within surveyed patches in Chapter 3. Species-specific use of individual habitats within the surrounding matrix was therefore not tested directly.

Table 4.1 GIS-derived landscape matrix predictors used in the Chapter 4 models. Predictors were calculated for 250 m and 500 m matrix rings surrounding each surveyed green space patch.

Predictor	Source layer	QGIS processing	Unit	Ecological role
Matrix green proportion	UKCEH Land Cover Map 2021, 10 m Classified Pixels	Zonal Histogram within 250 m and 500 m matrix rings; valid pixels retained; urban and suburban pixels used to calculate sealed cover	Proportion	Surrounding non-sealed habitat amount
Green patch count	Unified green space polygon layer established in Chapter 3	Count of neighbouring green space polygons intersecting each matrix ring; focal patch excluded	Count	Green space subdivision/configuration
Nearest green patch distance	Unified green space polygon layer established in Chapter 3	Euclidean edge-to-edge distance from focal patch boundary to nearest external green space polygon; focal patch excluded	Metres	Structural isolation
Matrix hedge density	UKCEH Land Cover Plus: Hedgerows 2016–2021 England	Hedgerow polylines clipped to matrix rings using intersection function; total length divided by matrix-ring area	m/ha	Potential woody linear structural connectivity
Matrix watercourse density	UKCEH 1:50,000 Digital River Network of Great Britain (UK Centre for Ecology & Hydrology, 2000), cross-referenced with the Environment Agency Statutory Main River Map (Environment Agency, 2025)	Watercourse polylines clipped to matrix rings using intersection function; total length divided by matrix-ring area	m/ha	Potential blue-green linear structural connectivity
Matrix road density	OS Open Roads, Ordnance Survey (2024)	Minor and major roads clipped to matrix rings using intersection function; densities summed	m/ha	Barrier / disturbance pressure

Matrix habitat composition was measured from the UKCEH Land Cover Map 2021 10 m Classified Pixels raster dataset (Marston et al., 2022), consistent with the raster dataset used in Chapter 3. Pixel counts were extracted within each matrix ring using the QGIS zonal histogram tool. Each 10 m × 10 m pixel represented 100 m^2. Cells with no data were excluded, and proportional habitat cover was calculated using the total number of valid classified pixels within each ring in line with the within-patch variables calculated in Chapter 3. Habitat area was therefore calculated as:

$$Habitat\ area\ (m^2)\ =\ pixel\ count\ \times\ 100$$

Habitat area was converted to hectares by dividing by 10,000, equivalent to multiplying the pixel count by 0.01. Percentage cover was then calculated by dividing the pixel count for each land cover class by the total number of classified pixels within the annular rings and multiplying by 100:

$$Habitat\ cover\ (\%) = \left(\frac{habitat\ class\ pixel\ count}{total\ classified\ ring\ pixel\ count}\right) X\ 100$$

For statistical modelling, habitat composition variables were retained as proportional cover values ranging from 0 to 1, calculated as the habitat class pixel count divided by the total classified pixel count within each matrix ring. Where matrix rings extended beyond the study-area boundary or beyond valid raster coverage, calculations were based only on the valid classified area within each ring, defined as within the urban study area baseline presented in Chapter 2 & 3. The final matrix composition variable was calculated as:

$$Matrix\ green\ proportion\ =\ \frac{(valid\ matrix\ pixels\ -\ sealed\ surface\ pixels)}{valid\ matrix\ pixel}$$

where sealed surface was derived from urban and suburban land cover classes. This provided a broad measure of non-sealed matrix cover and reduced redundancy among proportional land cover variables.

Linear feature predictors were processed separately from raster habitat composition. Features were intersected with the 250 m and 500 m matrix rings generated in QGIS. Matrix hedgerows were derived from UKCEH Land Cover Plus: Hedgerows 2016-2021 England, a polyline dataset of mapped woody linear features, including hedgerows, tree lines and semi-natural thickets, produced using Environment Agency lidar data (Broughton *et al.*, 2024). The Chapter 3 woody linear density variable represented a comparable set of features within surveyed patch boundaries, including hedgerows, scrub margins and tree lines, and was manually digitised from aerial imagery and field information. Chapter 4 therefore extends the assessment of woody linear structure into the surrounding matrix using an independently produced landscape-scale dataset. Matrix hedge density represents mapped woody linear features available within the UKCEH dataset and is not intended as a complete inventory of all woody linear vegetation within the surrounding matrix.

Watercourse density was calculated from the hydrological linear feature layer used in the wider study-area GIS framework (Chapter 3). This layer was derived from the UKCEH 1:50,000 Digital River Network of Great Britain (UK Centre for Ecology & Hydrology, 2000), with hydrological features cross-referenced against the Environment Agency Statutory Main River Map (Environment Agency, 2025). This vector layer was used to calculate matrix watercourse density and was therefore separate from the UKCEH Land Cover Map 2021 raster, which was used only for

matrix habitat composition calculations, including freshwater and other land cover classes (Marston et al., 2022).

Road density was calculated from OS Open Roads, which provides a vector representation of the Great Britain road network, including classified and unclassified roads as a structured link-and-node network representing road centrelines (Ordnance Survey, 2024). Minor and major roads were processed separately during GIS extraction and then combined to produce matrix road density. For each buffer scale, hedgerow, watercourse, minor road and major road layers were clipped to the 250 m and 500 m matrix-ring polygons using the QGIS Clip tool. Total clipped feature length was summarised by Site ID and divided by valid matrix-ring area to produce density variables expressed as metres per hectare:

$$Matrix\ feature\ density\ = \frac{total\ clipped\ feature\ length\ (m)}{valid\ matrix - ring\ area\ (ha)}$$

Minor and major road densities were initially calculated separately and then combined:

$$Matrix\ road\ desnity$$
$$= matrix\ minor\ road\ density\ +\ matrix\ major\ road\ density$$

Green patch count and nearest green patch distance were calculated in QGIS using the unified green space polygon layer established in Chapter 3. Green patch count was calculated using the Join attributes by location (summary) tool, with the 250 m and 500 m matrix-ring polygons as the input layer and the unified green space polygon layer as the join layer. A spatial intersects predicate was used to count the number of external green space polygons intersecting each matrix ring. The focal

surveyed patch was not counted as a neighbouring patch, meaning green patch count represented only additional green space polygons within the surrounding matrix. The nearest green patch distance was calculated using the QGIS Shortest line between features tool. Surveyed patch boundaries were used as the source layer, and the neighbouring green space polygons were used as the destination layer.

The focal patch was excluded only from the destination layer so that the distance represented the shortest Euclidean edge-to-edge distance from each surveyed patch to the nearest other green space polygon, rather than a zero-distance match to itself. This value was retained as Nearest Green Patch Distance. These variables represented green space subdivision and structural isolation. Nearest-neighbour distance is commonly used as a structural isolation metric in landscape ecology, although it does not directly measure functional movement between patches (Tischendorf and Fahrig, 2000; Moilanen and Nieminen, 2002; Bender et al., 2003; McGarigal et al., 2012).

All GIS-derived matrix outputs were exported from QGIS and joined in R (v.4.6.0) using Site ID. Separate model-ready datasets were created for the 250 m and 500 m matrix scales. Each dataset contained one row per Site ID × Repeat combination, with the biological response variables varying among repeat surveys and the matrix predictors attached at site level.

Predictor distributions were inspected visually before modelling. Collinearity among candidate predictors was assessed using Spearman correlations and variance inflation factors, following recommendations for ecological models where correlated predictors can inflate uncertainty and obscure interpretation (Zuur et al., 2010;

Dormann et al., 2013). Directly complementary or redundant variables were excluded from the final model set. For example, sealed surface was not included alongside matrix green proportion, and minor and major road densities were combined into a single road-density variable. All retained continuous predictors were standardised separately within each buffer-scale dataset using z-score transformation (Schielzeth, 2010; Kuhn & Johnson, 2013). Standardised variables were identified using the suffix _s. This allowed coefficient estimates to be compared among predictors measured on different scales, including proportions, distances, counts and line densities.

4.2.4 Statistical modelling

Separate matrix-only models were fitted for each response variable at each buffer scale, producing six models in total: species richness, total CPUE and Simpson's reciprocal diversity at 250 m, and the same three responses at 500 m.

The same fixed-effect structure was used for each response and buffer scale:

$$Response \sim matrix\ green\ proportion\ +\ green\ patch\ count + nearest\ green\ patch\ distance\ +\ matrix\ hedge\ density + matrix\ watercourse\ density\ +\ matrix\ road\ density + site\ identity\ as\ a\ random\ intercept\ (1|SiteID)$$

Models were fitted in R (v.4.6.0) using glmmTMB (Brooks et al., 2017). Type II Wald chi-square tests were used to evaluate fixed-effect predictors using the car package (Fox & Weisberg, 2019). The same predictor structure was applied at both buffer scales to allow direct comparison between immediate and broader matrix contexts. Table 4.2 includes the response variables and their respective model family and function.

Table 4.2 Model families used for Chapter 4 matrix-only models.

Response variable	Model family	Link function	Reason
Species richness	Poisson	Log	Count response
Total CPUE	Tweedie	Log	Continuous, non-negative, right-skewed CPUE response including zeros
Simpson's reciprocal diversity	Gaussian	Identity	Continuous community-diversity response

Model diagnostics were undertaken separately for each response variable and buffer scale. Simulated residual diagnostics were generated using DHARMa to assess residual uniformity, dispersion, outliers and zero inflation (Hartig, 2022). Additional collinearity checks were undertaken using the performance package (Lüdecke et al., 2021). Model outputs were summarised using AIC values, fixed-effect estimates, standard errors, confidence intervals and Type II Wald chi-square tests. Coefficients and confidence intervals were extracted using the broom.mixed package (Bolker and Robinson, 2022). Figures were produced using ggplot2 (Wickham, 2016), ggeffects (Lüdecke, 2018) and patchwork.

Fixed-effect coefficient plots were used to compare the direction and strength of matrix effects across response variables and buffer scales. Predicted response plots were generated for key CPUE predictors: nearest green patch distance, green patch count, matrix hedge density and matrix road density. A predicted response plot was also produced for the effect of 500 m watercourse density on Simpson's reciprocal diversity.

Species-specific matrix models were also explored for the four focal taxa but were not retained in the final Chapter 4 analysis. Capture frequencies were highly unequal among species and the less frequently recorded taxa contained many zero observations, producing unstable and inflated coefficient estimates and repeated convergence problems. The final landscape analysis was therefore restricted to species richness, total CPUE and Simpson's reciprocal diversity, for which the matrix models could be fitted reliably.

4.3 Results

4.3.1 Overview of landscape matrix effects

The matrix-only models identified significant associations between several landscape predictors and small mammal community responses at both buffer scales. Green patch count was negatively associated with species richness, CPUE and Simpson's reciprocal diversity at both 250 m and 500 m. Nearest green patch distance was negatively associated with species richness and Simpson's reciprocal diversity at both scales and with CPUE at 500 m. Matrix green proportion was positively associated only with Simpson's reciprocal diversity at 500 m.

CPUE showed two significant associations at 250 m, with negative associations with green patch count and road density. At 500 m, CPUE was negatively associated with green patch count, nearest green patch distance and road density, and positively associated with matrix hedge density. The 500 m Simpson's reciprocal diversity model also identified positive associations with matrix green proportion and watercourse density (Figure 4.2).

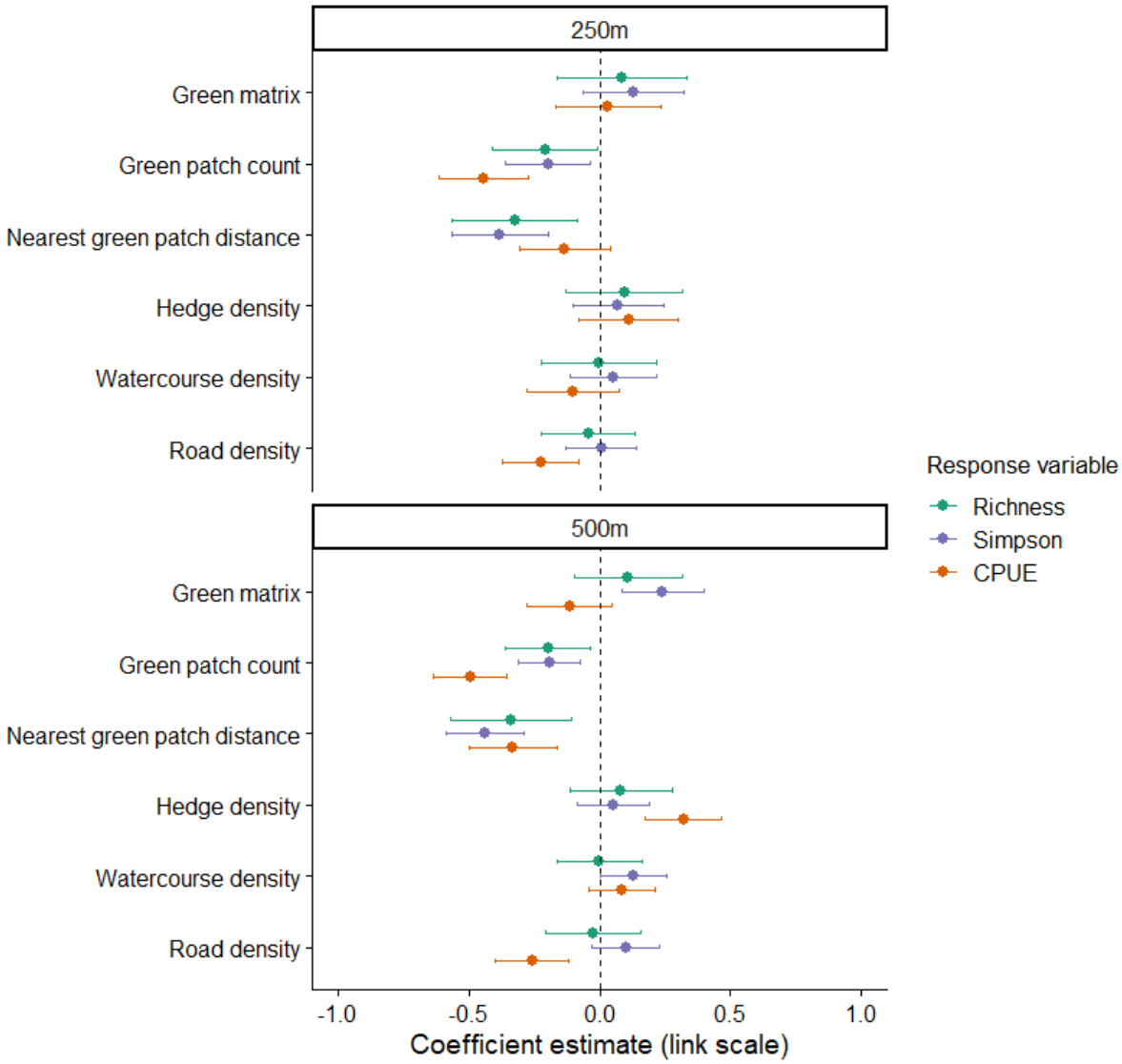


Figure 4.2 Landscape scale effects on small mammal community responses within the urban green space patches tested. Standardised coefficient estimates and 95% confidence intervals are shown from matrix only models fitted at 250 m and 500 m buffer scales for CPUE, species richness and Simpson's reciprocal diversity. The dashed vertical line represents zero effect. Negative coefficients indicate lower community responses with increasing predictor values, and positive coefficients indicate higher responses. Green patch count was negatively associated with all three community responses at both scales. Nearest green patch distance also showed negative coefficients across all responses, with significant associations with richness and Simpson's reciprocal diversity at both scales and with CPUE at 500 m. Matrix hedge density was positively associated with CPUE at 500 m. Green matrix represents the proportion of non-sealed green habitat within each matrix ring.

4.3.2 Species richness

Species richness showed a consistent pattern across both spatial scales (Figure 4.2).

At 250 m, richness declined significantly with increasing green patch count (β = -

0.210 ± 0.104 SE; $\chi^2 = 4.10$, $p = 0.043$) and increasing nearest green patch distance (β

= -0.326 ± 0.122 SE; χ^2 = 7.16, *p* = 0.007). Matrix green proportion, hedge density, watercourse density and road density were not significantly associated with richness. A similar pattern was observed at 500 m. Species richness declined with increasing green patch count (β = -0.199 ± 0.083 SE; χ^2 = 5.71, *p* = 0.017) and nearest green patch distance (β = -0.339 ± 0.117 SE; χ^2 = 8.32, *p* = 0.004). No significant associations were detected for the remaining matrix predictors.

4.3.3 Catch Per Unit Effort (CPUE)

CPUE showed significant associations with two matrix predictors at 250 m and four predictors at 500 m (Figures 4.2 and 4.3). At 250 m, CPUE declined significantly with increasing green patch count (β = -0.444 ± 0.087 SE; χ^2 = 26.19, p < 0.001) and road density (β = -0.226 ± 0.074 SE; χ^2 = 9.45, p = 0.002). Nearest green patch distance, hedge density and watercourse density were not significantly associated with CPUE at this scale.

At 500 m, CPUE declined significantly with increasing green patch count (β = -0.495 ± 0.072 SE; χ^2 = 47.25, p < 0.001), nearest green patch distance (β = -0.332 ± 0.085 SE; χ^2 = 15.31, p < 0.001) and road density (β = -0.259 ± 0.072 SE; χ^2 = 12.89, p < 0.001). Matrix hedge density was positively associated with CPUE (β = 0.322 ± 0.075 SE; χ^2 = 18.60, p < 0.001). Hedge density showed no significant association at 250 m. These fitted relationships are illustrated in Figure 4.3, showing the negative CPUE relationships with green patch count and road density at both scales, the stronger negative relationship with nearest green patch distance at 500 m, and the positive relationship with matrix hedge density at 500 m.

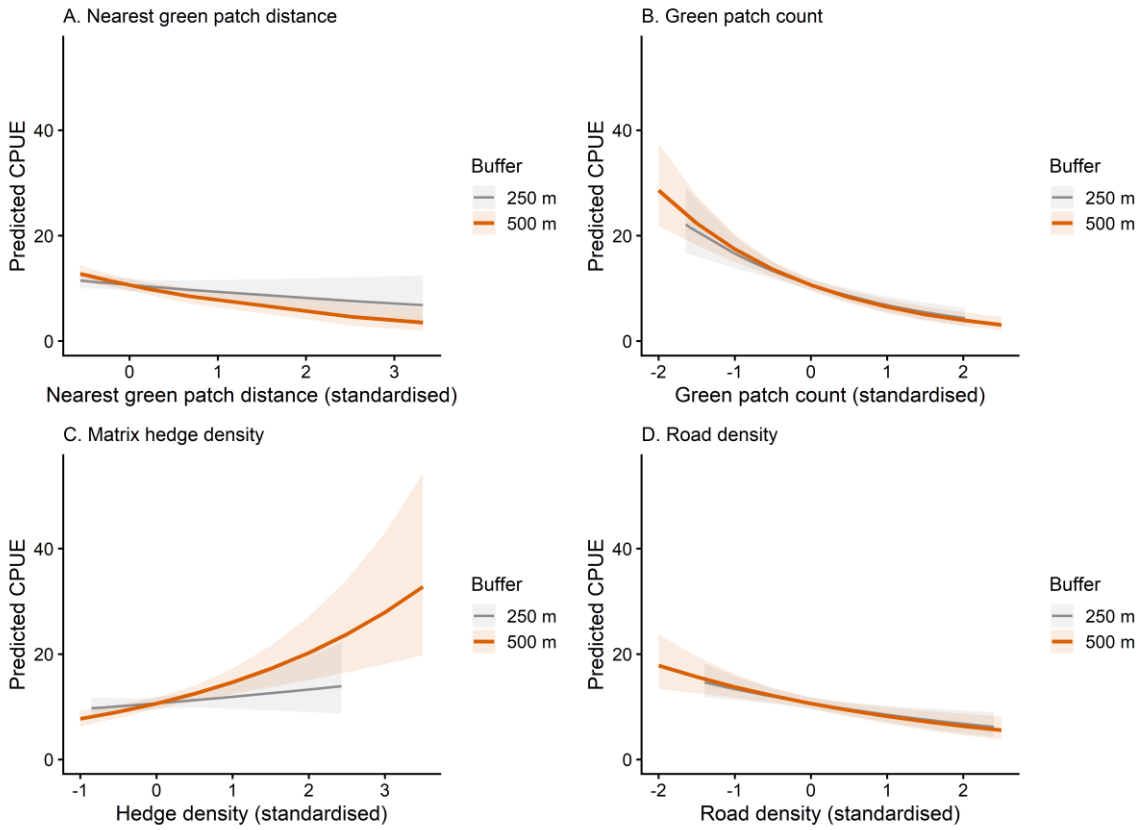


Figure 4.3 Predicted CPUE responses to (A) nearest green patch distance, (B) green patch count, (C) matrix hedge density and (D) matrix road density from the Chapter 4 matrix-only CPUE models. Lines show fitted model predictions and shaded areas show 95% confidence intervals. Nearest green patch distance was negatively associated with CPUE at 500 m but not at 250 m. Green patch count and matrix road density were negatively associated with CPUE at both scales, while matrix hedge density was positively associated with CPUE at 500 m but not at 250 m.

4.3.4 Simpson's reciprocal diversity

The relationships between Simpson's reciprocal diversity, green patch count and nearest green patch distance were consistent across both spatial scales (Figure 4.2). At 250 m, Simpson's reciprocal diversity declined significantly with increasing green patch count ($\beta = -0.199 \pm 0.082$ SE; $\chi^2 = 5.85$, $p = 0.016$) and nearest green patch distance ($\beta = -0.382 \pm 0.094$ SE; $\chi^2 = 16.52$, $p < 0.001$). No significant associations were detected for matrix green proportion, hedge density, watercourse density or road density.

At 500 m, Simpson's reciprocal diversity again declined with increasing green patch count (β = -0.192 ± 0.061 SE; χ^2 = 9.89, p = 0.002) and increasing nearest green patch distance (β = -0.439 ± 0.075 SE; χ^2 = 34.22, p < 0.001). Significant positive associations were also detected with matrix green proportion (β = 0.240 ± 0.080 SE; χ^2 = 8.90, p = 0.003) and matrix watercourse density (β = 0.128 ± 0.064 SE; χ^2 = 3.97, p = 0.046), with the fitted watercourse relationship illustrated in Figure 4.4. Nearest green patch distance had the largest negative coefficient and Wald χ^2 value in the Simpson's reciprocal diversity models at both scales.

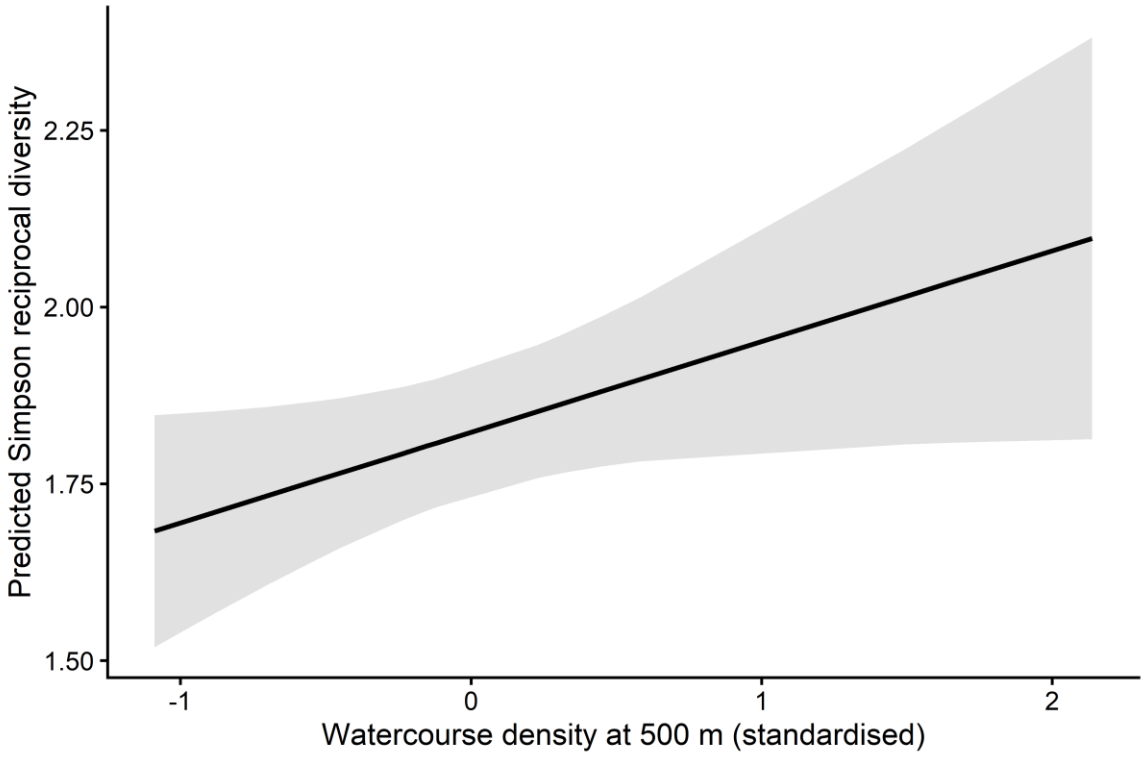


Figure 4.4 Predicted Simpson's reciprocal diversity in response to matrix watercourse density within the 500 m matrix buffer. The line shows the fitted model prediction and the shaded area represents the 95% confidence interval. Simpson's reciprocal diversity increased significantly with increasing matrix watercourse density (p = 0.046).

4.3.5 Summary of landscape effects

Across the Chapter 4 models, green patch count and nearest green patch distance showed the most consistent associations with small mammal community responses. Green patch count was negatively associated with richness, CPUE and Simpson's reciprocal diversity at both 250 m and 500 m. Nearest green patch distance was negatively associated with richness and Simpson's reciprocal diversity at both scales and with CPUE at 500 m. Matrix green proportion showed a significant positive association only with Simpson's reciprocal diversity at 500 m.

The number of significant matrix associations differed between the two buffer scales. At 250 m, significant relationships were largely restricted to green patch count, nearest green patch distance and road density. At 500 m, matrix hedge density was positively associated with CPUE, and matrix watercourse density and matrix green proportion were positively associated with Simpson's reciprocal diversity (Figures 4.3 and 4.4).

4.4 Discussion

Chapter 4 examined how the composition, configuration and potential permeability of the landscape surrounding surveyed urban green spaces were associated with small mammal richness, CPUE and Simpson's reciprocal diversity at 250 m and 500 m.

Chapter 4 addressed the landscape component of H3 developed in Chapter 1, which predicted that small mammal community responses would decline with increasing isolation, green space subdivision and road density, and increase with greater surrounding habitat amount and structurally continuous green or blue linear features.

These expectations were tested at 250 m and 500 m to examine whether relationships with the surrounding matrix changed between the immediate and broader local landscape. Chapter 4 tested these landscape associations independently of the within-patch predictors examined in Chapter 3. Whether patch and matrix predictors retain complementary effects when included together is addressed in Chapter 5.

The landscape component of H3 was broadly supported, but the predicted relationships were not shared equally among the three community measures or between spatial extents. These results show that several properties of the surrounding landscape were associated with the small mammal communities recorded within surveyed green spaces, with the pattern depending on the response measured and the spatial extent considered.

4.4.1 Green space patch subdivision and structural isolation were consistently associated with community responses

Green patch count and nearest green patch distance produced the most consistent negative landscape relationships in Chapter 4. Green patch count was negatively associated with species richness, CPUE and Simpson's reciprocal diversity at both 250 m and 500 m. Nearest green patch distance was negatively associated with richness and Simpson's reciprocal diversity at both scales, and with CPUE at 500 m. The recurrence of these relationships across the community measures supports the prediction under H3 that the configuration and separation of surrounding green space would be associated with small mammal communities within the surveyed patches.

Green patch count recorded the number of separate green space polygons within each matrix ring, whereas matrix green proportion represented the amount of broadly

non-sealed habitat. A higher count could therefore indicate greater subdivision, although it did not describe the size, quality or accessibility of individual green spaces.

The repeated negative association with green patch count is consistent with the idea that greater subdivision of the surrounding green space network is associated with lower community responses. Habitat subdivision alters the size, arrangement and separation of habitat units without requiring an equal reduction in total habitat amount (Fahrig, 2003; Fischer & Lindenmayer, 2007). Young and Jarvis (2001) described a comparable pattern within the Black Country, UK, a polycentric urban area west of Birmingham comprising several large, interconnected industrial towns. They suggested that urbanisation had produced a fine-grained mosaic of habitats differing in size, character and the number and quality of connections between them. Their work also showed why the number of mapped habitat units alone provides an incomplete representation of how those habitats are arranged within an urban landscape.

The direction of the relationship does, however, agree with evidence from urban small mammal studies. Gomes et al. (2011) recorded lower species richness and relative abundance with increasing urbanisation across isolated habitat patches in Porto and found that *A. sylvaticus* specifically benefited from habitat mosaics and pervious land in the surrounding matrix. Larson and Sander (2024) similarly identify fragmentation and low dispersal ability as important concerns for urban small mammals, citing evidence that less mobile or more specialised taxa become less frequent with increasing urbanisation. These studies support the wider expectation

that subdivision of urban habitat can alter the communities retained within individual green spaces.

The relationship is, however, not universal; Larson and Sander (2024) found no significant community-level relationship between their contagion-based fragmentation metric and either species richness or diversity in Iowa City, USA. Species diversity actually increased with human-modified land cover in their study, driven by sites where tall vegetation and water features supported additional prairie-associated species. Their species-level results also differed, with short-tailed shrew recruitment increasing at more-connected sites and deer mouse recruitment increasing at lower-connectivity sites. This does not conflict with the Chapter 4 results so much as demonstrate that fragmentation metrics do not have a fixed ecological meaning across urban landscapes. Their effects depend on the habitats being divided, the composition of the intervening landscape and the species forming the community. This supports the cautious interpretation of green patch count as a measure of subdivision within the Stoke-on-Trent and Newcastle-under-Lyme landscape, without assuming that a higher number of green spaces will always produce the same biological response.

Nearest green patch distance provides a separate measure because it describes the physical separation between the edge of the surveyed patch and the nearest neighbouring green space. Increasing distance was consistently associated with lower species richness and Simpson's reciprocal diversity, with the same negative relationship detected for CPUE at 500 m. The species richness response agrees with the isolation component of island biogeography, where increasing separation from

other habitat is expected to reduce immigration and contribute to lower species richness (MacArthur & Wilson, 1967). Metapopulation theory extends this expectation through the effects of inter-patch separation on migration, colonisation and population persistence (Hanski, 1998). Hanski specifically identifies patch area and isolation as established influences on migration, colonisation and extinction within fragmented landscapes. These processes were not measured directly here, but they provide an ecological basis for the repeated negative relationship with nearest green patch distance.

The island analogy cannot be completely applied to urban green spaces because the urban matrix comprises roads, sealed surfaces, gardens, verges, rough vegetation, watercourses, and other land cover types with distinct ecological characteristics. Ricketts (2001) developed the concept of effective isolation in relation to this problem. His mark-recapture study showed that different matrix types imposed different levels of resistance to movement, meaning that patches separated by an equivalent geographical distance could differ in their effective isolation. Responses also differed among taxa. Bender et al. (2003) and Bender and Fahrig (2005) similarly emphasised that geographical separation alone cannot fully represent inter-patch movement where matrix structure differs.

Evidence from more recent studies on *A. sylvaticus* provides a direct example within the small mammal species captured in this study. Wilson et al. (2016) found greater genetic differentiation among urban populations than among populations in arable landscapes. Euclidean and overland distance best explained differentiation in the arable landscape and measures incorporating variation in habitat quality performed

better within the urban landscape. The authors concluded that urban habitat fragmentation had increased population isolation and that simple geographical distance did not adequately describe the urban landscape experienced by *A. sylvaticus*. This provides strong support for interpreting nearest green patch distance as structural isolation while recognising that the biological effect of any given distance will also depend on what occurs between the two green spaces.

These results therefore offer an additional explanation for variation in small mammal communities among surveyed green spaces beyond the within-patch habitat and management associations identified in Chapter 3. Both the subdivision of surrounding green space and the physical separation from neighbouring green space were repeatedly associated with community responses. Green patch count and nearest green patch distance describe separate properties of that landscape structure and should not be treated as interchangeable measures. Their negative relationships support the Chapter 4 prediction that subdivision and structural isolation are associated with reduced responses of small mammal communities within urban green spaces. The composition of the intervening matrix is then important for understanding what separates those patches. Road density and matrix hedge density provide the clearest evidence for this within the CPUE models and are considered in the following section.

4.4.2 Road density and matrix hedge density were significantly associated with CPUE

Road density and matrix hedge density showed clear associations with CPUE, but their relationships differed in direction and spatial extent. Road density was negatively associated with CPUE at both 250 m and 500 m. Matrix hedge density was

positively associated with CPUE at 500 m and showed no significant relationship at 250 m. Neither predictor was significantly associated with species richness or Simpson's reciprocal diversity. These results provide support for the H3 predictions concerning road density and woody linear features, but the evidence was restricted to CPUE. The response therefore concerns variation in capture rate across the surveyed green spaces and does not demonstrate equivalent changes in species number or the relative distribution of captures among species.

The negative association with road density agrees with experimental and genetic evidence showing that roads can restrict small mammal movement. McDonald and St. Clair (2004) experimentally translocated three murid species across a major highway, a natural forest barrier and continuous habitats. Individuals were less successful at returning across the highway than across either continuous habitat or the natural barrier, although the strength of this response differed among species. Ascensão et al. (2016) found a similar effect using genetic data for *A. sylvaticus*. Their results indicated that highways acted as partial barriers, with road-surface avoidance identified as the main explanation for the observed genetic pattern. Some movement across highways still occurred, with the estimated migration rate between opposite roadsides around 5%. These studies show that roads do not need to form absolute barriers to affect small mammal movement between habitat areas, and the perception of them to the organism is often enough to deter less mobile species from crossing them to otherwise suitable habitat patches.

The Chapter 4 models did not measure road crossing, mortality, traffic volume or behavioural avoidance. Matrix road density represented the total length of mapped

roads within each matrix ring. The negative CPUE relationship therefore cannot identify which road-related process produced the observed pattern. It does show that green spaces surrounded by greater road density tended to have lower small mammal capture rates. This relationship remained significant at both spatial extents. The consistency of the direction agrees with H3 and with evidence that transport infrastructure can impede movements by small terrestrial mammals. It also adds ecological meaning to the subdivision and isolation relationships identified in Section 4.4.1. Separate green spaces may occur close together geographically, but the landscape between them can contain features that affect the ease with which animals move through it.

Road-associated land is not uniformly unsuitable for small mammals, though, as the structure of the vegetation surrounding a road can substantially alter its ecological role. Ascensão et al. (2012) recorded higher small mammal abundance in fenced highway verges than in adjacent land uses within an agricultural landscape. The authors attributed this pattern to the greater height and cover of herbaceous and shrub vegetation retained within the verges. Galantinho et al. (2020) demonstrated this complexity directly for *A. sylvaticus*. The road reduced fine-scale landscape connectivity and acted as a partial barrier to movement. Vegetated verges simultaneously promoted movements around the road network, and vegetation cutting reduced this function, which ties in with the reduced capture rates in intensely mown patches in Chapter 3. Because road density did not describe verge vegetation, the negative coefficient applies to road-dense matrices as a whole and does not imply that every road verge had a negative ecological effect.

Evidence from agricultural systems can also further provide reason for this increase along hedge and vegetated boundaries. Tattersall et al. (2002) found that field boundaries supported the highest species richness of the habitats they surveyed. *M. glareolus* was more abundant in linear hedgerows than in non-linear woodland, and individuals were also recorded in habitats adjoining those hedges. The study found no general effect of habitat linearity on *M. agrestis*, *A. sylvaticus* or *S. araneus*. This is useful for interpreting the Chapter 4 result because it shows that ecological value cannot be assigned simply from the linear form of a feature. Vegetation structure, surrounding habitat and species ecology remain important. The positive matrix hedge relationship detected here therefore provides evidence for an association between the amount of mapped woody linear structure and CPUE. It does not demonstrate that those hedges functioned as movement corridors.

The absence of a significant hedge relationship at 250 m in comparison to the 500 m result does not establish that small mammals travelled 500 m along hedgerows or identify 500 m as a movement distance. Increasing the analysis extent incorporates a greater portion of the surrounding hedgerow network. The 500 m measure may therefore describe broader variation in the availability of woody linear vegetation that was not captured within the smaller matrix ring. The Chapter 4 data, however, provide structural evidence only. The emergence of the hedge relationship at 500 m could indicate that woody linear features become more relevant when a greater extent of the surrounding urban landscape is considered. Connected landscape features can increase opportunities for movement between habitat areas and access to resources located beyond the immediate surroundings of individual patches (Baguette et al.,

2013; LaPoint et al., 2015). The wider issue of spatial extent is addressed in Section 4.4.4.

The matrix hedge result also extends an ecological pattern already identified within the surveyed green spaces. Chapter 3 found that woody linear density within patch boundaries was positively associated with richness, CPUE and Simpson's reciprocal diversity. Chapter 4 identified a positive association between mapped hedge density in the surrounding landscape and CPUE at 500 m. The variables were derived independently and covered different spatial areas, so they should not be treated as equivalent measures. They do show that woody linear vegetation remained associated with small mammal responses after the analysis moved beyond the surveyed patch boundary, regardless of within patch features, and the response detected in the surrounding matrix was restricted to CPUE and the 500 m extent.

Road density and hedge density, therefore, provide two complementary pieces of evidence concerning the landscape between urban green spaces. Direct movement was not measured, so neither relationship establishes the route taken by individual animals. The direction of both relationships aligns with H3, which expected that road-dense landscapes would be associated with lower community responses and that greater woody linear structure would support higher CPUE. Together with the subdivision and structural isolation results in Section 4.4.1, these findings show that the surrounding matrix contained information relevant to small mammal capture rates that was not represented by the location of neighbouring green spaces alone.

4.4.3 Matrix green proportion and watercourse density were associated with Simpson's reciprocal diversity at 500 m

The Simpson's reciprocal diversity models identified two positive landscape relationships that were not detected for species richness or CPUE. At 500 m, Simpson's reciprocal diversity increased with matrix green proportion and watercourse density. Neither variable was significantly associated with Simpson's reciprocal diversity at 250 m. These results provide partial support for the H3 prediction that greater surrounding habitat amount and blue linear features would support community diversity. The support was specific to Simpson's reciprocal diversity at the 500 m buffer applied to the surveyed green spaces (Figures 4.2 and 4.4). The Chapter 4 results therefore show that the amount of surrounding green habitat and the density of watercourses were associated with Simpson's reciprocal diversity without producing equivalent increases in the number of species captured or total CPUE.

The positive relationship between matrix green proportion and Simpson's reciprocal diversity is important when considered alongside the negative green patch count relationships discussed in Section 4.4.1. Increasing the amount of green habitat and increasing the number of separate green space units do not describe the same landscape condition. A 500 m matrix can contain a relatively large amount of green habitat distributed across fewer larger areas or divided among many separate units (Fahrig, 2003; Olff & Ritchie, 2002; Fahrig, 2013). Habitat amount and spatial configuration, therefore, represent separate components of landscape structure, even where they occur within the same surrounding area. Olff and Ritchie (2002), for

example, demonstrated that habitat cover and spatial fragmentation can vary independently and can produce different biodiversity responses.

The Chapter 4 models included both variables, and their opposing associations with Simpson's reciprocal diversity show that habitat amount and the spatial subdivision of that habitat contained different ecological trends within this urban landscape. The pattern suggests that communities tended to have higher Simpson's reciprocal diversity where the amount of surrounding green habitat increased and lower Simpson's reciprocal diversity where that habitat occurred as a greater number of separate green space units. Because matrix green proportion was not significantly associated with species richness, the positive Simpson relationship is consistent with changes in the relative distribution of captures among species being an important component of the response (Hillebrand et al., 2008; Morris et al., 2014). It cannot establish which species increased or declined because species-specific matrix relationships were not resolved in Chapter 4. The combined green proportion and patch-count results do indicate that the amount and spatial arrangement of surrounding green habitat can carry different effects on community diversity. This agrees with evidence that habitat amount and configuration can retain separate ecological effects within fragmented landscapes (Fahrig, 2003; Haddad et al., 2017; Fletcher et al., 2018; Rybicki et al., 2019).

The Habitat Amount Hypothesis proposed by Fahrig (2013) is again important to the findings of this research, which places greater emphasis on the total amount of suitable habitat within the surrounding landscape. Subsequent tests have produced mixed evidence. Haddad et al. (2017) reported experimental evidence that did not

support the Habitat Amount Hypothesis, whereas the global synthesis by Watling et al. (2020) found support for habitat amount as an important influence on species density. Chapter 4 is not a formal test of the Habitat Amount Hypothesis because matrix green proportion represented broad non-sealed habitat and was not defined separately according to the habitat requirements of each small mammal species. Green patch count and nearest green patch distance also retained significant relationships within the same Simpson's reciprocal diversity model. The positive matrix green relationship does, however, show that surrounding non-sealed habitat amount contributed to the observed community pattern at 500 m. Its absence from the richness and CPUE models also demonstrates that this contribution was not expressed uniformly across the three community measures.

Small mammal research from other modified landscapes supports the broader importance of landscape composition and also shows why broad green cover should not be expected to produce a single community response. Paniccia et al. (2022) found that landscape composition and configuration were associated with small mammal species composition and functional traits across Mediterranean landscapes. Species and trait composition varied along gradients of increasing artificial and agricultural land cover and decreasing seminatural habitat cover, demonstrating that changes in surrounding land cover composition were associated with changes in the structure of small mammal assemblages. Fischer et al. (2011) also found that small mammal abundance and community metrics responded to landscape complexity in agricultural systems, with responses varying among taxa and analytical extents. These studies support the broader expectation that the composition of the surrounding landscape can alter small mammal communities,

often in ways that are missed by abundance models, richness-only models, or combinations of the two, even when these models do not account for community evenness.

The positive association with matrix watercourse density adds a separate landscape component to the Simpson's reciprocal diversity response. Direct research examining these relationships within urban watercourse networks remains limited.

Bohdal et al. (2016) examined small mammal communities along five streams within a Central European cultural landscape and recorded 547 individuals from 10 species. Riparian vegetation was identified as an important linear landscape feature used by small mammals, and species diversity and community equitability increased with stream width. The study also recorded individuals crossing watercourses tens of metres wide, although crossing frequency for *A. flavicollis* declined as stream width increased. This species was recorded too infrequently in the present study for a comparable assessment. These findings, however, show that watercourse systems can contribute to riparian habitat and potential movement routes, but that the channel itself can also restrict movement between adjoining areas. The ecological role of a watercourse therefore depends on both its associated habitat and its physical form, which may be even more apparent in a landscape which is heavily modified by human activity.

Evidence from urban landscapes gives this relationship greater relevance to this Chapter 4 study. Larson and Sander (2024) found unexpectedly high small mammal richness at urban sites associated with streams in turfgrass parks and stormwater channels beside housing developments. These areas contained tall, dense

vegetation and supported prairie-associated species that were less frequent elsewhere in the urban landscape. The authors suggested that vegetation surrounding streams and stormwater channels could provide habitat or movement routes and could also concentrate small mammal activity within these linear areas. Their findings are important here because they show that water-associated features retained ecological value for small mammals even when embedded within developed land, and that the vegetation accompanying those features may influence both habitat availability and potential mobility.

Evidence from less urbanised river systems similarly indicates that riparian habitats can support diverse small mammal assemblages, although their effects on richness, abundance and evenness are not uniform (Hamilton et al., 2015; Romanowski et al., 2023).

This urban setting also adds complexity to these relationships due to the nature of the watercourses present in the study area. The Stoke-on-Trent watercourse network includes streams, canals and drainage systems crossing the built matrix, often with riparian or other edge vegetation associated with them (Staffordshire Wildlife Trust, 2024). A greater density of these features may therefore coincide with more linear habitat boundaries, riparian vegetation and structurally varied land across the surrounding matrix. It may also provide additional opportunities for movement between habitat areas (Bohdal et al., 2016; Larson & Sander, 2024). These characteristics were not measured separately in Chapter 4, and the mapped network also includes watercourses that may be heavily modified or channelised, characteristics known to alter the ecological condition of freshwater and riparian

systems (Riley et al., 2018). The positive Simpson relationship therefore indicates an association with the amount of watercourse structure across the 500 m matrix but cannot establish which characteristics of those watercourses produced the relationship. Further work separating channel type, bankside vegetation, degree of modification and direct small mammal movement would help determine how different components of the urban watercourse network contribute to community structure.

The absence of corresponding relationships with richness or CPUE is also important. Greater watercourse density was associated with Simpson's reciprocal diversity without a significant increase in the number of species captured or total capture rate. The relationship therefore appears to concern the relative structure of the recorded community, although the species responsible cannot be identified from the Chapter 4 models. Together with the positive matrix green proportion relationship, this provides further support for H3 at 500 m and leads directly to the question of why these associations emerged only at the broader spatial extent, which is considered in Section 4.4.4.

4.4.4 Matrix associations differed between 250 m and 500 m

Comparison of the 250 m and 500 m models showed a clear change in the number and type of significant associations detected as the area used to characterise the surrounding matrix increased (Figure 4.2). The change in spatial extent therefore added information to the matrix models without removing any of the relationships already detected at 250 m.

The clearest spatial-extent response occurred among hedge density, matrix green proportion and watercourse density. None was significantly associated with any community response at 250 m. At 500 m, hedge density was positively associated with CPUE, while matrix green proportion and watercourse density were positively associated with Simpson's reciprocal diversity (Figures 4.2-4.4). These findings indicate that the ecological information represented by those variables became detectable only when a larger area surrounding each surveyed patch was included. This is consistent with the wider research into scale-dependent landscape associations, as they can depend on the spatial extent over which environmental variables are measured, and different landscape variables can become informative over different extents (Wiens, 1989; Jackson & Fahrig, 2012; Miguet et al., 2016). The Chapter 4 models do not identify 500 m as a movement distance or biological threshold. They show that increasing the extent from 250 m to 500 m revealed additional associations with the surrounding matrix.

Comparable patterns have been reported in small mammal studies using multiple spatial extents. The Fischer et al. (2011) study is again especially relevant, as they analysed landscape complexity at 100 m, 250 m and 500 m and found that the extent at which significant relationships were detected differed among species. *A. agrarius* and *M. arvalis* showed relationships with landscape complexity at 100 m, while the relationship for *S. araneus* was detected at 500 m. Their results demonstrate that small mammal responses to the same broad landscape gradient can be detected over different surrounding areas. This is relevant to Chapter 4 because the additional 500 m relationships were also selective: they involved CPUE and Simpson's reciprocal diversity, not a uniform increase in the number of significant predictors

across every community response. Fischer et al. (2011) therefore provide empirical support for treating spatial extent as part of the ecological interpretation of a landscape relationship, without assuming that one extent is appropriate for every component of a small mammal community. It could be argued that additional scales could have been tested in this specific study, and 50 and 100 m were considered, but they offered no further explanation for the relationships observed with the focal taxa.

Other small mammal studies show that increasing extent does not necessarily reveal additional relationships, too. Bowman et al. (2002) examined small mammal abundance against landscape variables across several local extents, including 250 m and 500 m, but variables measured at 500 m were not significant in their study. Dorigo et al. (2021) also examined small mammal responses using both 250 m and 500 m landscape measures and obtained informative models at both extents. Together with Fischer et al. (2011), these studies show that there is no general expectation that the largest area examined should produce the strongest landscape signal. The outcome depends on the landscape being sampled, the environmental variable being measured and the biological response under consideration. The Chapter 4 pattern is therefore notable because expanding the matrix to 500 m retained all associations detected at 250 m and added relationships with hedge density, matrix green proportion and watercourse density.

Using both spatial extents consequently provided information that would have been lost if the surrounding matrix had been represented by a single buffer. The comparison therefore separates associations that were consistent across both analytical extents from those that became detectable only when the broader matrix

was considered. This supports the Chapter 4 prediction that associations with the surrounding matrix would vary with spatial extent and demonstrates the value of examining the same landscape predictors across more than one predefined area.

4.4.5 Limitations and management implications

The Chapter 4 analysis has several limitations that define the level at which the landscape relationships can be interpreted. Matrix predictors were calculated for 21 surveyed patches and represented broad properties of the surrounding landscape. Green patch count did not incorporate the size or habitat quality of neighbouring green spaces, nearest green patch distance measured geographical separation only, and road, hedge and watercourse densities described the amount of mapped linear structure without measuring traffic, vegetation condition, channel form or direct animal movement. Matrix green proportion similarly represented broad non-sealed cover and did not separate habitats according to their suitability for individual species. These variables are therefore useful indicators of landscape structure, but they cannot establish functional connectivity or the mechanisms responsible for the recorded associations (Tischendorf & Fahrig, 2000; Zeller et al., 2012). Species-specific matrix models were also explored but could not be retained reliably because the less frequently captured taxa produced sparse datasets, unstable coefficients and convergence problems. Future landscape-scale work based on greater species-specific capture numbers could therefore test whether the community-level matrix relationships identified here are shared among species or are driven by different responses to the surrounding urban landscape.

A further limitation concerns matrix hedge density, as the UKCEH Land Cover Plus: Hedgerows dataset represents mapped woody linear features associated with its

land cover boundary framework and is not a complete inventory of every hedge, treeline, scrub margin or vegetated boundary occurring within the surrounding matrix (Broughton et al., 2024). Matrix hedge density should therefore be interpreted as the amount of woody linear structure captured by that dataset, not the total extent of such features present on the ground. Despite this incomplete representation, matrix hedge density showed a strong positive association with community CPUE at 500 m ($\beta = 0.322$, $p < 0.001$), showing that the mapped surrounding woody network still contained ecological information relevant to the small mammal communities recorded within the focal green spaces.

Despite these constraints, the results have clear implications for the way urban green space networks are considered. Urban biodiversity planning should therefore consider the arrangement of green spaces alongside their total area and retain linear vegetation and ecologically valuable watercourse features where possible. This agrees with wider urban ecological work showing that biodiversity depends on the spatial arrangement, connectivity and quality of habitat distributed through the urban matrix (Angold et al., 2006; Beninde et al., 2015; LaPoint et al., 2015; Lepczyk et al., 2017).

4.4.6 Conclusions

Chapter 4 answers Research Question 2 by demonstrating that the composition, configuration and structural characteristics of the surrounding urban matrix were associated with small mammal richness, community CPUE and Simpson's reciprocal diversity within surveyed green spaces. The landscape component of H3 was supported across several of its predictions, although support differed among community responses and between the 250 m and 500 m analytical extents. H3 was

therefore supported in showing that landscape configuration, isolation, road density and selected components of surrounding green and blue structure were associated with community responses, but these relationships were not expressed uniformly across all three community measures.

Chapter 4 therefore extends the patch-scale findings of Chapter 3 by showing that variation among small mammal communities was associated not only with conditions within individual green spaces, but also with the landscape in which those patches were embedded. The repeated negative relationships with green space subdivision and structural isolation indicate that the amount of green space alone does not describe the surrounding landscape context, while the contrasting associations with roads, hedges, broader green cover and watercourses show that different components of the matrix contained separate ecological information. The comparison between 250 m and 500 m further demonstrated that some relationships were evident across both analytical extents, whereas others became detectable only when a broader portion of the surrounding matrix was considered. These findings do not establish functional connectivity or direct movement between patches, but they show that urban green spaces cannot be interpreted as ecologically independent units when assessing small mammal community responses. Chapter 5 therefore builds directly on this result by combining the landscape relationships identified here with the within-patch habitat and management relationships from Chapter 3 to determine which remain associated with community CPUE when considered within a single modelling framework.

Chapter 5 Spatially Explicit Predictive Modelling of Urban Small Mammal CPUE under Varying Grassland Management Scenarios

5.1 Introduction

Chapters 3 and 4 identified complementary associations between small mammal community responses and environmental conditions operating within surveyed green spaces and across the surrounding urban matrix. Chapter 3 showed that CPUE increased with within-patch woody linear density and grass cover, and decreased with patch area, path density and management intensity (Figures 3.2–3.5). Chapter 4 identified additional matrix-scale relationships with CPUE. At 250 m, CPUE decreased with green patch count and road density. At 500 m, these relationships remained, nearest green patch distance was also negatively associated with CPUE, and hedge density was positively associated with CPUE (Figures 4.2–4.4). These analyses were conducted separately, so they did not establish which patch-scale and matrix-scale predictors retained explanatory value when considered within the same model.

As outlined in Chapter 1, one of the wider gaps addressed by this thesis is the limited integration of local habitat conditions with surrounding landscape structure in urban fauna research. Studies frequently examine individual environmental variables or broad urban gradients, whereas mammals remain underrepresented relative to birds and invertebrates in the wider urban ecological literature (Lepczyk et al., 2017; Rega-Brodsky et al., 2022; Wang et al., 2025). This is directly relevant to small mammals because habitat conditions within individual green space patches and the characteristics of the surrounding matrix represent separate components of the

urban landscape. Chapter 5, therefore, tests these components together to determine whether associations identified independently in Chapters 3 and 4 remain evident once patch-scale and matrix-scale predictors are incorporated within a common model. It also provides a predictive response for the wider urban green space patches that were not surveyed during the research phase, to test whether small mammal distribution could be estimated across the wider landscape.

This combined analysis also addresses Research Question 3 and H4 established in Chapter 1. Research Question 3 asks whether local patch conditions and wider landscape structure retain complementary effects when considered within an integrated model, while H4 predicts that patch-scale habitat structure, grassland management and landscape configuration retain complementary effects within a combined predictive model. Chapter 5 is therefore the first stage of the thesis in which predictors derived from within surveyed green spaces and from the surrounding matrix are evaluated simultaneously. As established in Chapter 2, the spatial framework for this analysis was developed across a network of 1,280 mapped green space patches within Stoke-on-Trent and Newcastle-under-Lyme. Twenty-one patches were selected for repeated small mammal surveys, with the remaining 1,259 green spaces retained for subsequent spatial analysis and prediction (Figure 2.3). Environmental variables were also derived across unsurveyed green spaces where equivalent spatial data were available. This allowed relationships identified from surveyed sites to be transferred into the wider spatial framework once a combined model had been developed.

The analytical framework established in Chapter 2 also required variables to have ecological relevance and to be measured consistently across surveyed and unsurveyed green spaces. Patch-scale and matrix-scale variables were initially evaluated separately in Chapters 3 and 4 so that their respective associations with small mammal responses could be established before integration in Chapter 5. This provides the basis for the final modelling process used here. Predictors carried forward into the combined model were therefore informed by the preceding analyses and by whether equivalent variables could be derived across the unsurveyed prediction frame.

As further outlined in Chapter 1, spatial prediction provides a means of extending ecological relationships beyond surveyed locations where comparable environmental variables are available. Predictive habitat modelling links biological observations with environmental gradients and has developed alongside increasing availability of Geographic Information Systems (GIS) and remotely sensed environmental data (Guisan & Zimmermann, 2000; Elith & Leathwick, 2009). This is useful where ecological surveys sample only part of a landscape, but planning, conservation, or management decisions extend across many unsurveyed locations (Stoke-on-Trent City Council, 2013; Aronson et al., 2017; Łopucki & Kiersztyn, 2015; Stoke-on-Trent City Council, 2025). Chapter 5 applies this approach to the green space network established in Chapter 2 by using the relationships retained within the combined model to predict CPUE across unsurveyed green spaces. However, fitted model performance alone does not establish how well a model will predict at locations that contributed no biological information to model fitting (Roberts et al., 2017; Valavi et al., 2019; Araújo et al., 2019). This is especially relevant here because

repeated seasonal observations were collected within the same 21 survey sites. Random separation of individual observations between model fitting and validation could therefore retain information from the same sites in both datasets and produce an overly confident estimate of predictive performance. Chapter 5 addresses this through leave-one-site-out cross-validation, in which complete survey sites are withheld in turn before their CPUE is predicted from a model fitted to the remaining sites (Roberts et al., 2017; Araújo et al., 2019; Valavi et al., 2019).

The environmental range represented by the surveyed sites also limits where the fitted model can be applied with support from the training data (Elith et al., 2010; Mesgaran et al., 2014; Yates et al., 2018). As discussed in Chapter 1, statistical models can return predictions outside the environmental conditions represented during model fitting, but this does not establish that the fitted relationships continue under those new conditions (Araújo et al., 2019; Franklin, 2023). Chapter 5, therefore, compares the environmental characteristics of surveyed and unsurveyed green spaces before retaining final spatial predictions. Unsurveyed patches falling outside the environmental range represented by the training sites are identified separately and excluded from the final prediction maps.

Total CPUE was selected as the response for the combined model because Chapters 3 and 4 identified clear relationships between CPUE and predictors operating at both patch and matrix scales. CPUE also provides a continuous community-level capture response that can be applied consistently across the prediction frame. As outlined in Chapter 1, species richness, CPUE and Simpson's reciprocal diversity represent complementary components of the small mammal community and are not expected

to respond identically to environmental conditions (Hillebrand et al., 2008; Morris et al., 2014). Richness and Simpson's reciprocal diversity were therefore retained within Chapters 3 and 4 to evaluate variation in species number and community structure, but were not carried forward as additional spatial prediction responses in Chapter 5.

Individual species were also not carried forward into the spatial prediction framework. Species-specific responses were examined in Chapter 3, while attempts to extend these analyses to the surrounding matrix in Chapter 4 were limited by sparse captures and high numbers of zero observations for the less frequently recorded taxa, which resulted in unstable estimates and convergence problems. Small sample sizes can reduce predictive performance and increase variability in species-level models, while sparse ecological count data can create additional problems for fitting and evaluating complex models (Wisz et al., 2008; Bolker et al., 2009; Harrison et al., 2018). Extending these responses into spatial prediction models would therefore have introduced uncertainty that was not supported by the available data. The combined CPUE model subsequently provides a community-level prediction of relative capture rate and should not be interpreted as a prediction of species richness, Simpson's reciprocal diversity or the response of individual taxa (Hillebrand et al., 2008; Morris et al., 2014; Larson & Sander, 2024). The purpose of Chapter 5 is therefore to test which retained patch-scale and matrix-scale predictors remain associated with CPUE when included together, and to establish whether the resulting model can be applied across environmentally comparable unsurveyed green spaces.

The final model is applied under low, moderate and intensive grassland-management scenarios. Management intensity is varied while the remaining patch-scale and matrix-scale predictors are retained, allowing predicted CPUE to be compared under alternative mowing regimes across the supported prediction frame. This directly addresses Research Question 4 and the second component of H4 established in Chapter 1, which predict that environmentally bounded spatial prediction can be used to compare grassland-management scenarios and that predicted CPUE will decline from low through moderate to intensive mowing, with the magnitude of the management relationship varying among patches according to their structural condition and surrounding matrix.

Research questions addressed in this chapter are therefore:

1. Do patch-scale habitat and management predictors and matrix-scale predictors retain complementary associations with total small mammal CPUE when considered within an integrated model?
2. How well does the combined model predict CPUE at surveyed green spaces excluded from model fitting?
3. Which unsurveyed green spaces fall within the environmental range represented by the surveyed training sites?
4. How does predicted CPUE change across environmentally comparable unsurveyed green spaces under low, moderate and intensive grassland-management scenarios?

Based on the results of Chapters 3 and 4, greater within-patch woody linear density and matrix hedge density were expected to remain positively associated with CPUE, while increasing management intensity, green patch count and road density were expected to retain negative associations. In accordance with H4, predicted CPUE was also expected to decline from low through moderate to intensive mowing scenarios, with the size of the predicted change varying among green spaces according to their patch-scale and matrix-scale characteristics.

Chapter 5 therefore brings together the analytical components developed through the preceding chapters. Chapters 3 and 4 established patch-scale and matrix-scale relationships separately. Chapter 5 tests which of these relationships remain when both are included within the same model, evaluates prediction at withheld sites, identifies where spatial prediction is supported by the environmental range represented by the surveyed patches, and applies the resulting model across the wider urban green space network.

5.2 Methods

5.2.1 Predictive modelling framework

The workflow followed four stages: first, the patch-scale trapping dataset from Chapter 3 was joined with the landscape matrix dataset from Chapter 4; second, a final combined CPUE model was developed using retained patch- and matrix-scale predictors; third, model performance was assessed using observed-versus-fitted values and site-blocked cross-validation; and fourth, the fitted model was applied to environmentally comparable unsurveyed green space patches under alternative grassland management scenarios. The workflow is summarised in Figure 5.1.

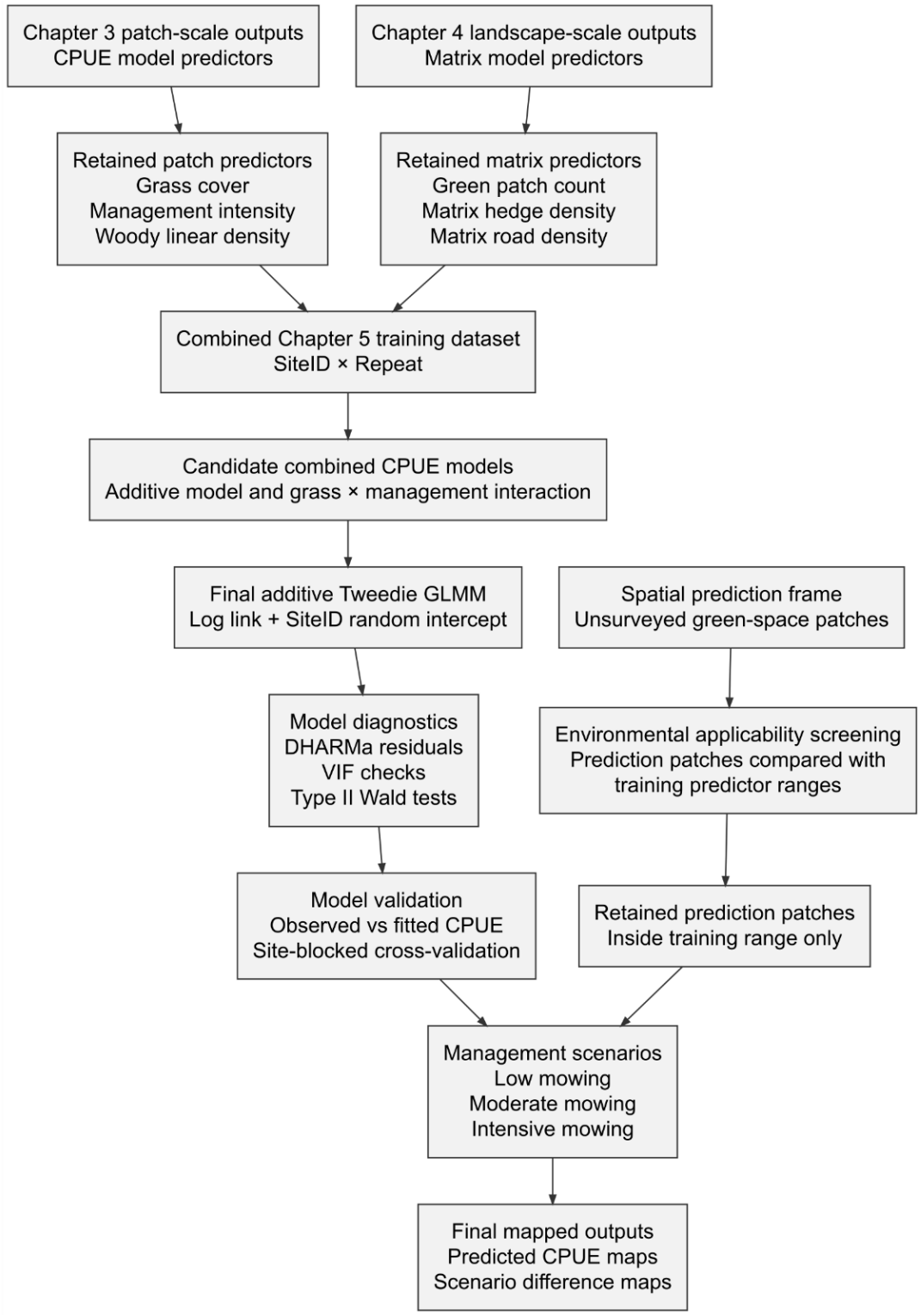


Figure 5.1 Predictive modelling workflow for Chapter 5. Patch-scale predictors from Chapter 3 and landscape-scale predictors from Chapter 4 were combined into a final CPUE model, validated using observed-versus-fitted values and site-blocked cross-validation, screened for environmental applicability, and then applied to unsurveyed green space patches under low, moderate and intensive mowing scenarios.

5.2.2 Assembly of the combined modelling dataset

All statistical analyses were undertaken in R (v.4.6.0). The combined modelling dataset was assembled by joining the Chapter 3 Site ID × Repeat CPUE dataset with the Chapter 4 matrix predictor dataset. This produced a single model-ready dataset in which CPUE varied among repeat surveys, while patch-scale and matrix-scale predictors were represented at site level.

Patch-scale predictors were carried forward from the Chapter 3 community CPUE analysis. Grass cover, management intensity and within-patch woody linear density were retained because each was associated with community CPUE and could also be derived consistently across the unsurveyed green space prediction frame. Within-patch path density was also negatively associated with CPUE in Chapter 3 but was not carried forward because an equivalent spatially complete path dataset was not available across the unsurveyed green spaces. Patch area was considered during development of the combined model but was not retained in the final predictor set. Tree cover was not carried forward because it was not significantly associated with community CPUE in Chapter 3. Within-patch watercourse density was positively associated with community CPUE in Chapter 3, but was not retained in the final Chapter 5 predictor set. The patch-scale component carried forward into Chapter 5 therefore comprised grass cover, management intensity and within-patch woody linear density. These represented habitat composition, the grassland-management gradient and internal woody linear vegetation structure, respectively.

Matrix-scale predictors were selected from the 500 m models presented in Chapter 4. At 250 m, green patch count and matrix road density were negatively associated with CPUE. Both relationships remained significant at 500 m, where nearest green patch

distance was additionally negatively associated with CPUE and matrix hedge density was positively associated with CPUE. The 500 m extent therefore retained the CPUE relationships identified at 250 m while also identifying additional associations with the surrounding matrix. Green patch count, nearest green patch distance, matrix hedge density and matrix road density were considered during development of the Chapter 5 combined model.

The use of the 500 m matrix extent does not identify an optimum biological scale or a movement distance for small mammals. As outlined in Chapter 4, landscape relationships can change according to the spatial extent over which environmental variables are measured, and no single extent is expected to represent every ecological response (Jackson & Fahrig, 2012; Miguet et al., 2016). Small mammal studies examining multiple surrounding extents have also reported variation in the spatial extent at which landscape relationships are detected (Fischer et al., 2011; Dorigo et al., 2021). The 500 m matrix was therefore used as the broader predefined extent that retained the full set of Chapter 4 CPUE associations considered for integration in Chapter 5.

As confirmed in Section 5.1, species richness, Simpson's reciprocal diversity and individual-species responses were not carried forward into the combined predictive modelling stage. Chapter 5 was developed specifically around total CPUE as a community-level capture response, while richness, Simpson's reciprocal diversity and focal-species responses remained evaluated within Chapters 3 and 4. The response variable was calculated at the Site ID × Repeat level using the method described in Section 3.2.5.1 and remained unchanged for Chapter 5.

The predictive modelling framework combined the surveyed green space patches used in Chapters 3 and 4 with the wider prediction frame of unsurveyed urban green spaces (Figure 5.2). The surveyed patches formed the model training dataset, while the remaining green space polygons represented candidate locations for spatial prediction. Predictor variables were derived across the prediction frame using the corresponding GIS methods described in Chapters 3 and 4 (Sections 3.2.3 and 4.2.3), allowing the same environmental variables used during model fitting to be calculated for unsurveyed green spaces.

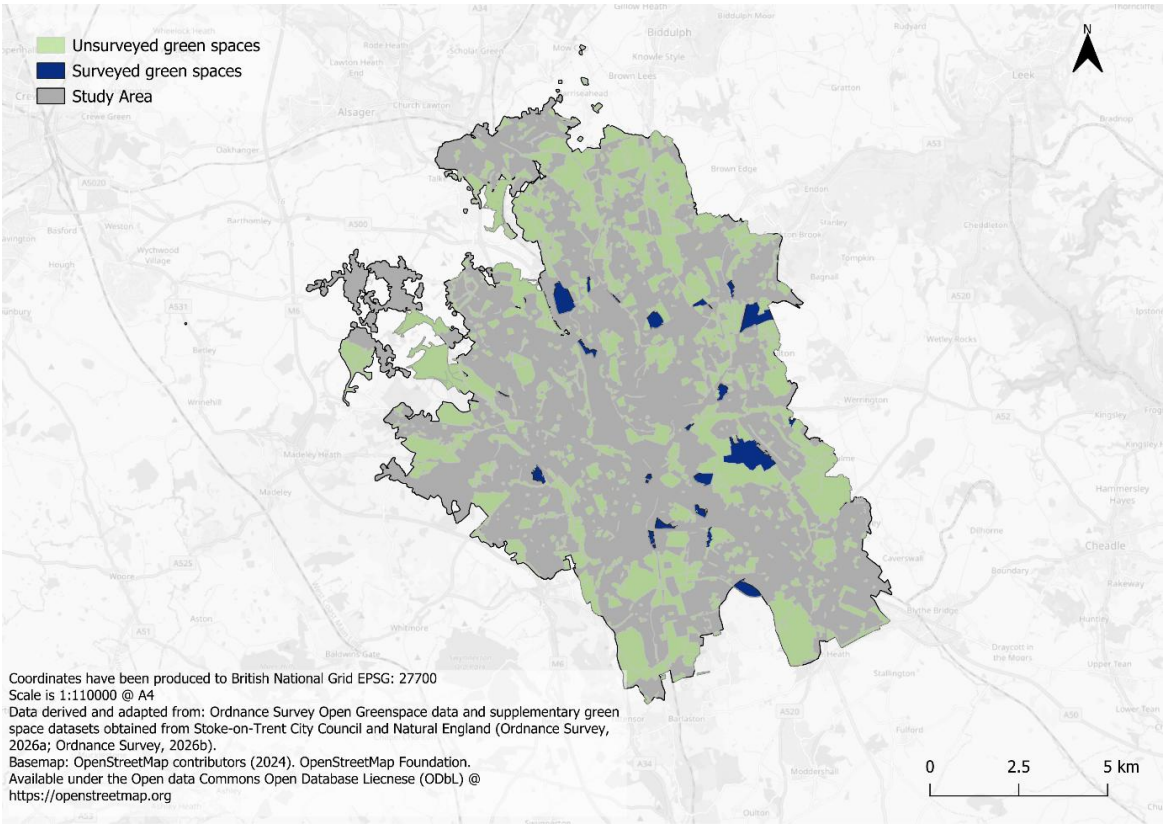


Figure 5.2 Spatial prediction framework showing the surveyed green space patches used to train the predictive model (blue) and the unsurveyed green space patches comprising the prediction frame (green) across the Stoke-on-Trent and Newcastle-under-Lyme study area. Predictor variables were derived for both surveyed and unsurveyed patches using identical GIS workflows prior to model fitting and environmental applicability screening.

Following the predictor-selection process described above, grass cover, management intensity, within-patch woody linear density, green patch count, matrix hedge density

and matrix road density were carried forward into the combined modelling stage. Nearest green patch distance was considered as part of the Chapter 4 candidate set but was not retained for the combined additive model. The variables carried forward therefore represented patch-scale habitat composition, grassland management and woody linear structure, together with green patch count and the density of hedgerows and roads within the 500 m matrix.

Continuous predictors were standardised using z-score transformation before model fitting. Standardisation was undertaken by subtracting the mean of the training dataset and dividing by the training standard deviation (Schielzeth, 2010; Cade, 2015). This allowed coefficient estimates to be compared across predictors measured on different scales, including proportional cover, line densities and patch counts. The same scaling values were saved and subsequently applied to the unsurveyed prediction dataset to ensure that spatial predictions were made on the same scale as the fitted model.

Before modelling, missing and non-finite values were checked for the response and all candidate predictors. No missing or non-finite values were present in the final training dataset. Collinearity was assessed using variance inflation factors with the performance package (Lüdecke et al., 2021). All final predictors had VIF values below 2.5, indicating low-to-moderate collinearity and supporting retention of the combined predictor set.

5.2.3 Final combined CPUE model

The final CPUE model was fitted using a generalised linear mixed model in glmmTMB (Brooks et al., 2017). A Tweedie error distribution with a log link was used because

CPUE was continuous, zero-inflated, non-negative and right-skewed, consistent with the modelling approach used for CPUE in Chapters 3 and 4 where zero-inflation occurred (Dunn & Smyth, 2005). Site ID was included as a random intercept to account for repeated survey observations from the same green space patches.

The starting model included the additive effects of management intensity, grass cover, within-patch woody linear density, green patch count, matrix hedge density and matrix road density. A grass cover × management intensity interaction was also tested because the thesis hypothesis predicted that the effect of grassland cover could depend on mowing intensity. The interaction model was compared with the additive model using AIC, fixed effect estimates and Type II Wald chi-square tests. The interaction model was not retained because the interaction did not improve model performance and the interaction term was not significant. The final model therefore retained grass cover and management intensity as separate additive predictors.

The final model structure was:

$$CPUEtotal \sim Management\ intensity\ +\ Grass\ cover\ +\ Within-patch\ woody\ linear\ density\ +\ Green\ patch\ count\ +\ Matrix\ hedge\ density\ +\ Matrix\ road\ density\ +\ (1\mid SiteID)$$

Fixed effects were assessed using Type II Wald chi-square tests with the car package (Fox & Weisberg, 2019). Model coefficients, standard errors and confidence intervals were extracted using broom.mixed (Bolker and Robinson, 2022). Simulated residual diagnostics were generated using DHARMa to assess residual uniformity, dispersion,

outliers and zero-inflation (Hartig, 2022). Model visualisations were produced using ggplot2, ggeffects and patchwork (Wickham, 2016; Lüdecke, 2018).

Model performance was assessed in two stages. First, fitted CPUE values were generated for the surveyed training data using fixed effects only, with the random site-level intercept excluded. This provided an observed-versus-fitted assessment of how well the fixed-effect component of the model reproduced the observed CPUE patterns. Model fit was calculated using Pearson's correlation, Spearman's rank correlation, root mean square error (RMSE) and mean absolute error (MAE) (Zuur et al., 2010; Dormann et al., 2013).

Second, predictive performance at withheld sites was assessed using leave-one-site-out, site-blocked cross-validation (Roberts et al., 2017; Valavi et al., 2019). Each surveyed site was withheld in turn, the model was re-fitted using the remaining 20 sites, and CPUE was predicted for the withheld site. This was repeated for all 21 sites and provided a stricter test of predictive performance than the observed-versus-fitted assessment because predictions were made for sites excluded from model fitting (Roberts et al., 2017). Within each cross-validation fold, predictor scaling was recalculated using only the training sites and then applied to the withheld site, preventing information from the test site being used during model fitting (Mahoney et al., 2023).

Cross-validation performance was summarised using the same metrics as the observed-versus-fitted assessment: Pearson's correlation, Spearman's rank correlation, RMSE and MAE (Roberts et al., 2017; Valavi et al., 2019; Hui et al., 2024). These validation steps were used to assess whether the combined model retained

sufficient predictive performance to support bounded spatial prediction across environmentally comparable green space patches within the study area.

5.2.4 Spatial prediction framework

Spatial predictions were generated for the unified urban green space prediction frame developed from the study-area GIS framework using the sf package (Pebesma, 2018). This prediction frame contained 1,259 unsurveyed green space polygons within the Stoke-on-Trent and Newcastle-under-Lyme study area. Each polygon represented a candidate green space patch to which the final CPUE model could potentially be applied.

Predictor variables for the prediction frame were derived using the same spatial extraction methods described in Chapters 3.2 and 4.2. Patch-scale variables were calculated within each green space polygon, while matrix-scale variables were calculated from the surrounding landscape using the same GIS-derived predictor definitions used for the surveyed sites. This ensured that predictor values in the prediction frame were directly comparable with those used to train the model. The prediction frame was then read into R (v.4.6.0) as a spatial sf object, transformed to British National Grid (EPSG:27700), checked for required predictor fields and assigned a unique patch identifier. The final model was then applied to the prediction frame using fixed effects only. Random effects were excluded during prediction because unsurveyed patches had no observed site-level random intercepts. Predictions were therefore based on the fixed-effect relationships estimated from the surveyed training data.

Before spatial prediction, the environmental range represented by the surveyed training data was compared with the corresponding predictor values across the 1,259 unsurveyed green space patches. Screening was undertaken using the retained predictors in the final combined model: grass cover, management intensity, within-patch woody linear density, green patch count, matrix hedge density and matrix road density. An unsurveyed patch was classified as inside the environmental training range only where all retained predictor values fell within the ranges represented by the surveyed training sites. Patches with one or more predictor values outside these ranges were classified as outside the supported prediction domain and were not retained for final mapped prediction. This screening identified 598 environmentally comparable patches, while 661 patches fell outside the training range (Appendix G, Table G.5).

Spatial predictions were generated under three grassland management scenarios: low mowing, moderate mowing and intensive mowing. These corresponded to the three management categories used throughout the thesis. For each scenario, the management intensity variable was fixed to the relevant category value across all retained prediction patches, while all other patch-scale and matrix-scale predictor values were left unchanged. This allowed predicted CPUE to be compared under alternative management assumptions while preserving the underlying spatial variation in habitat structure and landscape context among patches. Predicted CPUE was generated on the link scale and then back-transformed to the response scale. Approximate 95% confidence intervals were calculated from the prediction standard errors on the link scale and then back-transformed to CPUE units (McCullagh & Nelder, 1989; Wood, 2017).

Predictions were generated for all three scenarios and then filtered to retain only patches classified as inside the environmental training range. Summary statistics were calculated for each scenario, including minimum, mean, median and maximum predicted CPUE, mean confidence interval width and the number of patches exceeding the map display cap.

5.2.5 Spatial mapping and scenario comparison

Prediction maps were produced in R (v.4.6.0) using ggplot2, sf, ggspatial and patchwork (Wickham, 2016; Lüdecke, 2018). Only patches falling inside the environmental training range were mapped. Predicted CPUE was displayed using a common colour scale across all management scenarios to allow direct visual comparison. For map readability, predicted CPUE values were capped at 30 for visualisation only; uncapped predicted values were retained in the exported spatial and tabular outputs.

Difference maps were produced to compare each alternative management scenario against the moderate mowing scenario. Two difference layers were calculated: low mowing minus moderate mowing, and intensive mowing minus moderate mowing. Positive values therefore indicated higher predicted CPUE relative to moderate mowing, while negative values indicated lower predicted CPUE.

Final spatial outputs were exported as GeoPackage and CSV files. Figures were exported as 300 dpi PNG files for thesis production, with TIFF export available but disabled by default to reduce storage use during script development.

5.3 Results

5.3.1 Final combined CPUE model

The final model combined predictors representing within-patch habitat structure and management with characteristics of the surrounding landscape (Figure 5.3). Within-patch woody linear density was the strongest positive predictor of CPUE (β = 0.332 ± 0.068 SE; χ^2 = 23.58, p < 0.001), while management intensity was significantly negatively associated with CPUE (β = −0.296 ± 0.088 SE; χ^2 = 11.42, p < 0.001). Grass cover was not significantly associated with CPUE after the other predictors were included (β = 0.045 ± 0.097 SE; χ^2 = 0.22, p = 0.641).

Matrix-scale predictors also remained associated with CPUE after patch-scale predictors were included. Green patch count (β = −0.226 ± 0.080 SE; χ^2 = 8.03, p = 0.005) and matrix road density (β = −0.200 ± 0.101 SE; χ^2 = 3.95, p = 0.047) were negatively associated with CPUE. Matrix hedge density retained a positive but non-significant association (β = 0.127 ± 0.067 SE; χ^2 = 3.61, p = 0.057). The combined model therefore indicated higher CPUE in less intensively managed patches with greater within-patch woody linear density, lower green patch count and lower matrix road density.

The final combined model retained significant associations with predictors originating from both the Chapter 3 patch-scale analysis and the Chapter 4 matrix-scale analysis. Within-patch woody linear density and management intensity showed the largest standardised coefficient estimates, while green patch count and matrix road density also remained significantly associated with CPUE (Figure 5.3).

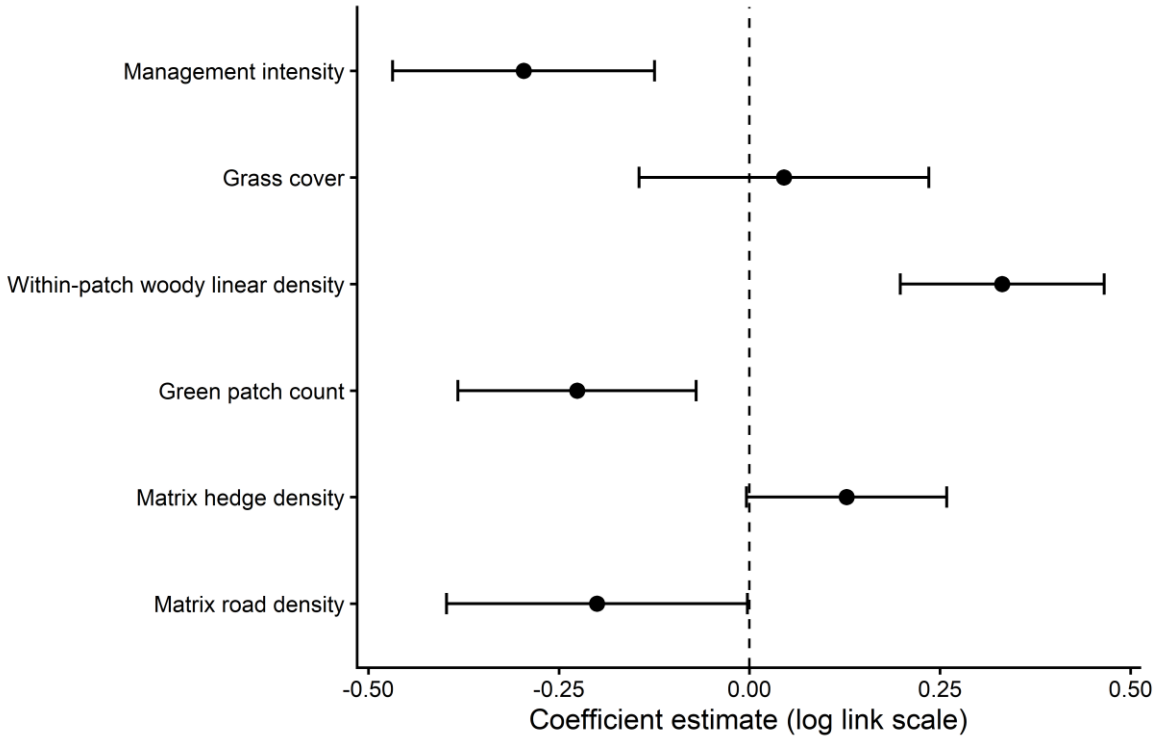


Figure 5.3 Coefficient estimates from the final combined CPUE model showing associations between patch-scale and matrix-scale predictors and total small mammal CPUE. Predictors comprise management intensity, grass cover, within-patch woody linear density, green patch count, matrix hedge density and matrix road density. Points show fixed-effect coefficient estimates on the log-link scale and horizontal bars show 95% confidence intervals. Positive estimates indicate associations with higher CPUE and negative estimates indicate associations with lower CPUE. The dashed vertical line indicates a coefficient estimate of zero.

5.3.2 Predicted responses across retained drivers

The partial-response plots showed predicted CPUE relationships for the four significant predictors retained in the final combined model, with the remaining predictors held constant (Figure 5.4). Predicted CPUE increased with within-patch woody linear density and decreased with increasing management intensity, green patch count and matrix road density. Within-patch woody linear density and management intensity showed the greatest changes in predicted CPUE across the plotted standardised ranges, consistent with their larger absolute coefficient estimates in the final model (Figure 5.3). Green patch count and matrix road density

showed smaller negative changes in predicted CPUE across their plotted standardised ranges.

Matrix hedge density was not included in Figure 5.4 because its relationship with CPUE was non-significant in the final combined model (p = 0.057).

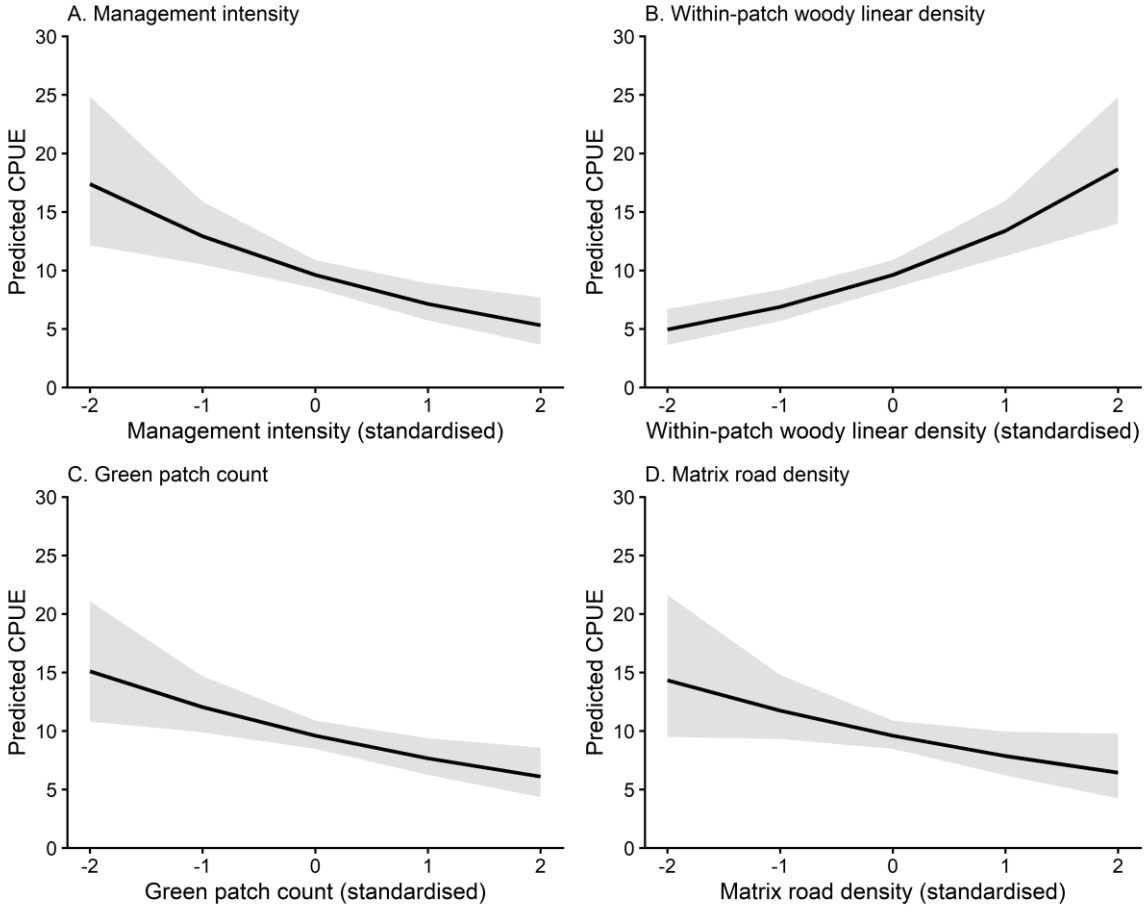


Figure 5.4 Model-predicted CPUE relationships for the significant predictors retained in the final combined CPUE model: (A) standardised management intensity, (B) standardised within-patch woody linear density, (C) standardised green patch count and (D) standardised matrix road density. Black lines show predicted mean CPUE and grey ribbons show 95% confidence intervals, with the remaining model predictors held constant. All panels use the same y-axis scale to facilitate comparison of predicted CPUE across the four relationships.

The management scenarios showed a progressive decline in predicted CPUE from low to moderate to intensive mowing (Figure 5.5). With the remaining model predictors held constant, predicted CPUE decreased from approximately 14 under

low mowing to approximately 7 under intensive mowing. This comparison shows the model-predicted change in CPUE across the three management categories before the scenarios were applied across the wider urban green space prediction frame.

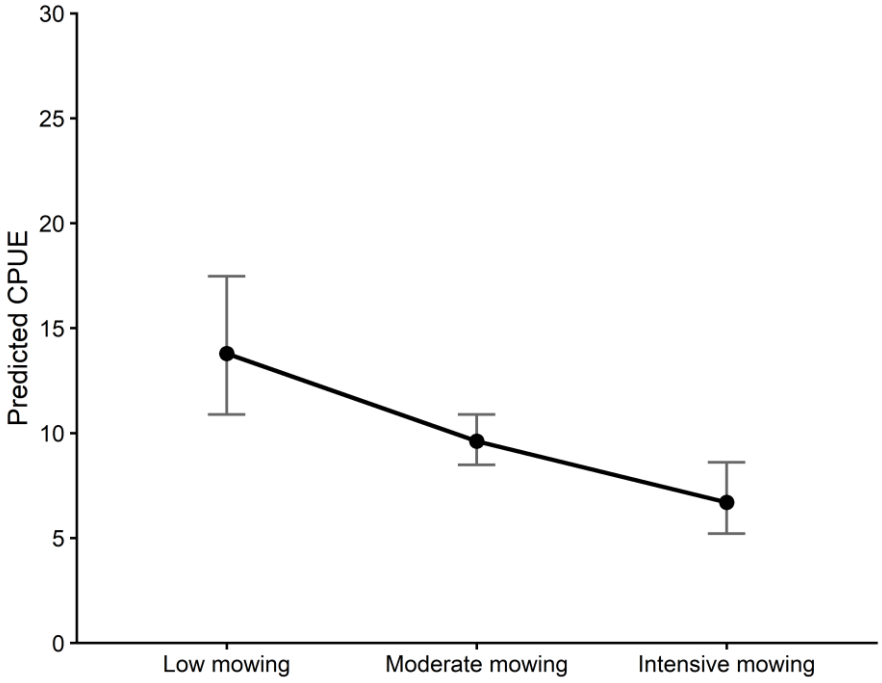


Figure 5.5 Model-predicted CPUE under low, moderate and intensive mowing scenarios, with the remaining predictors in the final combined CPUE model held constant. Points show predicted mean CPUE and vertical bars show 95% confidence intervals.

5.3.3 Model validation for untrained spatial prediction

Model performance was assessed using observed-versus-fitted comparisons and leave-one-site-out, site-blocked cross-validation (Figure 5.6). Observed and fitted CPUE values were positively correlated (Figure 5.6A), with a Pearson correlation of 0.850 and a Spearman rank correlation of 0.848. The model produced an RMSE of 4.63 and an MAE of 3.45. Fitted CPUE generally increased with observed CPUE, although scatter increased at higher observed values and several of the highest observed CPUE values were under-predicted.

Predictive performance at withheld sites was assessed using leave-one-site-out cross-validation, whereby each surveyed site was sequentially excluded from model fitting and predicted using a model fitted to the remaining sites (Figure 5.6B). Correlations between observed and cross-validated predicted CPUE were lower than those obtained from the fitted model, with a Pearson correlation of 0.719 and a Spearman correlation of 0.771. Prediction error also increased, with an RMSE of 6.31 and an MAE of 4.60. Cross-validated predictions therefore retained a positive relationship with observed CPUE, with greater prediction error than was recorded from the fitted model.

These validation results were used alongside the environmental-range screening described in Section 5.2.4 to determine the extent of subsequent spatial prediction across unsurveyed green space patches.

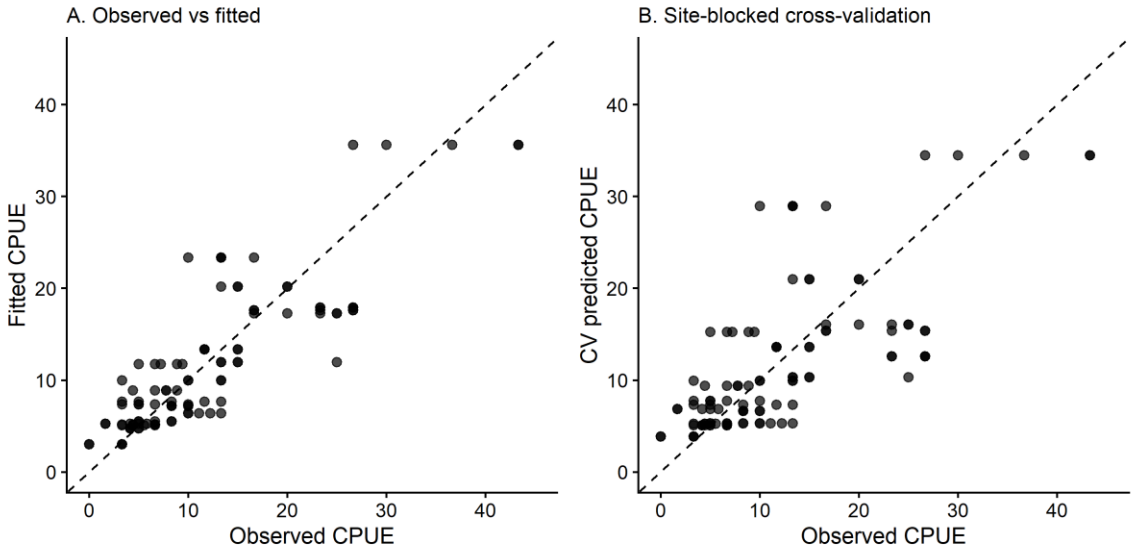


Figure 5.6 Validation of the final combined CPUE model. (A) Relationship between observed and fitted CPUE values from the final model. (B) Leave-one-site-out, site-blocked cross-validation showing observed CPUE against predicted CPUE for survey observations from withheld sites. Points represent individual Site ID × Repeat observations and dashed lines show the one-to-one relationship between observed and predicted CPUE.

5.3.4 Spatial prediction domain

Before generating spatial predictions, the environmental range of the unsurveyed prediction dataset was compared with that represented by the surveyed training data. Of the 1,259 unsurveyed green space patches assessed, 598 fell within the training range for the retained predictors and 661 fell outside the training range (Figure 5.7). Final mapped predictions were restricted to the 598 patches within the training range. The remaining 661 patches were excluded from subsequent mapped prediction outputs.

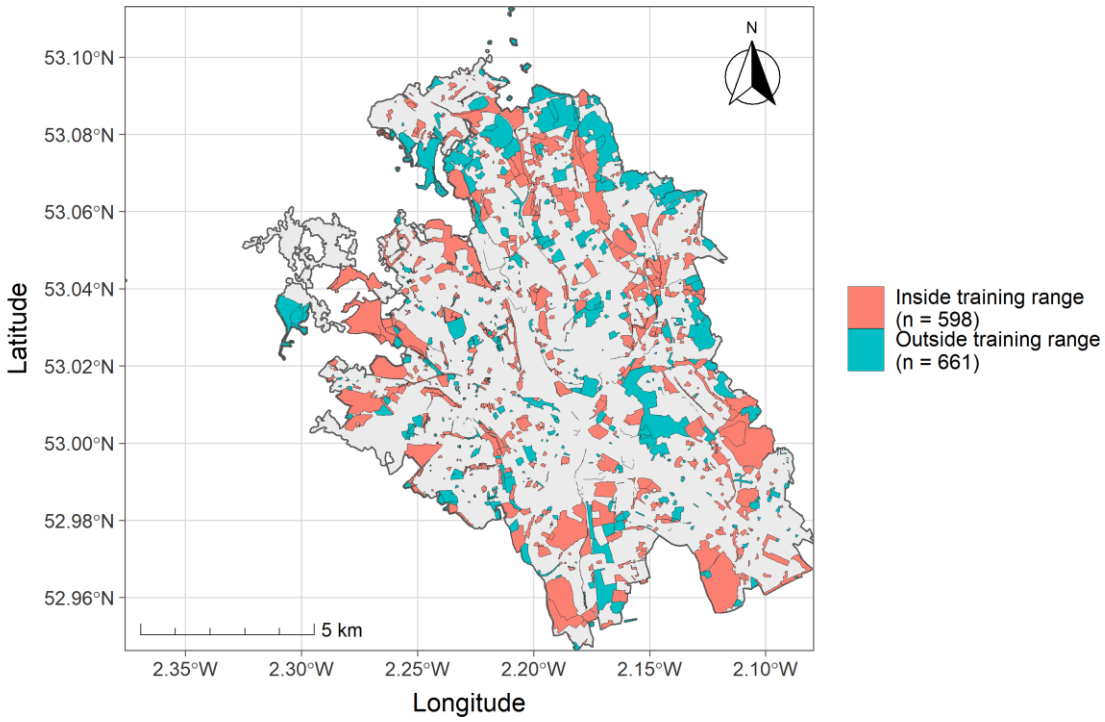


Figure 5.7 Environmental screening of the unsurveyed green space prediction dataset. Patches were classified according to whether retained predictor values fell within the range represented by the surveyed training data. Of the 1,259 unsurveyed patches assessed, 598 were within the training range and were retained for mapped spatial prediction; 661 were outside the training range and were excluded from mapped prediction outputs.

5.3.5 Predicted CPUE across management scenarios

Predicted CPUE declined consistently with increasing management intensity across the 598 environmentally comparable unsurveyed patches (Table 5.1; Figure 5.8). Mean predicted CPUE was highest under low mowing (mean = 9.83; median = 9.34), lower under moderate mowing (mean = 6.85; median = 6.51), and lowest under intensive mowing (mean = 4.77; median = 4.53). Maximum predicted CPUE also declined across the management gradient, from 27.89 under low mowing to 19.43 under moderate mowing and 13.54 under intensive mowing.

Higher predicted CPUE values were more widespread under the low mowing scenario (Figure 5.8a). Predicted values were lower across the retained patches under moderate mowing (Figure 5.8b) and lower again under intensive mowing (Figure 5.8c). No retained patches exceeded the CPUE map cap of 30 under any management scenario, so no predicted values were truncated for display.

Table 5.1 Summary of model-predicted CPUE across the 598 environmentally comparable unsurveyed green space patches retained for spatial prediction under low, moderate and intensive mowing scenarios.

Scenario	N Patches	Min CPUE	Mean CPUE	Median CPUE	Max CPUE	Mean CI Width
Low mowing	598	5.45	9.83	9.34	27.89	9.22
Moderate mowing	598	3.80	6.85	6.51	19.43	5.24
Intensive mowing	598	2.65	4.77	4.53	13.54	3.90

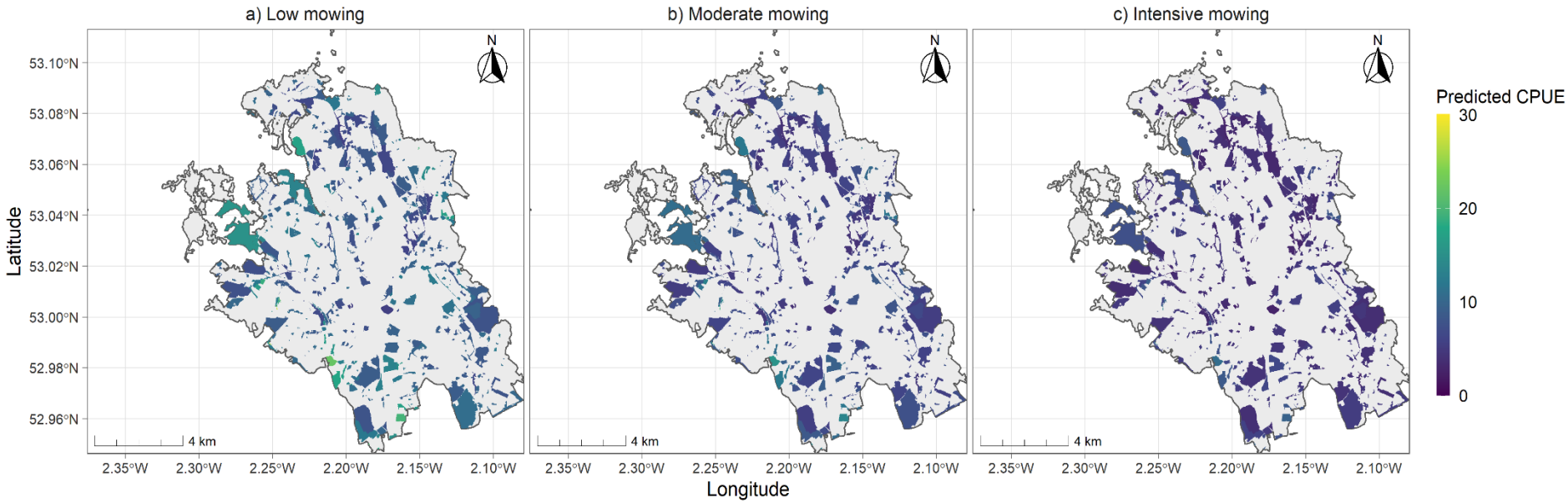


Figure 5.8 Model-predicted community CPUE across the 598 environmentally comparable unsurveyed green space patches under (a) low mowing, (b) moderate mowing and (c) intensive mowing scenarios. Predictions are shown only for patches retained following environmental screening against the surveyed training data. All panels use a common CPUE scale from 0 to 30.

5.3.6 Difference between management scenarios

Difference mapping showed that predicted CPUE was higher under low mowing than under moderate mowing across the retained prediction frame (Figure 5.9a). Predicted CPUE was lower under intensive mowing than under moderate mowing across the same patches (Figure 5.9b). The direction of change was spatially consistent across the retained patches, although the magnitude of the difference varied among patches.

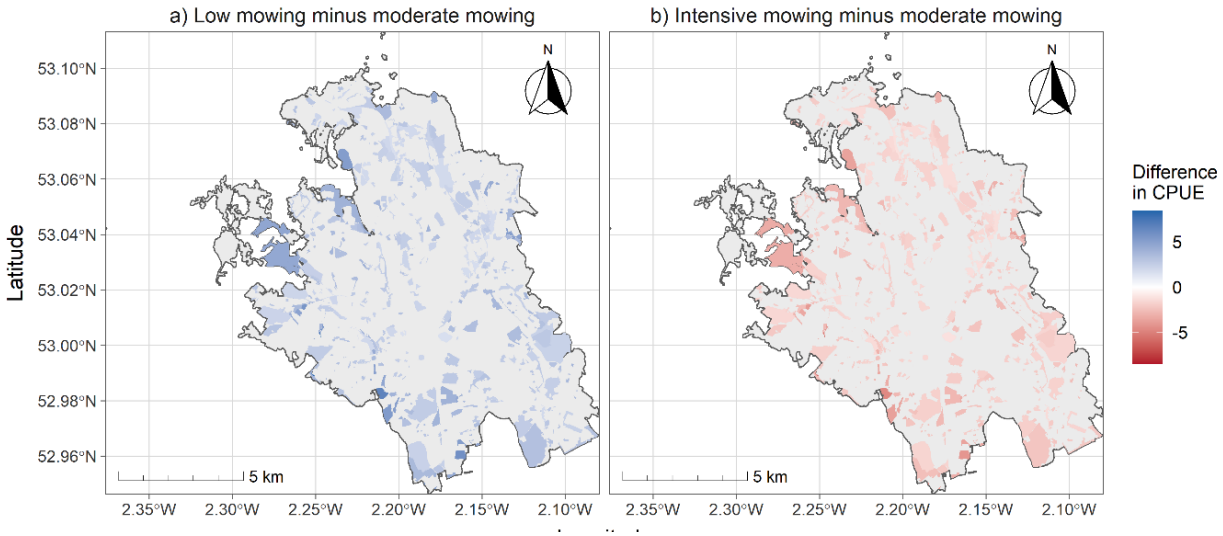


Figure 5.9 Difference in model-predicted community CPUE between management scenarios across the 598 environmentally comparable unsurveyed green space patches: (a) low mowing minus moderate mowing and (b) intensive mowing minus moderate mowing. Positive values indicate higher predicted CPUE relative to the moderate mowing scenario and negative values indicate lower predicted CPUE.

5.4 Discussion

Chapter 5 integrated the patch-scale relationships identified in Chapter 3 with the matrix-scale relationships identified in Chapter 4, before testing the resulting model at withheld sites and across unsurveyed urban green spaces. The combined analysis therefore changed both the ordering and statistical support of relationships identified in the separate patch and matrix-scale models.

The model also retained predictive ability when complete survey sites were excluded from model fitting, although prediction error increased relative to the fitted model (Figure 5.6). These results address the four questions established in the Chapter 5 Introduction: whether patch and matrix predictors retain complementary associations when combined, how well the model predicts at withheld sites, where spatial prediction is environmentally supported, and how predicted CPUE changes under the three management scenarios.

5.4.1 Combining patch and matrix predictors changed the hierarchy of relationships with CPUE

The first Chapter 5 question asked whether patch-scale habitat and management predictors and matrix-scale predictors retained complementary associations with community CPUE when considered together, while H4 predicted that patch-scale habitat structure, grassland management and landscape configuration would retain complementary relationships within the combined model. This was supported, but the relationships identified separately in Chapters 3 and 4 were not retained equally. Within-patch woody linear density remained the strongest positive predictor (β = 0.332, $p < 0.001$) and management intensity remained strongly negative (β = -0.296, $p < 0.001$), with both changing only modestly from their Chapter 3 coefficients (β = 0.372 and -0.278, respectively). Green patch count and road density also remained significantly negative, although their coefficients declined from β = -0.495 and -0.259 at 500 m in Chapter 4 to β = -0.226 and -0.200 in the combined model. In contrast, grass cover changed from a strong positive relationship in Chapter 3 (β = 0.404, $p < 0.001$) to no significant relationship (β = 0.045, $p = 0.641$), while matrix hedge density declined from β = 0.322 in Chapter 4 to β = 0.127 and also fell below statistical

significance (p = 0.057) (Figure 5.3). These differences are descriptive because the models contained different predictor sets, but they show that integration narrowed the hierarchy identified by the separate analyses rather than reproducing it unchanged.

Within-patch woody linear density and management intensity retained very similar coefficients after the matrix variables were incorporated, indicating that these local characteristics remained closely associated with differences in community CPUE even when the surrounding landscape was represented. Ecologically, this places the structure and management of habitat available within the occupied green space above broad grassland extent in the combined hierarchy. As discussed in Chapter 3, small mammals respond strongly to the structure of vegetation because dense ground cover, woody margins and structurally heterogeneous vegetation provide refuge and foraging habitat that broad habitat-area measurements do not necessarily describe (Dickman & Doncaster, 1987; Churchfield et al., 1997; Tattersall et al., 2002; Atkins et al., 2018).

This agrees with studies in which local and landscape variables were both associated with measures of small mammal abundance. Mérő et al. (2015) found differences in total small mammal abundance among grassland management regimes, while landscape composition also contributed to abundance, with higher values in landscapes containing more restored and natural grassland and lower values where roads and canals were more extensive. Scott et al. (2008), working in coastal wetlands in western Estonia, calculated community relative abundance as CPUE and found that it differed among trapping-habitat types and was correlated with habitat

composition within 50-200 m buffers surrounding the trap lines. Fischer et al. (2011) found that small mammal abundance was associated with field management and landscape complexity in an agricultural system, with the relationships detected differing across the spatial extents examined. These studies support the ecological interpretation of the Chapter 5 hierarchy: local habitat conditions can show the strongest relationships with community CPUE while landscape structure retains additional relationships with the same community response. In this study, patch structure and management showed stronger associations with CPUE than the matrix predictors, although green patch count and road density explained additional variation.

The persistence of green patch count and road density gives the landscape component a clear role in that hierarchy. Chapter 4 interpreted higher green patch count as greater subdivision of surrounding green space within this study area, while road density represented the concentration of transport infrastructure within the matrix. Both remained negatively associated with CPUE after local habitat structure and management were included. This means that green spaces with favourable internal conditions could still occur in landscape settings associated with lower community CPUE (Fischer et al., 2011; Mérő et al., 2015).

The smaller coefficients for green patch count and road density in Chapter 5 do not remove their ecological relevance. They show that the strongest relationships with CPUE occurred inside the focal patch, while subdivision and road density continued to describe landscape conditions associated with lower community CPUE after those local conditions were represented. They do, however, indicate that characteristics

associated with fragmentation of the surrounding landscape retained greater statistical support than matrix hedge density once the predictors were considered together, as the matrix hedge relationship was no longer significant. The Chapter 5 result therefore indicates that, within highly modified systems, the presence of connective habitat features alone may not offset the wider landscape conditions associated with habitat subdivision and transport infrastructure.

Matrix hedge density contrasted with within-patch woody linear density because it no longer retained significant statistical support in the combined model. Matrix hedge density represented mapped woody linear structure across the 500 m landscape, but the UKCEH layer was neither a complete inventory nor a measure of hedge condition or basal vegetation structure (Broughton et al., 2024). The within-patch woody variable was based on mapped hedgerows, treelines and scrub margins within the surveyed green spaces and therefore represented structural vegetation much closer to where the community was sampled. The Chapter 5 hierarchy therefore gives stronger support to woody structure within the focal patch than to the amount of mapped hedge across the surrounding matrix. It does not show that matrix hedges lacked ecological value; it shows that their relationship with community CPUE was weaker once local woody structure, management, green space subdivision and road density were considered together, strengthening the argument that the benefits associated with connective linear features may be constrained by the wider fragmented landscape in which they occur (Pardini et al., 2005; Mortelliti et al., 2011; Galantinho et al., 2020).

Grass cover also lost statistical support in the combined model. Its coefficient also declined from β = 0.404 in Chapter 3 to β = 0.045 in Chapter 5. This suggests that how grass-dominated patches were structured and managed was more closely associated with community CPUE than the proportion of the patch mapped as grassland once the wider environmental context was represented. The combined model refines the earlier interpretation: grassland amount was associated with CPUE when patch characteristics were considered alone, but once management intensity, woody structure, and the surrounding matrix were accounted for, grass cover itself contributed little to the remaining relationship. However, this does not make grassland ecologically unimportant; it indicates that broad grassland extent was a poorer discriminator of community CPUE than the structural and management characteristics that determine what that grassland actually provides (Garratt et al., 2012; Mérő et al., 2015).

Within the urban literature, however, the relative ordering of local and landscape relationships is inconsistent. As discussed in Chapter 3, Dondina et al. (2025) found local habitat characteristics to be more influential than the spatial variables examined across urban green areas, with non-synanthropic small mammals associated particularly with shrub-rich woodland while green-area size and distance from the city centre showed no consistent relationships. This agrees with the Chapter 5 ordering insofar as the two largest standardised coefficients were associated with conditions within the focal green space. It does not explain the whole result, because green patch count and road density remained significant after woody structure and management were included. On the other hand, the Chapter 4 discussion identified stronger matrix relationships in the urban study of Gomes et al. (2011), where small

mammal responses varied with surrounding pervious and impervious land cover and *A. sylvaticus* declined with increasing impervious cover while responding positively to pervious habitat. Chapter 5, therefore, falls between these urban examples: local structure and management retained the strongest statistical support, but characteristics of the surrounding matrix continued to explain variation in CPUE. Such variation among urban studies is expected where green spaces differ in vegetation structure, management, surrounding land cover and configuration, and where studies quantify different components of the patch and matrix (Beninde et al., 2015; Lepczyk et al., 2017; Liordos et al., 2021; Dondina et al., 2025).

The first Chapter 5 question and RQ3 were therefore answered positively because significant patch- and matrix-scale relationships remained in the integrated model. H4 was supported at the broader patch–matrix level and for the predicted directions of woody linear density, management intensity, green patch count and road density, but the specific prediction of a retained positive matrix hedge relationship was not supported.

5.4.2 Management scenarios provide a spatial application of the combined model

The management scenarios address the final step in the analytical framework established in Chapter 1: whether relationships identified from surveyed green spaces can be translated into spatially explicit comparisons across the wider urban network. H4 predicted that community CPUE would decline from low through moderate to intensive mowing, and this prediction was met across the 598 environmentally supported patches. Mean predicted CPUE declined from 9.83 under low mowing to 6.85 under moderate mowing and 4.77 under intensive mowing. The

management coefficient retained in the combined model was strongly negative (β = -0.296, $p < 0.001$), so the scenario pattern follows directly from a relationship that was first identified at the patch scale in Chapter 3 and remained after landscape structure was incorporated in Chapter 5. The progression across the thesis is therefore from identifying a management relationship within 21 surveyed green spaces, through testing whether that relationship persisted alongside matrix predictors, to examining its predicted consequences across a much larger set of unsurveyed green spaces. This fulfils the spatial component of Objective 4 and addresses Research Question 4, which asked whether the relationships identified through the empirical chapters could support environmentally bounded spatial prediction and comparison of alternative grassland-management scenarios.

The purpose of applying ecological models spatially is that planning decisions extend well beyond the locations at which biological surveys can be undertaken (Guisan et al., 2013). Chapter 1 identified this explicitly: GIS allows relationships derived from field observations to be examined across unsurveyed green spaces, provided that predictions remain within environmental conditions represented by the training data (Guisan & Zimmermann, 2000; Elith & Leathwick, 2009; Araújo et al., 2019; Franklin, 2023). This approach has already been applied directly to urban mammal conservation. Nelli et al. (2022) modelled habitat suitability and connectivity for grassland water voles in Glasgow, Scotland, UK, altered environmental layers to represent planned urban development, and generated new spatial predictions showing where suitable habitat and connectivity were expected to be lost. Their outputs were incorporated into an interactive mapping tool intended for conservation managers and planners assessing development proposals. The response and

modelling framework differ from those used here, but the planning principle is directly relevant: ecological relationships become more useful for decision-making when their consequences can be located geographically and compared under alternative scenarios (Nelli et al., 2022). Scenario analysis is now widely used in urban green space planning for this purpose, although a recent review of 157 studies found considerable variation in how scenarios are constructed, evaluated and translated into planning decisions (Wang & Wang, 2026).

The Chapter 5 scenarios add urban grassland management to that spatial framework. The green space boundary, woody linear density, grass cover and surrounding matrix characteristics of each patch remained fixed while mowing intensity was varied. The maps therefore do not simply identify areas predicted to contain relatively high or low community CPUE under existing conditions; they allow the predicted outcome under alternative management states to be compared for the same green space. Spatially explicit comparisons of management alternatives can change the usefulness of an ecological model because management resources rarely allow every site to be treated identically. Schulz et al. (2024), for example, modelled changes in mowing frequency across more than 1.3 million agricultural grassland fields in Germany and showed that the predicted biodiversity consequences of management extensification were spatially heterogeneous. Targeting the 30% of fields with the greatest predicted biodiversity response produced substantially greater gains in plant richness than applying the same reduction in mowing to randomly selected fields. Their study concerned agricultural grasslands and plant richness, and estimated context-dependent treatment effects using a different modelling approach, but it demonstrates why mapped management scenarios can provide information that a

single overall management coefficient cannot: they allow alternative interventions to be evaluated geographically and can reveal where management action may warrant greater consideration (Schulz et al., 2024).

H4 contained two related predictions about management. Figure 5.8 shows that increasing management intensity shifted predicted CPUE downward across the retained prediction frame without removing the wider spatial pattern in the predictions. Patches towards the upper end of the predicted range under low mowing generally remained towards the upper end under moderate and intensive mowing, while patches with lower underlying predictions remained lower across all three scenarios. Figure 5.9 shows that the absolute difference produced by changing management was also spatially variable: low mowing increased predicted CPUE relative to moderate mowing by 1.65–8.45 CPUE among patches, with a mean increase of 2.98, while intensive mowing reduced predicted CPUE by 1.15–5.89 relative to moderate mowing, with a mean reduction of 2.08.

The second prediction within H4 was that the magnitude of management differences would vary among patches according to their structural condition and surrounding matrix. Absolute scenario differences did vary among patches as predicted, but the final model was additive and did not contain management × woody structure or management × landscape interactions. The strong negative management relationship therefore applies across the prediction frame; the model does not show that its fitted strength changed according to road density, green space subdivision or woody linear density. Because the model used a log link, the same management coefficient was applied to different underlying predicted values generated by the other patch and

matrix predictors (McCullagh & Nelder, 1989). Patches with higher predicted CPUE consequently showed larger changes in raw CPUE units, which explains why the larger absolute differences in Figure 5.9 occurred among patches with higher underlying predictions in Figure 5.8.

The spatial pattern gives the scenario analysis a direct planning application. Figure 5.8 identifies how predicted community CPUE is distributed across environmentally comparable green spaces under each mowing regime, while Figure 5.9 identifies where changing the management input produces the largest absolute differences in predicted CPUE. Management intensity is therefore both one of the strongest retained predictors in the combined model and one of the few modelled characteristics that can be altered directly through management of an individual green space. The maps do not imply that reducing mowing can remove landscape constraints associated with road density or green space subdivision; they show where an empirically derived change in management is predicted to produce larger or smaller numerical changes within the environmental setting of each patch. This has clear parallels with wider urban grassland planning. Horák et al. (2022), working across 40 urban grasslands, found biodiversity relationships with grassland management alongside surrounding urban land use and argued for combining grassland management with spatial planning of the wider urban landscape. Their proposed measures included lower-intensity or mosaic mowing alongside protection of existing grasslands and management of development in their surroundings. Rada et al. (2024) found that retaining unmown portions within urban lawns increased pollinator richness and abundance and recommended mosaic management at the city scale. The Chapter 5 scenarios extend that planning logic to community CPUE by

locating where low, moderate and intensive mowing produce different predicted outcomes across a heterogeneous urban green space network for an underrepresented taxonomic group, while retaining the patch and landscape characteristics identified across Chapters 3 and 4.

RQ4 can therefore be answered positively within the supported prediction domain: relationships derived from the surveyed sites were transferable to environmentally represented unsurveyed green spaces for comparison of alternative management scenarios, while the resulting predictions continued to reflect differences in local woody structure, green space subdivision, matrix hedge density and road density.

5.4.3 Site-blocked validation and environmental screening define the supported prediction domain

The fitted model reproduced the surveyed data more closely than it predicted observations from sites excluded from model fitting. Across the training data, observed and fitted CPUE were strongly correlated (Pearson's $r = 0.850$; Spearman's $r = 0.848$), with an RMSE of 4.63 and MAE of 3.45. Under leave-one-site-out cross-validation, Pearson's r declined to 0.719 and Spearman's r to 0.771, with RMSE increasing to 6.31 and MAE to 4.60. The reduction in performance is informative for the intended spatial application of the model because the validation observations came from green spaces that had not contributed any observations to the corresponding model fit. Each surveyed patch was sampled repeatedly, so separating individual observations at random could place repeats from the same site in both training and validation data and provide an optimistic estimate of prediction to a genuinely unobserved patch (Roberts et al., 2017; Valavi et al., 2019).

The leave-one-site-out approach used here was not a full spatial blocking design, but it addressed the most immediate dependence in this dataset by keeping all repeated observations from a withheld green space outside the training fold. The cross-validation results therefore provide stronger evidence for prediction to another patch within the sampled urban system than the observed-versus-fitted relationship alone.

The retained correlations also show that the model preserved substantial variation in CPUE among withheld sites, despite the increase in prediction error. This is relevant to the purpose of Chapter 5. The model was developed to identify broad spatial differences in predicted community CPUE across unsurveyed green spaces and to compare management scenarios, not to provide an exact estimate of captures at every individual patch. A Spearman correlation of 0.771 under site-blocked validation indicates that much of the relative ordering from lower to higher CPUE was retained when entire sites were withheld, whereas the larger RMSE and MAE indicate poorer reproduction of the absolute predicted values at those sites (Roberts et al., 2017; Araújo et al., 2019). The model can therefore support spatial comparison within the study system, but the mapped CPUE values retain appreciable prediction error and should not be interpreted as precise estimates of local abundance.

Site-blocked validation within Stoke-on-Trent and Newcastle-under-Lyme does not establish that the fitted relationships would transfer unchanged to another urban area. Wenger and Olden (2012), Sequeira et al. (2018) and Yates et al. (2018), discussed earlier in relation to model transferability, show that predictive performance can deteriorate when models are applied under environmental conditions, spatial structures or species-environment relationships that differ from

those represented during calibration. Chapter 5 therefore supports prediction to unsurveyed patches within the study system under defined environmental limits. Application to another city would therefore require comparable biological observations against which the transferred predictions could be independently evaluated (Wenger & Olden, 2012; Yates et al., 2018).

Environmental screening imposed a second boundary on the spatial predictions. Of the 1,259 unsurveyed green space patches assessed, 598, or 47.5%, contained retained predictor values within the ranges represented by the surveyed sites. The remaining 661 patches, or 52.5%, exceeded the training range for at least one retained predictor and were removed from the final mapped prediction frame. This substantially reduced the area over which the model was applied, but it also prevented more than half of the candidate patches from being treated as equally supported by a model calibrated from 21 surveyed green spaces.

As outlined earlier in Chapter 5, Guillera-Arroita et al. (2015), Yates et al. (2018) and Araújo et al. (2019) identify extrapolation into poorly represented environmental conditions as a major source of uncertainty in spatial ecological prediction. Relationships estimated across one environmental range can become unreliable when projected beyond it because the fitted model has no empirical observations demonstrating how the response behaves under those conditions. The screening used here addresses that problem at the level of the individual retained predictors.

Environmental range is also only one source of uncertainty in predicting community CPUE. Ecological responses vary through both space and time, and relationships estimated from a finite set of surveys cannot represent every environmental process

contributing to observations at an individual site (Wenger & Olden, 2012; Yates et al., 2018). This is relevant to small mammals because live-trapping studies record substantial spatial and temporal variation in population numbers, including differences among sites and marked seasonal fluctuations associated with reproduction and resource availability (Flowerdew et al., 2004). Community CPUE in this thesis therefore represents the animals captured during defined sampling periods, not a census of every individual occupying a green space or a fixed population value that would be expected to remain unchanged through time. Predicting an exact CPUE for an unsurveyed patch is consequently a more demanding task than identifying whether patches are expected to occupy relatively higher or lower positions across the urban network.

This also places an important limit on comparisons with ecological relationships reported from agricultural and other modified landscapes. Such studies remain useful for identifying mechanisms that may operate in urban systems, including responses to mowing, woody linear vegetation, roads and habitat fragmentation. They cannot, however, be assumed to reproduce the strength of those relationships within an urban green space network. Urban landscapes contain fine-scale combinations of habitat, infrastructure and management that differ from more spatially uniform land-use systems, and ecological model transferability can deteriorate where environmental structure or species-environment relationships change between the calibration and application settings (Wenger & Olden, 2012; Yates et al., 2018).

The 661 excluded patches also have no implied ecological classification. Their exclusion does not indicate low CPUE, unsuitable habitat or low conservation value.

Their environmental characteristics extended beyond those represented by the surveyed training data, so the Chapter 5 model could not be applied to them with the same empirical support. This creates a clear boundary around the subsequent management-scenario maps: they describe predicted community CPUE across the 598 unsurveyed green spaces for which each retained predictor remained within the sampled environmental range. The excluded patches instead identify parts of the urban green space network where additional field sampling would be required to extend the prediction domain with empirical data.

5.4.4 Limitations and future development of the predictive framework

The main limitation of the predictive framework is the number of independent green spaces available for model training. Repeated seasonal sampling produced a substantially larger set of observations, but the environmental range of the model was still defined by 21 surveyed patches.

The screening procedure also defines environmental similarity using the observed range of each retained predictor. A patch classified within the prediction domain therefore fell within the individual training ranges for woody linear density, grass cover, management intensity, green patch count, matrix hedge density and road density. This does not demonstrate that every combination of those variables was equally well represented among the 21 survey sites. Environmental novelty can occur even where individual predictors remain within their observed ranges if combinations or correlations among those predictors differ from those represented during model fitting (Mesgaran et al., 2014). A green space could therefore contain individually familiar predictor values arranged in a combination that was uncommon within the training dataset. The mapped prediction domain should consequently be treated as a

first screening of environmental applicability, not evidence that uncertainty has been removed from every retained patch.

This limitation is relevant to the structure of the Stoke-on-Trent and Newcastle-under-Lyme study area. The urban green space network extends across historically distinct town centres, suburban areas, formal parks, nature reserves, post-industrial land and semi-natural corridors, while Stoke-on-Trent itself contains approximately 49% open green space (Bixby et al., 2015; Staffordshire Wildlife Trust, 2024; Stoke-on-Trent City Council, 2013, 2026). As established in Chapter 2, this high overall provision is distributed across a heterogeneous, polycentric urban landscape in which green spaces vary substantially in habitat composition, management and surrounding landscape context. Urban landscapes are widely recognised as spatially heterogeneous mosaics in which habitat quality and configuration can vary considerably among green spaces (Beninde et al., 2015; Lepczyk et al., 2017). Increasing the number and environmental spread of surveyed green spaces would improve representation of these combinations and provide a stronger test of model performance across the full urban network. However, retaining the repeated live-trapping design while substantially increasing the number of sites would require considerably greater survey time and field effort, so broader spatial replication represents a practical direction for larger or collaborative future studies.

Prediction also required a smaller set of variables than the ecological analyses in Chapters 3 and 4. Within-patch path density provides the clearest example. Path density was negatively associated with CPUE in Chapter 3, but no spatially complete equivalent could be generated consistently for the unsurveyed green spaces. It was

therefore excluded from the Chapter 5 model for reasons of spatial transferability, not because the earlier ecological relationship had ceased to be relevant. Similar problems occur widely when ecological models are moved from intensively surveyed sites to landscape-scale prediction because variables that can be measured accurately in the field are not always available at equivalent resolution across the prediction area (Elith & Leathwick, 2009; Yates et al., 2018). Chapter 5 therefore represents the portion of the patch and matrix relationships that could be modelled consistently across surveyed and unsurveyed green spaces. Future improvements in fine-resolution urban mapping could enable the incorporation of paths and other small structural features without altering the model's ecological basis.

Management intensity also introduces its own limitations. As discussed in Chapter 3, the three management classes provided a consistent way of representing broad differences in mowing across the surveyed green spaces, but they did not record every component of grassland management. Differences in cutting frequency, timing, vegetation height, biomass removal and the spatial arrangement of cut and uncut vegetation can all alter grassland structure and associated biodiversity (Norton et al., 2019; Watson et al., 2020; Hu & Lima, 2024). The Chapter 5 scenarios therefore compare the management categories represented by the fitted model. They do not identify an exact mowing frequency or management prescription for individual green spaces. A future version of the model, based on measured vegetation structure and more detailed management records, could test whether the broad relationship identified here can be resolved into specific management practices.

CPUE also represents relative capture rate and cannot be treated as an estimate of absolute population density, as discussed in Chapter 3 (Morris et al., 2014; Harkins et al., 2019). Chapter 5 also models total community CPUE, so the spatial predictions do not reproduce the species-specific relationships identified earlier in the thesis. Chapter 3 showed that *A. sylvaticus*, *M. agrestis*, *M. glareolus* and *S. araneus* differed in their responses to habitat structure and management, consistent with wider evidence that small mammal species respond differently to urban vegetation and landscape configuration (Fuentes-Montemayor et al., 2020; Larson & Sander, 2024; Dondina et al., 2025). A patch with relatively high predicted community CPUE should therefore not automatically be interpreted as supporting greater richness, diversity or equally favourable conditions for every species. Species-specific spatial models would provide an important extension where larger numbers of detections are available.

5.4.5 Conclusions

Within the limits outlined in the above section, Chapter 5 answers the two research questions that required integration of the preceding analyses. Local patch conditions and wider landscape structure retained complementary relationships with community CPUE when entered into the same model, answering RQ3. RQ4 was also addressed by demonstrating that these relationships could support prediction across environmentally comparable unsurveyed green spaces when site-blocked validation and environmental screening were used to define the supported prediction domain. The predictions under H4 were supported: patch structure, management and landscape configuration remained represented within the combined model, and predicted CPUE declined from low through moderate to intensive management

scenarios. Absolute differences between management scenarios also varied among patches, although the additive model did not test whether the strength of the management relationship itself changed according to structural condition or surrounding matrix. The spatial variation in these differences therefore cannot be interpreted as evidence that landscape structure statistically modified the management coefficient.

Chapter 5 therefore provides a bounded spatial synthesis of the patch- and landscape-scale results developed across Chapters 3 and 4. It identifies the environmental relationships that remained informative when the two spatial levels were considered together, demonstrates that the resulting model retained predictive ability when entire survey sites were withheld, and establishes where those relationships can be applied across environmentally comparable unsurveyed green spaces. The resulting maps are not a complete assessment of small mammal habitat or abundance across the city. They provide an empirically supported framework for identifying spatial variation in predicted community CPUE under different combinations of patch structure, landscape configuration and grassland management, while also identifying parts of the urban green space network where additional field data are required before the same inference can be extended. Chapter 5 therefore completes the progression from field-based ecological relationships to environmentally bounded spatial prediction, providing the basis for the wider synthesis of urban green space ecology and management developed in Chapter 6.

Chapter 6 General Discussion

6.1 Revisiting the thesis aims

The overarching aim of this thesis was to determine how local habitat structure and management interact with wider landscape configuration to shape small mammal communities within urban green spaces, and to translate these relationships into a spatially explicit evidence base for urban habitat management and nature-recovery planning. The aim developed from gaps identified in the urban ecological literature outlined in Chapter 1. Small mammals remain poorly represented in research examining urban habitat structure and vegetation management alongside the surrounding urban matrix (Wang et al., 2025). This limits understanding of a taxonomic group whose use of green spaces depends on conditions near ground level and whose movement through cities can be restricted by exposed surfaces, roads, and breaks in vegetation (Pearce & Venier, 2005; McDonald & St. Clair, 2004; Grilo et al., 2018).

Vegetation management formed a central component of the research because substantial ecological change can occur without any change in mapped green space area. Mowing, vegetation clearance and management of woody features alter vegetation height, density, succession and continuity while the same land remains classified as green space (Aronson et al., 2017; Watson et al., 2020). Broad measures of green space quantity can therefore conceal differences in the conditions available to ground-dwelling species. The same problem occurs at the landscape scale, where similar quantities of surrounding green cover can differ in subdivision, isolation, road pressure and the continuity of vegetated features (Fischer & Lindenmayer, 2007;

LaPoint et al., 2015; Galantinho et al., 2020). The research framework developed in Chapter 1 addressed both sources of variation by examining the internal condition of green spaces alongside the structure of the surrounding matrix. The applied component of the aim also reflected the increasing use of spatial nature-recovery frameworks in England, including Biodiversity Net Gain, Nature Recovery Networks and Local Nature Recovery Strategies (Environment Act, 2021; Department for Environment, Food and Rural Affairs, 2023). These approaches require evidence on where habitat enhancement is likely to provide ecological value within wider networks, giving direct relevance to the thesis focus on habitat condition, management and landscape configuration (Lawton et al., 2010; Environment Act, 2021; Department for Environment, Food and Rural Affairs, 2023).

Small mammals provided an appropriate assemblage for examining these relationships because the species occurring within urban green spaces differ in their dependence on herbaceous cover, woody vegetation, moisture and structurally protected movement routes (Dickman & Doncaster, 1987; Tattersall et al., 2002; Buesching et al., 2008; Atkins et al., 2018). Their responses can be examined at the community level while retaining species-specific differences in habitat association.

Chapter 2 established the biological and spatial framework used to test these questions. The spatial framework then placed the 21 stratified-random survey sites within a network of 1,280 mapped green space polygons, allowing the same study to move from field observations to landscape analysis and spatial prediction.

Chapter 3 provided the first direct test of the expectation developed in Chapter 1 that the ecological value of urban green space would depend on its internal structure and

management, with patch area alone providing an incomplete description of habitat suitability (Fahrig, 2003; Fischer & Lindenmayer, 2007; Concepción et al., 2015). This was broadly supported under H1, although the evidence was considerably more specific than the original hypothesis implied. Woody linear density was the only patch-scale predictor positively associated with richness, CPUE and Simpson's reciprocal diversity, while management intensity and path density were most clearly expressed through community CPUE. Patch area did not produce the expected positive species–area response, and broad tree cover was unrelated to the community and focal-species responses. The findings placed greater emphasis on the physical structure and management of vegetation than on mapped habitat extent, consistent with earlier small mammal work showing strong responses to dense ground-level cover and linear woody features (Dickman & Doncaster, 1987; Tattersall et al., 2002; Gelling et al., 2007b).

H2 was also supported at its broader level because the four focal taxa showed different habitat associations, confirming the Chapter 1 expectation that a community response could conceal substantial species-specific variation. The narrower prediction that *M. glareolus* and *M. agrestis* would respond more strongly to woody vegetation and lower mowing than *A. sylvaticus* was only partly suggested. *A. sylvaticus* itself responded positively to woody linear density and negatively to management intensity, while *M. agrestis* showed stronger associations with grassland and watercourses. The patch-scale work therefore answered RQ1 while also refining the ecological assumptions that produced H1 and H2: internal habitat structure and management mattered, but the assemblage did not respond along a simple gradient from generalist to specialist.

Chapter 4 tested whether that patch-scale explanation was sufficient by extending the analysis into the surrounding urban matrix. H3 was developed from the expectation that isolation, fragmentation and potential connectivity would influence small mammals beyond the conditions measured inside the focal green space (Hanski, 1998; Fahrig, 2003; Fischer & Lindenmayer, 2007; Galantinho et al., 2020). The strongest evidence supported the predicted negative relationships with green space subdivision (measured as green patch count), structural isolation (measured as nearest green patch distance), and road density, while the anticipated benefits of surrounding green amount, hedgerows, and watercourses were more selective and became clearer at the broader 500 m extent. This shifted the interpretation away from a simple expectation that more surrounding green cover would produce stronger communities (Fahrig, 2003; Betts et al., 2014; LaPoint et al., 2015; Galantinho et al., 2020), and towards the configuration and structural continuity of that cover. Chapter 5 then tested the central thesis proposition more directly by asking whether the local and landscape relationships retained information when considered together. Woody linear density and management intensity remained important within the patch, while green patch count and road density retained landscape-level relationships with CPUE.

Grass cover weakened substantially, and matrix hedge density became marginal and non-significant once the two spatial levels were integrated. This was an important development from the separate Chapter 3 and 4 analyses because it showed that relationships identified at one spatial level did not necessarily retain the same explanatory strength when local and landscape conditions were considered together. H4 was nonetheless supported: patch-scale habitat structure, grassland

management and landscape configuration all remained represented within the final model, and predicted CPUE declined from low through moderate to intensive mowing as expected. The absolute differences between management scenarios also varied among green spaces according to their combined patch and matrix characteristics, consistent with the spatial prediction developed in Chapter 5. The progression from Chapters 3 to 5 therefore moved beyond identifying separate local and landscape associations to establishing which components remained informative when the ecological framework was integrated, directly addressing RQ3 and providing the basis for the spatial application required by RQ4.

Chapter 5 also addressed RQ4 by testing whether the integrated model could support spatial comparison beyond the surveyed sites. Site-blocked validation showed reduced accuracy at withheld sites but retained useful relative ordering, while environmental screening restricted mapped predictions to environmentally comparable green spaces. The model therefore supports comparative prioritisation and management-scenario assessment rather than precise CPUE estimation at individual unsurveyed patches.

6.2 Development of the Research Framework

The methodological framework developed around the scale of inference established in Chapter 1: variation among whole urban green spaces and the landscapes surrounding them. Individual grasslands, hedgerows, scrub margins or woodland compartments were components of those green spaces rather than independent study units, and the field design was intended to characterise the small mammal assemblage using the range of suitable vegetation present within each patch.

Restricting site selection to green spaces containing an identical combination of habitats would have narrowed the sampling frame to an atypical subset of the wider urban network and removed part of the environmental heterogeneity the research was designed to investigate.

Urban green spaces frequently contain mixtures of habitat types that differ substantially between sites, and previous urban ecological studies have similarly shown that broad classifications such as park or green space can contain considerable variation in vegetation structure, disturbance and ecological condition (Angold et al., 2006; Concepción et al., 2015; Wood et al., 2018). The 21 survey sites were therefore selected randomly within three predefined grassland-management classes, giving representation across the management gradient without deliberately favouring nature reserves, structurally complex sites or locations already known to support small mammals. Within those sites, trapping was randomised through the suitable and accessible vegetated habitats present, including grassland, woody margins, scrub and woodland-edge vegetation where these occurred. A habitat absent from a green space could not be included in that site's trapping layout, allowing differences in habitat composition to remain part of the environmental variation being tested rather than imposing an artificial requirement for identical habitat representation among sites (Greenwood & Robinson, 2006; Mathews et al., 2006).

This approach also avoided assuming in advance that small mammals would occur only within habitats considered optimal from existing natural or semi-natural systems. Species have recognised habitat associations, but habitat use can broaden

or change with population density, competition, disturbance and the configuration of available vegetation, while urban environments contain combinations of resources and structures that do not always correspond closely with those occurring in less modified landscapes (Sundell et al., 2012; Concepción et al., 2015; Dondina et al., 2025). Restricting trapping exclusively to hedgerow bases, woodland edges or other locations judged visually favourable would therefore have made the sampling design partly dependent on the ecological relationships that the study was intended to test. Randomisation within the suitable vegetated domain allowed animals to be recorded across the range of usable habitat available within each green space without requiring live traps to be distributed across surfaces where the method could not function appropriately. Buildings, sealed surfaces, paths, open water and unsafe locations remained part of the mapped ecological characterisation of each patch even though they were excluded from trap placement. The trapping data thus represented the assemblage using suitable ground-level vegetation within the green space, while the GIS variables represented the wider composition and structure of the complete mapped patch (Flowerdew et al., 2004; Gurnell & Flowerdew, 2019). The design was not intended to estimate habitat-specific population density or to allocate trapping effort proportionally across every mapped land cover class.

Longworth live trapping was retained after the pilot phase because the response variables required repeated and sufficiently reliable species-level identification to calculate richness, CPUE, Simpson's reciprocal diversity and focal-species responses. Footprint tunnels, hair traps and remote cameras were considered during method development and could have provided useful complementary occurrence data, predominantly where large numbers of sites needed to be surveyed with

comparatively low handling effort (De Bondi et al., 2010; Chiron et al., 2018). Their outputs were less well matched to the analytical requirements of this thesis. Glennon et al. (2002), for example, found useful association between track-tube and live-trapping indices but also identified limitations in distinguishing closely related rodent and shrew taxa from tracks. Similar problems occurred during the pilot work, where mouse and vole footprints could not be assigned consistently to the species required for the later analyses, while processing hair samples added field and laboratory effort without producing an equivalent repeated measure of species-specific capture response. Retaining these methods throughout the full survey would therefore have increased survey effort without materially improving the richness, CPUE or diversity metrics used in Chapters 3–5. Longworth trapping provided the most appropriate compromise between direct species identification, repeated survey effort and the standardised capture information required by the research questions, although species with low trappability or ecological characteristics poorly represented by ground-level traps may still have been underestimated (Brouard et al., 2015; Kettel et al., 2016).

The physical arrangement of traps was governed by the same division between community comparison and fine-scale population estimation. Fixed grids are well suited to studies requiring estimates of population density, home range or spatial dispersion, but many of the urban green spaces surveyed here contained narrow vegetation strips, abrupt habitat boundaries, paths, water, hardstanding and irregular external geometry. Maintaining a regular grid across those sites would either have placed traps outside the intended vegetated sampling domain or restricted sampling to smaller portions of the green space (Pearson & Ruggiero, 2003; Sutherland, 2006).

Pearson and Ruggiero (2003) found that transects produced more captures, individuals, and species than grid arrangements for equivalent effort and identified their greater efficiency in assessing community composition and relative abundance. Transects were therefore better aligned with the objectives of this study and could be adapted to the geometry of each green space while maintaining the ten-metre inter-trap spacing. A grid-based design could have provided greater resolution of within-patch spatial organisation and supported density or home-range estimation (Pearson & Ruggiero, 2003; Gurnell & Flowerdew, 2006), but achieving that consistently across green spaces ranging widely in area and configuration would have required substantially greater trapping effort or a shift in the inferential unit from the whole green space towards individual habitat compartments.

Variation in patch area created a more substantive sampling problem because the surveyed green spaces ranged from 1.4 to 79.6 ha and differed markedly in the amount and arrangement of vegetation available for trapping. Applying the same number of traps to every site would have concentrated sampling effort within the smallest patches while providing progressively poorer spatial representation of the largest, whereas maintaining an identical trap density across the full patch area would have required considerably greater equipment, checking time and animal-handling capacity (Gurnell & Flowerdew, 2006; Sutherland, 2006). Sampling effort can materially influence the richness and abundance recorded during small mammal surveys, with more trapping locations increasing spatial coverage and the probability of encountering species that are distributed unevenly within a site (Bovendorp et al., 2017; Green et al., 2024).

More directly, small mammal studies have increased trapping effort with the area of the sampling unit to maintain adequate spatial representation across patches of substantially different sizes. Palmeirim et al. (2020), for example, increased the number of live-trap transects and pitfall arrays deployed in larger forest patches specifically to compensate for the reduction in sampling density that would otherwise occur as patch area increased. The area-based allocation used here followed the same underlying principle across the study's broad patch-size gradient: trap numbers increased with patch area, so that larger green spaces received greater sampling effort, without assuming that a fixed trap density could be maintained across all hectares of sites that differed substantially in geometry, vegetation and accessibility. The broad patch-area categories provided a reproducible guide to that increase rather than a rigid ecological threshold, with individual allocations adapted as necessary by the number of traps available and the amount and configuration of suitable vegetation. The underlying relationship was retained throughout, with sampling effort increasing across the patch-size gradient, culminating in a maximum allocation of 60 traps at the largest sites. The approach therefore preserved a structured relationship between patch size and survey effort while accommodating the practical variation inherent within heterogeneous urban green spaces.

This compromise was appropriate to a study comparing whole heterogeneous green spaces, although a larger trap stock would have allowed finer scaling of effort within the largest patches and reduced differences in the proportion of each green space encountered by the trapping layout. Future surveys could improve this element by determining trap allocation from pilot-based species-accumulation or effort curves, or by increasing the number of transects until additional spatial effort produced little

further increase in observed richness; both approaches would allow sampling intensity to be tuned empirically to the community response being measured (Bovendorp et al., 2017; Green et al., 2024). In this research, variation in deployed effort was incorporated through adjusted trap nights and CPUE following Nelson and Clark (1973), allowing capture responses to be expressed per unit of effective trapping effort. That standardisation improved comparability among sites with different numbers of deployed and functional traps, but it could not make the physical sampling footprint equivalent among green spaces or remove differences arising from patch geometry and the spatial distribution of suitable vegetation.

Repeated sampling strengthened temporal representation at the cost of the number of independent landscapes that could feasibly be surveyed. Five sessions at each site between May and October reduced the extent to which comparisons depended on a single period of activity, weather or seasonal population fluctuation and produced 6,966 adjusted trap nights across the study. The independent spatial sample nevertheless remained 21 green spaces. Repeated observations improved estimation of recurring patch-level relationships and were accommodated statistically through the site-level random effect, but they did not provide the additional environmental coverage that would have resulted from surveying a larger number of independent sites. This trade-off became most apparent when the Chapter 5 model was applied to the wider urban network. Small ecological samples can support useful predictive relationships, but model stability and transferability generally become more uncertain as predictor complexity increases relative to independent sample size (Wisz et al., 2008). A future study focused principally on landscape-wide prediction could therefore increase the number and environmental spread of surveyed green

spaces, potentially by reducing the number of repeat sessions or combining live trapping with complementary lower-effort detection methods. Such a change would improve spatial representation while sacrificing some of the temporal depth provided by repeated live trapping.

Extending the work through a larger field and research team would provide another route to increasing the number of independent green spaces without reducing repeat sampling to the same extent, by distributing trap checks, sample processing and image or track assessment among researchers with relevant expertise. Greater specialist input could improve consistency in footprint and hair identification, although it would not remove the inherent taxonomic limitations of indirect detection methods where morphologically similar species produce poorly distinguishable signs (Glennon et al., 2002).

Moving from field survey resolution to landscape-wide modelling introduced a second trade-off between ecological detail and reproducibility. Detailed habitat assessment could describe sward structure, scrub condition, basal vegetation, litter, management history and other characteristics that broad land cover datasets cannot reproduce, but equivalent field assessment of all 1,280 mapped green spaces was not feasible. National spatial datasets therefore provided the common framework required to calculate comparable variables across both surveyed and unsurveyed patches. UKCEH Land Cover Map classifications were used for broad habitat composition because they provided a nationally consistent and reproducible basis for extracting the same habitat variables across the full spatial framework (Marston et al., 2022). Roads were derived from OS Open Roads (Ordnance Survey, 2024), while

watercourses were represented using the UKCEH Digital River Network of Great Britain, cross-referenced with the Environment Agency Statutory Main River Map (UK Centre for Ecology and Hydrology, 2000; Environment Agency, 2025). Where information central to the thesis could not be represented adequately by these broader datasets, greater field-derived resolution was retained.

Grassland management was classified from Stoke-on-Trent City Council management schedules and guidance, supplemented by field assessment and the management information used to establish the three mowing categories in Chapter 3 (Stoke-on-Trent City Council, 2024). Within-patch woody linear features were mapped independently because broad raster tree cover classes could not distinguish the hedgerows, scrub margins and treelines relevant to the structural conditions being tested. The landscape-scale equivalent relied on the UKCEH Land Cover Plus: Hedgerows 2016–2021 dataset, which provided a reproducible mapped measure of woody linear features on field boundaries but was not a complete inventory of woody linear vegetation across the urban matrix (Broughton et al., 2024). The dataset is derived from lidar and mapped field-boundary features and excludes some land cover settings, including urban/suburban and woodland classes, so matrix hedge density represents the woody linear features captured by that dataset rather than all hedgerows, treelines and scrub margins surrounding each green space. Alternative approaches could have included manually digitising hedgerows and other woody linear features across the wider landscape or deriving them through further reclassification of existing land cover data. Manual digitisation would have provided greater spatial detail but was not feasible across the full study area within the available time and workload, and deriving additional woody features from broader

land cover classes would have introduced further uncertainty about whether the mapped features accurately represented physical hedgerows, treelines, or scrub margins on the ground.

Standardisation therefore reduced fine-scale ecological resolution while enabling the later predictive stage. A detailed Phase 1-style habitat survey of 21 sites might have distinguished more vegetation types and condition variables, but most of those measurements could not then have been reproduced across the unsurveyed network. The use of broader transferable variables allowed the same environmental framework to follow the analysis from field sampling to spatial prediction, although interpretation remained constrained by what those variables actually measured. Grassland cover, for example, represented habitat extent without directly describing sward height or litter structure, while matrix hedge density represented the quantity of mapped woody linear vegetation without demonstrating its condition or functional use. The resulting spatial model should therefore be understood as a standardised representation of measurable habitat and landscape characteristics rather than a replacement for detailed ecological assessment at individual green spaces.

6.2.1 Sequential modelling framework

The sequential modelling framework provided a further means of controlling complexity within the limits imposed by 21 independent spatial replicates. Candidate variables were established from the ecological literature, characteristics of the study system and pilot work before the final integrated model was fitted. Chapter 2 defined the requirements that predictors should have interpretable ecological meaning and, where they were intended for spatial prediction, an equivalent representation that could be derived across surveyed and unsurveyed green spaces. Chapter 3 first

tested the predefined patch-scale variables, while Chapter 4 deliberately separated the surrounding matrix and applied the same candidate predictor structure at the predefined 250 m and 500 m analytical extents.

The integrated Chapter 5 model was therefore the end of a narrowing analytical sequence in which local and landscape predictors had already been examined against the relevant biological responses before being considered together. This constrained the final model to variables with prior ecological justification, support from the preceding scale-specific analyses and, where required, sufficient spatial representation for application beyond the surveyed sites.

Fitting every available habitat, management, geometric and landscape metric simultaneously would have increased model complexity relative to the number of independent green spaces and increased the opportunity to fit variation specific to the surveyed sites. Ecological modelling studies have repeatedly cautioned against unrestricted variable selection and excessive model complexity when the number of candidate predictors is high relative to sample size, because apparently strong performance in the fitted model can coexist with poor transfer to independent observations (Whittingham et al., 2006; Wisz et al., 2008; Harrison et al., 2018). The staged framework reduced that risk by beginning with biologically defined candidate variables, removing redundant or poorly interpretable metrics, assessing patch and matrix components separately and only then combining a smaller set of supported and reproducible predictors. This was not an independent sequence of confirmatory tests: Chapter 5 drew on relationships identified from the same 21 green spaces, so the final predictor set remains conditional on the analytical pathway through

Chapters 3 and 4. The strength of the approach lies in limiting the number of candidate relationships carried into the integrated model, preserving their ecological interpretation and then testing whether the resulting model retained predictive performance when complete green spaces were withheld from model fitting.

Predefining the 250 m and 500 m matrix extents strengthened the consistency of the landscape analysis by fixing the spatial comparisons before the Chapter 4 models were examined. Landscape relationships are scale-dependent (Wiens, 1989; Jackson & Fahrig, 2012; Miguet et al., 2016), and testing a large series of alternative buffer distances would have afforded substantial flexibility to retain whichever scale yielded the strongest relationship within a relatively small spatial sample. The two extents were selected from ecological reasoning and comparable small mammal studies and the same predictor structure was applied at each, allowing the response to immediate and broader surrounding matrix contexts to be compared directly (Fischer et al., 2011; Fuentes-Montemayor et al., 2020; Larson & Sander, 2024). This provided a controlled test of scale dependence without multiplying the number of candidate models solely to identify an apparently optimal distance. The resulting 250 m and 500 m relationships should still be interpreted as responses measured within predefined analytical extents, not as estimates of home range, dispersal distance or universally optimal scales for urban small mammals (Jackson & Fahrig, 2012; Miguet et al., 2016). A larger independent site sample could support evaluation across a wider range of spatial extents, but the fixed two-scale framework was appropriate to the environmental coverage and sample size available here.

Site-blocked validation then tested whether the integrated model retained predictive value beyond the green spaces used to estimate its relationships. The five repeated surveys at each site created a hierarchical dataset in which observations from the same green space shared the same underlying patch and landscape characteristics. Randomly dividing those observations between training and validation datasets would therefore have allowed information from the same green space to contribute to both model fitting and evaluation, producing a test that was poorly matched to the intended application of predicting CPUE at an unsurveyed site. Cross-validation methods should reproduce the dependence structure of ecological data and the prediction task being evaluated, with random partitioning capable of underestimating predictive error where observations are spatially or hierarchically grouped (Roberts et al., 2017).

Leave-one-site-out validation addressed this directly by withholding every observation from one green space before predicting its CPUE from relationships estimated using the remaining sites.

6.3 Development of ecological understanding across the thesis

One of the clearest developments from the research was that mapped green space became an increasingly inadequate description of the habitat available to small mammals. Green spaces initially provided a necessary spatial unit through which habitat amount, management, and landscape position could be compared, but their biological value varied substantially according to characteristics that were poorly represented by area or by broad habitat classification alone. Dense woody margins, grassland structure, mowing and internal disturbance altered conditions close to

ground level without necessarily changing the mapped extent of the green space itself. The consistent association with woody linear density, contrasted with the weak evidence for broad tree cover and the absence of a simple positive patch-area relationship, provided the clearest empirical expression of this difference. Similar conclusions emerge from studies in modified landscapes where small mammal responses depend strongly on vegetation density, edge structure and sheltered ground-level cover (Dickman & Doncaster, 1987; Tattersall et al., 2002; Gelling et al., 2007b; Atkins et al., 2018). The ecological unit represented by a green space polygon was therefore useful for comparison, but its area and broad composition did not adequately describe the conditions experienced by organisms occupying it.

The research also moved away from the assumption that habitat quality could be represented along a single gradient shared by the whole assemblage. Community responses reflected several properties of the same green spaces, while the focal taxa differed in their associations with grassland, woody vegetation, watercourses, paths and management. These differences were more informative than the narrower expectation that the more specialised species would simply show stronger responses to structurally complex or less intensively managed habitat (Dickman & Doncaster, 1987; Fuentes-Montemayor et al., 2020; Dondina et al., 2025). *A. sylvaticus*, despite its broad habitat use and numerical dominance within the dataset, responded to several measured environmental characteristics, while the other focal taxa showed narrower and contrasting associations. The resulting picture is of an urban assemblage assembled from species responding to different components of habitat structure and disturbance, meaning that conditions associated with high community CPUE cannot automatically be interpreted as optimal for every member of the

community (Pearce & Venier, 2005; Rowe & Terry, 2014). Differences among richness, CPUE and Simpson's reciprocal diversity reinforce the same point because environmental relationships were not expressed uniformly across the three measures of community structure within surveyed patches.

Habitat value also became increasingly dependent on what occurred beyond the boundary of the focal green space. The surrounding matrix added information that could not be inferred from internal vegetation alone, with the repeated associations involving green space subdivision, structural isolation and road density showing that patches with otherwise comparable internal conditions occurred within substantially different landscape settings. Total surrounding green cover provided a less consistent explanation than how that cover was divided and separated, while positive relationships with hedgerows, watercourses and broader green cover occurred more selectively. This shifts the ecological interpretation from green spaces as discrete habitat islands towards a heterogeneous urban network in which both the condition of individual patches and the arrangement of habitat between them are associated with community structure. The analysis measured structural landscape relationships rather than movement itself, but the pattern agrees with work showing that the ecological consequences of geographical separation depend on the composition and permeability of the intervening matrix (Ricketts, 2001; McDonald & St. Clair, 2004; Wilson et al., 2016; Galantinho et al., 2020).

Vegetation management introduced a further conflict between reducing disturbance and maintaining the range of habitat conditions required by different species. Frequent mowing can simplify vegetation structure, reduce sward height and litter

accumulation, and repeatedly interrupt successional development (O'Sullivan et al., 2017; Winkler et al., 2024). Decreasing management does not, however, produce an indefinitely increasing gradient of habitat quality because vegetation succession changes the balance between herbaceous cover, tall vegetation and woody structure, with different consequences for species associated with those habitat conditions (Churchfield et al., 1997; Jasiulionis et al., 2011; Balčiauskas et al., 2017; Uchida & Ushimaru, 2014; Fuentes-Montemayor et al., 2020). The negative association between management intensity and community CPUE supports the importance of retaining less intensively managed vegetation, while the contrasting focal-species responses show that this relationship cannot be extended into a general expectation that progressively lower intervention will benefit the entire assemblage. Management could therefore be better understood as one process determining the structural state of urban habitat, with its ecological consequences depending on the vegetation already present and the species using it. The thesis adds support to an increasingly heterogeneous view of urban grassland management in which variation in disturbance and vegetation structure is ecologically more meaningful than treating either intensive maintenance or minimal intervention as a universal optimum.

The combined evidence supports an organism-centred view of urban green spaces as heterogeneous habitat units embedded within equally heterogeneous matrices. Their ecological value cannot be assigned from mapped extent, management intensity or apparent connectivity in isolation; it emerges from the combination of local structural conditions and the wider landscape in which those conditions occur. This accords with contemporary urban ecological frameworks that treat urbanisation as a spatially heterogeneous process involving interacting gradients of land use, vegetation,

infrastructure and disturbance rather than a single transition from natural to urban habitat (Grimm et al., 2008; Pickett et al., 2011; Beninde et al., 2015; Lepczyk et al., 2017).

6.3.1 Contrasting evidence, non-significant results and ecological uncertainty

Most of the principal ecological predictions were supported across the thesis, although the strength and consistency of individual relationships differed among species, community metrics and spatial scales. These contrasts are ecologically informative because the same environmental feature may influence different components of a community in different ways, and because urban habitat characteristics rarely operate independently (Dickman & Doncaster, 1987; Dondina et al., 2025; Wang et al., 2025). Non-significant relationships were treated as an absence of sufficient statistical support within the model in which they were tested, while differences in direction, magnitude and consistency across related analyses identified where ecological responses remained uncertain or context dependent. Tree cover, watercourse density, and matrix hedge density provide useful examples because expectations based on established small mammal ecology were not consistently expressed across all response or spatial scales.

The absence of any detectable relationship with proportional tree cover was one of the clearer departures from simple ecological expectation. Both *A. sylvaticus* and *M. glareolus* have established associations with woodland and woody habitat, yet tree cover was unsupported across the community and focal-species models, while woody linear density was positively associated with *A. sylvaticus*, *M. glareolus* and all three community measures. Previous studies help reconcile this apparent conflict

because use of woodland by these taxa varies with understorey development and vegetation density, and recovery of *A. sylvaticus* and *M. glareolus* populations has been associated with increasing structural complexity following changes in browsing pressure (Buesching et al., 2008; Bush et al., 2012; Fuentes-Montemayor et al., 2020). The non-significant tree-cover result therefore casts uncertainty on the value of proportional canopy extent as a predictor; it does not justify a broader conclusion that trees or woodland have little ecological relevance to these species.

Contrasts among the three community metrics created a second source of ecological uncertainty. CPUE responded to a wider range of patch characteristics than either richness or Simpson's reciprocal diversity, while the matrix analysis produced different combinations again. Management intensity, grassland cover and within-patch watercourse density were associated with CPUE without corresponding relationships across all community measures; at 500 m, matrix green proportion and watercourse density were associated with Simpson's reciprocal diversity without equivalent increases in richness or CPUE. The three responses measure different components of the assemblage, so agreement among them should not be expected automatically. The Chapter 3 results illustrate this directly: greater capture rate under particular habitat conditions was not consistently accompanied by an increase in the number of species or a more even community. This matters when interpreting ecological condition because a variable associated with greater overall activity can leave community composition largely unchanged, while another may alter the balance among species without producing more captures (Hillebrand et al., 2008; Morris et al., 2014).

Some of the clearest uncertainty arose from relationships that changed with spatial extent or after local and matrix conditions were considered together. Matrix hedge density showed no clear association with CPUE at 250 m, a strong positive association at 500 m, and a weaker positive estimate in the integrated model. Watercourse density showed an equally selective landscape response, being associated with Simpson's reciprocal diversity at 500 m without corresponding relationships with richness or CPUE. Comparable studies also report changes in the spatial extent at which small mammal landscape relationships are detected: Fischer et al. (2011) found different responses across 100 m, 250 m and 500 m extents, Dorigo et al. (2021) obtained useful relationships at both 250 m and 500 m, while Bowman et al. (2002) found no significant landscape associations at 500 m. The spatial extent at which a relationship is absent in one analysis cannot therefore be assumed to define the ecological relevance of that feature across other extents or urban systems. For hedgerows and watercourses, future work incorporating feature condition, vegetation structure and species-specific use could separate the ecological importance of the mapped feature from the limitations of representing it through density alone.

These contrasting results are consistent with the heterogeneity expected within urban landscapes, where equivalent mapped habitat classes can occur under different management, structural and surrounding landscape conditions. Unexplained variation is expected even where the principal ecological relationships are repeatable. Greater independent spatial replication, targeted sampling of the less frequently captured focal taxa and finer measurements of woody and riparian structure would reduce several of these uncertainties, while replication in other

urban systems would establish which relationships are locally contingent and which are more general features of urban small mammal ecology.

6.4 Spatial prediction, model performance and transferability limits

The spatial prediction in this thesis extended the ecological relationships identified within the surveyed green spaces into a form that could be evaluated across the wider urban network, but its value depended on retaining the empirical boundaries of those relationships. Spatial ecological models are commonly used to extend biological observations across unsurveyed areas because environmental variables can be measured continuously, where field observations cannot (Guisan & Zimmermann, 2000; Elith & Leathwick, 2009). The ability of a model to return a prediction at an unsurveyed location does not, however, establish that the underlying ecological relationship has been demonstrated under the conditions occurring there. Predictive performance can decline when models are transferred across environmental or spatial conditions that differ from those used during fitting, even where the same predictor variables can be calculated (Wenger & Olden, 2012; Sequeira et al., 2018; Yates et al., 2018). The modelling framework used here was designed to retain that separation between spatial coverage and empirical support.

Site-blocked validation provided the first test of how much of the fitted relationship remained when predictions were made for a complete green space that had contributed no trapping observations to model fitting. Performance declined relative to the fitted data, but the positive relationship between observed and predicted CPUE remained and the rank correlation remained comparatively strong. The model was therefore better at preserving relative differences among lower and higher CPUE

observations than at reproducing their exact values at withheld sites. That level of performance matched the intended spatial application, which was to identify broad differences among green spaces and compare management scenarios, not to estimate the number of captures expected at an individual unsurveyed patch.

Environmental screening placed a separate boundary around where those validated relationships were applied. Only 598 of the 1,259 unsurveyed green spaces contained retained predictor values within the environmental ranges represented by the survey data, leaving 661 patches outside the mapped prediction domain as described in Chapter 5. The scale of that exclusion is ecologically informative in its own right because more than half of the wider network contained environmental conditions that were not represented fully by the 21 surveyed green spaces. Urban green infrastructure may appear relatively uniform when reduced to mapped green space polygons, yet the predictor combinations relevant to small mammals varied sufficiently across one conurbation to prevent equal support for prediction throughout the whole green space network. Extrapolation beyond sampled environmental conditions is a recognised source of uncertainty in ecological models because fitted relationships can continue mathematically beyond the observations from which they were estimated without evidence that the biological response follows the same form (Guillera-Arroita et al., 2015; Yates et al., 2018; Araújo et al., 2019).

Restricting the mapped predictions did not reduce the usefulness of the spatial framework, though. The 598 retained patches identify where the existing field evidence can support comparative prediction, while the 661 excluded patches

identify where additional observations are needed before the prediction domain can be expanded. Those excluded green spaces should not be interpreted as lower-quality habitat or locations with low expected CPUE; their environmental characteristics were insufficiently represented by the training data to justify equivalent inference. Further surveys targeted across those environmental conditions could progressively extend the sampled range and provide a direct test of whether the relationships identified here continue, weaken or change under combinations currently absent from the field dataset. Spatial modelling can therefore serve both as an analytical endpoint and as a means of directing subsequent empirical sampling.

This has wider relevance to the use of GIS-derived ecological models in urban planning. Continuous prediction surfaces can imply a level of certainty that is not present in the biological observations underlying them, especially where model evaluation is based only on fitted performance or where environmental similarity between surveyed and prediction locations is assumed (Araújo et al., 2019; Franklin, 2023). Chapter 1 identified a related problem in connectivity modelling, where mapped corridors and resistance surfaces are frequently produced without independent biological data confirming that organisms respond to them as predicted (Laliberté & St-Laurent, 2020). The approach developed here treats spatial prediction as an evidence-led means of prioritising and comparing urban green spaces while retaining field survey as the basis for extending ecological inference. For applied urban ecology, knowing where a model requires new data can be as useful as producing predictions for the areas it already represents.

6.5 Wider contribution to urban ecological research and nature recovery

Urban ecological research remains uneven in the taxa used to assess green space quality, with birds and invertebrates receiving substantially greater attention than mammals and relatively few studies combining small mammal responses with vegetation management, internal habitat structure and the surrounding matrix (Zhao et al., 2022; Rega-Brodsky et al., 2022; Wang et al., 2025). Small mammals remain ecologically relevant within these systems because they occupy several trophic roles and respond directly to ground-level vegetation, disturbance and breaks in habitat continuity (Pearce & Venier, 2005; Rowe & Terry, 2014; Torre & Balčiauskas, 2024). The research adds evidence from a group that experiences urban green infrastructure differently from more mobile taxa and from assessments based primarily on vegetation extent or canopy cover. Their distributions and community structure provide evidence of whether green spaces retain the ground-level refuge, resources and continuity associated with functioning terrestrial habitat.

The ecological importance of widespread species should not be reduced because they attract less attention within routine protected-species assessment or urban planning. The focal assemblage contained taxa associated with grassland, woody vegetation, wetter ground and a broader range of vegetated habitats, allowing the same green spaces to be evaluated through several ecological requirements. High activity by an adaptable species such as *A. sylvaticus* did not guarantee an equally strong response across the wider assemblage, while *M. agrestis*, *M. glareolus* and *S. araneus* identified different components of habitat structure and disturbance. Common small mammals can therefore contribute useful evidence of ecological

condition, even where they are not the primary focus of routine protected-species assessment or urban planning. Their associations with rough grassland, scrub, woody margins and other structural characteristics also indicate whether heavily modified urban spaces retain features of less simplified ecological systems. Urban nature recovery should account for these widespread components of ecological communities alongside the species that already dominate survey and conservation attention. Chapter 1 similarly identified that high abundance of adaptable taxa can persist while species with narrower ecological requirements decline.

Nature recovery increasingly operates through spatial networks, including Local Nature Recovery Strategies and wider approaches seeking to retain, improve and reconnect habitat (Lawton et al., 2010; Department for Environment, Food and Rural Affairs, 2023). BNG also provides a structured means of accounting for habitat type, condition and strategic significance, although habitat metrics remain proxies for the ecological function experienced by individual taxa (UK Government, 2024; Wang et al., 2025). The results show why biological evidence remains useful alongside mapped habitat assessments. Narrow hedgerows, scrub margins, riparian vegetation and rough grassland can occupy relatively little mapped area while providing substantial ground-level structure, whereas intensive mowing, path construction or removal of basal vegetation can alter habitat conditions without changing the mapped area of the green space. Nature recovery based on habitat quantity alone may therefore retain the physical extent of green infrastructure while missing changes in its value to less mobile terrestrial fauna.

Management recommendations emerging from the thesis favour spatial variation over a single prescription applied across every urban green space. Woody linear features should be retained and managed to preserve basal cover and continuity, reflecting the consistent positive associations with woody linear density and previous evidence of small mammal use of structurally developed hedgerows and boundaries (Tattersall et al., 2002; Gelling et al., 2007b; Atkins et al., 2018). Intensive mowing can be reduced in selected areas while retaining shorter vegetation where recreation, access or other ecological objectives require it. Taken together, the findings support a mosaic incorporating shorter grass, taller grassland, rough margins and developing woody vegetation rather than uniform mowing across an entire patch. They also support retaining rough herbaceous and woody vegetation along watercourse margins where compatible with other management requirements, and avoiding unnecessary increases in path density through suitable vegetation. Management of linear vegetation outside formal green spaces also matters because clearance along roads, canals, railways and watercourses can alter continuity between patches even where the total area of mapped green space remains unchanged (O'Sullivan et al., 2017; Galantinho et al., 2020).

The spatial model allows these interventions to be targeted according to ecological context. Green spaces within the supported prediction domain can be compared to identify areas associated with relatively higher and lower predicted community CPUE. Higher-predicted patches provide reference points for examining combinations of management, woody structure and surrounding landscape conditions already associated with stronger small mammal responses. Lower-predicted patches can then be assessed against the same variables to identify whether actionable

characteristics, such as intensive mowing or limited woody structure, offer realistic opportunities for improvement. A low prediction does not identify a single cause or prove that management intervention will increase CPUE, as some influential landscape conditions cannot be altered readily. It does, however, provide a more informed basis for targeting investigation and intervention than applying identical management prescriptions throughout the urban network.

The same framework identifies where ecological knowledge remains insufficient. The practical contribution is therefore twofold: the model identifies where existing evidence can support targeted management for small mammal communities and where additional ecological information is required before equivalent recommendations can be made. Such an approach links nature recovery to an iterative cycle of survey, prediction, intervention and further evidence collection, while retaining the ecological heterogeneity that was central to the results of the thesis.

6.6 Thesis conclusion

This thesis set out to determine how habitat structure and management within urban green spaces combine with the surrounding landscape to shape small mammal communities, and whether those relationships could support spatially explicit management and nature-recovery planning. The results show that urban green spaces cannot be understood adequately from their mapped area or broad habitat composition alone. Small mammal responses were associated with the structure and management of vegetation within patches, the subdivision and isolation of surrounding green space, and the distribution of barriers and linear habitat across the

wider matrix. Woody linear structure provided the most consistent positive local relationship, while intensive management, paths, green space subdivision and road density were associated with reduced community responses. Species and community metrics did not respond uniformly, confirming that ecological value cannot be represented through a single measure applying equally to the whole assemblage.

Integrating these relationships showed that local habitat condition and wider landscape structure contain complementary ecological information. The resulting spatial model retained useful predictive performance when complete sites were withheld and allowed management scenarios to be compared across environmentally supported green spaces. Spatial modelling therefore provided both an evidence base for comparing and targeting management within the supported network and a means of identifying where further survey is needed before ecological inference can be extended.

The wider contribution of the research is to show how an underrepresented component of urban biodiversity can inform the management of heterogeneous green space networks. Small mammal communities respond to conditions that are easily obscured by broad measures of urban greenness, including ground-level vegetation structure, mowing, internal fragmentation and surrounding landscape configuration. Urban nature recovery should retain and develop these conditions through structurally varied vegetation, less intensive grassland management where appropriate, protected woody and riparian margins, and reduced fragmentation of suitable habitat. The thesis therefore provides an empirically bounded framework for

moving from mapped green space towards ecologically informed management, while recognising that effective urban nature recovery requires continued field evidence, species-specific understanding and management adapted to the conditions of individual green spaces.

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Appendices

The appendices provide supporting documentation, biological-record data, site-management information, survey metadata and full analytical outputs referred to throughout the thesis.

Appendix A: Biological records and data provenance

Small mammal occurrence records used to establish the contextual species assemblage described in Chapter 2 were obtained from the NBN Atlas and directly from Staffordshire Ecological Record. The NBN Atlas occurrence download contained 358 records and was initially accessed in December 2020, and updated and appended on 23rd January 2026.

NBN Atlas citation and data attribution

When using these data, the NBN Atlas download should be cited as:

NBN Atlas occurrence download at NBN Atlas, accessed on 23 January 2026.

Data were contributed by the following providers and datasets:

Records provided by the People's Trust for Endangered Species, accessed through the NBN Atlas website on 23 January 2026. Further information is available from enquiries@ptes.org or the NBN Atlas provider registry: https://registry.nbnatlas.org/public/show/dp35.

Records provided by the Biological Records Centre, accessed through the NBN Atlas website on 23 January 2026. Further information is available from brc@ceh.ac.uk or the NBN Atlas provider registry: https://registry.nbnatlas.org/public/show/dp77.

Staffordshire Ecological Record (2026). *SER Species-based Surveys*. Occurrence dataset accessed through the NBN Atlas on 23 January 2026. DOI: https://doi.org/10.15468/q8qen3. Data supplied under the Creative Commons Attribution–NonCommercial 4.0 licence. Further information is available from info@staffs-ecology.org.uk or https://registry.nbnatlas.org/public/show/dr1193.

Records provided by the Mammal Society, accessed through the NBN Atlas website on 23 January 2026. Further information is available from atlas@themammalsociety.org or https://registry.nbnatlas.org/public/show/dp8.

Mammal Society (2026). *Mammal Mapper App Sighting Records*. Occurrence dataset accessed through the NBN Atlas on 23 January 2026. DOI: https://doi.org/10.15468/mjw32e. Data supplied under the Creative Commons Attribution 4.0 licence. Further information is available from atlas@themammalsociety.org or https://registry.nbnatlas.org/public/show/dr2451.

Records provided by Staffordshire Ecological Record, accessed through the NBN Atlas website on 23 January 2026. Further information is available from info@staffs-ecology.org.uk or https://registry.nbnatlas.org/public/show/dp134.

Staffordshire Ecological Record (2026). *SER Site-based Surveys*. Occurrence dataset accessed through the NBN Atlas on 23 January 2026. DOI: https://doi.org/10.15468/h2yko0. Data supplied under the Creative Commons Attribution–NonCommercial 4.0 licence. Further information is available from info@staffs-ecology.org.uk or https://registry.nbnatlas.org/public/show/dr1399.

Records provided by iSpot, accessed through the NBN Atlas website on 23 January 2026. Further information is available from https://registry.nbnatlas.org/public/show/dp267.

Staffordshire Ecological Record (2026). *Stoke-on-Trent Environmental Survey results (1982–1984)*. Occurrence dataset accessed through the NBN Atlas on 23 January 2026. DOI: https://doi.org/10.15468/8gryb6. Data supplied under the Creative Commons Attribution–NonCommercial 4.0 licence. Further information is available from info@staffs-ecology.org.uk or https://registry.nbnatlas.org/public/show/dr1414.

iSpot (2026). *iSpot British Isles observations*. Occurrence dataset accessed through the NBN Atlas on 23 January 2026. DOI: https://doi.org/10.15468/uqdj62. Data supplied under the Creative Commons Attribution–NonCommercial 4.0 licence. Further information is available from https://registry.nbnatlas.org/public/show/dr2701.

National Mammal Atlas Project, online recording (2026). Occurrence dataset accessed through the NBN Atlas on 23 January 2026. DOI: https://doi.org/10.15468/i2eosa. Data supplied under the Creative Commons Attribution–NonCommercial 4.0 licence. Further information is available from https://registry.nbnatlas.org/public/show/dr671.

People's Trust for Endangered Species (2026). *Living with Mammals survey*. Occurrence dataset accessed through the NBN Atlas on 23 January 2026. DOI: https://doi.org/10.15468/kv7kgm. Data supplied under the Creative Commons Attribution–NonCommercial 4.0 licence. Further information is available from emily.thomas@ptes.org, https://registry.nbnatlas.org/public/show/dr227, and https://www.ptes.org/lwm.

Biological Records Centre (2026). *Mammal records from Britain from the Atlas of Mammals (1993), with some subsequent records*. Occurrence dataset accessed through the NBN Atlas on 23 January 2026. DOI: https://doi.org/10.15468/1qyxis. Data supplied under the Creative Commons Attribution 4.0 licence. Further information is available from brc@ceh.ac.uk or https://registry.nbnatlas.org/public/show/dr743.

Data anonymisation

To protect sensitive or potentially identifying information, locality names, site names, grid references, coordinates, recorder and determiner names, and free-text occurrence remarks were removed from the appendix. NBN Atlas record identifiers, taxonomic information, event dates, dataset provenance, rights-holder information, licence information and verification status were retained to preserve transparency and auditability.

The Staffordshire Ecological Record extract is presented separately from the NBN Atlas download. Some records supplied by Staffordshire Ecological Record are also represented within the NBN Atlas export. The totals from the two record tables should therefore not be combined without record-level duplicate resolution.

Table A.1 Anonymised small mammal occurrence records downloaded and updated from the NBN Atlas on 23 January 2026.

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
003724c9-2349-425d-86b3-f4122cc860f5	*Microtus agrestis*	Field Vole	2021-12-31	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
005e498a-bd51-4657-8daa-241039d9cd21	*Apodemus sylvaticus*	Wood Mouse	2024-11-06	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
00e07947-e102-40d2-846f-ea4248e18528	*Microtus agrestis*	Field Vole	2022-06-12	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
011550ae-b8b7-4aad-afdb-a49bb1cd5acb	*Microtus agrestis*	Field Vole	2019-10-12	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
01c0e30d-72f5-4620-b4a4-6c5fca7132a4	*Microtus agrestis*	Field Vole	2021-10-30	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
01e5bee4-1b62-4234-bbe0-9ffec7618400	*Microtus agrestis*	Field Vole	2020-03-07	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
01fa2968-558e-46fa-9e61-b71258bf5d34	*Microtus agrestis*	Field Vole	2021-12-31	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
02726c22-81a5-486b-8342-9d977b26999c	*Sorex araneus*	Common Shrew		SER Site-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
02838da5-2da3-49e8-a980-d57a15a6245f	*Microtus agrestis*	Field Vole	2021-02-03	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
0303493c-98a3-4d80-b7aa-8d3680bedbef	*Microtus agrestis*	Field Vole	2025-08-25	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
03e9de55-8453-45e7-b6bd-3c94aa56734b	*Microtus agrestis*	Field Vole	2022-06-12	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
03f93a7d-74e9-4b66-9e05-7f5915e633ff	*Apodemus sylvaticus*	Wood Mouse	2002-10-13	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
0460350e-ec93-440a-ad5a-74fff60d76ed	*Apodemus sylvaticus*	Wood Mouse	2025-09-24	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
04677e2a-752f-46ec-bf2f-73cd3ad59f00	*Microtus agrestis*	Field Vole	2005-12-01	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
06974d72-987a-4d94-aebf-1f8a32c8c5ee	*Sorex minutus*	Pygmy Shrew		SER Site-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
076cb17b-f8f3-4b46-a1c9-9a493a02970f	*Sorex araneus*	Common Shrew	2012-09-15	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
077652d6-5509-4e63-b27f-7663bec6493d	*Microtus agrestis*	Field Vole	2004-03-25	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
09708c08-002a-4e0b-a74b-5b79fc3341f5	*Microtus agrestis*	Field Vole	2019-01-12	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
09c245b6-24de-4ac9-8f29-8e6351e4d60e	*Micromys minutus*	Harvest Mouse	2023-11-12	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
0b57ecf5-595f-42cf-aa1f-fecee30a01d0	*Apodemus sylvaticus*	Wood Mouse	2023-05-22	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
0b95f069-a25a-4323-b960-b2127589c9f5	*Apodemus sylvaticus*	Wood Mouse		Mammal records from Britain from the Atlas of Mammals (1993), with some subsequent records	Biological Records Centre	CC-BY	Accepted
0c2719fd-520b-4870-b616-2ef286dfa866	*Sorex araneus*	Common Shrew	2020-07-17	National Mammal Atlas	Mammal Society	CC-BY-NC	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
				Project, online recording			
0cb1275b-92fd-45b6-8170-05ea124dc9cd	*Microtus agrestis*	Field Vole	2025-08-20	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
0e304d08-e3df-469a-84f5-391017c2cd70	*Microtus agrestis*	Field Vole	2022-06-12	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
0f99da21-2a03-4cc6-9b10-b4fd52609698	*Microtus agrestis*	Field Vole	2015-02-12	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted
0fb5bd7b-c3b5-403b-9b2e-dc0809408503	*Microtus agrestis*	Field Vole	2021-01-28	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
100eca82-4e8e-4103-a9d0-dda31b286e8b	*Apodemus sylvaticus*	Wood Mouse	2021-08-14	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
101665a4-9ab1-4f32-a748-e2bf33c066a2	*Micromys minutus*	Harvest Mouse	2023-10-01	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
12486a14-5862-4e81-b469-82be7fa91763	*Apodemus sylvaticus*	Wood Mouse		SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
144c9ce3-e42c-45d9-b5da-5e17bcddc71a	*Microtus agrestis*	Field Vole	2022-01-12	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
1458c5d3-a440-4304-b1fd-d7faf3ada211	*Apodemus sylvaticus*	Wood Mouse	2022-08-13	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
165171d9-39d0-4bae-be6e-4d0a06926810	*Sorex araneus*	Common Shrew		Mammal records from Britain from the Atlas of Mammals (1993), with some subsequent records	Biological Records Centre	CC-BY	Accepted
165b53ba-5837-4584-9084-40d062134cc8	*Microtus agrestis*	Field Vole	2020-06-02	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
16b2dc91-caf8-41f4-aa91-0864e14b64d2	*Sorex araneus*	Common Shrew	1983-09-20	Stoke-on-Trent Environmental Survey results (1982-1984)	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
1776eb51-49e8-40f1-8685-8b47e6a3da2e	*Myodes glareolus*	Bank Vole	2025-11-13	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
1884bf94-b3bc-4d54-	*Apodemus sylvaticus*	Wood Mouse	2022-04-25	National Mammal Atlas	Mammal Society	CC-BY-NC	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
a39b-5b02b5181d5c				Project, online recording			
18ff8f57-3e0c-4674-a3c8-52c5eb9b8e86	*Myodes glareolus*	Bank Vole	2021-02-23	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
1bac8319-eaa7-471d-b617-b294a365da00	*Sorex araneus*	Common Shrew	2018-10-13	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
1bcdddbc-4878-43b3-8d79-e8d11eabfe72	*Microtus agrestis*	Field Vole	2024-07-07	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
1ca649d8-2982-405a-8b52-f2c07f3c82da	*Microtus agrestis*	Field Vole	2021-12-30	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
1f35c75a-4f2d-4a39-a5fe-e31a77d349bd	*Apodemus sylvaticus*	Wood Mouse	2019-10-12	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
1f810f1c-1584-46ab-9fa3-a2b7841c473c	*Microtus agrestis*	Field Vole	2021-01-28	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
1fcc2a24-cd72-480f-8bd3-bf3859f428aa	*Sorex araneus*	Common Shrew	1965-04-10	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
20645d65-6a7f-443a-84ae-c2f09e0e5e91	*Micromys minutus*	Harvest Mouse	2023-12-19	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
2213a1c0-b687-436a-bbba-f167270cb135	*Microtus agrestis*	Field Vole	2015-11-22	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted
23df5adf-04ca-4d5b-a5e3-331835ced035	*Microtus agrestis*	Field Vole	2024-10-10	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
261b218f-cbcd-4fd6-a265-8f33db73d8e4	*Apodemus sylvaticus*	Wood Mouse	1996-03-26	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
26b6831c-1bdc-4a51-b051-b6447c42ec31	*Apodemus sylvaticus*	Wood Mouse	2013-04-16	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
26e09fed-5cb2-4109-9b3d-1ec3ede94de6	*Apodemus sylvaticus*	Wood Mouse	2002-05-18	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
27617c60-8b93-4c63-8101-316aed0a7a08	*Apodemus sylvaticus*	Wood Mouse	2011-11-06	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
278a1b3e-7a60-4aa9-acfb-eeb1cd96f632	*Sorex araneus*	Common Shrew	2020-05-01	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
28a69497-ddc0-44c8-8173-b0d40d2bbe23	*Sorex araneus*	Common Shrew	2022-08-21	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
28b1b898-bafb-44cd-bfa4-a7e2a4118055	*Sorex araneus*	Common Shrew	1983-09-20	Stoke-on-Trent Environmental Survey results (1982-1984)	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
28f29505-9da2-4862-a15f-e51ac561a851	*Microtus agrestis*	Field Vole	2021-03-17	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
29ca81f0-1cc2-4d7d-9ff0-290c87bca752	*Microtus agrestis*	Field Vole	2021-12-31	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
2a3fdd44-49e1-46c3-b468-b8b8df599089	*Myodes glareolus*	Bank Vole	2013-04-26	iSpot British Isles observations	iSpot	CC-BY-NC	Unconfirmed - not reviewed
2adfcbea-e9ef-49d4-967b-db03482e2d20	*Microtus agrestis*	Field Vole	2021-11-02	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
2bba7b0c-907a-42b3-9778-3ffcb34c51e7	*Apodemus sylvaticus*	Wood Mouse	2023-05-24	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
2c50f0c6-7d08-4393-a7fd-6f461ad83a2d	*Myodes glareolus*	Bank Vole	2022-09-18	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
2decf8d1-e3b8-4e31-8abd-9d1a42166b2e	*Microtus agrestis*	Field Vole	2025-05-17	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
2e4ce88e-853d-4503-b760-b509afa11929	*Microtus agrestis*	Field Vole	2021-02-01	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
2ea4f4f5-7f0f-4dbe-83b4-1b44cdb00897	*Apodemus sylvaticus*	Wood Mouse	2020-10-24	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
2f94df93-fa0c-4be7-b0d5-49a27e90df3d	*Apodemus sylvaticus*	Wood Mouse	2009-04-19	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
3086e26b-db66-470e-b37a-3b455a5d7fb0	*Apodemus sylvaticus*	Wood Mouse	2002-04-13	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
3109c12b-bcdc-42c0-ac4f-0f841cb1d074	*Apodemus sylvaticus*	Wood Mouse	2022-08-21	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
31c06ea0-8bb5-4af0-b512-ff9e5a6e7480	*Microtus agrestis*	Field Vole	2024-09-24	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
32784ee2-3f86-4714-be35-1810803a1188	*Sorex araneus*	Common Shrew		SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
33e62d8e-a0f8-4936-9792-7349c404e3e5	*Sorex araneus*	Common Shrew	1965-04-10	Mammal records from Britain from the Atlas of Mammals (1993), with some subsequent records	Biological Records Centre	CC-BY	Accepted
34297d96-c08b-47f3-8555-5db633a0adac	*Micromys minutus*	Harvest Mouse	2021-01-10	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
34bde9b9-cbdc-4bff-918e-2ffdbdd297aa	*Sorex araneus*	Common Shrew	2016-08-20	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
36bb35e2-2700-4f54-a28e-d156441fc50b	*Apodemus sylvaticus*	Wood Mouse	2021-06-13	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
37b0863f-4757-45c5-8bed-4c2f1b80a070	*Apodemus sylvaticus*	Wood Mouse	2022-07-19	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
38f59117-c2c6-4951-a13e-881f16e2ba20	*Sorex araneus*	Common Shrew	2022-08-29	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
393787b6-13b5-4f61-ba6c-916b3280837d	*Sorex minutus*	Pygmy Shrew	1984-06-19	Stoke-on-Trent Environmental Survey results (1982-1984)	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
39c6752c-7f9f-4271-9c91-2305c7149a96	*Micromys minutus*	Harvest Mouse	2022-12-14	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - considered correct
3a16f5ba-6246-44cf-92a8-5cb90e9d6376	*Microtus agrestis*	Field Vole		Mammal records from Britain from the Atlas of Mammals (1993), with some subsequent records	Biological Records Centre	CC-BY	Accepted
3ab7a31c-1127-4667-90f3-266d9c492476	*Microtus agrestis*	Field Vole	2006-07-23	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
3dbf421d-05eb-43c4-ac26-78ea77581a0e	*Myodes glareolus*	Bank Vole	2022-08-21	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
3e26d53c-49b1-40a8-a239-a6c7aaea4934	*Micromys minutus*	Harvest Mouse	2023-08-06	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
3e2ac1e7-2c8e-4be8-	*Microtus agrestis*	Field Vole	2021-11-02	National Mammal Atlas	Mammal Society	CC-BY-NC	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
85c1-fed99061cb54				Project, online recording			
3eedd0c1-4e8f-48bb-8fd0-1c7e3cd7014c	*Sorex araneus*	Common Shrew	2024-02-15	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
3f4c2aff-01c6-4dbb-8327-3446711576ba	*Sorex araneus*	Common Shrew	2018-06-27	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
3fa9807e-0979-46ab-bdcc-cbd86bb0c5c5	*Sorex araneus*	Common Shrew	2016-08-20	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
406eb136-7a6c-40fa-8138-6803678e9326	*Sorex minutus*	Pygmy Shrew	2013-07-05	SER Site-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
4185bcc2-2a65-4e43-9403-47e0f2969256	*Microtus agrestis*	Field Vole	2025-04-25	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
422bd4aa-cb87-430c-8ef2-9d889415f8da	*Apodemus sylvaticus*	Wood Mouse	1960-01-01	Mammal records from Britain from the Atlas of Mammals (1993), with some subsequent records	Biological Records Centre	CC-BY	Accepted

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
423bae73-d1d2-450b-988b-1db74a82c8e1	*Myodes glareolus*	Bank Vole	2021-02-04	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
428d7a18-dbda-48a4-b1de-fff51a2aff95	*Sorex araneus*	Common Shrew	2006-06-17	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
44aee5df-c126-4f8d-b2ba-c13424834acb	*Micromys minutus*	Harvest Mouse	2021-11-11	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
45f9b6ab-fe67-450b-899d-b9f7b2e880b5	*Sorex araneus*	Common Shrew	2019-10-12	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
466d50e1-989b-4be8-97ef-fba7e408139e	*Microtus agrestis*	Field Vole	2022-01-10	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
469a14cb-30c3-4d29-b5a0-9f361e55291f	*Myodes glareolus*	Bank Vole	2018-08-18	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
46ab7376-5c86-4343-8e37-53b4af301b34	*Sorex araneus*	Common Shrew	2024-02-01	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
46d0bdb8-13fc-4ea1-bd30-9540b0bfeefe	*Apodemus sylvaticus*	Wood Mouse	2018-08-27	National Mammal Atlas	Mammal Society	CC-BY-NC	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
				Project, online recording			
477cba40-a20d-40b0-a0ed-f6d6b8298a92	*Apodemus sylvaticus*	Wood Mouse	1983-09-12	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
47cfd7eb-8738-47f0-8dfc-a85adc62fdd4	*Apodemus sylvaticus*	Wood Mouse	2023-05-10	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
487f0601-4414-4bc6-8b5d-17afd4c6c0ac	*Microtus agrestis*	Field Vole	2006-11-30	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
4923105e-06f6-47f8-8903-59b60449aec9	*Sorex araneus*	Common Shrew	2023-07-16	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
4944658d-7f81-40d5-a7a4-520499393f4f	*Apodemus sylvaticus*	Wood Mouse	2018-10-13	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
49b2f507-95d7-45d2-8cf9-c05166f3288d	*Microtus agrestis*	Field Vole	2019-02-16	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
4a0178aa-4e19-4741-a09a-14c858f83c8f	*Microtus agrestis*	Field Vole	2025-10-20	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
4a9f7330-6b41-48c9-950a-b8ba950aaae1	*Microtus agrestis*	Field Vole	2021-09-01	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
4b9f5672-3fff-4648-96ba-5436e9e6c7dd	*Myodes glareolus*	Bank Vole	2012-09-15	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
4bbc9cdc-f37d-4a5b-bd6b-1cc43b737759	*Sorex araneus*	Common Shrew	2021-09-02	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
4d59d966-300f-47a7-afdc-52b4cfdefeba	*Sorex araneus*	Common Shrew	2024-09-25	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
4d8112ae-6b51-4899-954a-485e13d41608	*Sorex araneus*	Common Shrew		SER Site-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
4d858878-779a-4e0d-80b8-797d6f03a35b	*Microtus agrestis*	Field Vole	2024-09-27	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
4da8346a-377b-442e-8c82-b05951418555	*Apodemus sylvaticus*	Wood Mouse		SER Site-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
4e19fcde-6fcf-45db-9e42-f6f66763930c	*Micromys minutus*	Harvest Mouse	2022-02-22	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
4e298e2a-9fdd-4b92-a7fd-5f8fdaef6625	*Apodemus sylvaticus*	Wood Mouse	2007-01-14	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
4ec5c4ea-4b83-42e6-b806-fe72c32d5789	*Apodemus sylvaticus*	Wood Mouse	2022-05-21	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
4f814ced-1431-4b19-819a-1534a8ace6e7	*Apodemus sylvaticus*	Wood Mouse		SER Site-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
4ff8e89b-92db-4f8f-a43a-61d25e6e5785	*Microtus agrestis*	Field Vole	2025-05-29	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
50cfc40a-6075-4a8e-bb9c-5960dd244689	*Sorex araneus*	Common Shrew		Mammal records from Britain from the Atlas of Mammals (1993), with some subsequent records	Biological Records Centre	CC-BY	Accepted
50d55c8b-ea77-412b-8606-19fd021d0547	*Microtus agrestis*	Field Vole	2024-09-27	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
5131460b-8b56-4b74-ba30-c7ec08855e1f	*Myodes glareolus*	Bank Vole	2006-06-17	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
525d1764-0ee7-4f81-bd53-82408f5b1323	*Microtus agrestis*	Field Vole	2021-03-18	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
53f8df31-12a6-4c51-ba9a-036676cb3283	*Apodemus sylvaticus*	Wood Mouse	2019-04-13	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - considered correct
54371a36-996e-4612-9d7b-6a62fef83f82	*Sorex araneus*	Common Shrew	1984-04-13	Stoke-on-Trent Environmental Survey results (1982-1984)	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
544604db-f5bb-42e6-b364-3f8685f2a2a3	*Myodes glareolus*	Bank Vole	2020-02-22	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - considered correct
54d6b7c4-b7ea-4d0a-8d55-c457e5f027de	*Apodemus sylvaticus*	Wood Mouse	2007-02-18	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
55d621fc-4d32-4da9-8f17-4258490b83d2	*Microtus agrestis*	Field Vole		SER Site-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
570d7d54-89a6-4b2f-b560-65b9e36f68c1	*Microtus agrestis*	Field Vole	2021-12-30	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
5812b304-6685-42c6-9581-fdadad963c8f	*Apodemus sylvaticus*	Wood Mouse	2021-09-08	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
58afd96a-1311-4877-9544-d73b5b494da8	*Apodemus sylvaticus*	Wood Mouse	2006-06-17	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
5a5ef7e9-b18b-4a21-bfb5-09df0f61b93d	*Microtus agrestis*	Field Vole		Mammal records from Britain from the Atlas of Mammals (1993), with some subsequent records	Biological Records Centre	CC-BY	Accepted
5bb3557a-1bda-4711-810b-ffbadbfe9134	*Microtus agrestis*	Field Vole	2025-09-15	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
5caabc63-8809-4ad5-83e1-aaf27a4e19b8	*Sorex araneus*	Common Shrew	2020-08-02	National Mammal Atlas Project, online recording	Mammal Society	CC-BY	Accepted - correct
5cd46f18-a071-4937-9cfc-462752726cd1	*Mus musculus*	House Mouse	2023-04-17	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
5cd891a6-df18-4f52-a00e-5e37eef304c9	*Microtus agrestis*	Field Vole	2021-01-28	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
5d41a8d7-dc66-4604-	*Sorex araneus*	Common Shrew	2023-07-16	National Mammal Atlas	Mammal Society	CC-BY-NC	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
99f0-dd519a5d1cd5				Project, online recording			
5e2d8801-123d-4133-8c1a-858cf5e6c55a	*Microtus agrestis*	Field Vole	2012-09-15	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
5e55fb88-c3ef-4a96-ab22-0f65a93c0a65	*Microtus agrestis*	Field Vole	2021-01-19	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
5eeaf687-afe0-4f35-8554-703680fb1a3c	*Sorex minutus*	Pygmy Shrew	1960-01-01	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
60e923b3-78bd-4aca-9763-7a5a5b2d945b	*Apodemus sylvaticus*	Wood Mouse	2002-12-14	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
61928acb-0f43-4434-b24d-4c0f4462557e	*Mus*	Mouse	2023-10-21	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - considered correct
6285a01c-8fdd-4ceb-bdb6-25a25684bad4	*Apodemus sylvaticus*	Wood Mouse	2023-04-17	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
65fd329c-e440-40ab-89f4-d7d9db19b12a	*Sorex araneus*	Common Shrew		SER Site-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
674db188-b413-4f36-	*Myodes glareolus*	Bank Vole		Mammal records from Britain from the	Biological Records Centre	CC-BY	Accepted

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
a537-7da91086f80b				Atlas of Mammals (1993), with some subsequent records			
676d7fda-a146-47de-a6f4-2b54ff315a42	*Apodemus sylvaticus*	Wood Mouse	2025-05-17	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
67fa5e6b-7920-47cb-8cd3-67cf81facf2f	*Microtus agrestis*	Field Vole	2021-10-16	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
69442e65-8262-4fc1-8d6c-50a48ed14a07	*Apodemus sylvaticus*	Wood Mouse	2021-03-26	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
69940262-86bf-49e1-9405-24f69aeeaf20	*Apodemus sylvaticus*	Wood Mouse	2018-08-18	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
6a66f50f-c5c6-4eba-b41a-adbda74eb4cd	*Sorex araneus*	Common Shrew	2021-05-02	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
6a71c16f-7f87-44c4-82bd-2aef31b1edef	*Apodemus sylvaticus*	Wood Mouse	2006-04-12	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
6a790013-b1b1-4595-bca0-bee5e2c3b308	*Microtus agrestis*	Field Vole	2024-07-09	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
6b828c92-4e83-4c7c-9424-ab27b46232f0	*Sorex araneus*	Common Shrew	1975-09-09	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
6cd69e2d-d5ac-40bb-ac74-3630051bef07	*Sorex araneus*	Common Shrew	2021-11-21	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
6ced9bd5-0617-4d04-8c23-fb35de4d909a	*Apodemus sylvaticus*	Wood Mouse	2021-10-07	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
6d2c968c-a4cd-4d68-9f18-b22d098f53d6	*Microtus agrestis*	Field Vole	2021-10-25	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
6daa1d2c-8045-4234-bd76-f01bda3fd7ce	*Microtus agrestis*	Field Vole	2016-08-20	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
6e44109f-9f0c-499f-b659-9c5a5be626c6	*Sorex minutus*	Pygmy Shrew	2013-08-22	SER Site-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
6e4de4e7-6974-4a80-9af9-499fac614ae1	*Apodemus sylvaticus*	Wood Mouse	2022-06-25	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
6f0261b2-add7-4a11-a6b8-925baf42bfe7	*Myodes glareolus*	Bank Vole	1984-06-07	Stoke-on-Trent Environmental Survey results (1982-1984)	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
6f0813eb-eab3-410e-975b-5e9ab3635246	*Myodes glareolus*	Bank Vole	2021-02-23	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
6f8473d1-bfdd-4702-ab8d-48f2c93d00dd	*Apodemus sylvaticus*	Wood Mouse	2013-04-15	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
6fb7e04f-8111-4ab4-8b1e-9d5a13127c15	*Sorex*		1982-08-10	Stoke-on-Trent Environmental Survey results (1982-1984)	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
703871bc-d537-46d2-949c-0f615c0ddbd0	*Microtus agrestis*	Field Vole	2019-01-13	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
70a98943-5bb0-46df-a8e2-fc34fe442973	*Myodes glareolus*	Bank Vole	2021-06-13	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
70b9d637-3709-416d-ba5b-ae779917f2e6	*Microtus agrestis*	Field Vole	2021-05-30	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - considered correct
722530e1-11e1-4d86-91d2-ca41ff283572	*Myodes glareolus*	Bank Vole	2014-08-04	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
7227e789-022f-409d-a654-59a966a386ec	*Apodemus sylvaticus*	Wood Mouse	2024-06-06	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
72673d3d-87b0-462a-af25-a788aec22909	*Sorex araneus*	Common Shrew	2023-08-19	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
7315a198-cdfa-4d39-9700-de3b9f8c0f36	*Apodemus sylvaticus*	Wood Mouse	2007-04-25	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
731b0e38-ff27-408e-986a-b93b48ce14e7	*Sorex araneus*	Common Shrew	2025-08-27	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
735e0a56-4793-46cc-9aab-0a21453320d4	*Apodemus sylvaticus*	Wood Mouse	2016-08-20	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
74287fb5-06c3-4e14-b1e1-1ecfb15312e7	*Apodemus sylvaticus*	Wood Mouse	2012-09-15	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
75820bb1-6b6e-4390-a154-be0efdfbc11f	*Myodes glareolus*	Bank Vole	2009-04-19	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
75ded1ee-a957-4458-b051-3eb46e2933e6	*Micromys minutus*	Harvest Mouse	2007-01-14	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
76c7bac2-2188-4ea5-b069-fddc1bb076f9	*Microtus agrestis*	Field Vole	2009-04-19	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
76fb2865-93b6-48cf-9d7b-defd9b948105	*Microtus agrestis*	Field Vole	2021-04-08	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
77314bc0-f0a9-4fd3-83d5-1a5c76c35043	*Myodes glareolus*	Bank Vole	2006-06-17	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
77442a57-d440-4eb2-a12f-2058ed17be03	*Apodemus sylvaticus*	Wood Mouse	2007-03-19	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
7788dbbb-836b-40db-91d6-408251d8a422	*Sorex araneus*	Common Shrew	2024-07-31	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
784740f5-db98-4a43-a01e-62b11b266a0e	*Micromys minutus*	Harvest Mouse	2020-12-13	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
784c1665-a873-49f4-9d84-02a812bec260	*Apodemus sylvaticus*	Wood Mouse	2021-03-28	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
79d4a2e8-5e5b-46a3-ad19-de888c02930e	*Sorex araneus*	Common Shrew	2016-08-20	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
7a168537-470c-4051-8fba-936cc661198c	*Sorex araneus*	Common Shrew	2018-08-11	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
7de1df1e-7960-414f-83f5-9bd81b9b7e81	*Sorex araneus*	Common Shrew	2022-08-13	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
7f491970-026d-4d5d-8f05-04d205d747f8	*Myodes glareolus*	Bank Vole	2023-03-30	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
808eeb52-eeb7-466e-aea6-f6446eb7ce8d	*Myodes glareolus*	Bank Vole	2007-01-16	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
8191c087-6779-4fb3-a995-270f212841b7	*Microtus agrestis*	Field Vole	2024-02-17	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
82b7bead-8cab-4b81-96e3-ee8f546b336e	*Myodes glareolus*	Bank Vole	2024-07-07	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
843a5d35-599a-4e6a-920a-5fa7d6ed4165	*Myodes glareolus*	Bank Vole	2006-06-17	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
84a42f4c-5f1d-4932-9c60-173fb238002a	*Microtus agrestis*	Field Vole		SER Site-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
856508ca-949b-48fc-b3a2-e96c8b9b2ffb	*Apodemus sylvaticus*	Wood Mouse	2006-09-25	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
85ee8501-b1be-4154-bead-c5c34f62201f	*Apodemus sylvaticus*	Wood Mouse	2009-04-19	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
8697b6b3-c013-4586-8bb4-fdd755facb6c	*Microtus agrestis*	Field Vole	2021-11-30	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
86fd89a0-ca8e-42e3-9156-a7bbbe506da8	*Apodemus sylvaticus*	Wood Mouse	2021-11-21	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
87805e73-815b-4c2b-b3c8-b4708b908791	*Sorex araneus*	Common Shrew	1982-09-01	Stoke-on-Trent Environmental Survey results (1982-1984)	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
87e74cad-71fd-48a7-a8aa-ec9a9b48a7fb	*Micromys minutus*	Harvest Mouse	2009-10-09	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
887f1fc6-366f-49ad-8468-f4da8ff99b43	*Apodemus sylvaticus*	Wood Mouse	2004-03-25	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
8a2bc3de-886f-4068-aa7c-0b5984c1f490	*Microtus agrestis*	Field Vole	2004-04-19	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
8a4a1c86-4f3f-4577-9e4c-98219ac52f45	*Apodemus sylvaticus*	Wood Mouse	2023-08-19	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
8a5b46c1-79db-4942-89bf-6a6d47c419be	*Microtus agrestis*	Field Vole	2021-04-07	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
8ab1f567-d229-4b5a-a444-9d4dd74663ca	*Microtus agrestis*	Field Vole	2022-10-09	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
8bd2b191-50e0-4238-9ac6-8fd612b061d6	*Myodes glareolus*	Bank Vole	2006-06-17	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
8e072936-31c9-4598-9147-f440320857c8	*Myodes glareolus*	Bank Vole	2022-01-12	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
8edd9758-c950-4a92-801c-f56492d36516	*Myodes glareolus*	Bank Vole	2016-08-20	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
8ef03168-7ed1-4cff-9fec-65239d95fc03	*Microtus agrestis*	Field Vole	2021-11-15	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
8f01c4bd-9e15-4253-8d8f-8cd5b8fccca6	*Microtus agrestis*	Field Vole		Mammal records from Britain from the	Biological Records Centre	CC-BY	Accepted

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
				Atlas of Mammals (1993), with some subsequent records			
8f7c66fc-f39a-446c-bf64-fc4aee1c2dfd	*Micromys minutus*	Harvest Mouse	2023-11-12	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
90a6efe7-8a4c-44c9-9b66-8273dcbee0a5	*Sorex minutus*	Pygmy Shrew		Mammal records from Britain from the Atlas of Mammals (1993), with some subsequent records	Biological Records Centre	CC-BY	Accepted
90d1e2c3-5fb3-4f4f-9e4b-40a1a8471388	*Mus musculus*	House Mouse	2021-05-10	Living with Mammals survey	People's Trust for Endangered Species	CC-BY-NC	Unconfirmed - not reviewed
91ecdb69-d9c2-4ca7-bd05-b00218b0e07a	*Apodemus sylvaticus*	Wood Mouse	2018-07-29	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
921d336b-d7aa-4c14-98fe-ada60525bcc3	*Myodes glareolus*	Bank Vole	2025-09-24	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
923d90a9-a7a8-40d9-af8d-5e9831ae03ce	*Apodemus sylvaticus*	Wood Mouse		Mammal records from Britain from the Atlas of Mammals (1993), with some subsequent records	Biological Records Centre	CC-BY	Accepted
925e529a-e66b-481a-8eae-8506fb998658	*Mus musculus*	House Mouse	1983-03-23	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
937b344c-be0c-4e46-a6b6-4ed5c37583ba	*Microtus agrestis*	Field Vole	2013-08-22	SER Site-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
93afcbe7-6d20-4b52-9acf-bf8ef004b69c	*Microtus agrestis*	Field Vole	2021-10-25	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
944005df-cdf6-4e6e-9435-b5f540cad154	*Microtus agrestis*	Field Vole	2020-12-13	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
954291cb-55a6-44df-9a94-92051f184422	*Microtus agrestis*	Field Vole	2025-09-11	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
958b9807-f24f-4e94-87cc-d9fb21f40719	*Sorex minutus*	Pygmy Shrew	2025-08-05	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
95f407aa-1a1d-4506-a276-1e47fa7bc3da	*Sorex araneus*	Common Shrew	1982-07-05	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
97d26934-cd46-46a3-ab8f-4a63818e5a47	*Micromys minutus*	Harvest Mouse	2024-03-05	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - considered correct
980de5cc-5618-43e4-aaec-6a1ac64e1652	*Sorex minutus*	Pygmy Shrew	1984-06-19	Stoke-on-Trent Environmental Survey results (1982-1984)	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
987581e9-0de6-4d84-8524-390538ed422d	*Sorex araneus*	Common Shrew	2021-09-08	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
9879995d-24aa-4f93-8fe0-d63de8d2b3b3	*Apodemus sylvaticus*	Wood Mouse	2023-04-28	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
9890b214-332a-4406-9938-baaeedbc2a1c	*Apodemus sylvaticus*	Wood Mouse	2021-11-21	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
9acd14aa-7976-4e7e-aaab-1fb862a7f145	*Microtus agrestis*	Field Vole	2022-06-12	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
9b470167-1ce9-4e6b-93f5-e50a2a5acc92	*Microtus agrestis*	Field Vole	2022-06-12	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
9bc73a0a-6b02-4fc1-8ea2-2c3392799f51	*Apodemus sylvaticus*	Wood Mouse	2016-08-20	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
9be98c6a-89fa-4fa2-b8d3-c7bb6381f149	*Apodemus sylvaticus*	Wood Mouse	2025-08-13	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
9c02037e-1e2a-4ba4-991a-bd88bf421733	*Myodes glareolus*	Bank Vole	2021-12-02	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
9cf18018-a0bc-4e6d-acfc-4d3efcb1228d	*Apodemus sylvaticus*	Wood Mouse	2006-08-15	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
9e9e693d-787c-4e8c-ad01-178a203dc331	*Myodes glareolus*	Bank Vole	2018-08-11	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
9edff219-98e9-4413-b24b-d8ce05ce9e14	*Microtus agrestis*	Field Vole	2021-11-27	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
9ee3eff8-254f-4e19-b4db-f7f328c77919	*Apodemus sylvaticus*	Wood Mouse	2024-11-05	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - considered correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
9ef6e4f6-6dbc-4792-b42f-ac5092c87e14	*Sorex araneus*	Common Shrew	2006-07-09	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
9faeb7e6-2ff9-4053-a428-573a22893c92	*Micromys minutus*	Harvest Mouse	2019-12-31	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
9fe3f178-db57-4590-b30d-ef07034d5d0f	*Apodemus sylvaticus*	Wood Mouse	2006-06-17	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
a0c17540-7c96-45aa-b018-73e57226b7c4	*Apodemus sylvaticus*	Wood Mouse	2005-09-26	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
a15860e8-8687-4dc8-a113-76fd2c942781	*Microtus agrestis*	Field Vole	2021-09-02	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
a16c1764-27c8-48e8-9fa7-0237623e6e93	*Myodes glareolus*	Bank Vole	2023-08-19	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
a175b910-735c-46a1-b271-e5931479cfd8	*Sorex araneus*	Common Shrew	2024-10-10	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
a1cdd8f0-f634-4292-a1fb-53b68f797948	*Myodes glareolus*	Bank Vole	2013-04-26	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
a1f14fbc-ef61-4c13-ba29-303768ec91cc	*Micromys minutus*	Harvest Mouse	2021-01-06	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
a2020934-ebf4-4649-baa1-ff0700a6db91	*Sorex araneus*	Common Shrew	1983-10-05	Stoke-on-Trent Environmental Survey results (1982-1984)	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
a222c62f-5319-4610-a82c-5dc1e9c05ae6	*Microtus agrestis*	Field Vole	2021-11-15	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
a267d6d0-552d-4c2a-9625-14f92345ca70	*Apodemus sylvaticus*	Wood Mouse	2021-10-30	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
a2d6d38f-149c-4f7a-b043-6421e8106d41	*Micromys minutus*	Harvest Mouse	2024-03-05	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
a44e8f06-4140-4f11-a56f-999fb1d52417	*Apodemus sylvaticus*	Wood Mouse	2022-06-22	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
a458c8fa-0b5a-483c-a6ac-8e9f76d5a154	*Mus musculus*	House Mouse	1983-03-23	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
a53b9c19-c4f7-4c93-8f1c-b2fd013ef837	*Microtus agrestis*	Field Vole	2024-08-01	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
a57d9e89-a7ac-47cc-b3cd-e763554f6132	*Apodemus sylvaticus*	Wood Mouse	2021-03-27	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
a773c1de-6cb2-4436-aeaa-a84aa7d17840	*Apodemus sylvaticus*	Wood Mouse		Mammal records from Britain from the Atlas of Mammals (1993), with some subsequent records	Biological Records Centre	CC-BY	Accepted
a7b239d0-4832-4690-bb0e-30592926d91a	*Sorex araneus*	Common Shrew	1965-04-10	Mammal records from Britain from the Atlas of Mammals (1993), with some subsequent records	Biological Records Centre	CC-BY	Accepted
ac62078a-51b3-49ab-a7e6-a4f6ae4c49a3	*Microtus agrestis*	Field Vole	2020-02-22	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - considered correct
aca20f0a-1c9c-4e3e-9844-7974e30141cc	*Apodemus sylvaticus*	Wood Mouse	2021-07-14	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
ad704e66-4ed1-4fb0-ab44-2d21c76496dd	*Microtus agrestis*	Field Vole	2022-01-10	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
adf24295-f733-4f09-894e-1c0877fce426	*Micromys minutus*	Harvest Mouse	2004-03-25	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
adfa15ff-cea1-40d1-98a5-d9e48a6e3d6f	*Microtus agrestis*	Field Vole	2007-01-14	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
ae087378-84cd-4b9d-8dcb-ca646a091bac	*Apodemus sylvaticus*	Wood Mouse	2007-01-06	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
ae23e0cd-bdc6-4c16-8967-0ec369e9a683	*Sorex araneus*	Common Shrew	1965-04-10	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
ae8b27c1-0bd5-43b9-9ebc-58a61f55c292	*Apodemus sylvaticus*	Wood Mouse	2023-05-10	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
aefb0868-e18a-47d6-a6aa-cbc4b700f538	*Microtus agrestis*	Field Vole	2021-12-30	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
af29d739-9f24-4cdf-a14a-f6792ca5bc46	*Apodemus sylvaticus*	Wood Mouse	2025-08-19	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
b28b3ef1-3461-44e2-b06a-860ec8eb2e2b	*Micromys minutus*	Harvest Mouse	2023-11-12	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
b31f25c4-f827-45ca-afa6-d73384a9916f	*Microtus agrestis*	Field Vole	2021-12-29	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
b3683bcc-dd75-452b-a92a-d1eeaf1851e6	*Apodemus sylvaticus*	Wood Mouse		SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
b3ad8e28-6449-455d-a3c7-d133bd01f52e	*Apodemus sylvaticus*	Wood Mouse	2022-09-18	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
b42ed292-61d0-489e-adc4-4ee0dc7effa0	*Myodes glareolus*	Bank Vole	2019-10-12	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
b4e07ac5-21d6-418d-9334-dc7d3007f2c5	*Myodes glareolus*	Bank Vole	2021-03-28	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
b54bf302-7932-44db-a2f6-c90e49d084ca	*Sorex minutus*	Pygmy Shrew	1996-08-24	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
b58dc744-2840-436f-b47b-902aee7e9c7f	*Microtus agrestis*	Field Vole		Mammal records from Britain from the Atlas of	Biological Records Centre	CC-BY	Accepted

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
				Mammals (1993), with some subsequent records			
b66c2220-7330-41d5-a062-6d78cf0cc3e8	*Apodemus sylvaticus*	Wood Mouse	2018-08-12	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
b6920311-3a47-4a92-8156-278d324c303f	*Myodes glareolus*	Bank Vole	2021-06-13	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
b72a3a3e-75a5-4116-97bd-82717c922b5d	*Microtus agrestis*	Field Vole	2020-04-03	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
b8327e96-8ae5-4f61-abb2-904fefbae782	*Apodemus sylvaticus*	Wood Mouse	2006-06-17	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
b8ab9d67-a441-4508-9515-8537e07cac49	*Apodemus sylvaticus*	Wood Mouse	2022-05-20	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
b9fee496-c19a-4978-8989-ee38087d0ead	*Sorex araneus*	Common Shrew	1964-01-01	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
ba2d0c31-fbb6-4d71-	*Microtus agrestis*	Field Vole	2023-11-23	Mammal Mapper App	Mammal Society	CC-BY	Accepted - considered correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
ab78-6845d66729d0				Sighting Records			
baa3e198-ecbe-4af1-b3b8-daa9388bd434	*Myodes glareolus*	Bank Vole	2021-10-30	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
bb7e1313-711c-4403-99c2-f317fe5aa0ca	*Microtus agrestis*	Field Vole		SER Site-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
bc176b57-a63a-4e08-a2cd-5cdb6dcd7df4	*Apodemus sylvaticus*	Wood Mouse	2018-08-11	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
be83323a-93d9-4b67-9f83-83f09fe8f2ef	*Apodemus sylvaticus*	Wood Mouse	1993-08-31	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
bf197ac7-84bc-461e-b53c-e228b1fcd45d	*Microtus agrestis*	Field Vole	2002-05-26	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
bf36dde8-1646-46fc-97c3-926bc00dd68b	*Apodemus sylvaticus*	Wood Mouse		SER Site-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
bf41484b-ed8f-4580-9b6b-835956fda365	*Myodes glareolus*	Bank Vole	2014-06-20	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
bfbd0e0a-9bc3-4d7a-b2a2-244fde24bdb1	*Microtus agrestis*	Field Vole	2018-08-12	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
bff57687-54ed-4ec7-81cc-bda82b7ce960	*Apodemus sylvaticus*	Wood Mouse	2021-10-16	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
c2272b6d-5414-46d7-978b-476dadfcee21	*Sorex araneus*	Common Shrew	2006-11-30	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
c3e5fa87-89cc-49f8-afaf-4077531588e0	*Myodes glareolus*	Bank Vole	2003-10-19	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
c4ddaa26-42d4-424d-bb3b-ffba556e0f44	*Apodemus sylvaticus*	Wood Mouse	2006-06-17	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
c64546d6-705e-4d71-9cd4-7595b3bb606e	*Sorex araneus*	Common Shrew	1983-10-31	Stoke-on-Trent Environmental Survey results (1982-1984)	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
c7017d40-3cb2-40d7-8613-56c7ac5c31a7	*Micromys minutus*	Harvest Mouse	2021-09-11	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
c7e00ee6-a4c7-4296-a396-4ad25f509266	*Microtus agrestis*	Field Vole		SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
c8050721-647c-4000-aaf8-4b1f29f0ce76	*Sorex araneus*	Common Shrew	2002-07-20	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
c8b6ec0d-d9bf-478a-a036-843ddc6395e0	*Sorex minutus*	Pygmy Shrew	2023-03-23	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
c928b3b9-61ea-48ef-af34-6b02c5d8f4c8	*Apodemus sylvaticus*	Wood Mouse	2024-05-07	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
c94069b3-1f48-464f-9419-33b535cf3932	*Sorex araneus*	Common Shrew	2013-07-05	SER Site-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
c96cc1b7-7a3b-410c-9e10-ee9f321b4e9e	*Microtus agrestis*	Field Vole	2025-09-03	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
c97b9e9d-ab74-4f7e-863a-20f743b2cef7	*Microtus agrestis*	Field Vole	2021-10-25	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
ca25200a-222a-4427-8cfd-c7acbad02279	*Microtus agrestis*	Field Vole	2014-07-31	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted
ca98bf81-98b5-422c-b059-83170ed2518c	*Myodes glareolus*	Bank Vole	2016-08-20	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
cb0bd35c-0439-4daa-8b7b-f33257e29bed	*Microtus agrestis*	Field Vole	2021-03-19	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
cb3aa4d1-255f-4c27-8bd7-d52cad04c6fd	*Apodemus sylvaticus*	Wood Mouse	2007-04-25	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
cb8b5ab7-9a90-43a8-bd55-4505e90ee10f	*Apodemus sylvaticus*	Wood Mouse	1983-06-19	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
ce0cb1a2-8504-4cf4-ab03-59f175b29ff8	*Apodemus sylvaticus*	Wood Mouse	2023-09-11	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - considered correct
cea02fee-9dcd-4f6f-9ebc-df056f869e8e	*Microtus agrestis*	Field Vole	2020-02-22	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - considered correct
d12b192b-2e2d-420f-bf5d-754d94546d85	*Apodemus sylvaticus*	Wood Mouse	2016-08-20	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
d1e0c748-e424-42d9-a789-ac91f53c3953	*Apodemus sylvaticus*	Wood Mouse	2023-11-15	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
d234f390-d8c5-489c-abdb-047750e6f8c7	*Sorex minutus*	Pygmy Shrew	2020-11-02	Living with Mammals survey	People's Trust for Endangered Species	CC-BY-NC	Unconfirmed - not reviewed
d429af70-8a20-4f81-8ffe-a0616b5abd16	*Sorex araneus*	Common Shrew	1983-08-08	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
d456a90a-2bdc-4b23-b763-8a1391c5f3fd	*Apodemus sylvaticus*	Wood Mouse	2024-09-28	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - considered correct
d613e39c-df86-4816-a6b4-1f19f5bcd7db	*Sorex minutus*	Pygmy Shrew	2021-03-28	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
d9615f5e-b10a-428d-9672-fd0101efcafe	*Sorex araneus*	Common Shrew	2010-10-08	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
daae71dc-96b3-4be4-97dd-3659be66533c	*Micromys minutus*	Harvest Mouse	2021-11-02	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
db9ce1b1-9e20-45c7-acfa-8992656a1d31	*Microtus agrestis*	Field Vole	2021-03-17	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
dcaadd99-d329-41ef-92a1-1d463090a5c1	*Sorex araneus*	Common Shrew	2020-08-30	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
dd6a6ac2-2945-42aa-bdb6-e42c7346183b	*Myodes glareolus*	Bank Vole		SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
df206a0a-5819-470a-8d84-9bbfe6c8c0b3	*Apodemus sylvaticus*	Wood Mouse	2024-07-06	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
df7af5f9-254a-4dd3-bddb-7e28fe5c9be6	*Sorex araneus*	Common Shrew	2006-06-17	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
df993c33-570f-489a-b4f2-9fb9d1851217	*Sorex minutus*	Pygmy Shrew		SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
dfa5a6d5-85b2-43aa-aae9-047d3981b89d	*Sorex araneus*	Common Shrew	2013-08-22	SER Site-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
dfb3fc06-4496-4f8e-82df-25ca08fec5b0	*Microtus agrestis*	Field Vole	2021-12-30	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - considered correct
e0137539-ff58-4fde-bc58-43c2c2bada8a	*Apodemus sylvaticus*	Wood Mouse	2002-04-24	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
e0d3e793-2893-4d69-b00e-225a35e06549	*Microtus agrestis*	Field Vole	2022-01-10	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
e1726792-2444-44ed-8ef2-0269c96b7754	*Sorex araneus*	Common Shrew	2019-07-17	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
e1aec0b5-8c37-4c51-a4c2-38d652149747	*Micromys minutus*	Harvest Mouse	1736-01-01	Mammal records from Britain from the Atlas of Mammals (1993), with some	Biological Records Centre	CC-BY	Accepted

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
				subsequent records			
e1f42fe3-e3ea-4739-b135-4bc5dcfba51e	*Microtus agrestis*	Field Vole	2021-11-27	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
e3a1f598-f43c-4227-8e42-ba62de14d217	*Microtus agrestis*	Field Vole	2021-02-23	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
e4863e95-77fa-4c0e-8a2e-d3f620966d24	*Apodemus sylvaticus*	Wood Mouse	2007-04-25	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
e53c8b2b-1803-446f-a0ba-7910e30f91d2	*Mus*	Mouse	2022-08-28	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
e5b310a1-8bd3-4810-960f-662402343432	*Sorex araneus*	Common Shrew	1983-07-17	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
e6156b6e-7cf5-4b22-8147-657bcee3a6fd	*Apodemus sylvaticus*	Wood Mouse	2021-06-13	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
e671ad53-3fd0-4e8f-88a1-122d56ca22be	*Apodemus sylvaticus*	Wood Mouse	2020-10-31	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
e6aa09a5-6669-45da-aa97-9109ed8c7dc6	*Myodes glareolus*	Bank Vole	2016-08-20	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
e6fc7e2b-7fd4-4b43-8cf3-314e3cd7a425	*Microtus agrestis*	Field Vole	2021-03-25	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
e7a3906a-7f88-41cc-bd80-1873c7131225	*Microtus agrestis*	Field Vole		Mammal records from Britain from the Atlas of Mammals (1993), with some subsequent records	Biological Records Centre	CC-BY	Accepted
e7aadbbe-d214-4c06-ad81-05326825e6fe	*Microtus agrestis*	Field Vole	2015-02-05	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted
eb74a9c1-c8e2-4a96-a66c-14c5485dc227	*Apodemus sylvaticus*	Wood Mouse	2005-06-20	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
ec144170-833c-434c-a3f5-2281da969fa8	*Apodemus sylvaticus*	Wood Mouse	2019-02-22	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
ec651cd2-23de-4569-	*Microtus agrestis*	Field Vole	2021-11-02	National Mammal Atlas	Mammal Society	CC-BY-NC	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
be02-e14bad1cfd4a				Project, online recording			
edb48452-6380-4eee-b32a-49018018ee84	*Myodes glareolus*	Bank Vole	2014-07-30	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
ee2a9bd3-c7a2-422c-8944-98c296e2c852	*Apodemus sylvaticus*	Wood Mouse	2025-01-11	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
ee7c8c4d-2529-4452-9812-ba9cd9e7c9f7	*Sorex araneus*	Common Shrew	2025-05-29	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
ef052b07-ef84-4b31-a21f-17e595c3f93c	*Microtus agrestis*	Field Vole	2022-06-12	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - correct
ef4aaa39-0c52-4c97-8965-f9c7c0d677b5	*Microtus agrestis*	Field Vole	2022-05-21	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
f14b7198-ef2f-4799-b3f1-550bbfbce8a3	*Microtus agrestis*	Field Vole	2021-03-18	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
f3201b05-6296-432f-ac99-55fd2d410d61	*Apodemus sylvaticus*	Wood Mouse	2006-06-17	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
f3c82399-6310-4f04-912c-e1c92eb18534	*Apodemus sylvaticus*	Wood Mouse	2021-04-07	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
f43b741a-d11d-4630-805b-c0278c82e13e	*Apodemus sylvaticus*	Wood Mouse	2021-10-15	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
f4779ad0-a2d6-4b11-ad05-0ac7c557c4e3	*Microtus agrestis*	Field Vole	2022-02-26	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
f4982abd-6c81-4c72-afaa-eef931ed7b26	*Sorex araneus*	Common Shrew	1982-07-04	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
f5faf26a-d554-48f1-bf91-b8b14a510722	*Microtus agrestis*	Field Vole	2024-08-16	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
f64d8b97-4552-4f28-aa58-0d123c6eb792	*Sorex araneus*	Common Shrew	2015-01-08	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted
f729b5a0-ac23-46da-a151-d130afeeccca	*Sorex minutus*	Pygmy Shrew	2005-02-28	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
f75386c8-66e4-48e2-9d36-5bb1f55eb4b7	*Apodemus sylvaticus*	Wood Mouse	1983-10-04	Stoke-on-Trent Environmental Survey results (1982-1984)	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
f8a4d554-fb89-41c9-8fdf-eca9d07c731d	*Apodemus sylvaticus*	Wood Mouse	1983-06-19	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
f8d213b3-bf3b-469d-9b92-f8b157461751	*Myodes glareolus*	Bank Vole	1983-09-12	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
f9c061d4-c4b9-458b-ac28-77fea98152de	*Sorex araneus*	Common Shrew	1991-11-03	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
fa2edfdd-eebd-4cac-b475-9aff3ed9cc53	*Myodes glareolus*	Bank Vole	2018-08-12	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
fc2bdbb4-0bd0-4ba5-b3da-ddd75d0200a3	*Microtus agrestis*	Field Vole	2021-12-31	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
fcba9af1-5f8b-44b4-a1f0-972a41a9d282	*Apodemus sylvaticus*	Wood Mouse	2025-10-11	Mammal Mapper App Sighting Records	Mammal Society	CC-BY	Accepted - considered correct
fe760c6d-ce2e-471f-9383-8ba820a01102	*Apodemus sylvaticus*	Wood Mouse	2021-09-11	National Mammal Atlas Project, online recording	Mammal Society	CC-BY-NC	Accepted - correct
fe7ef92f-9675-457e-880c-b4534eb0ed2f	*Apodemus sylvaticus*	Wood Mouse	2007-01-16	SER Species-based Surveys	Staffordshire Ecological Record	CC-BY-NC	Unconfirmed - plausible
ff45ad70-ed9a-4f57-b5fa-ae618f1a7cd5	*Microtus agrestis*	Field Vole	2022-06-12	Mammal Mapper App	Mammal Society	CC-BY	Accepted - correct

NBN Atlas record ID	Scientific name	Common name	Event date	Dataset	Provider	Licence	Verification status
				Sighting Records			

Table A.2 Anonymised small mammal occurrence records supplied directly by Staffordshire Ecological Record.

SER record ID	Scientific name	Common name	Record date	Survey/source	Record method	Abundance/details	Status
SR00001100002AV6	*Apodemus sylvaticus*	Wood Mouse	26/03/1996	Staffordshire BRC data	trapped in Longworth	2 Count	Original
SR0000110002I3TA	*Apodemus sylvaticus*	Wood Mouse	27/08/2018	iRecord	Photographic record	1 Count	Considered Correct
SR0000110002I0MA	*Apodemus sylvaticus*	Wood Mouse	13/04/2019	iRecord	Photographic record	1 Count of Adult	Considered Correct
SR0000110002I3I5	*Apodemus sylvaticus*	Wood Mouse	29/07/2018	iRecord	Photographic record	1 Count	Considered Correct
MMEEE3TWEEELTQBF	*Apodemus sylvaticus*	Wood Mouse	25/09/2006	Staffordshire Mammal Group (3tw/9r6)	Field Observation	1 Count of Adult	Original
SR0000110002HZ13	*Apodemus sylvaticus*	Wood Mouse	22/02/2019	iRecord	Photographic record	1 Count	Considered Correct
MMEEE9R6EEEJ6B7T	*Apodemus sylvaticus*	Wood Mouse	19/04/2009	Staffordshire Mammal Group (3tw/9r6)	Live trapping	9 Count of Adult	Original
MMEEE3TWEEEGW6OT	*Apodemus sylvaticus*	Wood Mouse	18/05/2002	Staffordshire Mammal Group (3tw/9r6)	Field Observation	1 Count of Adult	Original
MMEEE3TWEEEK9I8N	*Apodemus sylvaticus*	Wood Mouse	12/04/2006	Staffordshire Mammal Group (3tw/9r6)	Field Observation	1 Count of Adult	Original

SER record ID	Scientific name	Common name	Record date	Survey/source	Record method	Abundance/details	Status
SR0000110001FH87	*Apodemus sylvaticus*	Wood Mouse	15/04/2013	County Recorders (JEH)	Field Observation		Original
MMEEE3TNEEEBCO69	*Apodemus sylvaticus*	Wood Mouse	06/01/2007	County Recorders (JEH)	Field Observation	1 Count of Adult	Original
MMEEE3TNEEE05NWN	*Apodemus sylvaticus*	Wood Mouse	19/03/2007	County Recorders (JEH)	Field Observation	1 Count of Adult	Original
MMEEE3TNEEEKIZ22	*Apodemus sylvaticus*	Wood Mouse	20/06/2005	County Recorders (JEH)	Field Observation	1 Count of Adult	Original
SR0000110001500O	*Apodemus sylvaticus*	Wood Mouse	06/11/2011	County Recorders (GCS)	Field Observation		Original
SR00001100002OF2	*Apodemus sylvaticus*	Wood Mouse	13/10/2002	County Recorders (GCS)	Field Observation	1 Count of Dead	Validation
MMEEE3TWEEE8B524	*Apodemus sylvaticus*	Wood Mouse	13/04/2002	Staffordshire Mammal Group (3tw/9r6)	Field Observation	1 Count of Adult	Original
MMEEE3TWEEEUZ3JI	*Apodemus sylvaticus*	Wood Mouse	14/12/2002	Staffordshire Mammal Group (3tw/9r6)	Field Observation	1 Count of Adult	Original
MMEEE3TWEEE72W7T	*Apodemus sylvaticus*	Wood Mouse	24/04/2002	Staffordshire Mammal Group (3tw/9r6)	Field Observation	1 Count of Adult	Original
SR00001100002FW9	*Apodemus sylvaticus*	Wood Mouse	June 1986	Staffordshire BRC data	Field Observation		Original
SR0000110001FH88	*Apodemus sylvaticus*	Wood Mouse	16/04/2013	County Recorders (JEH)	Field Observation		Original
SR00001100002HWW	*Apodemus sylvaticus*	Wood Mouse	31/08/1993	Staffordshire BRC data	Field Observation	1 Count of Juvenile	Original
SR000011000056YZ	*Apodemus sylvaticus*	Wood Mouse	04/10/1983	Stoke-on-Trent Environmental Survey	Field Observation		Original

SER record ID	Scientific name	Common name	Record date	Survey/source	Record method	Abundance/details	Status
SR00001100002F38	*Apodemus sylvaticus*	Wood Mouse	19/06/1983	Staffordshire BRC data	Field Observation		Validation
SR0000110002I0PY	*Apodemus sylvaticus*	Wood Mouse	13/10/2018	iRecord	Photographic record	23 Count	Considered Correct
MMEEE3TWEEEWVML8	*Apodemus sylvaticus*	Wood Mouse	16/01/2007	Staffordshire Mammal Group (3tw/9r6)	Live trapping	5 Count of Adult	Original
MMEEE3TWEEEMMUTB	*Apodemus sylvaticus*	Wood Mouse	25/03/2004	Staffordshire Mammal Group (3tw/9r6)	Nuts feeding	1 Count of Adult	Original
SR00001100002F39	*Apodemus sylvaticus*	Wood Mouse	19/06/1983	Staffordshire BRC data	Field Observation		Validation
MMEEE9R6EEEQ53BP	*Apodemus sylvaticus*	Wood Mouse	20/09/2009	Staffordshire Mammal Group (3tw/9r6)	Live trapping	12 Count of Adult	Original
SR00001100022I7J	*Apodemus sylvaticus*	Wood Mouse	25/03/1981	Staffordshire BRC data	Dead	1 Count of Dead	Original
SR000011000051JU	*Sorex*	Sorex	10/08/1982	Stoke-on-Trent Environmental Survey	Field Observation		Original
SR0000110002I1ES	*Sorex araneus*	Eurasian Common Shrew	17/07/2019	iRecord	Field Observation	1 Count of Adult	Considered Correct
MMEEE3TWEEE6DSKS	*Sorex araneus*	Eurasian Common Shrew	20/07/2002	Staffordshire Mammal Group (3tw/9r6)	Field Observation	1 Count of Adult	Original
SR0000110002HZT6	*Sorex araneus*	Eurasian Common Shrew	27/06/2018	iRecord	Photographic record	1 Count	Considered Correct

SER record ID	Scientific name	Common name	Record date	Survey/source	Record method	Abundance/details	Status
SR000011000056ZU	*Sorex araneus*	Eurasian Common Shrew	05/10/1983	Stoke-on-Trent Environmental Survey	Field Observation		Original
SR0000110001FI36	*Sorex araneus*	Eurasian Common Shrew	April 2011	County Recorders (RobG)	Field Observation	1 Count of Adult	Original
SR000011000056KT	*Sorex araneus*	Eurasian Common Shrew	20/09/1983	Stoke-on-Trent Environmental Survey	Field Observation		Validation
SR000011000056JN	*Sorex araneus*	Eurasian Common Shrew	20/09/1983	Stoke-on-Trent Environmental Survey	Field Observation	1 Count of Dead	Validation
SR0000110002HYYF	*Sorex araneus*	Eurasian Common Shrew	13/10/2018	iRecord	Photographic record	1 Count	Considered Correct
SR00001100005BGA	*Sorex araneus*	Eurasian Common Shrew	13/04/1984	Stoke-on-Trent Environmental Survey	Field Observation		Original
SR00001100002GIY	*Sorex araneus*	Eurasian Common Shrew	06/01/1989	Staffordshire BRC data	Field Observation		Original
MMEEE9R6EEET8883	*Sorex araneus*	Eurasian Common Shrew	20/09/2009	Staffordshire Mammal Group (3tw/9r6)	Live trapping	1 Count of Adult	Original
SR00001100002AYD	*Sorex minutus*	Eurasian Pygmy Shrew	24/08/1996	Staffordshire BRC data	Field Observation	1 Count of Dead	Original
SR00001100005DGH	*Sorex minutus*	Eurasian Pygmy Shrew	19/06/1984	Stoke-on-Trent Environmental Survey	Field Observation		Original

SER record ID	Scientific name	Common name	Record date	Survey/source	Record method	Abundance/details	Status
SR00001100005DGI	*Sorex minutus*	Eurasian Pygmy Shrew	19/06/1984	Stoke-on-Trent Environmental Survey	Field Observation		Original
MMEEE9R6EEET4Q7H	*Myodes glareolus*	Bank Vole	19/04/2009	Staffordshire Mammal Group (3tw/9r6)	Live trapping	6 Count of Adult	Original
SR0000110001TUDO	*Myodes glareolus*	Bank Vole	26/04/2013	iRecord	Field Observation		Original
SR00001100005D6T	*Myodes glareolus*	Bank Vole	07/06/1984	Stoke-on-Trent Environmental Survey	Field Observation		Original
MMEEE3TWEEE39177	*Myodes glareolus*	Bank Vole	16/01/2007	Staffordshire Mammal Group (3tw/9r6)	Live trapping	1 Count of Adult	Original
SR00001100002EV5	*Mus musculus*	House Mouse	23/03/1983	Staffordshire BRC data	Field Observation		Original
SR00001100022I7K	*Mus musculus*	House Mouse	21/04/1980	Staffordshire BRC data	Field Observation		Original
MMEEE3TWEEE8N8KG	*Micromys minutus*	Harvest Mouse	25/03/2004	Staffordshire Mammal Group (3tw/9r6)	Nest	1 Count of Adult	Original
SR0000110000Z8IE	*Micromys minutus*	Harvest Mouse	09/10/2009	SER General Records 2009	Nest	multiple Count of Young	Original
SR0000110002E6P4	*Microtus*	Microtus	20/11/2014	Consultants (FPCR)	Field Observation	1 Count of Burrow; 1 Count of Feeding remains; 1 Count of Latrine	Original
MMEEE9R6EEE67SUV	*Microtus agrestis*	Field Vole	19/04/2009	Staffordshire Mammal Group (3tw/9r6)	Tunnel	1 Count of Adult	Original

SER record ID	Scientific name	Common name	Record date	Survey/source	Record method	Abundance/details	Status
MMEEE3TWEEED2FKG	*Microtus agrestis*	Field Vole	26/05/2002	Staffordshire Mammal Group (3tw/9r6)	Field Observation	1 Count of Adult	Original
SR00001100002G04	*Microtus agrestis*	Field Vole	July 1986	Staffordshire BRC data	Field Observation		Original
SR00001100022I56	*Microtus agrestis*	Field Vole	08/11/2017	County Recorders (DRS)	trapped in Longworth	1 Count of Adult	Original
SR0000110001WPIN	*Microtus agrestis*	Field Vole	17/10/2015	County Recorders (DRS)	Field Observation	2 Count	Original
MMEEE3TWEEESSNWU	*Microtus agrestis*	Field Vole	25/03/2004	Staffordshire Mammal Group (3tw/9r6)	Droppings	1 Count of Adult	Original
MMEEE9R6EEEVTG60	*Microtus agrestis*	Field Vole	20/09/2009	Staffordshire Mammal Group (3tw/9r6)	Live trapping	3 Count of Adult	Original
SR00001100022I8O	*Microtus agrestis*	Field Vole	12/04/1975	Staffordshire BRC data	Pellet	1 Count of bones	Original
MMEEE3TWEEEGJDJ0	*Microtus agrestis*	Field Vole	15/07/2006	Staffordshire Mammal Group (3tw/9r6)	Cat brought in	1 Count of Adult	Original
MMEEE3TWEEE3F3CH	*Microtus agrestis*	Field Vole	13/11/2007	Staffordshire Mammal Group (3tw/9r6)	Cat brought in	1 Count of Adult	Original

Appendix B: Fieldwork ethics and Stoke-on-Trent City Council Approval

This appendix provides supporting documentation for the ethical approval and site-access arrangements underpinning the small mammal fieldwork described in Chapter 3. Institutional ethical approval was granted by the Staffordshire University School of Life Sciences and Education Ethics Panel on 19 April 2021. Stoke-on-Trent City Council subsequently confirmed its support for the research and agreed to facilitate access to council-managed green spaces representing different grassland-management practices.

Appendix B.1: Ethical approval for trapping small mammals

Life Sciences and Education

ETHICAL APPROVAL FEEDBACK

Researcher name:	Baylee Jennings
Title of Study:	Small mammals in city: Assessing the biodiversity value of urban greenspace and the effects of differing management regimes
Status of approval:	Approved

Thank you for addressing the committee's comments. Your research proposal has now been approved by the Ethics Panel and you may commence the implementation phase of your study. You should note that any divergence from the approved procedures and research method will invalidate any insurance and liability cover from the University. You should, therefore, notify the Panel of any significant divergence from this approved proposal.

You should arrange to meet with your supervisor for support during the process of completing your study and writing your dissertation.

When your study is complete, please send the ethics committee an end of study report. A template can be found on the ethics BlackBoard site.



Signed: Prof. Roozbeh Naemi **Date: 19.04.2021**

Ethics Coordinator

Appendix B.2: Stoke-on-Trent City Council support and site access confirmation

FW: Surveying

Paul Evison <Paul.Evison@stoke.gov.uk>
Wed 17/03/2021 09:43

To: JENNINGS Baylee <baylee.jennings@research.staffs.ac.uk>
Cc: Carol Gibbs <Carol.Gibbs@stoke.gov.uk>; Craig Howell <Craig.Howell@stoke.gov.uk>; Rob Hambleton <Rob.Hambleton@stoke.gov.uk>; FOI <foi@stoke.gov.uk>; Helen Meharg <Helen.Meharg@stoke.gov.uk>; Jonathan Ayres (Stoke City Council Ecologist) <Jonathan.Ayres@stoke.gov.uk>

1 attachments (33 KB)
PU city_A500 flat_72dpi.jpg;

Hi Baylee

Thank you for your email.

This sounds a really interesting and relevant research project which we would be happy to support and facilitate your access to complete, all I ask is that you keep me updated on your findings and share a copy of your final document.

The City Council has and continues to find alternative management practices for its POS in order to help offset carbon emissions and enhance biodiversity, this includes the neutralisation of open space, the establishment of meadows/wildflower areas as well as tree planting.

We will be able to signpost you to different sites that have a range of cutting and management practices in place it will be really useful to gain a further insight into the impact some of these management and maintenance changers have had.

Please let me know your thoughts, I've copied in Craig Howell and Rob Hambleton North and South Area managers for Grounds, Helen Meharg – Countryside Team Leader and Jon – the Councils Ecologist, all should be able to sign post you to relevant sites/information.

Kind Regards

Paul Evison
Service Manager- Streetcare and Greenspace
City of Stoke-on-Trent Council
Admin Block, Cromer Road Depot, Cromer Road, Stoke on Trent ST1 6QN

T: 01782 235243
e: paul.evison@stoke.gov.uk

www.stoke.gov.uk
www.twitter.com/sotcitycouncil
www.facebook.com/sotcitycouncil

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Appendix C: Natural England GL01 licence documentation

This appendix provides the Natural England GL01 licence documentation relevant to the trapping of shrews during the small mammal fieldwork described in Chapter 3.

GOV•UK

NATURAL ENGLAND

Statutory guidance

Licence to take shrews (GL01)

Updated 1 January 2026

Applies to England

Contents

Overview
Legislation
Licence terms and conditions
Definitions used in this licence
Licence conditions
Information and advice specific to this licence
Information and advice for all class and general licences
Contact Natural England

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This publication is available at https://www.gov.uk/government/publications/shrews-licence-to-take-them/licence-to-take-shrews-gl01

General licence GL01: to take shrews (Soricidae) for scientific or educational purposes, or for the purpose of ringing or marking, or examining any ring or mark.

Overview

This licence permits persons with appropriate knowledge and experience to trap and mark shrews for scientific or education purposes. Users must comply with licence terms and conditions.

Registration

Users do not need to register to use this licence.

Recording and reporting

Users are required to report fatalities – see Condition 3(c).

Reference: WML - GL01

Legislation

Statute(s) Wildlife and Countryside Act 1981 (as amended) ('the 1981 Act').

Section(s) This licence is issued under sections 16(3)(a) and (b), and 16(5).

Licence terms and conditions

Valid for the period: 1 January 2026 to 31 December 2030 (inclusive).

Area valid in: All counties of England (landward of the mean low water mark).

Purpose(s) for which this licence is issued: This licence can only be used for:

- science or education
- conserving wild animals
- ringing and marking or examining any mark on wild animals

What this licence permits

Subject to all the terms and conditions of this licence and solely for the purpose(s) stated above, this licence permits Authorised Persons to:

- take shrews by the use of appropriate live capture traps
- mark shrews using appropriate humane methods

Who can use this licence

Only an Authorised Person* can use this licence.

You must contact Natural England to apply for an individual licence if this licence does not cover the action you want to take.

*see the section 'Definitions used in this licence'

Who cannot use this licence

You cannot use this licence if you have been convicted of a wildlife offence (https://www.cps.gov.uk/legal-guidance/wildlife-offences)* on or after 1 January

2010, unless one or both of the following applies:

- you have served your conviction and are a rehabilitated person under the Rehabilitation of Offenders Act 1974
- a court has made an order discharging you of the wildlife offence

*the section 'Definitions used in this licence'

Definitions used in this licence

"Authorised person" is a person with appropriate knowledge and experience of shrews, and the trapping and marking techniques permitted, or under the direct supervision of such a person.

"Wildlife offence" means any offence under the Invasive Alien Species (Enforcement and Permitting) Order 2019, or any of the following legislation (all as amended):

- Conservation of Habitats and Species Regulations 2017 ('the Habitats Regulations')
- Wildlife and Countryside Act 1981
- Protection of Badgers Act 1992
- Deer Act 1991
- Hunting Act 2004
- Wild Mammals (Protection) Act 1996
- Animal Welfare Act 2006
- Protection of Animals Act 1911

Licence conditions

1. This licence does not authorise the use of any method of taking which is prohibited by section 11 of the 1981 Act except those listed under 'What this licence permits'.
2. No work shall be carried out under this licence on a National Nature Reserve except with the prior written permission of Natural England.
3. At each inspection:

a. Any shrew taken under this licence shall be liberated at the site of capture immediately after examination and marking.

b. Any other species caught in a trap must be released, unharmed, at the site of capture immediately after examination and marking, unless they are listed a non-native species. See notes o to p in the section 'Information and advice'.

c. Any dead animal caught in the trap must be removed from it. The licensee must report shrew deaths to Natural England (see contact details below) using form GL-01 Annex to help assess the trapping method.

4. All relevant animal welfare legislation must be complied with at all times, including the Animal Welfare Act 2006 and the Wild Mammals (Protection) Act 1996 (see note r in the section 'Information and advice').

5. When in use, every trap used under this licence must be physically inspected as follows:

   a. Where food and bedding are provided: at least twice every day at intervals of no less than 3 hours and no more than 12 hours (see note g to i in the section 'Information and advice')

   b. Without food and bedding: at intervals less than 3 hours

6. All equipment used for the purpose of this licence shall be so constructed and maintained as to avoid cruelty and distress to wild animals (see notes e and f in the section 'Information and advice').

7. Where a trap is not in use, it must be rendered incapable of holding or catching shrews or other animals. Any bait, food, or water must also be removed (see note n in the section 'Information and advice').

Important

This licence authorises acts that would otherwise be offences under the legislation referred to above. Failure to comply with its terms and conditions:

i. may be an offence against the 1981 Act or mean that the licence cannot be relied upon and an offence could therefore be committed. The maximum penalty available for an offence under the 1981 Act is, at the time of the issue of this licence, an unlimited fine, a six-month custodial sentence, or both

ii. may result in your permission to use this licence being withdrawn. Natural England will inform any person or organisation whose permission to use this licence is withdrawn in writing. This sanction may be applied to other similar licences

iii. may mean that you are not able to rely on this licence as a defence in respect to the prohibitions within the Animal Welfare Act 2006 or the Wild Mammals (Protection) Act 1996. If the activity that you wish to undertake is not covered by this licence, or if you are unable to comply with any of the terms and conditions which apply to the use of this licence, then you will need to apply to Natural England for an individual licence

This licence is not a consent for the purposes of Part II of the Wildlife and Countryside Act 1981 (as amended) in respect to SSSIs. It is your responsibility to get consent or assent if required before this licence can be used on any SSSI. See notes za to zc in the section 'Information and advice' for further information.

Issued by and on behalf of Natural England on 1 January 2026.

Information and advice specific to this licence

Coronavirus – Best practice

a. To avoid transmitting COVID-19 from people to wild mammals, you should: stay at least 2 metres from live wild mammals at all times; only capture or handle live wild mammals if necessary; and avoid going within 2 metres of known breeding or regularly used resting sites of mammals. If you cannot follow these guidelines, then you should minimise the time spent in close proximity and handling mammals; limit the number of people that come into close proximity or handle mammals; and keep animals separate from other individuals as far as possible.

b. Persons acting under this licence should follow good hygiene practices, including wearing of disposable gloves, or clean and disinfected non-disposable gloves; wearing a face covering when handling and in close proximity to any mammal; and washing hands before and after close proximity or handling of any mammal. Face coverings, non-disposable

gloves and overalls or other clothing should be washed or disinfected between uses when handling mammals.

c. You should not carry out activities under this licence if you feel unwell, test positive for COVID-19 or live with someone who has or might have COVID-19.

d. Any products used to clean and disinfect equipment should be safe for use where there could be contact with animals and used at an effective and safe dilution rate. 'Safe4' is considered to be an appropriate disinfectant for this purpose and is considered effective against the virus at a dilution of 1:50. Other products can be used to clean and disinfect equipment.

Trap construction

e. It is the responsibility of each person relying on this licence to use appropriate equipment (trap) and to ensure that they are competent to use such equipment so as to avoid causing unnecessary suffering. Appropriate live-traps are those that are designed, set and operated in such a manner as to avoid death or injury. This licence is intended to be used with traps manufactured commercially for the purpose of catching small mammals. 'Home made' traps should not be used unless they are equivalent to the specifications of those commercially available. When using 'home made' traps, records of measurements, materials and the live-trap mechanism used should be logged and provided to Natural England, if requested.

f. All traps should be regularly maintained and checked to ensure correct operation before use.

Trap use

g. Food: Leave plenty of suitable food in the trap nest-box. Blowfly larvae or pupae (Calliphora spp., also known as "casters") are best. These can be frozen and used as required. Frozen food must be fully thawed before use. Use in sufficient numbers to provide 10 g per trap.

h. Bedding: leave in the nest-box to provide insulation. Hay is best. Ensure it is dry.

i. Any person acting under this licence should have regard to the inspection requirements contained in Condition 5. Shrews need to eat every 3 to 4 hours, so inspect the trap at suitably frequent intervals.

(i.) If sufficient food and bedding are left, twice daily inspections (ideally at dusk and early morning) should be sufficient in accordance with licence condition 5.

(ii.) If for any reason food cannot be left, shrews are likely to die in the trap if left there for more than 4 hours, and visits should be correspondingly frequent (at intervals of less than 3 hours) - particularly if trapping from July to September when more shrews may be caught. Remember, however, that very frequent visits to the trap will cause disturbance to the habitat and this may affect trapping success.

j. When setting traps you should carefully record the location of each one and ensure that all those set are collected at the end of trapping. You may wish to mark the location with small flags or coloured tape on a stick.

k. Traps should be stable and the nest chamber of any traps used should be sloped to prevent water running into the bedding.

l. Traps should be covered with surrounding vegetation or other material, to insulate against extremes of temperatures. When trapping following periods of cold weather, particularly when the ground has been frozen, ensure that the trap has sufficient insulation from the ground, which will remain colder than the air temperature. This is particularly important when using metal traps. Within the constraints of licence condition 5, inspection intervals should be more frequent in hot, cold or wet weather. When trapping, avoid setting a trap during periods of extreme hot, cold or wet weather, or when such conditions are likely.

i. At each visit, check bedding and food within the trap and cover outside the trap and replace as required.

m. If shrew deaths occur, adjust the amounts of food and bedding, or increase the frequency of inspections, as appropriate. If despite this, deaths continue, suspend trapping and seek advice. (See Conditions 3(c) and 7 on this licence).

n. In order to render any trap incapable of holding or catching shrews or other animals when out of use, it is necessary to either secure the door in a fully open or closed position or to remove the door completely.

Non-native species

o. Under section 14 of the 1981 Act it is an offence to release into the wild any animal included in Schedule 9 of the 1981 Act. Any animals of species listed on Schedule 9 accidentally caught by licensed trapping should be treated as follows:

- Species listed in Part 1 must not be released or allowed to escape into the wild. It is preferable that these animals are killed in a quick and humane manner as soon as reasonably practicable after discovery. These animals may only be released under an individual licence obtained from Natural England.
- Species listed in Part 1A (which are native to Great Britain) must be released immediately upon discovery. This is permitted under general licence WML- GL22 (https://www.gov.uk/government/publications/wild-birds-and-animals-licence-to-release-certain-schedule-9-species-gl22).
- Species listed in Part 1B (which are former native species) may be released immediately upon discovery at the point of capture (this is permitted by general licence WML- GL22), killed humanely or kept in captivity. These animals may only be released at another location under an individual licence obtained from Natural England.

p. If you catch a species listed as an invasive non-native (alien) animal (https://www.gov.uk/guidance/invasive-non-native-alien-animal-species-rules-in-england-and-wales#list) you must not release it. You must quickly and humanely dispatch it, or give it to a premises licensed to keep it. For a list of licensed premises contact Natural England: invasive-alien-species-licence@naturalengland.org.uk.

Biosecurity

q. Users of this licence should consider the risks of activities to biosecurity and take relevant precautions when visiting heathland and woodland sites to avoid the spread of invasive non-native species, and tree and plant pests and diseases (https://www.gov.uk/guidance/find-a-specific-tree-pest-or-disease) such as Phytophthora and Ash dieback Chalara (https://www.forestresearch.gov.uk/tools-and-resources/fthr/pest-and-disease-resources/ash-dieback-hymenoscyphus-fraxineus/). General biosecurity advice (https://www.nonnativespecies.org/biosecurity/) can be found on the GB Non-Native Species Secretariat website. Further information on plant biosecurity (https://www.gov.uk/guidance/plant-health-controls) is available on the GOV.UK website.

The law

r. Natural England has issued this licence in exercise of the powers conferred by the 1981 Act. Visit www.legislation.gov.uk (http://www.legislation.gov.uk/ukpga/1981/69) to view the full text of this legislation.

s. Section 16 of the 1981 Act provides that the offences in Part 1 of the 1981 Act shall not apply to anything done under and in accordance with the terms of a licence granted by the appropriate authority, which is, in England, Natural England.

t. This licence complies with Article 9(1) of the Bern Convention (http://www.coe.int/en/web/bern-convention) (the exception applied in this case is 'for the purposes of research and education').

Relevant legislation and good practice

u. Persons acting under a licence should have regard to legislation and good practice relevant to the action(s) undertaken, including animal welfare and the Animal Welfare Act 2006. It is an offence to cause any unnecessary suffering to an animal (including birds) under the control of man (section 4 of the 2006 Act). This applies to the humane dispatch of captured animals and the treatment of animals held in traps or nets, including decoy birds and non-target animals. The application of Animal Welfare Act 2006 to wildlife management activities is explained in Natural England leaflet Wildlife Management Advice Note: The Animal Welfare Act 2006: what it means for wildlife (https://www.gov.uk/government/publications/wildlife-management-advice-notice-the-animal-welfare-act-2006-wml-gu02) (WML-GU02).

Information and advice for all class and general licences

General information

v. Ordinarily, licences will be reissued or updated on 1 January each year. You should check the terms and conditions of the licence before you use it for the first time each year, in case of amendments. Note that licences can be modified or revoked at any time by Natural England or the Secretary of State, but this will not be done unless there are good reasons for doing so.

w. The common name of the species given in a licence is included by way of guidance only; in the event of any dispute or proceedings, it is the scientific name of a species only that will be taken into account.

The limits of licences

x. Licences permit action only for the purposes specified on that licence.

y. Licences do not permit actions prohibited under any other legislation, nor do they confer any right of entry upon land.

z. Unless otherwise stated the provisions of Natural England licences only apply landward of the mean low water mark in England. The Marine Management Organisation is responsible for all licensing seaward of the mean low water mark.

Protected sites

za. You can search for and view details about all SSSIs by using Natural England's Designated Sites system (https://designatedsites.naturalengland.org.uk/SiteSearch.aspx). The notification documents for each SSSI contain a list of operations that require Natural England's prior consent. Owners and occupiers of land notified as SSSIs are required to give written notice to Natural England before either beginning any of these operations, or allowing someone else to carry out those operations. SSSI consent can only be given to a SSSI owner or occupier. It may be given with or without conditions, or in some cases, consent may not be granted.

A similar process applies to public bodies and statutory undertakers - as defined under Section 28G of the Wildlife and Countryside Act 1981 (as amended) and this obligation applies even where the operations are carried out on land outside of the SSSI.

zb. Please note that as the licensee you will not be able to undertake the licensed activity on a SSSI until the owner or occupier of the SSSI has applied for, and received, Natural England's written SSSI consent. If you do so, you may be at risk of committing an offence. As the licensee, if you wish to exercise this licence on a SSSI you must contact the relevant owners or occupiers of the SSSI and ensure they give written notice to Natural England of their proposal to permit you to carry out licensed activity on their SSSI. You should wait until a SSSI consent decision has been received by the SSSI owner/occupier before you begin to exercise this licence on a SSSI. See GOV.UK for further information on how to get SSSI consent (https://www.gov.uk/guidance/protected-areas-sites-of-special-scientific-interest) from Natural England.

zc. In considering whether to issue consent or assent for activities likely to affect a SSSI that is a European site, in other words a Special Protection Area (SPA) or Special Area of Conservation (SAC), Natural England will carry out a Habitats Regulations Assessment, as required by the Conservation of Habitats and Species Regulations 2017 (as amended) ('the Habitats Regulations') to ensure there will be no adverse effects on the European site.

Further information

The Mammal Society booklet: Live Trapping Small Mammals - A Practical Guide by J Gurnell and J R Flowerdew (2006), contains many useful hints. Available from The Mammal Society (https://mammal.org.uk/publications). If there is a difference between a licence condition and published guidelines, you must follow the licence condition.

Contact Natural England

For licensing enquiries:

> Wildlife licensing (Science, education and conservation)
>
> Natural England
> Horizon House
> Deanery Road
> Bristol
> BS1 5AH
>
> Email wildlife.scicons@naturalengland.org.uk
>
> Telephone 0300 060 3900

For any other enquiries, contact Natural England (https://www.gov.uk/government/organisations/natural-england#org-contacts).

WML-GL01 [version January 2026].





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Appendix D: Grassland management classification source material

The management information used to support the classification of study sites was derived from Stoke-on-Trent City Council grass-cutting regimes and associated management information. The current council guidance is publicly accessible from the following webpage:

Stoke-on-Trent City Council (n.d.). Grass cutting. Available at: https://www.stoke.gov.uk/info/20056/road_problems/412/grass_cutting (Accessed: 20 July 2026).

A detailed explanation of how this information was interpreted and applied to the three grassland-management categories used in the study is provided in Section 3.2.2 of Chapter 3.

Appendix E: Supplementary statistical outputs for Chapter 3

This appendix presents the complete fixed-effect estimates, Type II Wald tests, diagnostic summaries and collinearity assessments for the final patch-scale models described in Chapter 3. Predictors were standardised before modelling, and Site ID was included as a random intercept to account for repeated survey observations within green space patches.

Appendix E.1: Chapter 3 - model formulations

Table E3.1. Model structures used for the final Chapter 3 community and species-level analyses. All models included the same standardised patch-scale fixed-effect structure comprising patch size, grassland cover, tree cover, woody linear density, watercourse density, path density, management intensity, and the grassland cover × management intensity interaction. Site identity was included as a random intercept to account for repeated survey observations within patches. Community models were fitted at the patch × repeat level, while species-specific models were fitted at the species × patch × repeat level.

Model	Level	Distribution	Link	Fixed effects	Random effect	Analysis unit
Species richness	Community	Poisson	Log	Patch size + grassland cover + tree cover + woody linear density + watercourse density + path density + management intensity + grassland cover × management intensity	(1 | SiteID)	Patch × repeat survey
Total CPUE	Community	Tweedie	Log	Patch size + grassland cover + tree cover + woody linear density + watercourse density + path density + management intensity + grassland cover × management intensity	(1 | SiteID)	Patch × repeat survey
Simpson's reciprocal diversity	Community	Gaussian	Identity	Patch size + grassland cover + tree cover + woody linear density + watercourse density + path density + management intensity + grassland cover × management intensity	(1 | SiteID)	Patch × repeat survey
A. sylvaticus	Species	Tweedie	Log	Patch size + grassland cover + tree cover + woody linear density + watercourse density + path density + management intensity + grassland cover × management intensity	(1 | SiteID)	Species × patch × repeat survey
M. agrestis	Species	Tweedie	Log	Patch size + grassland cover + tree cover + woody linear density + watercourse density + path density + management intensity + grassland cover × management intensity	(1 | SiteID)	Species × patch × repeat survey

Model	Level	Distribution	Link	Fixed effects	Random effect	Analysis unit
M. glareolus	Species	Tweedie	Log	Patch size + grassland cover + tree cover + woody linear density + watercourse density + path density + management intensity + grassland cover × management intensity	(1 | SiteID)	Species × patch × repeat survey
S. araneus	Species	Tweedie	Log	Patch size + grassland cover + tree cover + woody linear density + watercourse density + path density + management intensity + grassland cover × management intensity	(1 | SiteID)	Species × patch × repeat survey

Appendix E.2: Chapter 3 - community-level model outputs

Table E.2. Full fixed-effect outputs for the Chapter 3 community-level models. Estimates are standardised fixed-effect coefficients. Wald χ^2 statistics and *p*-values are derived from Type II tests. Species richness was modelled using a Poisson distribution with a log link, total CPUE using a Tweedie distribution with a log link, and Simpson's reciprocal diversity using a Gaussian distribution with an identity link.

Response	Predictor	Estimate	SE	CI low	CI high	Wald chisq	Df	p value	Significance
Species richness	Patch size	0.139	0.101	-0.058	0.337	1.914	1	0.167	
Species richness	Grassland cover	0.182	0.096	-0.006	0.370	2.309	1	0.129	
Species richness	Tree cover	0.140	0.129	-0.113	0.392	1.178	1	0.278	
Species richness	Woody linear density	0.308	0.144	0.027	0.590	4.613	1	0.032	*
Species richness	Watercourse density	0.076	0.138	-0.194	0.347	0.307	1	0.580	
Species richness	Path density	-0.172	0.089	-0.347	0.002	3.748	1	0.053	†
Species richness	Management intensity	0.001	0.083	-0.163	0.165	0.112	1	0.738	
Species richness	Grassland cover × management intensity	-0.232	0.140	-0.507	0.042	2.751	1	0.097	†
Total CPUE	Patch size	-0.235	0.089	-0.409	-0.061	7.018	1	0.008	**
Total CPUE	Grassland cover	0.404	0.078	0.252	0.556	24.358	1	<0.001	***
Total CPUE	Tree cover	0.091	0.110	-0.125	0.308	0.683	1	0.408	
Total CPUE	Woody linear density	0.372	0.120	0.136	0.608	9.527	1	0.002	**
Total CPUE	Watercourse density	0.241	0.118	0.010	0.472	4.172	1	0.041	*
Total CPUE	Path density	-0.257	0.075	-0.405	-0.109	11.643	1	<0.001	***

Response	Predictor	Estimate	SE	CI low	CI high	Wald chisq	Df	p value	Significance
Total CPUE	Management intensity	-0.278	0.069	-0.414	-0.142	19.280	1	<0.001	***
Total CPUE	Grassland cover × management intensity	-0.210	0.117	-0.440	0.020	3.215	1	0.073	†
Simpson's reciprocal diversity	Patch size	0.076	0.121	-0.160	0.313	0.401	1	0.527	
Simpson's reciprocal diversity	Grassland cover	0.134	0.109	-0.080	0.349	0.966	1	0.326	
Simpson's reciprocal diversity	Tree cover	0.040	0.157	-0.267	0.348	0.065	1	0.798	
Simpson's reciprocal diversity	Woody linear density	0.388	0.168	0.059	0.717	5.349	1	0.021	*
Simpson's reciprocal diversity	Watercourse density	0.142	0.169	-0.189	0.473	0.704	1	0.402	
Simpson's reciprocal diversity	Path density	-0.222	0.116	-0.448	0.005	3.683	1	0.055	†
Simpson's reciprocal diversity	Management intensity	0.099	0.099	-0.096	0.293	0.689	1	0.407	
Simpson's reciprocal diversity	Grassland cover × management intensity	-0.170	0.167	-0.497	0.156	1.046	1	0.307	

Significance codes: † $p < 0.10$; * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$. The dagger (†) indicates a marginal effect that did not meet the conventional $p < 0.05$ threshold. Blank cells indicate $p \geq 0.10$.

Appendix E.3: Chapter 3 - Species-specific model outputs

Table E.3. Full fixed-effect outputs for the Chapter 3 focal-species CPUE models. Estimates are standardised fixed-effect coefficients from Tweedie mixed-effects models with a log link. Wald χ^2 statistics and *p*-values are derived from Type II tests. Significance codes: † $p < 0.10$; * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$. The dagger (†) indicates a marginal effect that did not meet the conventional $p < 0.05$ threshold. Blank cells indicate $p \geq 0.10$.

Species	Predictor	Estimate	SE	CI low	CI high	Wald chisq	Df	p value	Significance
A. sylvaticus	Patch size	-0.199	0.094	-0.383	-0.015	4.480	1	0.034	*
A. sylvaticus	Grassland cover	0.386	0.081	0.227	0.545	20.587	1	<0.001	***
A. sylvaticus	Tree cover	0.113	0.117	-0.116	0.342	0.933	1	0.334	
A. sylvaticus	Woody linear density	0.332	0.129	0.080	0.585	6.656	1	0.010	**
A. sylvaticus	Watercourse density	0.155	0.126	-0.091	0.402	1.525	1	0.217	
A. sylvaticus	Path density	-0.194	0.080	-0.350	-0.038	5.946	1	0.015	*
A. sylvaticus	Management intensity	-0.340	0.074	-0.484	-0.196	24.603	1	<0.001	***
A. sylvaticus	Grassland cover × management intensity	-0.197	0.125	-0.443	0.048	2.485	1	0.115	
M. agrestis	Patch size	-0.472	0.158	-0.782	-0.163	8.926	1	0.003	**
M. agrestis	Grassland cover	0.606	0.161	0.291	0.920	14.748	1	<0.001	***
M. agrestis	Tree cover	0.243	0.190	-0.129	0.615	1.636	1	0.201	
M. agrestis	Woody linear density	0.075	0.205	-0.326	0.476	0.135	1	0.714	
M. agrestis	Watercourse density	0.638	0.209	0.227	1.048	9.271	1	0.002	**
M. agrestis	Path density	-0.487	0.129	-0.739	-0.235	14.306	1	<0.001	***
M. agrestis	Management intensity	0.071	0.125	-0.175	0.317	0.206	1	0.650	

Species	Predictor	Estimate	SE	CI low	CI high	Wald chisq	Df	p value	Significance
M. agrestis	Grassland cover × management intensity	-0.092	0.203	-0.491	0.307	0.205	1	0.651	
M. glareolus	Patch size	0.110	0.443	-0.758	0.979	0.062	1	0.803	
M. glareolus	Grassland cover	0.441	0.332	-0.210	1.093	0.794	1	0.373	
M. glareolus	Tree cover	0.431	0.523	-0.594	1.456	0.679	1	0.410	
M. glareolus	Woody linear density	1.736	0.632	0.497	2.975	7.537	1	0.006	**
M. glareolus	Watercourse density	-0.440	0.522	-1.463	0.583	0.712	1	0.399	
M. glareolus	Path density	-0.213	0.289	-0.780	0.353	0.545	1	0.460	
M. glareolus	Management intensity	-0.105	0.302	-0.696	0.486	0.142	1	0.706	
M. glareolus	Grassland cover × management intensity	-1.634	0.599	-2.808	-0.461	7.449	1	0.006	**
S. araneus	Patch size	0.251	0.329	-0.393	0.895	0.583	1	0.445	
S. araneus	Grassland cover	-0.217	0.262	-0.731	0.297	0.706	1	0.401	
S. araneus	Tree cover	-0.328	0.386	-1.085	0.429	0.721	1	0.396	
S. araneus	Woody linear density	0.674	0.445	-0.198	1.547	2.294	1	0.130	
S. araneus	Watercourse density	0.268	0.424	-0.563	1.098	0.399	1	0.528	
S. araneus	Path density	-0.646	0.248	-1.132	-0.159	6.774	1	0.009	**
S. araneus	Management intensity	-0.577	0.242	-1.051	-0.102	5.209	1	0.022	*

Significance codes: † $p < 0.10$; * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$. The dagger (†) indicates a marginal effect that did not meet the conventional $p < 0.05$ threshold. Blank cells indicate $p \geq 0.10$.

Appendix F: Supplementary statistical outputs for Chapter 4

This appendix presents the model structures and complete fixed-effect outputs for the final Chapter 4 matrix-only analyses. Models were fitted separately using landscape predictors measured within 250 m and 500 m matrix buffers surrounding each surveyed green space patch. The retained predictors comprised matrix green proportion, green patch count, nearest green patch distance, matrix hedge density, matrix watercourse density and matrix road density. All continuous predictors were standardised before modelling, and analyses were undertaken at the patch × repeat-survey level.

Appendix F.1: Chapter 4 – model formulations

Table F.1. Model structures used for the final Chapter 4 matrix-only analyses. Separate models were fitted at the 250 m and 500 m buffer scales using a consistent set of standardised landscape predictors. Species richness was modelled using a Poisson distribution with a log link, total CPUE using a Tweedie distribution with a log link, and Simpson's reciprocal diversity using a Gaussian distribution with an identity link.

Buffer	Model	Distribution	Link	Fixed effects	Random effect	Analysis unit
250 m	Species richness	Poisson	Log	Matrix green proportion + green patch count + nearest green patch distance + matrix hedge density + matrix watercourse density + matrix road density	Site ID (random intercept)	Patch × repeat survey
250 m	Total CPUE	Tweedie	Log	Matrix green proportion + green patch count + nearest green patch distance + matrix hedge density + matrix watercourse density + matrix road density	Site ID (random intercept)	Patch × repeat survey
250 m	Simpson's reciprocal diversity	Gaussian	Identity	Matrix green proportion + green patch count + nearest green patch distance + matrix hedge density + matrix watercourse density + matrix road density	Site ID (random intercept)	Patch × repeat survey
500 m	Species richness	Poisson	Log	Matrix green proportion + green patch count + nearest green patch distance + matrix hedge density + matrix watercourse density + matrix road density	Site ID (random intercept)	Patch × repeat survey
500 m	Total CPUE	Tweedie	Log	Matrix green proportion + green patch count + nearest green patch distance + matrix hedge density + matrix watercourse density + matrix road density	Site ID (random intercept)	Patch × repeat survey
500 m	Simpson's reciprocal diversity	Gaussian	Identity	Matrix green proportion + green patch count + nearest green patch distance + matrix hedge density + matrix watercourse density + matrix road density	Site ID (random intercept)	Patch × repeat survey

Appendix F.2: Chapter 4 – matrix-only models fitted for the 250 m buffer

Table F.2. Full fixed-effect outputs for the final Chapter 4 matrix-only models fitted at the 250 m buffer scale. Estimates represent standardised fixed-effect coefficients. Confidence limits are 95% confidence intervals, and Wald χ^2 statistics and *p*-values were derived from Type II tests.

Buffer	Response	Predictor	Estimate	SE	CI low	CI high	Wald chisq	Df	p value	Significance
250 m	Species richness	Matrix green proportion	0.085	0.126	-0.163	0.332	0.448	1	0.503	
250 m	Species richness	Green patch count	-0.21	0.104	-0.413	-0.007	4.101	1	0.043	*
250 m	Species richness	Nearest green patch distance	-0.326	0.122	-0.564	-0.087	7.163	1	0.007	**
250 m	Species richness	Matrix hedge density	0.094	0.115	-0.132	0.319	0.665	1	0.415	
250 m	Species richness	Matrix watercourse density	-0.004	0.112	-0.223	0.216	0.001	1	0.975	
250 m	Species richness	Matrix road density	-0.044	0.091	-0.223	0.135	0.234	1	0.628	
250 m	Total CPUE	Matrix green proportion	0.031	0.102	-0.17	0.232	0.091	1	0.763	
250 m	Total CPUE	Green patch count	-0.444	0.087	-0.614	-0.274	26.186	1	<0.001	***
250 m	Total CPUE	Nearest green patch distance	-0.133	0.09	-0.309	0.042	2.221	1	0.136	
250 m	Total CPUE	Matrix hedge density	0.11	0.096	-0.079	0.299	1.307	1	0.253	
250 m	Total CPUE	Matrix watercourse density	-0.102	0.09	-0.279	0.075	1.265	1	0.261	
250 m	Total CPUE	Matrix road density	-0.226	0.074	-0.371	-0.082	9.445	1	0.002	**
250 m	Simpson's reciprocal diversity	Matrix green proportion	0.129	0.1	-0.066	0.324	1.682	1	0.195	
250 m	Simpson's reciprocal diversity	Green patch count	-0.199	0.082	-0.36	-0.038	5.853	1	0.016	*
250 m	Simpson's reciprocal diversity	Nearest green patch distance	-0.382	0.094	-0.566	-0.198	16.518	1	<0.001	***

Buffer	Response	Predictor	Estimate	SE	CI low	CI high	Wald chisq	Df	p value	Significance
250 m	Simpson's reciprocal diversity	Matrix hedge density	0.071	0.088	-0.102	0.244	0.65	1	0.420	
250 m	Simpson's reciprocal diversity	Matrix watercourse density	0.052	0.084	-0.113	0.217	0.384	1	0.535	
250 m	Simpson's reciprocal diversity	Matrix road density	0.005	0.07	-0.133	0.143	0.005	1	0.943	

Significance codes: † $p < 0.10$; * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$. The dagger (†) indicates a marginal effect that did not meet the conventional $p < 0.05$ threshold. Blank cells indicate $p \geq 0.10$.

Appendix F.3: Chapter 4 – matrix-only models fitted for the 500 m buffer

Table F.3. Full fixed-effect outputs for the final Chapter 4 matrix-only models fitted at the 500 m buffer scale. Estimates represent standardised fixed-effect coefficients. Confidence limits are 95% confidence intervals, and Wald χ^2 statistics and *p*-values were derived from Type II tests.

Buffer	Response	Predictor	Estimate	SE	CI low	CI high	Wald chisq	Df	p value	Significance
500 m	Species richness	Matrix green proportion	0.109	0.106	-0.099	0.317	1.058	1	0.304	
500 m	Species richness	Green patch count	-0.199	0.083	-0.361	-0.036	5.713	1	0.017	*
500 m	Species richness	Nearest green patch distance	-0.339	0.117	-0.569	-0.109	8.324	1	0.004	**
500 m	Species richness	Matrix hedge density	0.081	0.1	-0.114	0.276	0.66	1	0.417	
500 m	Species richness	Matrix watercourse density	-0.003	0.083	-0.166	0.16	0.001	1	0.969	
500 m	Species richness	Matrix road density	-0.024	0.093	-0.205	0.158	0.067	1	0.796	
500 m	Total CPUE	Matrix green proportion	-0.116	0.083	-0.278	0.046	1.978	1	0.160	
500 m	Total CPUE	Green patch count	-0.495	0.072	-0.637	-0.354	47.246	1	<0.001	***
500 m	Total CPUE	Nearest green patch distance	-0.332	0.085	-0.498	-0.166	15.306	1	<0.001	***
500 m	Total CPUE	Matrix hedge density	0.322	0.075	0.176	0.468	18.601	1	<0.001	***
500 m	Total CPUE	Matrix watercourse density	0.085	0.065	-0.042	0.212	1.718	1	0.190	
500 m	Total CPUE	Matrix road density	-0.259	0.072	-0.4	-0.118	12.889	1	<0.001	***
500 m	Simpson’s reciprocal diversity	Matrix green proportion	0.24	0.08	0.082	0.398	8.896	1	0.003	**
500 m	Simpson’s reciprocal diversity	Green patch count	-0.192	0.061	-0.312	-0.072	9.89	1	0.002	**
500 m	Simpson’s reciprocal diversity	Nearest green patch distance	-0.439	0.075	-0.586	-0.292	34.224	1	<0.001	***

Buffer	Response	Predictor	Estimate	SE	CI low	CI high	Wald chisq	Df	p value	Significance
500 m	Simpson's reciprocal diversity	Matrix hedge density	0.052	0.07	-0.085	0.19	0.555	1	0.456	
500 m	Simpson's reciprocal diversity	Matrix watercourse density	0.128	0.064	0.002	0.254	3.971	1	0.046	*
500 m	Simpson's reciprocal diversity	Matrix road density	0.099	0.065	-0.029	0.227	2.307	1	0.129	

Significance codes: † $p < 0.10$; * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$. The dagger denotes a marginal effect and not statistical significance at $p < 0.05$. Blank cells indicate $p \geq 0.10$.

Appendix G: Supplementary statistical outputs for Chapter 5

This appendix presents supplementary outputs for the final Chapter 5 combined CPUE model and its spatial validation. It includes the comparison between the additive model and the alternative grassland cover × management intensity interaction model, complete fixed-effect estimates for the retained model, overall model-fit statistics, observed-versus-fitted and leave-one-site-out validation results, environmental applicability screening, and summaries of predictions under alternative mowing scenarios. The final model combined standardised patch-scale and 500 m landscape-scale predictors and was fitted using a Tweedie distribution with a log link, with Site ID included as a random intercept to account for repeated surveys within green space patches.

Appendix G.1: Chapter 5 - comparison of candidate models evaluated during development

Table G.1 Comparison of candidate models evaluated during development of the final Chapter 5 combined CPUE model. The retained additive model was compared with an alternative model containing a grassland cover × management intensity interaction. Model comparison statistics include the number of estimated parameters, Akaike Information Criterion (AIC), Bayesian Information Criterion (BIC), log-likelihood and the Type II Wald test for the interaction term. The additive model was retained because inclusion of the interaction did not improve model performance and the interaction was not statistically significant.

Model	Selected	AIC	ΔAIC	BIC	Log-likelihood	Interaction estimate	SE	Wald χ^2	p value
Final additive model	Yes	459.119	0.000	485.659	−219.560	—	—	—	—
Grassland cover × management interaction model	No	461.119	2.000	490.313	−219.560	0.001	0.115	0.000	0.9962

Appendix G.2: Chapter 5 – final overall model outputs

Table G.2. Complete fixed-effect outputs for the final Chapter 5 combined CPUE model. Estimates are standardised fixed-effect coefficients on the log-link scale from a Tweedie generalised linear mixed model. Standard errors and 95% confidence intervals are shown alongside Wald χ^2 statistics and *p*-values derived from Type II tests. Site ID was included as a random intercept.

Term	Estimate	SE	CI low	CI high	Wald_Chi_square	Df	p value	Significance
Intercept	2.263	0.064	2.138	2.387				
Management intensity	-0.296	0.088	-0.468	-0.124	11.417	1.000	0.0007	***
Grassland cover	0.045	0.097	-0.145	0.235	0.218	1.000	0.6406	
Within-patch woody linear density	0.332	0.068	0.198	0.466	23.578	1.000	0.0000	***
Green patch count	-0.226	0.080	-0.383	-0.070	8.034	1.000	0.0046	**
Matrix hedge density	0.127	0.067	-0.004	0.259	3.614	1.000	0.0573	†
Matrix road density	-0.200	0.101	-0.397	-0.003	3.949	1.000	0.0469	*

Significance codes: † 0.05 ≤ *p* < 0.10; * *p* < 0.05; ** *p* < 0.01; *** *p* < 0.001. The dagger denotes a marginal effect rather than statistical significance at *p* < 0.05. Blank cells indicate *p* ≥ 0.10.

Table G.3. Summary of model structure, fit and convergence for the final Chapter 5 combined CPUE model. Marginal R^2 represents the variation explained by the fixed effects, while conditional R^2 represents the variation explained by the fixed and random effects together. The positive-definite Hessian indicated successful numerical convergence of the fitted model.

Model information	Value	Fit statistic	Value
Response variable	Total CPUE	AIC	459.119
Distribution and link	Tweedie; log	BIC	485.659
Survey observations	105	Log-likelihood	−219.560
Survey sites	21	Marginal R^2	0.747
Random-intercept SD	0.252	Conditional R^2	0.870
Dispersion parameter	0.831	Positive-definite Hessian	Yes

Table G.4. Predictive performance of the final Chapter 5 combined CPUE model. Performance was evaluated using an observed-versus-fitted comparison for the surveyed training data and leave-one-site-out site-blocked cross-validation. Metrics comprise root mean square error (RMSE), mean absolute error (MAE), Pearson correlation and Spearman rank correlation between observed and predicted CPUE. Cross-validation predictions were generated for each site after fitting the model to the remaining sites, with predictor standardisation recalculated within each training fold.

Validation	n	RMSE	MAE	Pearson r	Spearman r
Observed versus fitted (fixed effects only)	105.000	4.635	3.447	0.850	0.848
Leave-one-site-out site-blocked cross-validation	105.000	6.313	4.596	0.719	0.771

Table G.5. Environmental applicability of the Chapter 5 spatial prediction dataset. Unsurveyed green space patches were classified according to whether all retained predictor values fell within the environmental range represented by the surveyed training data. Counts and percentages are presented for patches inside and outside the supported prediction domain. Only patches inside the training range were retained for final mapped prediction.

Prediction domain	Patches	Percentage
Inside training range	598.000	47.4980
Outside training range	661.000	52.5020
Total unsurveyed prediction frame	1259.000	100.0000

Table G.6. Summary of predicted CPUE under low, moderate and intensive mowing scenarios across environmentally comparable unsurveyed green space patches. Results are restricted to patches classified as inside the environmental range of the training data. For each management scenario, the table presents the number of patches, minimum, mean, median and maximum predicted CPUE, mean 95% confidence-interval width and the number of predictions exceeding the map-display cap. Summary statistics are based on uncapped model predictions; the cap of 30 was applied only when displaying the maps.

Management scenario	Patches	Min CPUE	Mean CPUE	Median CPUE	Max CPUE	Mean 95 CI width	Patches above display cap
Low mowing	598.000	5.450	9.828	9.338	27.887	9.222	0.000
Moderate mowing	598.000	3.798	6.849	6.507	19.433	5.241	0.000
Intensive mowing	598.000	2.646	4.772	4.534	13.541	3.902	0.000

Table G.7. Differences in predicted CPUE between alternative mowing scenarios and the moderate-mowing reference scenario. Differences were calculated for each environmentally comparable unsurveyed patch as low mowing minus moderate mowing and intensive mowing minus moderate mowing. Positive values indicate higher predicted CPUE relative to moderate mowing, whereas negative values indicate lower predicted CPUE. The table summarises the distribution and magnitude of predicted management-related changes across the retained spatial prediction frame.

Comparison	Patches	Min difference	Mean difference	Median difference	Max difference
Intensive mowing minus moderate mowing	598.000	-5.891	-2.076	-1.973	-1.151
Low mowing minus moderate mowing	598.000	1.652	2.980	2.831	8.454

End of Thesis