Cardiff school of Bioscience

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Masters thesis

Evaluating the potential of road infrastructure greening and the relative importance of wildlife-vehicle collision data monitoring.

This thesis is submitted for the degree of Master of Philosophy from the Cardiff school of Bioscience.

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Thesis summary

The construction of linear transport infrastructure separates and fragments natural habitats, reducing carrying capacity and increasing mortality through wildlife-vehicle collisions. Wildlife mitigation structures like under and overpasses for roads, can help increase connectivity and reduce the number of animals crossing roads. However, these structures are typically difficult to design effectively, expensive to make, and time consuming to implement. An alternate solution to reduce the impact of roads on wildlife, could be to modify existing 'grey' infrastructure such as footbridges that are already in place. Grey infrastructure was identified throughout the UK using digital datasets and interpreted in GIS software. Once an extensive data set was complied, it was used to model connectivity changes of broadleaf and coniferous woodland networks on a 25km^2 scale using Condatis, a circuit theory application. Modelling showed an average increase in connectivity of 8 to 12% across networks after infrastructure was modelled to have been modified for wildlife or 'greened'. This theoretical modelling shows the potential of such methods but requires practical field data to confirm the simulated benefits.

Additionally, wildlife-vehicle collisions (WVCs) were investigated for their value to species monitoring. Using WVC data from UK citizen science scheme 'The Road Lab', spatial distributions for 33 species were compared to National Biodiversity Network (NBN) data. Spatial overlap of species distributions between NBN and WVCs was above 95%. Despite the smaller sample size of WVC data, it still contained 541 species presence datapoints (5km^2 grids) across the UK that were not represented by other NBN sources. WVC data also ranked highly for abundance of data points when compared to other NBN data providers. The high spatial overlap and relative abundance suggests WVCs are a suitable and complementary data source to standard species presence surveys, providing useful distribution insights that could be otherwise missed.

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Contents

Chapter 1: Introduction

1.1 Ecological importance of connectivity.

Anthropogenic disturbances and human developments are one of the largest contributors to global biodiversity loss and wildlife population declines (Ceballos and Ehrlich, 2002; Pimm *et al.*, 2006). By removing and degrading natural habitats, the carrying capacity for species within them becomes limited or actively lowered in relation to resource restriction and habitat lost (Benitez-Lopez *et al*., 2010; Crooks, 2002). However, the impact of human activity goes beyond the physical footprint of any constructions.

Habitat fragmentation is the act of separating one previously large area into smaller pieces, often with new barriers built between (Carvajal *et al.*, 2018). A common form of fragmentation occurs with the construction of linear transport infrastructure, such as roadways and rail lines (Carter *et al.,* 2022). While the active footprint may be small and the habitat lost minimal, these constructions can still have a significant impact on ecosystem functions through artificial separation and restricted movement (Fischer and Lindenmayer, 2007, Underhill and Angold, 2011).

Fragmenting a habitat increases the risk of separating vital resources, and lowers the quality and stability of the overall ecosystem (Preau *et al.*, 2022). Reductions in habitat quality and stability interfere with trophic interactions beyond a single species (Zarnetske *et al.*, 2017) and can interfere with populations beyond the region where fragmentation occurs (Carvajal *et al*., 2018). As such, the impact of fragmentation can be greater than many other forms of habitat loss, due to its indiscriminatory nature for species and range. Some estimations have shown biodiversity to be impacted up to five kilometres away from road constructions, indicating a 10km buffer may exist around major roadways where habitats are degraded for many species (Benitez-Lopez *et al.,* 2010). This effect was termed the road effect zone by Forman and Alexander (1998).

The splitting of habitats and construction of barriers such as roads has also been linked to the genetic deterioration of wildlife populations (Yumnam *et al.*, 2014). By limiting movement and dispersal routes, inbreeding pressures increase and the potential for immigration lowers (Joshi *et al.*, 2013). In combination, these greatly impact the health and survivability of a population, if it becomes separated due to a fragmented habitat (Suttidate *et al.*, 2021). In some cases, it can even lead to local population extinctions where a separated group becomes unviable (Fulgione *et al.,* 2009).

Understanding habitat connectivity is critical for conservation planning, as it allows habitat generation and land protections to be targeted in the most important locations. Beyond core habitat protections, ensuring unbroken links between patches through corridors can help mitigate fragmentation or the loss of overall habitat size (Rathore *et al.,* 2012). These movement networks are not only able to facilitate dispersal and geneflow, but also help reduce the impacts of climate change which is altering species distributions (Hamilton *et al.*, 2018, Dickson *et al.,* 2019). Maintaining habitat connectivity also protects ecosystem stability and ensures the functioning of critical mechanisms within them (Breckheimer *et al*., 2014; Harihar and Pandav, 2012). Therefore, the ability to identify and quantify habitat connectivity is an important tool for wildlife conservationists.

1.2 Review of common methods used in connectivity modelling.

Connectivity modelling is an emerging field which continues to evolve as new methods and software become available. Common methods include basic Euclidian distance measurements, least cost analysis, and circuit theory applications like Circuitscape and Condatis (Suttidate *at al.,* 2021; Travers *et al.,* 2021). The overall aim of these processes is to monitor and measure the extent to which a landscape facilitates the movements of individuals through it. Simplistic approaches like Euclidean distance measurements are able to provide an overview of habitat separation, but lack the analytical power to give a true representation of habitat connectivity (Adriaensen *et al*., 2003). Newer methods like least cost mapping and circuit theory offer a more robust option for analysing movement networks, by including environmental variables that impact modelled routes. These approaches are becoming more common in literature, with Circuitscape software being used frequently to assess and evaluate movement corridors (Wilcox *et al.*, 2023; McRae *et al.,* 2008; McRae *at al*., 2007). Novel approaches continue to be created, with each having advantages and limitations. Each of these techniques provide useful insights to measure habitat connectivity, but the type of analysis needs to match the data sources and intended outputs to be successful (Dertien and Baldwin, 2023; Lechner *at al.*, 2016).

Least cost paths became widely used in the 1990's (Knaapen *et al.* 1992) but are often limited by the assumption of informed decision making in the individuals being modelled (Mahmoodi *et al.,* 2023). While possible that any individual for a given species has knowledge of its home range, it is unlikely to have perfect knowledge of the surrounding habitats and routes beyond (Kumar *et al.,* 2022). This is amplified by least cost modelling which typically generates a single proposed route through an area, rather than multiple branching routes (Suttidate *at al.,* 2021). As such, least cost modelling of large movement networks is inherently limited in its accuracy (Dickson *et al.,* 2019).

Circuit theory modelling offers a possible solution, by adjusting the landscape to shape movement pathways with ecological data. As such, they can offer a more complex connectivity model where movement is impacted by habitat features and the method allows for multiple branching routes (Cushman *et al*., 2013; Naidoo *et al.*, 2018). However, many of the limitations of least cost modelling still occur within circuit theory, while others are introduced depending on methodology and study area (Poor *et al.*, 2012; Shirk *et al.*, 2015).

In many large-scale circuit theory models, movement outputs closely resemble species presence or habitat suitability models (Moilanen, 2011; Wade *et al.*, 2023). This similarity is due to lower resistance values inherently attracting the electrical current used in circuit theory models to those areas, with a pooling effect happening near current nodes that are typically placed on historical species observations (Grafius *et al.*, 2017; Kumar and Cushman, 2022; Milanesi *et al.*, 2017). By using observations as current nodes, it creates an inherent bias as species surveys may have only been completed in a small portion of the range and where a species is expected to occur. As such, it can lead to inaccurate predictions if surveys miss smaller populations or alternate habitats the species utilises for movement or dispersal (McClure *et al.* 2016; Carroll *et al.,* 2020). Movement of individuals is also largely dependent of the surrounding habitats, with movement strategies actively changing depending on the range, purpose and scale of movements (Zeller *et al.*, 2014). Pumas (*Puma concolor*) have been shown to utilise habitats beyond those considered suitable during dispersal, with some data indicating habitat choice is based on immediate perception when outside of their home range (Burdett *et al.*, 2010; Wilmers *et al.*, 2013; Zeller *et al.*, 2014). This suggests an individual will often choose a sub optimal habitat to move through immediately, rather than actively search for an alternate route or outright refuse to move through habitats which would be modelled with high resistance (Burdett *et al.*, 2010; Sweanor *et al*., 2010; Wilmers *et al*., 2013). To further complicate model making, some species exhibit different movement behaviours between sexes not just individuals (LaPoint *et al.,* 2013; Wilmers *et al.,* 2013; Kumar *et al*., 2022). Failing to take behavioural choices into account, overlooks major ecological considerations and can reduce the accuracy of movement predictions created in this way (Kumar and Cushman, 2022; Kumar *et al.*, 2022).

Many Circuitscape studies use habitat suitability as an inverse resistance layer, increasing the likelihood of survey location bias directly impacting connectivity models. Surveys are unlikely to occur near roads for many species, artificially inflating their resistance when using habitat suitability techniques like Maxent (Philips *et al*., 2006), which often includes distance to road as a key metric (Nelli *et al., 2022*; Zhang *et al.,* 2021, Linkie *et al.*, 2006). Any impact is further exacerbated by the nature of habitat suitability analysis being inherently different to the needs of most connectivity models (Hunter-Ayad and Hassall, 2020). Habitat suitability is a measure of the landscapes ability to support the needs of a population living within it, not just those moving through it. Therefore, suitability can be a poor predictor of species movement and habitat connectivity (Sartor *et al.,* 2022; Carroll *et al.,* 2020; Keeley *et al.,* 2017; Merrick and Koprowski, 2017). These issues can also lead to models being inaccurate to real world behaviours, by under or overestimating relative resistances of landscape features based on unsuitable data or data interpretations (Hanks and Hooten, 2013; Rudnick *et al.,* 2012; Wasserman *et al.,* 2010). Unless modelling habitat specialists, little can sometimes be determined through these types of analysis (Keeley *et al*.,2016; Shirk *et al.*, 2015; Brotons *et al,* 2004). This is especially true when modelling habitat generalists who may use much of the landscape, making resistance hard to rank accurately (Hanks and Hooten, 2013; Way *et al.,* 2004; Stevenson-Holt *et al.*, 2004; Merrick and Koprowski, 2017; Milanesi *et al.*, 2017). Many models also use habitat land cover maps that are typically categorised broadly and may not accurately model semi-continuous habitat or gradual succession between patches

(Moilanen, 2011; Sawyer *et al.,* 2011). This again can lead to features being ranked higher for resistances or not being used by electrical movement models, despite the likelihood of being used in the real world. Beyond these potential weaknesses within the models, resistance modelling requires large environmental datasets to create accurate movement estimates (Dickinson *et al.*, 2018). In having this requirement, it lowers the suitability for this approach in developing countries or understudied regions, which may not have the resources to establish the baseline data needed. It may also lead to the overuse of unsuitable datasets to cover knowledge gaps.

Programmes that use resistance layers such as Circuitscape are vulnerable to researcher bias when assigning resistance values to landscape features. Generated circuit theory outputs vary greatly depending on the methods and in some cases aims of the research being carried out (Braaker *et al.,* 2014). Resistance values are largely influenced by the quantification of barriers, a non-standardised and potentially subjective measure of the difficulty of movement. When comparing values across literature, roads were given different resistance values that ranged from 8/100 to 100/100 depending on the study (App *et al.*, 2022; Braaker *et al*., 2014; Braaker *at al*., 2017; Charney, 2010; Grafius *et al.,* 2017; Moqanaki and Cushman, 2016; Tarabon *et al.*, 2021). While some variation will be species specific, resistance values still differ across studies that evaluate the same species and those that look at habitats rather than a chosen taxon. Studies specifically measuring the impact of a feature like roads on connectivity, typically gave roads a higher value of resistance than those looking at general connectivity. Unsurprisingly, this then generates outputs that show roads are a significant barrier and seemingly validates the hypothesis put forwards by such papers. To further complicate resistance value setting of roads, studies using GPS data have shown that some species exhibit a form of growing habituation to roads the longer a home range is established (Dickson and Beier, 2002). This means relative resistance values of the same road can be different for each individual within a population of the same species.

Alternately, habitat connectivity studies that use previous papers to assign resistance values can become too generalised by assuming similar results. Fey *et al* (2016) indicated roads posed no barrier to red squirrel (*Sciurus vulgaris*) distribution in urban environments, but was subsequently used in Tarabon *et al.*, 2021 to set a low value of road resistance in an only partially urbanised environment. This type of value setting through literature is common, and due to the limited number of original datasets it can lead to an oversimplification or overuse of data beyond the scope of original papers (Merrick and Koprowski, 2017). Even within the field of barrier classifications there can be variation; some studies separated roads by size while others classified them all uniformly irrespective of size, traffic flowrate and speed. This type of subjective analysis allows data to be heavily biased and greatly influences generated outputs, which are mathematical and not behaviourally generated through novel observation datasets (Laliberte and St-Laurent, 2020). The limited number of studies that have compared resistance based models to real world observation / GPS data, have shown them to be largely inaccurate or too generalised for many species (Laliberte and St-Laurent, 2020, LaPoint *et al.,* 2013, Merrick and Koprowski, 2017; Pullinger and Johnson, 2010; Teitelbaum *et al.,* 2020; Keeley *et al.*,2016; Wasserman *et al.,* 2010; Zeller *et al.,* 2014). Additional comparisons to models made with expert opinions have been created, but often note significant differences to circuit theory approaches (Keeley *et al*.,2016; Stevenson-Holt *et al*., 2004). Several of these papers conclude their model has greater accuracy than the expert modelled routes without strong supporting evidence (Sartor *et al.*, 2022; Milanesi *et al.*, 2017), while other studies suggest that expert knowledge has a negative impact on model accuracy (Seoane *et al,* 2005). Attempts to validate models using further statistical tests such as Monte Carlo cross validation techniques, are often unsuitable for connectivity models. This type of cross validation uses pseudo-random sequences of the data to test for randomness and model fit (Guo, Lui and Lu, 2019). However, due to the nature of Circuitscape spatially modelling each possible node combination, generated outputs are likely to suffer from the same biases as the original dataset and always result in a high model fit. Despite this, such methods can be useful in validating resistance calculations through statistical analysis, rather than subjective attribute evaluation (*Kuroe et al.,* 2010). However, due to the complicated nature of these tests, use of them within the literature appears to be limited.

High levels of confliction within literature, lack of uniformity in methods and potential bias when performing analyses, clearly indicate significant risks with the validity of such methods and their use for conservation planning (Rudnick *et al.*, 2012). While circuit theory applications are still useful, they require sound ecological knowledge of the species present, and must be based on the conditions at a specific site (Hunter-Ayad and Hassall, 2020). As such, without expert opinion, high detail species presence information, and high detail species absence data, using these methods may create more problems than solutions (Zeller *et al.,* 2014). Therefore, a simpler structural landscape connectivity analysis technique may be more useful and provide more reliable insight (Charney, 2010).

For the work carried out in this study, connectivity was modelled on a habitat network basis using Condatis. Condatis is a specialist connectivity software that uses the principles of circuit theory to model the speed of a population moving through a habitat network (Travers *et al.,* 2021). Unlike other models which use subjective resistance layers to shape movement networks, Condatis uses conductivity which is based on habitat patch size (Area) and location (Figure 1.1). The larger a habitat patch is, the higher its conductive value within the model, while distance and direction of movement are also considered. Current is applied directionally and will identify the best route for movement as that which contains the largest patches with the least space between them and most direct route.

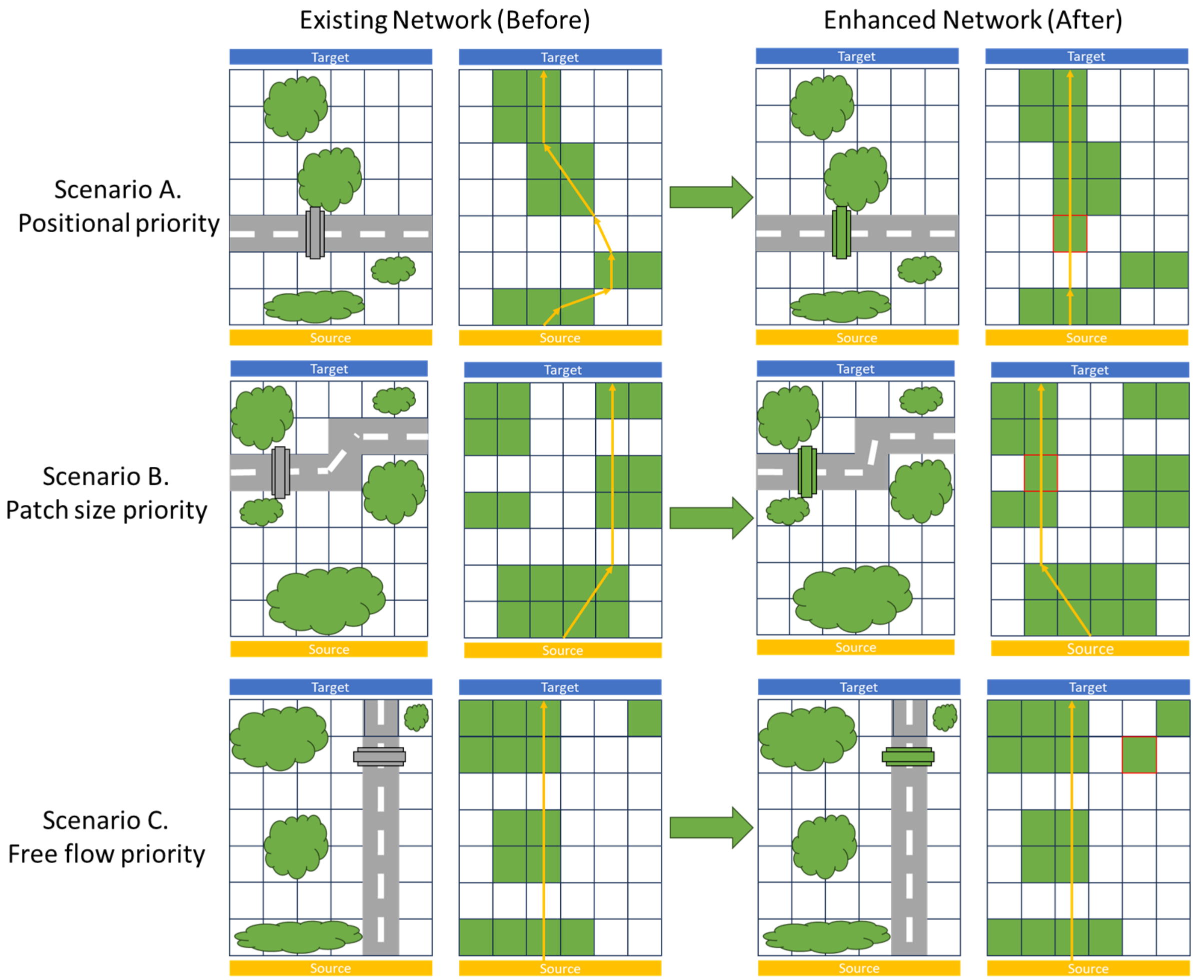


Figure 1.1 Contatis model output representations, demonstrating positional priority within the model (A) where location and proximity are taken into account, patch size priority (B) where current is preferentially attracted to larger habitat patches in the direction of current flow, and the models free flow analysis (C) where movement is calculated from source to target without being forced through any particular points and travels freely through the habitat patch network. (White patches represent non target habitat that are not conductive, green squares represent target habitat presence within the model and are conductive, red outlined squares are simulated habitat generation, yellow line indicates current flow through the modelled network)

Alternate circuit theory applications largely use the random walk approach, with each cell of a surrounding point being evaluated in any direction for its suitability (McRae *et al.,* 2008). This can create large aberrations within current flow with routes avoiding short high resistance movements, preferring significantly longer but lower resistance routes (Mahmoodi *et al.,* 2023). In many cases this results in outputs which are opposed by field observations of animal movement within the literature (McClure *et al.* 2016). By applying current directionally as seen in Condatis, it allows the model to simulate choices and movements of individuals that are travelling with purpose. Modelling dispersal in this way also demonstrates routes that result in gene flow, as opposed to routes that may be mathematically generated but lack any ecological reason for movement (Beier *et al*., 2008). This theory is used throughout chapter two to evaluate and quantify connectivity of broadleaf woodland and coniferous woodland across the UK.

As with many aspects of theoretical modelling, there are limitations and assumptions made for each step of circuit theory modelling. Despite these, it is still one of the most robust and accessible ways of modelling species movements through the landscape. While the assumptions and limitations of the method should be considered, there are currently few other approaches that offer the same benefits when modelling theoretical movement networks, without GPS data for tracked individuals.

1.3 Potential for wildlife-vehicle collision data use in species distribution monitoring.

To enact practical conservation processes, it is necessary to understand species distributions and ecosystem health. To increase our understanding, numerous survey methods are used to monitor species or habitats of interest. One potentially underutilised survey data type is wildlife-vehicle collisions (WVCs), despite its potential for increasing knowledge and understanding of many species (González-Gallina *et al*. 2016; González-Gallina *et al*. 2018). The presence and analysis of WVCs contributes significant evidence of species distributions and provides a unique opportunity to study biodiversity and population health (Schwartz *et al*., 2020).

Roadkill and WVC monitoring have previously been used to track disease outbreaks (Brockie *et al*. 2009), monitor invasive species (Caley *et al*. 2015) and to provide information on elusive species that can be difficult to monitor using conventional methods (Canova and Balestrieri, 2019). In some cases, roadkill surveying has even outperformed other types of ecological monitoring such as mammal trapping when quantifying species richness (González-Gallina *et al*. 2016). Additionally, WVC monitoring projects are one of the few types of surveying that covers multiple taxonomic groups and provides an insight into biodiversity, which are not constrained by specific survey areas or habitats (Medrano-Vizcaíno *et al.,* 2023). Yet, other than specifically designed WVC survey studies, this type of data collection is rarely used in tandem with other methods. By comparing WVC data to other sources of publicly available species survey data, this study aims to demonstrate the utility and potential of WVC data for wildlife conservation.

1.4 Aims and objectives.

Roads are barriers that lower connectivity and cause increased mortality for wildlife populations. The aim of this study is to investigate the potential of greening current road infrastructure to benefit habitat connectivity, while also evaluating wildlife-vehicle collision (WVC) data for use within species monitoring.

To achieve this aim, the study will:

1. Determine the extent of grey infrastructure presence in the UK.
2. Estimate current connectivity values of target habitat networks (broadleaf and coniferous woodland) using specialist circuit theory software Condatis.
3. Quantify the potential changes in connectivity when identified grey infrastructure is incorporated into existing habitat networks.
4. Evaluate infrastructure adaptation and its potential benefits for conservation.
5. Collate ecological and WVC data for multiple species throughout the UK from ‘The Road Lab’ and National Biodiversity Network (NBN).
6. Compare WVC spatial data distribution and data prevalence to other sources of species presence data within the NBN database on a species-by-species level.
7. Evaluate the relative importance of WVC data to species monitoring and its ability to provide supplementary distribution data to standard species monitoring techniques.

Chapter 2: Quantifying the potential benefits of integrating road infrastructure into wildlife movement networks.

2.1 Introduction

Maintaining habitat connectivity is a key aspect of conservation planning, due to its importance for vital ecological processes. Population health has been implicitly linked to a species ability to move, with barriers impacting genetic health through restricted dispersal, as well as physical health through resource restrictions (Fahrig, 2003). Urbanisation and land development are leading causes of decline in habitat connectivity, separating large untouched areas into smaller fragmented patches (Moqanaki and Cushman, 2016). This degradation of natural habitats is a driving force of global biodiversity loss, on both large and small scales (Kaszta *et al.*, 2020). One of the most common forms of habitat fragmentation is the construction of linear transport infrastructure. These features are typically built in both untouched habitats to facilitate faster movement through remote areas, and in heavily developed locations to enable movement in high traffic areas (Bennett, 2017). As such, the construction of transport infrastructure can impact a wider range of species and habitats, when compared to other forms of concentrated development like housing or industrial constructions (Zhuo *et al.*, 2022).

Linear transport infrastructure negatively impacts wildlife and ecosystem functions by increasing mortality rates and lowering habitat quality (Polak *et al.,* 2014; Gonçalves *et al.,* 2022 Moore et al., 2023). Roadways are one of the most detrimental forms of transport infrastructure for wildlife, due to their high occurrence rates, construction densities and traffic flow rates (Taylor and Goldingay, 2010; Cooke *et al.,* 2020). Global estimates indicate that more than 64 million kilometres of paved roads have been constructed around the world, a value set to increase up to 60% by 2050 (Meijer *at al.*, 2018; Laurance *et al.*, 2014). This not only represents a significant loss of natural habitat, but is indicative of a high level of habitat degradation and fragmentation (Jaarsma and Wilems, 2002; Didham, 2010). The consequences of fragmenting a habitat can include indirect mortality from resource scarcity (Horvath *et al*., 2019; Passoni *et al.*, 2021), or direct mortality due to wildlife-vehicle collisions (WVC's) (Cullen Jr. *et al.,* 2016). Despite this, practical implementation of mitigation strategies to lessen the impact of roads are relatively infrequent (White and Hughes, 2019) and vary by country and in some cases economic conditions rather than conservation value (Rytwinski *et al*., 2016).

Mitigation strategies exist in a variety forms, with more than 40 designations globally (Rytwinski *et al.,* 2016), although the exact design and implementation varies by region and available funding. As such, more research and higher rates of implementation of mitigation strategies have occurred in higher income countries compared to low and middle income nations (Collinson *et al.,* 2019).Common strategies include overpasses and underpasses which are placed above or below roads to decrease the barrier effect, by enabling the safe passage of wildlife from one side to the other (Paemelaere *et al.,* 2023). Additional features like fencing and driver warning mechanisms can also be implemented, however fencing can lower connectivity if poorly placed (Leblond *et al.,* 2007) and warning signs have inconclusive benefits (Bond and Jones, 2013; Collinson *et al.,* 2019). A combination of approaches designed to work in tandem is often more affective (*Boyle et al.,* 2021; Rytwinski *et al*., 2016), but can become financially unsustainable for large projects (Knifka *et al*., 2023). This increases resistance to their implementation, especially if data on their success rates is lacking (*Van der Grift et al.*, 2013; Knifka *et al*., 2023).

An alternative or supplementary approach to mitigation creation, is the modification of existing structures or 'grey infrastructure' like pedestrian underpasses and footbridges (Li *et al.,* 2017). These structures are often built around highly developed areas with low pedestrian connectivity, or in areas where roads pose a significant barrier or threat to humans (Salamak and Fross, 2016). By adapting these 'grey' structures to also benefit wildlife, habitat connectivity could be increased at a relatively low cost when compared to traditional mitigation structure creation (Lu *et al.*, 2023; Karthaus, 1985). This approach has already been trialled in the Netherlands where existing footbridges and passages were adapted through the addition of wood planks and natural substrate on the bridge floor, to benefit native species of amphibians (Veenbaas and Brandjes, 1999). Once adapted, 77% were used by amphibians during annual migrations the following year, indicating the concept could be successful (Veenbaas and Brandjes, 1999). This is further supported by camera trapping studies in Sweden (Bhardwaj *et al*., 2020) and Portugal (Grilo *et al.,* 2008) that found many species already use infrastructure not specifically designed for them.

This study uses a theoretical approach to estimate the connectivity changes associated with the 'greening' of existing road infrastructure throughout the UK. Circuit theory software Condatis was used to model changes in habitat connectivity before and after proposed greening has occurred. Circuit theory is a conceptual model where habitat networks are treated similarly to circuit boards, with current flow being used as an analogous measure for species movement (McRae et al., 2008). Site connectivity was calculated at a 'before' stage that did not include proposed infrastructure, and at an adjusted 'after' stage where infrastructure was incorporated into habitat networks and represented the simulated infrastructure greening efforts. This approach gives a comparative overview of the value of 'greening' existing infrastructure and investigates it as a potential method for increasing habitat connectivity.

2.2 Methods

2.2.1 Grey infrastructure identification

To determine the potential benefit of greening existing infrastructure, a data search was completed to identify current locations of grey infrastructure. For this investigation suitable 'grey infrastructure' was defined as any road crossing structure that would be accessible to wildlife, but that had not been purpose built for wildlife mitigation or movement. These structures included 28 designations such as pedestrian underpasses, bridges and towpaths. Structure information and locations came from a variety of sources including, National Resources Wales, Ordnance survey data and Open Street Map. (See Appendix 1 for a full list of structure designations and data sources). Culverts were also identified but were not included in the analysis due to a lack of structural uniformity, and inconsistent availability of information. Among the 1.363 million culverts identified, diameter ranged from 0.005m diameter to 50m, although most lacked accurate size data. Given the missing information on their design, it was unclear if they could provide passage to wildlife and therefore, they were excluded from the analysis. Data were identified through online GIS layer searches, and then combined into a single dataset.

The final dataset included 150,561 structures throughout the UK. Virtual ground truthing of structures was carried out using Google maps to check suitability of structures in terms wildlife access, and potential for modifications. A total of 4,810 (3%) randomly selected structures were evaluated, being classed as either suitable or unsuitable for greening. Structures that were inaccessible to wildlife such as those over enclosed sewers were deemed unsuitable, along with those labelled as bridges but that showed no evidence of above ground structures being present. These structures may either be mislabelled within the data source, or in some cases represent engineering structures beneath the surface of a road, but that are not possible to modify for wildlife movement. Ground truthing found a rate of 4 suitable (accessible to wildlife) structures for every 1 unsuitable (inaccessible to wildlife). Due to the size of the dataset and time limitations, it was unfeasible to check each structure individually. Therefore, unsuitable structures remain within the dataset used in this analysis at an assumed rate of 1:4.

2.2.2 Habitat network creation

Due to the size and scale of the proposed analysis, the UK was split into smaller areas to lower computational demands and increase variation of habitat networks analysed. One hundred 25km x 25km squares (each 62,500 hectares in area) were generated at random within ArcGIS Pro (Version10.2.1.), using the random point generation tool. These squares (termed study sites) were created throughout England, Scotland and Wales, with a minimum distance between central points of each study site of 12,500m (Figure 2.1). This minimum distance removes the opportunity of duplicate sites being created and increases the number of novel areas for analysis. These study sites were used to identify naturally existing habitat networks, and the grey infrastructure present around them. Analysis was limited to 100, 25km x 25km square study sites (62,500 hectares), due to constraints placed on analysis by the computational limitations of the software. Preliminary analysis of larger study site sizes (50km x 50km, 40km x 40km and 30km x 30km) were attempted but failed due to technical limitations of the Condatis software. 25km x 25km was the maximum study site size possible within the limitations of the software and as such was selected as the optimal size for analysis. 100 sites were selected due to the processing time available, with each site taking 12 to 16 hours to analyse. In total, these 100 sites cover 6,250,000 hectares or around 25% of the total land of the UK. However, some overlap does occur between study sites but was limited by the minimum distance mentioned above.

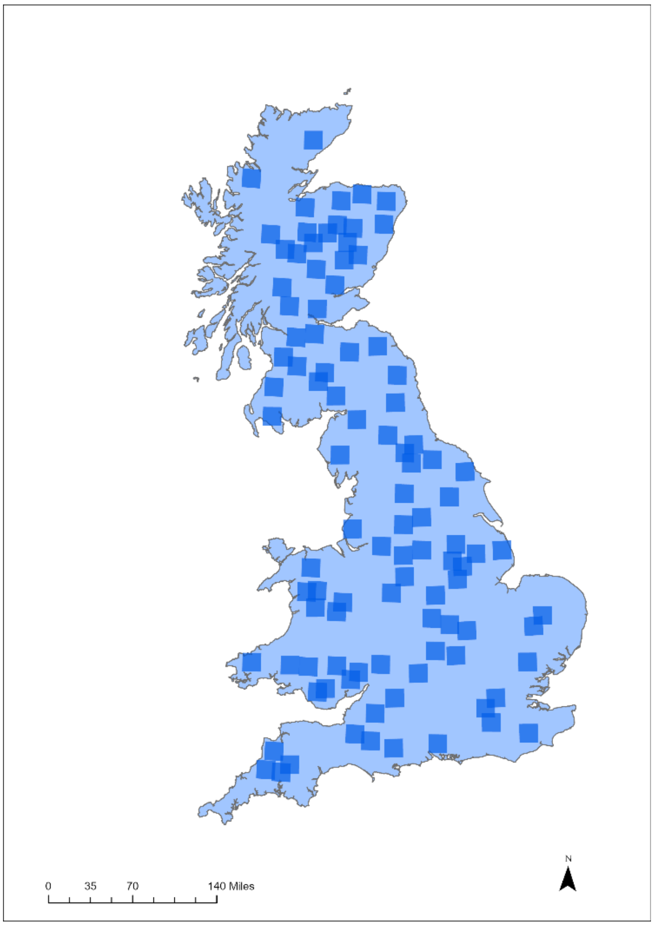


Figure 2.1 Study site distribution throughout England, Wales and Scotland. Site locations and extent are marked with dark blue squares.

Habitat networks were individually created for broadleaf and coniferous woodland habitat, using land cover data from the UK Centre for Ecology and Hydrology at a 25m resolution (UKCEH, 2021). These woodland habitats were chosen due to the relative abundance, distribution and to investigate potential differences in distinct terrestrial habitats that have differing values to biodiversity. Other habitat types such as grassland were considered for analysis, but rejected due to difficulty in identifying an appropriate level of classification. A broad characterisation of all grassland produces expansive habitat networks which are too large to be analysed by Condatis. Separating them into classifications such as acid, neutral and calcareous grasslands, limits direct comparisons because these classifications were not present in all study areas.

After being modelled individually, all habitat networks were then combined in the final dataset, which compared variables to each unique network. Habitat networks were created using existing woodland patches to represent the 'before' network, which contained the current (as of 2021) extent of habitats without modification. The grey structure dataset was then merged into the habitat network layers to create a second theoretical and enhanced 'after' network, containing both the target habitat patches and grey structures present in each study site. This second 'after' network represented the integration of structures through simulated 'greening' and was used to quantify the relative impact on connectivity when compared to the 'before' network within the same study site.

2.2.3 Connectivity modelling

To quantify the impact of structure greening on connectivity, each habitat network was modelled using Condatis 1.2, a circuit theory modelling software (Wallis and Hodgson, 2012). Circuit theory analysis is based on the assumption that habitat networks can be similarly modelled to electrical circuits, with suitable habitat patches conducting current and unsuitable areas typically acting as a resistor to current flow (Hodgson *et al., 2012*). Current travels through the simulated habitat network as a proxy for animal movement. Unlike commonly used Circuitscape software (Anantharaman *et al*., 2020), Condatis applies directional current to the habitat network rather than relying on artificially placed nodes. Current passes between patches and generates a value of conductivity as a measure of the speed of movement through an area (Hodgson *et al.,* 2012). This value for movement is used as a measure of structural habitat connectivity throughout this study.

Condatis models were run based on habitat networks rather than species specific ecological requirements, and without barrier regions of resistance. By choosing this approach, the model represents a range of species and focuses on the benefits to overall network connectivity as proposed in Travers *et al.,* (2021) and supported by Gonçalves *et al.,* (2022). Default values were used within Condatis, including a 5km dispersal range from habitat patches. Dispersal values define the electrical current range used to generate connected pathways between woodland patches within the model. Values of connectivity within Condatis prioritise patch size, proximity between patches and directionality/simplicity (Figure 2.2). Large patches have a higher value for conductivity compared to smaller patches, therefore attracting more of the current which is being used to model movement networks. Beyond the single patches, habitat networks with a direct route of travel and with smaller gaps between patches, also result in the highest values of connectivity/conductivity. Networks of small, sparsely located patches, that do not allow current to flow in the direction of travel, have low overall current flow. This functionally means that there is a trade-off between these three characteristics which enables the simulated network to behave more naturally than other circuit theory resistance models.

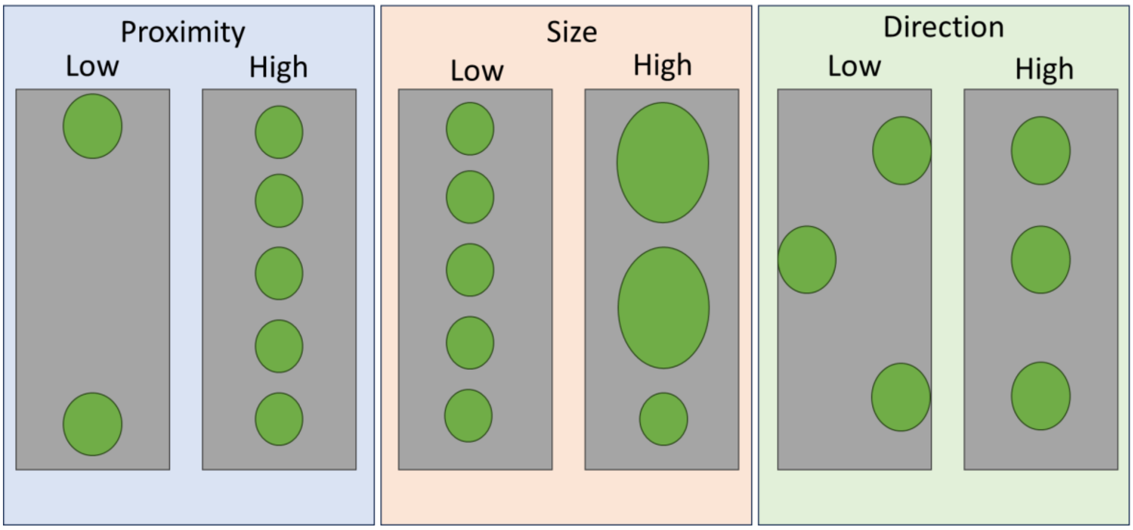


Figure 2.2 Representation of current flow considerations within Condatis software while calculating network connectivity and simulated movement pathways. Green areas represent target habitat patch size and network complexity, with the highest current and lowest current variations identified.

Mean connectivity values were calculated for each network within a study site using longitudinal (South to North) and latitudinal (East to West) current flow. Sites with less than 500 hectares (1% study site coverage) of target habitat were excluded from analysis due to limitations within the software and a high margin for error. This accounted for six study sites which could not be analysed, reducing the sample size to 94. Cell size was set at 100m resolution, with all cells containing at least 51% coverage of the target habitat included within the connectivity model. Comparable measures of connectivity were calculated before and after simulated greening occurred. The comparisons of each network model were used to quantify connectivity changes after infrastructure was incorporated into the habitat networks. For analysis, both habitat types were merged into a single dataset, comparing changes across individual networks before and after. Multiple habitats were used as a variable to determine if the impact of greening was habitat specific, and to increase the variety and structure of networks being analysed. Designations of the types of grey structures included in the analysis can be found in Appendix 1.

2.2.4 Data analysis

Once connectivity was quantified for each individual network, the change in connectivity between the before and after networks was calculated. This change in connectivity was analysed to identify potentially significant associations with landscape variables. With change in connectivity as the dependent variable, a general linear model (GLM) was fitted with habitat type (broadleaf/coniferous), habitat abundance (Hectares of target habitat within a study site), Urbanisation (Hectares of developed land including roads within a study site) and number of 'greened' structures added to each 'after' network as the independent variables. Model assumptions were checked by exploring the distribution of residuals, and led to selection of a gaussian error family and identity link function. Variables were removed from the model using a stepwise deletion approach based on a p-value threshold of <0.05, leading to the removal of habitat type. Preliminary data exploration showed that road density was highly correlated with urbanisation, therefore road density was not included in the model. Urbanisation was included instead of road density due to its overall larger footprint in the connectivity model and presence throughout study sites. All analyses were carried out in R version 4.3.1 (R Core Team, 2022).

2.3 Results

2.3.1 Characteristics of habitat networks.

Habitat abundance varied between study sites, producing a wide range of networks analysed for connectivity, with total range and coverage shown in table 2.1. Broadleaf woodland had a mean abundance of 3,810 hectares (SE ± 291), while coniferous woodland had a larger mean abundance of 4,688 hectares (SE ± 324) per study site. The amount of infrastructure (greened structures) modelled per site varied between 9 and 4,498 structures in a single study site.

Table 2.1: Habitat characteristics of study sites, including mean abundance (hectares) and total coverage of a study site (% value) per habitat classification.

Habitat type	Mean abundance (Hectares)	Mean site coverage	Minimum abundance (Hectares)	Minimum site coverage	Maximum abundance (Hectares)	Maximum site coverage
Broadleaf woodland	3,810 SE ± 291	6% SE ± 0.4%	27	0.1%	13,172	21%
Coniferous woodland	4,688 SE ± 324	7.5% SE ± 0.5%	1	0%	22,791	36%

2.3.2 Impact of infrastructure greening on habitat connectivity.

Existing network connectivity values ('before') varied from 0.3 to 99.6, expressing the wide range of network structures present (Figure 2.3). Once infrastructure was incorporated into the networks, the indicated mean connectivity across all existing habitat networks increased from 25.78 (SE ±2.04), to 27.34 (SE ±1.96). The mean connectivity per study site increased by 12% after structure inclusion. However, due to a skewed distribution, a median of 8% change is likely more representative of the total dataset and reduces the impact of potential outliers.

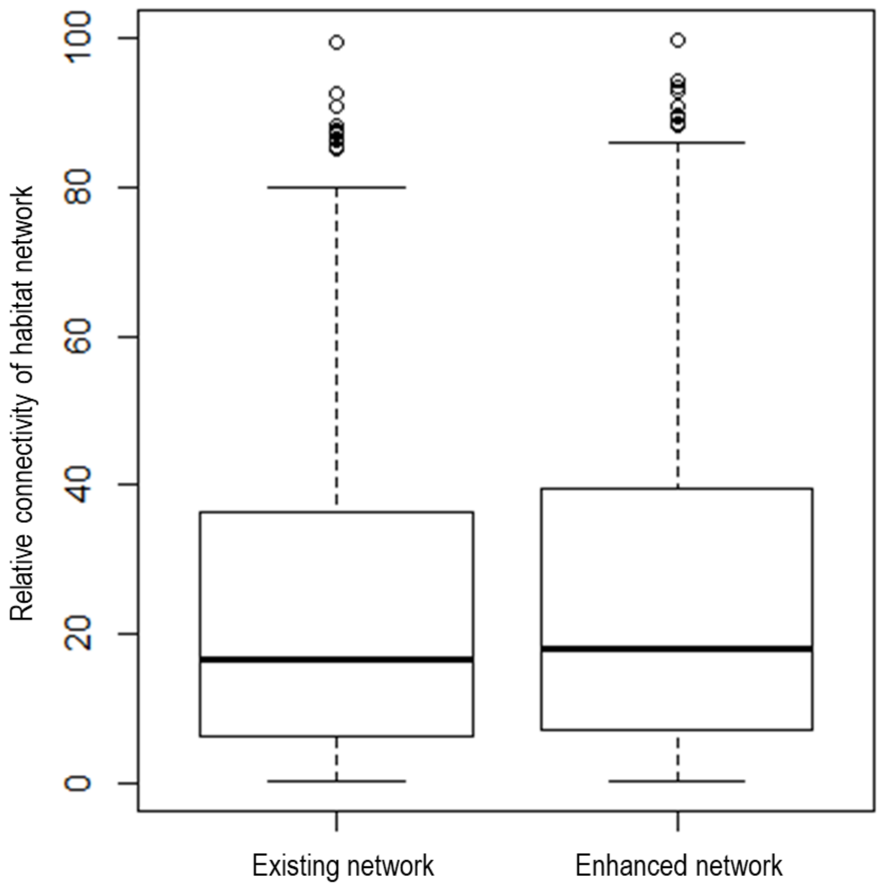


Figure 2.3 Habitat connectivity of all modelled networks before (existing network) and after (enhanced network) infrastructure was incorporated into habitat networks. Black circles identify potential outliers within the data.

2.3.3 Statistical analysis

A Generalised linear model was carried out to compare the change in connectivity against habitat type (broadleaf/coniferous), habitat abundance (total hectares), number of structures and amount of urbanisation (Hectares of developed land). The GLM described significant variation in connectivity change, in association to the variables tested ($F_{4, 168}$ = 81.17, p <0.001, Rsq = 0.681)

Analysis indicated a significant positive correlation between increasing network connectivity and the number of structures added to the network (t=4.341, df = 169, p = 0.002). Habitat abundance was also significantly associated with positive increases in habitat connectivity, as the amount of target habitat increased (t=6.637, df = 169, p < 0.001). The amount of urbanisation was borderline significant with a negative impact on connectivity as urbanisation increased (t= -2.393, df = 169, p=0.049). Habitat type was not significant to changes in connectivity (t= -0.680, df = 169, p=0.48).

To investigate relationships between habitat abundance and number of structures, networks were compared using these variables in relation to connectivity change (Figure 2.4). Sites ranked lowest for connectivity change (plotted in red below) had minimal existing habitat abundance and/or minimal numbers of structures added. Those with a moderate value for both variables, typically resulted in higher connectivity changes within the modelled dataset.

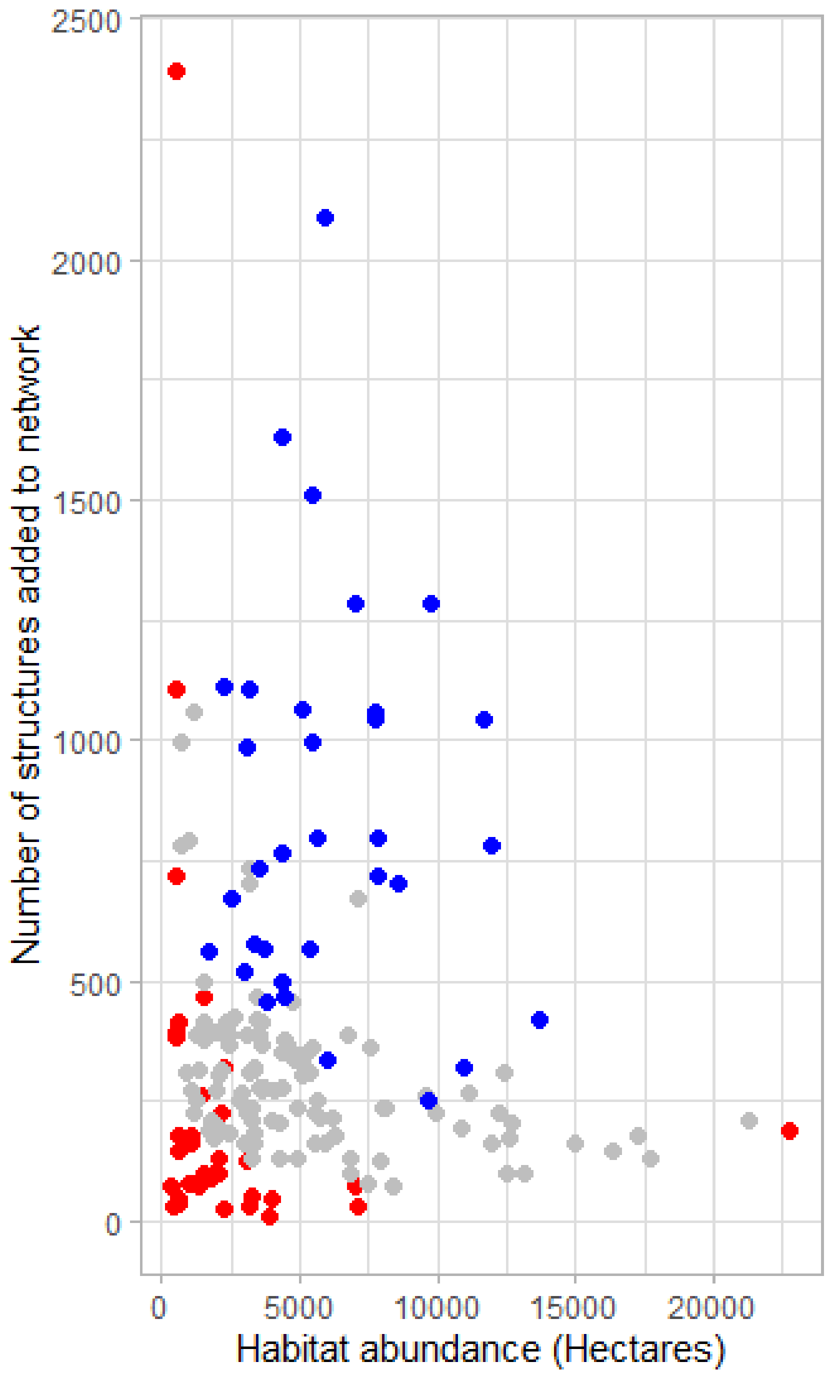


Figure 2.4 Comparison of habitat abundance and number of structures across the top 20% (blue) and bottom 20% (red) of sites ranked for connectivity change. Grey dots are remaining 60% of data points between chosen categories.

2.4 Discussion

Connectivity analysis demonstrated infrastructure greening has the potential to increase connectivity for wildlife and enhance existing habitat networks throughout the UK. This impact appears to be independent of habitat type, with a median increase in connectivity of 8% over a 25km^2 area. This represents a significant increase considering the relatively small amount of land needed compared to other network assessments (Augustynczik, 2021; Nguyen *et al.,* 2021). Each structure was measured at a 100m x 100m resolution (1 hectare) as used by the Dutch habitat defragmentation program (Sijtsma *et al.,* 2020), and resulted in connectivity benefits similar those seen in other green infrastructure studies (MacKinnon *et al*., 2023; Nguyen *et al.,* 2021). The impact per hectare indicates current infrastructure designed for human connectivity, may be placed in locations that could benefit wildlife if adapted accordingly (Wang *et al.,* 2018).

Statistical analysis identified that habitat abundance (hectares of target habitat), and the number of structures, are significant variables at predicting positive connectivity change. In tandem, these could be used to find the most suitable areas for structures to be greened. Ideal circumstances are indicated to be those with large but fragmented habitat networks, where there are also large numbers of structures to be greened. This makes logical sense as modifying structures in this type of environment helps connect patches at multiple points, to create larger networks overall. The data suggests regions with 2,500 – 10,000 hectares of target habitat and over 300 structures benefit most from this type of greening. Areas with high habitat abundance but low numbers of structures, as well as those with a small amount of target habitat but large numbers of structures, did not appear to benefit significantly from structure greening.

Sites with fewer than 200 structures (54) showed a mean increase in connectivity of just 6%, half the average of the total dataset. While 21 of

26 sites (81%) containing fewer than 102 structures, resulted in a less than 3% increase. These comparable values indicate that small scale greening of a handful of grey structures may not provide much value to wider network connectivity, when compared to extensive greening programmes. However, as the number of structures increases so does the relevant costs to modify them.

Habitat generation costs listed by the Environment Agency (2015) indicate the per hectare cost of broadleaf woodland is £9,174 and coniferous woodland is £8,674 (including land purchases). This is a relatively low value when compared to designing and building new mitigation structures, which can often cost over £5 Million (Brennan *et al*., 2022). In the Netherlands, a national programme designed to reduce habitat fragmentation through the construction of wildlife crossing bridges, spent £240 million to generate 1,734 hectares of crossing structures (Sijtsma *et al.,* 2020). This equates to a cost per hectare of £138,400. By using existing structures, the cost of greening would be substantially lowered and as such pose a cost effective alternative or supplementary approach to structure creation. In locations without existing grey structures, new mitigation measures such as underpasses would still be necessary to enhance connectivity. Designing these to work in tandem with proposed structure greening, would increase the effectiveness of both forms of mitigation.

While connectivity is seen to increase across study sites in this project, the real-world impact of physical greening is still uncertain. One of the limitations of this research is the lack of supporting fieldwork looking at current species crossing rates. Camera trapping studies have previously been carried out on mitigation structures to monitor effectiveness (Hamilton *et al.,* 2024) and could similarly be used to monitor grey structures before greening. By completing monitoring before and after any proposed structure modifications in this manner, crossing rates and species utilisation could be directly compared and the effectiveness of greening evaluated. However, without additional field work and observations such as these, it is not possible to conclusively say that connectivity will improve after greening. Species may avoid structures or still cross roads directly even after greening and more research would be needed to determine if structures are used after greening. Despite this, information can be taken from literature and the parallel success of specifically designed mitigation schemes. Studies have shown that many species use mitigation crossing structures (Kusak *et al.,* 2009; Sawyer *et al.,* 2016), while others have also shown wildlife to inherently use grey infrastructure (Wang *et al.*, 2018; Bhardwaj *et al*., 2020). It is therefore possible to assume that grey to green structures would be utilised in such a way by wildlife, if modified correctly and based on ecological data for the area involved (Bhardwaj *et al.*, 2020; Grilo *et al.*, 2008).

While literature exists on mitigation structure usage by wildlife, many of these studies were not performed in the UK, limiting their direct applicability. Studies frequently focus on endangered species without similar allegories in the UK such as tigers (*Panthera tigris*) (Saxena and Habib, 2022) and elephants (*Loxodonta africana*) (Okita-ouma *et al.,* 2022), or those which pose a risk to drivers like moose (*Alces alces*) (Bhardwaj *et al*., 2020) or bears (Morales-González *et al.,* 2020). Data from these studies are useful to increase our understanding, but findings often indicate usage factors are species and in some cases regionally specific (Jurečka *et al.,* 2024; Hamilton *et al.,* 2024; Bar-ziv *et al.,* 2022; Prokopenko *et al.,* 2016). However, these studies from other countries highlight the fact that variables from construction dimensions, ground substrate, vegetation proximity and human presence, create a complex matrix of variables which need to be considered. To fully understand and maximise the potential benefits for native wildlife, the utilisation of structures would need to be studied in the UK. This research would benefit both specifically designed wildlife mitigation structures and identify features to include in structure greening.

There is a lack of information describing either how widespread or effective purpose-built wildlife mitigation strategies are in the UK. Mitigation structures are likely to be common across the UK due to legal requirements and inclusion as compensation strategies within environmental impact assessments (Matos *et al.,* 2017). Despite this, there is no legal requirement to report any data after construction. Most structures are not monitored, and there is no coordinated retention of information on their locations after construction. Data requests made, during the current research project, to the governing bodies of roadways in England and Wales (Highways England and Welsh Trunk Road Agents) resulted in the identification of only 27 structures, excluding wildlife fencing, and with no information on historical monitoring. Responses did not include structures that were identified from published literature which had been created with the support of these agencies, suggesting a lack of communication and centralised knowledge about this topic in the UK.

Similarly to the assumptions of this study benefiting from mitigation research, so too does it fill a current gap within the wider literature. Many studies looking at structure passage rates do not take wider connectivity into account (Paemelaere *et al.,* 2023; Stewart *et al.,* 2016; Polak *et al.*, 2014). However, the findings of this study indicate that single structures may not pose significant benefits to landscape scale connectivity on their own, even if they mitigate wildlife vehicle collisions and help connect two patches. A lot of focus is currently used to identify the largest barriers to movement, but do not quantify connectivity benefits or look beyond those structures during analysis. More effort may be needed to assess the wider relevance of structures in their role of improving access to additional habitat patches and networks. If used in tandem, small scale barrier mitigation studies and larger scale connectivity analysis could create a better understanding of structure priorities. Crossing rate studies help identify structure properties and site importance on a small local scale (Chapman and Hall, 2022), while connectivity analysis on a larger scale helps put them into a context of entire movement networks (Wilmers *et al.,* 2013; Rathore *et al.,* 2012). Further studies and work on this topic are needed to fully validate this method and the findings of this study.

2.5 Conclusion

Greening road infrastructure is shown to increase connectivity in woodland habitats, despite the restrictions of being placed on roads and without having been targeted to benefit wildlife when constructed. With more than 150,000 such structures present around the UK, they present a high potential network of infrastructure to be greened on a national level. While passage rates were not investigated in this study and more research is needed to demonstrate the real world benefit, the potential for conservation is high and should be investigated further.

Chapter 3: The value of citizen science collated wildlife-vehicle collision data as a tool for species monitoring.

3.1 Introduction

Wildlife-vehicle collisions (WVCs) can be seen as a byproduct of habitat loss and fragmentation. Bisecting natural habitats with roads exposes wildlife to increased risk of collisions during dispersal or to reach isolated resources (Neumann *et al*., 2012). As such, mortality rates often increase in areas where roads are present (Hill *et al*., 2019). The presence of roadkill, however, offers useful evidence for conservationists of a species presence. It also provides a unique opportunity to study wildlife behaviour, biodiversity, species distributions and population health (Raymond *et al*., 2021; Schwartz *et al.*, 2020; González-Gallina *et al*. 2016).

The relevance and potential uses of WVC data go beyond informing road mitigation strategies, by providing insights into native and invasive species which other survey methods may miss (Kindberg *et al*., 2009;_Guisan *et al.*, 2013). Ecological data collection is typically expensive, time consuming, and requires a high level of training and competency to maximise its effectiveness (Canova and Balestrieri, 2019; Zhou and Griffiths, 2007). These limitations can cause gaps to exist in our understanding of species distributions, as projects and surveys are often designed to maximise efficiency of a few technically skilled surveyors (Dri *et al.*, 2022;_Jones 2011). For example, many species in the UK have designated survey seasons when the species is most active. However, in some cases species may also be active outside of these time periods. Newt surveying typically happens in breeding ponds between March and June, but they can be active earlier on land as they travel back to ponds (Harper *et al.*, 2019). As such, entire parts of their life cycles and yearly distribution are often missed out in preference to monitoring selective breeding sites, potentially missing useful information (Jones 2011). Effort consolidation is necessary in wildlife conservation, as much of this effort is performed by volunteers and charity organisations who lack resources to perform extensive and constant surveys over large areas (Milda *et al.,* 2020).

A possible solution to this problem, is the use of citizen science projects to report ecological data and species sightings. Citizen science is increasingly used to supplement formal survey recording methods, and has been shown to be an effective tool in scientific data collection (Follett and Streznov, 2015). Methods of involvement can include direct surveying techniques like camera trapping where equipment is sent to volunteers (Parsons *et al.*, 2018), or ad-hoc data collection through software and apps including eBird, iNaturalist and iRecord (Sullivan *et al.*, 2009). Just these three apps have recorded almost one billion observations from more than three million reporters since 2002 (Sun *et al.*, 2021). However, even these types of data collection methods require a certain level of specialist knowledge to successfully identify species and ensure a high accuracy, something that can hinder the effectiveness and appeal of these projects to a wider audience.

Comparatively, citizen science roadkill survey projects offer a lower skill, low cost, high yield option for data collection that gives constant species presence information (Shilling *et al.*, 2015; *George et al.,*2011). The passive nature of data collection and reporting reduces barriers for participation, and can provide insights into species across large areas rather than a specific surveyed location (Canova and Balestrieri, 2019). Wildlife-vehicle collisions have been monitored since the 1960's in the UK (Hodson and Snow, 1965) and extensive databases are being constructed in many countries as the impact of roads on conservation is better realised. There are currently 12 known roadkill monitoring projects worldwide (Schwartz *et al.,* 2020), totalling over 458,000 reports. The Road Lab currently ranks 3rd globally for total number of reports, with over 96,000 roadkill reported from across the UK. Comparatively, the National Biodiversity Network contains more than 125 million ecological records, collected through almost 200 partner organisations.

Despite the comparatively small size of WVC databases in relation to general ecological records, they are one of the few types of surveying that covers multiple taxonomic groups. Untargeted roadkill monitoring also benefits buy not being constrained by previous ecological assumptions of where a species should be present. By having no restrictions spatially or temporally or for particular species, it increases the likelihood of obtaining novel distribution data due to the lack of assumptions that may occur during standardised species survey planning. This project aims to quantify the relative contribution of WVC data to species distribution information in the UK, in terms of spatial extent of the data as well as species population numbers.

3.2 Methods

Between January 2013 and January 2023 citizen science project 'The Road Lab' received 85,941 roadkill reports from members of the public. These data included information on over 200 species and from a significant portion of the UK (Figure 3.1).

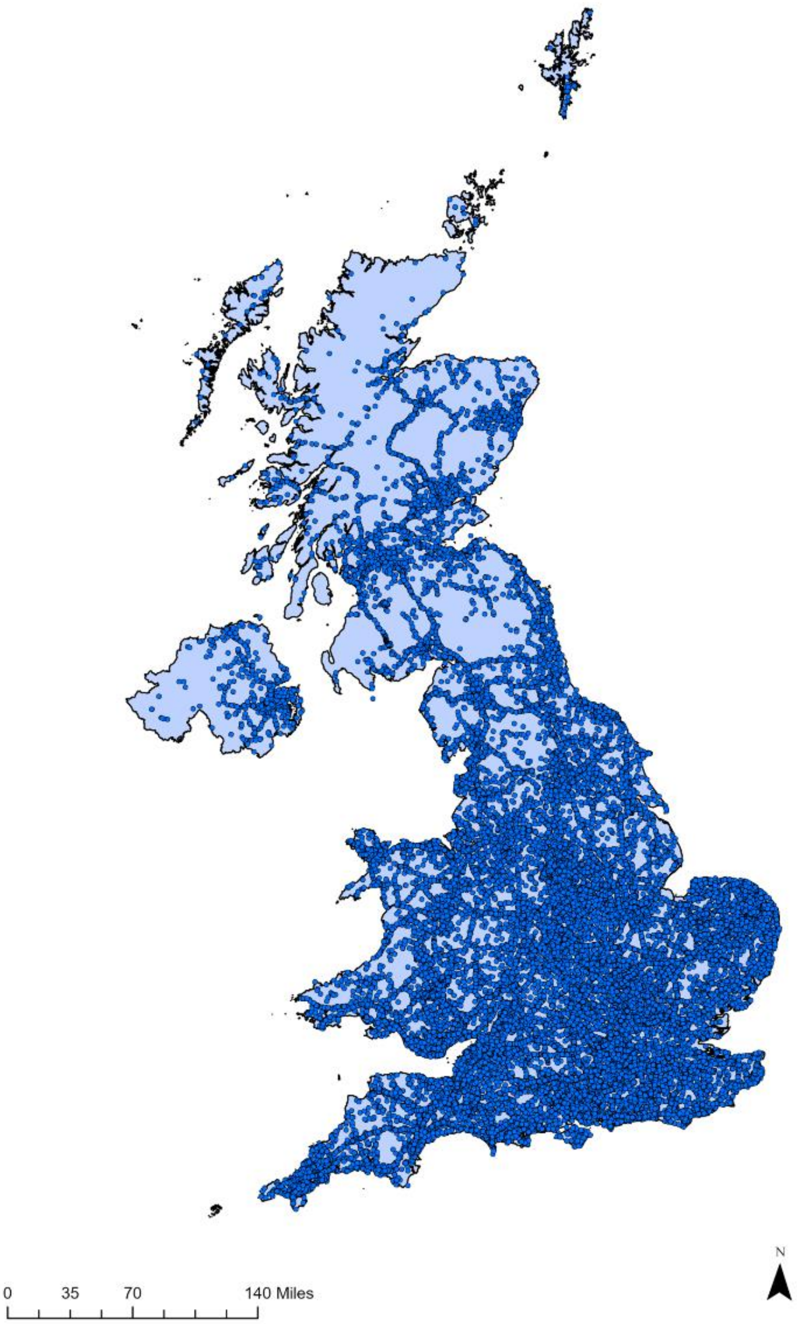


Figure 3.1 Road Lab data point distribution of reported roadkill across the UK, dark blue dots are individual reports.

From the total Road Lab database, a subset of the 33 most reported species with over 60 reports each was identified and included 62,333 roadkill data points (see table 3.1). These data were compared numerically and spatially against 1,152,220 survey data points (of the same 33 species) collected from the National Biodiversity Network (NBN). To maximise comparability, NBN data were limited to the 'human observations' category that had been confirmed within the database between January 2013 and January 2023. Human observations include recorded field signs, direct human observations and remote sensing such as camera trapping, but exclude museum records or preserved samples (NBN, 2024). Of the listed data providers, those marked as "UK Roadkill records" were removed as these were duplicates from The Road Lab data present within NBN.

To compare species distributions, spatial data points within the two data sets were first mapped and then buffered within ArcGISpro software (Version10.2.1.). This buffer was set to a 2.5km radius, creating a larger area around each point to account for variations within the spatial accuracy of reports. These areas were then rasterized, a process which converts the raw data into a matrix of cells, each holding a specific value to represent presence or absence of data. By converting the data type in this way, it enabled comparisons to be made using a 5km^2 grid cell layer, identifying cells that contained no reports, reports from a single dataset, or reports from both datasets. Coverage overlap was calculated as the percentage of Road Lab cells that coincided with NBN data, identifying how much of a species predicted range was unique to Road Lab.

3.3 Results

National Biodiversity Network datasets contained more species records overall, but represents submissions from dozens of organisations and charities (Table 3.1). Despite this, 'Data source rank' indicates roadkill data from The Road Lab, ranks highly on a one-to-one comparison to each organisation that is providing data to NBN. Of the 33 species monitored, the Road Lab is in the top 10 reporting organisations for 22 species (66%). Number of reports within NBN was typically larger for avian species, due to consistent and large-scale annual surveys performed by Royal Society for the Protection of Birds (RSPB) and the British Ornithology trust (BTO). However, The Road Lab still ranks highly for several bird species including barn owls (*Tyto alba)* where it is the 5th largest contributor of data. Mammals typically had lower total data counts from NBN, combined with fewer reporting organisations. Chinese water deer (*Hydropotes inermis*) only had 7 additional datasets to WVC Road Lab data. However, the ratio of reported roadkill against other NBN data was low (1:26), compared to other species like badger (*Meles meles*) which have a Road Lab to NBN ratio of 1 WVC record to 3.5 despite 102 reporting organisations. As such, the relative impact of WVC records differs from species to species.

Table 3.1. List of species included within data analysis, with number of datapoints contained within the Road Lab database, data source rank of Road Lab data within NBN (based on total counts of individuals between 2013-2023), total number of NBN data points from data providers and a total count of data suppliers to NBN for the species over this period.

Species	**Scientific name**	**Road Lab data points**	**Data source rank**	**NBN Data points**	**Total datasets**
Badger	*Meles meles*	13,163	1st	45,262	102
Barn owl	*Tyto alba*	658	5th	85,577	121
Brown hare	*Lepus europaeus*	846	7th	39,906	117
Chinese water deer	*Hydropotes inermis*	71	5th	1,886	8

Common frog	*Rana temporaria*	269	12th	76,094	138
Common toad	*Bufo bufo*	337	8th	28,704	122
Fallow deer	*Dama dama*	322	5th	6,685	65
Fox	*Vulpes vulpes*	9,276	3rd	77,999	122
Grass snake	*Natrix natrix*	69	11th	8,545	84
Grey Squirrel	*Sciurus carolinensis*	4,382	6th	241,358	118
Hedgehog	*Erinaceus europaeus*	7,492	5th	204,785	115
Herring gull	*Larus argentatus*	478	7th	481,445	145
House sparrow	*Passer domesticus*	107	22nd	429,305	143
Kestrel	*Falco tinnunculus*	122	22nd	331,645	148
Lesser black backed gull	*Larus fuscus*	139	19th	411,287	130
Mink	*Neovison vison*	112	8th	4,058	102
Mountain Hare	*Lepus timidus*	83	7th	5,090	46
Muntjac Deer	*Muntiacus reevesi*	1,174	5th	18,439	54
Otter	*Lutra lutra*	1,828	2nd	17,871	130
Pheasants	*Phasianus colchicus*	10,982	4th	496,953	140
Pine Marten	*Martes martes*	73	7th	4,113	55
Polecat / polecat-Ferret	*Mustela putorius / furo*	985	1st	1,676	80
Rabbit	*Oryctolagus cuniculus*	6,633	5th	94,405	121
Red Deer	*Cervus elaphus*	145	9th	5,744	58
Red kite	*Milvus milvus*	83	17th	127,450	92
Red legged partridge	*Alectoris rufa*	194	10th	87,342	96
Red squirrel	*Sciurus vulgaris*	212	10th	66,879	81
Roe-Deer	*Capreolus capreolus*	1,227	6th	44,201	98
Slow-worm	*Anguis fragilis*	61	10th	19,153	90
Sparrowhawk	*Accipiter nisus*	78	20th	321,201	136
Stoat	*Mustela erminea*	261	5th	6,825	113
Tawny Owl	*Strix aluco*	396	11th	111,501	128
Weasel	*Mustela nivalis*	75	7th	2,935	89

3.3.2 Species distributions

The species with the largest distribution within the Road Lab database was the badger (*Meles meles*), including 3,608 5km^2 grids (Table 3.2). This is expected as badgers were also the most reported species within the dataset. However, number of records did not necessarily predict range as species may be reported multiple times in the same grid squares. Mountain hare (*Lepus timidus*) had the smallest identified range from Road Lab data, but had more occurrence records than 7 other species which were analysed. High numbers of reports in a small region may be impacted by dedicated reporters in regions with rare or less common species are present. Of all mountain hare reports, 48% were provided by a single recorder in a single region.

The mean overlap in predicted ranges across all species was 95.7%, while coverage overlap of the databases for badgers (*Meles meles*) was 98.61%, demonstrating a strong correlation between the two datasets and predicted ranges for the species. Of the unique distribution grids for badgers in The Road Lab data, 94% were contiguous to NBN range and just three 5km^2 grids were not directly bordering the NBN data. The lowest overlap in distributions occurred for weasel (*Mustela nivalis*) 80.58%, mink (*Neovison vison*) 78.26% and polecat (*Mustela putorius*) 75.43%. These are species that have some of the smallest datasets from NBN, but that Road Lab ranks highly in comparisons to other data set sizes. The small amount of data present for these species increases the likelihood of datasets containing different ranges, as overall distributions are limited. However, with more than 75% overlap these distributions can still be considered similar.

Table 3.2 Species range estimations and coverage overlap, identifying how much of the Road Lab data fell within the wider NBN predicted range for each species.

Species	Scientific name	NBN range (5km Grids)	Road Lab range (5km grids)	Coverage overlap (%)
Badger	*Meles meles*	9,008	3,608	98.61
Barn owl	*Tyto alba*	9,744	459	99.56
Brown hare	*Lepus europaeus*	8,439	513	97.86
Chinese water deer	*Hydropotes inermis*	1,367	28	96.43
Common frog	*Rana temporaria*	9,645	83	98.80
Common toad	*Bufo bufo*	8,333	108	93.52
Fallow deer	*Dama dama*	3,401	133	86.47
Fox	*Vulpes vulpes*	9,459	2,644	99.21
Grass snake	*Natrix natrix*	4,255	60	96.67
Grey Squirrel	*Sciurus carolinensis*	8,282	1,655	99.82
Hedgehog	*Erinaceus europaeus*	9,949	2,531	100.00
Herring gull	*Larus argentatus*	10,491	207	100.00
House sparrow	*Passer domesticus*	10,595	69	100.00
Kestrel	*Falco tinnunculus*	10,768	108	100.00
Lesser black backed gull	*Larus fuscus*	10,494	72	100.00
Mink	*Neovison vison*	5,569	92	78.26
Mountain Hare	*Lepus timidus*	1,941	24	87.50
Muntjac Deer	*Muntiacus reevesi*	3,484	507	97.24
Otter	*Lutra lutra*	9,456	1,290	95.81
Pheasants	*Phasianus colchicus*	10,291	2,379	100.00
PineMarten	*Martes martes*	2,920	61	90.16
Polecat / polecat-Ferret	*Mustela putorius / furo*	3,532	586	75.43
Rabbit	*Oryctolagus cuniculus*	9,935	2,287	99.69
Red Deer	*Cervus elaphus*	4,695	104	93.27
Red kite	*Milvus milvus*	8,284	295	100.00
Red legged partridge	*Alectoris rufa*	8,010	680	99.56
Red squirrel	*Sciurus vulgaris*	3,996	763	98.03
Roe-Deer	*Capreolus capreolus*	8,055	2,964	99.12
Slow-worm	*Anguis fragilis*	5,125	336	97.92

Sparrowhawk	*Accipiter nisus*	10,773	560	100.00
Stoat	*Mustela erminea*	7,932	1,329	98.65
Tawny Owl	*Strix aluco*	9,334	2,088	100.00
Weasel	*Mustela nivalis*	6,290	520	80.58

The median NBN species distribution (Number of 5km2 grids) was 16.4 times larger than the Road Lab data. However, despite the much larger overall coverage of NBN data, 24 of the 33 species had unique locations identified by the WVC data that were not present within other NBN survey datasets (Figure 3.2). In total 541 WVCs were reported outside of the NBN predicted ranges, these data points represent novel areas where the species is present but was not reported by other organisations. Excluding polecat (*Mustela putorius*) data, 87% of novel areas were contiguous with NBN ranges, either filling in small gaps in existing distributions or slightly expanding known ranges. Polecat (*Mustela putorius*) was the only species to show substantial variation, with only 37% being contiguous and 63% of novel areas being discontinuous from NBN ranges. The majority of NBN data for Polecat could be found near and around National Parks and large green areas, while Road Lab data included records from more urbanised areas such as Cardiff, Exeter and Leeds (Figure 3.3).

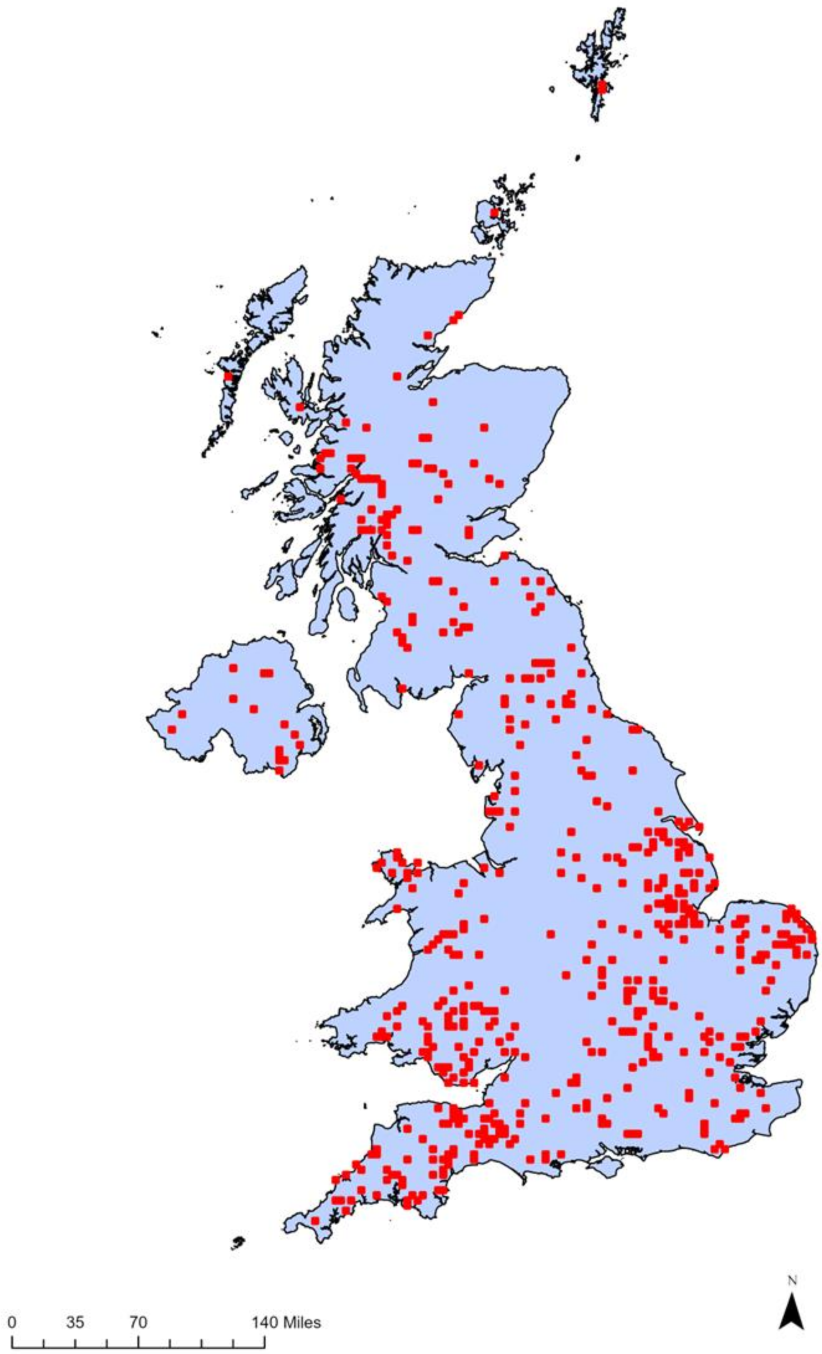
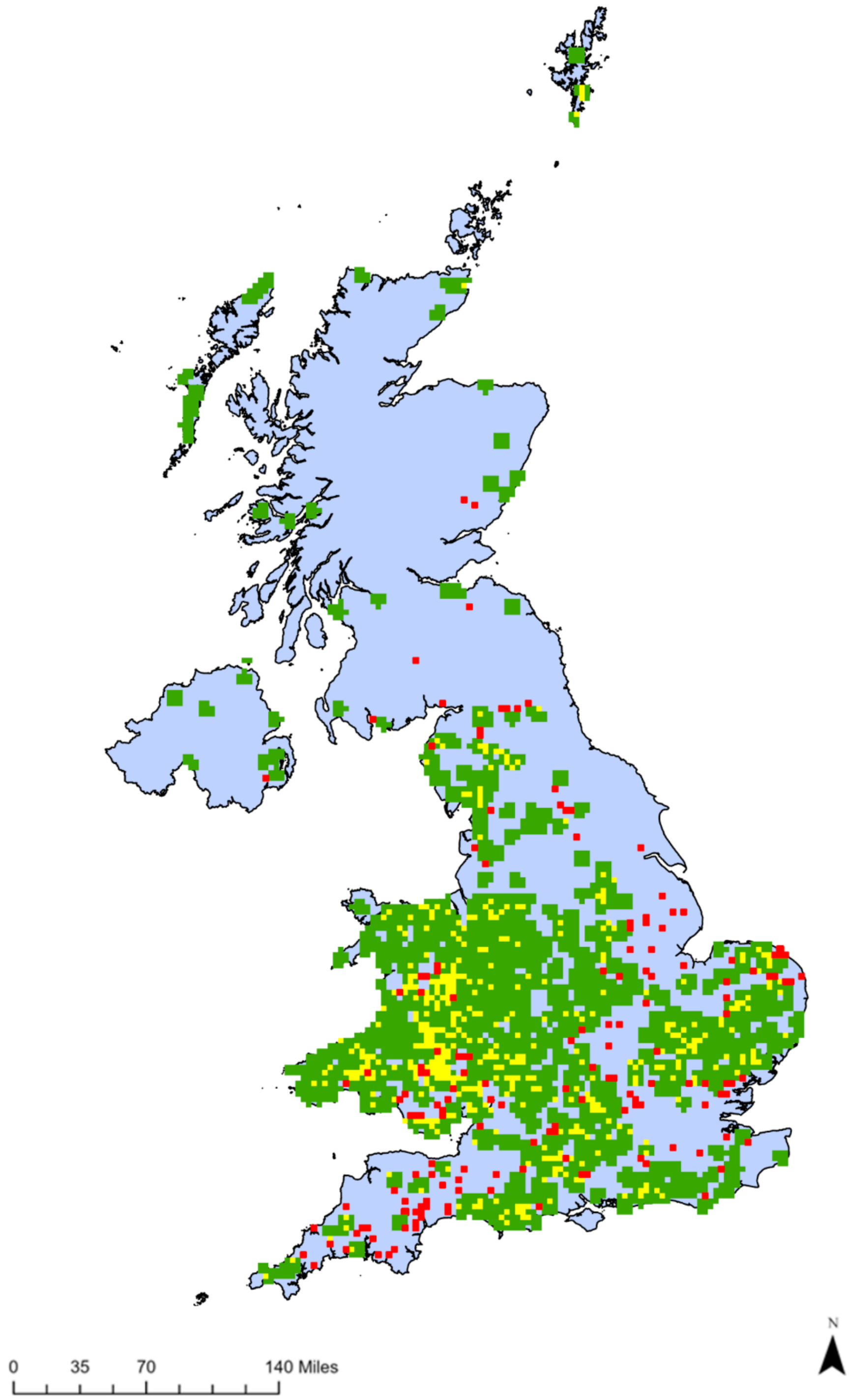


Figure 3.2 Distribution of unique Road Lab data points for 33 target species across the UK. Red squares indicate 5km^2 grid squares where Road Lab data was present for a species, but there was no corresponding data for the same species in the NBN database.

Figure 3.3



Distribution of NBN and Road Lab data points for polecat across the UK. Red squares indicate 5km^2 grid squares where Road Lab data were present for a species, but there was no corresponding data for the same species in the NBN database. Green areas represent 5km^2 grids where data is present within NBN database but not Road Lab. Yellow areas represent 5km^2 grids where both NBN and Road Lab data was present.

3.4 Discussion

When comparing data on WVCs to other NBN datasets, it is apparent that most species are more readily recorded through other survey methods and organisations. However, despite significantly larger databases for all species investigated, WVC data still included 541 spatial datapoints not present within other NBN datasets. The presence of novel distribution data when compared to datasets twenty times the size, is a clear indication that even with a small number of ad-hoc recorders WVC data can help supplement more standardised species survey methods. This is likely a result of the pseudo-random nature of WVC survey effort, as citizen scientists are unlikely to perform dedicated roadkill surveys. Combined with a broad remit of collecting WVC data on all wildlife, it increases the chances of detecting species in new and potentially unexpected areas. Wildlife surveys are rarely performed in locations which are not predetermined to be the most likely to record species presence, due to effort consolidation within data collection methods. However, even within these locations species can be missed, as potentially shown by the high level of contiguous locations between the Road Lab and NBN ranges.

Less than 1% of WVC reports were outside of the predicted range of NBN data. The substantial overlap of datasets is an indication of the validity of the data collection methods involved with citizen science WVC surveying. Additionally, the Road Lab species database was the largest reporter of badger (*Meles meles*) and polecat data (*Mustela putorius*), while being the second largest reporter for otters (*Lutra lutra*), when compared individually to all other data providers. This could be due to several factors, polecats (*Mustela putorius*) and otters (*Lutra lutra*) are illusive species with many standard survey observations based on field signs or breeding dens, which reduces the total count of direct observations. Badgers (*Meles meles*) are a species that are highly protected, but do not appear to have many dedicated monitoring schemes reporting field survey observations to NBN. This is likely true for other species that Road Lab ranks highly in, such as foxes (*vulpes vulpes*) and rabbits (*Oryctolagus cuniculus*). Despite evidence of its impacting on WVC reporting (Barrientos *et al.,* 2018), body size does not appear to influence reporting to The Road Lab, with the most reported species ranging from hedgehogs (*Erinaceus europaeus*) to badgers (*Meles meles*) to deer. Equally, no clear ecological factors such as time of activity are obvious in the 33 species subset. However, it has been shown that temporal trends exist within WVC events and reporting for many species (Raymond *et al.,* 2021).

Bias may exist within the data as WVC reporters could be impacted by various factors, for example size and ease of noticing roadkill while travelling in a car. The reporting methodology (incidental observation reported using a smartphone or social media) may also restrict data collection in some cases, for example if driving alone, or unable to stop and record sightings. On many roads, stopping safely is not possible which further reduces the chances of reporting or clarifying observations. Bias may also be introduced based on what species each individual feels motivated to report, with novel or 'interesting' species having the potential to be reported more than common species (Egerer *et al.,* 2019), although other studies suggest common species are overrepresented in citizen science (Callaghan *et al.,* 2021). Overall, detectability of species is likely to be influenced by several factors including species, size, scavenger activity, time of day and the personal interests of the surveyor. However, the variety of reporters submitting to the Road Lab may mitigate some of these biases.

Surveys for nocturnal species within NBN are more likely to have occurred during optimum times, compared to a reduced number of drivers and reporters to The Road Lab overnight. Some trends do exist within taxonomic groups, with birds reported more than mammals to NBN, while number of Road Lab reports are similar when compared across all species. Amphibians and Reptiles are seen in small numbers within the Road Lab database, but due to designated organisations like the Amphibians and Reptiles Group (ARG) reports are higher in NBN. It is possible that the level of protections or conservation interest for a species could be having an impact on NBN data collection. While the interest and fondness of a species in the general public may be more likely to impact number of reports to the Road Lab (Austen *et al*., 2016). Work to potentially identify this bias could be useful as an indicator of the wider value of WVC reporting for under-surveyed species not typically deemed to be of conservation concern.

Outside of major charities and organizations such as the Royal Society for the Protection of Birds, and British Trust for Ornithology, Road Lab data was regularly the largest supplier of ecological data for the 33 species analysed. This is despite the small nature of the organisation and reliance on citizen science. It is clear that there is a large potential for this type of data collection, as evidenced by the high data ranking of Road Lab data within NBN, as well as the development of other citizen science applications. A potential limitation of this analysis and citizen science WVC data collection in general, is the potential for misidentification (Farr *et al.,* 2022; Austen *et al*., 2016). Without receiving image verification, the chances of a species being reported incorrectly may be high (Gorleri and Areta, 2021). However, the project overcomes this by enabling reporters to use varying levels of certainty. By using categories such as 'unidentified' or 'bird of prey' within the data recording app, it overcomes many of the issues faced within other databases by creating a scale of certainty and reduces the likeness of forced misidentification. As such, species level identification is considered reliable in this case. Previous studies have shown a high level of competency and accuracy within nature-based citizen science projects (Lewandowski and Specht, 2015), supported by direct comparisons against regular trained observers where both groups identified the same trends in species WVC presence and abundance (Guinard *et al.*, 2023; Périquet *et al.,* 2018). These similarities in findings indicate the use of citizen science to identify species for WVC and other ecological monitoring projects can be considered accurate and suitable for analysis (Valerio *et al.,* 2021; Collinson *et al.*, 2018). Shin *et al.*, (2022) also proposed that citizen science overcomes the potential bias of planned surveys and can aid in finding novel distribution data, although there is some evidence to suggest coverage can be limited in some cases (Prenda *et al.,* 2024). To establish the level of over or under reporting of species within the Road Lab data, future research should be conducted with dedicated surveyors. This would enable comparisons to be made between data submitted by citizen scientists and trained surveyors, establishing a baseline frequency of reports for all species.One potential limitation in this analysis is the combination of multiple survey methods used within NBN data collection. To fully understand the relative usefulness of WVC data collection, a standardised survey could be completed to compare WVC reports to species monitoring techniques over the same time period. Additionally, the current extent of WVCs and proportion reported to the Road Lab is uncertain. A further study quantifying the ratio of true WVC events and relative reports through citizen science to the Road Lab, would help increase the accuracy of these findings.

3.5 Conclusion

The 99% overlap of Road Lab WVC and NBN survey data demonstrates that citizen science collated WVC reports, can be considered as valid as other data sources. Additionally, the presence of novel spatial data within the Road Lab database, indicates the potential for WVC data to supplement more standardized ecological survey methods, and help identify species ranges and distributions beyond what might be expected. The potential for WVC data collection is further increased by the cost effective and low expertise needed to perform such surveys. As such, WVC surveys should be considered applicable for use in future projects and species monitoring as a supplementary technique.

Chapter 4: project conclusions

The research and analysis performed during this study were completed to answer two main questions. What are the potential benefits to habitat connectivity of greening grey infrastructure, and are wildlife-vehicle collisions a useful data source for species presence knowledge.

4.1 Benefits of greening infrastructure.

To answer the first question, connectivity was successfully quantified and identified a median 8% increase across 25km2 square study sites. This increase in connectivity indicates that greening existing infrastructure could provide a cost-effective way to enhance current habitat networks throughout the UK. Studies attempting to estimate connectivity loss during urban expansion have indicated connectivity can be lowered by as much as 14% during development for some species (Nelli *et al.,* 2022). If greening structures can compensate for over half of this loss, it would be a significant benefit to nature. Single mitigation structures have previously been demonstrated to reduce the barrier effect of roads (Soanes *et al.,* 2024) which restricts movement and lowers local connectivity (Ree *et al.,* 2007). However, few studies look at landscape level connectivity in relation to purpose built crossing structures (Kor *et al.,* 2022). As such, there is a fundamental gap in our understanding of how these structures impact movement beyond the single location at which they are implemented (Harker *et al.,* 2021). There is evidence that mitigation planning on well used migration routes is successful at reducing the impact of new roads and reducing mortality (Glista *et al.,* 2009; Taylor and Goldingay, 2009). If these benefits and reduction in wildlife mortality can be achieved in addition to the indicated 8% connectivity increase, the impact of greening existing infrastructure to benefit nature could be substantial if carried out across the UK.

4.2 Limitations and future research on structure greening.

The exact modifications and designs of grey to green structures will need to be species dependent and habitat dependent (Cork *at al.,* 2024; Neumann *et al.,* 2012; Taylor and Goldingay, 2009; Yanes, Velasco and Suarez, 1995). Cain *et al.,* (2003) demonstrated that road crossing points were not randomly selected by bobcats (*Lynx rufus*) and shared common environmental variables, while Eldridge and Wynn (2011) found that vegetation type and cover impacted the frequency of underpass usage by badgers. These studies suggest that species specific modifications are possible, although highlight there may be a trade off between which species benefit most. Alterations which benefit one species, may not benefit, or could reduce structure utilisation by other species. It is vital that future research assesses this gap in knowledge before practical implementation occurs.

Some assumptions of the proposed effectiveness of structure greening have been made in this project due to the lack of literature focused on this specific topic. We suggest that in-depth field monitoring of wildlife crossing rates on grey structures should be carried out. Identifying the current baseline crossing rates would help provide the necessary context to the theoretical modelling performed in this study. Monitoring studies of mitigation structures typically use a combination of camera trapping techniques and footprint tunnels (Pomezanski and Bennett, 2018; Jumeau *et al.,* 2024). Species movement patterns change throughout the year depending on behaviours and sexual dispersal, these should be considered when choosing when to conduct surveys (Hamilton *et al.,* 2024). Alternately and depending on resources, GPS tracking of species could help generate a more thorough picture of how individuals move through the study areas and may even demonstrate links between different structures (Jin *et al.,* 2022). Combined, these monitoring techniques would overcome the limitations of this study by providing the real-world context needed to validate the connectivity benefits indicated.

4.3 Suitability of wildlife-vehicle collision monitoring for ecological data collection.

In the current study, wildlife-vehicle collisions were investigated for their validity and use within broader ecological data collection. When compared to data collected using more standard surveying techniques, there was a 99% overlap in species distribution data. This is a clear indication that the two datasets were spatially similar, and the data collected through WVC monitoring can be considered comparative. The relatively high abundance of data provided per species by WVC methods compared to other NBN data providers, also showed its importance as a survey method. For some species like badgers, polecat and otters, WVC reports were the largest or second largest contributor of data to NBN. This suggests a wider adoption of WVC surveying would increase our understanding of species distributions and help fill in knowledge gaps that currently exist.

Beyond their use in species range predictions, WVC data can also be used to inform conservation actions. Hotspot analysis can identify areas with the highest rate of reported collisions, highlighting regions where mitigation may have the biggest impact on reducing mortality rates (Kim *et al.,* 2023; Balciauskas *et al.,* 2020). In the context of the research carried out in this project, hotspots may also be helpful to identify priority areas for structure greening. By comparing hotspot presence within study sites, it is possible to determine the locations of grey infrastructure which could have the most benefit if they are greened (figure 4.3.1). However, it is important to consider that hotspots may not be truly representative due to ad hoc nature of the WVC data collection. Instead, they may identify regions with dedicated reporters or high reporter/population density. A single reporter who submits their sightings each week will likely appear as a hotspot when compared to regions with less reporters. This does not necessarily mean the rate of WVC’s is substantially higher in that region, and future research establishing baseline reporting with standardised surveys should be completed.

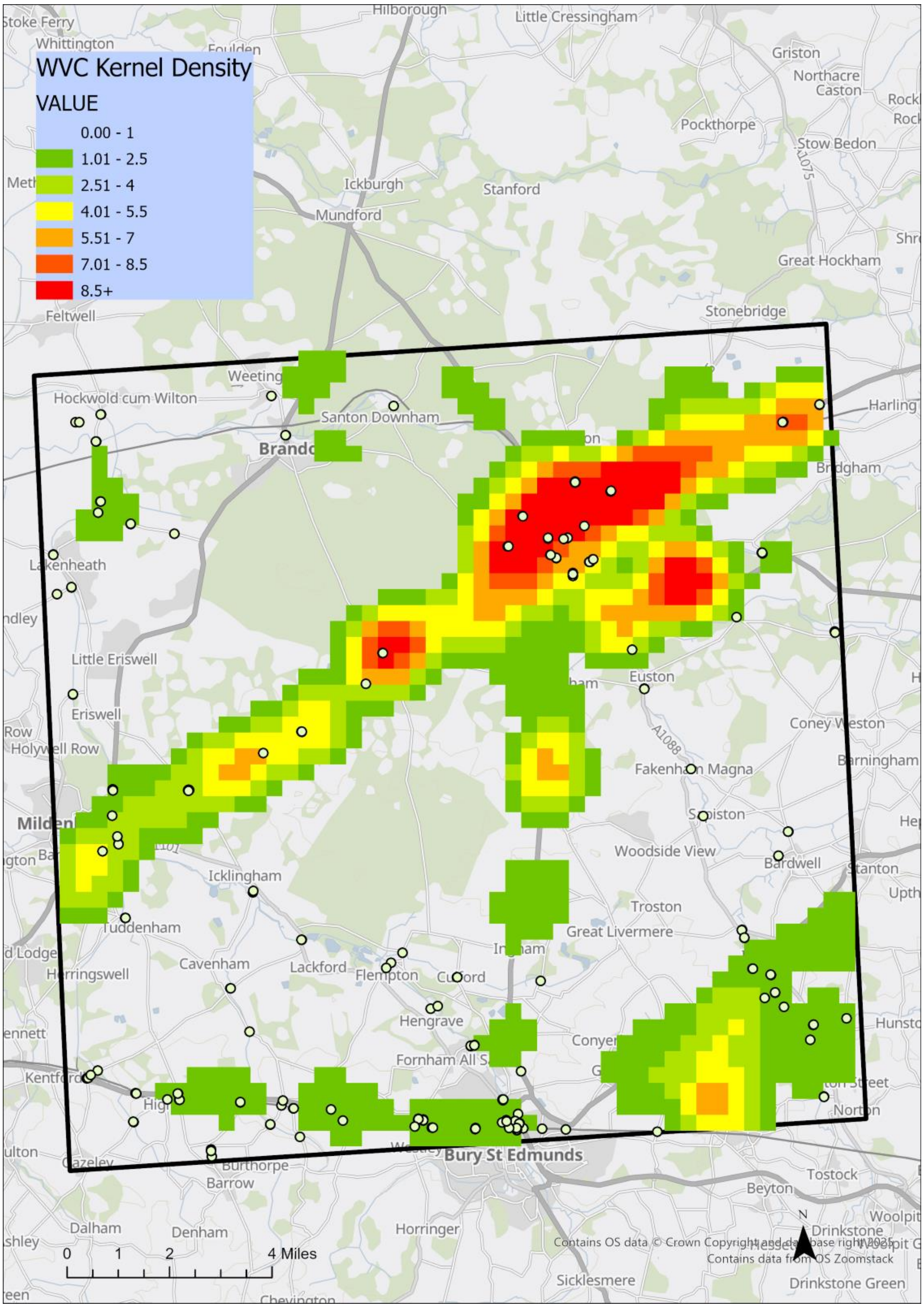


Figure 4.3.1 Hotspot map (Kernel density estimation) of WVC records within study site 41, identifying overlapping grey infrastructure locations (white dots). Those structures present within red areas could be considered a priority as their location overlaps with areas of high wildlife mortality.

The lack of standardisation within citizen science WVC data collection can generate resistance to its utilisation due to its sporadic nature and lack of assurance with data quality, but as a whole it has been proven to be an extremely useful tool (Vercayie and Herremans 2015). While reviewing citizen science data against other collection methods including standard ecological transect surveys, Aceves-Bueno *et al.,* (2017) found that 72% of the correlations identified in data collected through citizen science, were also present within data collected by professional researchers. However, this meta-analysis included a variety of methods and training types for the citizen scientists. The researchers also cautioned that the needs of the research should be considered when using citizen scientists. Complex methodologies with high barriers of entry for knowledge may not be suitable, with several studies highlighting potential limitations (Hochachka *et al.* 2012, Vermeiren *et al.* 2016). These studies also indicate a need for more reviews of the accuracy of citizen science data sets by using direct comparisons to data collected by experts and professionals in the field.

In the case of our study, the data collated by Road Lab are presence only, with the only necessary skill being that of species identification. The added difficulty of identification caused by speed of vehicle, lighting and other distractions is likely to be the same for all surveys, trained or otherwise. Therefore, it relies more on baseline species identification skills which have in other roadkill studies been indicated to be over 90% accurate even in untrained individuals (Vercayie and Herremans 2015). This high level of accuracy even in untrained individuals, may be due to the type of person interested in taking part in these research projects already having an inherent interest in nature (Prenda *et al.,* 2024). Overall, citizen science offers a methodology to collect data which can occur even in the absence of resources and funding typically required to carry out expansive surveys. In many cases, the alternate is to not carry out the research or to significantly reduce data collection.

4.4 Limitations and solutions to the use of citizen science for Road Lab data collection

Literature broadly supports the idea that citizen science is accurate and a reliable tool for monitoring wildlife-vehicle collisions (Collinson *et al.,* 2018), although many studies are performed with dedicated teams of surveyors and not a truly random selection of citizens as with the Road Lab (Guinard *et al.,* 2023; Valerio *et al.,* 2021; Petrovan *et al.,* 2020).The Road Lab does not control or influence who can report to the programme, potentially increasing the chances of surveyor errors. To fully validate the data collected by Road Lab and used in this study, it would be necessary to carry out assessments using the same surveyors who have been actively reporting to the project. Future research using images verified by experts could be shown to surveyors who are asked how they would categorise them. This would evaluate and quantify the likeliness of correct identification within the existing Road Lab dataset and therefore the outcomes of this project. While spatial trends have been assessed in this study, the accuracy of temporal or overall WVC rates has not been investigated. Future work using dedicated surveyors at regular locations and times, would allow estimates to be made for WVC detection rate of ad-hoc citizen science. By completing these surveys, more accurate estimates could be made for total roadkill and their impact on nature within the UK.

4.5 Outcomes and future work.

While the work carried out during this project indicates the potential for infrastructure greening and the value of wildlife-vehicle collision monitoring, more research is required. To aid with this, a clear change in the transparency of mitigation structure monitoring is needed at a regulatory level. Making data available on the location, species and crossing rates before and after mitigation implementation, would provide the much-needed context for this investigation and future research. The results of the research performed here, also suggests that wildlife-vehicle collision monitoring should be integrated into large scale species monitoring efforts carried out by charities and conservation organisations. WVC's are indicated by our data to provide novel spatial data while maintaining a high level of reliability, although more work should be completed to determine detection rates and possible total WVC events across the UK. Both structure greening and the use of WVC monitoring have clear potential to benefit nature in the UK.

5.0 References

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

Adriaensen, F., Chardon, J.P., De Blust, G., Swinnen, E., Villalba, S., Gulinck, H. and Matthysen, E. 2003. The application of 'least-cost' modelling as a functional landscape model. *Landscape and Urban Planning*. 64(4), pp.233-247. https://doi.org/10.1016/S0169-2046(02)00242-6.

Anantharaman, R., Hall, K., Shah, V.B., Edelman, A., 2020. Circuitscape in Julia: High Performance. Connectivity Modelling to Support Conservation Decisions. *Proceedings of the JuliaCon Conferences*, 1(58). https://doi.org/10.21105/jcon.00058

App, M., Strohbach, M.W., Schneider, A.-K., Schröder, B., 2022. Making the case for gardens: Estimating the contribution of urban gardens to habitat provision and connectivity based on hedgehogs (Erinaceus europaeus). *Landscape and Urban Planning.* 220, 104347. https://doi.org/10.1016/j.landurbplan.2021.104347

Augustynczik, A.L.D., 2021. Habitat amount and connectivity in forest planning models: Consequences for profitability and compensation schemes. *Journal of Environmental Management.* 283, 111982. https://doi.org/10.1016/j.jenvman.2021.111982

Austen, G.E., Bindemann, M., Griffiths, R.A. and Roberts, D.L. 2016. Species identification by experts and non-experts: comparing images from field guides. *Scientific reports*. 6, 33634. https://doi.org/10.1038/srep33634

Baker, E., Drury, J.P., Judge, J., Roy, D.B., Smith, G.C., Stephens, P.A., 2021. The Verification of Ecological Citizen Science Data: Current Approaches and Future Possibilities. Citizen Science: Theory and Practice 6. https://doi.org/10.5334/cstp.351

Balčiauskas, L., Stratford, J., Balčiauskienė, L., Kučas, A., 2020. Importance of professional roadkill data in assessing diversity of mammal roadkills. Transportation Research Part D: Transport and Environment 87, 102493. https://doi.org/10.1016/j.trd.2020.102493

Beier, P., Majka, D., Spencer, W., 2008. Forks in the Road: Choices in Procedures for Designing Wildland Linkages. *Conservation biology*. 22, pp.836–51. https://doi.org/10.1111/j.1523-1739.2008.00942.x

Benítez-López, A., Alkemade, R. and Verweij, P.A. 2010. The impacts of roads and other infrastructure on mammal and bird populations: A meta-analysis. *Biological Conservation*. 143 (6), pp.1307-1316. DOI:10.1016/j.biocon.2010.02.009.

Bennett, V.J., 2017. Effects of Road Density and Pattern on the Conservation of Species and Biodiversity. *Current Landscape Ecology*. 2, pp.1–11. https://doi.org/10.1007/s40823-017-0020-6

Bergès, L., Avon, C., Bezombes, L., Clauzel, C., Duflot, R., Foltête, J.-C., Gaucherand, S., Girardet, X., Spiegelberger, T., 2020. Environmental mitigation hierarchy and biodiversity offsets revisited through habitat connectivity modelling. Journal of Environmental Management 256, 109950. https://doi.org/10.1016/j.jenvman.2019.109950

Bhardwaj, M., Olsson, M., Seiler, A., 2020. Ungulate use of non-wildlife underpasses. *Journal of Environmental Management*. 273, 111095. https://doi.org/10.1016/j.jenvman.2020.111095

Bond, A.R.F., Jones, D.N., 2013. Wildlife Warning Signs: Public Assessment of Components, Placement and Designs to Optimise Driver Response. *Animals (Basel).* 3, pp.1142–1161. https://doi.org/10.3390/ani3041142

Boyle, S.P., Keevil, M.G., Litzgus, J.D., Tyerman, D., Lesbarrères, D. 2021. Road-effect mitigation promotes connectivity and reduces mortality at the population-level. *Biological Conservation*. 261, 109230. https://doi.org/10.1016/j.biocon.2021.109230

Braaker, S., Moretti, M., Boesch, R., Ghazoul, J., Obrist, K. and Bontadina, F. 2014. Assessing habitat connectivity for ground-dwelling animals in an urban environment. *Ecological applications*. 24(7), pp.1561-1877. https://doi.org/10.1890/13-1088.1

Braaker, S., Kormann, U., Bontadina, F. and Obrist, M.K. 2017. Prediction of genetic connectivity in urban ecosystems by combining detailed movement data, genetic data and multi-path modelling. *Landscape and Urban Planning*. 160, pp.107–114. https://doi.org/10.1016/j.landurbplan.2016.12.011

Breckheimer, I., Haddad, N.M., Morris, W.F., Trainor, A.M., Fields, W.R., Jobe, R.T., Hudgens, B.R., Moody, A. and Walters, J.R. 2014. Defining and Evaluating the Umbrella Species Concept for Conserving and Restoring Landscape Connectivity. *Conservation Biology*. 28, pp.1584-1593. https://doi.org/10.1111/cobi.12362

Brennan, L., Chow, E. and Lamb, C. 2022. Wildlife overpass structure size, distribution, effectiveness, and adherence to expert design recommendations. *PeerJ*. 10, e14371. https://doi.org/10.7717/peerj.14371

Brockie, R., Sadleir, R.M. and Linklater, W.L. 2009. Long-term wildlife road-kill counts in New Zealand. *New Zealand Journal of Zoology*. 36, pp.123-134. DOI: 0301-4223/09/3602-0123.

Brotons, L., Thuiller, W., Araújo, M.B. and Hirzel, A.H. 2004. Presence-absence versus presence-only modelling methods for predicting bird habitat suitability. *Ecography.* 27, pp.437–448. https://doi.org/10.1111/j.0906-7590.2004.03764.x

Bull, J.W., Brauneder, K., Darbi, M., Van Teeffelen, A.J.A., Quétier, F., Brooks, S.E., Dunnett, S., Strange, N., 2018. Data transparency regarding the implementation of European 'no net loss' biodiversity policies. Biological Conservation 218, 64–72. https://doi.org/10.1016/j.biocon.2017.12.002

Burdett, C.L., Crooks, K.R., Theobald, D.M., Wilson, K.R., Boydston, E.E., Lyren, L.M., Fisher, R.N., Vickers, T.W., Morrison, S.A. and Boyce, W.M. 2010. Interfacing models of wildlife habitat and human development to predict the future distribution of puma habitat. *Ecosphere.* 1(4). https://doi.org/10.1890/ES10-00005.1

Burgess, H.K., DeBey, L.B., Froehlich, H.E., Schmidt, N., Theobald, E.J., Ettinger, A.K., HilleRisLambers, J., Tewksbury, J., Parrish, J.K., 2017. The science of citizen science: Exploring barriers to use as a primary research tool. Biological Conservation, The role of citizen science in biological conservation 208, 113–120. https://doi.org/10.1016/j.biocon.2016.05.014

Caley, P., Ramsey, D.S.L. and Barry, S.C., 2015. Inferring the Distribution and Demography of an Invasive Species from Sighting Data: The Red Fox Incursion into Tasmania. *PLOS ONE.* 10, e0116631. https://doi.org/10.1371/journal.pone.0116631

Callaghan, C.T., Poore, A.G.B., Hofmann, M., Roberts, C.J., Pereira, H.M., 2021. Large-bodied birds are over-represented in unstructured citizen science data. Sci Rep 11, 19073. https://doi.org/10.1038/s41598-021-98584-7

Carter, N.H., Pradhan, N., Hengaju, K., Sonawane, C., Sage, A.H. and Grimm, V. 2022. Forecasting effects of transport infrastructure on endangered tigers: a tool for conservation planning. *PeerJ.* Doi: 10.7717/peerj.13472

Carvajal, M.A., Alaniz, A.J., Smith-Ramírez, C. and Sieving, K.E. 2018. Assessing habitat loss and fragmentation and their effects on population viability of forest specialist birds: Linking biogeographical and population approaches. *Diversity and Distributions*. 24, pp. 820–830. https://doi.org/10.1111/ddi.12730

Canova, L. and Balestrieri, A. 2019. Long-term monitoring by roadkill counts of mammal populations living in intensively cultivated landscapes. *Biodiversity Conservation*. 28, pp.97–113. https://doi.org/10.1007/s10531-018-1638-3

Carroll, K.A., Hansen, A.J., Inman, R.M., Lawrence, R.L. and Hoegh, A.B. 2020. Testing landscape resistance layers and modeling connectivity for wolverines in the western United States. *Global Ecology and Conservation.* 23, e01125. https://doi.org/10.1016/j.gecco.2020.e01125

Ceballos, G. and Ehrlich, P. 2002. Mammal Population Losses and the Extinction Crisis. *Science*. 296, pp.904-907. DOI:10.1126/science.1069349.

Chapman, C. and Hall, J.W. 2022. Designing green infrastructure and sustainable drainage systems in urban development to achieve multiple ecosystem benefits. *Sustainable Cities and Society.* 85, 104078. https://doi.org/10.1016/j.scs.2022.104078

Charney, N.D. 2012. Evaluating expert opinion and spatial scale in an amphibian model. *Ecological Modelling.* 242, pp.37–45. https://doi.org/10.1016/j.ecolmodel.2012.05.026

Cooke, S.C., Balmford, A., Donald, P.F., Newson, S.E., Johnston, A., 2020. Roads as a contributor to landscape-scale variation in bird communities. Nat Commun 11, 3125. https://doi.org/10.1038/s41467-020-16899-x

Cork, N.A., Fisher, R.S., Strong, N., Ferranti, E.J.S., Quinn, A.D., 2024. A systematic review of factors influencing habitat connectivity and biodiversity along road and rail routes in temperate zones. Frontiers in Environmental Science 12. https://doi.org/10.3389/fenvs.2024.1369072

Crooks, K.R. 2002. Relative Sensitivities of Mammalian Carnivores to Habitat Fragmentation. *Conservation Biology*. 16(2), pp. 488-502. DOI: 10.1046/j.1523-1739.2002.00386.x

Cullen Jr, L., Stanton, J.C., Lima, F., Uezu, A., Perilli, M.L.L. and Akçakaya, H.R. 2016. Implications of Fine-Grained Habitat Fragmentation and Road Mortality for Jaguar Conservation in the Atlantic Forest, Brazil. *PLOS ONE.* 11, e0167372. https://doi.org/10.1371/journal.pone.0167372

Cushman, S.A., McRae, B., Adriaensen, F., Beier, P., Shirley, M. and Zeller, K. 2013. Biological corridors and connectivity. *Key Topics in Conservation Biology 2* (eds D.W. Macdonald and K.J. Willis). https://doi.org/10.1002/9781118520178.ch21

Dertien, J.S. and Baldwin, R.F. 2023. Does scale or method matter for conservation? Application of directional and omnidirectional connectivity models in spatial prioritizations. *Frontiers in Conservation Science*. 4. https://doi.org/10.3389/fcosc.2023.976914

Dickson, B.G., Albano, C.M., Anantharaman, R., Beier, P., Fargione, J., Graves, T.A., Gray, M.E., Hall, K.R., Lawler, J.J., Leonard, P.B., Littlefield, C.E., McClure, M.L., Novembre, J., Schloss, C.A., Schumaker, N.H., Shah, V.B. and Theobald, D.M. 2019. Circuit-theory applications to connectivity science and conservation. *Conservation Biology*. 33, pp. 239-249. https://doi.org/10.1111/cobi.13230

Dickson, B. and Beier, P. 2002. Home-Range and Habitat Selection by Adult Cougars in Southern California. *The Journal of Wildlife Management.* 66, 1235. https://doi.org/10.2307/3802956

Didham, R.K. 2010. Ecological Consequences of Habitat Fragmentation, in: *Encyclopedia of Life Sciences*. John Wiley & Sons, Ltd. https://doi.org/10.1002/9780470015902.a0021904

Dri, G.F., Blomberg, E.J., Hunter, M.L., Vashon, J.H. and Mortelliti, A. 2022. Developing cost-effective monitoring protocols for track-surveys: An empirical assessment using a Canada lynx Lynx canadensis dataset spanning 16 years. *Biological Conservation.* 276, 109793. https://doi.org/10.1016/j.biocon.2022.109793

Egerer, M., Lin, B.B., Kendal, D., 2019. Towards better species identification processes between scientists and community participants. Science of The Total Environment 694, 133738. https://doi.org/10.1016/j.scitotenv.2019.133738

Eldridge, B., Wynn, J., 2011. Use of badger tunnels by mammals on Highways Agency schemes in England. Conservation evidence, 8, pp.53-57.

Environment Agency. 2016. Cost estimation for habitat creation summary of evidence. Report SC080039/R14. Available at: https://assets.publishing.service.gov.uk/media/6034ef5ee90e0766033f2ea7/Cost_estimation_for_habitat_creation.pdf

ESRI. 2022. ArcGISPro: Release 10. Redlands, CA: Environmental Systems Research Institute.

Fahrig, L. 2003. Effects of Habitat Fragmentation on Biodiversity. *Annual Review of Ecology, Evolution, and Systematics*. 34, pp.487–515. https://doi.org/10.1146/annurev.ecolsys.34.011802.132419

Farr, C.M., Ngo, F. and Olsen, B., 2023. Evaluating Data Quality and Changes in Species Identification in a Citizen Science Bird Monitoring Project. *Citizen science, theory and practice.* 8(24). https://doi.org/10.5334/cstp.604

Fey, K., Hämäläinen, S., Selonen, V., 2016. Roads are no barrier for dispersing red squirrels in an urban environment. *Behavioral Ecology* 27, pp.741–747. https://doi.org/10.1093/beheco/arv215

Fischer, J. and Lindenmayer, D.B. 2007. Landscape modification and habitat fragmentation: a synthesis. *Global Ecology and Biogeography*. 16, pp. 265-280. https://doi.org/10.1111/j.1466-8238.2007.00287.x

Fulgione, D., Maselli, V., Pavarese, G., Rippa, D. and Rastogi, R.K. 2009. Landscape fragmentation and habitat suitability in endangered Italian hare (Lepus corsicanus) and European hare (Lepus europaeus) populations. *European Journal of Wildlife Research*. 55, pp.385–396. https://doi.org/10.1007/s10344-009-0256-5

George, L., Macpherson, J., Balmforth, Z. and Bright, P. 2011. Using the dead to monitor the living: Can road kill counts detect trends in mammal abundance? *Applied Ecology and Environmental Research*. 9, pp.27–42. https://doi.org/10.15666/aeer/0901_027041

Glista, D.J., DeVault, T.L., DeWoody, J.A., 2009. A review of mitigation measures for reducing wildlife mortality on roadways. Landscape and Urban Planning 91, 1–7. https://doi.org/10.1016/j.landurbplan.2008.11.001

González-Gallina, A., Benítez-Badillo, G., Hidalgo-Mihart, M.G., Equihua, M., Rojas-Soto, O.R., 2016. Roadkills as a complementary information source for biological surveys using rodents as a model. *Journal of Mammalogy*. 97, pp.145–154. https://doi.org/10.1093/jmammal/gyv165

González-Gallina, A., Hidalgo-Mihart, M.G. and Castelazo-Calva, V., 2018. Conservation implications for jaguars and other neotropical mammals using highway underpasses. *PLOS ONE* 13, e0206614. https://doi.org/10.1371/journal.pone.0206614

Gorleri, F.C., Areta, J.I., 2022. Misidentifications in citizen science bias the phenological estimates of two hard-to-identify Elaenia flycatchers. *Ibis*. 164, pp.13–26. https://doi.org/10.1111/ibi.12985

Grafius, D.R., Corstanje, R., Siriwardena, G.M., Plummer, K.E. and Harris, J.A. 2017. A bird’s eye view: using circuit theory to study urban landscape connectivity for birds. *Landscape Ecology*. 32, pp.1771–1787. https://doi.org/10.1007/s10980-017-0548-1

Grilo, C., Bissonette, J.A., Santos-Reis, M., 2008. Response of carnivores to existing highway culverts and underpasses: implications for road planning and mitigation. *Biodiversity Conservation.* 17, pp.1685–1699. https://doi.org/10.1007/s10531-008-9374-8

Guinard, E., Billon, L., Bretaud, J.-F., Chevallier, L., Sordello, R., Witté, I., 2023. Comparing the effectiveness of two roadkill survey methods on roads. Transportation Research Part D: Transport and Environment 121, 103829. https://doi.org/10.1016/j.trd.2023.103829

Guisan, A., Tingley, R., Baumgartner, J.B., Naujokaitis-Lewis, I., Sutcliffe, P.R., Tulloch, A.I.T., Regan, T.J., Brotons, L., McDonald-Madden, E., Mantyka-Pringle, C., Martin, T.G., Rhodes, J.R., Maggini, R., Setterfield, S.A., Elith, J., Schwartz, M.W., Wintle, B.A., Broennimann, O., Austin, M., Ferrier, S., Kearney, M.R., Possingham, H.P., Buckley, Y.M., 2013. Predicting species distributions for conservation decisions. *Ecology Letters.* 16, pp.1424–1435. https://doi.org/10.1111/ele.12189

Guo, L., Liu, J. and Lu, R. 2019. Subsampling Bias and The Best-Discrepancy Systematic Cross Validation. *ArXiv*.1907.02437 https://doi.org/10.48550/arXiv.1907.02437

Hamilton, C.M., Bateman, B.L., Gorzo, J.M., Reid, B., Thogmartin, W.E., Peery, M.Z., Heglund, P.J., Radeloff, V.C. and Pidgeon, A.M. 2018. Slow and steady wins the race? Future climate and land use change leaves the imperilled Blanding's turtle (Emydoidea blandingii) behind. *Biological Conservation*. 222, pp.75-85. https://doi.org/10.1016/j.biocon.2018.03.026.

Hamilton, K.M., Bommarito, T., Lewis, J.S., 2024. Spatial and temporal factors influencing wildlife use of overpass crossing structures and landscape siphons along a major canal. Biological Conservation 292, 110481. https://doi.org/10.1016/j.biocon.2024.110481

Hanks, E.M. and Hooten, M.B. 2013. Circuit Theory and Model-Based Inference for Landscape Connectivity. *Journal of the American Statistical Association*. 108, pp. 22–33. https://doi.org/10.1080/01621459.2012.724647

Harihar, A. and Pandav, B. 2012. Influence of Connectivity, Wild Prey and Disturbance on Occupancy of Tigers in the Human-Dominated Western Terai Arc Landscape. *PLOS ONE*. https://doi.org/10.1371/journal.pone.0040105

Harker, K.J., Arnold, L., Sutherland, I.J., Gergel, S.E., 2021. Perspectives from landscape ecology can improve environmental impact assessment. FACETS 6, 358–378. https://doi.org/10.1139/facets-2020-0049

Harper, L.R., Downie, J.R. and McNeill, D.C. 2019. Assessment of habitat and survey criteria for the great crested newt (Triturus cristatus) in Scotland: a case study on a translocated population. Hydrobiologia 828, pp.57–71. https://doi.org/10.1007/s10750-018-3796-4

Helldin, J.O., 2022. Are several small wildlife crossing structures better than a single large? Arguments from the perspective of large wildlife conservation. Nature Conservation. https://doi.org/10.3897/natureconservation.47.67979

Hill, J.E., DeVault, T.L. and Belant, J.L., 2019. Cause-specific mortality of the world's terrestrial vertebrates. *Global Ecology and Biogeography.* 28, pp.680–689. https://doi.org/10.1111/geb.12881

Hodgson, J.A., Thomas, C.D., Dytham, C., Travis, J.M.J. and Cornell, S.J. 2012. The Speed of Range Shifts in Fragmented Landscapes. *PLOS ONE*. 7, e47141. https://doi.org/10.1371/journal.pone.0047141

Hodson, N.L. and Snow, D.W. 1965. The Road Deaths Enquiry, 1960–61. Bird Study 12, pp.90–99. https://doi.org/10.1080/00063656509476091

Horváth, Z., Ptacnik, R., Vad, C. and Chase, J. 2019. Habitat loss over six decades accelerates regional and local biodiversity loss via changing landscape connectance. *Ecology Letters.* 22, pp.1019–1027. https://doi.org/10.1111/ele.13260

Hunter-Ayad, J. and Hassall, C. 2020. An empirical, cross-taxon evaluation of landscape-scale connectivity. *Biodiversity Conservation*. 29, pp.1339–1359. https://doi.org/10.1007/s10531-020-01938-2

Jaarsma, C.F. and Willems, G.P.A. 2002. Reducing habitat fragmentation by minor rural roads through traffic calming. Landscape and Urban Planning, Fragmentation and Land Use Planning: Analysis and beyond? 58, pp.125–135. https://doi.org/10.1016/S0169-2046(01)00215-8

Jäckel, D., Mortega, K.G., Sturm, U., Brockmeyer, U., Khorramshahi, O., Voigt-Heucke, S.L., 2021. Opportunities and limitations: A comparative analysis of citizen science and expert recordings for bioacoustic research. PLOS ONE 16, e0253763. https://doi.org/10.1371/journal.pone.0253763

Jones, J.P.G. 2011. Monitoring species abundance and distribution at the landscape scale. *Journal of Applied Ecology.* 48, pp.9–13. https://doi.org/10.1111/j.1365-2664.2010.01917.x

Joshi, A., Vaidyanathan, S., Mondol, S., Edgaonkar, A. and Ramakrishnan, U. 2013. Connectivity of Tiger (Panthera tigris) Populations in the Human-Influenced Forest Mosaic of Central India. *PLOS ONE*. 8(11). https://doi.org/10.1371/journal.pone.0077980.

Jumeau, J., Petrod, L., Handrich, Y., 2017. A comparison of camera trap and permanent recording video camera efficiency in wildlife underpasses. Ecol Evol 7, 7399–7407. https://doi.org/10.1002/ece3.3149

Jurecka, M. Influence of land use intensity on ecological corridors and wildlife crossings' effectiveness: comparison of 2 pilot areas in Austria, 2024. . Nature Conservation 57, 143–171. https://doi.org/10.3897/natureconservation.57.117154

Karthaus, G. 1985. Protective measures for migrating amphibians from danger from road traffic - observations and experiences. *Natur und Landschaft: Zeitschrift fur Naturschutz und Landschaftspflege*, 60, pp.242-247.

Kaszta, Ż., Cushman, S.A., Macdonald, D.W., 2020. Prioritizing habitat core areas and corridors for a large carnivore across its range. *Animal Conservation*. 23, pp.607–616. https://doi.org/10.1111/acv.12575

Keeley, A.T.H., Beier, P., Gagnon, J.W., 2016. Estimating landscape resistance from habitat suitability: effects of data source and nonlinearities. *Landscape Ecology*. 31, pp.2151–2162. https://doi.org/10.1007/s10980-016-0387-5

Keeley, A.T.H., Beier, P., Keeley, B.W., Fagan, M.E., 2017. Habitat suitability is a poor proxy for landscape connectivity during dispersal and mating movements. *Landscape and Urban Planning.* 161, pp.90–102. https://doi.org/10.1016/j.landurbplan.2017.01.007

Keeley, A.T.H., Beier, P., Jenness, J.S., 2021. Connectivity metrics for conservation planning and monitoring. Biological Conservation 255, 109008. https://doi.org/10.1016/j.biocon.2021.109008

Kim, I.R., Kim, K., Song, E., 2023. An Analysis of the Effectiveness of Mitigation Measures at Roadkill Hotspots in South Korea. Diversity 15, 1199. https://doi.org/10.3390/d15121199

Kindberg, J., Ericsson, G., Swenson, J.E., 2009. Monitoring rare or elusive large mammals using effort-corrected voluntary observers. *Biological Conservation* 142, pp.159–165. https://doi.org/10.1016/j.biocon.2008.10.009

Knaapen, J.P., Scheffer, M and Harms, B. 1992. Estimating habitat isolation in landscape planning. *Landscape and urban planning*. 23(1), pp.1-16. https://doi.org/10.1016/0169-2046(92)90060-D.

Knifka, W., Karutz, R., Zozmann, H. 2023. Barriers and Solutions to Green Facade Implementation—A Review of Literature and a Case Study of Leipzig, Germany. *Buildings*. 13, 1621. https://doi.org/10.3390/buildings13071621

Kor, L., O'Hickey, B., Hanson, M., Coroi, M., 2022. Assessing habitat connectivity in environmental impact assessment: a case-study in the UK context. Impact Assessment and Project Appraisal 40, 495–506. https://doi.org/10.1080/14615517.2022.2128557

Kumar, S.U. and Cushman, S. A. 2022. Connectivity modelling in conservation science: a comparative evaluation. *Nature*. 16680. https://doi.org/10.1038/s41598-022-20370-w

Kumar, S.U., Turnbull, J., Davies, O.H., Hodgetts, T. and Cushman, A. 2022. Moving beyond landscape resistance: considerations for the future of connectivity modelling and conservation science. *Landscape Ecology*. 37, pp.2465–2480. https://doi.org/10.1007/s10980-022-01504-x

Kuroe, M., Yamaguchi, N., Kadoya, T. and Miyashita, T. 2011. Matrix heterogeneity affects population size of the harvest mice: Bayesian estimation of matrix resistance and model validation. *Oikos*. 120, pp.271–279. https://doi.org/10.1111/j.1600-0706.2010.18697.x

Laliberté, J. and St-Laurent, M.H. 2020. Validation of functional connectivity modeling: The Achilles' heel of landscape connectivity mapping. *Landscape and Urban Planning*. 202, 103878. https://doi.org/10.1016/j.landurbplan.2020.103878

LaPoint, S., Gallery, P., Wikelski, M. and Kays, R., 2013. Animal behavior, cost-based corridor models, and real corridors. *Landscape Ecology*. 28, pp.1615–1630. https://doi.org/10.1007/s10980-013-9910-0

Laurance, W.F., Clements, G.R., Sloan, S., O'Connell, C.S., Mueller, N.D., Goosem, M., Venter, O., Edwards, D.P., Phalan, B., Balmford, A., Van Der Ree, R., Arrea, I.B., 2014. A global strategy for road building. *Nature*. 513, pp.229–232. https://doi.org/10.1038/nature13717

Leblond, M., Dussault, C., Ouellet, J.-P., Poulin, M., Courtois, R. and Fortin, J., 2007. Electric Fencing as a Measure to Reduce Moose-Vehicle Collisions. *The Journal of Wildlife Management*. 71, pp.1695–1703.

Lechner, A.M., Sprod, D., Carter, O. and Lefroy, E. 2017. Characterising landscape connectivity for conservation planning using a dispersal guild approach. *Landscape Ecology*. 32, pp.99–113. https://doi.org/10.1007/s10980-016-0431-5

Lee, T.S., Jones, P.F., Jakes, A.F., Jensen, M., Sanderson, K., Duke, D., 2023. Where to invest in road mitigation? A comparison of multiscale wildlife data to inform roadway prioritization. Journal for Nature Conservation 71, 126327. https://doi.org/10.1016/j.jnc.2022.126327

Lewandowski, E., Specht, H., 2015. Influence of volunteer and project characteristics on data quality of biological surveys. Conservation Biology 29, 713–723. https://doi.org/10.1111/cobi.12481

Li, F., Liu, X., Zhang, X., Zhao, D., Liu, H., Zhou, C., Wang, R., 2017. Urban ecological infrastructure: an integrated network for ecosystem services and sustainable urban systems. Journal of Cleaner Production, Urban ecological infrastructure for healthier cities: governance, management and engineering 163, S12–S18. https://doi.org/10.1016/j.jclepro.2016.02.079

Linkie, M., Chapron, G., Martyr, D.J., Holden, J. and Leader-Williams, N., 2006. Assessing the viability of tiger subpopulations in a fragmented landscape. *Journal of Applied Ecology*. 43, pp.576–586. https://doi.org/10.1111/j.1365-2664.2006.01153.x

Lu, S., Yue, Y., Wang, Y., Zhang, D., Yang, B., Yu, Z., Lin, H., Dai, Q., 2023. The Factors Influencing Wildlife to Use Existing Bridges and Culverts in Giant Panda National Park. *MDPI*. 15, 487. https://doi.org/10.3390/d15040487

MacKinnon, M., Pedersen Zari, M. and Brown, D.K. 2023. Improving Urban Habitat Connectivity for Native Birds: Using Least-Cost Path Analyses to Design Urban Green Infrastructure Networks. *MDPI*. 12, 1456. https://doi.org/10.3390/land12071456

Mahmoodi, S., Shadloo, S., Rezaei, S. and Shabani, A.A. 2023. Prediction of habitat suitability, connectivity, and corridors in the future to conserve roe deer (Capreolus capreolus) as a locally endangered species in northern Iran. *Journal for Nature Conservation*. 71, 126313. https://doi.org/10.1016/j.jnc.2022.126313

Mancini, F., Hodgson, J.A., Isaac, N.J.B., 2022. Co-designing an Indicator of Habitat Connectivity for England. Front. Ecol. Evol. 10. https://doi.org/10.3389/fevo.2022.892987

Martinez-Cillero, R., Siggery, B., Murphy, R., Perez-Diaz, A., Christie, I., Chimbwandira, S.J., 2023. Functional connectivity modelling and biodiversity Net Gain in England: Recommendations for practitioners. Journal of Environmental Management 328, 116857. https://doi.org/10.1016/j.jenvman.2022.116857

Matos, C., Petrovan, S., Ward, A.I., Wheeler, P., 2017. Facilitating permeability of landscapes impacted by roads for protected amphibians: patterns of movement for the great crested newt. PeerJ 5, e2922. https://doi.org/10.7717/peerj.2922

McClure, M., Hansen, A. and Inman, R., 2016. Connecting models to movements: testing connectivity model predictions against empirical migration and dispersal data. *Landscape Ecology*. 31. https://doi.org/10.1007/s10980-016-0347-0

McRae, B. and Beier, P. 2007. Circuit theory predicts gene flow in plant and animal populations. *Biological sciences*. 104(50), pp. 19885-19890. https://doi.org/10.1073/pnas.0706568104

McRae, B., Dickson, B., Keitt, T. and Shah, V. 2008. Using circuit theory to model connectivity in ecology, evolution, and conservation. *Ecology*. 89, pp. 2712-2724. DOI:10.1890/07-1861.1.

Medrano-Vizcaíno, P., Grilo, C., Brito-Zapata, D., González-Suárez, M., 2023. Landscape and road features linked to wildlife mortality in the Amazon. Biodivers Conserv. https://doi.org/10.1007/s10531-023-02699-4

Merrick, M.J. and Koprowski, J.L. 2017. Circuit theory to estimate natal dispersal routes and functional landscape connectivity for an endangered small mammal. *Landscape Ecology.* 32, pp.1163–1179. https://doi.org/10.1007/s10980-017-0521-z

Milanesi, P., Holderegger, R., Caniglia, R., Fabbri, E., Galaverni, M. and Randi, E. 2017. Expert-based versus habitat-suitability models to develop resistance surfaces in landscape genetics. *Oecologia.* 183, pp.67–79. https://doi.org/10.1007/s00442-016-3751-x.

Milda, D., Ramesh, T., Kalle, R., Gayathri, V. and Thanikodi, M., 2020. Ranger survey reveals conservation issues across Protected and outside Protected Areas in southern India. *Global Ecology and Conservation*. 24, e01256. https://doi.org/10.1016/j.gecco.2020.e01256

Moilanen, A., 2011. On the limitations of graph-theoretic connectivity in spatial ecology and conservation. *Journal of Applied Ecology.* 48, pp.1543–1547. https://doi.org/10.1111/j.1365-2664.2011.02062.x

Moore, L.J., Petrovan, S.O., Bates, A.J., Hicks, H.L., Baker, P.J., Perkins, S.E., Yarnell, R.W., 2023. Demographic effects of road mortality on mammalian populations: a systematic review. *Biological Reviews*. 98, pp.1033–1050. https://doi.org/10.1111/brv.12942

Morales-González, A., Ruiz-Villar, H., Ordiz, A., Penteriani, V., 2020. Large carnivores living alongside humans: Brown bears in human-modified landscapes. Global Ecology and Conservation 22, e00937. https://doi.org/10.1016/j.gecco.2020.e00937

Moqanaki, E.M., Cushman, S.A., 2017. All roads lead to Iran: Predicting landscape connectivity of the last stronghold for the critically endangered Asiatic cheetah. *Animal Conservation.* 20, pp.29–41. https://doi.org/10.1111/acv.12281

Naidoo, R., Kilian, J.W., Preez, P.D., Beytell, P., Aschenborn, O., Taylor, R.D. and Stuart-Hill, G. 2018. Evaluating the effectiveness of local- and regional-scale wildlife corridors using quantitative metrics of functional connectivity. *Biological Conservation.* 217, pp.96-103. https://doi.org/10.1016/j.biocon.2017.10.037.

Nelli, L., Schehl, B., Stewart, R.A., Scott, C., Ferguson, S., MacMillan, S., McCafferty, D.J., 2022. Predicting habitat suitability and connectivity for management and conservation of urban wildlife: A real-time web application for grassland water voles. *Journal of Applied Ecology*. 59, pp.1072–1085. https://doi.org/10.1111/1365-2664.14118

Neumann, W., Ericsson, G., Dettki, H., Bunnefeld, N., Keuler, N.S., Helmers, D.P., Radeloff, V.C., 2012. Difference in spatiotemporal patterns of wildlife road-crossings and wildlife-vehicle collisions. *Biological Conservation.* 145, pp.70–78. https://doi.org/10.1016/j.biocon.2011.10.011

Nguyen, T.T., Meurk, C., Benavidez, R., Jackson, B., Pahlow, M., 2021. The Effect of Blue-Green Infrastructure on Habitat Connectivity and Biodiversity: A Case Study in the Ōtākaro/Avon River Catchment in Christchurch. *New Zealand Sustainability* 13, 6732. https://doi.org/10.3390/su13126732

Oliveira Gonçalves, L., Kindel, A., Augusto Galvão Bastazini, V., Zimmermann Teixeira, F., 2022. Mainstreaming ecological connectivity in road environmental impact assessments: a long way to go. *Impact Assessment and Project Appraisal* 40, pp.475–480. https://doi.org/10.1080/14615517.2022.2099727

Paemelaere, E.A.D., Mejía, A., Quintero, S., Hallett, M., Li, F., Wilson, A., Barnabas, H., Albert, A., Li, R., Baird, L., Pereira, G., Melville, J., 2023. The road towards wildlife friendlier infrastructure: Mitigation planning through landscape-level priority settings and species connectivity frameworks. *Environmental Impact Assessment Review*. 99, 107010. https://doi.org/10.1016/j.eiar.2022.107010

Parsons, A.W., Goforth, C., Costello, R., Kays, R., 2018. The value of citizen science for ecological monitoring of mammals. *PeerJ*. 6, e4536. https://doi.org/10.7717/peerj.4536

Passoni, G., Coulson, T., Ranc, N., Corradini, A., Hewison, A.J.M., Ciuti, S., Gehr, B., Heurich, M., Brieger, F., Sandfort, R., Mysterud, A., Balkenhol, N., Cagnacci, F.,

2021. Roads constrain movement across behavioural processes in a partially migratory ungulate. *Movement Ecology*. 9, 57. https://doi.org/10.1186/s40462-021-00292-4

Paemelaere, E.A.D., Mejía, A., Quintero, S., Hallett, M., Li, F., Wilson, A., Barnabas, H., Albert, A., Li, R., Baird, L., Pereira, G., Melville, J., 2023. The road towards wildlife friendlier infrastructure: Mitigation planning through landscape-level priority settings and species connectivity frameworks. Environmental Impact Assessment Review 99, 107010. https://doi.org/10.1016/j.eiar.2022.107010

Périquet, S., Roxburgh, L., le Roux, A., Collinson, W.J., 2018. Testing the Value of Citizen Science for Roadkill Studies: A Case Study from South Africa. Front. Ecol. Evol. 6. https://doi.org/10.3389/fevo.2018.00015

Pernat, N., Kampen, H., Jeschke, J.M., Werner, D., 2021. Citizen science versus professional data collection: Comparison of approaches to mosquito monitoring in Germany. Journal of Applied Ecology 58, 214–223. https://doi.org/10.1111/1365-2664.13767

Petrovan, S.O., Vale, C.G., Sillero, N., 2020. Using citizen science in road surveys for large-scale amphibian monitoring: are biased data representative for species distribution? Biodivers Conserv 29, 1767–1781. https://doi.org/10.1007/s10531-020-01956-0

Phillips, S.J., Anderson, R.P., Schapire, R.E., 2006. Maximum entropy modeling of species geographic distributions. *Ecological Modelling. 190*, pp.231–259. https://doi.org/10.1016/j.ecolmodel.2005.03.026

Pimm, S., Raven, P., Perterson, A., Sekercioglu, C.H. and Ehrlich, P.R. 2006. Human impacts on the rates of recent, present, and future bird extinctions. *PNAS*. 103(29), pp. 10941-10946. DOI: 10.1073/pnas.0604181103.

Polak, T., Rhodes, J.R., Jones, D., Possingham, H.P., 2014. Optimal planning for mitigating the impacts of roads on wildlife. *Journal of Applied Ecology.* 51, pp.726–734. https://doi.org/10.1111/1365-2664.12243

Poor, E.E., Loucks, C., Jakes, A. and Urban, D.L. 2012 Comparing Habitat Suitability and Connectivity Modeling Methods for Conserving Pronghorn Migrations. *PLOS ONE*. 7(11): e49390. https://doi.org/10.1371/journal.pone.0049390

Préau, P., Tournebize, J., Lenormand, M., Alleaume, S., Boussada, V. G., and Luque, S. 2022. Habitat connectivity in agricultural landscapes improving multi-functionality of constructed wetlands as nature-based solutions. *Ecological Engineering*. 182. https://doi.org/10.1016/j.ecoleng.2022.106725

Prenda, J., Domínguez-Olmedo, J.L., López-Lozano, E., Fernández de Villarán, R., Negro, J.J., 2024. Assessing citizen science data quality for bird monitoring in the Iberian Peninsula. Sci Rep 14, 20307. https://doi.org/10.1038/s41598-024-70827-3

Pullinger, M.G., Johnson, C.J., 2010. Maintaining or restoring connectivity of modified landscapes: evaluating the least-cost path model with multiple sources of ecological information. *Landscape Ecology*. 25, pp.1547–1560. https://doi.org/10.1007/s10980-010-9526-6

R Core Team. 2021. R: A language and environment for statistical. Computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.

Rathore, C.S., Dubey, Y., Shrivastava, A., Pathak, P., Patil, V., 2012. Opportunities of Habitat Connectivity for Tiger (Panthera tigris) between Kanha and Pench National Parks in Madhya Pradesh, India. *PLOS ONE.* 7, e39996. https://doi.org/10.1371/journal.pone.0039996

Raymond, S., Schwartz, A.L.W., Thomas, R.J., Chadwick, E., Perkins, S.E., 2021. Temporal patterns of wildlife roadkill in the UK. *PLOS ONE.* 16, e0258083. https://doi.org/10.1371/journal.pone.0258083

Rudnick, D., Ryan, S., Beier, P., Cushman, S., Dieffenbach, F., Epps, C., Gerber, L., Hartter, J., Jenness, J., Kintsch, J., Merenlender, A., Perkl, R., Preziosi, D., Trombulak, S., 2012. The Role of Landscape Connectivity in Planning and Implementing Conservation and Restoration Priorities. *Issues in Ecology*. 16, pp.1–20.

Rytwinski, T., Soanes, K., Jaeger, J.A.G., Fahrig, L., Findlay, C.S., Houlahan, J., Ree, R. van der, Grift, E.A. van der, 2016. How Effective Is Road Mitigation at Reducing Road-Kill? A Meta-Analysis. PLOS ONE 11, e0166941. https://doi.org/10.1371/journal.pone.0166941

Salamak, M., Fross, K., 2016. Bridges in Urban Planning and Architectural Culture. Procedia Engineering, World Multidisciplinary Civil Engineering-Architecture-Urban Planning Symposium 2016, WMCAUS 2016 161, pp.207–212. https://doi.org/10.1016/j.proeng.2016.08.530

Sartor, C.C., Wan, H.Y., Pereira, J.A., Eizirik, E., Trigo, T.C., Freitas, T.R.O. de, Cushman, S.A., 2022. Landscape genetics outperforms habitat suitability in predicting landscape resistance for congeneric cat species. *Journal of Biogeography.* 49, pp.2206–2217. https://doi.org/10.1111/jbi.14498

Sawyer, S.C., Epps, C.W., Brashares, J.S., 2011. Placing linkages among fragmented habitats: do least-cost models reflect how animals use landscapes? *Journal of Applied Ecology*. 48, pp.668–678. https://doi.org/10.1111/j.1365-2664.2011.01970.x

Saxena, A., Habib, B., 2022. Crossing structure use in a tiger landscape, and implications for multi-species mitigation. Transportation Research Part D: Transport and Environment 109, 103380. https://doi.org/10.1016/j.trd.2022.103380

Schmidt, G.M., Lewison, R.L., Swarts, H.M., 2021. Pairing long-term population monitoring and wildlife crossing structure interaction data to evaluate road mitigation effectiveness. Biological Conservation 257, 109085. https://doi.org/10.1016/j.biocon.2021.109085

Schwartz, A.L.W., Williams, H.F., Chadwick, E., Thomas, R.J., Perkins, S.E., 2018. Roadkill scavenging behaviour in an urban environment. Journal of Urban Ecology 4, juy006. https://doi.org/10.1093/jue/juy006

Schwartz, A.L.W., Shilling, F.M. and Perkins, S.E., 2020. The value of monitoring wildlife roadkill. *European Journal Wildlife Research*. 66, 18. https://doi.org/10.1007/s10344-019-1357-4

Seoane, J., Bustamante, J., Díaz-Delgado, R., 2005. Effect of Expert Opinion on the Predictive Ability of Environmental Models of Bird Distribution. *Conservation Biology*. 19, pp.512–522. https://www.jstor.org/stable/i369584

Shirk, A.J., Schroeder, M.A., Robb, L.A., Cushman, S.A., 2015. Empirical validation of landscape resistance models: insights from the Greater Sage-Grouse (Centrocercus urophasianus). *Landscape Ecology.* 30, pp.1837–1850. https://doi.org/10.1007/s10980-015-0214-4

Shin, Y., Kim, K., Groffen, J., Woo, D., Song, E., Borzée, A., 2022. Citizen science and roadkill trends in the Korean herpetofauna: The importance of spatially biased and unstandardized data. Front. Ecol. Evol. 10. https://doi.org/10.3389/fevo.2022.944318

Sijtsma, F.J., van der Veen, E., van Hinsberg, A., Pouwels, R., Bekker, R., van Dijk, R.E., Grutters, M., Klaassen, R., Krijn, M., Mouissie, M., Wymenga, E., 2020. Ecological impact and cost-effectiveness of wildlife crossings in a highly fragmented landscape: a multi-method approach. *Landscape Ecology*. 35, pp.1701–1720. https://doi.org/10.1007/s10980-020-01047-z

Simpson, N., Stewart, K., Schroeder, C., Cox, M., Huebner, K., Wasley, T., 2016. Overpasses and underpasses: Effectiveness of crossing structures for migratory ungulates: Crossing Structures and Migratory Ungulates. *The Journal of Wildlife Management*. 80. https://doi.org/10.1002/jwmg.21132

Soanes, K., Rytwinski, T., Fahrig, L., Huijser, M.P., Jaeger, J.A.G., Teixeira, F.Z., van der Ree, R., van der Grift, E.A., 2024. Do wildlife crossing structures mitigate the barrier effect of roads on animal movement? A global assessment. Journal of Applied Ecology 61, 417–430. https://doi.org/10.1111/1365-2664.14582

Stevenson-Holt, C.D., Watts, K., Bellamy, C.C., Nevin, O.T., Ramsey, A.D., 2014. Defining Landscape Resistance Values in Least-Cost Connectivity Models for the Invasive Grey Squirrel: A Comparison of Approaches Using Expert-Opinion and Habitat Suitability Modelling. *PLOS ONE*. 9, e112119. https://doi.org/10.1371/journal.pone.0112119

Sullivan, B.L., Wood, C.L., Iliff, M.J., Bonney, R.E., Fink, D., Kelling, S., 2009. eBird: A citizen-based bird observation network in the biological sciences. *Biological Conservation.*142, pp.2282–2292. https://doi.org/10.1016/j.biocon.2009.05.006

Suttidate, N., Steinmetz, R., Lynam, A.J., Sukmasuang, R., Ngoprasert, D., Chutipong, W., Bateman, B.L., Jenks, K.E., Baker-Whatton, M., Kitamura, S., Ziółkowska, E. and Radeloff, V.C. 2021. Habitat connectivity for endangered Indochinese tigers in Thailand. *Global Ecology and Conservation*. 29. https://doi.org/10.1016/j.gecco.2021.e01718.

Sun, C.C., Hurst, J.E., Fuller, A.K., 2021. Citizen Science Data Collection for Integrated Wildlife Population Analyses. *Frontiers in Ecology and Evolution 9*.

Sweanor, L.L., Logan, K.A., Bauer, J.W., Millsap, B., Boyce, W.M., 2008. Puma and Human Spatial and Temporal Use of a Popular California State Park. *The Journal of Wildlife Management*. 72, pp.1076–1084. https://doi.org/10.2193/2007-024

Tarabon, S., Dutoit, T. and Isselin-Nondedeu, F., 2021. Pooling biodiversity offsets to improve habitat connectivity and species conservation. *Journal of Environmental Management*. 277, 111425. https://doi.org/10.1016/j.jenvman.2020.111425

Taylor, B.D., Goldingay, R.L., 2010. Roads and wildlife: impacts, mitigation and implications for wildlife management in Australia. Wildl. Res. 37, 320–331. https://doi.org/10.1071/WR09171

Teitelbaum, C.S., Hepinstall-Cymerman, J., Kidd-Weaver, A., Hernandez, S.M., Altizer, S., Hall, R.J., 2020. Urban specialization reduces habitat connectivity by a highly mobile wading bird. *Movement Ecology.* 8, 49. https://doi.org/10.1186/s40462-020-00233-7

Travers, T.J.P., Alison, J., Taylor, S.D., Crick, H.Q.P., Hodgson, J.A., 2021. Habitat patches providing south–north connectivity are under-protected in a fragmented landscape. *Proceedings of the Royal Society B: Biological Sciences.* 288, 20211010. https://doi.org/10.1098/rspb.2021.1010

Trouwborst, A., 2010. Managing the Carnivore Comeback: International and EU Species Protection Law and the Return of Lynx, Wolf and Bear to Western Europe. Journal of Environmental Law 22, 347–372.

Underhill, J. and Angold, P. 2011. Effects of roads on wildlife in an intensively modified landscape. *Environmental Reviews*. 8, pp.21-39. DOI: 10.1139/a00-003.

Unnithan Kumar, S., Cushman, S.A., 2022. Connectivity modelling in conservation science: a comparative evaluation. *Scientific Reports*. 12, 16680. https://doi.org/10.1038/s41598-022-20370-w

Unnithan Kumar, S., Turnbull, J., Hartman Davies, O., Hodgetts, T., Cushman, S.A., 2022. Moving beyond landscape resistance: considerations for the future of connectivity modelling and conservation science. *Landscape Ecology*. 37, pp.2465–2480. https://doi.org/10.1007/s10980-022-01504-x

Valerio, F., Basile, M., Balestrieri, R., 2021. The identification of wildlife-vehicle collision hotspots: Citizen science reveals spatial and temporal patterns. Ecological Processes 10, 6. https://doi.org/10.1186/s13717-020-00271-4

van der Grift, E.A., van der Ree, R., Fahrig, L., Findlay, S., Houlahan, J., Jaeger, J.A.G., Klar, N., Madriñan, L.F., Olson, L., 2013. Evaluating the effectiveness of road mitigation measures. Biodiversity Conservation. 22, pp.425–448. https://doi.org/10.1007/s10531-012-0421-0 van der Ree, R., Cesarini, S., Sunnucks, P., Moore, J., Taylor, A., 2010. Large Gaps in Canopy Reduce Road Crossing by a Gliding Mammal. *Ecology and Society*. 15. https://doi.org/10.5751/ES-03759-150435

Vercayie, D., Herremans, M., 2015. Citizen science and smartphones take roadkill monitoring to the next level. Nature Conservation 11, 29–40. https://doi.org/10.3897/natureconservation.11.4439

Veenbaas, G. and Brandjes, J. 1999. Use of fauna passages along waterways under highways. Proceedings of the International Conference on Wildlife Ecology and Transportation, *Florida Department of Transportation, Tallahassee, Florida USA*, 253-258.

Wade, B. S., Carter, E. T., Derolph, C. R., Byrd, G., Darling, S. E., Hayter, L. E., Jett, R. T., Herold, J. M., and Giffen, N. R. 2023. Advancing wildlife connectivity in land use planning: a case study with four-toed salamanders. *Journal of Wildlife Management.* 87:e22456. https://doi.org/10.1002/jwmg.22456

Wang, Y., Guan, L., Chen, J., Kong, Y., 2018. Influences on mammals frequency of use of small bridges and culverts along the Qinghai–Tibet railway, China. *Ecological Research.* 33, pp.879–887. https://doi.org/10.1007/s11284-018-1578-0

Wasserman, T., Cushman, S., Schwartz, M., Wallin, D., 2010. Spatial scaling and multi-model inference in landscape genetics: Martes americana in northern Idaho. *Landscape Ecology*. 25, pp.1601–1612. https://doi.org/10.1007/s10980-010-9525-7

Way, J.G., Ortega, I.M., Strauss, E.G., 2004. Movement and Activity Patterns of Eastern Coyotes In a Coastal, Suburban Environment. *NENA*. 11, pp.237–254. https://doi.org/10.1656/1092-6194(2004)011[0237:MAAPOE]2.0.CO;2

White, I.C., Hughes, S.A., 2019. Trial of a bridge for reconnecting fragmented arboreal habitat for hazel dormouse Muscardinus avellanarius at Briddlesford Nature Reserve, Isle of Wight, UK. *Conservation Evidence*. 16, pp.6-11.

Wilcox, M., Jeffery, N.W., DiBacco, C., Bradbury, I., Lowen, B., Wang, Z., Beiko, R. and Stanley, R. 2023. Integrating seascape resistances and gene flow to produce area-based metrics of functional connectivity for marine conservation planning. *Landscape Ecology*. 38, pp.1-17. DOI:10.1007/s10980-023-01690-2.

Wilmers, C.C., Wang, Y., Nickel, B., Houghtaling, P., Shakeri, Y., Allen, M.L., Kermish-Wells, J., Yovovich, V. and Williams, T., 2013. Scale Dependent Behavioral Responses to Human Development by a Large Predator, the Puma. *PLOS ONE.* 8, e60590. https://doi.org/10.1371/journal.pone.0060590

Yumnam, B., Jhala, Y.V., Qureshi, Q., Maldonado, J.E., Gopal, R., Saini, S., Srinivas, R. and Fleischer, C. 2014. Prioritizing Tiger Conservation through Landscape Genetics and Habitat Linkages. PLOS ONE. 9(11). https://doi.org/10.1371/journal.pone.0111207

Zarnetske, P.L., Baiser, B., Strecker, A., Record, S., Belmaker, J. and Tuanmu, M. 2017. The Interplay Between Landscape Structure and Biotic Interactions. *Current Landscape Ecology*. 2, pp.12–29. https://doi.org/10.1007/s40823-017-0021-5

Zeller, K.A., McGarigal, K., Beier, P., Cushman, S.A., Vickers, T.W., Boyce, W.M., 2014. Sensitivity of landscape resistance estimates based on point selection functions to scale and behavioral state: pumas as a case study. *Landscape Ecology.* 29, pp.541–557. https://doi.org/10.1007/s10980-014-9991-4

Zhang, J., Jiang, F., Cai, Z., Dai, Y., Liu, D., Song, P., Hou, Y., Gao, H., Zhang, T., 2021. Resistance-Based Connectivity Model to Construct Corridors of the Przewalski's Gazelle (Procapra Przewalskii) in Fragmented Landscape. *Sustainability.* 13, 1656. https://doi.org/10.3390/su13041656

Zhou, S., Griffiths, S.P., 2007. Estimating abundance from detection−nondetection data for randomly distributed or aggregated elusive populations. *Ecography*. 30, pp.537–549. https://doi.org/10.1111/j.0906-7590.2007.05009.x

Zhuo, Y., Xu, W., Wang, M., Chen, C., da Silva, A.A., Yang, W., Ruckstuhl, K.E. and Alves, J., 2022. The effect of mining and road development on habitat fragmentation and connectivity of khulan (Equus hemionus) in Northwestern China. *Biological Conservation.* 275, 109770. https://doi.org/10.1016/j.biocon.2022.109770

Zimmermann Teixeira, F., Kindel, A., Hartz, S.M., Mitchell, S., Fahrig, L., 2017. When road-kill hotspots do not indicate the best sites for road-kill mitigation. Journal of Applied Ecology 54, 1544–1551. https://doi.org/10.1111/1365-2664.12870

6.0 Appendices

Appendix 1 – Grey infrastructure terms and sources.

Data source	Terms included	Reference
Ordnance survey data	Access point (bridge), Archway (bridge), Aqueduct, Aqueduct (disused), Aqueduct (scrub), Bridge, Bridge (Access), Bridge (canal), Bridge (coniferous), Bridge (disused), Bridge (scrub), Bridge (towpath), Bridge (woodland), Bridge (bridleway), Farm underpass, Footbridge, Underpass, Underpass (Civilian), Underpass (agriculture), Underpass (access), Underpass (road), Tunnel, Tunnel (road), Tunnel (foot), Tunnel (bike),	Edina digimap. 2023. OS MasterMap,Digimap OS Collection. https://digimap.edina.ac.uk
National resources Wales	Bridge, Bridge (Forestry) Bridge (Private) Overpass, Tunnel, Walkway - Bridge Public walking route.	National Resources Wales. 2023. Forest Roads and Planned Forest Roads. https://datamap.gov.wales/maps/new?layer=inspire-nrw:NRW_FOREST_ROADS#/ And National Resources Wales. 2023. Active travel network. https://datamap.gov.wales/maps/active-travel-network-maps/view#/
Open Street map	Bridge: Access Bridge: Cantilever Bridge: Covered Bridge: Foot Bridge: Man_Made Bridge: Movable Bridge: Structure Bridge: Support Bridge: Yes Building: Bridge Footbridge Crossing: Raised Man_Made: Bridge Road: Crossing_Structure Structure: Bridge.	Open street map. 2023. openstreetmap.org/copyright

Canal and Rivers Trust	Bridge Overbridge Towpath_bridge	Canal and Rivers Trust. 2023. Bridge database. https://data-canalrivertrust.opendata.arcgis.com/datasets/CanalRiverTrust::bridges-21/about
Natural England	Bridge Footbridge Tunnel Underpass	Natural England. 2023. Maps and data on the natural environment. https://www.gov.uk/guidance/how-to-access-natural-englands-maps-and-data
Scottish Government	Bridge	Spatial Data.gov.scot. National Forest Estate Bridges GB. https://www.spatialdata.gov.scot/geonetwork/srv/api/records/ed9c3a33-f504-472c-b7a7-bcce1c6212fb