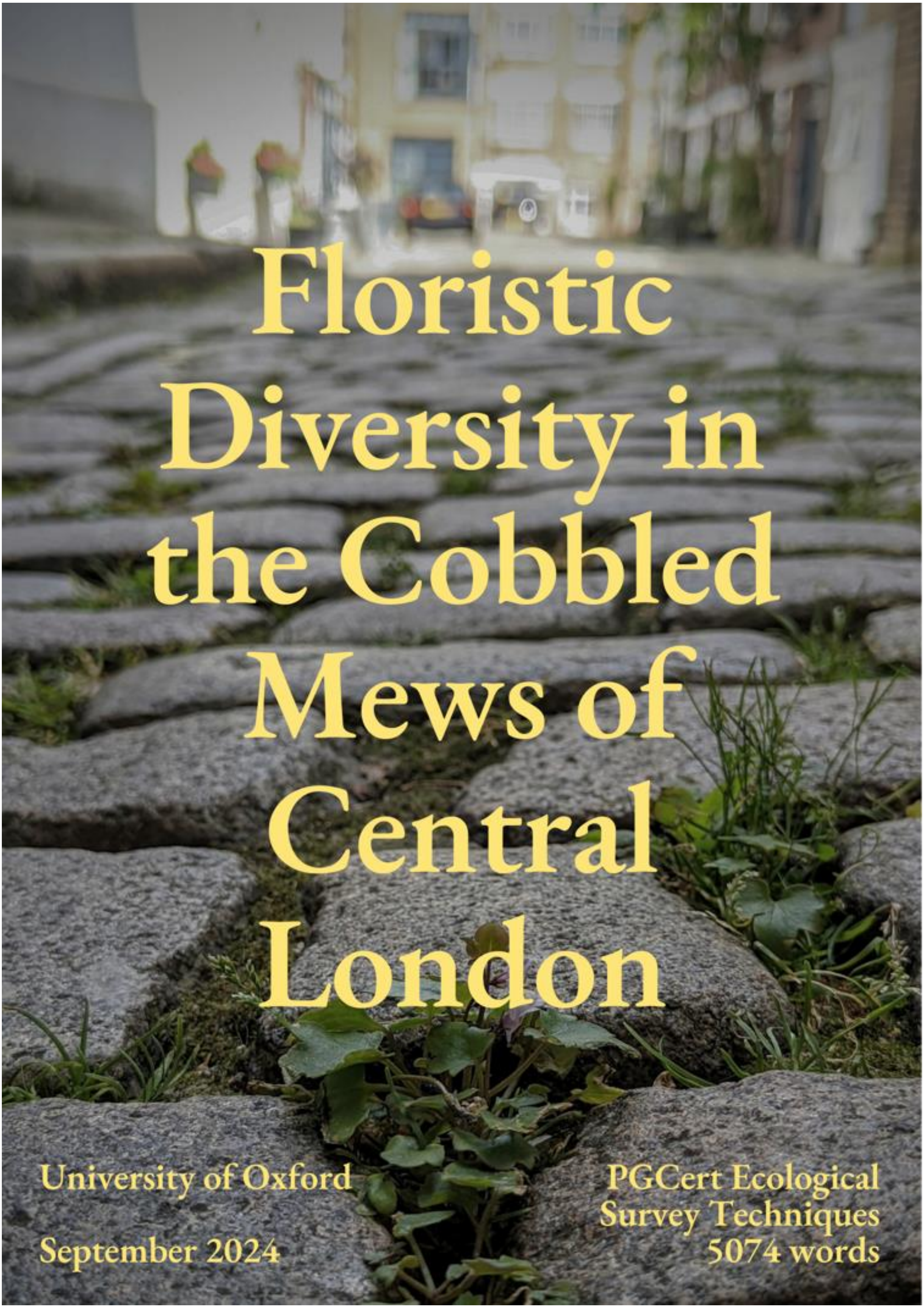

Floristic Diversity in the Cobbled Mews of Central London

University of Oxford

September 2024

PGCert Ecological Survey Techniques

5074 words

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Cover image: Ivy-leaved toadflax in a cobbled mews (author's own)
Word count: 5074

Abstract

Impervious sealing of soil surfaces in urban environments has an inhibiting effect on plant life, leading to reduced urban biodiversity. Pioneer species often establish on broken surfaces such as the cobbled mews of central London. This study uses a citizen science methodology to compare the biodiversity and percentage cover of vascular plants in ten pairs of mews, in which one is cobbled and the other tarmacked. To do so, three parallel transect lines were laid along the full length of each mews and sampled with quadrats every 20m. The influence of distance from the entrance and the edge of the mews is also explored.

Both percentage cover and biodiversity are significantly higher in cobbled mews, but there is no significant impact of the location within a mews on cover. Management recommendations to maintain and conserve cobbled mews are suggested.

Introduction

Plant Diversity on Urban Hard Surfaces

Urban ecosystems are increasingly common – and thus important – but often remain overlooked (Goddard, Dougill and Benton, 2010; Soanes *et al.*, 2019). Cities epitomise global ecological trends – for example, the urban species mix is increasingly homogenous and non-native (McKinney, 2006) – and highlight the impact of anthropogenic disturbance. A meta-analysis (Hou *et al.*, 2023) found urbanisation to have a variable but overall negative impact on plants, especially native and herbaceous plants. Studies of city vegetation often focus on formal green spaces such as parks and gardens (e.g. Wilby and Perry, 2006; Goddard, Dougill and Benton, 2010; Aronson *et al.*, 2017) but there are many different urban ecosystems (Godefroid and Koedam, 2007) and 15% of global urban areas are impervious, such as roadways and buildings (Liu *et al.*, 2014). Such pavements are sometimes considered to have "zero productivity" (McKinney, 2010, p.290) with resource constraints constraining growth in these environments (Wheater, 2020). Impervious surfaces suffer from less infiltration, more runoff and higher temperatures (Chithra *et al.*, 2015). Altogether, it has been suggested that surfacing streets is more environmentally destructive than logging, mining and farming (McKinney, 2006).

Nonetheless, urban surfaces are subject to colonisation by pioneer species (**Error! Not a valid bookmark self-reference.**), undergoing early stages of primary succession (Gilbert, 1991). Productivity is *not* zero; the abundance of plants growing on paved ground can be remarkable. Uchida *et al* (2014) found 113 plant species on urban paving cracks in Fukuoka, Japan while Krüsi and Trachsel (2012) reported 213 species of vascular plants using a systematic ArcGIS grid over Zurich, Switzerland. This so-called spontaneous urban vegetation can provide important ecosystem services, including climate regulation and habitat provisioning, often better than other urban flora such as lawns or even urban forest (Robinson and Lundholm, 2012). The little research that exists points to the importance of surface type (e.g. Godefroid and Koedam, 2004; Godefroid, Monbaliu and Koedam, 2007; Bonthoux *et al.*, 2019). For example, Bonthoux *et al* (2019, p.169) "found pavement type to be the strongest determinant of plant cover and species richness". De Cauwer *et al* (2014) even found, in an experimental study, that altered paving type could be used as an effective weed-suppressor.

Figure 1 Oxalis corniculata growing in a cobbled mews

Many other factors also strongly influence establishment on paved surfaces, including dispersal ability, age of stonework, construction methods, weathering, position, temperature, nutrient and moisture availability, pollution and trampling (Wheater, 2020). Trampling by cars can be particularly impactful, since it not only causes direct harm but also redistributes or compacts soil, minimising bacterial nitrification (Benvenuti, 2004; Lundholm, 2011). Exhaust emissions and salting of roads add to the impact from cars (Gilbert, 1991). The intensity of use of a road can therefore influence the community (Fagot *et al.*, 2011).

Plants that grow within the cracks in road surfaces have a wide variety of adaptations to survive in this habitat, from the production of adventitious roots in *Veronica arvensis* (Harris and Lovell, 1980) to ballistic seed dispersal in *Cardamine hirsuta* (Vaughn, Bowling and Ruel, 2011). Many of these plants are spring or early-summer annuals that require exposure to light in order to germinate (Baskin and Baskin, 1983; Boutin and Harper, 1991) and so are well adapted to germination on bare surfaces. *Oxalis corniculate* (**Error! Not a valid bookmark self-reference.**), *Veronica arvensis, Draba verna, Stellaria media* and several of the other most common mews plants have a small, tough and spreading growth form which mitigates against trampling.

Mews

A potential haven for spontaneous vegetation in London are the many mews (Figure 2). Mews are residential streets that were historically used as accommodation for horses or falcons and the stablehands who looked after them, but were converted to highly desirable housing (McManus, 2004; Fowler, 2019). Across London there are 432 of the original mews still remaining, of which nearly half (195) are in Westminster (Brown, 2021a). They are generally, but not exclusively, under 100m long and - crucially - often retain original features such as cobbled surfaces, within which soil accumulates and a range of herbaceous pioneer species are able to survive. Cobbled mews thus represent what Gilbert (1991, p.2) terms an 'ecological' (as opposed to 'technological' or 'gardenesque') urban landscape, as "unofficial wild spaces where nature has gained a foothold".

Qualitative surveying for this study indicated an approximately equal number of thoroughfare and cul-de-sac mews. The majority - 71% - of mews are cobbled (Brown, 2021a), but many are paved with tarmack and not all cobbles are original and so the shape of, and the mortar between, cobbles varies. All surveyed mews are open to traffic, which generally drives down the centre of the mews, potentially concentrating trampling and pollution there.

Figure 2 Wythburn Place, a typical cobbled thoroughfare mews

Citizen Science

Bonthoux *et al* (2019) recommend raising awareness of the value of spontaneous vegetation and in this study an opportunity was provided to engage teenage citizen scientists in learning about these neglected ecosystems through the collection of authentic data. Not only does citizen science allow for the collection of more data, but there is evidence to suggest that curriculum-linked citizen science improves the science knowledge and skills of the scientists themselves (Bonney *et al.*, 2015). This is done most effectively when the participation goes beyond just data collection and engages citizens in authentic and holistic co-construction of the research (Loiselle, Thornhill and Bailey, 2016; Phillips *et al.*, 2019). Citizen science data can be comparably reliable to professional data collection (Aceves-Bueno *et al.*, 2017) but this relies on preconditions such as comprehensive training (Brown and Williams, 2019). Both of these points hold true for adolescents, although it is important that tasks are adapted for accessibility and that data quality is monitored (Castagneyrol *et al.*, 2020).

Aims and Research Questions

This study seeks to identify and analyse the factors influencing urban plant abundance and biodiversity, to inform conservation management strategies. This is in line with the City of Westminster's Greener City Action Plan 2015-2025, which supports the Mayor of London's aim of "increasing the vegetated cover of the Central Activities Zone by 5% by 2030" (City of Westminster, 2015, p.30). Cobbled mews are an under-studied habitat – London's own guidance about "plants on roofs, walls and pavements" (Johnston and Newton, 2004) barely even mentions paved surfaces – and a better understanding of their status could inform conservation management strategies both in London and urban settings in general.

Citizen Science approaches are in line with UK government policy objectives, for example the 25 Year Plan set out by the previous government aimed to encourage children to be "close to nature" and have "contact with local natural spaces" whilst also aiming at "greening our towns and cities"(HM Government, 2018, p.13). This study seeks to enhance all three of these areas.

Four research questions (hereafter RQs) are posed:
RQ1 - What is the relationship between *surface type* and *percentage cover* of plants?
RQ2 - What is the relationship between *surface type* and *Simpson diversity index* of vascular plants?
RQ3 - What is the relationship between *distance from mews entrance* and *percentage cover* of vascular plants in cul-de-sac mews?
RQ4 - What is the relationship between *proximity to mews edge* and *percentage cover* of plants?

RQ1 and 2 seek to understand the impact of surface type on floristic diversity. RQ3s and 4 address the impact of trampling by vehicles on floristic diversity.

Methodology

This study compared the biodiversity and percentage cover of vascular plants in ten pairs of mews, in each of which one of the pair is cobbled and the other is tarmacked (Figure 3). Three parallel transect lines were laid along the full length of each mews and sampled with quadrats every 20m by citizen scientists in collaboration with the lead researcher. All references below to 'biodiversity' and "percentage cover" refer to plants only.

Data were collected between 28 March and 17 May 2024.

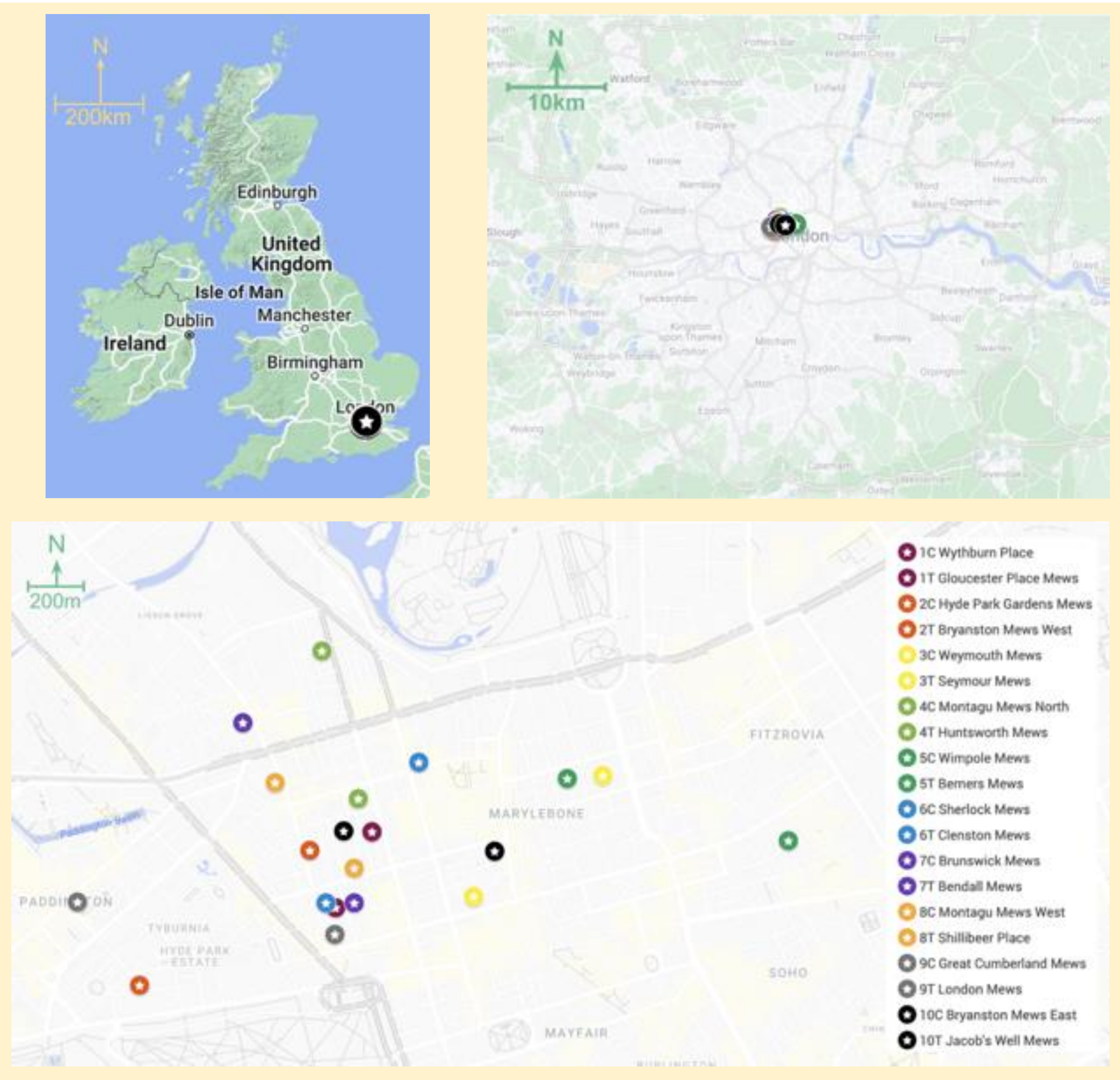

Figure 3 Location maps for UK, London and the sampled mews within Westminster (Google Maps, 2024)

Constraints

All research is subject to constraints that influence the choice of methodology. Here the need to fit sampling within the two-hour science lessons of the citizen scientists and their relative inexperience in natural history and ecological surveying are key constraints.

Choice of sites

All sample sites were in the City of Westminster, London, UK (Figure 3) and within twenty minutes walk of Halcyon School (51.5161° N, 0.1612° W), which the citizen scientists attend. The area is close to Oxford Street (a busy shopping street) and Hyde Park (a large urban green space) and so is typical of London, which is approximately 18% public open space (GiGL, 2024).

Sample mews were identified by walking surveys (akin to Godefroid, Monbaliu and Koedam, 2007) and the extensive use of Google Maps™ and Google Streetview™ to rapidly visit multiple distant locations (Curtis *et al.*, 2013; Berland and Lange, 2017). Ten pairs of mews were selected using the following criteria, which allowed for valid comparison while ensuring sufficient data.

Criteria

- Half of the pairs should be cul-de-sacs and half thoroughfares.
- Ten mews should have a cobbled surface, ten should have a tarmacked surface.
- Prioritise straight mews.
- Prioritise mews at least 50m long.
- Side branches of a mews may be surveyed as part of a thoroughfare only.

Exclude any mews with:

- Many potted plants (e.g. Southwick Mews).
- Potentially dangerous levels of traffic cars (e.g. Brook's Mews).
- Private road status (e.g. William Mews).
- Mixed surface type (e.g. Quebec Mews) or surface other than cobbles or tarmac.
- Active construction.

Krüsi and Trachsel (2012) used a similar approach to the present study, selecting the closest 'individual' (sampling area) that met a set of criteria that would allow for the collection of sufficient relevant data.

Pairs were established on the following basis:

- One was cobbled, the other tarmacked.
- Both were thoroughfares or both were cul-de-sacs.
- Sampling took place on the same day.
- The mews were in relatively close proximity.
- The length of the mews was similar.

Further shared features such as building height were noted but ultimately too restrictive to form sufficient pairs.

A paired design is appropriate because the ecological community in street vegetation can vary depending on proximity to human activities (Pauleit and Breuste, 2011) and time of year (Han et al., 2023) and so it was felt to be important to avoid comparison where these were an explanatory factor for differences.

It would have been desirable to record values for abiotic conditions in each of the mews to quantify their effects on biodiversity (and thus disentangle those effects from surface type) but in preliminary trials it was found to be impossible to collect both biotic and abiotic data in the time available. An alternative would be to use Ellenberg indicator values, which use the species present as indicators of the abiotic conditions, this is unfeasible in the present study (Hill, 1999; Godefroid, Monbaliu and Koedam, 2007).

Training of citizen scientists

Four groups of students aged 13-17 were trained to participate in data collection. Only students aged 13+ participated since evidence suggests that children get more accurate in their species identification as they age (Dickinson, Zuckerberg and Bonter, 2010). All citizen scientists were given at least one hour of training in plant ID and the experimental methodology and then participated in a practice data collection trial (Figure 4) For further details see appendix 1.

Figure 4 Citizen scientist practicing data collection techniques.

Transect set-up (sampling strategy)

Parallel transects can be used to sample biodiversity, sometimes with quadrats placed randomly along the transect (e.g. Bridle *et al.*, 2009) and sometimes with them placed systematically, for example every 10m along a 100m transect (Collaborative Forest Restoration Program, 2003). In this study, the latter approach was taken, enabling correlation between distance from entrance and biodiversity.

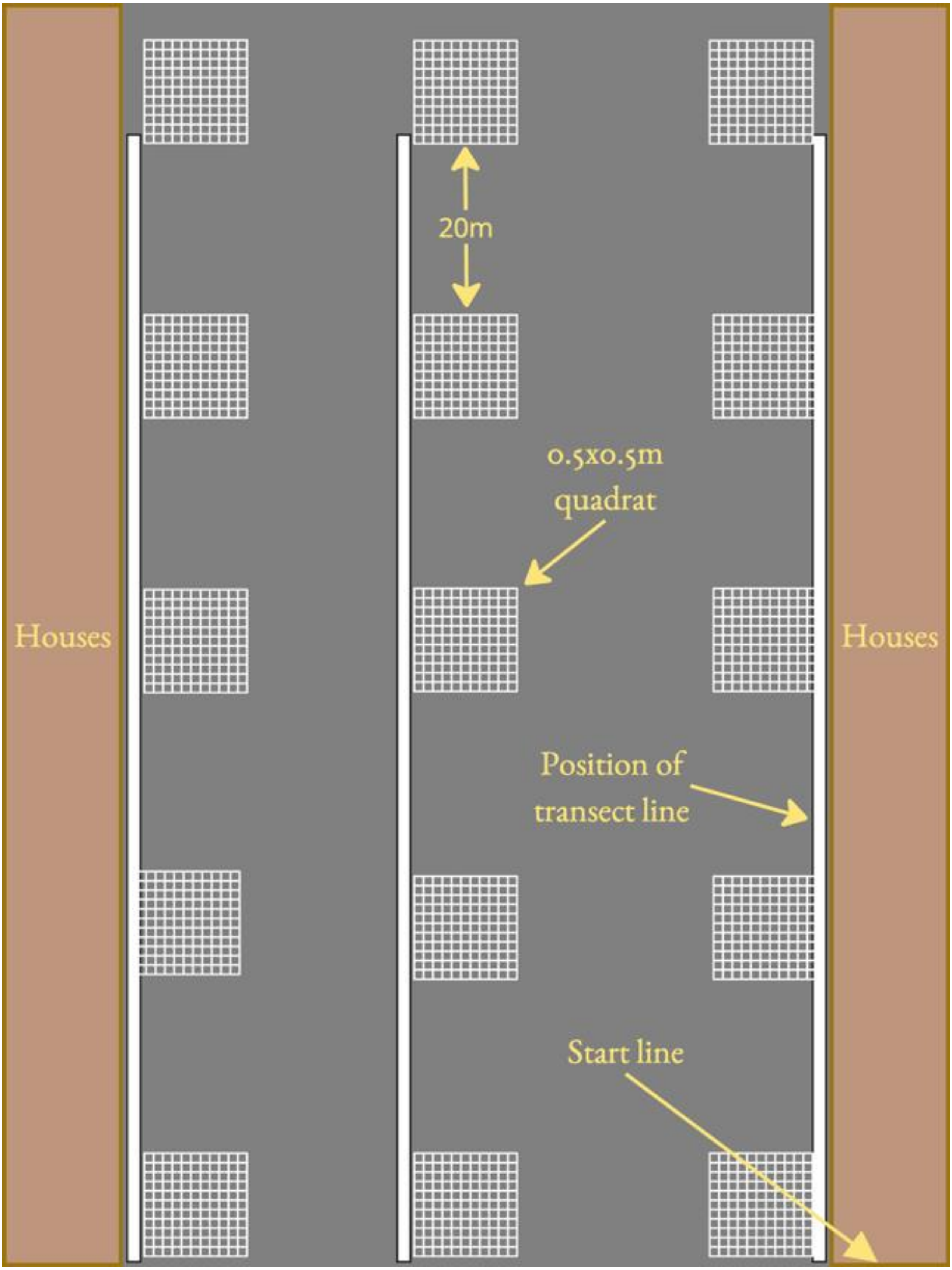

Figure 5 Set up of transects. Not to scale. Transect line shown is for illustration - 20m surveyor tape was used. Shape of mews is idealised; not all are perfectly rectangular and transect lines flexibly tracked the edge in those cases.

Three transect lines were laid within each mews, one in the centre and one tracking each edge (Figure 5). For consistency, the start of the transect was defined as being in line with the outer (gable-end) wall of the first buildings in the mews (for example, the shaded wall on the far left of Figure 2). Every 20m along each transect, a quadrat was placed, and observations made (see below). The mean length of mews in the present study was 94 metres (the mode was 80) and so sampling every 20m ensured a sufficient but feasible number of data points per mews. This method was preferable to dividing the length by a fixed value as it ensures an equal proportion of each mews is surveyed. A 20m surveyor tape was laid down the centre of the mews and used to mark out the distance to each new sample point - a short tape allowed rapid movement in response to traffic. Citizen scientist spotters guided the placement of quadrats in line with the 20m mark.

The edge transects were positioned as close as possible to the side walls; a metre ruler was used to indicate a maximum allowable 1m distance from the wall in rare cases where obstacles prevented them being flush with the wall. Having two edge quadrats reflects the anecdotal observation from preliminary investigations that vegetation in mews tends to be concentrated at the edges (perhaps due to trampling by cars driving down the middle, since mews are typically single-lane roads). This observation led to the formulation of RQ4, which asks whether this is a statistically significant relationship.

Sampling within quadrats (including plant ID)

Two approaches are typically available for determining optimum quadrat size - a species-area curve or the use of a generalised table for different lifeforms (Kent, 2012). Within mews, the vegetation tends to be clumped in small patches and this lack of homogeneity in the vegetation - and the analysis of twenty different sites - means that a species-area curve is not suitable. In accordance with lifeform-based guidance in Kent (2012) 0.5x0.5m quadrats were therefore used. These had a mesh size of 50x50mm, providing 100 squares and facilitating determination of percentage cover.

When surveying spontaneous urban plants, it is common to use the Braun-Blanquet method (e.g. Godefroid, Monbaliu and Koedam, 2007; Uchida *et al.*, 2014), which estimates cover within categories rather than as raw percentages. However, the categories of Braun-Blanquet are insufficiently granular to distinguish the small differences in cover seen in this study.

Six simultaneous groups of students worked in pairs (Figure 6), filling out a data sheet (see appendix 3) by recording the following:

1. The location of, and length along, the transect.
2. The percentage coverage of all plants (including grasses and bryophytes). Each 1% square within the grid that was more than half covered by vegetation counted as 1% cover.
3. The percentage frequency for each species of plant (excluding grasses, due to their growth form and inability to distinguish the species of saplings). The total number of 50x50mm squares in which the species was present at all was recorded.

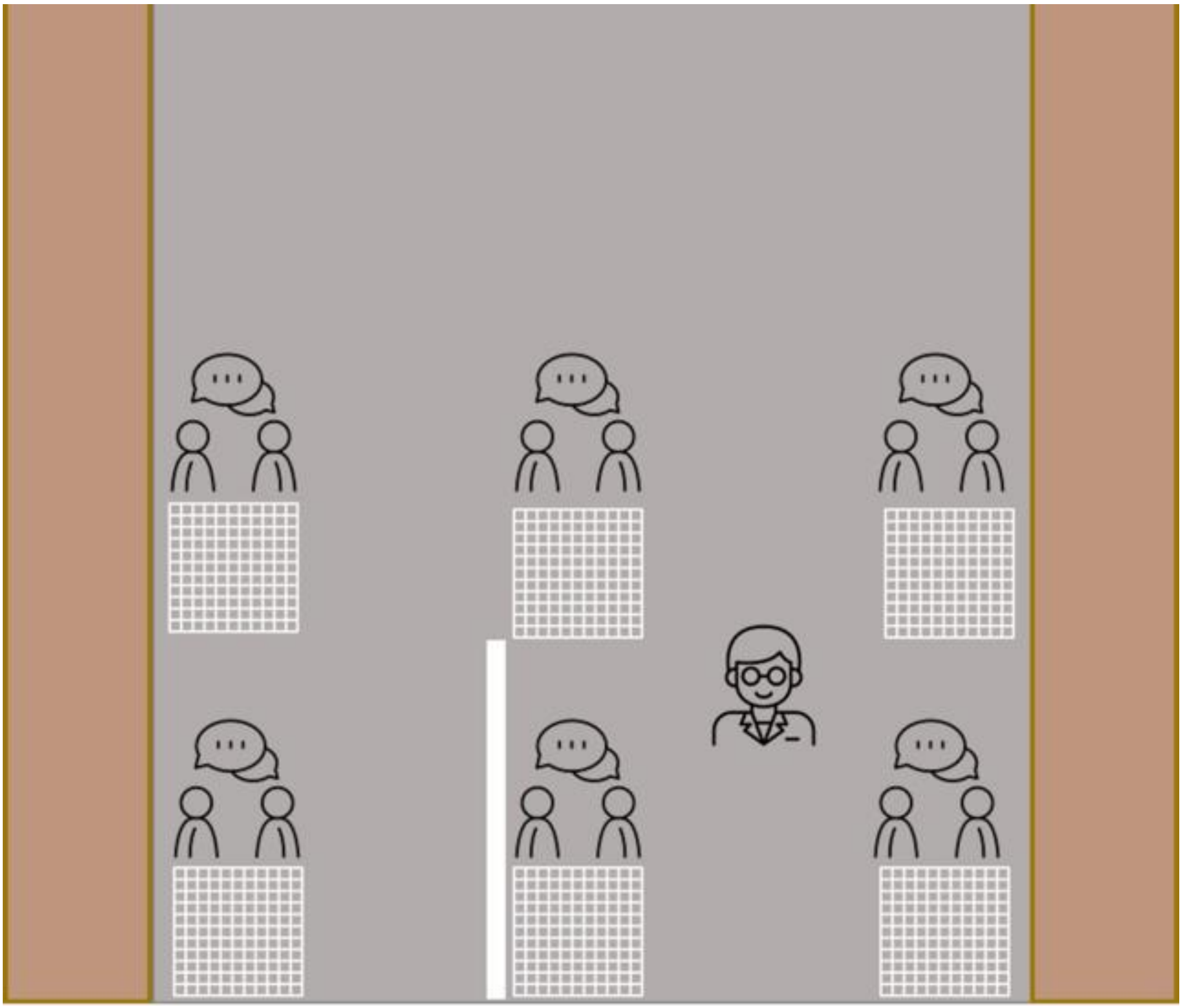

Figure 6 Six simultaneous pairs of students collaborating before digitally recording data. Groups smaller than 12 still worked in pairs but completed fewer quadrats simultaneously. Icon credit: Soontorn (2020).

Pairs were asked to individually determine values and then reach a consensus to record in the data sheet, to improve reliability of estimates. Data recording was done using shared online documents on mobile phones, allowing data quality to be monitored in the field. The lead researcher checked the data of the six simultaneous pairs after each round before collectively moving on to the next set of quadrats; controlling the pace at which tasks are completed mitigates some of the risk of difference in sampling effort that Dickinson *et al* (2010) highlight as an issue with citizen science.

The lifeform of each species was subsequently recorded (Dřevojan *et al.*, 2023 following Raunkiær, 1934) and each species was classified as native or non-native with reference to published literature (Darlington, 2020; Stroh *et al.*, 2020; NBN Trust, 2024) but data were unavailable for more granular distinctions based on when non-natives were introduced, as used by Kent (1999).

Following Krüsi and Trachsel (2012) and Kent (1999) mosses and liverworts were not identified to species level and were excluded from most quantitative analyses.

One mews, Wythburn Place, was re-recorded because of insurmountable issues with data quality. The data for its partner, Gloucester Place Mews, could not be formally repeated due to time constraints but was observed to remain very low in biodiversity.

Data Analysis

RQ1 - Relationship between surface type and percentage cover

For each mews, a mean percentage cover was found and from this a "mean of means" for each surface type was calculated (Table 1). For comparison, a "true mean" was also calculated by summing the percentage cover for all quadrats in all cobbled/tarmacked mews and dividing by the total number of quadrats. The values are similar regardless of the approach taken. Graphing (Figure 14) uses the "mean of means" because we are comparing the means of *pairs* of mews in the statistical analysis.

A paired t-test was used to compare the means of the two sets of paired data. This assumes the *differences* in percentage cover are normally distributed (McDonald, 2014), confirmed by the Q-Q plot in Figure 7 and a Shapiro-Wilk test (W = 0.885, n = 10, P = >0.05).

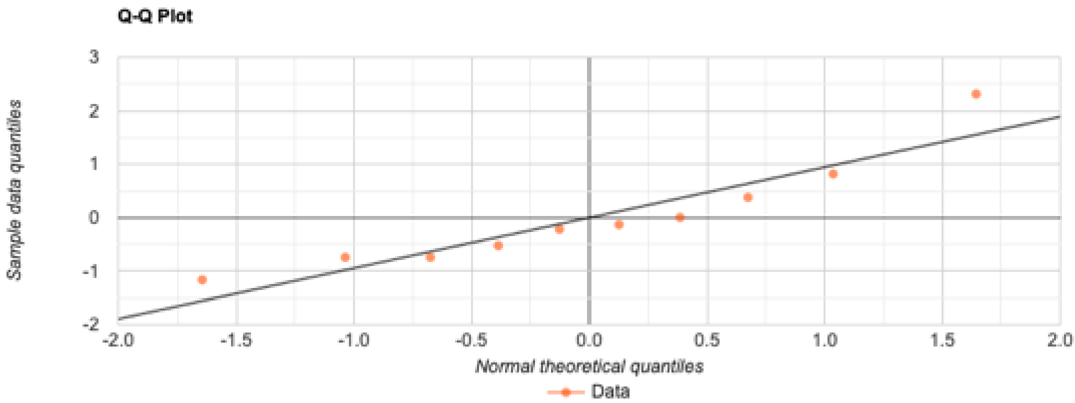

Figure 7 Probability plot for paired differences in percentage cover between surface types.

There was a single outlier in the differences, pair 4 at 15.9 (Tukey Fence, k=1.5) but this was not considered an error as it simply represents natural variation.

Since the aim is to test if cobbled mews have greater cover, and there is a theoretical basis to support this, a one-tailed test was used. Effect size was calculated to quantify the magnitude of the effect (Sullivan and Feinn, 2012).

RQ2 - Relationship between surface type and Simpson diversity index

Diversity was measured using the reciprocal Simpson diversity index ('SDI'), otherwise known as 'Hill-Simpson' or 'effective number'; this has a number of advantages over the standard Simpson index, including that it intuitively has the number of species as its maximum value and that it scales linearly with the loss of species (Jost, 2006; Roswell, Dushoff and Winfree, 2021). Since this value is sensitive to abundant species (Morris *et al.*, 2014), and because of their distinctive ecology, Bryophytes s.s. were excluded from the calculation of diversity.

Diversity was measured for each mews population as a whole and was therefore not suitable for RQs 3 and 4 (which used percentage cover instead).

As for RQ1, we compared means of two sets of paired data for which we assume cobbled mews is greater. However, this time the differences deviated from normal (Shapiro-Wilk test, $W = 0.806$, $n = 10$, $P = <0.05$ and Figure 8) and so even though paired-t tests are not sensitive to small deviations from normal (McDonald, 2014), the effect size here was large (Kolmogorov-Smirnov, KS - $D = 0.2391$). The data were therefore log(10) transformed, which produced normally-distributed differences (Shapiro-Wilk test, $W = 0.971$, $n = 10$, $P = >0.05$) so a one-tailed paired t-test was used.

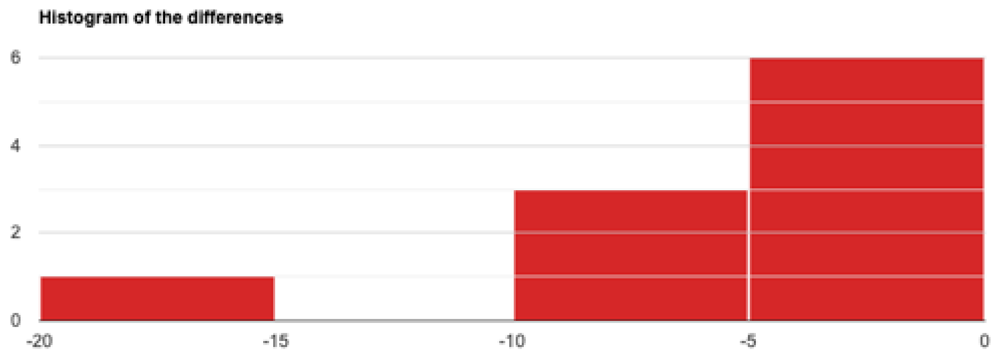

Figure 8 Histogram clearly shows non-parametric distribution of untransformed data

RQ3 - Relationship between distance from mews entrance and percentage cover

Figure 9 and a Shapiro-Wilk test ($W = 0.476$, $n = 141$, $P = <0.05$) showed the percentage cover variable deviated significantly from a normal distribution. This remained the case after $\log_{10}+1$ (+1 because of zeroes) and square-root transformations. A Spearman's rank correlation was therefore used.

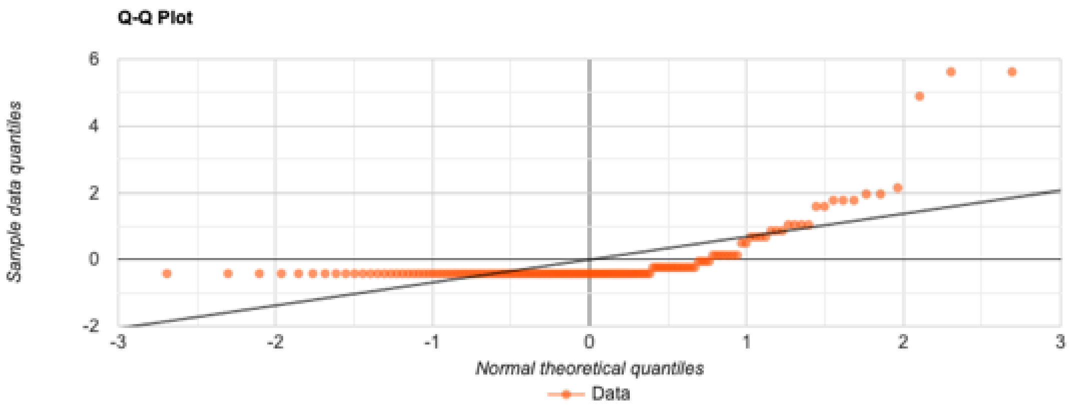

Figure 9 Probability plot of percentage cover variable shows non-parametric distribution.

RQ4 - Relationship between proximity to mews edge and percentage cover

Data exist for left, centre and right transects (Table 4) and thus we first tested the assumption that left and right are not significantly different and can thus be combined as 'edge'. Differences were normally distributed (Shapiro-Wilk test, $W = 0.921$, $n = 10$, $P = <0.05$) and a paired t-test showed no significant difference between left and right edges ($t = 0.192$, $DF = 9$, $P = >0.05$).

When combining left and right, pairs are lost and so normality must be tested within each sample; neither set was normally distributed (Shapiro-Wilk test, $W = 0.718$, $n = 20$, $P = >0.05$ and $W = 0.589$, $n = 10$, $P = >0.05$) even after $\log_{10}+1$ and square-root transformations. Therefore, a two-tailed Mann-Whitney U test was performed.

Results

Across all mews, 37 species were recorded (Figure 12). Half of all individual plants observed were of only three species (Figure 11) and 25% were *Draba verna* (Figure 13). As discussed below, almost all this diversity was in cobbled mews (Figure 10).

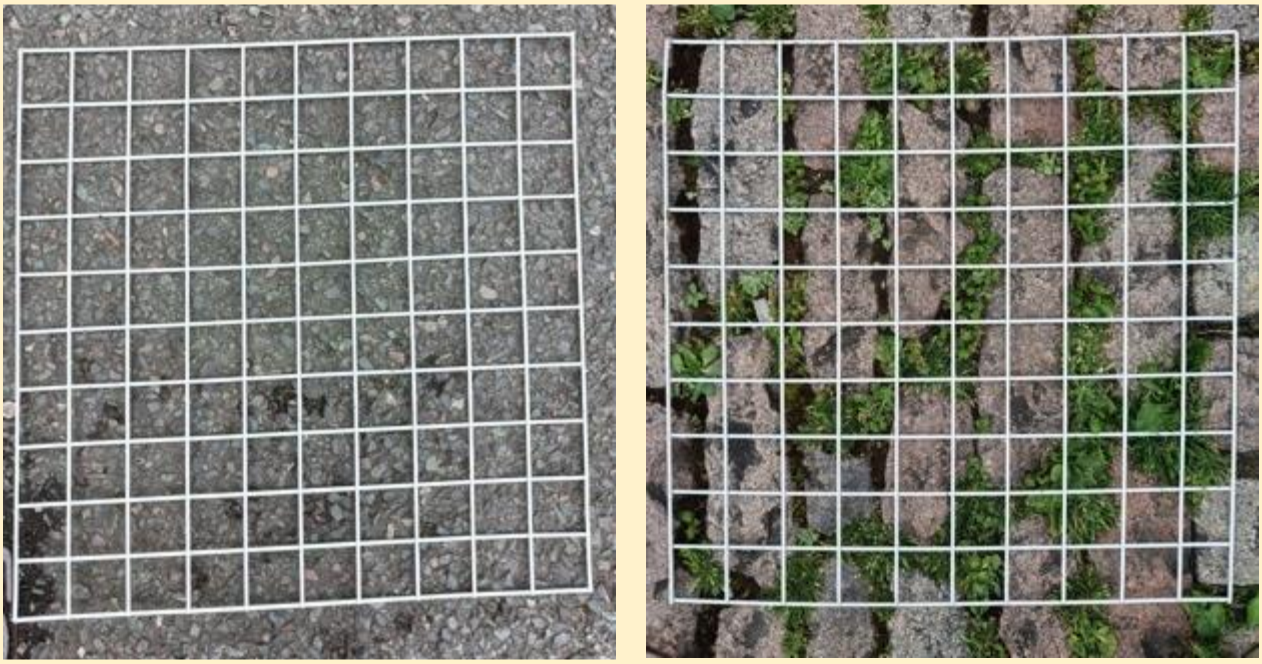

Figure 10 The two extreme ends of biodiversity; a typical quadrat in a tarmacked mews and an especially biodiverse one in a cobbled mews.

Figure 11 The four most abundant species in Westminster mews. Anti-clockwise from top left: Draba verna (Lockton, 2017) , Veronica arvensis (Siegmund, 2012), Plantago major (Taylor, 2012), Stellaria media (Hugo.arg, 2007).

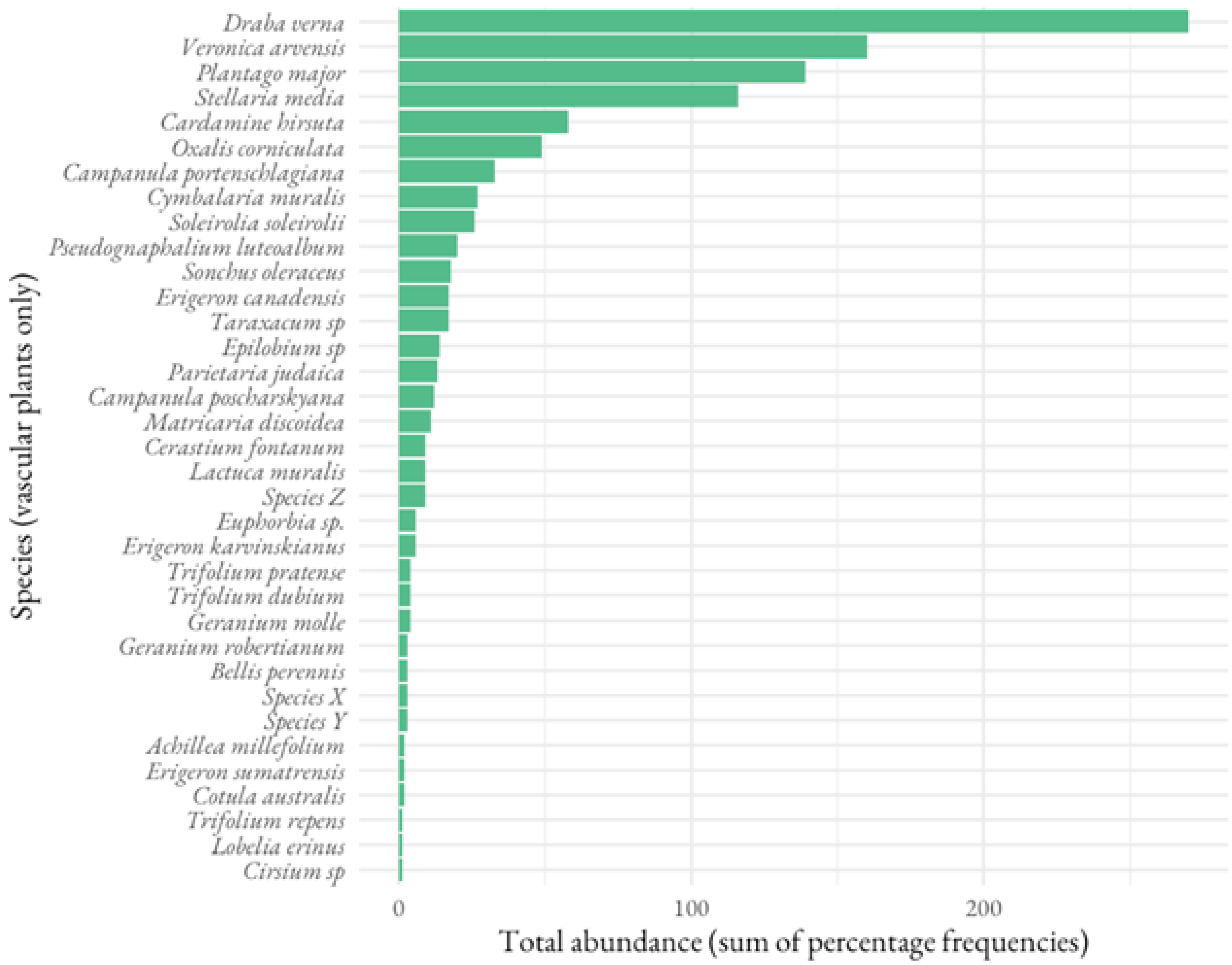

Figure 12 All 37 species identified, ranked by abundance.

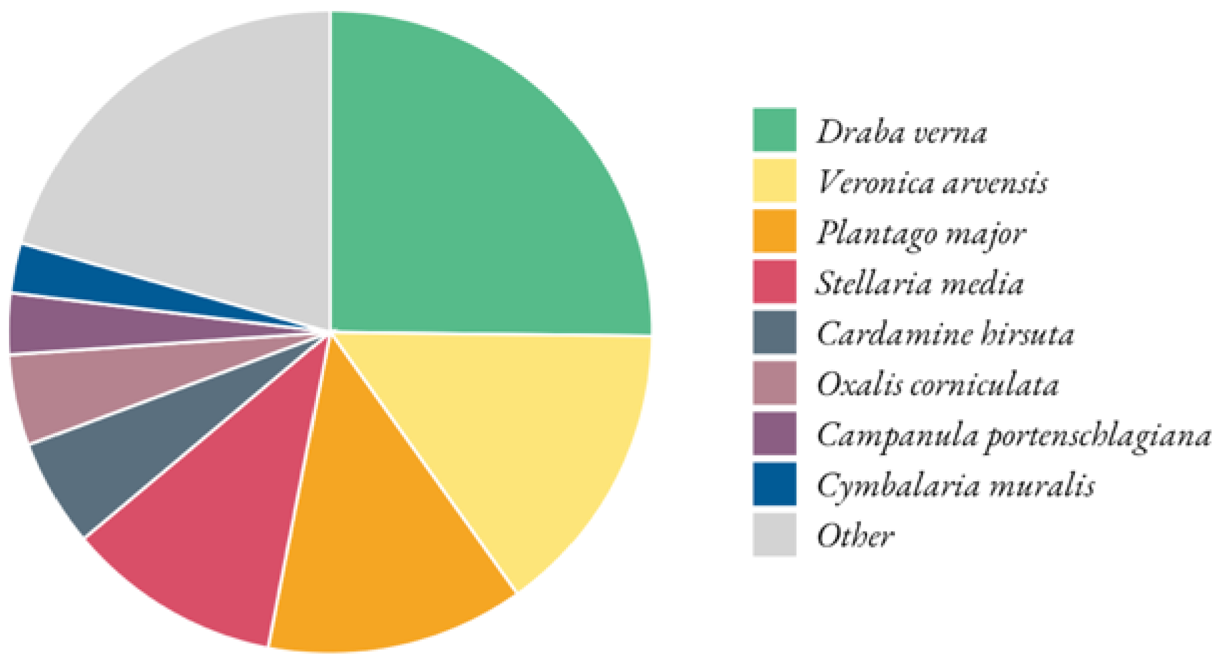

Figure 13 Pie-chart of abundances emphasises dominance of a few species.

RQ1 - Relationship between *surface type* and *percentage cover*

Percentage cover in cobbled mews is significantly larger than in tarmacked mews ($t = 3.76$, $DF = 9$, $P = <0.01$). The effect size (Cohen's d, 1.19) was large (Sullivan and Feinn, 2012).

Table 1 Mean percentage cover in mews with different surface types

Pair	Mean Percentage Cover	
	Cobbled	**Tarmacked**
1	4.5	0.1
2	7.3	2.0
3	4.9	0.1
4	16.0	0.1
5	3.2	0.1
6	10.7	1.6
7	7.3	0.2
8	2.3	0.3
9	0.2	0.1
10	2.1	0.1
Mean of Means	5.8	0.5
True' Mean	5.3	0.5
Standard Deviation	4.7	0.7
95% confidence interval	2.9	0.4

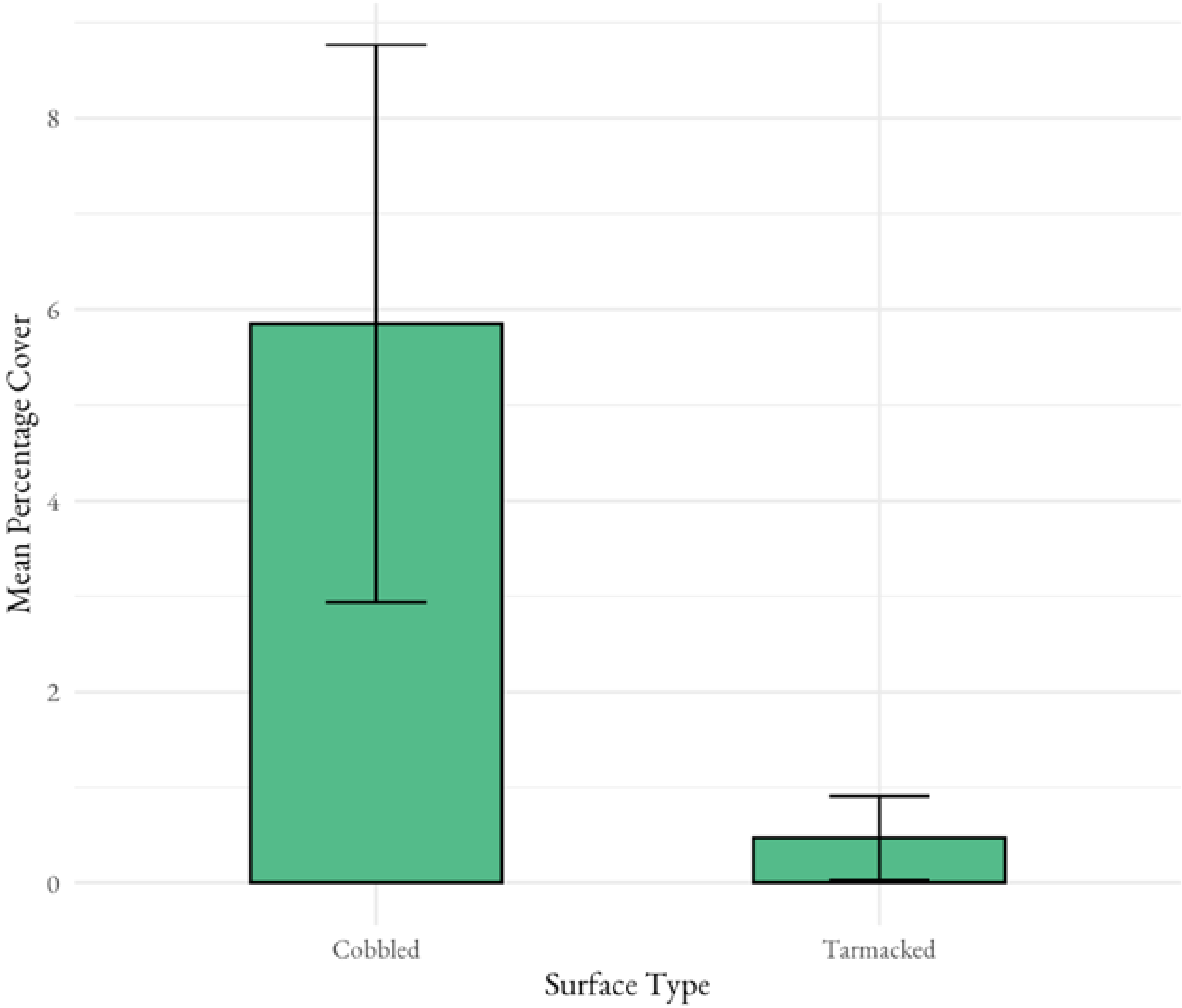

Figure 14 Mean percentage cover is much greater in cobbled mews . Error bars show 95% confidence intervals; variability is large.

RQ2 - Relationship between *surface type* and *Simpson diversity index*

SDI in cobbled mews is significantly larger than in tarmacked mews (t = -5.89, DF = 9, P = <0.01). The effect size (Cohen's d, 1.86) was large.

A box plot of the untransformed data (Figure 15) revealed minimal spread or overlap in the data. One outlier (cobbled 18.3) pulled the mean up to 6.0 from 4.6 without it.

Table 2 SDI in mews with different surface types – values vary but cobbled appears visibly much higher

Pair	Reciprocal Simpson Diversity Index	
	Cobbled	**Tarmacked**
1	2.2	1.0
2	18.3	2.7
3	3.7	3.6
4	3.7	1.0
5	6.1	1.0
6	3.9	1.0
7	3.5	1.0
8	8.0	1.0
9	3.9	2.1
10	6.6	0.6
Mean	6.0	1.5
Standard Deviation	4.7	1.0
95% CI	2.9	0.6

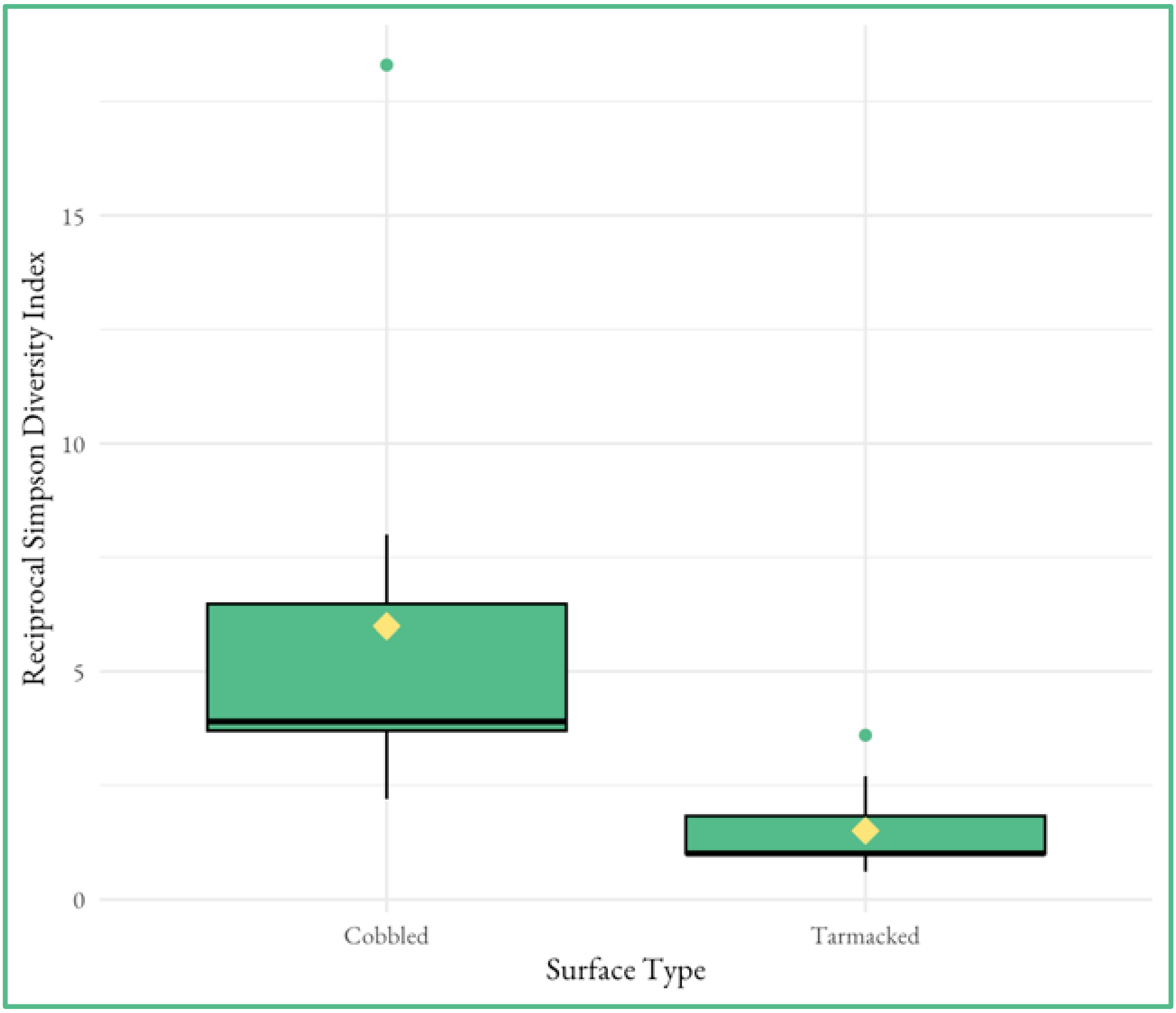

Figure 15 Boxplot of surface type and SDI highlights minimal variability. Means in yellow, outliers in green.

RQ3 - Relationship between *distance from mews entrance* and *percentage cover*

Spearman's rank correlation found no significant correlation between distance from entrance and percentage cover (r = -0.0560, DF = 141, P = >0.05). So-called 'p hacking' should be avoided (Head *et al.*, 2015) but there was no significance even for only the cobbled cul-de-sacs (r = -0.089, DF = 72, P = >0.05) or excluding the small-sample-size 100m values (r = -0.106, DF = 69, P = >0.05).

A boxplot of these data (Figure 16) shows that percentage cover values are clustered at low values but for almost every distance there are several noteworthy outliers.

Table 3 Distance from entrance and percentage cover

Distance from entrance (m)	Mean Percentage cover	Sample size
0	2.3	30
20	2.7	30
40	2.9	30
60	2.3	30
80	0.6	18
100	3.0	3

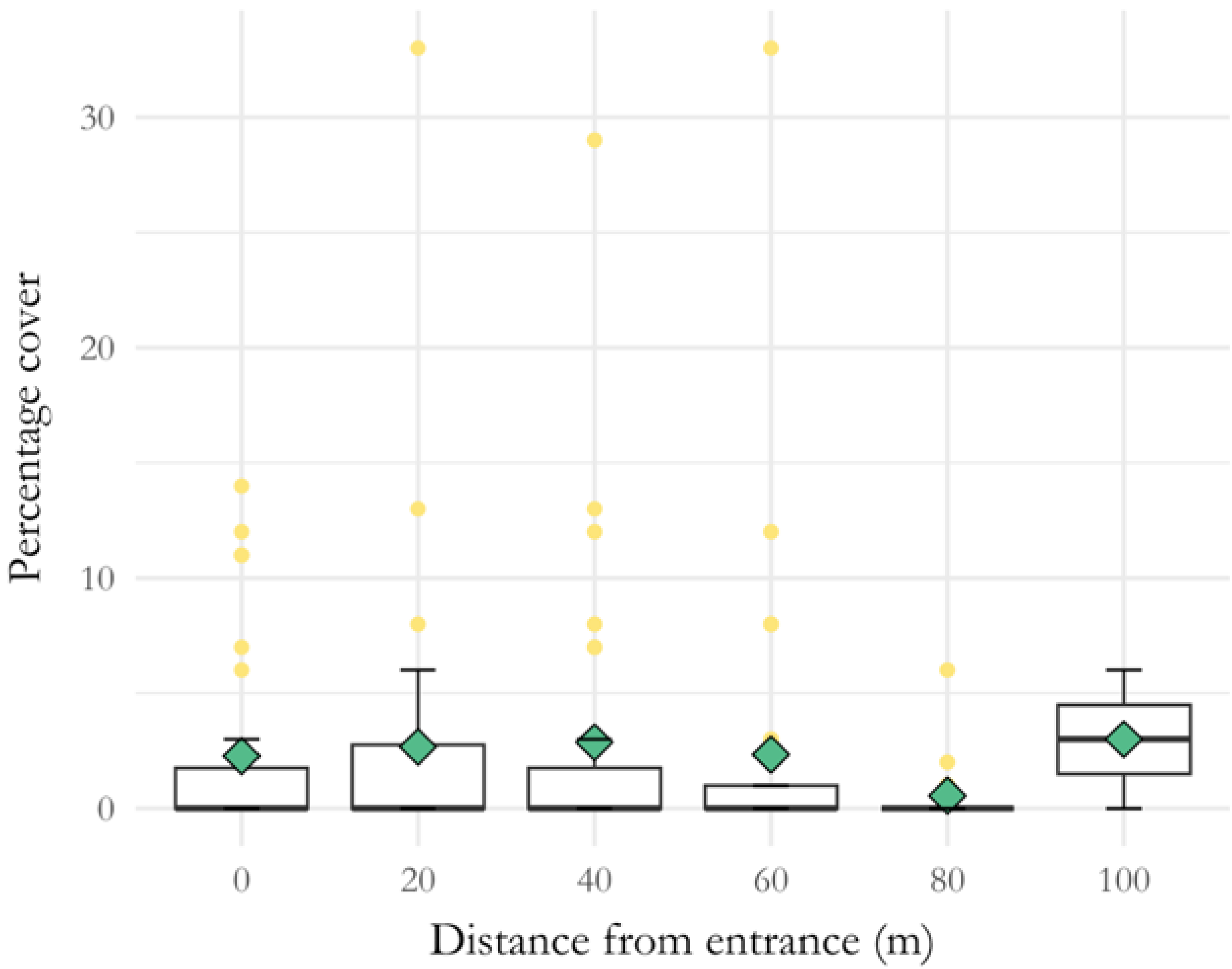

Figure 16 Boxplot of distance from entrance and percentage cover. Means in green, outliers in yellow.

RQ4 - Relationship between *proximity to mews edge* and *percentage cover*

The null hypothesis was accepted, so we conclude that vegetation cover does not significantly differ between the edge and centre of mews ($U = 318$, $n_1 = 40$, $n_2 = 20$, $P = >0.05$). A Kruskal Wallis Test similarly found no significant difference in medians between the three separate transects ($H = 2.20$, $DF = 2$, $P = >0.05$). The similarity of each transect can be seen in table 4 and figure 17. Mann-Whitney U tests for all subdivisions of the data (e.g. cobbled only, throughfare only etc.) found no significant difference between edge and centre.

Table 4 Heat-map table showing proximity to edge and percentage cover. Left table orders mews by surface type, right table orders by road type. The darker the shade of green, the greater the cover.

		Mean Percentage Cover by Surface Type		
		Left	**Centre**	**Right**
Cobbled Mews	C1	6.3	4.5	4.0
	C2	5.1	2.1	13.8
	C3	6.0	0.0	8.8
	C4	21.3	14.5	12.3
	C5	2.3	6.0	1.2
	C6	2.3	25.5	4.3
	C7	5.0	8.3	8.5
	C8	2.5	3.0	1.3
	C9	0.0	0.0	0.6
	C10	3.8	0.8	1.8
	Mean	**5.5**	**6.5**	**5.6**
Tarmacked Mews	T1	0.0	0.0	0.4
	T2	5.3	0.0	0.6
	T3	0.2	0.2	0.0
	T4	0.0	0.0	0.2
	T5	0.4	0.0	0.0
	T6	0.4	0.0	4.4
	T7	0.0	0.0	0.5
	T8	0.0	0.0	0.8
	T9	0.0	0.2	0.0
	T10	0.0	0.0	0.4
	Mean	**0.7**	**0.0**	**0.8**

		Mean Percentage Cover by Road Openness		
		Left	**Centre**	**Right**
Through Roads	1C	6.3	4.5	4.0
	1T	0.0	0.0	0.4
	2C	5.1	2.1	13.8
	2T	5.3	0.0	0.6
	3C	6.0	0.0	8.8
	3T	0.2	0.2	0.0
	4C	21.3	14.5	12.3
	4T	0.0	0.0	0.2
	5C	2.3	6.0	1.2
	5T	0.4	0.0	0.0
	Mean	**4.7**	**2.7**	**4.1**
Cul de Sacs	6C	2.3	25.5	4.3
	6T	0.4	0.0	4.4
	7C	5.0	8.3	8.5
	7T	0.0	0.0	0.5
	8C	2.5	3.0	1.3
	8T	0.0	0.0	0.8
	9C	0.0	0.0	0.6
	9T	0.0	0.2	0.0
	10C	3.8	0.8	1.8
	10T	0.0	0.0	0.4
	Mean	**3.1**	**3.2**	**3.2**

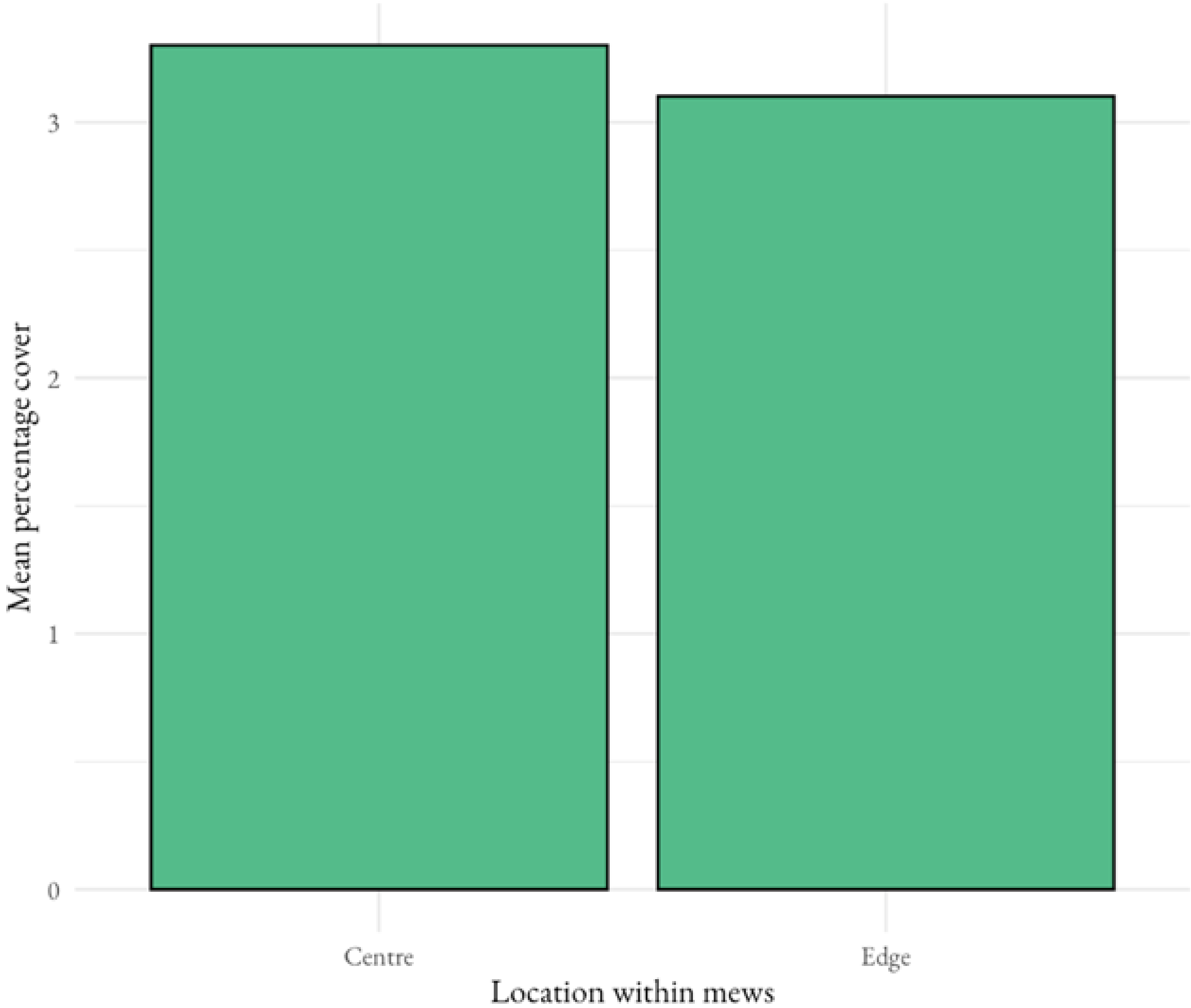

Figure 17 Bar chart of proximity to edge and percentage cover. 95% CIs not shown due to non-normal distribution; whilst range or interquartile range can be used as alternatives, a box plot shows this better and here the emphasis is on similar means..

Additional analysis

Further analysis is detailed in appendix 2.

Discussion

Interpretation

Clear evidence is provided that cobbled mews have significantly more abundant and biodiverse plant life than tarmacked mews. In the most extreme cases, the amount of vegetation between cobbles is remarkable and strongly supports an ecological benefit of retaining cobbled surfaces.

Variables other than surface type do not appear to influence vegetation cover, perhaps because the species that thrive in mews are sufficiently tolerant to trampling and pollution to survive anywhere in the mews.

Equally, the study provides evidence of the general impact of covering soil by any means. Even the most abundant mews had large patches with no vegetation and the near-total absence of life in tarmacked mews was striking.

Comparison to other research

Cover is higher than McKinney's (McKinney, 2010, p.290) "zero productivity" and similar to Melander's (2009) 2% but Gilbert (1991) claims 32% vegetation cover in built-up Berlin in 1979, which is significantly higher than even the cobbled mews (5.3%). The total number of plants found is lower than in some similar studies (Gilbert, 1991; Kent, Stevens and Zhang, 1999; Krüsi and Trachsel, 2012; Uchida *et al.*, 2014) but here we analysed a much smaller total area of 85.5m^2. Gilbert (1991) reports 380 species of vascular plant *per km^2* but it is hard to compare like-for-like because we cannot assume data scale up linearly. Doing so would predict more species would be found in London per km^2 than the total number of vascular plant species on earth, so we must assume it levels off at some undetermined point. Similar studies tend not to report SDI values.

Several species were also recorded by Krüsi and Trachsel (2012) and Fagot *et al* (2011), who found *Taraxacum officinale*, *Plantago major* and *Erigeron canadensis* amongst their most common species, and also found Bryophyta to cover the most area. No similar studies report *Draba verna* or *Veronica arvensis*, suggesting their dominance may be unique to London mews. Benvenuti (2004) also found a mix of annuals and perennials. That 25% of species were non-native strengthens McKinney's (2006) claim that urban plant communities are becoming more globally homogenous. Gardens are a significant source of alien plant species in urban environments (Kowarik and Fischer, 2020) - and it is likely that some non-natives were intentionally introduced by mews residents. Roads themselves can be dispersal corridors for invasives (Meunier and Lavoie, 2012).

That surface type is a key causal factor in the difference between mews is in line with the complete literature review above. A more complete multivariate analysis could identify other explanatory factors - Fagot *et al* (2011), for example, analysed usage intensity, light intensity and joint width together and Kent *et al* (1999) accounted for altitude, substrata and distance from water and the city centre.

The industrialised northern hemisphere is over-represented on the literature on urban flora (McDonnell, Hahs and Breuste, 2009) and this study does not address that.

Management Recommendations

The results support a three-pronged management strategy. First, 'do no harm', avoiding converting more mews to tarmac where possible, perhaps by raising awareness of the ecological value of cobbled surfaces. Second, biodiversity within mews should be maximised – and further research may be needed to identify the most effective interventions. Krüsi and Trachsel (2012, p.69) suggest "(1) reducing care and maintenance, (2) not sealing paving joints and (3) using grass pavers in parking lots or tram routes."

Finally, broader efforts to are required to mitigate negative impacts of urbanisation on the environment, for example Johnston and Newton (2004, p.63) suggest using paving that incorporates vegetation or even vegetation instead of “hard landscaping”. What is true in mews is likely true for other broken surfaces (Figure 18) and so the findings here should have broader implications.

Figure 18 Spontaneous roadway vegetation in Chitose, Japan.

Evaluation

Systematic sampling misses some variation within the mews. Many mews have plant abundance clustered in gutters that run down each side, but these were all missed due to the sampling strategy. Although the same for all mews, this systematic error likely reduces the perceived size of the differences, because tarmacked mews had almost no cover but cobbled mews likely have more cover than was recorded.

Similarly, another systematic error is introduced by the method of determining percentage cover; the cracks between cobbles rarely cover more than half a quadrat square, leading to an under-estimate of cover. Uchida *et al* (2014) define the size of a crack in the road as under 20mm - by this definition any plants growing in a crack could never cover more than 50% of a 50x50mm square. And since “floristic composition… is up to a point dependent on the width of the spaces between the stones or bricks” (Segal, 1969, p.239) it may be that biodiversity is also underestimated. Using smaller squares could overcome this but would take longer.

Further inaccuracies arise from not recording grasses when determining SDI. This decision was made because of the difficulty distinguishing species, but most similar studies record (only) *Poa annua* and grasses could perhaps have been identified at least to family level if not species. *Sagina procumbens* is a common pavement plant not recorded in this study – its seedings are highly alike *Poa annua* and it was likely missed.

The data for (cobbled) Great Cumberland Mews were unusually low, revealing a limitation. No control for age of cobbles or for sealant type was included in the experimental design and although many mews have original granite setts, a number of cobbled mews such as Great Cumberland have newer or different materials (Brown, 2021b) this nuance suggests areas for further research.

Further study

The study invites additional questions for further research. For example, since mews are small, unconnected spaces are there is debate about whether such small spaces may create ecological traps, drawing animals away from higher quality habitats to their peril (Lepczyk *et al.*, 2017).

It was observed that plants in the mews generally remained quite small during the whole season relative to individuals in less disturbed locations. Further research into the morphological impacts of mews life could provide further insight into the ecology of these streets.

Humans are an integrated component of urban ecosystems and drive the emergent properties of the system; in that context it is vital to manage cities with a whole-system approach, integrating ecological and sociological considerations (Alberti, 2008). As Grimm et al (2000, p.573) note, "Without understanding interactions and feedbacks between human and ecological systems, our view of ecosystem dynamics both at local and global scales will be limited—as will be our ability to apply these insights to public policy and land management." To do this is somewhat beyond the scope of the current study but must be borne in mind. There is often resistance to 'messy' spontaneous vegetation and many studies of it focus on removing "weeds" (e.g. Benvenuti, 2004; Rask and Kristoffersen, 2007; Melander *et al.*, 2009; Fagot *et al.*, 2011; De Cauwer *et al.*, 2014) but informal conversations with the residents of mews during this study suggested that in these locations plants are actively welcomed and there is scope for sociological research about the reasons for this openness to spontaneous vegetation.

Conclusion

In line with previous studies, it was found that the surface type is an important driver of urban plant biodiversity. When mews have a cobbled surface, there is more opportunity for plants to thrive and this results in higher cover and biodiversity. The location within the mews appears to not impact biodiversity. The study adds further evidence that urbanisation is associated with reduced species richness and increased prominence of non-native species. High-quality data to support these conclusions was successfully collected by young citizen scientists.

Biodiversity of spontaneous vegetation and the community that depends upon it can be maintained by preserving cobbled mews.

References

Aceves-Bueno, E., Adeleye, A.S., Feraud, M., Huang, Y., Tao, M., Yang, Y. and Anderson, S.E. (2017) 'The Accuracy of Citizen Science Data: A Quantitative Review', *The Bulletin of the Ecological Society of America*, 98(4), pp. 278–290. Available at: https://doi.org/10.1002/bes2.1336.

Alberti, M. (2008) *Advances in urban ecology: integrating humans and ecological processes in urban ecosystems*. New York, NY: Springer.

Aronson, M.F., Lepczyk, C.A., Evans, K.L., Goddard, M.A., Lerman, S.B., MacIvor, J.S., Nilon, C.H. and Vargo, T. (2017) 'Biodiversity in the city: key challenges for urban green space management', *Frontiers in Ecology and the Environment*, 15(4), pp. 189–196. Available at: https://doi.org/10.1002/fee.1480.

Baskin, J.M. and Baskin, C.C. (1983) 'Germination Ecology of Veronica Arvensis', *Journal of Ecology*, 71(1), pp. 57–68. Available at: https://doi.org/10.2307/2259963.

Benvenuti, S. (2004) 'Weed dynamics in the Mediterranean urban ecosystem: ecology, biodiversity and management', *Weed Research*, 44(5), pp. 341–354. Available at: https://doi.org/10.1111/j.1365-3180.2004.00410.x.

Berland, A. and Lange, D.A. (2017) 'Google Street View shows promise for virtual street tree surveys', *Urban Forestry & Urban Greening*, 21, pp. 11–15. Available at: https://doi.org/10.1016/j.ufug.2016.11.006.

Bonney, R., Phillips, T., Ballard, H. and Enck, J. (2015) 'Can citizen science enhance public understanding of science?', *Public understanding of science*, 25(1). Available at: https://doi.org/10.1177/0963662515607406.

Bonthoux, S., Voisin, L., Bouché-Pillon, S. and Chollet, S. (2019) 'More than weeds: Spontaneous vegetation in streets as a neglected element of urban biodiversity', *Landscape and Urban Planning*, 185, pp. 163–172. Available at: https://doi.org/10.1016/j.landurbplan.2019.02.009.

Boutin, C. and Harper, J.L. (1991) 'A Comparative Study of the Population Dynamics of Five Species of Veronica in Natural Habitats', *Journal of Ecology*, 79(1), pp. 199–221. Available at: https://doi.org/10.2307/2260793.

Bridle, K., Fitzgerald, M., Green, D., Smith, J., McQuillan, P. and Lefroy, T. (2009) 'Relationships between site characteristics, farming system and biodiversity on Australian mixed farms', *Animal Production Science*, 49(10), pp. 869–882.

Brown, E.D. and Williams, B.K. (2019) 'The potential for citizen science to produce reliable and useful information in ecology', *Conservation Biology*, 33(3), pp. 561–569. Available at: https://doi.org/10.1111/cobi.13223.

Brown, M. (2021b) *A Typical London Mews*, *Everchanging Mews*. Available at: https://everchangingmews.com/about/glossary/ (Accessed: 28 July 2024).

Brown, M.J. (2021a) *Mews Facts*, *Everchanging Mews*. Available at: https://everchangingmews.com/about/faq/ (Accessed: 27 June 2024).

Castagneyrol, B., Valdés-Correcher, E., Bourdin, A., Barbaro, L., Bouriaud, O., Branco, M., Centenaro, G., Csóka, G., Duduman, M.-L., Dulaurent, A.-M., Eötvös, C.B., Faticov, M., Ferrante, M., Fürjes-Mikó, Á., Galmán, A., Gossner, M.M., Harvey, D., Howe, A.G., Kaennel-Dobbertin, M., Koricheva, J., Löveï, G.L., Lupaştean, D., Milanović, S., Mrazova, A., Opgennoorth, L., Pitkänen, J.-M., Popović, M., Roslin, T.V., Scherer-Lorenzen, M., Sam, K., Tahadlová, M., Thomas, R. and Tack, A.J.M. (2020) 'Can School Children Support Ecological Research? Lessons from the Oak Bodyguard Citizen Science Project', *Citizen Science: Theory and Practice*, 5(1). Available at: https://doi.org/10.5334/cstp.267.

Chithra, S., Nair, M.H., Amarnath, A. and Anjana, N. (2015) 'Impacts of impervious surfaces on the environment', *International Journal of Engineering Science Invention*, 4(5), pp. 27–31.

City of Westminster (2015) 'Greener City Action Plan 2015-2025'. Available at: https://www.westminster.gov.uk/sites/default/files/media/documents/greener_city_action_plan_main%20%282%29.pdf (Accessed: 19 July 2024).

Collaborative Forest Restoration Program (2003) *Multiparty Monitoring and Assessment Guidelines for Community Based Forest Restoration in Southwestern Ponderosa Pine Forests*. U.S. Department of Agriculture, Forest Service, Southwestern Region.

Curtis, J.W., Curtis, A., Mapes, J., Szell, A.B. and Cinderich, A. (2013) 'Using google street view for systematic observation of the built environment: analysis of spatio-temporal instability of imagery dates', *International Journal of Health Geographics*, 12(1), p. 53. Available at: https://doi.org/10.1186/1476-072X-12-53.

Darlington, R.W. (2020) *Wild Flower Identification Guide*, *Wildflower Finder*. Available at: https://wildflowerfinder.org.uk/ (Accessed: 18 June 2024).

Davis, C.R., Belote, T., Williamson, M., Larson, A. and Esch, B. (2016) 'A rapid forest assessment method for multiparty monitoring across landscapes', 114, pp. 125–133. Available at: https://doi.org/10.5849/jof.14-118.

De Cauwer, B., Fagot, M., Beeldens, A., Boonen, E., Bulcke, R. and Reheul, D. (2014) 'Reduced weed growth with different paving constructions', *Weed Research*, 54(2), pp. 151–161. Available at: https://doi.org/10.1111/wre.12059.

Dickinson, J.L., Zuckerberg, B. and Bonter, D.N. (2010) 'Citizen science as an ecological research tool: challenges and benefits', *Annual review of ecology, evolution, and systematics*, 41(1), pp. 149–172. Available at: https://doi.org/10.1146/annurev-ecolsys-102209-144636.

Dřevojan, P., Čeplová, N., Štěpánková, P. and Axmanová, I. (2023) *Life Form*, *FloraVeg.eu*. Available at: https://floraveg.eu/ (Accessed: 18 June 2024).

Fagot, M., Cauwer, B.D., Beeldens, A., Boonen, E., Bulcke, R. and Reheul, D. (2011) 'Weed flora in paved areas in relation to environment, pavement characteristics and weed control', *Weed Research*, 51, pp. 650–660.

Fowler, S. (2019) 'Looking for Your Own Front Door in London? Consider a Mews.', *The New York Times*, 14 May. Available at: https://www.nytimes.com/2019/05/14/realestate/london-mews-carriage-houses.html (Accessed: 27 June 2024).

GiGL (2024) *Key London Figures - GIGL, Greenspace Information for Greater London CIC*. Available at: https://www.gigl.org.uk/our-data-holdings/keyfigures/ (Accessed: 19 July 2024).

Gilbert, O.L. (1991) *The Ecology of Urban Habitats*. Dordrecht, Netherlands: Springer Netherlands. Available at: http://ebookcentral.proquest.com/lib/oxford/detail.action?docID=3566470 (Accessed: 19 July 2024).

Goddard, M.A., Dougill, A.J. and Benton, T.G. (2010) 'Scaling up from gardens: biodiversity conservation in urban environments', *Trends in ecology & evolution*, 25(2), pp. 90–98. Available at: https://doi.org/10.1016/j.tree.2009.07.016.

Godefroid, S. and Koedam, N. (2004) 'The impact of forest paths upon adjacent vegetation: effects of the path surfacing material on the species composition and soil compaction', *Biological Conservation*, 119(3), pp. 405–419. Available at: https://doi.org/10.1016/j.biocon.2004.01.003.

Godefroid, S. and Koedam, N. (2007) 'Urban plant species patterns are highly driven by density and function of built-up areas', *Landscape Ecology*, 22(8), pp. 1227–1239. Available at: https://doi.org/10.1007/s10980-007-9102-x.

Godefroid, S., Monbaliu, D. and Koedam, N. (2007) 'The role of soil and microclimatic variables in the distribution patterns of urban wasteland flora in Brussels, Belgium', *Landscape and Urban Planning*, 80(1), pp. 45–55. Available at: https://doi.org/10.1016/j.landurbplan.2006.06.001.

Google Maps (2024) 'Mews Maps'. Available at: https://www.google.com/maps/d/edit?mid=1Y_md6EU72Xa-kKYSLSIiknV9Kb70Z48&usp=sharing.

Grimm, N.B., Grove, J.G., Pickett, S.T.A. and Redman, C.L. (2000) 'Integrated Approaches to Long-Term Studies of Urban Ecological Systems: Urban ecological systems present multiple challenges to ecologists—pervasive human impact and extreme heterogeneity of cities, and the need to integrate social and ecological approaches, concepts, and theory', *BioScience*, 50(7), pp. 571–584. Available at: https://doi.org/10.1641/0006-3568(2000)050[0571:IATLTO]2.0.CO;2.

Harris, G.R. and Lovell, P.H. (1980) 'Adventitious Root Formation in Veronica Spp.', *Annals of Botany*, 45(4), pp. 459–468. Available at: https://doi.org/10.1093/oxfordjournals.aob.a085844.

Head, M.L., Holman, L., Lanfear, R., Kahn, A.T. and Jennions, M.D. (2015) 'The Extent and Consequences of P-Hacking in Science', *PLOS Biology*, 13(3), p. e1002106. Available at: https://doi.org/10.1371/journal.pbio.1002106.

Hill, M.O. (ed.) (1999) *Ellenberg's indicator values for British plants*. Abbots Ripton, Huntingdon: Center for Ecology and Hydrology, Natural Environment research Council (Ecofact, v. 2, technical annex).

HM Government (2018) *A green future: our 25 year plan to improve the environment*. London: Defra. Available at: https://assets.publishing.service.gov.uk/media/65fd713d65ca2f00117da89e/CD1.H_HM_Government_A_Green_Future_Our_25_Year_Plan_to_Improve_the_Environment.pdf (Accessed: 19 July 2024).

Hou, Y., Li, J., Li, G. and Qi, W. (2023) 'Negative effects of urbanization on plants: A global meta-analysis', *Ecology and Evolution*, 13(4), p. e9894. Available at: https://doi.org/10.1002/ece3.9894.

Hugo.arg (2007) *StellariaMedia001.JPG (2816×2112)*, *Wikimedia Commons*. Available at: https://upload.wikimedia.org/wikipedia/commons/7/7d/StellariaMedia001.JPG (Accessed: 19 July 2024).

Johnston, J. and Newton, J. (2004) *Building green: a guide to using plants on roofs, walls and pavements*. London: Greater London Authority. Available at: https://building-green.org.uk/wp-content/uploads/2017/07/johnstone-and-newton-building-green.pdf#page=83.11.

Jost, L. (2006) 'Entropy and Diversity', *Oikos*, 113(2), pp. 363–375. Available at: https://doi.org/10.1111/j.2006.0030-1299.14714.x.

Kent, M. (2012) *Vegetation description and data analysis: a practical approach*. second edition. Oxford Chichester Hoboken: Wiley-Blackwell.

Kent, M., Stevens, R. and Zhang, L. (1999) 'Urban plant ecology patterns and processes: a case study of the flora of the City of Plymouth, Devon, UK', *Journal of Biogeography*, 26(6), pp. 1281–1298. Available at: https://doi.org/10.1046/j.1365-2699.1999.00350.x.

Kowarik, I. and Fischer, L. (2020) 'Alien Plants in Cities', in I. Douglas (ed.) *Handbook of urban ecology*. London ; New York: Routledge/Taylor & Francis Group, pp. 472–482. Available at: https://ebookcentral.proquest.com/lib/oxford/reader.action?docID=6374123.

Krüsi, B.O. and Trachsel, T. (2012) 'Erstaunliche Vielfalt in einem unscheinbaren Lebensraum: die Pflasterfugen-Flora der Stadt Zürich', *Vierteljahrsschrift der Naturforschenden Gesellschaft in Zürich*, 157(3/4), pp. 59–72.

Lepczyk, C.A., Aronson, M.F.J., Evans, K.L., Goddard, M.A., Lerman, S.B. and MacIvor, J.S. (2017) 'Biodiversity in the City: Fundamental Questions for Understanding the Ecology of Urban Green Spaces for Biodiversity Conservation', *BioScience*, 67(9), pp. 799–807. Available at: https://doi.org/10.1093/biosci/bix079.

Liu, Z., He, C., Zhou, Y. and Wu, J. (2014) 'How much of the world's land has been urbanized, really? A hierarchical framework for avoiding confusion', *Landscape Ecology*, 29(5), pp. 763–771. Available at: https://doi.org/10.1007/s10980-014-0034-y.

Lockton, A. (2017) *Draba_verna_whole.jpg (1549×1467)*, *Wikimedia Commons*. Available at: https://upload.wikimedia.org/wikipedia/commons/b/bb/Draba_verna_whole.jpg (Accessed: 19 July 2024).

Loiselle, S., Thornhill, I. and Bailey, N. (2016) 'Citizen science: advantages of shallow versus deep participation', *Front. Environ. Sci.*, 4, pp. 1–2. Available at: https://doi.org/10.3389/conf.FENVS.2016.01.00001.

Lundholm, J. (2011) 'Vegetation of Urban Hard Surfaces', in J. Niemelä, J.H. Breuste, T. Elmqvist, G. Guntenspergen, P. James, and N.E. McIntyre (eds) *Urban Ecology: Patterns, Processes, and Applications*. Oxford University Press, pp. 93–102. Available at: https://doi.org/10.1093/acprof:oso/9780199563562.003.0012.

McDonald, J.H. (2014) *Handbook of Biological Statistics*. Baltimore: Sparky House Publishing. Available at: https://www.biostathandbook.com/pairedttest.html (Accessed: 16 July 2024).

McDonnell, M.J., Hahs, A.K. and Breuste, J.H. (2009) *Ecology of cities and towns: a comparative approach*. Cambridge University Press.

McKinney, M. (2006) 'Urbanization as a major cause of biotic homogenization.', *Biological Conservation*, 127, pp. 247–260. Available at: https://doi.org/10.1016/j.biocon.2005.09.005.

McKinney, M.L. (2010) 'Urban futures', in K.J. Gaston (ed.) *Urban Ecology*. Cambridge: Cambridge University Press (Ecological Reviews), pp. 287–308. Available at: https://doi.org/10.1017/CBO9780511778483.013.

McManus, R. (2004) 'Windows on a hidden world: Urban and social evolution as seen from the mews', *Irish Geography*, 37(1), pp. 37–59. Available at: https://doi.org/10.1080/00750770409555828.

Melander, B., Holst, N., Grundy, A.C., Kempenaar, C., Riemens, M.M., Verschwele, A. and Hansson, D. (2009) 'Weed occurrence on pavements in five North European towns', *Weed Research*, 49(5), pp. 516–525. Available at: https://doi.org/10.1111/j.1365-3180.2009.00713.x.

Meunier, G. and Lavoie, C. (2012) 'Roads as Corridors for Invasive Plant Species: New Evidence from Smooth Bedstraw (Galium mollugo)', *Invasive Plant Science and Management*, 5(1), pp. 92–100. Available at: https://doi.org/10.1614/IPSM-D-11-00049.1.

Morris, E.K., Caruso, T., Buscot, F., Fischer, M., Hancock, C., Maier, T.S., Meiners, T., Müller, C., Obermaier, E., Prati, D., Socher, S.A., Sonnemann, I., Wäschke, N., Wubet, T., Wurst, S. and Rillig, M.C. (2014) 'Choosing and using diversity indices: insights for ecological applications from the German Biodiversity Exploratories', *Ecology and Evolution*, 4(18), pp. 3514–3524. Available at: https://doi.org/10.1002/ece3.1155.

NBN Trust (2024) *Species search | NBN Atlas*, *The National Biodiversity Network (NBN) Atlas*. Available at: https://species.nbnatlas.org/ (Accessed: 18 June 2024).

Papworth, S. k., Rist, J., Coad, L. and Milner-Gulland, E. j. (2009) 'Evidence for shifting baseline syndrome in conservation', *Conservation Letters*, 2(2), pp. 93–100. Available at: https://doi.org/10.1111/j.1755-263X.2009.00049.x.

Pauly, D. (1995) 'Anecdotes and the shifting baseline syndrome of fisheries', *Trends in ecology and evolution*, 10(10), p. 430.

Phillips, T.B., Ballard, H.L., Lewenstein, B.V. and Bonney, R. (2019) 'Engagement in science through citizen science: Moving beyond data collection', *Science Education*, 103(3), pp. 665–690. Available at: https://doi.org/10.1002/sce.21501.

Poppenwimer, T., Mayrose, I. and DeMalach, N. (2023) 'Revising the global biogeography of annual and perennial plants', *Nature*, 624(7990), pp. 109–114. Available at: https://doi.org/10.1038/s41586-023-06644-x.

POWO (2024) *Plants of the World Online. Facilitated by the Royal Botanic Gardens, Kew*, *Plants of the World Online*. Available at: https://powo.science.kew.org/ (Accessed: 18 July 2024).

Rask, A.M. and Kristoffersen, P. (2007) 'A review of non-chemical weed control on hard surfaces', *Weed Research*, 47(5), pp. 370–380. Available at: https://doi.org/10.1111/j.1365-3180.2007.00579.x.

Robinson, S.L. and Lundholm, J.T. (2012) 'Ecosystem services provided by urban spontaneous vegetation', *Urban Ecosystems*, 15(3), pp. 545–557. Available at: https://doi.org/10.1007/s11252-012-0225-8.

Roswell, M., Dushoff, J. and Winfree, R. (2021) 'A conceptual guide to measuring species diversity', *Oikos*, 130(3), pp. 321–338. Available at: https://doi.org/10.1111/oik.07202.

Segal, S. (1969) 'Vegetation of streets: a comparison', in S. Segal (ed.) *Ecological Notes on Wall Vegetation*. Dordrecht: Springer Netherlands, pp. 239–265. Available at: https://doi.org/10.1007/978-94-017-6232-8_8.

Siegmund, W. (2012) *Veronica_arvensis_7998.JPG (2848×4272)*, *Wikimedia Commons*. Available at: https://upload.wikimedia.org/wikipedia/commons/0/0c/Veronica_arvensis_7998.JPG (Accessed: 19 July 2024).

Soanes, K., Sievers, M., Chee, Y.E., Williams, N.S.G., Bhardwaj, M., Marshall, A.J. and Parris, K.M. (2019) 'Correcting common misconceptions to inspire conservation action in urban environments', *Conservation Biology*, 33(2), pp. 300–306. Available at: https://doi.org/10.1111/cobi.13193.

Soontorn, T. (2020) *'Consulting' Icon - Free PNG & SVG 4095106*, *The Noun Project*. Available at: https://thenounproject.com/icon/consulting-4095106/ (Accessed: 23 July 2024).

Stroh, P.A., Humphrey, T.A., Burkmar, R.J., Pescott, O.L., Roy, D.B. and Walker, K.J. (2020) *BSBI Online Plant Atlas 2020*, *BSBI Online Plant Atlas 2020*. Available at: https://plantatlas2020.org/atlas/2cd4p9h.f8p (Accessed: 18 June 2024).

Sullivan, G.M. and Feinn, R. (2012) 'Using Effect Size—or Why the P Value Is Not Enough', *Journal of Graduate Medical Education*, 4(3), pp. 279–282. Available at: https://doi.org/10.4300/JGME-D-12-00156.1.

Taylor, J. (2012) *Broadleaf_Plantain_(Plantago_major)_growing_in_crack_in_sidewalk.JPG (3240×4320)*, *Wikimedia Commons*. Available at: https://upload.wikimedia.org/wikipedia/commons/b/b3/Broadleaf_Plantain_%28Plantago_major%29_growing_in_crack_in_sidewalk.JPG (Accessed: 19 July 2024).

Uchida, T., Xue, J.H., Hayasaka, D., Arase, T., Haller, W.T. and Gettys, L. (2014) 'The relation between road crack vegetation and plant biodiversity in urban landscape', *International Journal of GEOMATE*, 6, pp. 885–891. Available at: https://doi.org/10.21660/2014.12.3177.

Vaughn, K.C., Bowling, A.J. and Ruel, K.J. (2011) 'The mechanism for explosive seed dispersal in Cardamine hirsuta (Brassicaceae)', *American Journal of Botany*, 98(8), pp. 1276–1285. Available at: https://doi.org/10.3732/ajb.1000374.

Wheater, C.P. (2020) 'The invasion of walls, pavements and building surfaces by organisms', in I. Douglas (ed.) *Handbook of urban ecology*. London ; New York: Routledge/Taylor & Francis

Group, pp. 265–277. Available at: https://ebookcentral.proquest.com/lib/oxford/reader.action?docID=6374123.

Wilby, R.L. and Perry, G.L. (2006) 'Climate change, biodiversity and the urban environment: a critical review based on London, UK', *Progress in physical geography*, 30(1), pp. 73–98.

Appendices

Appendix 1: Citizen Science

The Dimensions of Engagement Framework (Phillips *et al.*, 2019) was utilised to apply best practice in engaging citizen scientist learnersFigure 19. Students collected data (behavioural), worked in collaborative pairs (social), learned research skills and natural history facts (learning) and reflected how the experience affected them (affective).

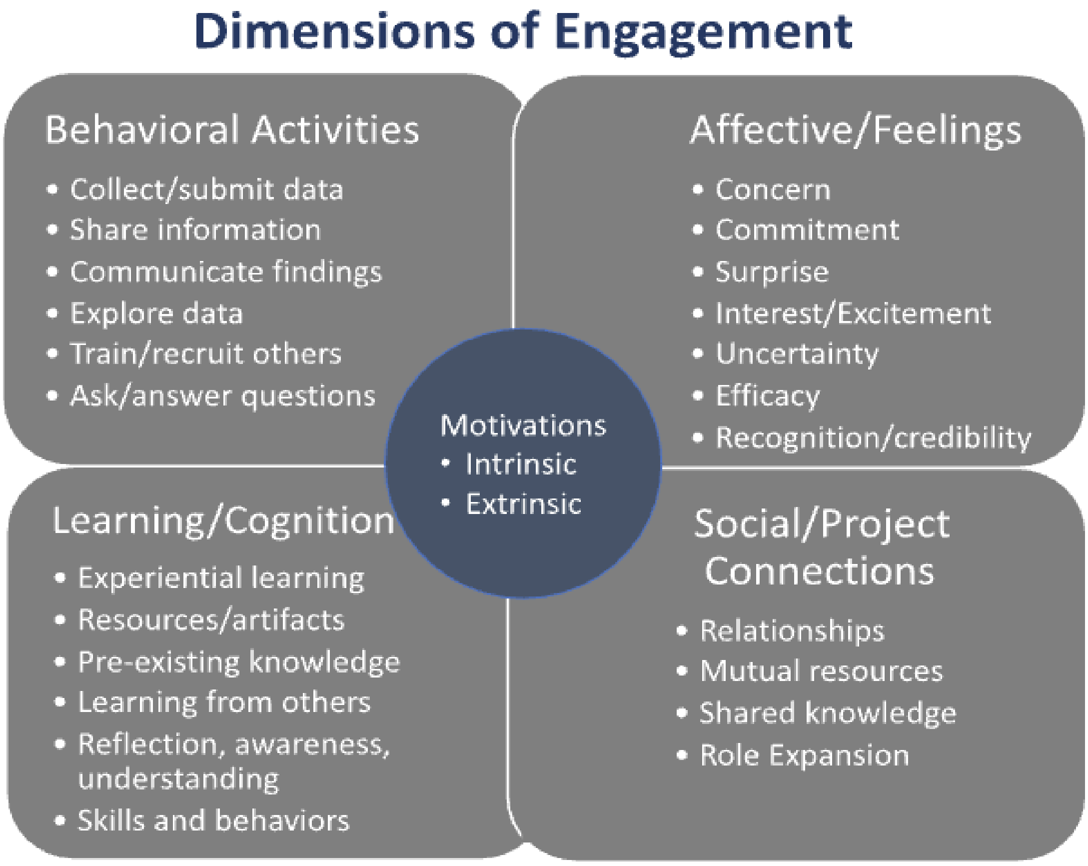

Figure 19 Dimensions of Engagement Framework (Phillips et al., 2019)

Following Davis *et al* (2016), they were given a simple identification guide (see appendix 3) and were extensively supervised in the field. One group was selected (based on alignment with curriculum goals) to co-construct methods and thus helped to shortlist sites, create the identification guide and conduct preliminary trials to refine the methodology. Timing constraints did not allow for students to take a identification proficiency test, which could have improved the reliability of measurements (Dickinson, Zuckerberg and Bonter, 2010).

Citizen scientists reported generally positive feelings about their participation in the study. One noted, for example, that "I liked that… we were actively doing real science" and another that "I felt fully immersed in the scientific process and that I had responsibility for the application of each of its aspects." Showing students older and newer mews surfaces also hopefully mitigates 'shifting baseline syndrome' (Pauly, 1995; Papworth *et al.*, 2009).

Appendix 2: Additional Tables, Statistics and Analysis

A Principal Component Analysis was not conducted because most of the relevant variables are categorical and conversion to binary values would lead to excessive differences in scaling. Non-Metric Multidimensional Scaling would be suitable but analysis in R indicated that there was insufficient variability in the data and too few usable samples (since many data points are effectively blank). Regardless, the existing analysis is considered sufficient to answer the research questions.

Time of year as a variable was analysed (Figure 20). The paired design was implemented because the pairs of mews have a shared time and (approximate) space and when plant biodiversity was informally observed before and after the study period (in March and June) it appeared considerably lower than during the study period. However, it is worth subjecting this question to statistical analysis. SDI deviates significantly from normal (Shapiro-Wilk test, $W = 0.691$, $n = 10$, $P = <0.05$) so a Spearman's rank correlation was used and found no significant correlation between day of year and reciprocal Simpson Diversity Index ($r = -0.643$, $DF = 10$, $P = >0.05$). This suggests that it may not be necessary to use a paired design (it is maintained as a precautionary measure). Finding statistical significance even with the paired design adds greater confidence to the findings. It was unexpected to find no relationship with date (Figure 20), but this is likely to be influenced by the choice of sampling period, which is within a narrow window of peak growth.

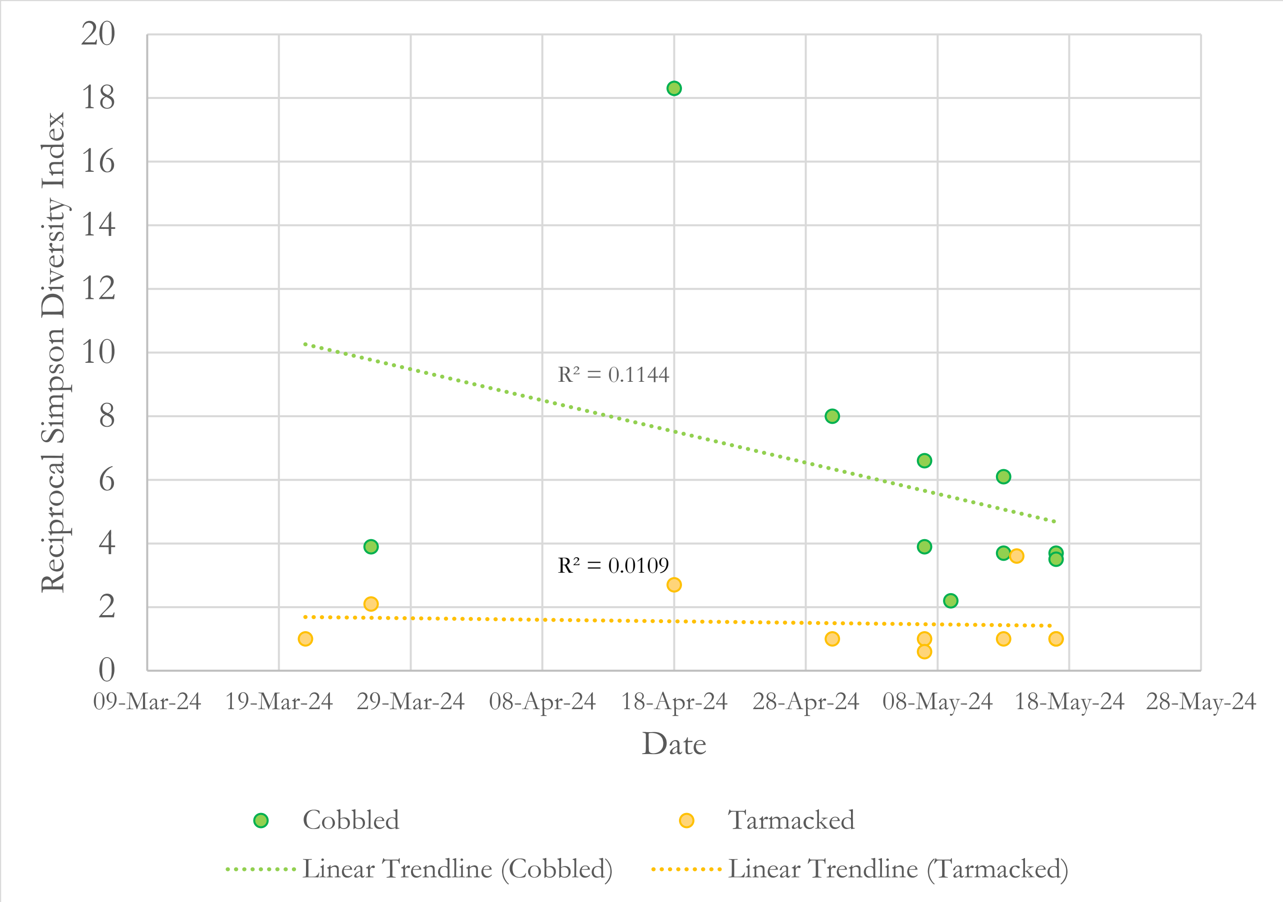

Figure 20 Relationship between date and SDI. Low R^2 values imply weak correlation.

In addition to cover and SDI, species richness for the two surface types (Table 5) also differs significantly ($t = -8.99$, $DF = 9$, $P = <0.01$).

Table 5 Surface type and species richness; richness is much higher in cobbled mews

Pair	Species Richness	
	Cobbled	**Tarmacked**
1	10	4
2	17	8
3	9	1
4	15	1
5	10	1
6	10	5
7	14	2
8	11	0
9	5	0
10	9	2
Mean	11	2

A total of 37 different species were found (Figure 12), with an abundance ranging from appearing in a single 5x5mm square to appearing in 270 (or 984 if mosses are included). 25% of the individual plants that were found were *Draba verna.* The three most common plants (Figure 11) represented more than 50% of the total individuals found (Figure 13) and 90% of all individuals came from just 14 species. Three species could not be identified either in the field or with extensive later investigation (see appendix 5), but the research questions depend only on knowing that they are distinct to the other species, and so they are recorded simply as species X, Y and Z (named in order of discovery).

Nine of the 37 plant species are non-native to the UK (NBN Trust, 2024; POWO, 2024). Exactly half of the plants are annual therophytes and half are perennial hemicryptophytes. Annuals represent only 6% of plants in general and are often found in arid conditions (Poppenwimer, Mayrose and DeMalach, 2023) and indeed many of the non-native species found in mews are from more arid regions (Table 6) and it seems that the bare rock of the mews may echo a habitat found in drier parts of the world - perhaps implying the mews have relative resilience to climate change.

Table 6 Place of origin for the non-native species; often arid regions

Species	Origin
Campanula portenschlagiana	Croatia
Soleirolia soleirolii	Italy
Erigeron canadensis	North America
Campanula poscharskyana	Serbia
Matricaria discoidea	North America
Erigeron karvinskianus	Mexico
Erigeron sumatrensis	Central/South America
Cotula australis	Southern Africa/Australia
Lobelia erinus	Africa

Overall, 66% of quadrats studied had absolutely zero cover - only 21% were like this in cobbled mews but 87% were in tarmacked mews.

Appendix 3: Data recording sheet

Size reduced to fit on a single page (document used in field was in pageless digital format)

London Mews Floristic Diversity Investigation: Data Sheet

Location	
Date	
Weather	

Left Edge Transect				
Distance along transect	Percentage cover all species	Species present (names and percentage frequency)	Recorded by (initials)	Photograph

Central Transect				
Distance along transect	Percentage cover all species	Species present (names and percentage frequency)	Recorded by (initials)	Photograph

Right Edge Transect				
Distance along transect	Percentage cover all species	Species present (names and percentage frequency)	Recorded by (initials)	Photograph

Notes, including images of unknown species

Pavement Plant ID Guide

5

Quad. Ref.	Notes

Appendix 4: Raw data

A digital copy of the raw data, which does not fit on A4, is viewable at tinyurl.com/mews-data

Appendix 5: Unidentified Species

Three species could not be reliably identified in the field or with extensive later research. The conclusions of the report do not depend on the ability to name species, but only to determine uniqueness (for calculation of SDI) and so these were simply recorded as X, Y and Z.

Species X Found in Hyde Park Gardens Mews, 18 April 2024	
Species Y Found in London Mews 28 March 2024	
Species Z Found in Bryanston Mews West 18 April 2024	Uploaded image file was corrupted but the plant was observed by the principal scientist in the field and determined to be unique.