ELUCIDATING THE DRIVERS, CONTEXTUAL SENSITIVITY AND RESILIENCE OF URBAN ECOLOGICAL SYSTEMS

by

JAMES DAVID HALE

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School of Geography, Earth and Environmental Sciences

College of Life and Environmental Sciences

University of Birmingham

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ABSTRACT

As the global population urbanises, the benefits derived from contact with nature increasingly depend upon the presence of diverse urban ecological communities. These may be threatened by changes in land-cover and the intensification of land-use. A key question is how to design and manage cities to retain desirable species, habitats and processes. Addressing this question is challenging, due to the dominant role of humans in shaping spatially and temporally complex urban landscapes.

Earlier research identified ecological patterns along urban–rural gradients, often using simplified measures of built form and disturbance. The central theme within this thesis is that we require a more mechanistic understanding of the processes that created today‘s ecological patterns, which recognises the interactions between social and ecological sub-systems.

Using bats (Chiroptera) as a case study group, I identified a broadly negative association between bat activity and built density. Urban tree networks appeared beneficial for one species, and further work revealed that their role in facilitating movement depended upon the size of gaps in tree lines and their illumination level. Resilience analyses were used to map diverse dependencies between the functioning of urban bat habitats and human social factors; illustrating the value of a more mechanistic, systems-based approach.

DEDICATION

I dedicate this thesis to my urban family. In particular, I would like to thank my wife Sabrina, for her incredible support, understanding and love.

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I would like to thank Professor Jon Sadler for his support over many years and his willingness to let me explore more than a few tangents along the way! He provided my first opportunity to work as part of an academic research group and gave me the impression that he always had complete confidence in me, even though he must have had a few doubts at times.

I am also incredibly grateful to Professor Christopher Rogers, who has tirelessly fought to include ecological (and broader environmental) research as part of his EPSRC funded research projects. As a result, I have been exposed to a wide variety of research ideas from the engineering and social sciences, and had the pleasure of meeting some wonderful researchers.

I would also like to thank the co-authors of the five publications included in this thesis, as they really helped me to take a broader systems-view of urban areas. Particular thanks need to go to Ali Fairbrass and Tom Matthews for their help with fieldwork in the depths of the urban jungle.

Primary data collection has often depended on the goodwill of both friends and strangers. In particular, I would like to thank volunteers associated with the Birmingham and Black Country Bat Group (BBCBG) and the Open Air Laboratories (OPAL).

Numerous private landowners supported our surveys and analyses, as well as staff from Birmingham City Council (notably Nick Grayson), Lancaster City Council, the Royal Institute of British Architects, and the Building Research Establishment. I am also indebted to a range of data providers including EcoRecord - the biological record centre for Birmingham and the Black Country, the Birmingham Environmental Partnership and BBCBG. A range of geospatial data and advice were provided by the Ordnance Survey (GB), Bluesky International Limited and the UK Environment Agency Geomatics Group, with much support from the Joint Information Systems Committee (JISC) funded Landmap service.

Finally, I would like to thank the UK Engineering and Physical Sciences Research Council (EPSRC) for their funding of this research (numbers EP/F007426/1, EP/J017698/1 and EP/I035129/1). Their support for interdisciplinary research on urban areas has enabled me to work with a broad range of very capable researchers, exposing me to a variety of disciplinary perspectives which I am convinced has greatly benefitted this thesis.

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INTRODUCTION

Urban landscapes are far from natural, yet they contain semi-natural elements that are often of great value to those that live and work in cities (Tzoulas et al., 2007). One might therefore expect that management to enhance desirable species, habitats, and ecosystem processes would be a common characteristic of modern cities. However, despite considerable efforts to *green* urban areas, gaps in our understanding of how urban ecological patterns and processes have emerged are preventing their full potential being realised (Breuste et al., 2008, Niemela, 1999).

Ecologists have historically avoided the study of nature within cities (Collins et al., 2000, Grimm et al., 2008, Niemela, 1999), possibly because models for how more natural ecosystems operate are not easily transferred to such highly modified landscapes (Shochat et al., 2006). Recently, ecosystems within urban areas have received greater research attention (Gaston, 2010), supported by a more holistic view of the environment that urban organisms experience. Habitats within cities are not limited to green and blue patches, fragmented by a matrix of concrete, tarmac and rubble. Rather, the high diversity of urban land-covers and human activities (Breuste et al., 2008) has created, often unintentionally, a variety of novel habitat features, food webs and communities. For example, residential houses can provide excellent nesting spaces for bats (Chiroptera) (Altringham, 2003), urban roads can provide useful sources of carrion for coyotes (*Canis latrans*) (Morey et al., 2007), and the species richness of a brownfield can be many times greater than a nearby park (Donovan et al., 2005). Explicit recognition of the various roles that humans play in structuring urban ecological systems is evident within many recent studies. Models for nutrient cycling have been adapted to include the artificial fertilisation of gardens and the removal of *green waste* (Templer et al., 2015), impacts of domestic cats have been accounted for within studies of avian reproductive success (Bonnington et al., 2013), and research on urban seed dispersal has explored the role of vehicles entering and leaving the city (von der Lippe and Kowarik, 2008).

The importance of nature within urban areas

Cities and towns are habitats for people, and are where the majority of the global population now reside. By 2050 two thirds of the global population will live in urban areas (UN, 2014). In many parts of the world, rural populations are declining whilst urban populations are increasing, raising questions about the impact of urban expansion and densification on human wellbeing (Kent and Thompson, 2014, Moore et al., 2003). The liveability of these cities is therefore of considerable importance and the role that semi-natural components can play in enhancing residents' quality of life is increasingly being recognised (Bolund and Hunhammar, 1999, Gaston et al., 2013). At coarse spatial scales, human population density tends to correlate positively with species richness (Luck, 2007). One implication of this is that human settlements may threaten this local biodiversity though expansion and disturbance, yet these local opportunities for citizens to enjoy positive wildlife experiences might promote more environmentally responsible behaviour (Dunn et al., 2006, Nisbet et al., 2008). These findings support the argument that the conservation and enhancement of semi-natural habitats within cities may have benefits that range from local to global. Interest in conserving semi-natural features, communities and ecological processes within cities is therefore high in many parts of the world

(McDonnell and Hahs, 2013), as reflected by the various urban planting projects (Pincetl, 2010), planning policies (Conway and Urbani, 2007, Sadler et al., 2010), professional networks (Müller and Kamada, 2011) and journals (e.g. Urban Forestry & Urban Greening) that focus on this subject. If we accept that many forms of nature within cities are desirable, the next step is to understand the urban forms and processes that need to be modified to deliver and retain this diversity.

The complexity of urban systems

Cities are meta-stable, self-organising systems, whose multiple forms, flows and functions can change rapidly (Grimm et al., 2008, Ahern, 2013). Economic cycles and periods of physical redevelopment (Dallimer et al., 2011), coupled with changing technologies and social practices, can result in stresses to existing habitats and species (Partecke et al., 2006), whilst at the same time creating conditions for the development of new (often novel) behaviours (Fuller et al., 2007) and ecological communities (Goddard et al., 2010). These threats and opportunities are also enhanced through the large movement of materials and people into the city, bringing with them (often unintentionally) a variety of species, along with new ideas about how the city should be managed. An additional layer of complexity is delivered by the multiple scales at which urban areas are managed (Borgström et al., 2006, Ernstson et al., 2010, Sadler et al., 2010), and at which spatial and temporal patterns and impacts occur (Grimm et al., 2008, Luck, 2010, McDonnell and Pickett, 1990). For example, decisions relating to private gardens typically take place at the scale of individual households, whilst many of the species that use these gardens operate over larger spatial extents (Goddard et al., 2010). The complexity of cities presents a variety of challenges to the conservation, enhancement and creation of urban biodiversity and ecosystem services. Clear description of the urban context for surveys and experiments is important, yet an immediate barrier is that no clear definition of ‘urban’ exists, and definitions of what constitutes ‘urban’ are rarely provided within ecology studies (McIntyre et al., 2000). High-densities of humans, sealed surfaces and infrastructure are widely accepted urban characteristics; yet the boundaries between urban, suburban, urban-fringe and rural are largely arbitrary (Gaston, 2010). Less subjective descriptions of urban form and its intensity of use are increasingly possible thanks to advances in remote sensing and mapping. Broad measures of urbanisation are useful for supporting comparative research, although these are often too simplistic to shed light on mechanisms that are driving a particular ecological response (McDonnell and Hahs, 2008). However, more choices for how to describe urban areas may unintentionally create even greater difficulty in drawing comparisons between studies. Additional complexity is introduced by the desire to better account for the human drivers of urban ecological processes, as ecological models are expanded to include political, economic and cultural dimensions (Collins et al., 2000, Pickett et al., 2001).

Urban ecology as interdisciplinary research

Ecological models for urban areas still struggle with identifying elements of the human sub-system that are most relevant, and with understanding how and why these vary in space and time (Collins et al., 2000). The human dimensions of urban systems tend to be heavily simplified by ecologists, and incorporating information and tools from the social sciences may improve the ability of ecologists to capture key information about the cultural, social, economic and built context for their research (McIntyre et al., 2000, Pickett et al., 2001). I argue that to undertake effective ecological research within urban environments requires an explicit acknowledgement that humans are central to shaping the structure and function of this ecosystem type. It is also likely that the answers to key questions within urban ecology lie outside the traditional disciplinary boundaries of ecological research, and that working across disciplines may be beneficial. An interdisciplinary approach to urban ecology has been widely advocated (McIntyre et al., 2000, Pickett et al., 2001, Alberti et al., 2003, Tzoulas et al., 2007), and collaboration between the ecological and social sciences would be expected to be mutually beneficial (Niemela, 1999). However, it is not clear that interdisciplinary research is always required, as much can be gained by multiple disciplines working together without aiming for coherence or synthesis of disciplinary knowledge (Petts et al., 2008). Interdisciplinary studies of urban areas have been supported by funding agencies within North America (Grimm et al., 2000, Collins et al., 2000) and Europe (Petts et al., 2008). However such collaborative work is recognised as being particularly challenging, not only due to differences in how disciplines frame and address research questions, but also due to the dominance of traditional disciplinary views on how the quality of the resulting research should be evaluated (Petts et al., 2008, Boyko et al., 2014). The papers presented within this thesis are the direct result of two cross-disciplinary projects funded by the Engineering and Physical Sciences Research Council: Urban Futures and Liveable Cities (grants EP/F007426/1 and EP/J017698/1).

Within the five papers that form this thesis, I have endeavoured to incorporate this broader view on the role of humans in shaping semi-natural communities within urban landscapes. These papers range from a straightforward ecology-in-cities approach (Grimm et al., 2000), which includes anthropogenic variables in the modelling of ecological patterns (paper II); to an ―ecology of cities" perspective that treats the city as a social–ecological system (papers II and V). I have also focused on the topic of artificial outdoor lighting, a pervasive characteristic of urban areas and a strong indicator of human activity (Elvidge et al., 1997, Sutton et al., 2001). Extensive and intensive lighting of human settlements has radically altered the cycles of darkness and light that natural ecosystems have experienced for hundreds of millions of years, yet comparatively little is known about its ecological impacts (Hölker et al., 2010). A major barrier to exploring the landscape-scale impacts of artificial lighting is the huge variability in the brightness, density and spectral quality of lamps. This diversity is an indication of the multiple of reasons why such lighting has been introduced (e.g. safety, security, aesthetic and amenity) and is also the result of partial lamp replacement as new lighting technologies become available. Paper III describes how high-resolution aerial night photography for the entire city of Birmingham was used to explore basic patters in urban lighting, to identify sites for field surveys and to estimate city-wide ecological impacts (paper IV).

Aims and nature of the research

The central theme within this thesis is that whilst identifying broad urban ecological patterns is important, it is insufficient as an endpoint if the ultimate goal is to support conservation practice in cities. To realise the long-term improvement of urban biodiversity, I argue that we require a more mechanistic understanding of the processes that have created the patterns we observe today. In addition, research methods are needed that can clarify the social, environmental and economic factors upon which these processes depend.

We chose bats as a study group, as they are highly mobile and their presence at a particular location may provide a broad indication of its ecological health. In addition, mammals, nocturnal and cryptic taxa are poorly represented in the literature on the ecological impacts of urban intensification (McDonnell and Hahs, 2008). The variety of bats known to be present within the study region also provided the opportunity to compare their response to different configurations of the built form, and to empirically test earlier observations that some bat species are sensitive to the structural connectivity of tree cover (Verboom and Huitema, 1997) and lighting (Stone et al., 2009).

Key aims for this thesis are to:

1. Describe the spatial patterning of the bat community within a UK city and identify the land-cover and land-use variables with which they are associated.
2. Explore the underlying mechanisms that shape these patterns.
3. Identify broad urban system conditions upon which the bat community depends.

THEORETICAL FRAMEWORK

Given the complexity of urban areas and their unusual characteristics in comparison to other landscapes, some have asked whether new theoretical approaches are required to support ecological studies in cities and towns (Collins et al., 2000, Niemela, 1999, Wu, 2014), and to support urban research more generally (Short, 2014). The position I adopted within this thesis is that existing ecological theories are still appropriate, as long as urban landscapes are reconceptualised by ecologists to better account for the dominant role that humans play in shaping ecological patterns and processes (Alberti et al., 2003, Niemela, 1999). However, the point at which this integration of knowledge from several urban disciplines becomes so transformative that it forms new theory is unclear. The papers within this thesis apply standard landscape ecology theory to urban areas (Breuste et al., 2008), but incorporate data on human modified land-cover and human practices that impact ecological function. In addition, these papers make use of resilience theory (Gunderson, 2000) to identify dependencies between urban ecological and social sub-systems, and to explore the vulnerability of ecological mitigation and enhancement initiatives.

Landscape ecology and urban areas

Urban areas differ from other landscapes, not only in terms of the diversity (Breuste et al., 2008) and intensity of human activity, but also in spatial characteristics such as patchiness (Niemela, 1999). The size of land-cover parcels tends to reduce towards the centre of cities, whilst their physical isolation tends to increase (Luck and Wu, 2002, Zhang et al., 2004). Metapopulation theory (Hanski and Gilpin, 1991) and island biogeography theory (MacArthur, 1967) predict that small and isolated habitat patches have the lowest biodiversity and lowest rates of re-colonisation, yet these theories were not specifically developed for urban landscapes. A recent meta-analysis of the factors influencing biodiversity levels within cities found that positive relationships do exist between species diversity and habitat patch area, but that there is little evidence for an effect of distance between patches (Beninde et al., 2015). The implication here is that the probability of movement within urban landscapes is not simply a function of patch distance, and that other factors influence movement choices and events. Studies of movement within cities indicate that the nature of the urban land-cover between habitat patches (i.e. the matrix) may be particularly important. For example, radio-tracking and genetic techniques have revealed that urban roads may not only act as physical barriers to carnivore movement, but that despite their spatial proximity, populations on either side of the road may become genetically isolated from each other (Riley et al., 2006). The results of such studies could be scaled to model ‗esistance‘ to movement across an entire landscape, recognising that the energetic costs and risks associated with movement may vary at fine spatial scales (Zeller et al., 2012). However, few studies have attempted to model landscape resistance within urban landscapes (but see Verbeylen et al. (2003)).

Several approaches have been adopted that attempt to describe the urban landscape in a manner that facilitates a more structured investigation of intra-urban ecological performance, and inter-city comparison. The most notable of these is the urban-rural gradient (McDonnell and Pickett, 1990); at its simplest it can be thought of a transect that runs from the edge to the centre of a city, but more typically it involves the selection of sampling locations that vary in the level of ‗urbanisation' of their local context (McDonnell and Hahs, 2008). For example, survey sites might be selected along a gradient of percentage built land cover, population density or air pollution within a specified distance from the site boundary. This diversity of metrics illustrates a key drawback of this approach, in that urban intensification is a complex anthropogenic gradient, along which many types of disturbance and habitat modification occur. Such anthropogenic variables often co-vary at multiple spatial scales (Andersson et al., 2009). The gradient approach can therefore be useful when seeking to stratify sampling evenly along a particular axis of disturbance, especially when testing for intermediate disturbance effects (see Wilkinson (1999)). However, it is unlikely to reveal the underlying mechanisms behind the patterns it identifies.

These landscape ecology theories and hypotheses are directly relevant to paper I, where the extent and structural connectivity of habitat patches is considered in the analysis of urban bat activity along an urbanisation gradient. In addition, within paper IV the fine-scale impacts of structural fragmentation of tree cover and lighting are modelled and used to create a resistance surface for an entire city.

Cities as systems, and ecological resilience

It could be argued that ecology as a discipline is particularly well placed to study heterogeneous human dominated landscapes such as cities. Ecologists deal with complex systems, disturbances, feedback loops, energy and material flows, and processes that operate at multiple scales (Ahern, 2013, Pickett et al., 2001). Ecological theory has been ‗borrowed‘ and applied to largely unrelated areas. Models for nutrient cycling within food webs have inspired the field of industrial ecology (Nielsen, 2007), and ecological concepts such as competition and succession have been used by social scientists to understand human social organisation (Wu, 2014).

Whilst it is clear that many recent ecological studies in urban areas have taken into account the impact of built structures or human activity in some way, several authors have argued that greater integration of human and ecological systems is needed at a conceptual level. A variety of models have been proposed in an attempt to capture the key social and semi-natural components of cities and their often complicated interactions (Alberti et al., 2003, Grimm et al., 2000, McDonnell and Pickett, 1990, Pickett et al., 1997). For example, patterning of urban habitats may be influenced by land-use planning decisions, which may themselves be influenced by the values and aspirations of local residents, combined with political and market pressures, and so on. In this case, to better understand habitat patch dynamics in urban areas, research would be needed to model the impact of, for example, economic cycles, as their effects cascade through the various levels of urban decision-making. This radical conceptual shift has been referred to as the ‗ecology of cities‘ (Grimm et al., 2000, Pickett et al., 2001), where cities themselves are treated as ecosystems and where the performance of particular natural or human components can only be properly understood by identifying their interactions with the broader urban context. One benefit from treating cities as systems is that it allows them to be approached from different theoretical perspectives. The concept of system resilience may be of particular value when exploring urban sustainability (Ahern, 2011), and whilst several definitions exist, it is used within this thesis to mean ‗the capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity, and feedbacks' (Walker et al., 2004). Resilience is a concept that is drawn from the study of ecological systems such as shallow lakes or savanna rangelands (Gunderson, 2000). Internal processes within these systems are able to prevent minor disturbances from fundamentally altering their state (referred to as their stability domain). Here I use the concept of resilience to explore the vulnerability of ecological features, processes and high-level functions within urban systems. Within papers II and V, I endeavour to identify the broader urban system conditions upon which various ecological processes and functions depend and to explore which dependencies are particularly sensitive to how the urban system evolves over time. In paper V, I then draw upon observations of resilience in natural ecosystems to identify strategies for future proofing the benefits of urban trees.

A conceptual framework for this thesis

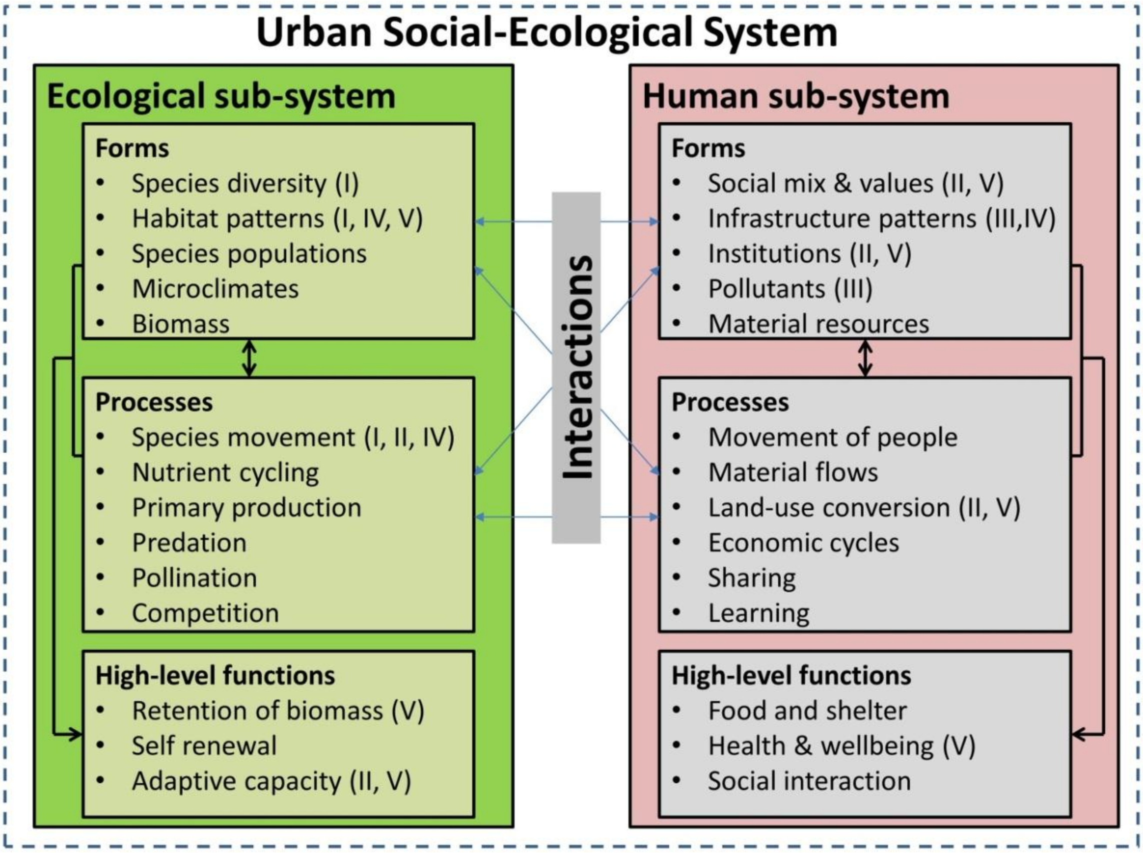
I have created a conceptual framework (Fig.1) to illustrate the human and ecological sub-systems that operate within an urban area, and to identify where the papers within this thesis contribute to knowledge about this social-ecological system. This figure is intended to emphasise the similarity between these two dimensions, and to highlight their potential to interact. I have used parentheses within this figure to indicate where my thesis papers fit within this framework. This figure has been informed by various conceptual models created to support a more integrated study of urban systems (Alberti et al., 2003) and to clarify the stocks and flows of materials through cities (Kennedy et al., 2011). Please note that the boundaries of these sub-systems are not defined, as they will necessarily vary depending on the research question being asked (Jax, 2005, Pickett et al., 2001).

Each sub-system has three elements: Forms, Processes and High-level functions.

- **Forms**. This element includes examples of the structure and composition of the sub-system. It can be thought of as a snapshot of the constituent parts of the urban system. It includes spatial patterns and some of the driver elements identified in the conceptual model of Alberti et al. (2003, Fig. 6), social institutions given in Pickett et al. (2001, Fig. 3), and resources listed in Kennedy et al. (2011, Fig. 2).
- **Processes**. These include nutrient cycling, the movement of people or materials, and social interactions. I included some of the drivers and processes from (Alberti et al., 2003), and this element captures the process of changing ecological conditions and human perceptions referred to by Grimm et al. (2000, Fig. 3).
- **High-level functions**. Within this model, processes interact with forms to create high-level functions. The use of the term *function* can be problematic in ecology, as it can have a variety of very different meanings (Jax, 2005). It is often used as a synonym for process, or to refer to the role that an individual component plays within the broader system. However, here we use it to denote higher level purposes or key outcomes of the sub-system as a whole

(Jax, 2005). Many of the intended functions of cities are clear – they provide a centralised access point to food, information and other resources, as well as security and opportunities for social interaction. Despite this, it is clear that they also have emergent properties and their high level functions may therefore change over time. In comparison, identifying the high-level functions of the ecological sub-system is more challenging. Natural ecosystems have not been designed for specific purposes, but have emerged as species and habitats have adapted to a set of biophysical conditions, or have been managed to meet human needs. I argue that the primary goal of natural systems is therefore to persist and that the high-level functions of ecosystems are those that are key to their persistence. For the ecological sub-system I suggest three high-level functions: the retention of biomass (e.g. via the process of primary production), self-renewal (e.g. via processes of pollination and succession) and adaptive capacity (e.g. via the maintenance of response diversity within the species pool).

Figure 1.



A conceptual model of an urban social – ecological system, divided into its ecological and human sub-systems. Roman numerals in parentheses indicate where the papers within this thesis fit within this model. For example, paper IV considers how patterns in tree cover (habitat) and lighting (infrastructure) combine to impact the process of movement.

PAPER SUMMARIES

This thesis is based upon five papers. Please note that I use the plural –we” within this section, as the papers that form this thesis were written with the support of co-authors from a broad variety of disciplinary backgrounds.

Paper I is a study of bat activity within the West Midlands conurbation of the United Kingdom. It combines a gradient and multi-scale approach to landscape analysis with a technique that explores habitat accessibility via tree networks. This study was motivated by the limited information on the impact of infill development and land-cover change on urban biodiversity. Studies along anthropogenic gradients have often lacked a clear description of how the gradient was defined (McDonnell and Hahs, 2008), and of the broader landscape context within which the data were collected. In addition, the spatial scale at which species respond to the landscape is vital information for conservation managers, particularly as species can select habitat at contrasting and multiple scales; yet estimates of habitat use are often made using a low number of scales due to practical considerations (Mayor et al., 2009). We therefore took care to stratify our sampling in a manner that could be replicated by others, and to measure clearly defined land-cover and land-use variables at multiple spatial scales.

Bats were surveyed at ponds located throughout the West Midlands. Site selection was informed by a GIS analysis of the land-cover and land-use within 1 km^2 circle surrounding candidate ponds. These ponds were then assigned to one of five landscape categories, and six ponds from each category were selected for surveys. Each site was surveyed three times, using two types of detector – a fixed detector that recorded the time and nature of bat calls for the entire night, and a handheld detector that was used to identify bat activity for 1.5 hours following sunset. These methods were complementary, with the fixed detector survey collecting more data, and the handheld detector allowing concentrations of bat activity in the vicinity of the pond to be surveyed more effectively. We identified broad correlations between bat activity and the physical structure and composition of the urban landscape. The analysis draws particular attention to the positive role that structural connectivity of tree cover appears to play for *Pipistrellus pipistrellus*, particularly in areas of high built-density. This species was also found to be particularly sensitivity to the nature of land-cover at a local scale, in comparison to the other species recorded. We also found a broadly negative impact of built surface cover on the urban bat community. However, *P. pipistrellus* activity exhibited a more complicated ‗humped' shape response, the cause of which is unknown. Whilst this study helps to fill a clear gap within urban gradient studies, the applicability of these results to fine-scale conservation efforts is still problematic, as the mechanism(s) by which high building-density appears to deter bats remain unclear. A key question that remains is whether the urban bat community can be enhanced simply by reducing levels of built density and planting more trees, or whether the underlying mechanisms that drive patterns in bat activity are more subtle and context specific?

Paper II explores the vulnerability of ecological features (such as artificial bat roosts) within an urban development case study in the city of Lancaster, UK. This is a relatively straightforward analysis of how ecological mitigation and enhancement is considered within an urban redevelopment proposal. However, it draws upon a variety of ideas about how urban systems can be conceptualised and tested. The site chosen for this study is called Luneside East, a post-industrial brownfield site near to the commercial centre of Lancaster. Sites like Luneside East can be found in many UK cities, being composed of abandoned and collapsed buildings, contaminated soils and encroaching vegetation. Such sites are often promoted by local and national government as development opportunities, yet they frequently contain species of conservation concern that are legally protected (Harrison and Davies, 2002). In the case of Luneside East, surveys by ecological consultants had revealed the presence of commuting or foraging *P. pipistrellus*, whilst some of the structures were also identified as potential winter roosts. Mitigation for potential roost losses, and enhancements to foraging habitats had been proposed by the developer, along with enhancements to bird nesting habitat. The central question within this paper is whether these ecological interventions would continue to function far into the future, once the development process and initial monitoring period had been completed.

To address this question we employed a systems approach, to identify the broader social, economic and environmental conditions upon which these interventions depended. For example, for an artificial bat roost to be of use, bats need to be able to commute from the roost to feeding areas in the surrounding landscape. We then questioned whether these conditions might be undermined in the future, using a set of future socio-political scenarios developed for UK urban areas. A resilient habitat, it is argued, is one that would continue to function despite radical changes to the values and behaviours of local residents, to the surrounding built form, to technology, or to the local economy. The resilience analysis highlighted several uncertainties over whether key habitat features would remain undisturbed, whether important microclimates would be preserved and whether functional habitat connectivity would be maintained. Some suggestions for improving the resilience of these features are made, such as locating artificial roosts in parts of the site where functional connectivity is less likely to be undermined. However, at the time of writing there was little empirical data on impact thresholds for factors such as artificial lighting, or on the sensitivity of *P. pipistrellus* to the fragmentation of urban tree cover. This paper serves to highlight the failure of current practice to consider the extent to which ecological mitigation and enhancement is sensitive to changes within and adjacent to the development site over time. In addition, it reveals important knowledge gaps about how protected species respond to fine scale environmental change within urban areas.

Paper III focuses on the subject of urban artificial lighting, which is known to have a range of ecological impacts (Rich and Longcore, 2006). We present the first ever spatial analysis of fine-scale multispectral lighting data for an entire city. High-resolution aerial night photography was captured for the city of Birmingham (UK) and analysed to identify spatial patterns in lighting associated with built density, grain of sample size and land-cover/land-use. We argued that a lack of high-resolution baseline lighting data for cities has prevented artificial lighting being included meaningfully within the impact assessments of development proposals (as illustrated in paper II). Such data are also essential for addressing a broad variety of research questions relating to human and ecological health, amenity and economic costs. We found positive relationships between artificial lighting and built density at coarse spatial scales, whilst at fine scales lighting varied depending on land-use. Of particular note was that industrial land uses are responsible for much of the bright lighting within the city, despite covering a relatively small area of land. This paper illustrates the value of generating city-wide data on ecological disruptors, as assumptions that street lighting is the dominant source of urban light pollution may not always be correct.

Paper IV draws together elements of the preceding papers to ask whether variations in artificial lighting and gaps in urban tree cover can combine to deter bat movement within urban areas. Again, the focus is on the city of Birmingham (UK). We were motivated to write this paper by the knowledge gaps around the fine-scale ecological impacts of lighting and urban form. These knowledge gaps had limited the detail at which the resilience analysis within paper II could be undertaken. In addition, we were keen to try and understand the mechanisms that were responsible for the low bat activity within heavily developed areas, described in paper I. This provoked us to undertake a more mechanistic and fine-scale study of lighting and bats, focusing on the potential impacts of street lighting on bat movement. This mechanistic approach then allowed us to scale the results to estimate landscape scale impacts.

Our broad hypothesis was that the commuting behaviour of bats such as *P. pipistrellus* would be impacted by the nature of gaps in urban tree lines. Bats were surveyed at gaps in tree cover, which varied in width and illumination level. Crossing events were recorded, and the probability of crossing was modelled as a function of gap width and illumination. We found that street lighting within gaps could create a barrier to bat movement. Importantly, this impact was found to be context dependent, as the barrier effect of lighting varied depending on the size of the gap. Models derived from this fieldwork were then applied to high-resolution tree and lighting data for the whole of Birmingham (derived from papers I and III), predicting a high resistance to movement in heavily developed areas. Finally, in recognition that urban systems are dynamic, often unpredictable and dominated by human activities, we generated different scenarios for urban lighting, to explore how landscape resistance might be impacted in the future. One interpretation of these results is that functional connectivity for bats need not always be poor within urban centres, and that improvements could be made simply through the strategic dimming of street lights and narrowing of gaps

Paper V considers the topic of urban tree cover and the factors which may influence its persistence and performance over time. Its focus is on UK urban areas, although it draws on a global literature on the potential benefits of urban trees to local residents, the impact of landscape context on the delivery of these benefits and on how urban tree cover has varied in the past. We have focused upon urban trees as these could be thought of as the archetypal urban sustainability intervention. They have multiple positive functions for human wellbeing and have the potential to facilitate more biodiverse cities (see papers I and IV), yet these benefits may take many decades to be realised. Relatively little is known about the factors that impact the delivery of benefits from urban street trees over large timescales, and this analysis is intended to highlight and challenge the social, environmental, and economic assumptions that are implicit within large-scale urban tree planting projects.

This study employs the same resilience analysis methods used in paper II, but in comparison this analysis was possible at a much finer level of detail. First, a range of potential benefits were identified through workshops and a literature review. For each intended benefit, the broad system conditions upon which it depends were then identified (e.g. aesthetic benefits may depend not just on the tree being visible, but also upon the values and past experiences of local residents). Finally, a set of future urban scenarios were used to test whether these conditions might be undermined in the future. These scenarios have the advantage over alternative approaches that use predictions of risks based upon trend analysis (e.g. in relation to climate change), as they also include socio-political changes that are much less predictable. We argue that by focusing upon the system conditions that these benefits depend, rather than the tree itself, we are better able to examine the underlying mechanisms that support the resilience of these benefits.

We find that many benefits appear to be dependent on continued levels of tree maintenance, on public values which are supportive, and on a built form which is relatively stable over time. For example, large trees and extensive canopy cover are important to the delivery of urban cooling and biodiversity goals, yet they create a range of conflicts with built infrastructure and human activities that need to be managed over long timescales. In future scenarios where funding for the management of urban street trees is not considered a priority, one might expect to see a reduction in the abundance of large trees within urban areas. We conclude by suggesting some changes that may improve the resilience of urban trees and their benefits, including the use of targeted payments for the retention of trees within priority neighbourhoods, and a more formal integration of members of the public, NGOs and municipal departments in the co-management of street trees.

This work can be seen as complementary to papers I and IV which flag up the importance of urban tree cover to the urban bat community, as it could be used to test the vulnerability of related urban habitat enhancement proposals. It also makes a broader point about the short-term focus of ‗million tree‘ planting projects, and the need to query whether such projects have fully considered the timescales required to deliver the benefits that are often promised.

CONCLUSION

Much of the discussion about sustainability and cities over recent decades has focused on the performance of particular classes of urban form, e.g. compact vs. sprawling cities (Gordon and Richardson, 1997, Burton, 2000, Neuman, 2005). This interest may well be a reaction to changes in urban development policy (Dallimer et al., 2011, Williams, 1999, Harrison and Davies, 2002) or to the consistency with which certain types of development are viewed to have failed in the past (Jacobs and Manzi, 1998). However, it is important to be alert to the danger of assuming that there are deterministic relationships between urban form and sustainability performance. Such an assumption appears to be common within the field of urban design, where the goal is often to reach a single idealised urban form (Ahern, 2013). This approach fails to recognise the dynamic nature of cities, and that sustainability goals may necessarily change over time. In the context of this thesis, I would argue that simply because low levels of bat activity were associated with heavily developed locations, it does not mean that these sites will always perform so poorly. Alternative states might well be possible where high chiropteran diversity could be sustained within urban centres, through a combination of strategic habitat creation and a reduction in disturbance from lighting and other factors currently associated with densification. Crucially, the persistence of more biodiverse cities may depend not just upon radical physical changes to the urban form, but also on a reinforced shift in the values, knowledge and expectations of local residents, workers and landscape managers.

The key aims for this thesis were to:

1. Describe the spatial patterning of the bat community within a UK city and identify the land-cover and land-use variables with which they are associated.
2. Explore the underlying mechanisms that shape these patterns.
3. Identify broad urban system conditions upon which the bat community depends.

I am confident that when taken as a whole, the papers that form this thesis have addressed each of these aims. Aim 1 was covered within in paper I, where stratified surveys of the urban bat community were undertaken and modelled against landscape variables. The patterns identified within this paper also highlighted potential causative mechanisms (Aim 2), namely movement via tree cover and disruption by urban intensification (possibly due to lighting – paper III). These mechanisms were explored in paper IV, through a more mechanistic study on the impact of gaps in tree cover and of lighting, on bat commuting behaviour. Aim 3 was addressed within papers II and V. The system conditions that the functioning of artificial bat roosts and new feeding areas depended upon were identified in paper III, along with their vulnerabilities. A similar approach was then applied to urban trees in general (V), which are a key habitat component and predictor of activity for several urban bats (paper I).

Areas for future research

A key aim of paper I was to survey the entire bat community, yet this was extremely challenging in practice. The calls (and ecology) of three UK bat species are so similar that we decided to treat them as a single guild for the purposes of analysis, but this is not an ideal outcome. In addition, other species such as the brown long-eared bat *Plecotus auritus* are known to be present in urban areas (Altringham, 2003), yet their calls are of such low volume that detector based surveys are rarely of use and this species was not recorded in this study. A more complete analysis might have been accomplished by trapping bats in flight, although logistically this would be extremely challenging. Improvements in detector technology and call analysis software (Obrist et al., 2004) would also be expected to facilitate a more complete survey of the urban bat community in the future.

The land-cover and land-use variables that were used within paper I were selected from mapping and remotely sensed data that were broadly available for UK urban areas at that time. This has the practical benefit that the results might be more easily applied to other cities. However, it is possible that alternative explanatory variables might have been more appropriate. For example, the best model of evening activity for *Pipistrellus pipistrellus* included the area of gardens within a 50m radius. Whether it was the gardens themselves that are beneficial (e.g. as feeding habitat) or whether garden area is actually an indicator of local roost availability in residential buildings is unclear. The causes of such associations need to be understood, and in this example the use of landscape data on building age, condition and thermal performance may well be beneficial for future studies. The extent and grain of analysis employed in urban landscape ecology studies can make a huge difference to the relationships detected between urban land-cover and species richness (Luck, 2007) and to models of habitat use (Mayor et al., 2009). Our analysis demonstrates the value of a multi-scale approach, as it highlights how different components of the urban bat community are sensitive to the composition of the landscape at different scales.

At the time of writing papers III and IV, there was much suspicion amongst bat researchers and practitioners that artificial lighting was detrimental to bat activity, yet very limited empirical data was available to support this. Paper III was particularly timely, as it was published at approximately the same time that the EU Loss of the Night COST Network (LoNNe) was established. It was therefore able to support a broader discussion about how best to measure light pollution, about where light pollution was most intense within cities, and which species and social groups were subjected to the greatest exposure. It also facilitated the mechanistic approach employed in paper IV, responding to calls for mechanistic studies that can be linked to ecosystem-scale environmental changes (Shochat et al., 2006). However, it is clear that a wide range of other mechanisms could be shaping the patterns we observed. For example, it would be interesting to explore how predation, competition and prey availability varied along this urbanisation gradient, and to undertake experimental manipulation of these factors.

Papers II and V captured a range of system conditions that are likely to impact the functioning of urban bat communities, yet the scenarios used to test their resilience still contain many assumptions in relation to human behaviour. Paper V identifies several studies that considered public attitudes to urban tree cover, which may be key to determining its long-term persistence. What became clear during the writing of this paper was that attitudes seem to vary widely, depending upon factors such as age of residents, household size, tenancy turn-over rates, employment and income (Zhang et al., 2007, Conway and Bang, 2014). Given the complexity and apparent context sensitivity of public attitudes to ecological features within urban areas, this subject would clearly benefit from more attention, with greater integration of ecological and social-science approaches (Wu, 2014).

If the ultimate goal of ecological research is to support conservation practice (which is my personal view), an important question is whether these research findings are of value to practitioners? Perhaps the most useful outcome of this thesis is the model presented in paper IV, as this could be used to create resistance surfaces for other cities, given data on lighting and tree cover. However, it should be noted that the data used to generate this model was collected from a single city and that we cannot assume it can be reliably applied elsewhere, particularly to cities outside of the UK. The need for comparative ecological studies involving multiple cities is well recognised (McDonnell et al., 2009), and I look forward to undertaking fieldwork in some more exotic cities in the future!

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PAPERS

List of papers

I. HALE, J. D., FAIRBRASS, A. J., MATTHEWS, T. J. & SADLER, J. P. 2012. Habitat Composition and Connectivity Predicts Bat Presence and Activity at Foraging Sites in a Large UK Conurbation. PLoS ONE, 7, e33300.

II. HALE, J. D. & SADLER, J. P. 2012. Resilient ecological solutions for urban regeneration. Engineering Sustainability, 165, 59-67.

III. HALE, J. D., DAVIES, G., FAIRBRASS, A. J., MATTHEWS, T. J., ROGERS, C. D. & SADLER, J. P. 2013. Mapping lightscapes: spatial patterning of artificial lighting in an urban landscape. PloS ONE, 8, e61460.

IV. HALE, J. D., FAIRBRASS, A. J., MATTHEWS, T. J., DAVIES, G. & SADLER, J. P. 2015. The ecological impact of city lighting scenarios: exploring gap crossing thresholds for urban bats. Global Change Biology, In Press.

V. HALE, J., PUGH, T., SADLER, J., BOYKO, C., BROWN, J., CAPUTO, S., CASERIO, M., COLES, R., FARMANI, R., HALES, C., HORSEY, R., HUNT, D., LEACH, J., ROGERS, C. & MACKENZIE, A. 2015. Delivering a Multi-Functional and Resilient Urban Forest. Sustainability, 7, 4600-4624.

Author contributions

James Hale was the lead author on all publications and in each case undertook at least 60% of the data collection, analysis and manuscript preparation (typically more). Specific contributions of co-authors are as follows for each paper:

I. James Hale and Jon Sadler wrote the paper. Alison Fairbrass and Thomas Matthews contributed to writing the manuscript. James Hale and Jon Sadler conceived and designed the field surveys. James Hale, Alison Fairbrass and Thomas Matthews undertook the field surveys. James Hale and Jon Sadler analysed the data.

II. James Hale and Jon Sadler wrote the paper. James Hale undertook the analysis.

III. James Hale and Jon Sadler wrote the paper. Gemma Davies, Alison Fairbrass, Thomas Matthews and Christopher Rogers contributed to writing the manuscript. James Hale, Alison Fairbrass and Thomas Matthews undertook the field surveys. James Hale undertook the analysis.

IV. James Hale and Jon Sadler wrote the paper. Alison Fairbrass, Gemma Davies and Thomas Matthews contributed to writing the manuscript. James Hale and Jon Sadler conceived and designed the field surveys. James Hale, Alison Fairbrass and Thomas Matthews undertook the field surveys. James Hale, Gemma Davies and Jon Sadler analysed the data.

V. James Hale, Rob MacKenzie, Thomas Pugh and Jon Sadler wrote the paper. Silvio Caputo, Richard Coles, and Russell Horsey provided additional text and substantial feedback on an early manuscript draft. Christopher Boyko, Julie

Brown, Silvio Caputo, Maria Caserio, Richard Coles, Raziyeh Farmani, James Hale, Chantal Hales, Dexter Hunt, Joanne Leach, Rob MacKenzie, Thomas Pugh, Christopher Rogers and Jon Sadler generated the data on benefits and necessary conditions and initiated the resilience analysis. James Hale and Thomas Pugh led the analysis.

PAPER I. HABITAT COMPOSITION AND CONNECTIVITY PREDICTS BAT PRESENCE AND ACTIVITY AT FORAGING SITES IN A LARGE UK CONURBATION

James D. Hale[1], Alison J. Fairbrass[2], Tom J. Matthews[3], Jon P. Sadler[1]

1. School of Geography, Earth and Environmental Sciences, The University of Birmingham, Birmingham, United Kingdom
2. Imperial College London, Silwood Park Campus, Ascot, United Kingdom
3. School of Geography and the Environment, University of Oxford, Oxford, United Kingdom

Abstract

Background: Urbanisation is characterised by high levels of sealed land-cover, and small, geometrically complex, fragmented land-use patches. The extent and density of urbanised land-use is increasing, with implications for habitat quality, connectivity and city ecology. Little is known about densification thresholds for urban ecosystem function, and the response of mammals, nocturnal and cryptic taxa are poorly studied in this respect. Bats (Chiroptera) are sensitive to changing urban form at a species, guild and community level, so are ideal model organisms for analyses of this nature.

Methodology/Principal Findings: We surveyed bats around urban ponds in the West Midlands conurbation, United Kingdom (UK). Sites were stratified between five urban land classes, representing a gradient of built land-cover at the 1 km^2 scale. Models for bat presence and activity were developed using land-cover and land-use data from multiple radii around each pond. Structural connectivity of tree networks was used as an indicator of the functional connectivity between habitats. All species were sensitive to measures of urban density. Some were also sensitive to landscape composition and structural connectivity at different spatial scales. These results represent new findings for an urban area. The activity of *Pipistrellus pipistrellus* (Schreber 1774) exhibited a non-linear relationship with the area of built land-cover, being much reduced beyond the threshold of ~60% built surface. The presence of tree networks appears to mitigate the negative effects of urbanisation for this species.

Conclusions/Significance: Our results suggest that increasing urban density negatively impacts the study species. This has implications for infill development policy, built density targets and the compact city debate. Bats were also sensitive to the composition and structure of the urban form at a range of spatial scales, with implications for land-use planning and management. Protecting and establishing tree networks may improve the resilience of some bat populations to urban densification.

1. Introduction

Fifty years of agricultural intensification, fragmentation and urbanisation have radically altered the landscape composition of the UK (Robinson and Sutherland, 2002). Urban areas have grown substantially over the last 20 years (Grimm et al., 2008) and now support the majority of the global population (UN, 2010). Urbanisation is characterized by an increase in sealed land-cover density (McKinney, 2002), geometric complexity, fragmentation of land-use patches (Luck and Wu, 2002, Zhang et al., 2004), and a reduction in patch size (Zhang et al., 2004, Sadler et al., 2006). Combined with varying disturbance levels (Williams et al., 2009), this results in a spatio-temporally complex land-use mosaic (Cadenasso et al., 2007, McDonnell and Pickett, 1990), with far-reaching consequences for species dispersal (Tremblay and St Clair, 2009), ecological function (McDonnell et al., 1997) and ecological service provision (Eigenbrod et al., 2011). In some countries, urbanisation has resulted in urban sprawl into agricultural land (Irwin and Bockstael, 2007), while in others policies favour compact city forms (Burton, 2000). Where greenbelts constrain urban sprawl (Sadler et al., 2010), there is evidence that urban landscapes have densified and lost greenspace, especially over the last decade (Dallimer et al., 2011). Little data exist that indicate how much densification the urban ecosystem can withstand before ecosystem function is substantially impaired. In terrestrial habitats increased urbanisation generally has a negative effect on species richness, although this pattern is not universal (McKinney, 2008). Organism responses to increasing urban land-cover are species and trait-specific, but generally differentiate between generalist species that thrive or show humped abundance patterns, and specialist species that exhibit declines (Williams et al., 2009, Bates et al., 2011, Tratalos et al., 2007, Croci et al., 2008, Evans et al., 2011, Niemela and Kotze, 2009).

Urban density thresholds for species presence and abundance are likely to be contingent upon sampling methodology and the spatial scale at which built density and landscape composition are measured. There is currently a multiplicity of approaches evident in the literature (Hahs and McDonnell, 2006, McDonnell and Hahs, 2008) clarification is needed in order to improve comparability between studies and to aid the translation of results into conservation practice. Clarity may be gained by studying taxa whose species are sensitive to different measures of urbanisation at a range of spatial scales, as well as to the surrounding landscape. There is already a considerable literature on birds and urbanisation (Tratalos et al., 2007, Evans et al., 2011, McDonnell and Hahs, 2008), but their life histories and responses to urbanisation do not always reflect those of other groups (Gagne and Fahrig, 2007). Mammals, nocturnal and cryptic taxa are poorly studied in this respect and bat (Chiroptera) communities are ideal candidates for research. They typically include species that exploit built structures (Altringham, 2003) are sensitive to landscape scale, patch effects (Gehrt and Chelsvig, 2003, Gehrt and Chelsvig, 2004), and to changes in structural connectivity (Verboom and Huitema, 1997, Gaisler et al., 1998). The few studies focusing on the effect of urbanisation on bat species indicate variability in response to changing urban form at a species, guild and community level. In studies of cities in the Czech Republic (Gaisler et al., 1998), Mexico (Avila-Flores and Fenton, 2005) and Australia (Threlfall et al., 2012) bat activity was lower in high density residential areas, than in low density areas (e.g. suburban, urban fringe) and semi-natural areas. In addition, lower species richness was reported in the urban centre and densely developed areas. This contrasts with other studies in the USA (Gehrt and Chelsvig, 2003, Gehrt and Chelsvig, 2004), where positive relationships have been reported between both overall bat activity and species richness of natural habitat fragments and the urban density of the surrounding landscape. Several studies have identified positive relationships between urbanisation and the activity of certain species (Gehrt and Chelsvig, 2003, Gehrt and Chelsvig, 2004, Avila-Flores and Fenton, 2005, Lookingbill et al., 2010) but other species clearly favoured semi-natural areas or exhibited a broad tolerance of urbanisation (Gaisler et al., 1998, Avila-Flores and Fenton, 2005). It has been suggested that these responses reflect differences in wing and call morphology, with species specialising in cluttered habitats avoiding brightly lit and poorly vegetated urban areas (Avila-Flores and Fenton, 2005).

Given the intensity of compositional change in urban areas (Grimm et al., 2008, Dallimer et al., 2011), and the associated high levels of fragmentation (Luck and Wu, 2002), one might expect that connectivity and linkage would be a central theme in urban ecology, as it is in other landscapes (Robinson and Sutherland, 2002, Boughey et al., 2011, Cornulier et al., 2011, Lawton et al., 2010). Indeed, connective features such as green networks and corridors have been influential in guiding city planning in many areas of the world (Turner, 2006, Fleury and Brown, 1997), and the creation and preservation of wooded corridors does seem to present an ideal opportunity for restoration aimed at enhancing spatial population resilience in cities (Marzluff and Ewing, 2001). However, there are very few studies that focus on this element. Studies on plants and invertebrates (Angold et al., 2006, Small et al., 2006) in UK greenways identified multiple structural and functional roles that were species specific in terms of habitat provision, but did not indicate a strong functional conduit role that enhanced movement and dispersal. Although evidence that wooded linear features such as streets and riparian corridors facilitate connectivity for birds in urban areas (Fernandez-Juricic, 2000, Shanahan et al., 2011), there are few studies pertaining to urban bats, although several have identified relationships between linear features and bat activity in agricultural areas (Verboom and Huitema, 1997, Limpens, 1991, Downs and Racey, 2006). Such features appear to have roles in both feeding and movement and thresholds for loss of functional connectivity are still unclear (Lookingbill et al., 2010, Oprea et al., 2009).

Here we explore the influence of urban landscape composition and structural connectivity on the presence and activity of bats at a range of spatial scales. We stratified sampling sites evenly across classes of urban form whose composition and extent were clearly defined a priori using a wide range of environmental data captured in a Geographical Information System (GIS). Foraging sites with similar local land-cover were selected to reduce the effect of confounding local variation in habitat type, and their landscape context measured consistently at multiple spatial scales. A proxy measure of functional connectivity was developed for each scale based on the traits of the species encountered. Both walking surveys and fixed position detectors were used to record bat activity. Using the assemblage and environmental data we addressed the following research objectives: (1) To characterize bat activity and presence in relation to urban density and landscape composition; (2) To assess the spatial scale at which species respond to the urban landscape; (3) To establish the significance of connectivity for bat activity in a heavily urbanised landscape.

We achieved our objectives and explore in the Discussion section the issues surrounding quantifying land-cover, land-use, functional connectivity and species specific responses to landscape change. Despite some progress with mapping key variables at a high spatial resolution and large spatial extent, our proxies for roost potential (large trees and residential buildings) and lighting (road area) did not add any explanatory power to the models and could be improved on. Future studies may benefit from data on tree species, building age and densities of lighting columns.

2. Results

We recorded bat calls within a total of 14,176 survey minutes using the fixed-point automatic detector (see section 5. Materials and Methods). Of these, 11,545 minutes contained calls identifiable to species or guild level. These included 9,950 active minutes (86% of the identifiable bat calls) of *P. pipistrellus* calls, 1,330 (11%) of *P. pygmaeus* and 345 (3%) belonging to the NSL (*Nyctalus noctula, Eptesicus serotinus, Nyctalus leisleri*) guild, with some minutes including calls from more than one species/guild. Walking surveys added an additional 1178 active minutes (75%) for *P. pipistrellus*, 190 (12%) for *P. pygmaeus*, 49 (3%) for the NSL guild and 163 (10%) Myotis calls, all of which where from bats observed feeding over the pond surface and were therefore confirmed as *Myotis daubentonii*. Bats were recorded at all of the thirty survey sites. Only *P. pipistrellus* was recorded at all sites, although *P. pygmaeus* was recorded at 93.3% of sites. The NSL guild was recorded at 73.3% of sites but was virtually absent from those within the Dense Urban land class (Fig. 1). *M. daubentonii* was present at 33.3% of sites and was negatively associated with increased urbanisation (Fig. 1). Full-night activity for both *P. pipistrellus* ($p = 0.043$) and the NSL guild ($p = 0.035$) was significantly higher in the Rural compared to the Dense Urban land class (Fig. 1). Evening activity for *P. pipistrellus* and the NSL guild followed similar patterns, although the differences between classes were not significant.

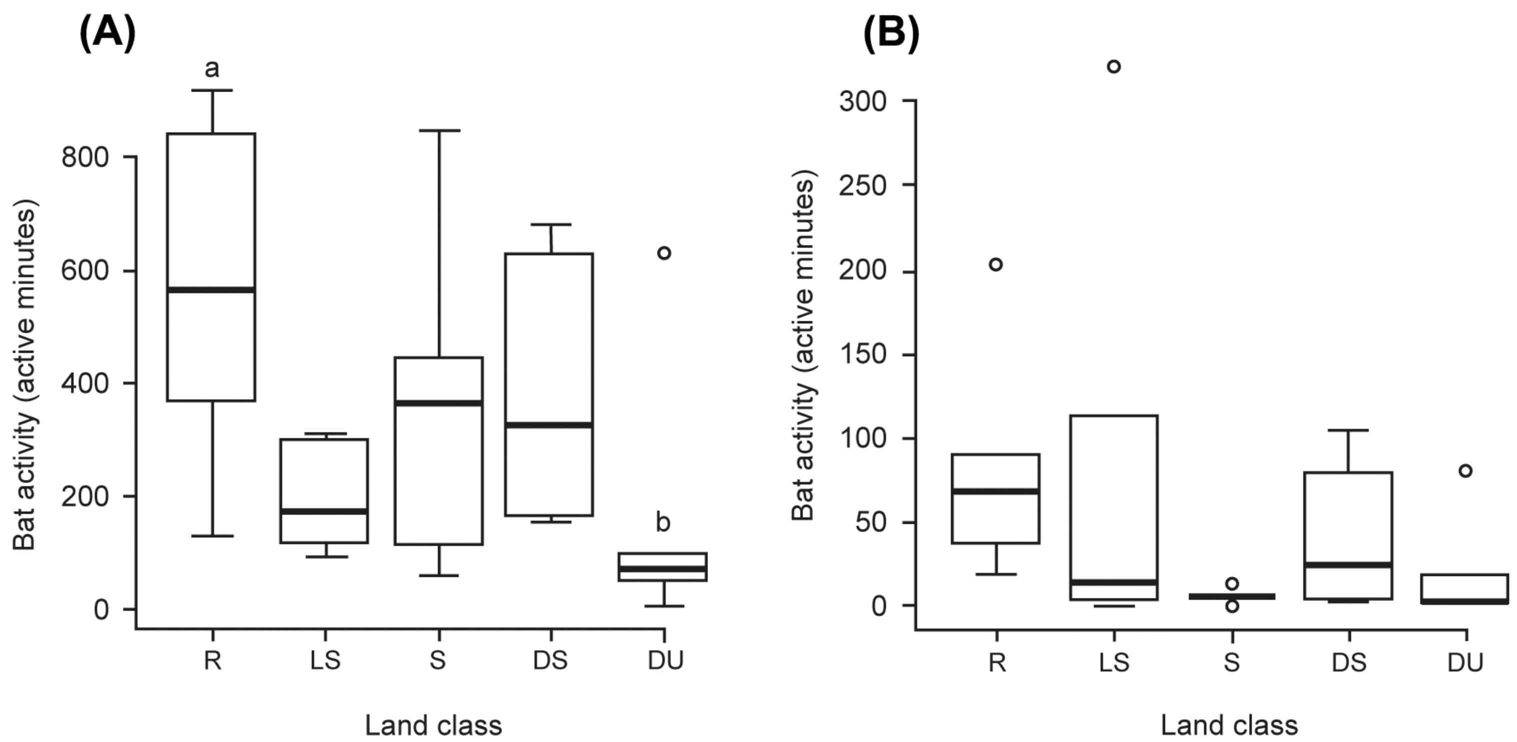


Figure 1. Bat activity adjacent to survey ponds based on full-night survey data. Land classes follow an urbanisation gradient from Rural (R) to Dense Urban (DU). Box plots represent total active minutes for (A) *P. pipistrellus*, (B) *P. pygmaeus*. Boxes that do not share a letter showed significant differences ($P<0.05$) between land classes.

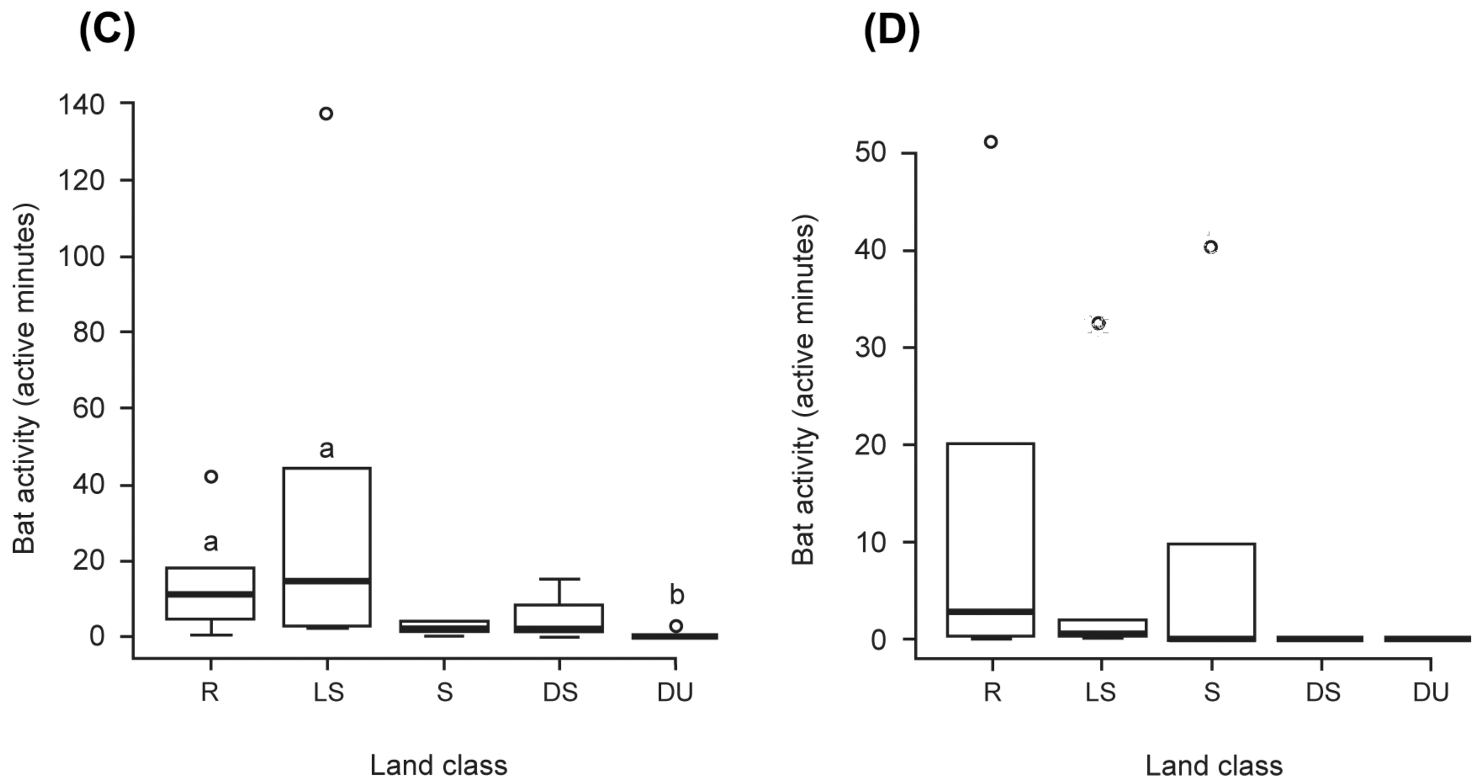


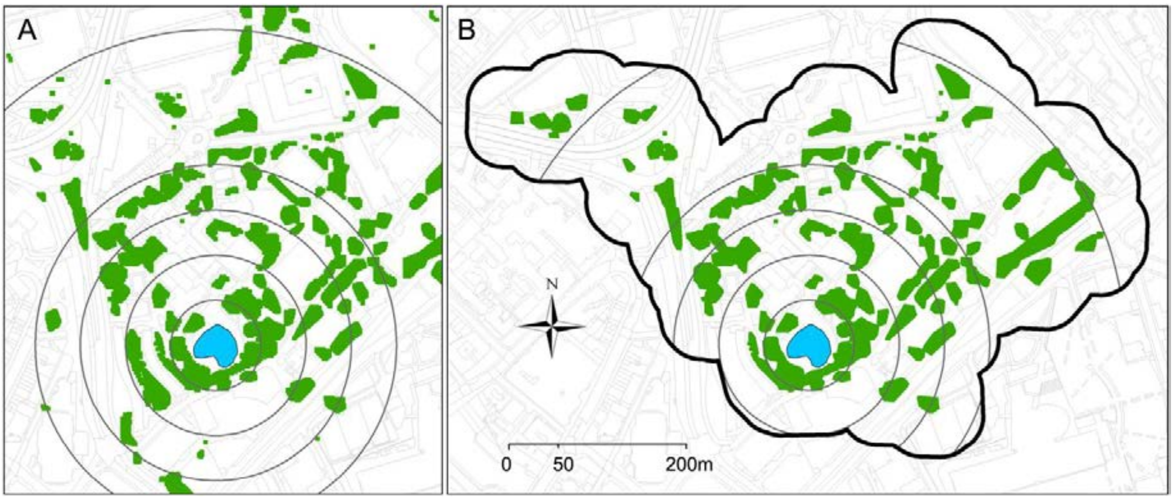
Figure 1 ctn. Bat activity adjacent to survey ponds based on full-night survey data. Land classes follow an urbanisation gradient from Rural (R) to Dense Urban (DU). Box plots represent total active minutes for (C) a group comprising *N. noctula*, *E. serotinus, N. leisleri* and (D) *M. daubentonii*. Boxes that do not share a letter showed significant differences (P<0.05) between land classes.

Multiple models were created for all species and guilds and the best of these are presented in Table 1. These models included data extracted using concentric buffers applied both to the landscape around each pond (concentric landscape) and restricted to the landscape intersected by a connectivity mask (connected landscape) (Fig. 2). In general, *P. pipistrellus* activity was highest at sites surrounded by low or moderate levels of built land-cover or at well-connected sites in highly urban areas. The best models all included the area of built land-cover within 350 m of survey ponds, with activity peaking at intermediate levels of built land-cover and being lower but more variable at high levels of urbanisation (Fig. 3). These models included a positive association with connected tree cover (>6 m high) within a radius of 150 m for sites in Dense Urban and Dense Suburban land classes (Fig. 4). Evening activity was also positively associated with connected garden area within both 50 and 500 m. These garden parameters were not present in the full-night model, but were replaced by a measure of connected vegetation cover within 200 m of the site (Table 1).

Pipistrellus pygmaeus activity was highest at ponds located within a highly vegetated landscape but poorly connected at a local level. All models included negative parameter coefficients for connected tree cover (>6 m high) within a radius of 100–200 m. Evening models included a positive relationship between activity and connected vegetation within 1 km, and the full-night models included a negative association with the area of built land-cover within 1 km.

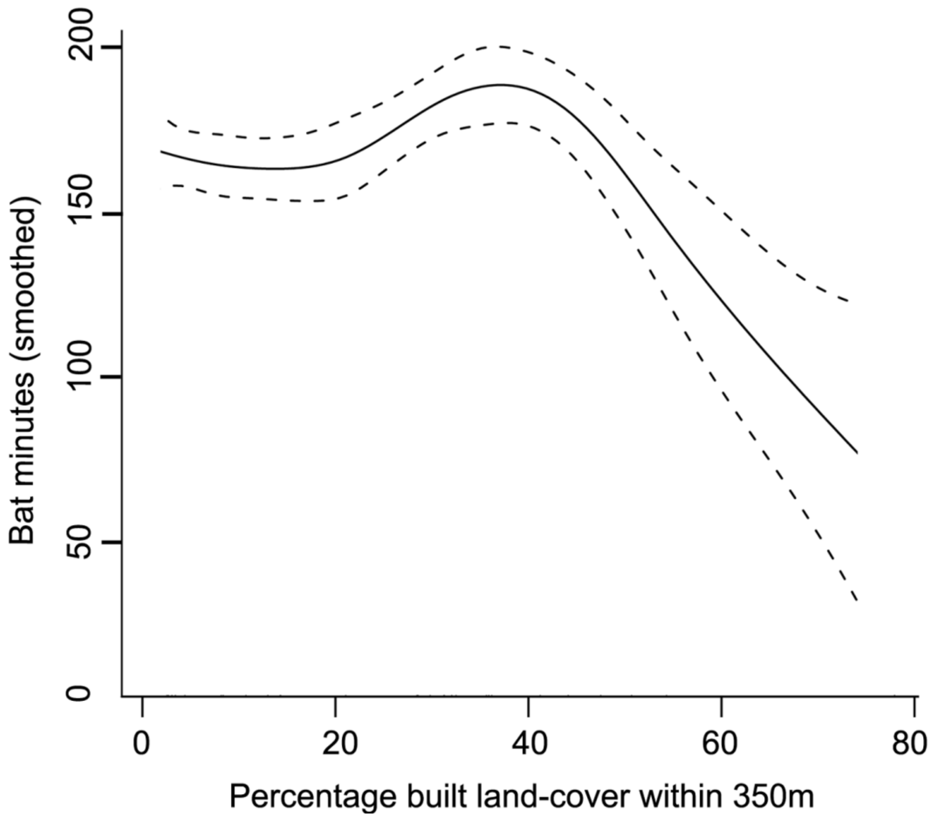
No valid GLM or GAM activity models were identified for *M. daubentonii*, or the NSL guild, but several logistic regression models were selected for evening presence data. The evening presence model for *M. daubentonii* included a positive relationship with natural land-cover (concentric) within 150 to 350 m of the survey sites (Table 1). For the NSL guild evening activity was negatively associated with built land-cover (concentric) within 350–750 m (Table 1). For these species, restricting the landscape analysis to the areas adjacent to tree networks (connected landscape) did not result in any valid models. Candidate all-night NLS presence-absence models all exhibited residual spatial patterning (e.g. Figure S1). We used a spatial correlation structure to compensate for this, but it did not improve either the residual spread or the AIC of the models, so all the models were rejected.

Figure 2.



A survey pond and two methods used to extract landscape data at multiple scales.
(A) An unrestricted extraction of landscape data using concentric buffers. (B) Connected (available) landscape mask created using tree networks buffered by 50 m. This polygon was used as a mask to restrict the landscape analysis to the area within this network. In both examples, landscape data were extracted at distances of 50, 100, 150, 200, 350 and 500 m from the pond centre.

Figure 3.

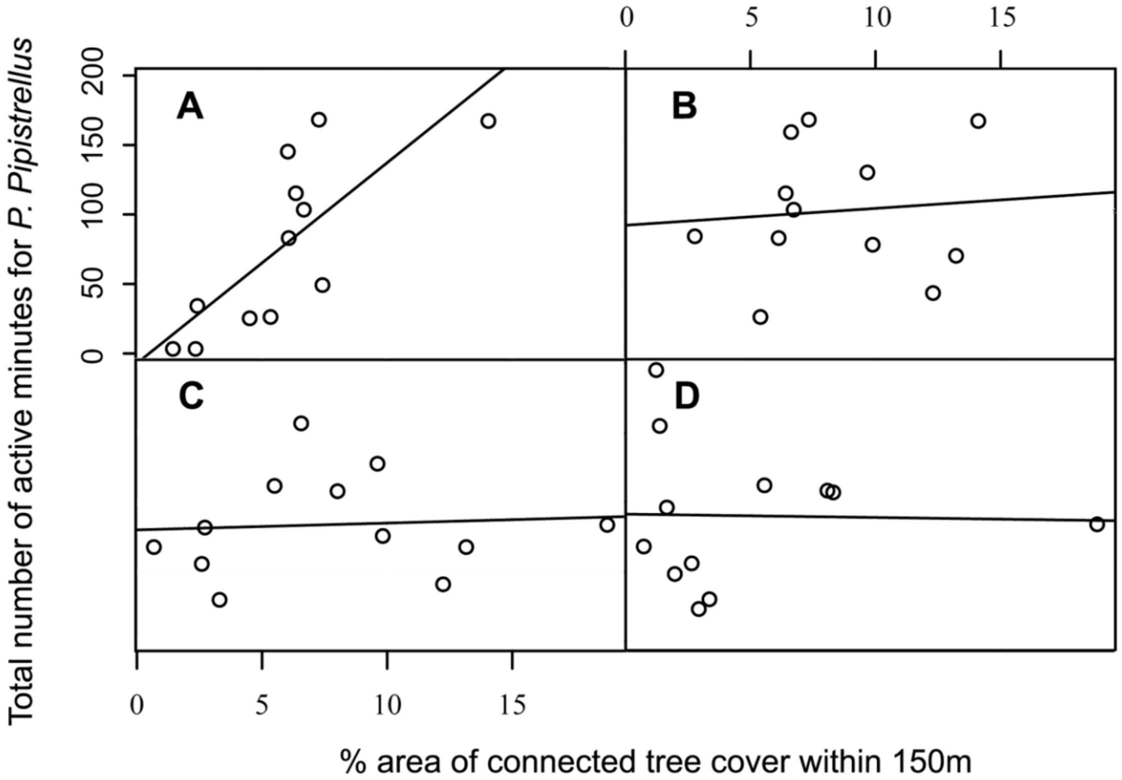


Partial plot of smoothed evening bat activity and percentage built land-cover within 350 m of surveys sites. This was included in the final model for evening activity of *P. pipistrellus* (see Table 1 for full model).

Res	Dur	Model type	Landscape variables and spatial scales (radii in m)								AIC
			50	100	150	200	350	500	750	1000	
PP	Ev	GAM Ne	Gard ***		Tree>6m ***s(Urb)		Bu***s‡	Gard*			283.6
PPy	Ev	GLM Ne		-Tree>6m **						Ve**	187.8
My	Ev	GLM Bi				Nat**‡					32.2
My	Ev	GLM Bi						Nat**‡			32.8
My	Ev	GLM Bi			Nat**‡						33
NSL	Ev	GLM Bi						-Bu**‡			35.2
NSL	Ev	GLM Bi							-Bu**‡		35.2
NSL	Ev	GLM Bi								-Bu**‡	35.3
PP	F-N	GAM Ne			Tr>6m ***s(urb)	Ve*	Bu***s‡				364
PPy	F-N	GLM Ne			-Tr>6m **					-Bu*‡	270.3
PPy	F-N	GLM Ne				-Tr>6m **				-Bu*‡	270.7

Table 1. Summary of the best-fit multi-scalar models for measures of bat activity and presence. ‡ indicates data extracted using simple concentric circular buffers, otherwise, a connectivity mask (a 50m buffer around tree networks) was used. * = P<0.05, ** = P<0.01, *** = P<0.001. The most parsimonious model for each response variable is shown, according to the Akaike Information Criterion (AIC) values. Resp = response group. PP indicates *P. pipistrellus,* PPy indicates *P. pygmaeus,* NSL a group comprising *N. noctula, E. serotinus, N. leisleri.* My indicates *M. daubentonii.* Dur = duration of sampling, where Ev = evening and F-N = full night. Models whose AIC values are =<2 of the optimum model are also included. Model type is either a generalised linear model (GLM) or a generalised additive model (GAM) with either negative binomial (Ne) distribution for activity data or binomial (Bi) for presence data. See Table 2. For variable definitions.

Figure 4.

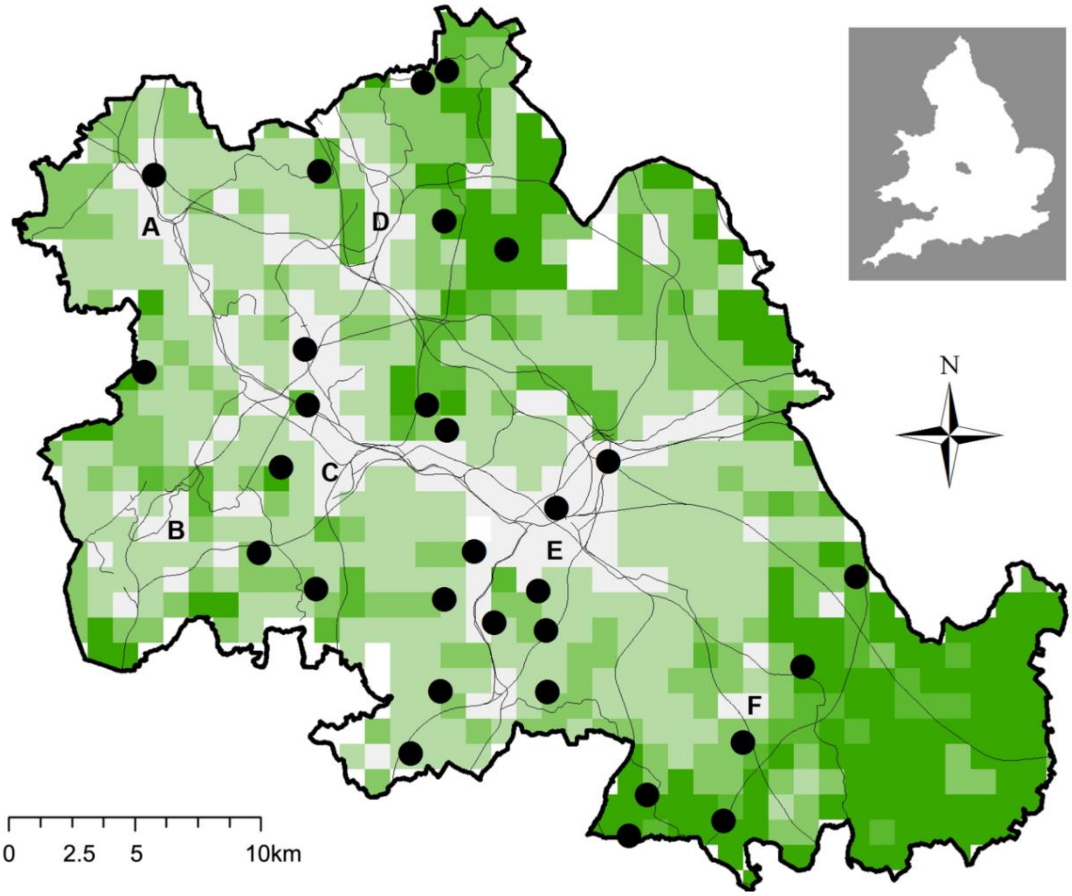


Conditional scatter plot of *P. pipistrellus* evening activity against connected tree cover (>6 m) within 150 m. (A) sites within Dense Urban and Dense Suburban (B) Dense Suburban and Suburban (C) Suburban and Light Suburban, (D) Suburban and Rural land classes.

3. Discussion

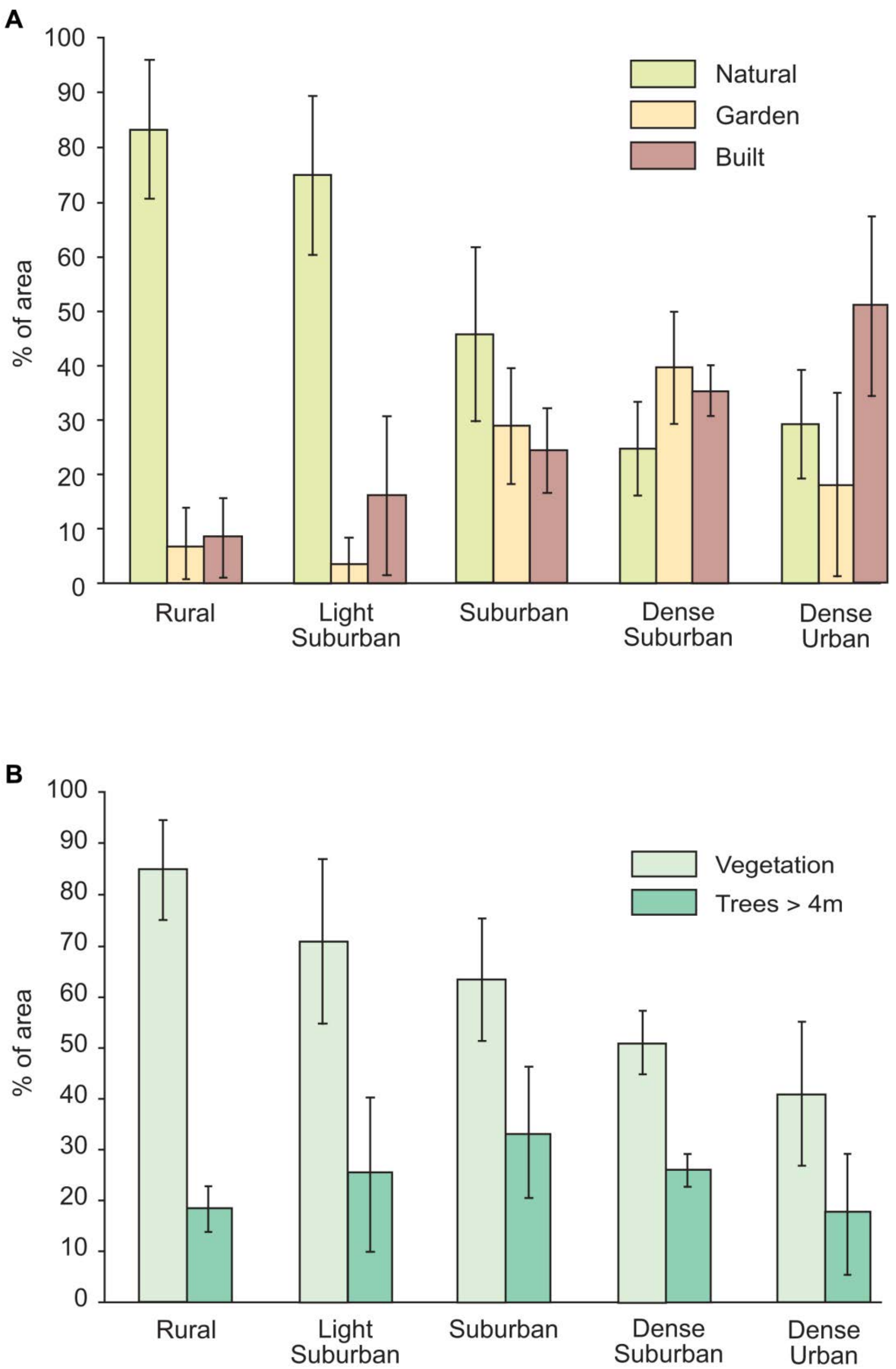
We investigated the response of a bat community to urbanisation, landscape composition and structural connectivity at a variety of spatial scales using standardized samples across five urban landscape classes (Figs. 5 & 6), targeting small ponds with consistent levels of adjacent riparian woodlands. All species were found to be sensitive to at least one measure of urbanisation and some were additionally influenced by landscape composition and structural connectivity at different spatial scales. For two species, habitat associations differed between evening and full-night models.

Figure 5.



The West Midlands metropolitan county study area. It includes the metropolitan borough centres of A Wolverhampton, B Dudley, C Sandwell, D Walsall, E Birmingham and F Solihull. Bat survey ponds are indicated by a black circle and were stratified by urban land classes, which are represented by a grid of 902×1 km^2 pixels covering the study area. These range from Dense Urban (white) to Rural (dark green). Canals and railways are indicated by fine black lines.

Figure 6.



Landscape data summaries for 1 km^2 circles surrounding each pond. (A) Mean area and SD for 3 of the 13 land-cover and land-use types derived from the Ordnance Survey Mastermap and used to assign ponds to land classes (see Table S1). (B) Mean area and standard deviation for vegetation cover and trees>4 m high.

3.1 Urban Density

Three broad measures of urbanisation were derived for this study (i) the area of built (sealed) land cover parcels derived from OS digital data, (ii) the area of natural (vegetated) land-cover parcels and (iii) the area of vegetated (remotely sensed) land-cover (Table 2). It is notable that all of the species/guilds in our study were found to be sensitive to at least one of these measures of urban density, given the limited evidence on the response of bat communities to urbanisation (Gehrt and Chelsvig, 2003, Oprea et al., 2009). Each measure provided significant explanatory power to different models, supporting calls for the use of multiple measures of urbanisation in gradient studies (Hahs and McDonnell, 2006). It is likely that these measures differ in their representation of key resources or ecological disruptors as they are broad and indirect measures of a complex anthropogenic gradient (McDonnell and Hahs, 2008). Although most species demonstrated a negative association with urbanisation, we found a non-linear relationship between *P. pipistrellus* activity and built surface cover. Activity peaked at ~40% built cover, yet at levels above ~60% activity rapidly reduced, implying the existence of a threshold or tipping point (Scheffer, 2010). This is the first report of such a relationship for a bat species, although non-linear relationships with urbanisation have been identified for other taxa (Tratalos et al., 2007). *Pipistrellus pipistrellus* could be described as an ‗urban adapter‘ (Blair, 1996), whilst the remaining species would be ―urban avoiders" of varying sensitivity. These results broadly agree with those of other studies, for example, that Myotis species tend to avoid villages (Vaughan et al., 1997) and urban centres (Gaisler et al., 1998). European work (Gaisler et al., 1998) suggests that small bats with low wing loadings are tolerant of even dense urbanisation, but large bats with high wing loadings generally avoid urban centres. This is, however, at odds with work in Australia which suggests the reverse (Threlfall et al., 2011), and is also in disagreement with studies of urban bird traits, which suggest large size/wings are a trait of urban adapters (Croci et al., 2008). These differences may be an artefact of differences in urban composition and morphology between European and Australian cities, the scale that urbanisation is measured at, how urban density is defined and the degree to which different species are willing to accept human subsidised resources (e.g. building roosts).

Variable	Abbreviation	Description
Water	Wat	Surface water features from OSM including canals, ponds and streams
Natural	Nat	Polygons dominated by vegetated/unsealed land-cover from OSM including roadside grass verges and parks, but excluding gardens
Garden	Gard	Gardens as defined from the Ordnance Survey Mastermap (OSM) layer in 2008
Roads	Rds	Roads from OSM
Buildings	Build	Built structures from OSM
Built	Bu	Polygons dominated by built landcover types from OSM including roads, buildings and pavements but excluding gardens
Vegetation		All vegetation cover at 2m pixel resolution, extracted from aerial near-infrared and colour photography 2007 (Bluesky International Limited, Leicestershire)
Urban density	Urb	Nominal variable (1 - 2) differentiating between highly urban (Dense Suburban and Dense Urban) and less urban land classes (Suburban, Light Suburban and Rural)
Trees	Tree	Five tree datasets created by selecting areas of the (above) vegetation dataset ≥ a specified height* above the ground, according to photogrammetrically derived data collected in 2007 (Bluesky)
Connected area	ConA	Area of connected tree cover > 4m in height buffered by 50m and intersecting each survey pond. Connectivity is defined here as a spatial network of tree patches separated by a maximum of 40m
Edge	Edge	Length of the perimeter of the connected tree patches described above

Table 2. Land-cover and land-use explanatory variables used in the analysis.
Each was measured as the total area (m2) or length (m) within a radius of 50, 100, 150, 200, 350, 500, 750, 1000, 1250, 1500, 1750, 2000, 3000 and 4000m
*Five datasets representing tree cover with a minimum height of 4, 6, 10, 15 or 20m.

3.2 Landscape composition

The often contradicting findings from studies of urban bat communities illustrate some of the broader challenges associated with attempting to identify ecological patterns along urbanisation gradients (McDonnell and Hahs, 2008). The descriptions of the urban form provided by Gaisler et al. (1998), Avila-Flores and Fenton (2005) and Gehrt and Chelsvig (2003) varied considerably in their detail. We addressed this issue by accessing high-resolution parcel based and remotely sensed data for land-cover and land-use for the entire study area.

Although all species demonstrated a negative response to broad measures of urbanisation, considerable differences in activity were evident between sites at similar points along these gradients. This suggests that more subtle variations in landscape composition may also be important. *Pipistrellus pipistrellus* evening activity was found to be positively associated with gardens, which might be expected given their propensity for roosting in buildings (Simon, 2004) and for early evening emergence. An additional explanation is that gardens might typically provide tree cover that facilitates early emergence and feeding. These findings pose further questions about the mechanisms behind associations between bat activity and residential land-covers reported elsewhere (Gehrt and Chelsvig, 2003, Gehrt and Chelsvig, 2004, Gaisler et al., 1998, Hourigan et al., 2008). As with other studies, differences were found between the landscape composition preferred by *P. pipistrellus* and *P. pygmeaus*. A positive association between *P. pipistrellus* and local tree cover was expected given their known use of edges as commuting and feeding areas (Verboom and Huitema, 1997, Gaisler et al., 1998), and their role in increasing the attractiveness of adjacent roost sites (Jenkins et al., 1998). The association of P. pygmeaus with aquatic habitats described elsewhere (Vaughan et al., 1997, Davidson-Watts and Jones, 2006, Sattler et al., 2007) was not observed in this study. However, this may reflect variations in the quality of riparian vegetation (Scott et al., 2010), which we were unable to measure at a landscape scale.

We expected *M. daubentonii* to demonstrate a strong positive association with the area of water in the vicinity of the survey sites (Dietz et al., 2006), yet we did not find support for this in our results. It is possible that by surveying ponds we removed water as a limiting variable or that the social dynamics of this species served to mask important habitat associations (Kapfer et al., 2008). Members of the NSL guild are reported to seek out pastures, parks and other open green spaces (Vaughan et al., 1997). Their relatively high wing loadings, medium-high aspect ratios and low call frequencies permitting them to hawk in the open, typically feeding on large insects (Altringham, 2003). Our presence models broadly support this, although we were unable to differentiate between ground vegetation types and management for the extent of our study area.

In general, we expected bat presence and activity to reflect the local availability of roosts (Jenkins et al., 1998) and foraging sites (Vaughan et al., 1997). Several of our landscape variables were intended to be proxy measures for these key resources, yet the value of our roost metrics was limited. There is an inevitable trade-off between the detail and availability of urban habitat data and the spatial scale of analysis (McDonnell and Hahs, 2008) and it is likely that greater effort is required to map roost potential effectively. In addition, both roosting bats (Jenkins et al., 1998), commuting bats (Gaisler et al., 1998) and their insect prey are sensitive to variations in microclimate, which in urban areas will be heavily influenced by human activity. Additional data that improves the integration of human processes such as land management and intensity of use into urban ecological models may therefore clarify how roosting or feeding potential varies within each land-cover type.

3.3 Landscape connectivity

Our approach employed proxy measures of functional connectivity to estimate the areas of the landscape theoretically available to bat species that commute along tree networks and is an extension of the accessible habitat model (Eigenbrod et al., 2008, Pascual-Hortal and Saura, 2006). Functional connectivity is concerned with the ability of individuals to move between resource patches within the landscape rather than explicitly measuring the structure of landscape elements, although structure is frequently used as a proxy for function (Tischendorf, 2001, Tischendorf and Fahrig, 2000). The measures of structural landscape connectivity used to extract landscape variables from the GIS appeared to be a good approximation of functional connectivity for the two Pipistrellus species. Euclidian distance may be a more appropriate for measuring accessible habitat for *M. daubentonii* and the NSL group. This supports previous studies highlighting the importance of linear landscape features (Verboom and Huitema, 1997, Limpens, 1991, Downs and Racey, 2006), but this study is unique in demonstrating the functional importance of structural connectivity of tree cover for bat species in urban landscapes. As *M. daubentonii* has a similar wing aspect ratio and loading to the two Pipistrellus species we had expected models for *M. daubentonii* to include measures related to connectivity. It is possible that structural connectivity is relevant to this species, but that our landscape measures were insufficient to detect this relationship. However, given that our site data for this species is limited to activity within the first 1.5 hours after dusk and that this species is a late emerger, such interpretations should be treated with caution.

Whilst the concept is relatively straightforward, measuring functional connectivity in urban landscapes is challenging, particularly as the patch/matrix distinction is often unclear and actual movement paths are not easily observed. Previous studies have successfully employed expert judgement to estimate landscape resistance values for different urban matrix types (FitzGibbon et al., 2007). Our approach is easily replicable and scalable, but relies on the accurate mapping of individual trees in three dimensions and on a consistent response by bats within a population to gaps in tree networks. For studies of highly mobile bird species, estimating the path of movement within the urban matrix has delivered improved models compared to more general landscape measures (Tremblay and St Clair, 2009) and it is worth noting that for both Pipistrellus species studied, the only valid models we identified were those that included variables measured using a connectivity mask. At sites where the built land-cover of the surrounding landscape was over 40%, structural connectivity was critical for maintaining high levels of *P. pipistrellus* activity. Urban density dependent relationships with connectivity such as this have not been demonstrated before.

3.4 Spatial scale

Gehrt and Chelsvig (2003) and Lookingbill et al. (2010) located bat survey sites within natural reserves along an urbanisation gradient. However, the direct comparison of these studies is difficult as the spatial extent used to define urbanisation and characterize the landscape differed between studies. We attempted to avoid such issues by measuring urban density at a wide range of spatial scales. This identified broad patterns, with edge specialists (*Pipistrellus* spp.) being sensitive to landscape composition even at small spatial scales (50–100 m). Their relatively fast and agile flight appears to allow them to utilise relatively small foraging areas, supported by a high wing aspect ratio, low wing loading and small size (Altringham, 2003). This may well explain their presence in densely built areas, as patch sizes tend to decrease with urbanisation (Zhang et al., 2004). That *P. pygmaeus* responds to the landscape at radii of up to 1 km may reflect a need to travel further to access preferred feeding habitats (Davidson-Watts and Jones, 2006) such as highly structured riparian vegetation (Scott et al., 2010). The NSL aerial hawkers guild seeking un-built land-cover at larger scales (≥500 m) would be expected, given that their large size and high wing loading is suited to efficient flight over large open areas (Altringham, 2003).

There are few studies that attempt to characterize the response of bats to urban landscapes at multiple radii that extend over a large spatial extent, although the response of groups such as birds has been explored (Hostetler and Holling, 2000). Our data suggest that individual species may be sensitive to changes in landscape composition at multiple spatial scales. For example, evening models for *P. pipistrellus* include connected garden area within a radius of 500 m and connected tree cover within 150 m. We speculate that the 500 m radius may indicate the ―roost catchment” of the pond i.e. that bats using the pond, tend to roost in houses within 500 m. The area within 150 m of a pond may be relevant to the quality and accessibility of the local feeding area, with ponds surrounded by a high density of tree networks being particularly desirable. These results corroborate other studies that conclude multiple spatial scales may be relevant to bats (Lookingbill et al., 2010) and urban mammals (Garden et al., 2010).

4. Conclusions

We have demonstrated that the density of landscape urbanisation, its composition and configuration are important to the urban bat community. These relationships are scale dependent and species-specific. The broadly negative associations with urbanisation for all species imply that a transition to more compact urban forms that reduce greenspace and habitat would inevitably impact the species richness of the urban bat community. This work informs the continuing debate about the sustainability of this approach to development (Neuman, 2005). The presence of thresholds for ecological function raises the possibility that development densities could be specified with ecological thresholds in mind, and that tipping points should be explored in more detail for other taxa. The importance of connectivity for the *Pipistrellus* species suggests that some ecological function could be retained even within high-density developments and that protected tree networks may deliver some spatial resilience (Nystrom and Folke, 2001) to the impacts of increased urban densities. It remains to be seen whether tree networks play a similar role for other organisms (but cf. (Tremblay and St Clair, 2009)). Our data on ecologically important land-covers, land-uses and spatial scales should support urban planners and managers in making spatially explicit decisions about urban conservation (Garden et al., 2010). We recommend that a multi-scale approach to planning and management be adopted, whilst recognising that this may be challenging given the typical spatial scales of urban land ownership (Ernstson et al., 2010, Goddard et al., 2010) and decision-making (Conroy, 2003). In particular, we suggest that when creating new urban bat habitats, consideration is given to ensuring that they remain functionally connected and therefore available to at least part of the urban bat community into the future.

5. Materials and Methods

5.1 Ethics Statement

The landowners gave permission for access to the sites. All Bat species are protected in the UK and licenses are needed if they are handled, mist-netted, or disturbed in their roosts. As our sampling involved only monitoring at foraging sites there were no licensing issues.

5.2 Study Area and site selection

The West Midlands metropolitan county (population ~2.3 million) is a highly urbanised region of the United Kingdom (UK), covering 902 km^2. As a centre of the industrial revolution it has undergone multiple cycles of development and distinct zones can be identified representing pre, wartime and post-war regeneration. The study area includes several urban centres (Fig. 5) with high levels of sealed land-cover, canals, railways, residential areas of varying housing density, industrial zones, parks, nature reserves and agricultural land on the urban fringe. Existing survey records for the study area indicate that several species of bat were present, including: *Pipistrellus pipistrellus* (Schreber 1774), *Pipistrellus pygmaeus* (Leach 1825), *Myotis daubentonii* (Kuhl, 1817), *Eptesicus serotinus* (Schreber, 1774), *Nyctalus leisleri* (Kuhl, 1817) and *Nyctalus noctula* (Schreber, 1774). These species vary considerably in their roosting, commuting and feeding behaviour (Table S1).

In order to stratify the survey sites along an urbanisation gradient we first classified the landscape using land-use and land-cover data from OS Mastermap (OSM) (Ordnance Survey, 2008), which is a high-resolution parcel based GIS dataset (Table S1). OSM polygon data were converted into a 2 m pixel resolution raster and displayed in a GIS (ArcGIS 9.2, ESRI Redlands, USA). A grid of 1 km^2 cells was used to extract raster summaries using Hawth's Analysis Tools (Beyer, 2004), as this is close to the average minimum foraging areas of both *P. pipistrellus* and *P. pygmaeus* (Davidson-Watts and Jones, 2006), which are the smallest foraging areas for the species we expected to encounter (Table S2). Five land classes were identified using a cluster analysis of landscape variable percentages (Table S1) in SPSS 18.0 (c.f. Owen et al., 2006) and excluding squares with greater than 30% water cover or 80% tree cover. These represented a gradient from Rural (R), Light Suburban (LS), Suburban (S), Dense Suburban (DS) and Dense Urban (DU) land classes (Figs. 5&6). In order to reduce the potential for variations in local habitat composition to obscure the effect of landscape context (Gagne and Fahrig, 2007, Duguay et al., 2007), survey sites were restricted to small (515–2146 m^2) unlit ponds with at least 30% riparian edge tree cover. This choice was a reflection of the attractiveness of aquatic, riparian and woodland edge habitats for foraging to all the species we expected to record (Russ, 1999) and the need to identify a foraging habitat patch that would be present in all land classes. Candidate ponds were assigned to one of the five land classes based on the land-cover and land-use percentages for a 1 km^2 circle surrounding each pond (Fig. 6) and six survey sites were then selected from each land class. All ponds were separated by at least one kilometre (pond centre to pond centre).

5.3 Bat sampling methods

Ponds were surveyed for bat activity fortnightly between May and August 2009. We avoided nights where strong rainfall or wind were predicted and surveyed several sample points within each site (Fischer et al., 2009). A variety of techniques have previously been applied to compare bat species presence and activity between sites, with detectors generally regarded as superior (Hourigan et al., 2008). We used a combination of walking transects (Gaisler et al., 1998, Vaughan et al., 1997) and fixed point detector surveys (Avila-Flores and Fenton, 2005), which allowed multiple microhabitats habitats to be surveyed and activity to be recorded from dusk to dawn.

Evening walking surveys were undertaken for a period of 1.5 hours following sunset using a Pettersson D240x ultrasound bat detector (Pettersson Electronic, Sweden), in heterodyne mode, alternating between 20 and 50 kHz. Sample calls of 3.4 seconds were recorded in time expansion mode and transferred to a Sony MZ_NH6000 Minidisk recorder (Sony, Japan). Walking routes circled each pond at varying distances (0- 50 m from edge), with the purpose of detecting and observing bats that were active in the close vicinity, as well as directly over the pond. Fixed point surveys were initiated at dusk and terminated at dawn, using an AnaBat SD1 frequency division bat detector (Titley Scientific, Australia) installed at the edge of the pond at a height of 1 m, using an acoustic reflector (Corben, 2007). Tests confirmed that bats active within at least 15 m (horizontal distance) and up to ~10 m above ground level were detectable, with bats calling at low frequencies (20–30 kHz) recorded at an unknown but greater distance.

5.4 Call analysis

Bat calls were identified to species level where possible, using parameters given in Russ (1999). Where species identification was not possible in the field, bat calls detected on walking surveys were recorded and analyzed using BatSound 3.31 (Pettersson Electronic, Sweden). Calls recorded using fixed point Anabat detectors were processed automatically using filters within AnalookW (Corben, 2009). Although Myotis sp. calls were identified at several sites the call quality was highly variable. Subsequent tests confirmed that using a reflector on the Anabat dramatically reduced the detectable range for this group, so Myotis calls from the fixed detector were excluded from analysis. Species or guild specific call filters were developed and their results compared to a 10% sample of the call dataset to estimate the percentage of bat calls incorrectly rejected by filters, calls allocated to the incorrect species/group, and files incorrectly identified as a bat call. Considerable caution was applied, preferring filters that discarded a greater percentage of calls. As considerable overlap in call parameters has been reported for *Nyctalus noctula*, *Eptesicus serotinus*, *Nyctalus leisleri*, and these species were rarely observed in flight (which would aid identification), we processed these (NSL) calls as a single functional group of large, early emerging bats with similar foraging behaviours (Table S2).

5.5 Landscape and connectivity environmental variables

Using the GIS we selected a range of variables that related to roosting, commuting and feeding resources or that could be used as broad measures of urbanisation (Table 2). Summaries of the area of built (sealed manmade) or natural (vegetated) surface cover (derived from the OSM landscape parcels) and the (remotely sensed) vegetation layer provided broad indications of urbanisation density. Whilst the parcel based mapping was useful for estimating dominant land-cover, parcel types such as gardens were excluded as they contained varying levels of built and semi-natural land-cover. The remotely sensed vegetation layer was therefore used to gain a better reflection of vegetation cover. For the species we encountered, roost sites are likely to be located either in buildings or trees. Buildings provide a variety of roost opportunities due to their varied age, materials, architectural style and degree of maintenance. In addition to buildings, we included gardens from the OSM as an indirect measure of residential building availability, which we hypothesized might offer an enhanced roosting resource compared to commercial or industrial structures. Tree cover with a minimum height of either 15 m or 20 m was also included as a variable, as this was expected to indicate roost potential in mature woodland.

Several bat species are reported to fly along tree-lines when commuting and feeding (Verboom and Huitema, 1997, Russ, 1999). We estimated suitable commuting habitats by identifying areas of the landscape where vegetation was greater than 4, 6, 10 or 15 m high, which correspond to the range of typical flight heights for these species (Table S2). In addition, the raster representing vegetation greater than 4 m high was converted to a polygon feature class and buffered by a distance of 20 m. This resulted in a layer representing tree networks separated by gaps of no more than 40 m, which was used as a measure of structural connectivity and a proxy for functional connectivity. Although gaps of this size are not likely to be problematic for most species when commuting within rural landscapes (Verboom and Huitema, 1997), we hypothesized that with increasing urbanisation, an increase in artificial lighting could be sufficient to deter bats from crossing such gaps (Stone et al., 2009). Whilst we were unable to access spatial lighting datasets, we used a road dataset derived from the OSM as an indirect indicator of lighting and traffic disturbance. Insect feeding potential was represented by OSM derived polygons depicting water bodies (canals, streams and still waters), natural land-cover (predominantly vegetated) and supplemented with a high-resolution remotely sensed vegetation layer. Finally, the perimeter length of tree patches within tree networks connected to each pond was estimated, as many species are known to feed along woodland edges (Downs and Racey, 2006, Russ, 1999).

Two approaches were taken to extract landscape variable summary data for the landscape surrounding each survey site. Firstly, using the GIS we created multiple concentric circular buffers around the ponds and extracted complete summaries of the underlying landscape data (Fig. 2A). Such a multi-scale approach is increasingly common (Garden et al., 2010, Cushman and McGarigal, 2004), although we used a particularly large number of radial extents (14, between 50 m and 4 km) in an attempt to accurately identify the spatial scales of relevance for each species. This approach assumes that all of the landscape is potentially available to the species concerned. Our second approach was to restrict the landscape analysis to the parts of the landscape adjacent to tree-lines. As both *P. pipistrellus* and *M. daubentonii* activity has been reported to occur predominantly within ~50 m of water and woodland edges (Gaisler et al., 1998, Downs and Racey, 2006), we buffered the tree networks connected to each pond by 50 m, creating a connectivity mask. Landscape variable summaries were again extracted at multiple radii around each pond centre, but this time the available landscape data were limited to the areas intersected by the connectivity mask (Fig. 2B).

5.6 Data analysis

The measure of bat activity used for each site was the total number of minutes in which a call was recorded, for each species or functional guild (hereafter termed active minutes). We make no assumptions that this is a measure of individual bat abundance. Variation in bat activity between land classes was analyzed in SPSS using either a one-way ANOVA or a Kruskal-Wallace test if variances were heterogeneous. Tukey or Nemenyi post hoc tests were used to identify which classes differed in activity (Wheater, 2000).

The relationships between the environmental measures and bat activity were modelled using a combination of Brodgar v2.6.4 (Highland Statistics, Newburgh, UK) and R (version 2.11.1) (R Development Core Team, 2010) using the mgcv and nlme libraries. We used up to three response variables per species (or guild): (i) the total minutes of bat activity recorded by the fixed detector for the first hour and a half following dusk (evening) (ii) total bat minutes recorded by the fixed detector from dusk to dawn (full-night) and (iii) presence using either the fixed detector or walking transect surveys (when bat activity data were either unavailable or did not produce valid models).

Data exploration was undertaken prior to statistical analyses, and as a result, we included an additional nominal explanatory variable differentiating between sites in highly urban (Dense Urban and Dense Suburban) and less urban land classes (Suburban, Light Suburban and Rural). This was used as a conditional variable in the fixed element of the models for some species (e.g. Table 1).

Initially we developed species or guild specific models independently for each spatial scale (50, 100, 150, 200, 350, 500, 750, 1000, 1250, 1500, 1750, 2000, 3000 and 4000 m). First, we used co-plots to inspect co-variation between explanatory variables at each scale and where variables had high correlations (>0.5) one of the pair was removed. We then assessed these against the response variables, removing explanatory variables with correlation scores of <0.3. This reduced the potential pool of explanatory variables considerably. None of the explanatory variables at higher spatial scales (>1000 m) showed significant relationships with any of our response variables. This left a pool of seven spatial variables at eight spatial scales (56 in total). We entered these into all the models and used Akaike Information Criterion (AIC) to identify the most parsimonious model for each species or guild, ensuring that model variables had variation inflation scores (VIFs) of <3 (Zuur, 2010). Where initial co-plots suggested linear relationships between the response and explanatory variables we used generalised linear modelling (GLM) (McCullagh and Nalder, 1983). Non-linear relationships were analyzed using generalised additive modelling (GAM) (Hastie, 1990). The deviance/degrees of freedom ratio was used to assess possible over-dispersion in the models (Zuur, 2010). We used negative binomial distributions to account for over-dispersion (Ne) (O'Hara, 2010) and logistic regression with a binomial (Bi) distribution for presence data (McCullagh, 1983). Finally, this process was repeated with the variables from the best models at each scale combined to derive multi-scale models, pooling site based, concentric and connected variables for each species. All Models were validated using graphical visualisation tools in Brodgar and R. We plotted the residuals against fixed values to assess model homogeneity, QQ-plots for normality and plotted residuals against environmental co-variables to test for independence (Zuur, 2009). Lastly, we used bubble plots in the gstat R library to examine each individual model for spatial autocorrelation (Pebesma, 2004). Where patterns indicated no spatial patterning the models were accepted (e.g. Figure S2). After validation we were left with a pool of 51 models: *Pipistrellus pipistrellus* evening (9), all-night (8); *Pipistrellus pygmaeus* evening (9), all-night (12), *M. daubentonii* evening (5) and NSL guild evening (8). Of these, candidate models with AIC≤2 of the optimum model were retained (Burnham, 2002) (Table 1).

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Supporting Information

Supporting information can be found in the Appendix at the end of this thesis.

Figure S1 Residual bubble plot for NSL all-night Anabat data from logit binomial presence-absence data. The plot shows clumping of similar size positive residuals in the middle of the plot, indicative of spatial structuring in the data. Negative residuals in black and positive residuals are grey. The size of the circles indicates the size of the residuals.

Figure S2 Bubble plot for P. pipistrellus all-night Anabat residuals from a GAM of bat activity minutes. The plot indicates no spatial structuring in the data. Negative residuals in black and positive residuals are grey. The size of the circles indicates the size of the residuals.

Table S1 Mean area (m^2) and standard deviation of Ordnance Survey (OS) land-cover type for each urban land class.

Table S2 Broad life history data for bat species recorded within the study area.

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PAPER II. RESILIENT ECOLOGICAL SOLUTIONS FOR URBAN REGENERATION.

James D. Hale[1], Jon P. Sadler[1]

1. School of Geography, Earth and Environmental Sciences, The University of Birmingham, Birmingham, B15 2TT, UK

Abstract

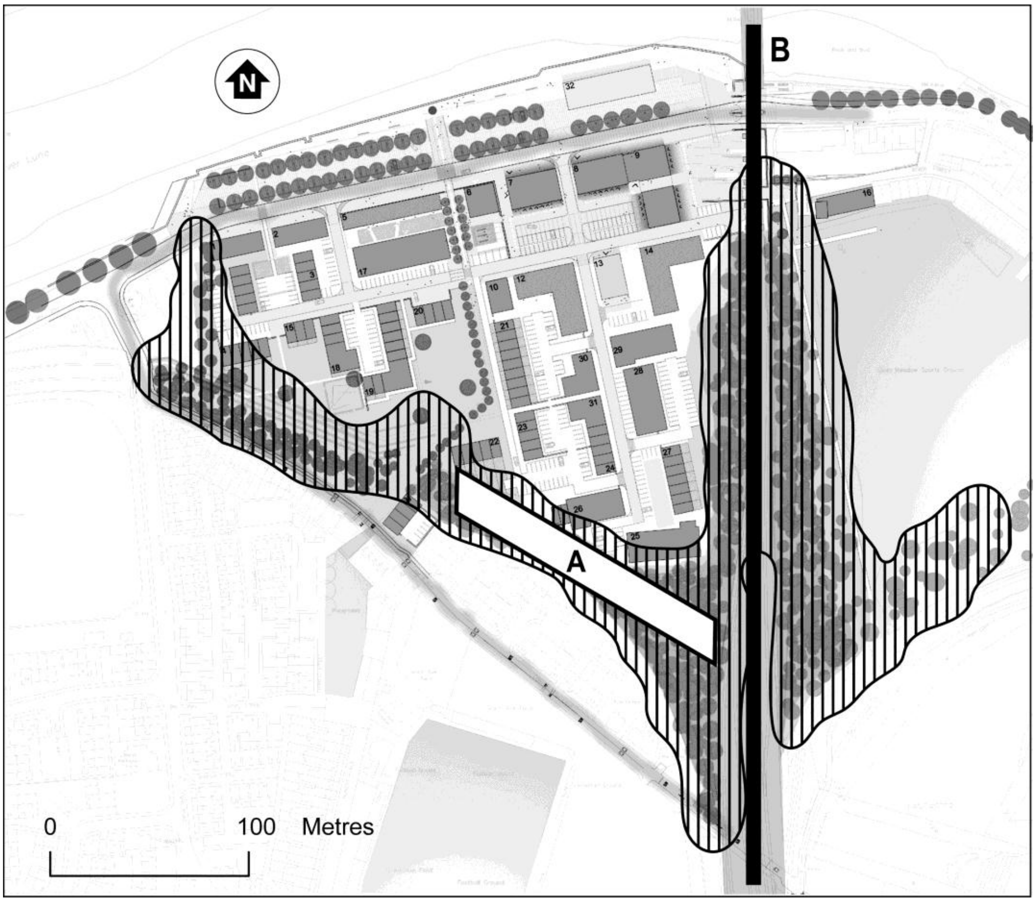
There is a need for biological conservation at the global scale, and urban conservation has the potential to support the delivery of this wider goal. Despite historic trends, efforts are underway to protect and enhance the quality, quantity and accessibility of green infrastructure within cities, including biodiversity features within new developments. However, there are questions over their long-term persistence and function. This paper applies an urban futures resilience analysis to a case study site to illustrate how such concerns may be explored and addressed in practice. The analysis identifies vulnerable sustainability solutions and clarifies the aspects that may be improved. The results suggest that the resilience of these solutions is questionable, even though resilience has clearly been considered. In particular, future compliance with, and enforcement of, planning conditions is questionable. The resilience of these ecological solutions may be improved by including some redundancy, designing for low maintenance, incorporating microclimate buffers and locating features in areas unlikely to be subject to future disturbance. The establishment of endowment funds or other dedicated funding mechanisms should

also be explored. The paper also recommends that a futures-based resilience analysis be included within the development planning process.

1. Introduction

The need for biological conservation at the global scale is clear, as rates of extinction, habitat loss and degradation show little sign of slowing (Butchart et al., 2010). Local-scale conservation efforts within urban areas have the potential to support the delivery of this wider goal. This may be by way of the direct protection and enhancement of species of conservation concern or through the development of accessible green spaces where people are able to experience a range of species and habitats. Urban landscapes provide many opportunities for direct conservation and enhancement, particularly through the regeneration process (Sadler et al., 2011). These include the protection of relict native habitats, the construction of natural habitat analogues (Lundholm and Richardson, 2010) such as brown roofs (Oberndorfer et al., 2007) and artificial roosts (Williams, 2010), and changes to the management of amenity green spaces (Sadler et al., 2011). It has also been argued that positive experiences with urban wildlife have indirect benefits for global conservation in the form of greater public support for related policies and campaigns (Dunn et al., 2006). In addition, the ecological services provided by urban wildlife and green spaces are relevant to the delivery of numerous sustainability goals (MEA, 2005) related to quality of life, social cohesion and sense of place (Miller, 2005). Ensuring a diverse and accessible urban wildlife community should therefore be central to strategies for both global biological conservation and sustainable development. The majority of the global population now reside in cities (UN, 2010) and the extent and density of urban areas are expected to continue to increase during this century (Irwin and Bockstael, 2007). Urbanisation is often characterised by high levels of impervious surfaces (McKinney, 2002), patch fragmentation (Luck and Wu, 2002, Zhang et al., 2004) and heterogeneity in land cover type over time and space (Cadenasso et al., 2007, McDonnell and Pickett, 1990). Despite considerable variability, increasing urbanisation generally results in a reduction in species richness (McKinney, 2008) and ecosystem services (Tratalos et al., 2007). A reduction in the area and accessibility of urban green spaces during the latter half of the twentieth century has been reported for the UK in general (UKNEA, 2011a) and across Europe (Fuller and Gaston, 2009). However, there are indications that some losses are now being reversed (UKNEA, 2011b). Efforts have been made to compensate for losses and to enhance biodiversity within new developments (DEFRA, 2007), focusing on the planning, design and installation of habitat structures (DCLG, 2010, Williams, 2010). However, relatively little is known about the long-term persistence of these structures and their ecological function post-development (Sadler et al., 2010). Recent high-profile failures of some artificial habitats (e.g. http://news.bbc.co.uk/2/hi/uk_news/england/london/8215035.stm) and analyses of post-mitigation success (e.g. Waring, 2011) highlight the need to consider whether such investments are sufficiently future-proofed. This paper applies an *urban futures* resilience analysis to a regeneration case study site in the UK in order to explore the vulnerability of a selection of ecological interventions (hereafter termed ecological sustainability solutions) that are commonly undertaken to deliver biodiversity goals within urban regeneration projects. The focus is on species of birds and bats that are protected under European and UK law and are frequently identified as targets for mitigation, compensation or enhancement during development schemes. While it is appreciated that a large number of ecological sustainability solutions may be included within regeneration projects, this paper focuses on three examples in order to illustrate how a futures-based resilience analysis can be applied in practice. The information available for these examples is therefore limited, reflecting the level of detail supplied in the various planning documents relevant to the case study site.

Figure. 1



Plan view of proposed site layout (modified from the Luneside East masterplan design code (LCC, 2007: p. 11, Figure 12) and reproduced by permission of Lancaster City Council). The hashed area indicates a major tree network expected to facilitate bat movement through the site. The white area labelled A indicates the embankment of a disused railway line and the black strip labelled B indicates the embankment of a mainline railway

2. Methodology

2.1 Case study site: Luneside East

Luneside East is a post-industrial site in Lancaster, UK, proposed for mixed-use regeneration. In 2004, it was Lancaster City Council's (LCC) largest single regeneration project, with a vision to transform the largely vacant and derelict site into a vibrant, well-used and integrated quarter (LCC, 2004). The site (owned by LCC) is 6.6 ha in area and is bounded by a mainline railway (owned by Network Rail), a disused railway embankment (owned by LCC), a river and an established residential area (Figure 1). The land cover is typical of many brownfield sites, with built structures of varying integrity, contaminated soils and a mix of bare ground, ephemeral vegetation, scrub and semi-mature trees (Rogers et al., 2012). The site has outline planning permission (granted in 2001), an environmental statement (2001), a development brief (2004) and a masterplan design code (2007). These documents were used to inform the analysis in this paper, although it is acknowledged that the plans are currently under review.

2.2 The urban futures resilience analysis methodology

The urban futures methodology addresses the question: will today's sustainability solutions deliver their intended benefits whatever the future brings? The analysis is divided into four steps (Boyko et al., 2012, Rogers et al., 2012). In step 1, the sustainability solutions are listed and their intended benefits are described. This step is particularly important because clarity on the nature of each solution and its intended purpose underpins the validity of subsequent steps in the analysis. The prerequisite conditions for the delivery of each intended benefit are outlined in step 2, including the key patterns and processes that need to be in place if each solution is to function effectively. Step 3 provides an analysis of whether these necessary conditions are likely to remain in place in the future. To provide a structured approach to this analysis, the following plausible, robust and divergent future scenarios have been defined for UK urban areas.

(a) Policy reform (PR). Government action is promoted in an attempt to reduce poverty and social conflict, although behaviour change is slow. There is a belief that markets require strong policy guidance and legislation/regulation to address inherent tendencies toward economic crisis, social conflict and environmental degradation. The tension between continuity of dominant values and greater equity for addressing key sustainability goals is not easily reconciled.

(b) Market forces (MF). The self-correcting logic of the market predominates, with individualism and materialism as core human values. Well-functioning markets are thus considered key to resolving social, economic and environmental problems. This scenario assumes that the global system in the twenty-first century evolves without major surprise and incremental market adjustments are able to cope with social, economic and environmental problems as they arise.

(c) Fortress world (FW). Powerful actors safeguard their own interests and resources at the expense of an impoverished majority who must live in ghettoes. The world is divided, with the elite in interconnected, protected enclaves and an impoverished majority outside. Armed forces impose order, protect the environment and prevent collapse.

(d) New sustainability paradigm (NSP). An ethos of ‗one planet living‘ pervades and a fundamental questioning of progress emerges in light of sustainability goals. New social–economic arrangements and fundamental changes in values result in changes to the character of urban industrial civilisation rather than its replacement.

These four scenarios were selected from the six scenario variants developed by the Global Scenarios Group (www.gsg.org) (Raskin et al., 1998) and adapted to reflect a UK urban context, as part of the urban futures project (www.urbanfutures.org). For each intended benefit, the necessary conditions are considered in the context of an extensive characteristics list developed to describe each future scenario (Boyko et al., 2012, Rogers et al., 2012). In the final step, if the necessary conditions are unlikely to be supported in some of the future scenarios then the solution is classed as vulnerable, prompting a revision of plans for its design, construction and maintenance. An example of how this methodology may be applied in practice is provided below, drawing on Luneside East regeneration as a case study.

3. Results

3.1 Ecological sustainability solutions suggested for the Luneside East regeneration site

Biodiversity concerns are referred to within the LCC core strategy (LCC, 2008) and several ecological sustainability solutions were proposed for the site following an environmental impact assessment (EIA) (Entec, 2001). These were intended either to mitigate/compensate for impacts on local biodiversity or to deliver ecological enhancements. These solutions and their intended benefits are most clearly stated within the Luneside East environmental statement (Entec, 2001) and a selection are summarised in Table 1. The analysis presented here is limited to the solutions with clearly stated intended benefits. This is vital because, without clarity on the purpose of each solution, its vulnerability cannot be assessed.

3.2 Conditions necessary for the solutions to deliver their intended benefits

3.2.1 Bats

A bat (Chiroptera) survey was undertaken to inform the EIA, as all bats are legally protected at European level under the EU 1992 habitats and species directive. All bats and their roosts are also legally protected in the UK under The Conservation of Habitats and Species Regulations 2010, with reckless or intentional disturbance in England an offence under the Wildlife and Countryside Act 1981 (as amended) and the Countryside and Rights of Way Act 2000. In addition, bats have a dedicated species action plan as part of the Lancashire biodiversity action plan. The survey identified common pipistrelle bats (*Pipistrellus pipistrellus*) commuting or foraging in several parts of the site and the possibility that some buildings may contain winter hibernation roosts. The associated development impacts, proposed solutions and their necessary conditions are now outlined.

(a) Artificial bat roosts. The most current proposals include the installation of ‗bat boxes' (artificial bat roosts) to compensate for possible loss of winter hibernacula, but do not specify their type or location. However, it is clear that any compensation for the loss of possible winter roosts (see Table 1) should include artificial roosts in structures that are undisturbed, with a cool and stable temperature. Disturbance may include physical movement, predation, poisoning from pest control or building treatment products, high-frequency noises, artificial lighting and changes in temperature or humidity. The artificial roosts must also be accessible to the bats and be retained on-site as features. Assuming these conditions will be met during installation, the success of this solution would be dependent on these conditions continuing indefinitely into the future.

(b) Bat foraging habitat. The proposed increase of, and enhancement to, foraging areas within the site are primarily intended to benefit bats that are active during the spring, summer and autumn. During this period, common pipistrelles typically roost in warm inhabited buildings, and the EIA report concluded that modern houses outside the boundary of the site were the most likely location of summer roosts for the bats recorded as foraging on-site. For the enhanced foraging areas to be successful, they must be available to common pipistrelles following completion of the Luneside East development. This requires that local summer roosts continue to be present, that bats can commute from these roosts to the Luneside East feeding areas and that the foraging habitats produce sufficient quantities of their insect prey.

3.2.2 Birds

Although no nesting sites were recorded as part of the EIA, enhancements are currently proposed to support several species that are listed within the Lancashire biodiversity action plan (Table 1). These birds are protected at European level under the EU 1992 habitats and species directive and the 2009 birds directive. They are legally protected in the UK under The Conservation of Habitats and Species Regulations 2010, with intentional killing, injury or damage of the birds, their eggs or active nests an offence in England under the Wildlife and Countryside Act 1981 (as amended) and the Countryside and Rights of Way Act 2000. The success of the bird nesting boxes proposed as ecological enhancements depends on conditions similar to those required for artificial bat roosts. Boxes must be retained on-site, accessible to the birds and remain undisturbed. All species that are intended to benefit from these enhancements at the Luneside East site require nesting sites that are out of direct sunlight (Williams, 2010). Some require unobstructed flight paths to the nests and others, such as swifts (*Apus apus*), require a site free of climbing plants that may give access to predators. Again, these conditions must continue to be present indefinitely into the future if the nest boxes are to function as intended.

3.3 Performance of the Luneside East ecological solutions within the urban future scenarios

The analysis indicates that, under certain scenarios, it is questionable whether the habitat features of interest will remain undisturbed, whether microclimates will be preserved and functional connectivity maintained (Table 2). Habitat management is considered unlikely to be undertaken in two of the scenarios and its presence is questionable in a third. It is only in the NSP scenario that all the conditions necessary for these sustainability solutions to function are likely to be present. The reasoning behind these results and implications for specific solutions are now discussed.

Table. 1

Ecological solution	**Intended benefits**	**Post-development retention mechanisms**
A Bat hibernation boxes	Compensation for possible loss of winter hibernation roosts within existing buildings on-site	Condition to be checked every 5 years by an ecologist. Planning controls used to ensure the required management, repair and replacement is undertaken
B Expansion and management of semi-natural vegetation as bat foraging habitat	To enhance the foraging habitat for the common pipistrelle (*Pipistrellus pipistrellus*) with new habitats created to complement those retained as part of the disused railway embankment	Planning controls to ensure the implementation of a management plan in perpetuity, with checks every 5 years. New habitats to be monitored annually for first 3 years by an ecologist
C Bird nesting boxes	Enhancements for local priority bird species such as swifts (*Apus apus*), house martins (*Delichon urbica*), house sparrows (*Passer domesticus*) and starlings (*Sturnus vulgaris*)	Condition to be checked every 5 years by an ecologist. Planning controls used to ensure cleaning, repair and replacement

Proposed ecological sustainability solutions for the Luneside East development, their intended benefits and evidence that retention (post-development) has been considered. This summarises information from the LCC environmental statement (Entec, 2001).

Table. 2

Necessary conditions for the success of proposed solution	Scenario			
	PR	MF	FW	NSP
Habitat features not intentionally disturbed[ABC]	✓	?	✓	✓
Habitat features not accidentally disturbed[ABC]	?	?	?	✓
Microclimates (light, temperature, moisture) are maintained[AC]	?	?	?	✓
Functional connectivity is retained[AC]	?	?	?	✓
Habitats are managed to deliver their intended ecological function[B]	?	X	X	✓

Summary of results from a futures-based sensitivity analysis of key local conditions. Superscripts A, B and C indicate the solutions listed in Table 1 that these conditions relate to. ✓ indicates where a condition is expected to be supported within a particular scenario, ? means that it is unclear whether the condition will be supported and X indicates that support for this condition is unlikely.

4. Discussion

4.1 Vulnerability of proposed ecological sustainability solutions

4.1.1 Artificial bat roosts

Bats rarely cause nuisance to householders and therefore the intentional disturbance of an artificial roost is considered unlikely. However, in future scenarios such as MF, in which materialism and individualism are valued over environmental concerns and planning enforcement is expected to be weak, artificial bat roosts may be removed if the structure is damaged or droppings impact the aesthetics of a building.

Accidental disturbance is considered to be a reasonable risk within three of the scenarios. In the PR scenario, policies to meet social sustainability goals (e.g. encouraging flexible building use) may result in warmer or inconsistent hibernation roost temperatures, undermining their success. In addition, apparent ‗holes‘ (roost entrances) in a building envelope may be inadvertently sealed during routine maintenance, to ensure good thermal performance. Artificial lighting of the roost or roost entrance is considered a risk in several scenarios, preventing or disturbing access for bats (see Waring (2011) for case studies where this has occurred elsewhere). In the PR scenario, this lighting may be intended to encourage walking as an alternative to night-time car use while, in the MF scenario, lighting may be used as a tool for raising the visual profile of the development or illuminating advertising boards. Artificial lighting of roost entrances may also occur in the FW scenario, but in this case may be used to increase site security or the perception of safety. The proposed planning conditions to require monitoring and maintenance of roosts on a five-yearly basis are unlikely to be enforced in either the MF or FW scenarios, as values and priorities lie elsewhere.

4.1.2 Bat foraging habitat

The current proposals imply that winter rather than summer roosts will be created as on-site compensation. Any new foraging habitat created on the Luneside site would therefore be used in the summer by bats that are roosting off-site in adjacent residential areas. However, roosts within off-site buildings are considered vulnerable in three of the four scenarios as they may be unintentionally lost during building renovation or changes to the immediate built environment. The loss or isolation of off-site roosts would make on-site feeding areas redundant from the perspective of bat conservation. For several UK bat species (including the common pipistrelle), unlit tree lines are important commuting routes between roosts and foraging areas. The bat survey and consultant's report included within the EIA identified the trees along the disused railway embankment and along the active railway line as particularly important in this respect (Figure 1).

In the current analysis, the function of the disused railway embankment as a commuting route is considered vulnerable in three of the four scenarios. Future tree losses may occur if their canopies are managed in the PR scenario to improve passive solar gain for adjacent buildings, in the MF scenario to maintain a desirable view or in the FW scenario as a local supply of fuel. In addition, artificial lighting may also increase in these scenarios, thus threatening the accessibility of foraging areas (Stone et al., 2009). Feeding areas are considered vulnerable to disturbance or degradation in three scenarios. Although planning policy in PR would generally support their retention, the loss of these areas may be permitted if it contributes to achieving targets for higher residential density and social equity. In the MF scenario, if the land value of these foraging areas were to be high, planning decisions would be likely to favour development over conservation. Should these foraging areas remain undeveloped, they are likely to be vulnerable to gentrification, typified by amenity planting with non-native species, frequent maintenance and low insect productivity (Donovan et al., 2005). Low land values would likely result in the abandonment of habitat management and potentially a reduction in foraging quality over time. The proposed planning conditions to monitor and maintain semi-natural vegetation in perpetuity are unlikely to be enforced in either the MF or FW scenarios, as values and priorities lie elsewhere.

4.1.3 Bird nesting boxes

As with the bat hibernation boxes, bird nesting boxes may be intentionally removed in scenarios where planning enforcement is weak and aesthetics are prioritised over the environment. Bird nesting boxes are potentially more vulnerable than artificial bat roosts, particularly those for house martins (*Delichon urbica*) and starlings (*Sturnus vulgaris*), which may be considered a nuisance due to their droppings and noise respectively (Williams, 2010). Again, accidental disturbance appears to be a greater threat, with exposure of nests to direct sunlight (following changes to tree or building cover) being of particular concern. Although the monitoring and repair of these features is inexpensive, a planning condition to ensure their maintenance on a five-yearly basis is unlikely to be enforced in either the MF or FW scenarios, as priorities lie elsewhere.

4.2 Resilience of selected ecological solutions proposed for Luneside East

Resilience is a term increasingly used in discussions about sustainable development, but is applied differently depending on the context of its use (Folke et al., 2010, Pickett et al., 2004). Walker et al. (2004: p. 1) define resilience as ‗the capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity, and feedbacks‘. In this paper, resilience is defined as the capacity of a sustainability solution to continue to deliver its intended benefits, despite changes to its environmental, social, economic or political context.

The results of the selective analysis described in this paper suggest that none of the ecological solutions proposed for Luneside East is particularly resilient, even though some consideration has clearly been given to sustaining their intended benefits post-development (Table 1). The difficulties in ensuring the long-term maintenance of biodiversity compensation and enhancements are well known among practitioners and issues such as the governance and management of urban green space have been explored in the academic literature (e.g. Hermy, 2011, James et al., 2009). The futures-based resilience analysis illustrated here may be a particularly valuable tool for improving the communication of these vulnerabilities among key decision makers. The next step is to explore how these solutions might be modified to improve their resilience, so that they deliver their intended benefits in any envisaged future.

4.2.1 Suggestions for improving the resilience of the proposed ecological solutions

While modification of the proposed solutions is not within the scope of this paper, some general approaches are considered below. Intentional disturbance is the threat to the functioning of habitat features that is perhaps the most difficult to respond to. Increased legal penalties for removing bat/bird boxes may be sufficient deterrent in some cases, but their effectiveness relies on feedback loops that may be degraded in some scenarios (e.g. residents may fail to report wildlife crime and responsible agencies may fail to act). A more reliable approach may involve designing these features in a manner that makes them more difficult to disturb, less likely to cause nuisance and easier to maintain. This could be as simple as integrating bird nesting boxes or artificial bat roosts into the building fabric (e.g. using bat bricks) rather than attaching them to outside walls (see Williams, 2010). Ensuring that people are aware that solutions are vulnerable to disturbance can be achieved through management agreements that specify community participation or warning signs incorporated into specific features that will be visible during building maintenance. However, in scenarios where development decisions are market led, awareness of such tensions may make little difference.

The strategy of locating key features in areas where conflicts are less likely to arise may be successful, particularly where this includes the transfer of ownership to a community land trust or where these features are likely to be valued and protected by multiple decision makers. In the case of Luneside East, the active railway embankment immediately adjacent to the site would appear to be ideal for providing resilient access for bats to foraging areas. The topography and adjacent land use makes future development pressure unlikely, while the dense vegetation would probably be valued by both residents as a screen from noise and the landowner as it impedes public access to the railway track. However, establishing a broader connected tree network would provide some useful redundancy, as tree lines in the surrounding landscape are still considered vulnerable. Similarly, locating artificial winter roosts throughout this network creates a diversity of accessible roost options, so should a roost be damaged or isolated, bats may respond by switching to a local alternative. In addition, creating artificial summer roosts on-site would have the benefit that the function of new on-site feeding areas is not reliant on bats roosting in off-site areas, which may be more vulnerable to loss or isolation.

Maintaining microclimates (such as temperature and moisture) within a particular range is crucial to the success of many ecological solutions. There is a need to buffer against extreme changes and it is clear from the analysis that feedback loops reliant on well-resourced and ecologically motivated planning authorities are particularly vulnerable. Alternatives include locating sensitive ecological features on sites where adjacent land use or topography is unlikely to change or to include lighting, thermal or moisture buffers as part of the solutions themselves (e.g. lighting shields around roost entrance, moisture-absorbent substrates and ceramic heat sinks). In future scenarios where resources are under pressure or public values are unsupportive, habitat management may be much reduced. Design may again play a useful role in improving resilience, with a focus on designing for longevity and low maintenance. Additional mechanisms to support long-term maintenance may also be explored, such as establishing endowment funds or the management of ecological features (e.g. as commercial woodland) to generate funds in perpetuity.

4.3 Resilience and the development planning process

Building resilience into a sustainability solution requires awareness that the drivers of its future success may be social, environmental or economic. It may therefore be necessary for professional input from a range of disciplines (e.g. legal, financial, design and communication). This is particularly true when identifying the conditions that need to be in place for a solution to function and for considering how the solution might be modified. Various attempts have been made to conceptualise urban areas in a manner that includes the human and ecological components on equal footing, to facilitate collaboration between disciplines (Alberti et al., 2003, Folke et al., 2005). Conceptualisation of cities as social–ecological systems and improving the collaboration between disciplines is a key ingredient to integrating ecological conservation into urban planning (Niemela, 1999) and providing a strong basis for managing system resilience (Folke et al., 2010). The urban futures resilience analysis methodology has therefore been developed to support broader systems thinking, to be as accessible as possible (avoiding discipline-specific language and concepts) and has been tested using a wide variety of sustainability solutions (as discussed elsewhere in this special issue). In principle, any sustainability solution could be analysed in this way as long as sufficient information is available to define the solution, its intended benefits and the condition necessary for these benefits to be delivered in the future.

As sustainability has become a key goal in urban planning policy (Bramley et al., 2006), it follows that resilience management for sustainability should play a prominent role in the planning process. Attempts to improve the longevity of ecological compensation and enhancement measures are evident in both urban planning policy and practice, yet their effectiveness is often questionable. Implicit within related planning conditions are assumptions about resources, values and governance; that is, that in the future funding will be available for the required management and there is the will and capacity to enforce these conditions. This is illustrated in Table 10.3 of the Luneside East environmental statement (Entec, 2001:

p. 98), which states that ‗planning controls should be used to ensure that the area (of semi-natural vegetation) is managed in perpetuity'. The implication is that a condition for continued management will be attached to any consent for development and monitored by LCC in perpetuity, yet there is no guarantee that LCC will have the capacity to do this in the future. Declines in the quality of green infrastructure reported in recent decades (DTLR, 2002) and reports of poor post-development compliance of mitigation features to planning conditions (e.g. Waring, 2011) indicate that the current system of ecological governance is failing. While there appears to be a broad awareness among practitioners that some mitigation and enhancement measures may be temporary, there are few tools that allow these concerns to be demonstrated to a diverse audience. It is therefore suggested that consideration of future-proofing should be explicitly included within the Royal Institute of British Architects' outline plan of work (RIBA, 2007) and that evidence of a resilience analysis be required as part of planning submissions for development consent.

As a cautionary note, careful consideration needs to be given to the appropriate level of resilience to incorporate into a particular sustainability solution. Increasing the resilience of one desirable component of a system may compromise the resilience of others (Folke et al., 2010). A balance is therefore required between future-proofing particular sustainability solutions and retaining the flexibility to adapt the regeneration site in the future.

5. Conclusions

In this paper, resilience is defined as the capacity of a sustainability solution to continue to deliver its intended benefits, despite changes to its environmental, social, economic or political context. Recent reports raise concerns as to whether the ecological sustainability solutions often implemented as part of regeneration projects will continue to deliver their intended benefits in the long term. Their performance may rely on questionable assumptions about resources, values and governance in the future and it is argued that there is a need for a tool that can make these vulnerabilities explicit.

The urban futures resilience analysis method illustrated here provides a structured approach to identifying vulnerable sustainability solutions and to clarifying the aspects of each solution that may need to be improved. The results of this selective analysis suggest that none of the ecological solutions proposed for the Luneside East case study is particularly resilient, even though some consideration has clearly been given to sustaining their intended benefits post-development.

In particular, the effectiveness of planning conditions and enforcement is questioned, given future scenarios where political and financial priorities may lie elsewhere. In terms of improving the resilience of these ecological solutions, the inclusion of some redundancy, designing for low maintenance, including microclimate buffers and locating features in areas unlikely to be subject to future disturbance may be particularly effective. The establishment of endowment funds or other dedicated funding mechanisms should also be explored.

Ensuring that current investments in sustainability solutions will continue to deliver their intended benefits into the future should be at the heart of sustainable development. It is thus recommended that resilience analysis techniques such as the one presented here be explicitly included within the development planning process.

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PAPER III. MAPPING LIGHTSCAPES: SPATIAL PATTERNING OF ARTIFICIAL LIGHTING IN AN URBAN LANDSCAPE

James D. Hale[1], Gemma Davies[2], Alison J. Fairbrass[3], Tom J. Matthews[4], Christopher D. F. Rogers[5], Jon P. Sadler[1],

1. School of Geography, Earth and Environmental Sciences, The University of Birmingham, Birmingham, B15 2TT, UK
2. Lancaster Environment Centre, Lancaster University, Lancaster, LA1 4YQ, UK
3. University College London, Silwood Park Campus, Ascot, SL5 7PY, UK
4. School of Geography and the Environment, University of Oxford, Oxford, OX1 3QY, UK
5. School of Civil Engineering, The University of Birmingham, Birmingham, B15 2TT, UK

Abstract

Artificial lighting is strongly associated with urbanisation and is increasing in its extent, brightness and spectral range. Changes in urban lighting have both positive and negative effects on city performance, yet little is known about how its character and magnitude vary across the urban landscape. A major barrier to related research, planning and governance has been the lack of lighting data at the city extent, particularly at a fine spatial resolution. Our aims were therefore to capture such data using aerial night photography and to undertake a case study of urban lighting. We present the finest scale multi-spectral lighting dataset available for an entire city and

explore how lighting metrics vary with built density and land-use. We found positive relationships between artificial lighting indicators and built density at coarse spatial scales, whilst at a local level lighting varied with land-use. Manufacturing and housing are the primary land-use zones responsible for the city‘s brightly lit areas, yet manufacturing sites are relatively rare within the city. Our data suggests that efforts to address light pollution should broaden their focus from residential street lighting to include security lighting within manufacturing areas.

1. Introduction

As the global population grows and becomes increasingly urban (Grimm et al., 2008, UN, 2010), cities are increasing in their spatial extent (Antrop, 2000), intensity of use (Dallimer et al., 2011) and physical heterogeneity (Wu et al., 2011). Measuring variation within urban systems (in terms of their composition, configuration and function) plays a vital role in supporting research and management for improved sustainability performance (McDonnell and Pickett, 1990, Bolund and Hunhammar, 1999, Cervero, 2001, Zhang et al., 2004, Glaeser, 2011). However, systematic urban data collection and interpretation is challenging (Weng and Lu, 2007) given the high spatial variability within (Cadenasso et al., 2007) and between urban areas (Fuller and Gaston, 2009), the co-variability between features of urbanisation (Hahs and McDonnell, 2006) and scale dependent relationships (Andersson et al., 2009). Multiple and diverse measures of urbanisation at a range of spatial scales are therefore desirable (McDonnell and Hahs, 2008).

One variable that is closely associated with urbanisation is outdoor artificial lighting. Remotely sensed data are good predictors of both urban extent (Imhoff et al., 1997, Small et al., 2005) and population size (Elvidge et al., 1997b, Sutton et al., 2001) at coarse spatial scales. Like urbanisation, the spatial coverage and intensity of artificial light pollution appear to be increasing (Cinzano, 2006, Holker et al., 2010); whilst the spectrum of the night sky is also changing due to the replacement of lighting infrastructure (Massey and Foltz, 2000). Lighting has strong cultural links to ideas of modernity and safety (Lyytimaki et al., 2012) and is a hallmark of development, giving people greater choices as to where, when and how long their activities can take place. However, lighting has other direct effects on health (Falchi et al., 2011, Stevens, 2009), culture and amenity (Cinzano et al., 2001, Mizon, 2002, RCEP, 2009), safety (Wanvik, 2009), security (Farrington and Welsh, 2002) and ecology (Longcore and Rich, 2004, Bruce-White and Shardlow, 2011) and indirect effects on economics (Gallaway et al., 2010) and carbon emissions (Elvidge et al., 1997a). Given the broad sustainability implications of increases in artificial lighting, research programs are emerging that examine this phenomenon from a range of disciplinary perspectives (e.g. www.verlustdernacht.de). However, strategies and policies for the management of artificial lighting are less comprehensive than might be expected (Lyytimaki et al., 2012). The lack of high resolution mapping of artificial lighting is increasingly recognised as an important barrier to related research and management (Elvidge et al., 2007). Datasets exist globally at a coarse spatial (~3km) and spectral resolution, allowing broad variations in urban lighting to be detected (Cinzano et al., 2001); but sub-city patterning cannot be explored effectively (Elvidge et al., 2007, Sutton et al., 2007). Numerous colour photographs are available from the

International Space Station with a spatial resolution of up to 60m (Elvidge et al., 2007). Although these images are starting to be used to detect demographic patterns within urban areas (Levin and Duke, 2012), individual lamps still cannot be identified (Elvidge et al., 2007). Finer spatial resolution data do exist, but typically have a limited spatial extent (Elvidge et al., 2007, Barducci et al., 2003) (but see (Kuechly et al., 2012)). This hinders the development of a strong evidence base to support urban lighting strategies, as cities can be highly heterogeneous even at fine spatial scales (Wu et al., 2011, Zhang et al., 2004). For example, little is known about how lighting varies with urban land-use (Kuechly et al., 2012, Doll, 2008, Luginbuhl et al., 2009) or along built density gradients. Improved baseline urban lighting maps are also needed in order to apply the results of published lighting research e.g. (Davies et al., 2012), to implement existing planning guidance on urban lighting zones (Luginbuhl et al., 2009, ILE, 2005), to enforce planning consents and legislation related to lighting nuisance (Morgan-Taylor, 2006, Flanders Government, 2012) and to monitor changes over time. Therefore, there is a need to secure lighting datasets at the city scale; and at a spatial and spectral resolution sufficient to advance lighting research and the planning and governance of urban lighting.

In this study our aims were: 1) to develop a method for securing fine resolution urban lighting datasets and 2) to undertake a city case study exploring how lighting varies with built density and land-use. Here, we present the finest resolution multi-spectral night-time photograph of an entire city, processed to derive estimates of surface illuminance and the location and nature of individual lamps. We found positive relationships between artificial lighting indicators and built density at coarse spatial scales, whilst at a local level lighting varied significantly with land-use.

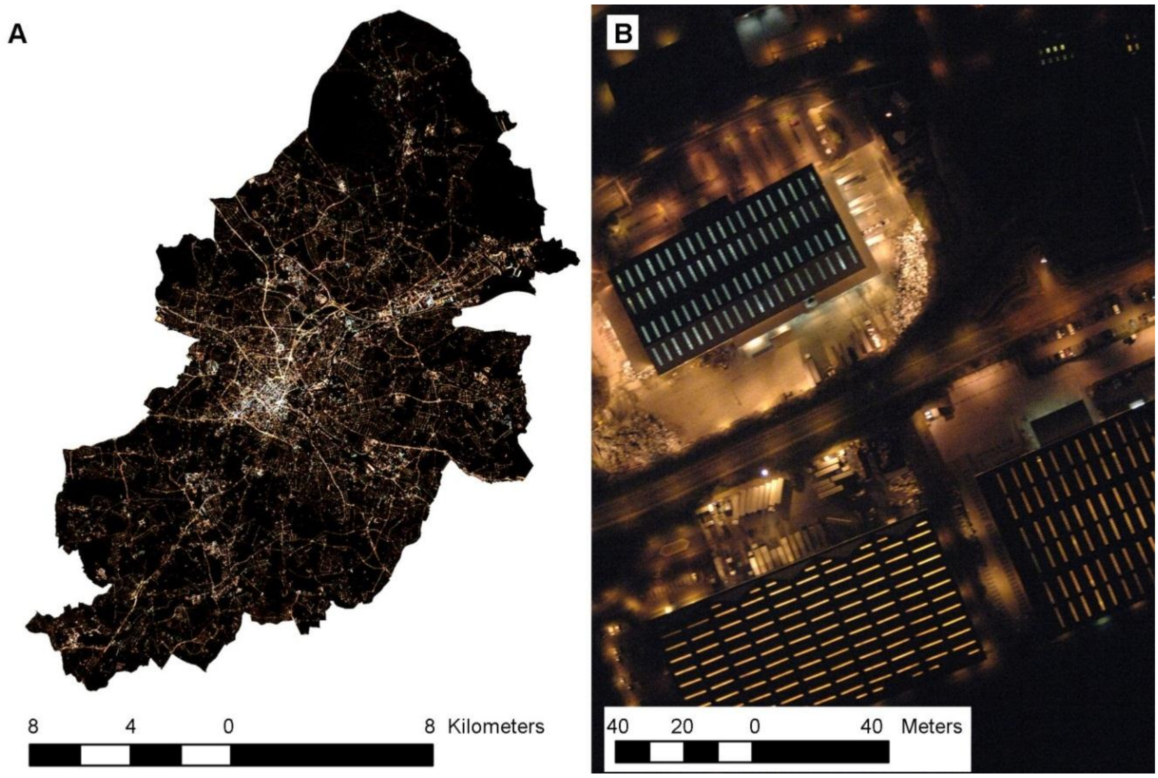
2. Methods

2.1 Data collection and processing.

Aerial night photography was collected in March 2009 by the UK Environment Agency (Fig.1). The target area was Birmingham, a large city (268km^2) within the highly urbanised West Midlands metropolitan county of the United Kingdom. Surveys were undertaken by plane at a height of ~900m, using a colour Nikon D2X digital camera, a 24mm AF Nikkor lens and a 1/100ths exposure. The resulting RGB images were orthorectified, mosaiced and re-sampled from 10cm to 1m pixel resolution. This single image was then processed to derive two landscape indicators of artificial lighting: a raster layer representing incident surface lux and a point layer representing the location and class of individual lamps. These indicators were considered to be of broad interest for those studying and managing lighting in urban landscapes.

Field surveys of ground incident lighting were undertaken in order to develop these indicators, using a USB2000+VIS-NIR Spectrometer (Ocean Optics, Florida, USA). Surveys were stratified over a range of lamps types located in both dense urban and residential neighbourhoods. Starting below each lamp, ground measurements of incident lux (lx) were collected at 1m intervals along a linear transect (total 400 measurements). Using a GIS (ArcGIS 9.2, ESRI Redlands, USA) these point survey data were superimposed onto a single band (greyscale) raster, generated by

Figure.1



Aerial night photography examples. (A) The city of Birmingham and (B) a retail distribution centre. Copyright Environment Agency 2009. All rights reserved.

averaging pixel values from the RGB image of the city using ER Mapper 7.2 (ER Mapper, San Diego, USA). The pixel value below each point was then extracted, allowing the relationship between incident lux and pixel value to be modelled. Model fit was found to improve when the measurements taken between 0 and 2m from the lamp were removed. This was likely due to inconsistent signal sources for the camera; in some cases the signal coming directly from unshielded lamps whilst in others from light reflected by the surfaces below a shielded lamp. The equation for the final model (Fig. S1) was then used to reclassify the greyscale raster to represent incident lux (hereafter referred to as the “lux layer”). To derive an estimate of noise we extracted raster summaries for 25ha of the greyscale raster corresponding to areas of the landscape known to be unlit. For these dark locations, 99% of greyscale pixels had values of less than 20 (Fig. S2). Pixel values < 20 were therefore considered to be unlit for the purposes of the landscape analysis. Three raster layers were generated representing areas lit to ≥10, ≥20 and ≥30lx.

To identify the point location of all lamps within the landscape, we used the focal statistics and raster calculator tools in ArcGIS to identify the brightest pixels at a processing resolution of 10m. First, a focal maximum layer was created using a circular roving window of 10m radius. The raster calculator tool was then used to identify pixels in this focal maximum layer whose values were identical to the original greyscale raster, which were then reclassified into a binary raster layer representing potential lamp locations (the candidate lamp layer). A 10m sample radius was chosen because street lamps are typically spaced at greater intervals and it was also found that this reduced the occurrence of false lamp signals due to highly reflective surfaces. Although generating this layer succeeded in identifying lamp locations, a large proportion of the candidate lamp pixels did not correspond to a lamp. These were the result of small variations in greyscale pixel values within dark areas such as parks and gardens. To address this, statistics for a selection of confirmed lamp locations were compared to a sample of these ―dark” locations. Focal statistics layers were created from the greyscale raster as well as from the individual red, green and blue layers of the mosaiced night photograph. These layers were generated using circular neighbourhoods of radii up to 7m, as well as annuli that excluded the neighbourhood centre. Using a CHAID classification tree (SPSS 18.0), we found that the majority (95.4%) of locations representing lamp centres had average green pixel values between 1m and 2m from the lamp of ≥ 14 whilst the majority (99.8%) of locations within unlit areas had values for this measure of < 14. This threshold was therefore used to remove dark locations within the candidate lamp layer and the remaining pixels were converted to a point layer representing 117,599 lamp centres within the city.

Elvidge et al. (2010), demonstrated the potential for discriminating major lamp types by using a 3 band sensor that broadly covered the visible light spectrum. Whilst the RGB bands in our image did not correspond exactly to the band widths proposed by Elvidge et al. (2010), we considered it feasible that they would be sufficient to differentiate between the major classes of street lamps present in the city: mercury vapour (MV), metal halide (MH), low pressure sodium (LPS) and high pressure sodium (HPS). Focal statistics were extracted for 240 lamp centres of known class and a CHAID classification tree was used to differentiate between lamp types (Fig. S3). The first discriminating variable was the green to red ratio (G:R) for pixels up to 1m from the lamp centre. A G:R of 0.96 separated the orange lamps (LPS and HPS) from white lamps (MH and MV), with an accuracy of 98.3% in both cases. LPS and HPS lamps were then differentiated based on the maximum greyscale pixel value between 2 and 4m from the lamp centre. Values <= 48 indicated an LPS lamp (96.7% correct), whilst HPS lamps typically had values > 48 (81.7% correct) (Table 1). MH and MV were differentiated based on the average blue pixel value up to 1m from the lamp centre. Values > 33.2 gave a 93.3% correct classification for MH, whilst values <= 33.2 gave a 98.8% correct classification for MV. These thresholds were then used to classify all city lamp centres into the 4 broad lamp classes.

Table. 1

Observed		**Predicted lamp class**				
Lamp class	Sample	LPS	HPS	MH	MV	Percentage Correct
LPS	60	58	1	0	1	96.7%
HPS	60	10	49	0	1	81.7%
MH	60	1	0	56	3	93.3%
MV	60	1	0	0	59	98.3%

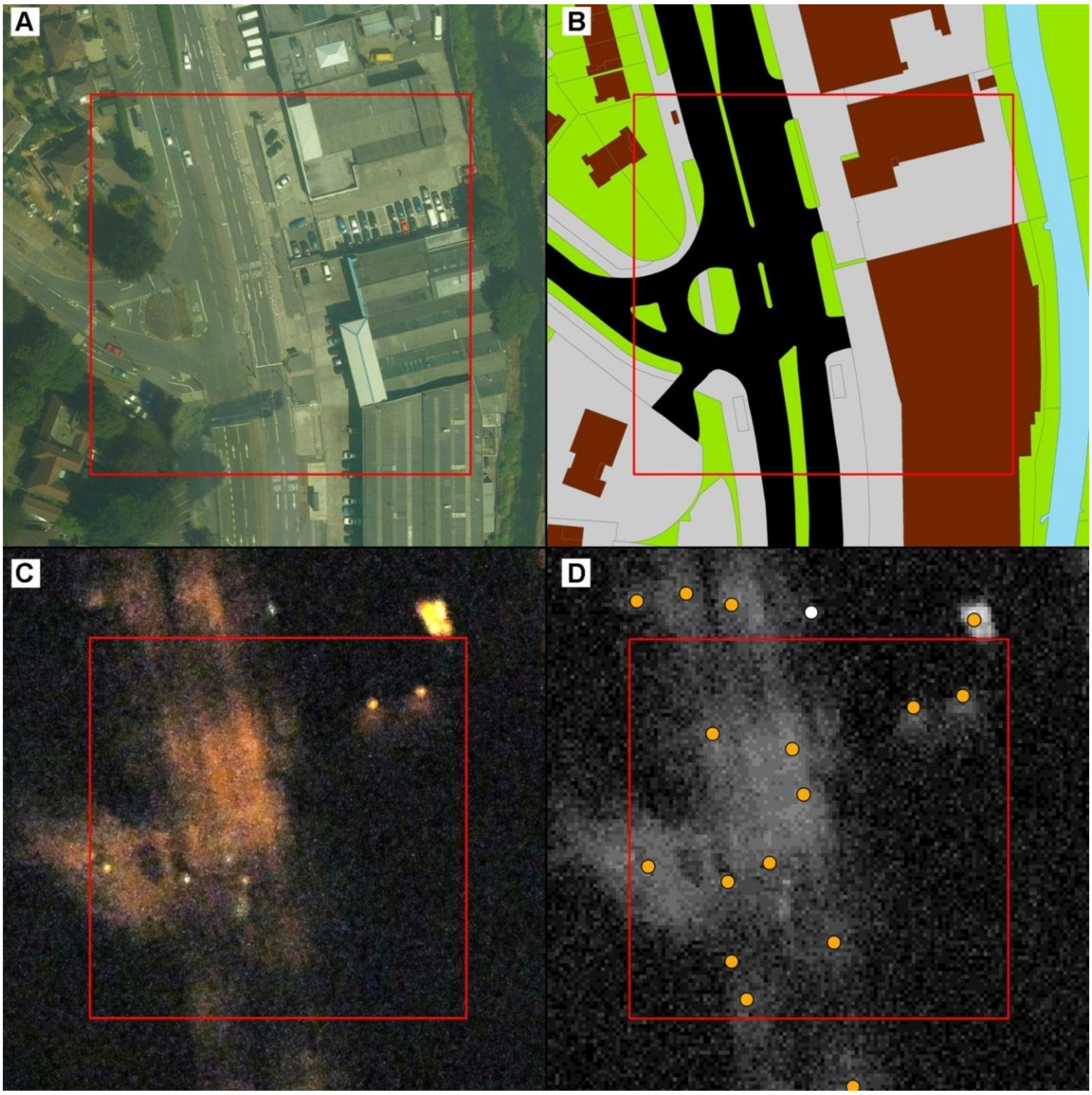
Classification of lamps using pixel values from the aerial night photograph, corresponding to individual lamp locations. Accuracy is estimated based on a sample of 60 known lamps for each lamp class. LPS = low pressure sodium, HPS = high pressure sodium, MH = metal halide, MV = mercury vapour.

2.2 Landscape analysis.

The sampling strategy was intended to reflect key scales and boundaries of urban ownership, management and decision-making (Cadenasso et al., 2007, Andersson, 2006). GIS analyses were undertaken to explore patterns between two broad lighting metrics (lit area and number of lamps) and measures of urban composition. To explore the effect of urban density, Ordnance Survey MasterMap (OSMM) land-cover and land-use parcels that were dominated by built land-cover (e.g. roads, car parks and buildings but not gardens) were combined into a single built category. These were then converted to a 1m resolution raster representing built land-cover for the entire city. Grid squares of increasing size (0.01km^2, 0.25km^2, 1km^2 and 4km^2) were then used to extract summaries of built land-cover and lighting metrics. Because broad urbanisation gradients typically fail to capture the effect of different land-uses (Alberti, 2005), we employed a complementary approach to measuring urban performance by sampling the lighting of land-use units. Importantly, we used two contrasting land-use classifications to maximise the utility of the results (Table S1);

National Land Use Database (NLUD) zones (Dunn and Harrison, 1995) and OSMM parcels. NLUD data included categories such as housing, manufacturing and education, which were available for the entire city as 100m grid squares ($0.01km^2$). OSMM parcels were typically smaller than $0.01km^2$ and irregularly shaped, representing features such as gardens, pavements and buildings. OSMM parcels were grouped to reflect five broad management categories (Table S1). Each $0.01km^2$ NLUD land-use zone was therefore typically composed of a number of smaller OSMM land-use parcels (Fig. 2). Lighting indicator summaries were extracted for both the land-use zones and land-use parcels at the city scale. These were used to estimate the percentage contribution of different land-uses to the total number of lamps and total lit area within the city. In addition, we calculated the lamp density and percentage lit area for each land-use zone and parcel type. These provided an indication of how intensely lit different land-uses were, irrespective of how much they contributed to lighting at the city scale.

Figure. 2



Aerial photography, mapping and lighting indicators for a 100m square manufacturing zone and road intersection. (A) A daytime aerial photograph (© Bluesky International Limited 2007), (B) Ordnance Survey MasterMap land-cover and land-use parcels (2008), (C) a night time aerial colour photograph (© Environment Agency 2009) and (D) a raster representing ground lux, overlain by a point layer representing lamp centres.

3. Results

3.1 Landscape scale patterns between lighting and built density.

8% of the total land surface of the city was found to be illuminated to at least 10lx. In addition, 65% of all lit surfaces (≥10lx) and 80% of all city lamps were directly associated with built land-cover. Lighting indicators demonstrate positive and often non-linear relationships with the density of built land-cover. The percentage of lit area increases in a non-linear fashion along these urbanisation gradients (Figs 3A & C), whilst lamp density increases linearly (Fig. 3B). As the scale of sampling (window size) increases, the fit of these models improves; although the relationships remain essentially the same (Fig. 3A & B). The results for sampling at the 0.01km^2 scale are presented for comparison in figure S4. The percentage of each sample square that is lit to ≥10lx rises from ~5% in rural or semi-natural areas to ~30% in densely built areas (Fig. 3A). Similarly, lamp density rises from ~0 lamps/ha in rural or semi-natural areas to ~15/ha in densely built areas (Fig. 3B). The composition of lamp types also changes along the 1km^2 urban gradient (Fig. 3D), with LPS lamps dominating provision at low built densities, shifting to (broader spectrum) HPS and MH lamps in densely built areas. Changes in the density of individual lamp types along the 1km^2 urban gradient are presented in figure S5.

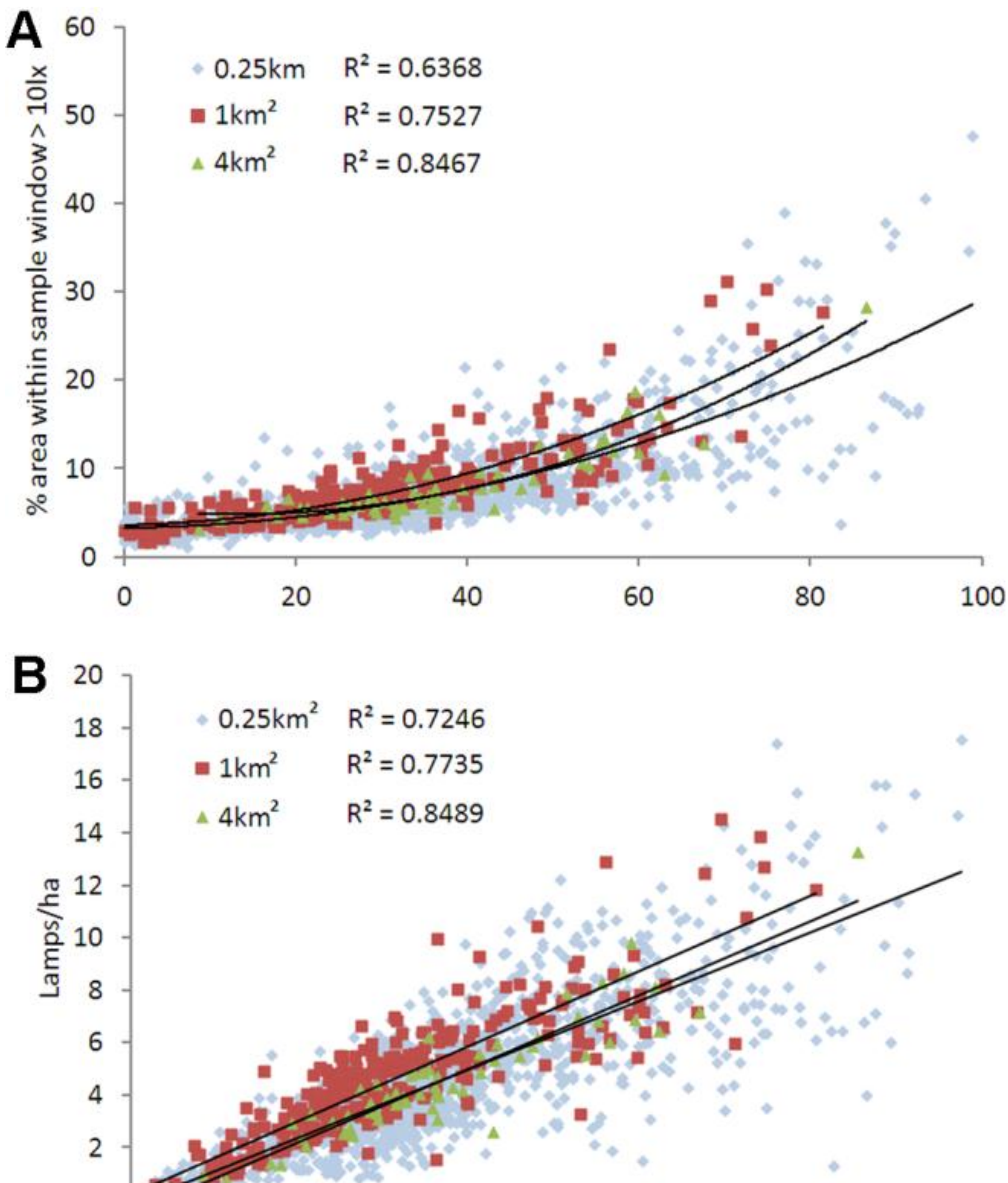
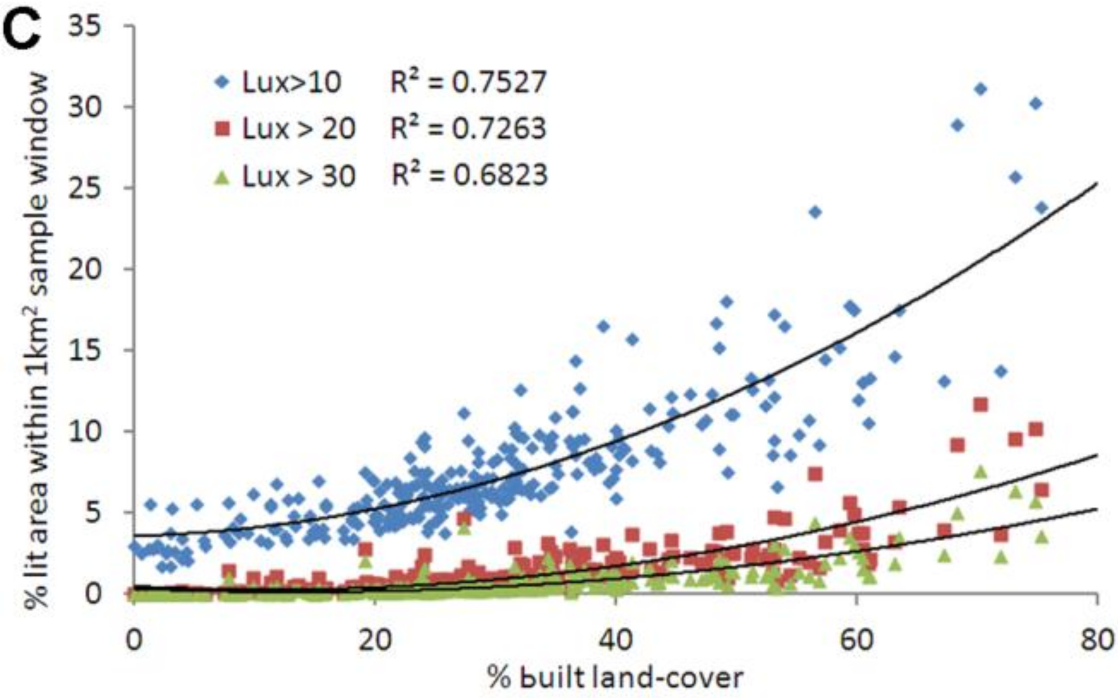


Figure. 3. Percentage built land-cover plotted against a variety of lighting metrics. (A) Percentage lit area (≥10lx) sampled at $0.25km^2$, $1km^2$ and $4km^2$ scales. (B) Density of lamps sampled at $0.25km^2$, $1km^2$ and $4km^2$ scales.



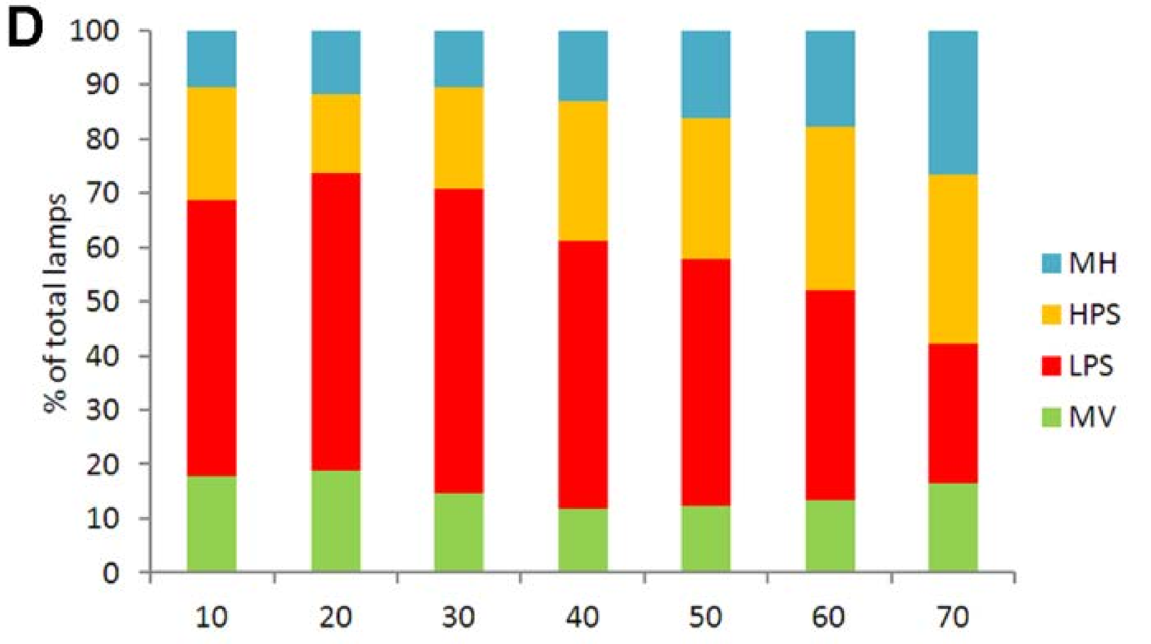
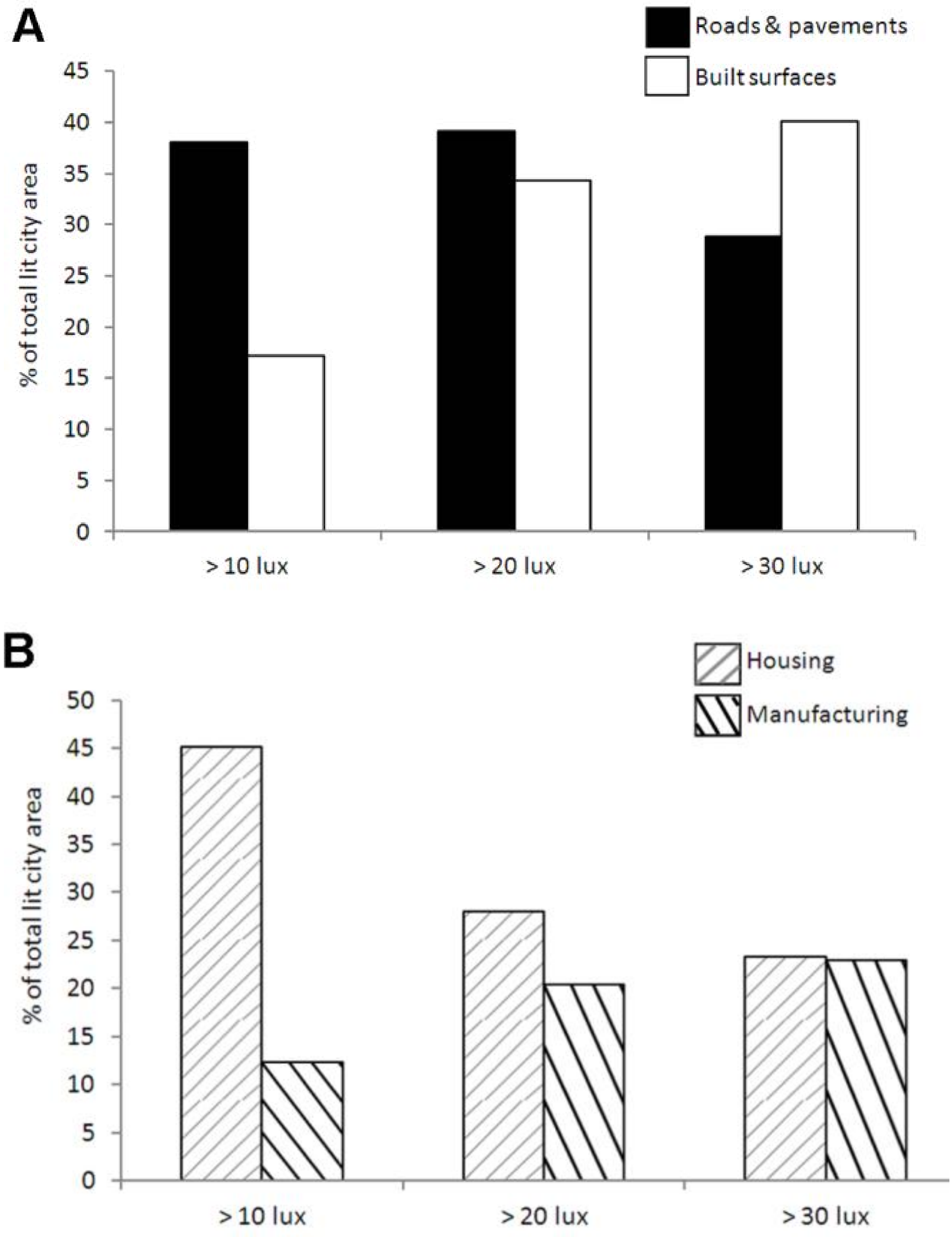


Figure. 3 ctn. Percentage built land-cover plotted against a variety of lighting metrics. (C) Percentage lit area ≥10, ≥20 and ≥30lx at the 1km^2 scale. (D) Lamp class sampled at the 1km^2 scale. LPS = low pressure sodium, HPS = high pressure sodium, MH = metal halide, MV = mercury vapour. Y axis values are standardised as a percentage of the total number of lamps within each built density class. Built density values represent class maximum (10 = 0-10% built land cover).

3.2 Lighting and land-use.

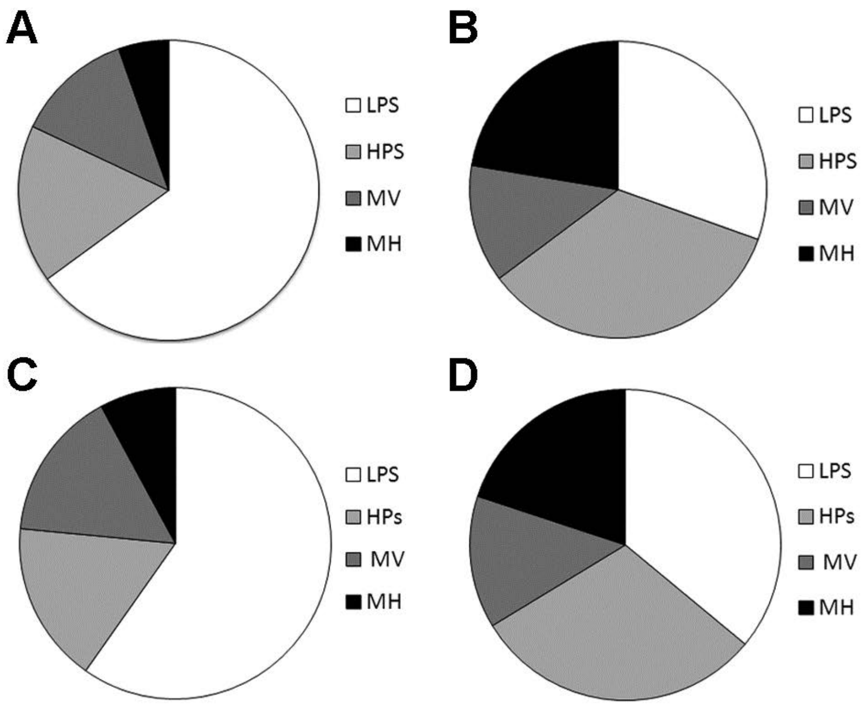
The contribution of different OSMM land-use parcels to the total lit surface area within the city varied, with roads/pavements (38%) and other built surfaces such as car parks (17%) contributing the majority of the total area ≥10lx (Fig. 4A). These land-uses are also the main sources of the city's brighter lighting, although roads/pavements are responsible for a lower percentage (29%) of areas ≥30lx than built surfaces (40%) (Fig. 4A). For NLUD land-use zones, housing (45%) and manufacturing (12%) areas were responsible for the majority of city lighting ≥10lx and approximately equal proportions of areas ≥30lx (Fig. 4B). The distribution of lamps between land-uses is similar to that for lit areas, with the majority of city lamps being associated with OSMM roads/pavements (52%) and other built surface parcels (14%). LPS lamps dominate the lighting of roads/pavements (Fig. 5A), whilst the lamp types associated with other built surface parcels are more evenly spread between LPS, HPS and MH (Fig. 5B). When considering NLUD land-use zones, 55% of city lamps are associated with housing and 11% with manufacturing. Whilst LPS lamps dominate lighting provision within housing zones (Fig. 5C), the lamps in manufacturing zones are more evenly divided between LPS, HPS and MH (Fig. 5D). A more detailed breakdown of lighting and land-use at the city scale is presented for comparison in Table S1.

Figure. 4.



Percentage contribution of land-uses to the total area of the city ≥10, ≥20 and ≥30lx. (A) Roads/pavements and built surface Ordnance Survey MasterMap (OSMM) land-use parcels. (B) Housing and manufacturing National Land Use Database (NLUD) zones.

Figure. 5



The relative proportions of lamp classes associated with different land-uses. (A) Roads/pavements and (B) other built surface OSMM land-use parcels, located within (C) housing and (D) manufacturing NLUD zones.

Although OSMM roads/pavements and other built surface parcels within NLUD housing and manufacturing zones are responsible for the majority of lighting within the city, other land-uses are still intensely lit and therefore may contribute significantly to lighting at local scales (Fig. 6). For example, although office land-use is limited in terms of urban areal extent (<1% of total city area) (Table S1), a $0.01km^2$ office zone typically has over twice the lamp density and five times the brightly illuminated surface area than the average land-use zone within the city (Fig. 6B). In contrast, a typical $0.01km^2$ area of housing (which is the dominant land-use zone in the city) has just half the brightly lit area than the city average.

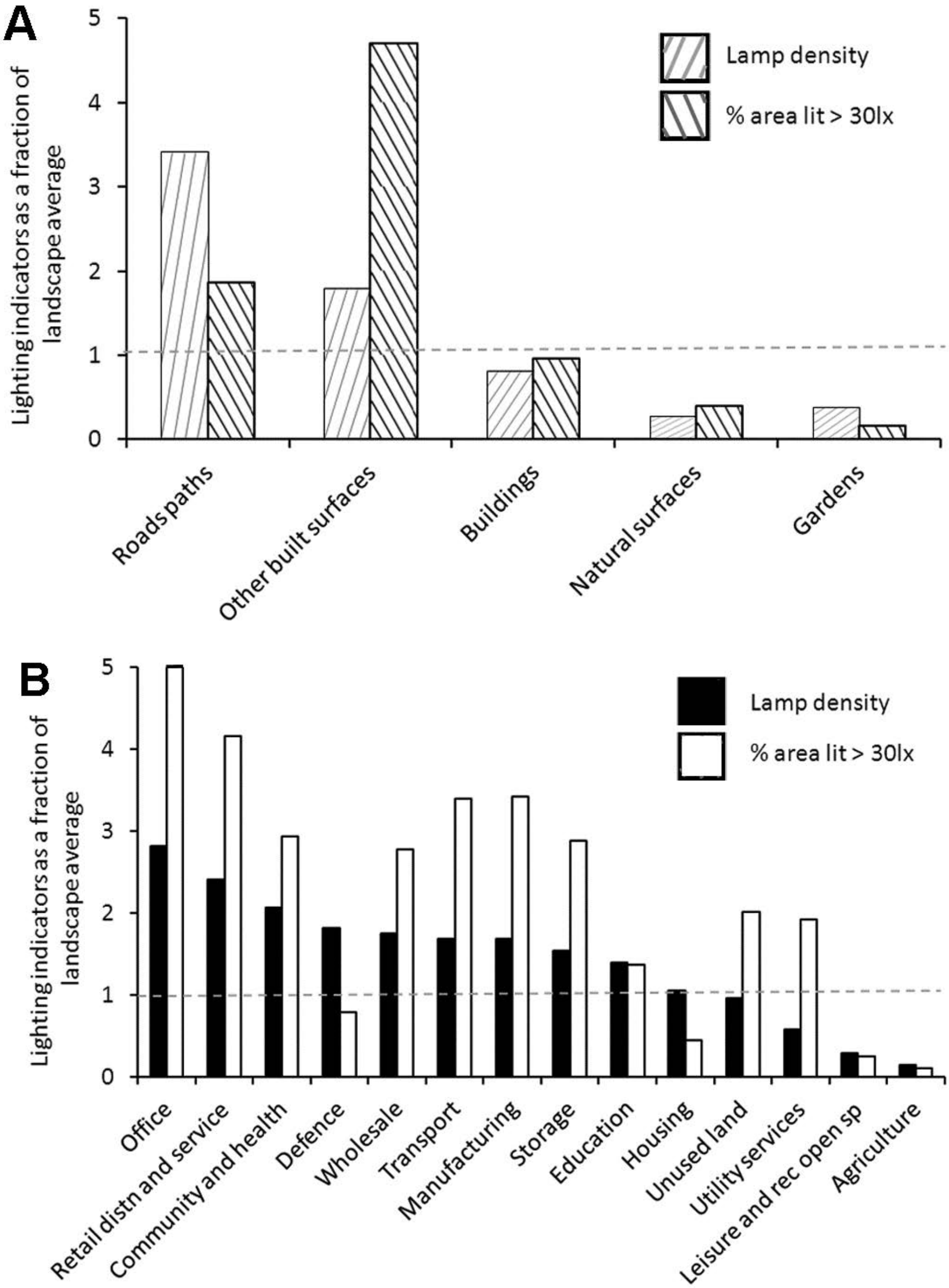


Figure. 6. Lamp density and percentage illuminated area (≥30lx) for total city area covered by different land-uses. (A) OSMM land-use parcels and (B) NLUD land-use zones. Values have been standardised, with values > 1 indicating abundance is greater than the landscape average.

4. Discussion

The earth is experiencing a step-change in artificial lighting provision (Holker et al., 2010, RCEP, 2009, Pimputkar et al., 2009). The replacement and expansion of lighting infrastructure raises the possibility of unintended and broad scale impacts on human health and wellbeing (Cinzano et al., 2001, Foster and Wulff, 2005) and on ecosystem processes (Longcore and Rich, 2004). Although beneficial for many social applications; strong, broad spectrum and extensive lighting at night can interrupt key physiological and behavioural processes for species of plants and animals, including humans (Falchi et al., 2011, Navara and Nelson, 2007). Point sources can be a cause of nuisance due to glare and lighting trespass (Morgan-Taylor, 2006) whilst diffuse sky glow can obscure views of the night sky (Cinzano et al., 2001) and eliminate natural cycles in lunar illumination (Kyba et al., 2011). It is therefore vital that baseline lighting data are collected, against which to measure these changes and to support research into understanding the implications for social and ecological systems. A major advance has been the collection of global data on the extent and magnitude of night lighting (Cinzano et al., 2001). However, many key urban research questions require higher resolution data (Elvidge et al., 2007). Advances in high-specification digital camera technology have now made broad-scale aerial night photography a possibility (Kuechly et al., 2012). For the first time we are able to explore patterns between lighting and urban land-use, using metrics and scales that are relevant to those involved in research, planning and management of cities.

4.1 Built density

The results of this case study indicate that high built densities are associated with more extensive, brighter and broader spectrum lighting. This has implications for debates about the sustainability performance of the compact city (Neuman, 2005); with the economies that arise from dense urban development (Williams, 1999) potentially being accompanied by greater light pollution. The co-variability between lighting and built density also has implications for studies employing urban-rural gradients (McDonnell and Pickett, 1990); which should take steps to avoid potential confounding effects of lighting on the social or environmental variables of interest. At fine spatial scales (< $0.25km^2$), built density is a poorer predictor of urban lighting. Spatial patterning is therefore nested, with small dark spaces existing even within densely built, brightly lit neighbourhoods. Lighting at fine scales is socially and ecologically relevant and appears to be related to land-use.

4.2 Land-use

The results of our analysis of OSMM parcels and NLUD zones illustrate which land-uses are predominantly responsible for lighting at the city scale and which have a strong local effect. As might be expected, roads/pavements and other built surface parcels within housing and manufacturing zones are responsible for a large proportion of the lamps and brightly lit surfaces within the city, reflecting the role that lighting plays in transport, safety and building security. This suggests that these land-uses should be the target of proactive strategies to reduce light pollution, such as dimming (RCEP, 2009), shielding (ILE, 2005) and switch-off (Smith, 2009). The large-scale replacement of LPS suburban street lighting underway in the UK presents an opportunity to reduce some aspects of light pollution, although it may cause others to increase. The replacement lamps are generally well shielded (Mizon, 2002), and their timing and brightness is more easily altered. However, public opposition to switch-off has been considerable (RCEP, 2009). The use of broader spectrum lamps is being driven by the desire for improved colour perception, but may result in greater disturbance to natural processes (Falchi et al., 2011). Whilst efforts to address current light pollution should continue to focus on suburban street lighting, our research suggests that the security lighting of manufacturing areas may warrant similar attention. These areas occupy a small fraction of the city with relatively few lamps, yet are responsible for a large proportion of bright urban lighting. Concerns have already been raised about light pollution arising from the security lighting of commercial areas (Mizon, 2002, RCEP, 2009, Luginbuhl et al., 2009), and our study provides strong evidence that this is an issue at the city scale. Retail and distribution land-use zones alone account for 11% of all brightly lit surfaces, rising to 34% when manufacturing areas are included; yet these account for just 10% of the city landscape (Table S1). Similar results have been found for central Berlin (Kuechly et al., 2012). In contrast to street lighting, modifications to the positioning and triggering of security lamps may well be more publicly acceptable as well as more effective from a security perspective than current practice (Mizon, 2002, Morgan-Taylor, 2006). Although they are relatively infrequent land-uses within the case study area; built surfaces within office, retail, transport, community/health, manufacturing and storage zones have lamp and lighting densities that are considerably higher than the landscape average. This has implications for land-use planning as such development may have strong local effects; and future growth in the service and retail sectors has the potential to deliver greater pollution at the city scale.

Whilst useful for raising awareness of the likely lighting implications of development proposals, it is still not known how well these findings transfer between cities and to what extent the lighting characteristics of land-uses described here are fixed. For example, large-scale replacement of lighting infrastructure in the future is likely to result in brighter and broader spectrum lighting (RCEP, 2009, Pimputkar et al., 2009), although the reverse may be true in some cases (Luginbuhl et al., 2009).

4.3 Future applications of urban lighting indicators

Although not the focus of this paper, there are a range of additional research and planning applications for the lighting datasets described here. Light maps have the potential to address several topical issues in urban studies and the diversity of applications for remotely sensed lighting data is illustrated by research resulting from the interdisciplinary EU MANTLE project (Doll, 2008). Similar questions might be addressed using higher resolution data, but as urban relationships and management priorities can be scale dependent, additional questions might also be explored. Higher resolution data have the potential for characterising urban forms (Kruse and Elvidge, 2011) and for generating lighting inventories for infrastructure management. They might also be used to scale the results of field studies and research experiments to explore their implications for an entire city. Remotely sensed lighting maps are considered to be unique in their ability to reflect human activity (Doll, 2008). As research into urban areas tends to underplay their social dimensions (Alberti, 2005), the collection and use of lighting maps may help to better integrate these aspects into the modelling of urban systems. From an applied perspective, high resolution mapping would also enable the development of more sophisticated urban lighting masterplans, tailoring lighting to meet the needs of the community at a fine spatial scale and to improve urban lighting governance (Morgan-Taylor, 2006, Aubrecht et al., 2010). Changes to the nature and operation of lighting infrastructure also have the potential to permit considerable financial and carbon savings (Gallaway et al., 2010), although some public opposition might be expected (RCEP, 2009). How environmental information is presented can be key to facilitating behavioural change (RCEP, 2007) and striking images of cities, neighbourhoods and streets at night could play a useful role in encouraging a broader social debate about lighting, energy and climate change. Combined with analyses such as those presented here, these images may also be useful in challenging false assumptions on the causes and magnitude of artificial lighting and its associated impacts (Lyytimaki et al., 2012).

Artificial lighting can play an important role in shaping urban sustainability, yet little is known about how it varies with land-cover and land-use. In this paper we have demonstrated that aerial night photography can be effective in clarifying these relationships and in challenging conventional approaches to tackling unnecessary or problematic urban lighting.

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Supporting information

Supporting information/supplementary figures can be found in the Appendix at the end of this thesis.

Figure S1. Ground incident lux plotted against corresponding greyscale pixel value for survey locations within Birmingham. The equation for the best fit line (y = 0.0128X2+0.2246X +0.8517) was used to reclassify the greyscale raster. R2 = 0.9146. A 95% confidence interval is also indicated.

Figure S2. The distribution of greyscale pixel values for known —dark” locations (lit to <1lx).

Figure S3. CHAID classification tree for lamp classes. 1 = low pressure sodium (LPS), 2 = high pressure sodium (HPS), 3 = metal halide (MH) and 4 = mercury vapour (MV). The first discriminating variable was the green to red ratio (G:R 0–1 m) for pixels up to 1 m from the lamp centre. LPS and HPS were then differentiated based on the maximum greyscale pixel value between 2 and 4 m (GS 2–4 m) from the lamp centre. MH and MV were differentiated based on the average blue pixel value up to 1 m from the lamp centre (BL 1 m).

Figure S4. The results for sampling of lighting metrics at the 0.01 km^2 scale. (A) Percentage area ≥10lx and (B) density of lamps, both plotted against percentage built land-cover.

Figure S5. Changes in the density of lamp classes along the 1 km^2 urban gradient. (A) MH and LPS lamps and (B) MV and HPS lamps.

Table S1. Land-uses and lighting metrics for the city of Birmingham. Land-uses are given as a fraction of total city area, along with their contribution to the total city area lit ≥30lx and to the total number of city lamps. Two alternative measures of land-use are given; land-use parcels based upon the Ordnance Survey MasterMap (OSMM) (2008) and land-use zones based on the National Land Use Database (NLUD) categories (1995).

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PAPER IV. THE ECOLOGICAL IMPACT OF CITY LIGHTING SCENARIOS: EXPLORING GAP CROSSING THRESHOLDS FOR URBAN BATS.

James D. Hale[1], Alison J. Fairbrass[2], Thomas J. Matthews [34], Gemma Davies[5], Jon P. Sadler[1]

1. School of Geography, Earth and Environmental Sciences, The University of Birmingham, Birmingham, West Midlands, United Kingdom.
2. Centre for Urban Sustainability and Resilience, University College London, London, United Kingdom.
3. School of Geography and the Environment, University of Oxford, Oxford, Oxfordshire, United Kingdom.
4. Centre for Ecology, Evolution and Environmental Change / Azorean Biodiversity Group, Universidade dos Açores, Angra do Heroísmo, Portugal.
5. Lancaster Environment Centre, Lancaster University, Lancaster, Lancashire, United Kingdom.

Abstract

As the global population urbanises, dramatic changes are expected in city lighting and the urban form, which may threaten the functioning of urban ecosystems and the services they deliver. However, little is known about the ecological impact of lighting in different urban contexts. Movement is an important ecological process that can be disrupted by artificial lighting. We explored the impact of lighting on gap crossing for *Pipistrellus pipistrellus*, a species of bat (Chiroptera) common within UK cities. We aimed to determine whether the probability of crossing gaps in tree cover varied with crossing distance and lighting level, through stratified field surveys. We then used the resulting data on barrier thresholds to model the landscape resistance due to lighting across an entire city and explored the potential impact of scenarios for future changes to street lighting. The level of illumination required to create a barrier effect reduced as crossing distance increased. For those gaps where crossing was recorded, bats selected the darker parts of gaps. Heavily built parts of the case study city were associated with large and brightly lit gaps, and spatial models indicate movement would be highly restricted in these areas. Under a scenario for brighter street lighting, the area of accessible land-cover was further reduced in heavily built parts of the city. We believe that this is the first study to demonstrate how lighting may create resistance to species movement throughout an entire city. That connectivity in urban areas is being disrupted for a relatively common species raises questions about the impacts on less tolerant groups and the resilience of bat communities in urban centres. However, this mechanistic approach raises the possibility that some ecological function could be restored in these areas through the strategic dimming of lighting and narrowing of gaps.

1 Introduction

Urban areas are now home to over half of the world's population (UN, 2010), are the drivers behind much of the global CO_2 emissions and resource demands (Wackernagel et al., 2006; Hoornweg et al., 2011) and are highly modified environments (Grimm et al., 2008). They are therefore at the heart of debates about climate change, resource security, nature conservation and human wellbeing (Newman, 2006; Grimm et al., 2008; Hodson & Marvin, 2009; Glaeser, 2011). Given the diversity and complexity of change within urban areas (Dallimer et al., 2011), there is a need to explore how their sustainability performance might vary under alternative scenarios for their future structure and operation (Lombardi et al., 2012). In this paper we explore how the disruption of the nocturnal urban environment by different levels of artificial lighting can impact species movement – a key ecological process.

Growth, sprawl, compaction and fragmentation of the built form varies within and between urban areas (Williams, 1999; Luck & Wu, 2002; Couch et al., 2005; Irwin & Bockstael, 2007; Adams et al., 2010; Seto et al., 2011) and changes in built extent, density and land-use may occur over relatively short time periods (Pauleit et al., 2005; Seto & Fragkias, 2005; Dallimer et al., 2011). In addition to shifts in urban form, changing technologies and social practices also radically alter urban environments (Gandy, 2004). One important example is outdoor artificial lighting, a pervasive yet diverse characteristic of cities that is changing in many regions (Bennie et al., 2014a; Kyba et al., 2014). Remotely sensed measures of light emissions from the earth's surface have been found to correlate with built land-cover (Hale et al., 2013), population density (Sutton et al., 1997), electric power consumption (Elvidge et al., 1997) and per capita income (Ebener et al., 2005). Outdoor artificial lighting also varies considerably within cities depending on land-cover and land-use (Luginbuhl et al., 2009; Kuechly et al., 2012; Hale et al., 2013; Levin et al., 2014). Intensification and expansion of lighting is evident at both local and global scales (Hölker et al., 2010a; Bennie et al., 2014a), a process fuelled by the emergence of cheaper and more efficient lighting technologies (Tsao et al., 2010; Kyba et al., 2014). The large-scale introduction of such technologies would also be expected to result in changes to the dominant spectral composition of outdoor lighting (Stone et al., 2012). However, despite a broad trend of growth in artificial lighting, some locations are becoming darker (Bennie et al., 2014a) as lamps are shielded, dimmed or even removed to reduce light pollution, running costs and carbon emissions (RCEP, 2009; Falchi et al., 2011; Gaston et al., 2012). Changes in artificial lighting can impact city performance in a variety of ways (Smith, 2009; Falchi et al., 2011), yet many of the potential sustainability impacts remain unexplored (Hölker et al., 2010a; Lyytimaki et al., 2012). The nature of lighting infrastructure and its operation has obvious implications for energy demands and costs (Gallaway et al., 2010; Tsao et al., 2010). However, artificial lighting also has numerous positive and negative impacts on social practices and human health; lighting has enabled greater flexibility in the timing of work and leisure activities, although at the cost of disruption to circadian rhythms, behaviours and physiological processes (e.g. Navara & Nelson, 2007; Falchi et al., 2011; Cho et al., 2013). Less is known, however, about how natural systems are disturbed and the resulting effects on ecological function and service provision (Rich & Longcore, 2006; Hölker et al., 2010b; Gaston et al., 2012).

In this paper we focus on ecological impacts of artificial lighting in urban areas and explore how these may vary with different levels of illumination and configurations of the built form.

The value of the semi-natural components of cities is increasingly recognised, particularly from the perspective of those ecosystem functions with strong links to human wellbeing (Carpenter et al., 2006; Sadler et al., 2010; Haase et al., 2014). Given the known effects of artificial lighting on a variety of species and habitats (Longcore & Rich, 2004; Hölker et al., 2010b; Gaston et al., 2012) and the rapid changes to urban street lighting already underway, research is needed that explores the potential disruption of ecological processes at the city-scale. Individuals of most species are sensitive to natural cycles of day and night (Hölker et al., 2010b), with light acting both as information and a resource (Gaston et al., 2013). For some species the disruption of these cycles by artificial lighting can impair particular parts of their life history e.g. feeding and growth (Boldogh et al., 2007), commuting to foraging sites (Stone et al., 2009) or the timing of reproduction (Kempenaers et al., 2010). Conversely, lighting can bring direct advantages such as concentrating prey (Blake et al., 1994; Jung & Kalko, 2010) or for diurnal and crepuscular species it may extend the hours of activity (Negro et al., 2000). A further complication is that lighting may deliver both costs and benefits to a single individual, making the net impact challenging to estimate. For example, artificial lighting has been found to delay roost emergence in the bat *Pipistrellus pygmaeus* (Downs et al., 2003), but also to provide foraging locations for the same species (Bartonička et al., 2008). Impacts on individual fitness may be sufficient to alter populations and even community composition (Perkin et al., 2011; Davies et al., 2012), with the potential to affect important ecosystem functions and services such as pollination (Eisenbeis, 2006) or seed dispersal (Lewanzik & Voigt, 2014). However, population or ecosystem-scale research related to artificial lighting is rare (Gaston et al., 2013; Lyytimäki, 2013). One further notable research gap relates to lighting thresholds for ecological impacts and their spatial extent (Gaston et al., 2013).

Here we examine the impact of lighting on animal movement within urban areas as movement is a process relevant to individual fitness, population resilience and to broader ecosystem structure and function (Nathan et al., 2008). Despite the importance of movement for enabling organisms to forage, disperse and ensure gene flow between populations, the direct measurement of functional connectivity is not always practical (Nathan et al., 2008; Zeller et al., 2012). Tracking and genetic studies may provide evidence that some patches within a landscape are functionally connected, but on their own these approaches fail to explain why movement may have been recorded in some contexts but not in others. Understanding the factors that affect movement between habitat patches is therefore important for conservation practice (Rayfield et al., 2010; Watts et al., 2010), particularly in landscapes undergoing rapid environmental change (Zeller et al., 2012). This can be highly complicated as movement may not only depend on patterns of land-cover and land-use within a landscape, but also on the motivation and ability of individuals to move (Tischendorf & Fahrig, 2000; Nathan et al., 2008; Pe'er et al., 2011).

The flight behaviour of several bat species may be influenced by artificial lighting (Kuijper et al., 2008; Stone et al., 2009, 2012; Polak et al., 2011) which can cause deviation of the flight path to avoid the most heavily lit area (Kuijper et al., 2008; Stone et al., 2009), or barrier effects where approaching bats turn and fly in the opposite direction (Stone et al., 2009). Barrier effects on commuting bats have also been demonstrated for structures such as motorways that bisect habitat networks (Kerth & Melber, 2009). Several European bat species are known to fly along woodland edges and tree lines when commuting between their roost and feeding locations (Racey & Entwistle, 2003), and the activity of some species is higher with increasing proximity to these corridor features (Verboom & Spoelstra, 1999; Downs & Racey, 2006). This suggests that movement for nocturnal bat species might be simultaneously impacted by the structural fragmentation of habitat networks and by the artificial lighting of commuting routes, both of which are common within urban areas, potentially increasing levels of landscape resistance. Here we modelled the effect of both crossing distance and illumination level on the crossing behaviour of a common urban bat (*Pipistrellus pipistrellus*) at gaps in urban tree networks. The resulting model was then used to explore the landscape-scale implications of different urban lighting scenarios for movement.

The study objectives were to:

1) Determine whether the probability of bats crossing gaps in tree lines varies with crossing distance and illumination, and to model any barrier effects;

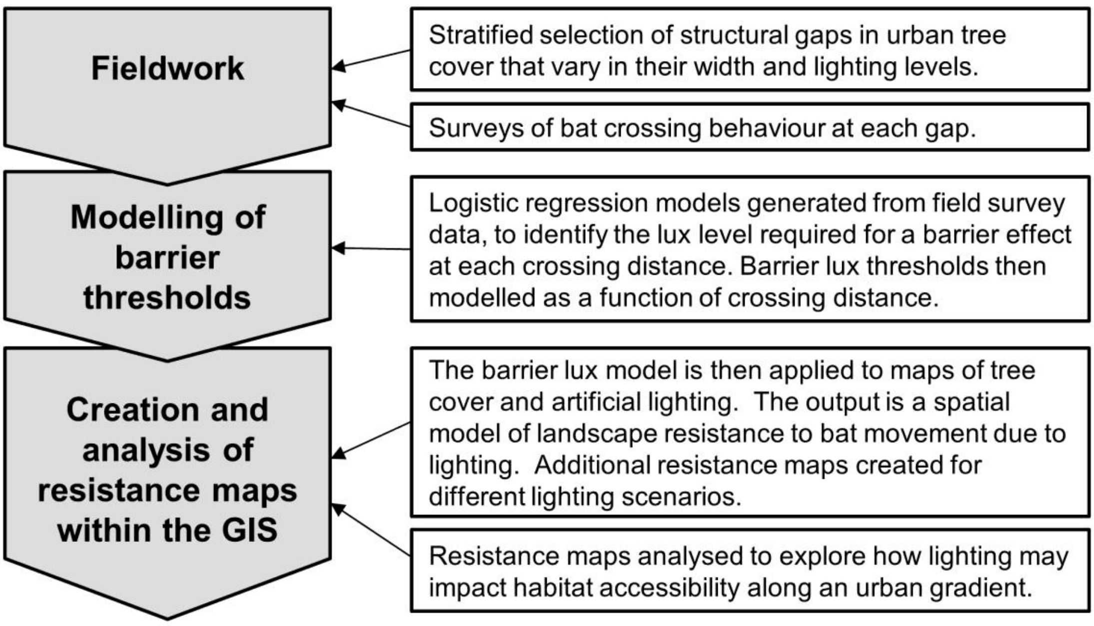
2) Develop spatial models for landscape resistance due to artificial lighting;

3) Explore the implications of these resistance models for habitat accessibility along an urban gradient.

2. Materials and methods

These methods are divided into five distinct sections (Fig. 1): (1) the selection of survey gaps within networks of urban tree cover, (2) surveys of gap crossing events by bats, (3) the development of statistical models for gap crossing probability that identify distance-dependant lighting thresholds for barriers to movement, (4) the translation of this barrier lux model into spatial GIS models for landscape resistance under contrasting lighting scenarios for a case study city, and (5) an analysis of how these scenarios for landscape resistance may impact habitat accessibility along an urban gradient.

Figure. 1



A flow diagram representing key steps within the methods

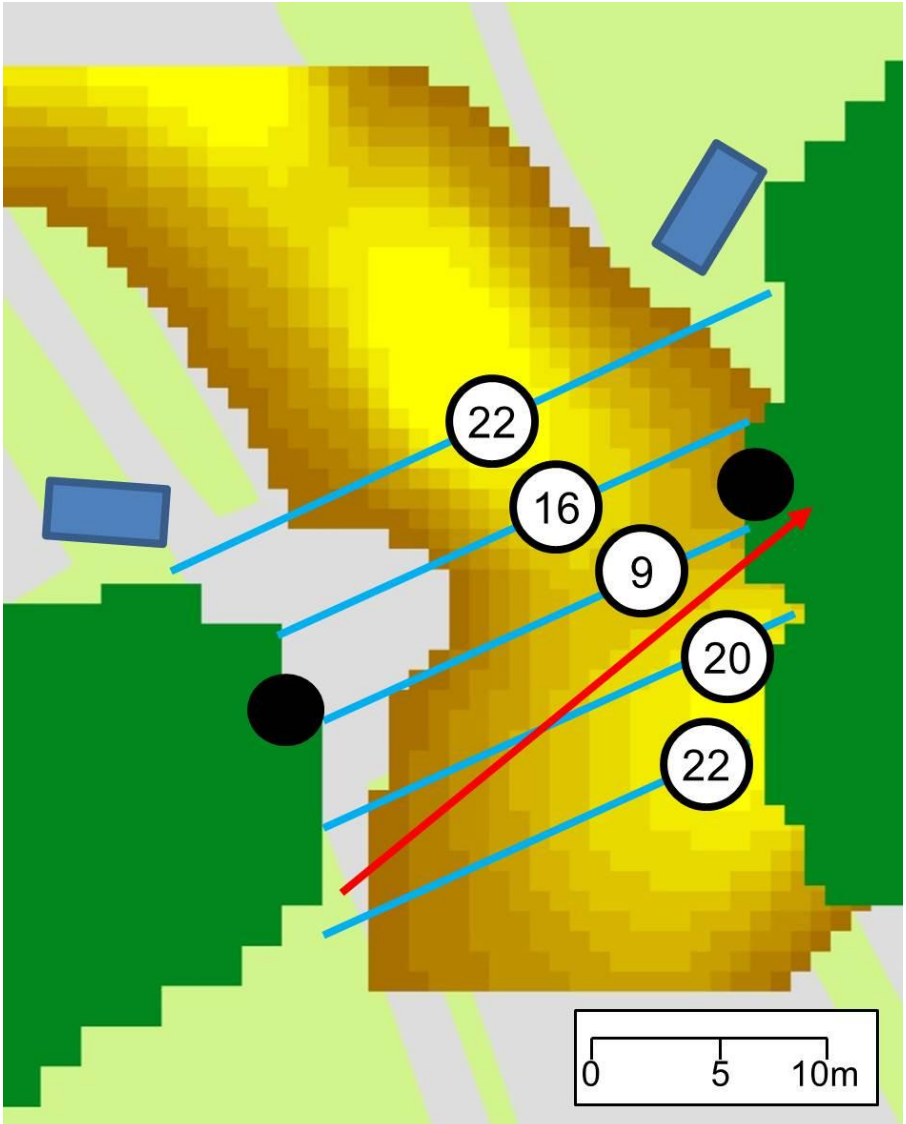
2.1 Selection of survey gaps

For *P. pipistrellus*, movement between resource patches is facilitated by linear features such as tree lines (Verboom & Spoelstra, 1999) and therefore the patch-matrix-corridor model (Forman, 1995) would appear to be an appropriate starting point for exploring some of the mechanisms that deliver functional connectivity for this species. A key assumption within this model is that the matrix creates resistance to the movement of individuals between habitat patches and that this resistance is reduced by the presence of linear habitat features that form structural connections between patches. To directly measure functional connectivity between bat roosts and feeding areas within an urban area would be extremely challenging, particularly as both feeding areas and roosts may be difficult to identify or gain access to, given their frequent association with private built infrastructure (Blake et al., 1994; Altringham, 2003). This therefore led us to focus on corridor features k*now*n to facilitate movement and to explore the degree to which structural gaps in these features and lighting within the intervening matrix could influence crossing behaviour. Field observations were undertaken in the summer of 2010 within the West Midlands of the United Kingdom (UK), a highly urbanised metropolitan county covering 902 km^2 with a population of ~2.3 million (S1). *P. pipistrellus* is a species of bat that is broadly distributed over Europe and the Near East (Altringham, 2003), is commonly found within UK cities and can be found throughout the UK West Midlands (Hale et al., 2012). It is nocturnal and easily surveyed and was therefore chosen as a model species for exploring the impacts of lighting on bat movement in urban areas. Bats were surveyed at gaps in networks of tree cover, as this species is known to follow the edges of tree lines when commuting between roosts and feeding areas (Downs &

Racey, 2006). Tree cover is ubiquitous within the West Midlands, with the exception of the most densely built areas. Trees are typically located along road edges, railway embankments and waterways, in gardens and recreational green spaces and within the broader amenity planting of commercial areas. Such trees are rarely isolated, but tend to form linear features that follow existing or historic land-use boundaries such as the perimeter of a park or residential development. These lines of trees are readily identifiable from aerial photography and their canopy typically forms a structural network that connects a variety of urban land-covers. Despite this high structural connectivity, gaps within this network are evident. Tree lines were selected that were at least 20m wide and composed of trees >4m in height, which we consider ideal commuting features for *P. pipistrellus* (c.f. Verboom & Spoelstra, 1999). Gaps in tree lines were defined as locations where a tree line terminated, but where after a break of at least 20m a second tree line continued along approximately the same direction. In some cases it is likely that such tree lines had originally formed a single boundary feature, which was subsequently bisected by the building of a road. Gaps were illuminated to varying levels (S1) by sodium-vapour street lamps (the dominant source of outdoor artificial lighting within the city (Hale et al., 2013)).

Our aim was to explore the impact of different gap widths and lighting conditions on crossing behaviour. Rather than experimentally manipulating gap characteristics, we identified a selection of gaps within which to undertake surveys, stratified by width and illumination level. To support this stratification process, gaps were each assigned single values for width and illumination as follows: (1) A variety of gaps in tree lines were identified in ArcGIS 9.2 (ESRI, USA) using a raster layer representing tree cover >4m in height derived from remotely sensed 1m resolution colour and near-infrared photography (Bluesky International Limited, UK, 2007) and LiDAR data (The GeoInformation Group, UK, 2006). Gaps where the built land-cover within a 350m radius exceeded 60% were excluded, as activity for *P. pipistrellus* tends to be lower in these areas (Hale et al., 2012). (2) Measurements of surface illumination within each gap were collected in the field following a 2m interval grid of survey points, using a USB2000+RAD spectroradiometer (Ocean Optics, USA). (3) These point measurements were subsequently digitised within the GIS and spline interpolation was used to generate a 1m resolution raster layer representing surface lux within each gap (Fig. 2). (4) Five transect lines crossing each gap were created in the GIS at 5m intervals parallel to the main axis of the tree line (Fig. 2), and the length of each transect line was recorded. (5) Each transect line was then intersected with the lux raster to identify the maximum lux encountered, using Hawth's Analysis Tools (Beyer 2004). The result of this process was the calculation of five width and five lux values for each gap (S1). From these, the median width and median lux value were used to characterise each gap, in order to provide typical values to inform the final stratified selection of survey gaps. (6) 27 survey gaps were then chosen to ensure strong coverage across three width and three illumination categories (S1).

Figure. 2



A gap in an urban tree line caused by a road, as represented in the GIS. Dark green areas represent tree cover >4m high and the variation in road surface lux is indicated by the yellow gradient. Parallel transects (blue lines) were used to provide an indication of potential crossing routes, the distance of these routes and the maximum lux that they encounter (indicated by numbered circles). Actual bat crossing routes (red arrow) were mapped based on surveyor (black circle) observations and confirmed based on camera (blue rectangle) recordings.

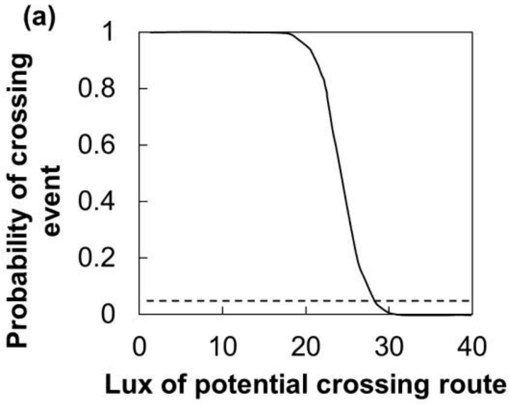
2.2 Gap crossing surveys

In order to record crossing behaviour of *P. pipistrellus*, surveys were undertaken at each gap for a 1.5h period following dusk (c.f. Berthinussen & Altringham, 2012). Surveyors were positioned at either end of the gap and used Batbox Duet detectors (Batbox, UK) to be alerted to approaching bats. As directionality of bat detectors is generally poor, it was necessary for surveyors to identify and record the crossing route of each bat, which was later digitised onto the GIS. This species typically commutes at a height of 2.5-10m (Russ, 1999; Verboom & Spoelstra, 1999; Berthinussen & Altringham, 2012) and individuals were visible when crossing lit gaps. However, when bats crossed in groups or when dark gaps were surveyed, the crossing routes were confirmed using video recordings. Two cameras were used: a Thermovision A20M thermal camera (FLIR systems, United States) and a DCR-HC19E digital video camera (Sony, Japan) in NightShot mode, with additional near-infrared (NIR) lighting provided by a 70 degree angle 850nm IR LED flood lamp (Camsecure, UK). Such lighting is routinely used in bat surveys (Berthinussen & Altringham, 2012) and we found no research to indicate NIR sensitivity for any bat species. The potential for mammals to sense NIR wavelengths has been raised by Newbold and King (2009) and the possibility of NIR lighting impacting bat behaviour should therefore not be excluded, although we emphasise that our research design was systematic across all sites. Bat calls were recorded using a pair of AnaBat SD1 frequency division bat detectors (Titley Scientific, Australia) positioned at either end of the gap, allowing each crossing event to be attributed to a species or species group. Calls were identified in AnalookW (Corben, 2009) using bespoke filters (Hale et al., 2012).

2.3 Models for crossing behaviour

Two analyses were undertaken to explore the response of bats to potential crossing routes that differed in their width and illumination level, using data from the gap crossing surveys. The primary analysis sought to identify barrier thresholds for gap crossing, using logistic regression to estimate the probability of a barrier effect (c.f. Awade et al., 2012). First, we created a single dataset of distance and lux values (referred to as the ‗crossing distance‘ and ‗crossing lux‘ respectively) for crossing events and failures. For crossing events, these values were extracted from the GIS using the digitised crossing routes. For survey gaps where no crossings were recorded, distance and lux data were extracted from the GIS using the gap transect lines. As lux levels could be highly variable within a gap, we extracted the maximum lux value encountered along each crossing route or gap transect. This data was then used to generate a series of binary logistic regression models in R 2.11.1 (R Core Team, 2010) as follows, using the MASS library (Venables & Ripley, 2002). To explore whether the level of illumination required for a barrier effect (the ‗barrier lux‘) varied with crossing distance, subsets of data were selected for modelling using a 20m moving window (see Fig. 3 for examples). The barrier lux was defined as the lux level required for a crossing probability of 5% or less. The barrier lux and mid-range distance from each logistic regression model were then used to model barrier lux as a linear function of crossing distance (henceforth referred to as the ‗barrier lux model‘) (Fig. 4).

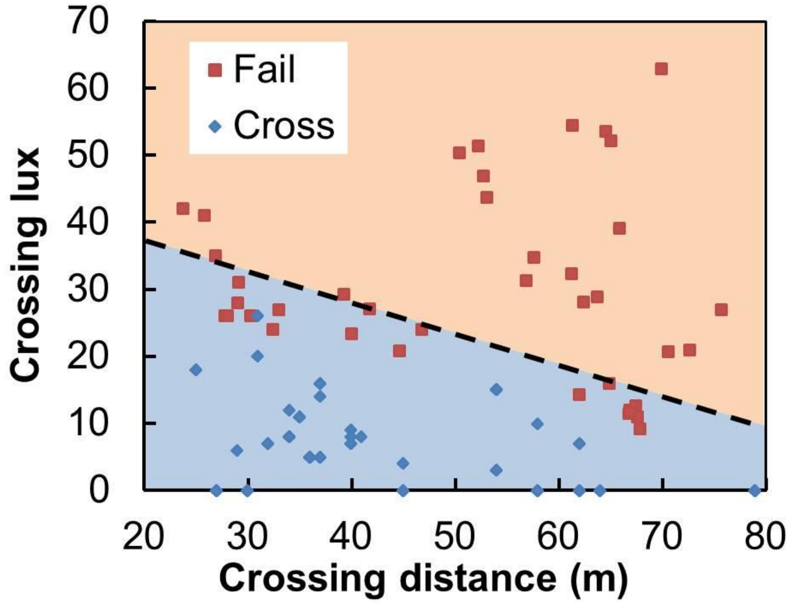
Figure. 3



Examples of binary logistic regression models for the probability of gap crossing by *P. pipistrellus* at different lux levels. Models are given for crossing distances of (a) 20-40m, and (b) 60-80m. The dashed lines indicates where the probability of crossing = 0.05.

The second analysis aimed to explore whether the routes taken by bats crossing survey gaps differed from the typical values for the corresponding gaps, in terms of lighting and distance. In order to highlight potential biases in crossing behaviour, values for the distance of each digitised crossing route were plotted against the median width of the gap being crossed, calculated using distance data from the 5 gap transects. Similarly, the lux of each crossing route was plotted against the median lux value of the gap being crossed.

Figure. 4



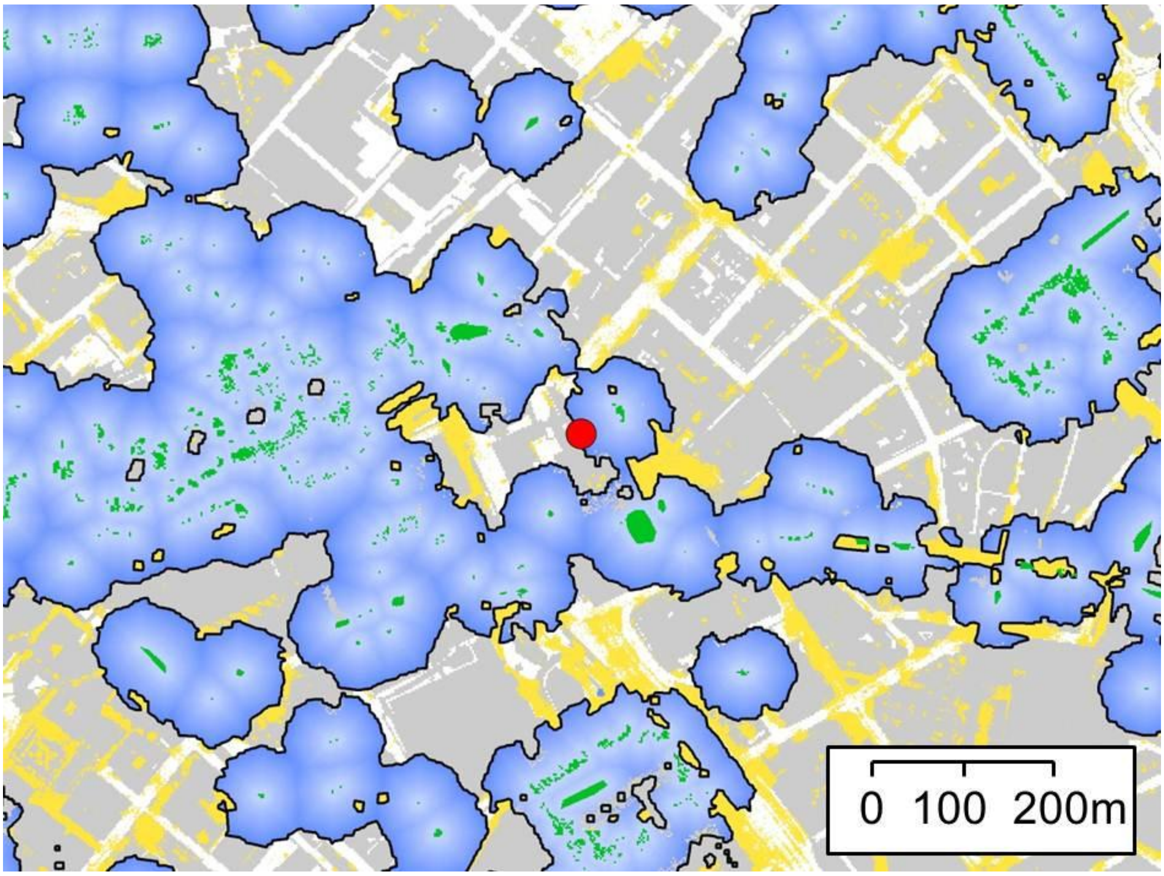
Gap crossing successes and failures for *P. pipistrellus*. For crossing events, the distance and maximum lux for each crossing route are plotted (blue diamonds). For gaps where no crossing events were recorded the distance and maximum lux for gap transects are plotted (red squares). The dashed line represents an estimate of the barrier lux for any given crossing distance, generated using the linear regression equation from the barrier lux model.

2.4 Spatial models for the impact of artificial lighting on landscape resistance

Generating resistance surfaces is an increasingly popular way to provide quantitative estimates of how different environmental parameters such as land-cover type or human population density may impede animal movement (Zeller et al., 2012). Spatial environmental data are typically combined with biological data from surveys to generate cost surfaces that can be interpreted as maps of resistance/barriers to movement. In this case, we created a resistance surface to represent the combined effect of distance from tree cover and illumination by artificial lighting on bat movement. We generated this resistance surface for the city of Birmingham, as it is within the broader West Midlands metropolitan county where our gap surveys were undertaken, and high-resolution lighting and tree cover data are available for the full extent of the city (Hale et al., 2012; Hale et al., 2013). Our aim was to use the barrier lux model to generate a resistance surface by using rasters representing distance to tree cover and incident lux as input values for the model variables, from which we could classify the landscape into either accessible or inaccessible patches of land-cover. A key assumption within this model was that lighting would have no barrier effect on individuals of *P. pipistrellus* commuting along contiguous tree lines and woodland edges (c.f. Stone et al., 2012), but that lighting had the potential to act as a barrier to the crossing of open areas between tree cover.

First, the ArcGIS Cost Distance tool was used to generate a 1m resolution raster layer for the entire city representing distance to the nearest tree cover >4m high. In most cases the output raster values represented linear distance to tree cover. However, non-linear distance calculations were also permitted in order to recognise that euclidian distance measures would be inappropriate for locations where tall buildings would create a barrier to straight line flight at typical commuting height. To achieve this, parts of buildings > 30m in height were selected from the 2008 Ordnance Survey MasterMap (OSMM) land-use dataset and saved as NoData values within a 1m resolution raster layer. All other raster cells were assigned a value of 1 and this layer was then used as an input cost raster as part of the cost distance calculations. Next, the distance value attributed to each pixel was doubled to represent the minimal possible flight distance for a bat leaving and returning to tree cover via that pixel location. This distance layer was then used to calculate the lux level that would be required for a barrier effect at each pixel location, using the Raster Calculator tool to apply the regression equation from the barrier lux model to the distance value of each pixel value. The resulting barrier lux layer was compared to a second layer representing incident lux (2009) for the entire city at 1m resolution, estimated from aerial night photography (Hale et al., 2013). When the lux value for a pixel from the 2009 lighting dataset was equal to or greater than the corresponding pixel value within the barrier lux layer, the pixel was classified as inaccessible to our study species. The resulting resistance surface was converted to a polygon layer representing zones surrounding urban tree cover that would be expected to be accessible to bats, based upon the lighting levels in 2009 (Fig. 5). This process was repeated to generate resistance surfaces for two contrasting urban lighting scenarios. The first scenario was for a city without any lighting (the Dark City) and was intended to serve as a baseline model for the independent effect of the structural connectivity of tree cover on landscape resistance. The second was for a heavily lit scenario (the Bright City). This Bright City scenario used the 2009 lighting layer as a starting point, but the surface lighting values of all roads were increased to a minimum of 20 lux; representing a plausible but extreme scenario for future urban road lighting.

Figure. 5



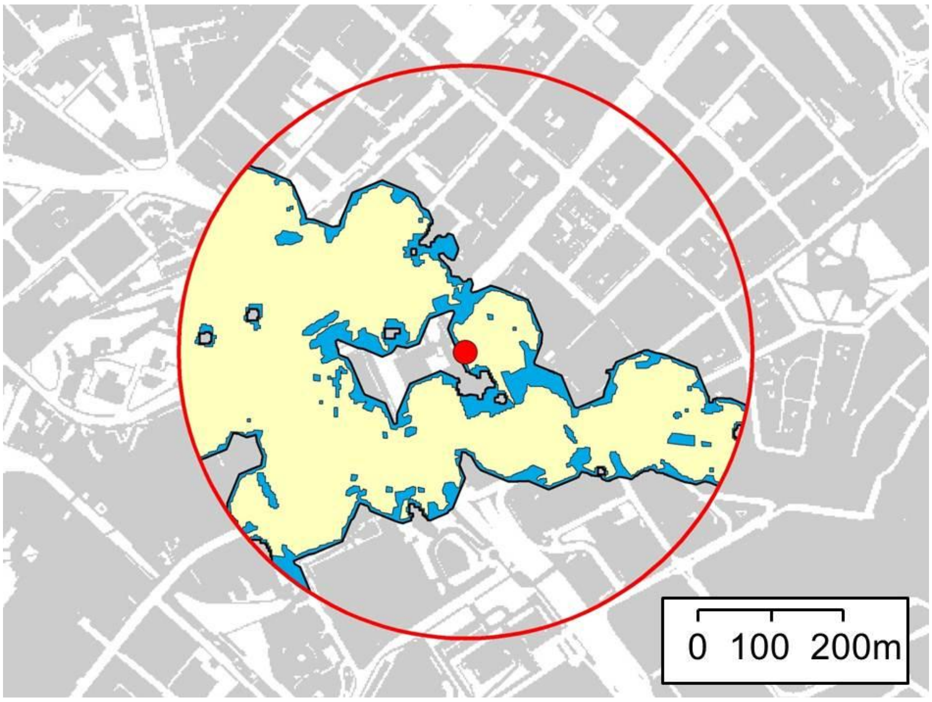
Zones surrounding urban tree cover where the lighting levels are predicted to be insufficient to act as a barrier to movement for *P. pipistrellus*. This was derived from a resistance surface generated within the GIS at 1m resolution, by applying the barrier lux model to a raster representing distance to tree cover (2006/7) and by comparing the output to a map of incident lux (2009). Key: green = trees >4m, blue network = areas surrounding tree cover where lighting has no barrier effect, blue gradient indicates distance from tree cover, yellow = surface illuminance >20lx, grey = buildings, red dot = an urban pond used by bats for foraging. Building outlines derived from OS MasterMap land-cover and land-use parcels reprinted from original mapping with permission from the Ordnance Survey (2008).

2.5 Habitat accessibility along an urban gradient

Urban gradient analyses have been extensively used as a means for exploring the impact of ‗intensification' on species presence or abundance. Such approaches are a practical response to concerns about the increasing density and extent of urban areas, yet as many ecologically relevant variables co-vary along such gradients (Hahs & McDonnell, 2006), it is rarely clear how these variables combine to drive the ecological patterns observed. The aim of this analysis was to use GIS analyses to explore how the landscape resistance resulting from variations in urban tree cover and lighting could impact habitat accessibility along a gradient of built land-cover. Sampling was centred on small ponds (maximum area 2000m^2), as these are potential foraging sites for *P. pipistrellus* and are distributed throughout the city. The underlying assumption of this analysis was that ponds would have greater value as foraging habitats if the surrounding landscape had low resistance to bat movement. All ponds within Birmingham were identified from OSMM land-use polygons using the GIS and each pond centre was buffered by 350m, a key spatial scale for predictive models of *P. pipistrellus* activity identified in an earlier study (Hale et al., 2012). The percentage built land-cover within 350m of each pond was then estimated using OSMM polygon data and each pond was assigned to one of seven ‗density classes' ranging from a low density class of 10-20% built land-cover, to a class of ponds surrounded by between 70 and 80% built land-cover. 35 of these ponds were then selected for use in the gradient analysis, 5 from each density class. A greater number of ponds could not be selected without causing uneven sampling, because few ponds were present in heavily built areas.

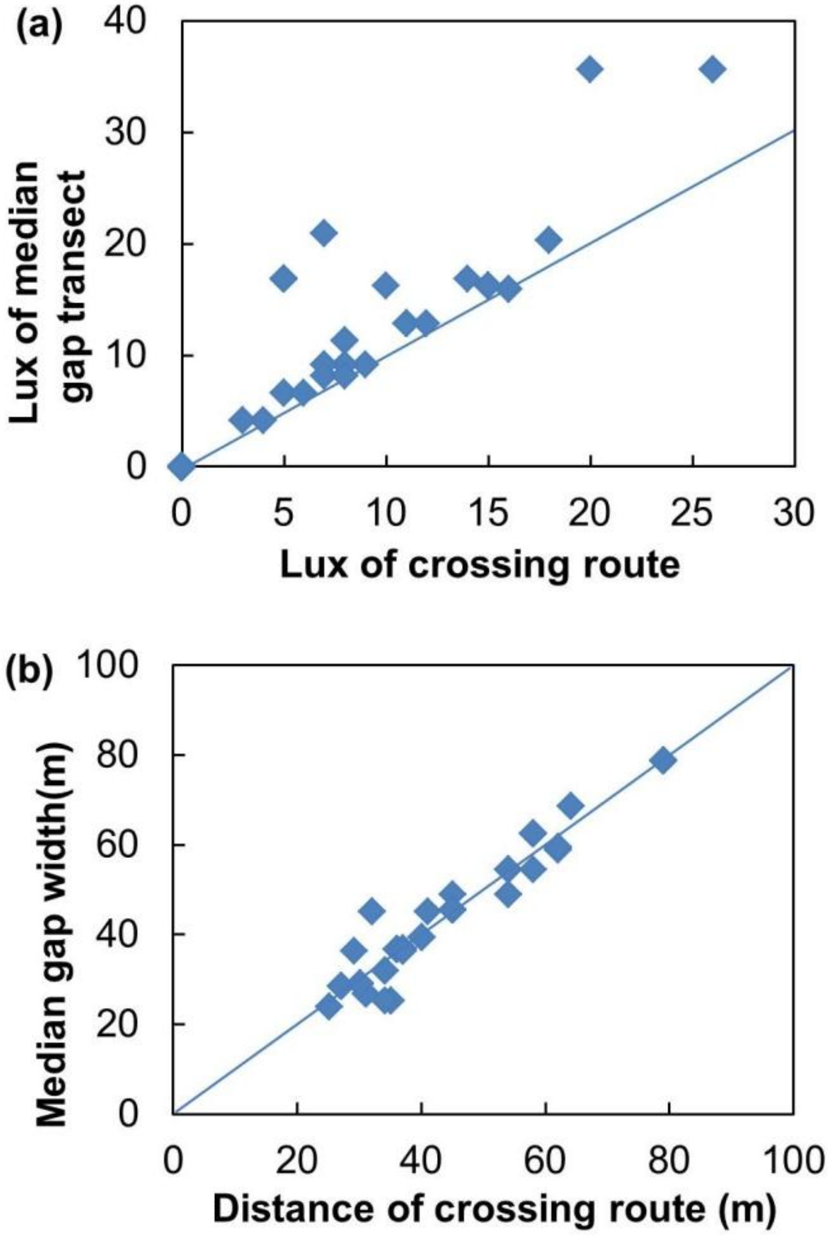
The polygon layer representing patches of land-cover predicted to be accessible under 2009 lighting levels was then clipped by a 350m buffer zone surrounding each pond and those patches that intersected the pond were retained (Fig. 6). The total area of accessible land-cover connected to each pond was then recorded as a percentage of the total surface area within 350m of the pond. This was modelled against the percentage built land-cover within the 350m buffer zone using a generalised additive model (GAM) in R 2.11.1, using the MGCV library (Wood, 2006). This process was then repeated for the accessible land-cover models generated for the Dark City and Bright City scenarios.

Figure. 6



Two spatial models for areas of accessible land-cover within 350m (red circle) of an urban pond (red dot), under a Dark City scenario (blue) and a Bright City scenario (yellow). In this example when no lighting is present 44% of the local landscape is predicted to be accessible from the pond, shrinking to 36% in the brightly lit scenario.

Figure. 7



(a) Maximum lux for bat crossing routes vs. maximum lux of the median gap transect. The line indicates where the crossing route lux and gap lux values are equal. (b) Distance of each crossing route vs. the median gap width (based upon gap transects). The line indicates where the crossing route distance and gap width are equal.

3. Results

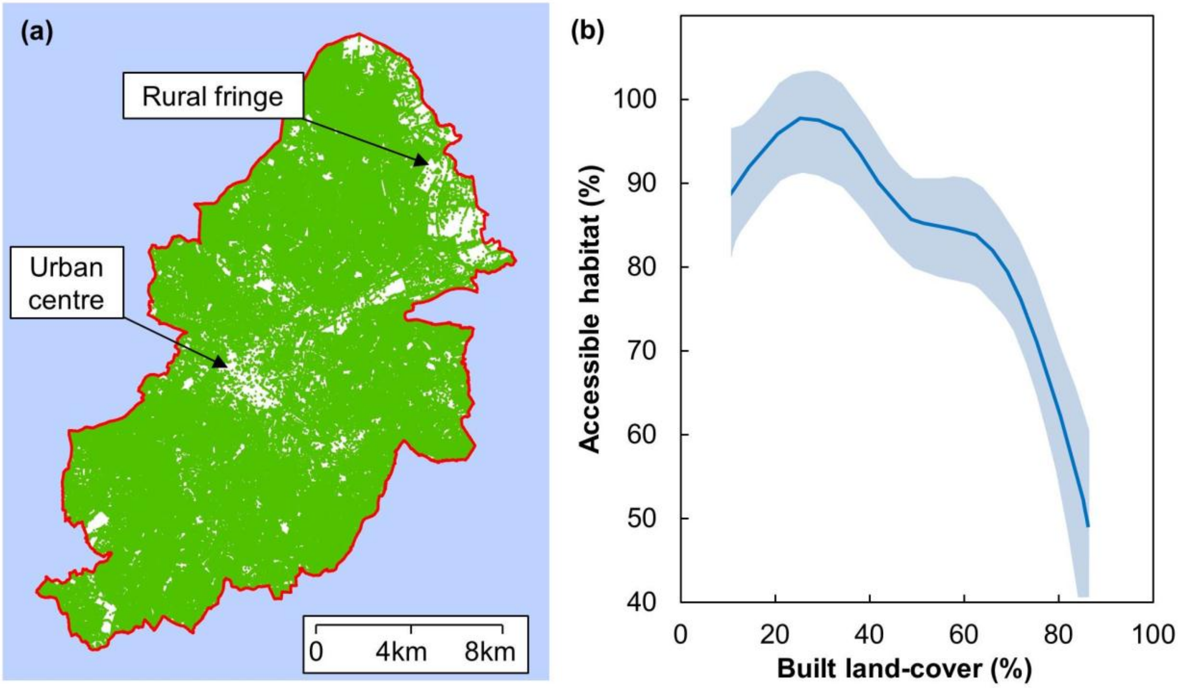
3.1 Crossing behaviour

The majority of the bats that were recorded crossing gaps were *P. pipistrellus* and therefore all results presented here relate to this species. Individuals of *P. pipistrellus* were recorded in the vicinity of all survey gaps, but were only observed crossing 19 of the 27 gaps. The lighting threshold for a barrier effect reduced with increasing crossing distance (Fig. 4), following the linear model: barrier lux = -0.46*crossing distance + 46.2, where the barrier lux is the lux value at which the probability of crossing is 5%. The majority of bats (95.6%) selected crossing routes that were darker than the median gap lux value (Fig. 7a), indicating that bats were choosing to cross in the darker parts of gaps; whereas the length of crossing routes was not consistently larger or smaller than the median gap width (Fig. 7b).

3.2 Landscape connectivity analysis

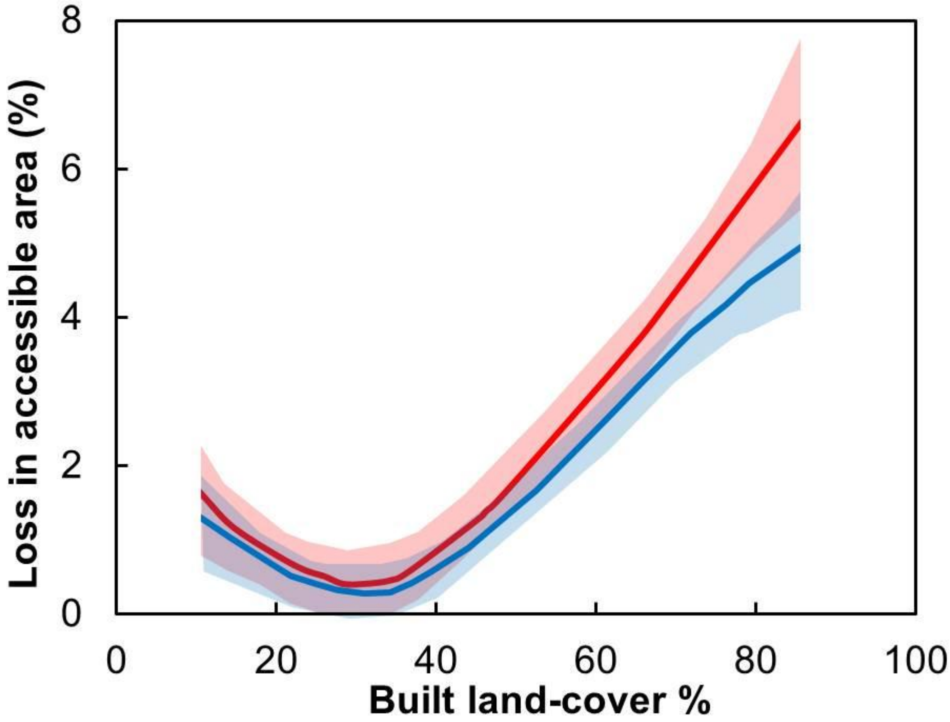
Landscape resistance for *P. pipistrellus* varied within the City of Birmingham (Fig. 8a) along a gradient of built density (Fig. 8b), as a result of the fine grained arrangement of trees and lighting (S2). When modelled using 2009 lighting data, accessible land-cover was highest in areas where built surfaces account for less than 25% of the landscape, but dropped markedly when built land-cover was greater than 65%. Much of this effect is due to the abundance and arrangement of tree cover, although the impact of lighting is clear at higher built densities (Fig. 9). Compared to a Dark City model, lighting levels in 2009 further reduce the percentage of accessible land-cover surrounding ponds by up to 5% in heavily built areas, and by up to 7% under a Bright City scenario (Fig. 9).

Figure.8



(a) A spatial model of areas within the City of Birmingham where accessibility for *P. pipistrellus* is not restricted (indicated by green networks) by artificial lighting levels present in 2009. Accessible land-cover is poor in the urban centre and other highly built up areas, as well as at the rural fringe. (b) Estimates of habitat accessibility along a gradient of built surface cover, based on measurements for 35 typical ―foraging ponds". Habitat accessibility is defined as the percentage of surface area within a 350m radius of each pond that the model predicts to be available to bats under a given lighting scenario, and that also intersects the pond. Shaded areas represent 95% confidence intervals.

Figure. 9



The impact of lighting on the area of accessible land-cover connected to urban ponds under Bright (Red) and 2009 (Blue) city lighting scenarios, compared to the levels found under a Dark City scenario.

4. Discussion

Outdoor artificial lighting is one of many urban characteristics that are changing rapidly across the globe, yet relatively little is known about its unintended consequences for environmental wellbeing. There is a need for research to identify these potential impacts, and to contextualise the results in a way that allows mitigation to be targeted effectively. Our analysis demonstrates that lighting can affect landscape resistance in cities, even for a species of bat (*P. pipistrellus*) that that has been recorded in many urban land-cover types (Gaisler et al., 1998; Hale et al., 2012). The greatest impacts on this species are likely to be in brightly lit areas where structural connectivity of tree cover is already low, characteristics typical of heavily built areas such as urban centres.

4.1 Bats, connectivity and lighting

There is a need to better understand those factors that influence the ability of organisms to move between resource patches and for tools that can predict the impacts of changes at a landscape-scale (Adriaensen et al., 2003). Central to this is the recognition that functional connectivity of habitats is dependent on both landscape structure and individual behaviour (Tischendorf & Fahrig, 2000). To our knowledge, this is the first study to quantify the effect of lighting on gap crossing in bats and to explore how barrier effects may accumulate across a landscape. Distance thresholds for gap crossing have been identified in the field for groups such as birds (Creegan & Osborne, 2005; Awade & Metzger, 2008) and mammals (van der Ree et al., 2004) and then translated into maps of accessible habitat (Awade & Metzger, 2008). However, few attempts have been made to model landscape resistance for bats (but see Frey-Ehrenbold et al., 2013), or to integrate lighting into gap crossing models.

Measures of tree/hedge connectivity have been used to model bat activity in both rural and urban landscapes. A connectivity index for rural trees and hedgerows was developed by Frey-Ehrenbold et al. (2013), and used to identify a positive association between connectivity and activity patterns for three bat guilds. Their results indicate that the distance between patches impacts their likelihood of use. In addition, a connectivity measure used by Hale et al. (2012) found a significant effect of connected urban tree cover on bat activity, based upon the assumption that bats could cross gaps in tree cover of <40m. In both cases the connectivity model was developed using weightings or distance thresholds chosen to broadly reflect what was known of the species movement ecology, although the results of this study suggests that the inclusion of lighting in such connectivity models could be beneficial. Researchers have also experimentally tested the effect of lighting on bat movement (e.g. Stone et al., 2009) and others have modelled the effect of lighting on the movement of nocturnal species by using street lamp locations to create spatially explicit lightscapes (Bennie et al., 2014b); however no studies have considered lighting thresholds for gap crossing. Stone et al. (2009) used experimental lighting of rural hedge lines to disrupt movement for the relatively slow flying lesser horseshoe bat (*Rhinolophus hipposideros*), demonstrating a significant barrier effect. In a later study (Stone et al., 2012) they found no effect of lighting on *P. pipistrellus* despite using similar illumination ranges to our study. The study by Stone et al. (2012) differs to this study in two important ways: firstly in terms of the structural connectivity of the hedges/tree lines (continuous vs. fragmented), and secondly the landscape context (rural vs. urban). It is possible that illuminating a tree line to 50 lux is insufficient to disrupt the commuting behaviour of *P. pipistrellus*, but that the creation of a similarly lit gap may be enough to deter crossing. Moreover, it is possible that a small section of lit hedge in an otherwise dark rural landscape may be of little concern to the fast flying *P. pipistrellus*, whereas the perceived predation risk from crossing a lit gap in an already extensively lit urban area may be high enough to deter crossing.

4.2 Habitat accessibility and urban context

Ecological studies along urbanisation gradients are relatively common and typically indicate a reduction in species richness or abundance at high levels of built density (McKinney, 2008). However, given that many variables such as land-cover and disturbance co-vary (Hahs & McDonnell, 2006; Hale et al., 2013) it is often unclear which underlying mechanisms are responsible for the ecological patterns observed (Threlfall et al., 2011). Here we found that along a gradient of increasing built land-cover, the area of tree canopy cover reduces whilst brightly lit surfaces increase (S2) and that these combine to increase the resistance to movement within heavily built areas.

4.3 Implications for conservation

Relating movement patterns to measures of landscape structure is desirable (Kindlmann & Burel, 2008), particularly as habitat features are often easily mapped. However, it is clear that simple maps of contiguous habitat do not necessarily correspond to functionally connected areas (Tischendorf & Fahrig, 2000) as individuals may move between habitat patches for a wide variety of reasons (Nathan et al., 2008), crossing a potentially hostile matrix in the process. Networks of tree cover along with broader elements of ‗green infrastructure' are commonly recognised in urban planning policy as ‗wildlife corridors', although the evidence base for their efficacy is mixed (Angold et al., 2006; Gilbert-Norton et al., 2010). Whether such structural features actually function to reduce landscape resistance has been a much debated question in landscape ecology (Beier & Noss, 1998). Awareness of the potential impacts of habitat fragmentation (Kerth & Melber, 2009) and lighting (Stone et al., 2009) on bat movement has led to a range of mitigation practices, yet in some cases they appear ineffective (Berthinussen & Altringham, 2012). The ability to commute from roost to feeding areas is crucial to the survival of *P. pipistrellus* and commuting distances >1km are not uncommon (Davidson-Watts & Jones, 2006). It is therefore plausible that restrictions on movement in parts of a city could have fitness impacts at the individual level, as well as limiting the size and extent of urban populations. This highlights the need for a stronger evidence base to support work to protect and improve landscape permeability for urban bats. Whilst bat roosts within the European Union are legally protected under the EU Habitats Directive (1992/43/EEC), the level of protection afforded to commuting routes is less clear (Garland & Markham, 2007). Analyses such as those presented here could support the development of related policy, by clarifying the likely location of commuting routes and the thresholds for their disturbance. These results suggest that networks of urban trees support the movement of *P. pipistrellus*, even when they contain gaps of up to 80m. However, it is clear that access to feeding habitats may be undermined by lighting within the surrounding landscape, even if the structural elements of the tree network remain unchanged. Although the impacts of lighting demonstrated here are subtle, the approach used to characterise barriers was conservative and lower thresholds for identifying impacts on movement may be more appropriate for conservation purposes. This is supported by the finding that individuals consistently crossed in the darker parts of the gap, even when those gaps were poorly lit, suggesting that all crossing events may be associated with costs (e.g. greater predation risks) that commuting individuals attempt to minimise. The strategic dimming of lights in the vicinity of gaps, combined with the narrowing of gaps through tree planting might therefore be reasonable conservation measures for this species in urban areas. Such an approach may also have benefits for other bat species that are even less tolerant of lighting such as Myotis spp (Stone et al., 2012). However, the impacts on *P. pipistrellus* of a broader scale reduction in urban lighting may be more complex, given that this species is able to exploit concentrations of its insect prey surrounding individual lamps (Blake et al., 1994). Species of bats may respond to gaps (Kerth & Melber, 2009) and also lighting (Stone et al., 2012) very differently; therefore whilst this approach could be used to model the impact of lighting on landscape resistance for other species, further research is needed to identify appropriate threshold values. Similarly, it is unknown whether the barrier lux model developed here is suitable for all individuals of *P. pipistrellus*, or for different times of the night. Movement is a key component of functional connectivity and it is important to recognise that a range of factors may influence movement events. Whilst patterns of tree cover and lighting appear to be important, further work is needed to identify how resistance may vary with different land-covers or the impact of habitat quality and social structure on movement decisions.

The use of contrasting lighting scenarios to explore potential impacts on landscape resistance could be incorporated into practical conservation measures at a variety of scales. Scenarios are commonly used in sustainability research and practice to test the resilience of infrastructure, communities, resources and natural systems to a variety of stressors (Nakicenovic & Swart, 2000; Carpenter et al., 2006; Hunt et al., 2012). The ecological impacts of different scenarios for land-cover have been explored by other authors (Adriaensen et al., 2003; Kong et al., 2010; Sushinsky et al., 2013) but we believe this is the first study that has explored the impacts of different urban lighting scenarios at the city scale. This approach may be useful for exploring the impact of specific proposals for changes to urban lighting (Gaston et al., 2012) or tree cover (Pincetl, 2010). However, the limited knowledge of how these characteristics can change over time (Gaston et al., 2012; Gillespie et al., 2012) means that a broader sensitivity analysis may be required to identify network connections that are particularly vulnerable or resilient.

Given the rapid changes underway in cities, urban biodiversity is often faced with multiple ecological disruptors that may be changing simultaneously; disentangling the impacts of these disruptors presents a major challenge. For conservation to shift from a largely reactive to a more proactive approach, it must move on from detecting broad patterns in urban biodiversity to a more mechanistic understanding of the processes that drive them (McDonnell & Hahs, 2013). The results of this study indicate that the structural connectivity of tree cover and the levels of lighting within the intervening matrix combine to affect gap crossing behaviour for a common urban bat. In the case study city, this model predicts that as a result, habitat accessibility may reduce with increasing built density, although the potential exists for de-coupling this relationship in the future. This has implications for conserving urban biodiversity in cities that are becoming brighter and more densely developed.

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Supplementary Information

The following SI Figures can be found in the Appendix at the end of this thesis.

Figure S1. Information on survey gap locations and characteristics.

Figure S2. Changes in tree cover and lighting along a built density gradient.

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PAPER V - DELIVERING A MULTI-FUNCTIONAL AND RESILIENT URBAN FOREST

James Hale[1,11], Thomas Pugh[2], Jon Sadler[1,11], Christopher Boyko[3], Julie Brown[4], Silvio Caputo[5], Maria Caserio[6], Richard Coles[6], Raziyeh Farmani[7], Chantal Hales[8], Russell Horsey[9], Dexter Hunt[10], Joanne Leach[10], Christopher Rogers[10], Rob MacKenzie[11]

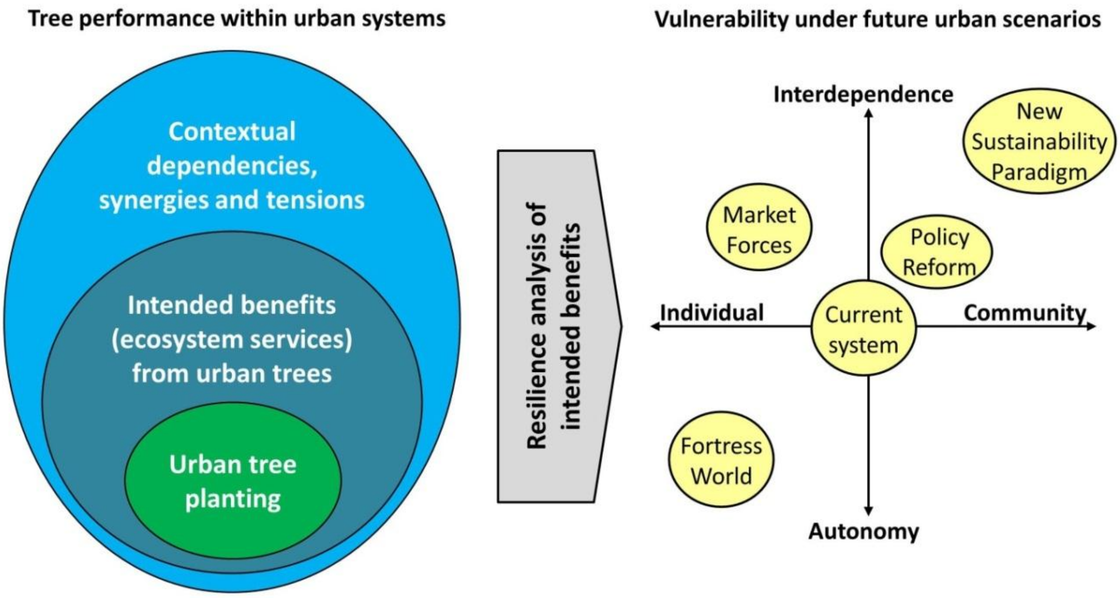
1. School of Geography Earth and Environmental Sciences, The University of Birmingham, Edgbaston, Birmingham, B15 2TT, United Kingdom
2. Institute of Meteorology and Climate Research/Atmospheric Environmental Research, Karlsruhe Institute of Technology, Garmisch-Partenkirchen, Germany
3. Imagination Lancaster, Lancaster Institute for the Contemporary Arts, Lancaster University, Lancaster, LA1 4YW, United Kingdom
4. Faculty of the Creative Industries, Southampton Solent University, East Park Terrace, Southampton SO14 0YN, United Kingdom
5. Department of Civil Engineering, Architecture and Building, Coventry University, Coventry, CV1 5FB, United Kingdom
6. Birmingham Institute of Art & Design, Birmingham City University Birmingham, B4 7BD, United Kingdom
7. College of Engineering, Mathematics and Physical Sciences, University of Exeter, Exeter, EX4 4QF, United Kingdom
8. London School of Commerce, London, SE1 1NX, United Kingdom
9. Institute of Chartered Foresters, Brunswick Court, Brunswick Square, Bristol, BS2 8PE, United Kingdom

10. Department of Civil Engineering, The University of Birmingham, Edgbaston, Birmingham, B15 2TT, United Kingdom
11. Birmingham Institute of Forest Research, University of Birmingham, Edgbaston, B15 2TT, United Kingdom

Abstract

Tree planting is widely advocated and applied in urban areas, with large-scale projects underway in cities globally. Numerous potential benefits are used to justify these planting campaigns. However, reports of poor tree survival raise questions about the ability of such projects to deliver on their promises over the long-term. Each potential benefit requires different supporting conditions - relating not only to the type and placement of the tree, but also to the broader urban system within which it is embedded. This set of supporting conditions may not always be mutually compatible and may not persist for the lifetime of the tree. Here, we demonstrate a systems-based approach that makes these dependencies, synergies and tensions more explicit, allowing them to be used to test the decadal-scale resilience of urban street trees. Our analysis highlights social, environmental, and economic assumptions that are implicit within planting projects; notably that high levels of maintenance and public support for urban street trees will persist throughout their natural lifespan, and that the surrounding built form will remain largely unchanged. Whilst the vulnerability of each benefit may be highly context specific, we identify approaches that address some typical weaknesses, making a functional, resilient, urban forest more attainable.

Graphical abstract.



1. Introduction

Greening our cities might be thought of as the archetypal urban sustainability solution. The potential of trees, in particular, to deliver a range of social, environmental and economic benefits is recognised by both researchers and practitioners, yet it is clear that the dynamic nature of urban areas may threaten the survival of trees to maturity and/or undermine their delivery of key benefits to society. Here we demonstrate an approach that: (i) integrates different disciplinary perspectives on the benefits and drawbacks of urban street tree planting, (ii) identifies system conditions upon which these depend, and (iii) tests the vulnerability of these conditions using contrasting scenarios for urban futures.

1.1. The benefits of urban tree cover

In recent years a multitude of large-scale urban tree planting campaigns have been initiated in cities around the world (e.g. the New York City Million Trees program, the UK Big Tree Plant, and Global ReLeaf), and support for urban greening can be found at both local and national levels of government (Stewart et al., 2004, Kang and Cervero, 2009, James et al., 2009, Conway and Urbani, 2007, Petts et al., 2008, Escobedo et al., 2008, Young, 2011). Whilst the financial (McPherson et al., 1997) and natural resource (Pincetl et al., 2013) costs may be considerable, such planting programmes typically claim that a wide range of sustainability benefits will be delivered; including, but not limited to: building energy savings, improved air quality, carbon capture, increased biodiversity, improved water quality, and greater human health and wellbeing (McPherson et al., 2008, Young, 2011, Pincetl et al., 2013). Discipline-specific studies provide evidence to support these claims (Dwyer et al., 1991, Akbari, 1992, Donovan et al., 2005, Currie and Bass, 2008, Escobedo et al., 2010, Savard et al., 2000, Matteo et al., 2006, Price, 2003) and the broad identification, quantification and valuation of potential ecosystem services supplied by the urban forest has received substantial attention (McPherson et al., 1997, Xiao and McPherson, 2002, Nowak et al., 2008, Sander et al., 2010, Escobedo et al., 2011, Roy et al., 2012, 2008, Mullaney et al., 2015). There are also drawbacks associated with increasing urban tree cover that need to be considered, typically referred to as disbenefits or disservices (Roy et al., 2012, Escobedo et al., 2011). These include: health and safety risks (and associated fears), public nuisance (e.g. fallen leaves sticking to parked cars), financial costs from maintenance and infrastructure damage, and environmental impacts relating to waste, pollutants and the introduction of pests (Roy et al., 2012).

1.2. Threats to urban tree cover

Tree cover within some cities has undergone periods of expansion and contraction in recent decades (Myeong et al., 2006, Gillespie et al., 2012, Díaz-Porras et al., 2014, Merry et al., 2014), and the level of tree cover in several US cities has been found to be in decline (Nowak and Greenfield, 2012). Estimates of annual tree mortality rates are highly variable (Roman and Scatena, 2011), with reported losses of 3% to >50% for newly planted street trees, depending on local land-uses and social influences (Nowak et al., 2004, Lu et al., 2010). This raises the question of whether large-scale urban tree planting can succeed in delivering benefits over the long-term, given the impermanence of past urban tree cover. A diverse range of factors influence urban tree survival, ranging from vandalism or removal of the tree itself, to restricted access to key resources such as soil moisture (TDAG, 2014). Threats are also emerging or intensifying as a result of globalisation, urbanization, and population growth (Tubby and Webber, 2010, TDAG, 2014). A recent review of the success of large-scale urban tree planting initiatives in the US points to the problem of uncertainties regarding long-term funding and political support (Young, 2011). Future risks from factors such as pests, diseases and climate change have been addressed at many levels by researchers (Tubby and Webber, 2010), urban forestry and arboriculture professionals (TDAG, 2014), and regulatory organisations. For example, the European Parliament is currently considering a revision of its Plant Health Regime to address concerns about emerging risks related to pests, diseases and the spread of non-native species. In addition, databases exist to help identify pests and diseases that are considered a risk to tree health (DEFRA, 2015) and to support more climate and disease resistant choices for urban tree planting (Forest Research, 2015). However, threats to trees associated with changes to the built form, urban governance and social values appear less well addressed.

1.3. Contextual and temporal sensitivity of ecosystem services supplied by trees

The nature and magnitude of the sustainability benefits delivered by urban trees can be strongly influenced by their urban context, in its broadest sense (e.g. their built, cultural, ecological or economic context). It should therefore be recognised that large-scale urban tree planting projects may include a wide variety of planting locations and tree types, and involve a pool of stakeholders with different motivations, expectations and resources (Pincetl, 2010). Recognising this context is important, as key biophysical processes can be influenced by local land-covers, land-uses and social practices (Supplementary Information S1). For example, transpiration and shade from trees can benefit people and infrastructure via summertime cooling (Akbari et al., 2001, Armson et al., 2012), but may be disrupted in situations where built infrastructure damages tree roots or where sealed surfaces reduce soil moisture levels (Mullaney et al., 2015). The presence of receptor groups (beneficiaries) is also an important consideration. For example, trees planted in residential neighbourhoods may deliver benefits through the cooling of houses in summer, which they would be unable to deliver if planted within other land-use types (McPherson et al., 2011). In addition, the degree to which urban trees effect net urban carbon emissions depends not only on their size, health and species, but also on the surrounding land-use, on the energy requirements of adjacent buildings and on how the resulting green ‘waste’ is managed (McPherson and Kendall, 2014).

Given that the benefits delivered by urban trees can vary depending on their context, what happens when this context changes? Are some benefits particularly sensitive to future social, environmental or economic changes? Redevelopment, densification, population increases and demographic shifts are common characteristics of many cities (Dallimer et al., 2011), potentially impacting the production and consumption of urban ecosystem services. In addition, the ways that citizens value trees may change over time (Ordóñez and Duinker, 2010). Even if the built and social context of an urban tree were to remain stable and supportive over the short term, some benefits may still take many years to accumulate (McPherson et al., 2011), as they often scale with the size or maturity of the tree (McPherson et al., 1997). A key challenge is therefore to ensure that the potential benefits of urban tree planting are realized over the following decades and centuries, in the face of a complex, uncertain and changing urban context (Grimm et al., 2008).

1.4. Trees, urban systems and resilience thinking

Studies that consider threats to the longevity and performance of urban trees often focus on technical questions and solutions related to the tree itself, such as identifying planting techniques that will improve the chances of long-term survival and growth (Grabosky and Bassuk, 1995). This reflects a broader pattern within sustainable urban forestry, to focus on technical and numerical standards related to the trees themselves (Ordóñez and Duinker, 2010). However, a much broader range of social, environmental and economic factors are clearly relevant to the persistence and functioning of urban trees (Young, 2011, Pincetl et al., 2013), as evidenced by the variability in tree cover and survival between land-use types (Nowak et al., 2004), built densities (Díaz-Porras et al., 2014) and land ownership (Gillespie et al., 2012). There is therefore a need for approaches that can explore the current and future performance of the urban forest in a way which acknowledges its diverse range of values (Ordóñez and Duinker, 2010), and also the dynamic nature of the landscape within which these are embedded.

Cities are complex, metastable systems with highly-coupled flows of mass, energy, people and capital (Pickett et al., 2001, Alberti and Marzluff, 2004). Analyses of the risks to the ecosystem services supplied by urban trees must therefore recognise that trees are embedded within this broader ‗system of systems', and may benefit from identifying key system components, dependencies, processes and outputs. For any given benefit to be sustained, a set of system conditions needs to persist, which extend beyond the simple presence of an urban tree (McPherson et al., 1997, Pincetl et al., 2013, Escobedo et al., 2011, Nowak and Dwyer, 2007, Conway and Bang, 2014). A culture of planting 'the right tree in the right place' recognises the importance of context and is clearly embedded in the psyche of many arboriculturists and foresters (TDAG, 2014, James et al., 2012). However, systematic recording and analysis of these contextual dependencies and their vulnerability has thus far been absent.

Addressing sustainability challenges within urban areas typically requires the integration of a variety of perspectives within analyses and decision making, and calls have been made for more interdisciplinary research and collaboration in relation to urban ecosystems (James et al., 2009, Ahern, 2013). However, such interdisciplinary collaboration in both research and practice can be extremely challenging (Boyko et al., 2014, Petts et al., 2008, Pincetl, 2010). ‗Resilience' is a concept that can be used to stimulate interdisciplinary research, to support understanding, management and governance of complex linked systems of people and nature, and to guide development pathways in changeable and uncertain environments (Folke, 2006). Resilience has multiple definitions, but here it is used to mean continuity in the desirable aspects of system performance, despite disturbance or re-structuring of the system itself. When applied to sustainability in cities it helps to emphasise the inherent instability of urban spaces and their uses, and that the performance of an idealized sustainable urban form may depend on its capacity to tolerate, adapt, or even to provoke change (Ahern, 2013). An urban tree could therefore be conceptualised as being part of, and intimately linked to, a broader socio-economic and biophysical system, which if disturbed sufficiently may prevent the tree from delivering key benefits to society. Using the concept of resilience to frame a discussion about risks to urban tree performance also helps to highlight the difference between the intervention (i.e. the planting of a tree), its intended benefits and the conditions upon which these benefits depend. Distinguishing the intervention from its intended benefits makes explicit that it is the benefits that the tree delivers that need to be resilient, even if the tree itself and the urban system within which it is embedded undergo changes in the future.

This study aims to illustrate an approach that can be used to integrate different perspectives on the risks to the long-term performance of urban tree cover, and to provide insight into the resilience of potential benefits associated with street tree planting in the UK. We apply a recently developed method for analysing urban ‗sustainability solutions' (Lombardi et al., 2012, Rogers et al., 2012) that: (i) explicitly pairs a sustainability solution (in this case urban street tree planting) with its intended benefits; (ii) identifies system conditions necessary for the delivery of each benefit and (iii) uses future urban scenarios to systematically test the vulnerability of these conditions.

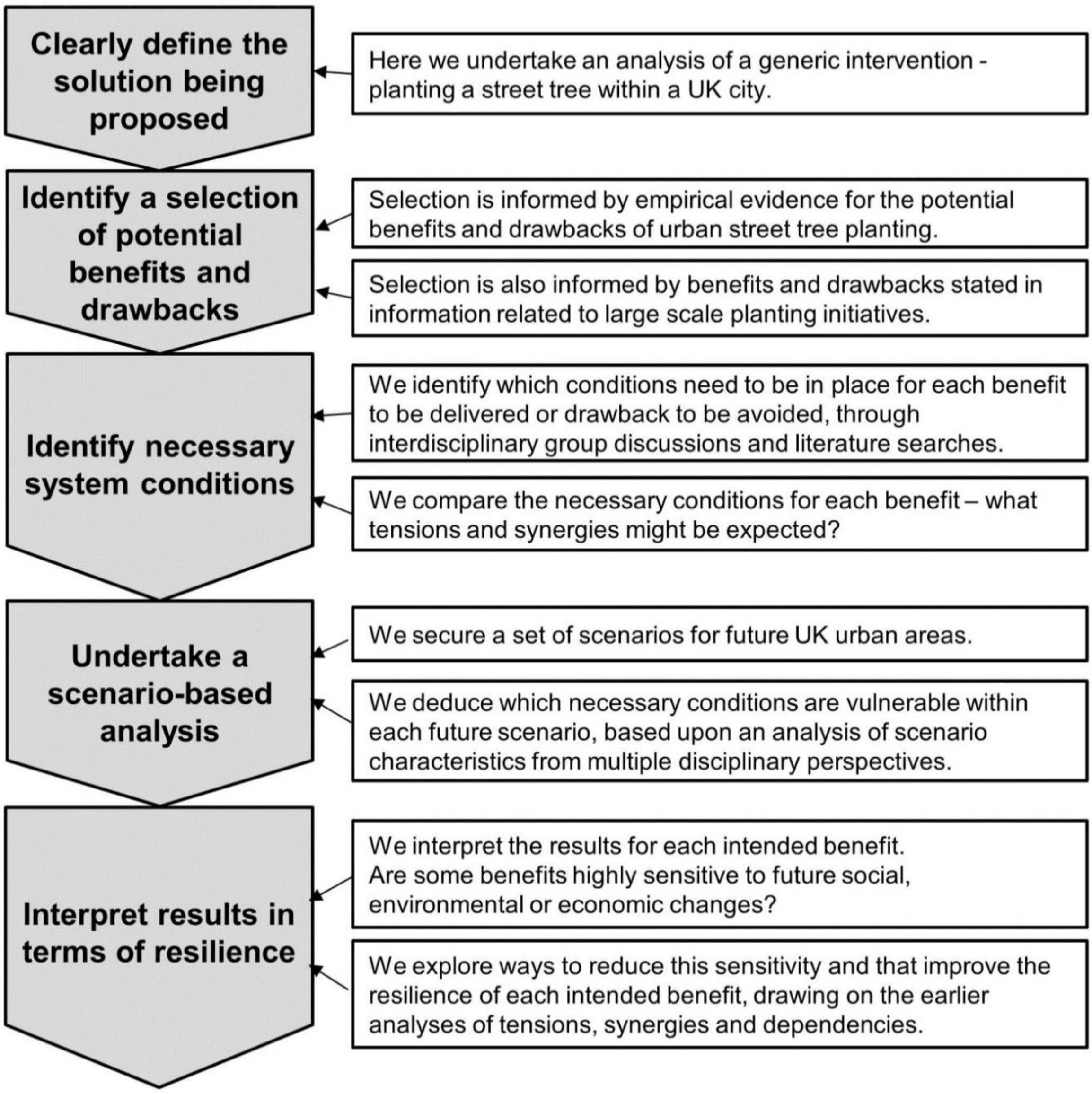
We conclude that the potential benefits of urban street tree planting are often dependant on the presence of system conditions related to the level of tree maintenance, public values, local government policies, and the density and configuration of the surrounding built form. Key conditions may not persist within future urban scenarios where market forces are more dominant, where individualist attitudes prevail or where poverty and inequality are high. We suggest that resilience might be increased by broadening planting locations to include private green spaces immediately adjacent to streets and improving the co-management of street trees by individuals, NGOs and municipal departments. This could be supported by the introduction of market-based systems to incentivise the participation of a broad range of stakeholders in the long-term protection and management of urban street trees. In addition, planting techniques that reduce the need for supplementary watering, reduce maintenance requirements, isolate roots from potentially polluted urban soils and that facilitate transplantation, have the potential to improve the resilience of urban street tree benefits.

2. Materials and Methods

2.1. Diverse perspectives on threats to the benefits of urban street trees

In order to explore how urban street tree performance might become vulnerable over time we followed steps 1-4 of the Designing Resilient Cities Method (Lombardi et al., 2012, Rogers et al., 2012, Pugh et al., 2012), as outlined at www.designingresilientcities.co.uk. At the heart of this method is the recognition that cities are complex systems and that the success of urban interventions may depend on multiple factors. It is therefore desirable to seek input from a broad range of perspectives at each methodological step (Fig. 1). In the present case, we sought UK practitioner input from experts in the fields of architecture (Royal Institute of British Architects), town planning (Lancaster City Council) and the built environment (Building Research Establishment), via workshops led by members of the Urban Futures project (Rogers et al., 2012). Participants were invited to question the resilience of specific development proposals (Hale and Sadler, 2012) or previously implemented solutions (Lombardi et al., 2012), and to identify conditions upon which their performance depended. Participant numbers ranged from approximately 20-40 and were selected by making contact with professional groups/institutions and local municipalities. Our final workshop included the entire Urban Futures academic team and covered the following disciplines: Forestry, Air Quality, Design, Architecture, Civil Engineering, Spatial Planning, Environmental Psychology, Human Geography, Ecology, Utility Services and Economic Development. At this workshop, participants drafted a formal list of the intended benefits of urban street tree planting and associated necessary conditions, drawing upon the outcomes of earlier workshops, as well as their own knowledge from related research and practice. This draft list was then circulated for comments to a wider pool of academic and practitioner project partners.

Figure 1.



An overview of key steps within the methods.

2.2. Urban street trees and their intended benefits

The first step within this methodology is to clearly define the ‗sustainability solution' that is being tested and to state explicitly its intended benefits. Tree planting within urban areas can be highly varied, from commercial forestry within a large temperate park to amenity planting within the business district of a tropical city. We therefore narrowed the scope of our analysis by analysing a generic solution of planting a single street tree within a UK urban area. We define a street tree as any tree growing immediately adjacent to a road. In the UK such trees are often planted in pits dug directly into the paved pedestrian walkway that runs parallel to the road. Previous authors have identified a need for increased planting within UK cities (Britt and Johnston, 2008) and urban greening has received considerable support at the UK government level over recent decades (Tubby and Webber, 2010). In addition, urban street trees are associated with a large set of benefits and challenges (Dandy, 2010, Mullaney et al., 2015) that would be interesting to explore from a systems perspective. This solution was considered broad enough to capture many of the likely threats to urban tree performance, whilst providing sufficient context to make the results useful to those addressing concerns about the legacy of today's urban planting initiatives.

Urban street trees are multi-functional (Mullaney et al., 2015), and are therefore introduced or retained in cities for a variety of reasons, by a variety of actors (TDAG, 2014). This multi-functionality is both desirable and unavoidable, but it may also create confusion about which ecosystem services a particular street tree is being managed to deliver, and which services should be prioritised. We developed a list of potential benefits and drawbacks associated with urban street tree planting, based upon group discussions and workshops, claims within publicity material for large-scale urban tree planting initiatives and evidence from the academic literature (S1), and then screened these to ensure their relevance to UK urban areas.

2.3. Necessary system conditions

Next, we identified conditions that would need to be in place in order for each of these benefits to be delivered, and for key drawbacks to be avoided. These ‗necessary conditions' can be as simple as the continued presence of a tree, or as specific as a particular type of on-going maintenance. For example, street trees planted adjacent to a building can reduce heating requirements during cold weather, on the condition that they are an evergreen species and that they are positioned so as not to inhibit possible solar gain. For each intended benefit, the necessary conditions were initially identified through group discussions and workshops, followed up by literature searches. This process of identifying necessary conditions is subjective and partial, given the many dimensions of the urban system; furthermore, outcomes will be influenced by the professions and disciplines contributing to the process. However, it has the advantage that assumptions about dependencies are made explicit and recorded. Revision in light of new data is straightforward and indeed recommended. Care was taken to avoid duplication and overlap as much as possible (i.e. listing a condition that implicitly includes another listed condition), although this is difficult to eliminate completely when considering such a complex system. Whilst it would not compromise the efficacy of the methodology, avoiding duplication is important, both for simplifying the analysis and for clarifying thinking about which characteristics of the urban system are most relevant. The list of benefits, drawbacks and associated necessary conditions was then arranged as a matrix, similar to the score-matrix method used to support design decision-making in engineering. This benefit-condition matrix was used to identify particular conditions that were necessary for certain benefits to be delivered, conditions that might be required only in particular contexts, and those which had the potential to compromise the delivery of other benefits. This process facilitates the identification of synergies in, and tensions between, delivering multiple benefits from urban street tree planting. The literature evidence base to support this analysis is given Supplementary Information S2.

2.4. Scenario-based resilience analysis

We then undertook a scenario-based resilience analysis to identify those necessary conditions that might not be supported in the future, and to make the reasons for their vulnerability more apparent. Four plausible and internally consistent scenarios for UK urban areas in 2050 were considered, derived from a broader scenario set developed by the Global Scenarios Group (Gallopin et al., 1997). The broad characteristics of these four global scenarios were retained (Fig. 2). However, as part of the Urban Futures project (Lombardi et al., 2012) their characteristics were adapted and expanded to make them more relevant to UK urban areas, covering themes such urban form, natural environment, technology, policy, governance, social values and economy. These urban UK scenarios had been created to allow for the pressure-testing of sustainability solutions against an uncertain future (Boyko et al., 2012, Rogers et al., 2012). Such scenarios are distinct from predictions, extrapolations or any other formal forecasting method (Carpenter et al., 2006, Hunt et al., 2012) and have the advantage that they allow for the inclusion of shocks, phase changes and tipping points that can occur within complex socio-ecological systems (c.f. Carpenter et al., 2006, Fischer-Kowalski and Haberl, 2007, Renaud et al., 2010). Whilst climate change is considered here, its nature is identical within each scenario. However, its impacts on urban trees may vary between the urban scenarios depending on their capacity to respond to this threat. The use of multiple urban scenarios recognises that the future cannot be predicted with any degree of certainty, whilst still providing a framework for exploring whether solutions put in place today could still function within a future that we may not necessarily expect or desire. The combinations of drivers that underpin these scenarios are intended to differ from those typically found within UK cities and in some cases they result in radically different visions of the future. These scenarios can be mapped to fall along a gradient of social values that range from an individual to a community focus, or a gradient that ranges from an open and globalised economy to one that is much more localised (Hunt et al., 2012). We contend that together these scenarios define the boundaries of a likely plausibility space for UK urban areas in 2050. The future scenarios we used are described briefly below, with further details provided in Boyko et al. (2012). The scenario characteristics we consider particularly relevant to urban trees are given in Supplementary Information S3.

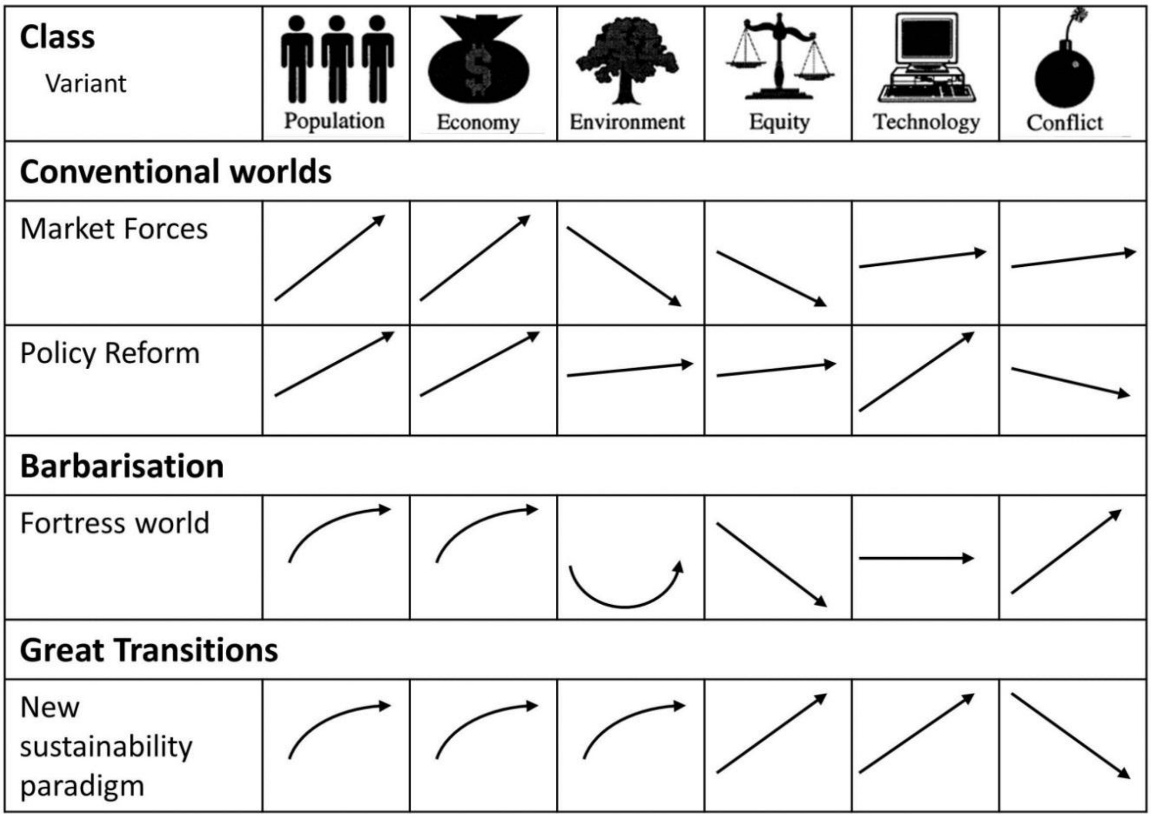
(a) Policy Reform (PR). Government action attempts to reduce poverty and social conflict within the confines of a globalized free market. Individual behaviours are slow to move from materialistic self-interest although it is widely accepted that markets require strong regulation to avert economic crisis, social conflict and environmental degradation. Tensions continue to grow between continuity of the dominant social values and the desire for greater equity to address key sustainability goals.

(b) Market Forces (MF). There is strong belief in the ‗hidden hand‘ (i.e. self-correcting logic) of the free market as key to resolving social, economic and environmental problems. Individualism and materialism are core human values. This scenario assumes that the global system in the twenty-first century evolves without major surprise. Incremental market adjustments have (so far) been able to cope with major social, economic and environmental problems as they have arisen.

(c) Fortress World (FW). As a result of the (partial) breakdown in world order, powerful and self-interested actors protect their resources whilst an impoverished majority are (literally or effectively) disenfranchised and live in ghettos. In this divided world, the elite live in an interconnected network of enclaves and the impoverished majority scratch a living outside. Armed forces impose order, protect those parts of the environment valued by the elite, and prevent complete collapse of society.

(d) New Sustainability Paradigm (NSP). An ethos of sustainability (of ‗one-planet living‘), has taken root throughout society, bringing with it a fundamental questioning of materialism. New socio–economic patterns follow from these fundamental changes in values. In order to maintain global communication and economies of scale, cities are transformed rather than abandoned or replaced.

Figure 2.



Broad characteristics of the future scenarios used within this analysis. Edited and reproduced with permission of Gallopin et al. (1997).

For each necessary condition, we searched the database of scenario characteristics for those that were deemed to be most relevant. The characteristics of each scenario that were considered to either support or undermine each necessary condition are given in S3. Using these characteristics, we deduced whether each necessary condition was likely to be ‗Vulnerable‘, ‗Partially Vulnerable‘ or ‗Supported‘ within a particular scenario. The classification of ‗Partially Vulnerable‘ was used where a condition was considered likely to be supported in some urban contexts but not in others, within the same scenario. By using multiple scenarios with a wide range of characteristics, we were able to subject the necessary conditions to a much more rigorous test of vulnerability than would be the case if only current conditions, or those conditions pertaining to predictions for a single future scenario, were considered.

Finally, the results of this scenario analysis were interpreted in terms of resilience. Those benefits whose necessary conditions were found to be vulnerable under a range of future scenarios were identified as potentially lacking resilience. Options for addressing these vulnerabilities are explored in the discussion.

3. Results

3.1. Necessary conditions

Overall, the most common conditions that were identified as being necessary for delivering particular benefits were those that related to the presence and health of the street tree (Table 1), such as access to sufficient light and water. Other frequently identified conditions were that that the tree is large or mature, that high levels of canopy cover exist in the surrounding urban area, that the tree is maintained for amenity, that people are present nearby, and that the tree is visually accessible to the public. However, several conditions that are necessary for delivering one intended benefit are considered likely to undermine the delivery of other benefits. These conditions include those that relate to the presence of a large or mature tree, the presence of large-scale tree cover, the tree being maintained for wildlife, the surrounding area being built to high-density and the tree being physically accessible to the public. As a result, it is clear that delivering any of the benefits considered here has the potential to undermine other benefits, or to result in drawbacks.

3.2. Scenario analysis

When subjected to the scenario based resilience analysis, all necessary conditions were considered to be vulnerable to some degree and most were found to be at least partially vulnerable within three of the four future scenarios (Table 2). These vulnerabilities are most evident within the Market Forces (MF) and Fortress World (FW) scenarios, where economic and security interests (respectively) are prioritised over environmental concerns. Several conditions relating to the species of street tree appear to be particularly vulnerable. Even in the Policy Reform (PR) and New Sustainability Paradigm (NSP) scenarios where careful consideration is given to the most appropriate tree species for a particular location, the large-scale replacement of street trees with more appropriate species is unlikely to take place. This means that the species of street trees planted as part of current initiatives are those that would broadly be expected to be present in 2050, under these scenarios. In addition, within the MF and FW scenarios, the species of tree that are retained and planted are likely to be those that happen to be in fashion or that have particular value in terms of timber or fuel. Other conditions which appear particularly vulnerable are the continued presence of a street tree at the original planting site, the tree‘s roots not spreading excessively, its maintenance to benefit wildlife and its structural connectivity to a broader tree network. Whilst a necessary condition may be vulnerable in several scenarios, the reasons for this were not always the same. For example, the structural connectivity of tree networks may not always be supported within the PR scenario because high-density land redevelopment to deliver key social goals takes priority over environmental concerns. In this scenario, tree networks may become structurally fragmented as policies are implemented to achieve a more compact urban form and to deliver new public transport systems. Although policy would specify the need for mitigation, trade-offs would be expected where infrastructure projects have a particularly high social value. Tree networks may also be removed in the MF scenario, but for different reasons such as the avoidance of damage to built infrastructure, the reduction of litigation risks and the widening of major transport corridors. The protection and management of urban tree networks is also less likely within the FW scenario, in which a ‗tragedy of the commons' has unfolded within much of the city; street trees in many areas are illegally taken for timber and fuel by the impoverished citizens, with the government unable or unwilling to prevent this.

Benefits to achieve and drawbacks to avoid	Necessary conditions																										
	Consistent water supply for healthy growth	Tree's access to light is maintained	Low stress from soil pollution	Low stress from air pollution	Root growth not substantially impeded	A tree is still present	Tree is large or mature	People are present nearby	Tree is visually accessible to public	Tree is maintained for amenity	Large-scale tree-cover across urban area	Tree is maintained for wildlife	Surrounding area built to high-density	Tree is physically accessible to public	Tree not in a street canyon with busy road	Tree does not overhang road or pavement	High canopy	Tree blocks solar access to building	No artificial lighting	Tree is part of a densely-vegetated barrier	No persistent noise	Tree is connected to a broader tree network	Species is native	Species is low a VOC emitter	Lateral root spread not excessive	Tree is growing in a pervious surface	Species is evergreen
Provide feeding resource for native birds and bats	*	*	*	*	*	✓	✓	!		!	✓	✓	!	!	✓				✓		✓	✓	✓			*	
Cool buildings (shade)	✓	✓	✓	✓	✓	✓	✓								!			✓									*
Reduce heating requirements during cold weather	✓	✓	✓	✓	✓	✓	✓	✓					!				!	!		✓							✓
Attenuate noise throughout the year	✓	✓	✓	✓	✓	✓	✓			!							!			✓						*	✓
Reduce net CO_2 emissions	✓	✓	✓	✓	✓	✓	✓			!	✓															*	
Reduce stormwater runoff rate/volume	✓	✓	✓	✓	✓	✓	*				✓															✓	*
Summertime cooling	*	✓	✓	✓	✓	✓	✓				✓				!	!		✓									
Reduce exposure to air pollutants (NO_2, O_3, PM)	✓	✓	✓	✓	✓	✓	✓	✓		*	*				✓									✓			*
Reduce psychological stress	✓	✓	✓	✓	✓	✓	*	✓	✓					*			✓										

Table 1. Conditions necessary for securing key benefits from urban street trees and avoiding drawbacks. See S2 for supporting justifications. **KEY** ✓ = Condition is typically necessary for delivering intended benefit. * = Condition may be necessary in some contexts. ! = Potential conflict between condition and a particular benefit.

Benefits to achieve and drawbacks to avoid	Necessary conditions: Consistent water supply for healthy growth	Tree's access to light is maintained	Low stress from soil pollution	Low stress from air pollution	Root growth not substantially impeded	A tree is still present	Tree is large or mature	People are present nearby	Tree is visually accessible to public	Tree is maintained for amenity	Large-scale tree-cover across urban area	Tree is maintained for wildlife	Surrounding area built to high-density	Tree is physically accessible to public	Tree not in a street canyon with busy road	Tree does not overhang road or pavement	High canopy	Tree blocks solar access to building	No artificial lighting	Tree is part of a densely-vegetated barrier	No persistent noise	Tree is connected to a broader tree network	Species is native	Species is low a VOC emitter	Lateral root spread not excessive	Tree is growing in a pervious surface	Species is evergreen
Decrease perceptions of overcrowding	✓	✓	✓	✓	✓	✓	*	✓	✓			!	✓		!	!											
Create desirable environments for recreation	✓	✓	✓	✓	✓	✓	*	✓	✓	✓		!		✓			✓					✓			✓		
Improve urban aesthetics	✓	✓	✓	✓	✓	✓	*	✓	✓	✓		!													✓		
Increase property values	✓	✓	✓	✓	✓	✓	*	✓	*	✓		!															
Reduce crime	✓	✓	✓	✓	✓	✓	!	✓	✓	✓	✓	!					✓		!	!							
Increase economic investment within surrounding area	✓	✓	✓	✓	✓	✓		✓	✓	✓	✓	!							!		!						
Stabilise cuttings/embankments						✓	✓							!													
Avoid root interference with built infrastructure/paving	✓	!	!	!	!	!	!				!		!					!				!			✓		
Avoid shrink-swell damage to buildings/infrastructure	✓	!	!	!		!	!				!		!													✓	
Avoid public hazard due to leaf/fruit fall	✓	!	!	!	!	!	!			*	!	!		!		*											
Avoid injury/damage due to branch/tree fall	✓	✓	✓	✓	✓	!	!			✓	!	!	!	!		✓	!										

Table 1 ctn.

Table 2.

Necessary Conditions	**Future scenarios**			
	PR	MF	FW	NSP
Species is native	*	*	*	*
Species is a low VOC emitter	*	*	*	*
Species is evergreen	*	*	*	*
A tree is still present	*	*	x	✓
Lateral root spread is not excessive	*	x	x	✓
Tree is connected to a broader tree network	*	x	x	✓
Tree is maintained for wildlife	*	x	x	✓
Tree is not in a street canyon with busy road	*	*	x	✓
Tree is maintained for amenity	*	✓	*	x
Consistent water supply for healthy growth	*	*	*	✓
Root growth not substantially impeded	*	*	*	✓
Tree's access to light maintained	*	*	*	✓
Tree is large or mature	*	*	x	✓
High canopy	*	*	x	✓
Tree is part of a densely-vegetated barrier	*	*	x	✓
No persistent noise	*	*	*	✓
No artificial lighting	*	*	*	✓
Tree blocks solar access to building	*	*	*	✓
Surrounding area built to high-density	✓	*	*	*
Tree does not overhang road or pavement	*	✓	✓	x
Low stress from air pollution	✓	*	x	✓
Low stress from soil pollution	✓	*	*	✓
Tree is physically accessible to public	✓	*	*	✓
Tree is growing in a pervious surface	✓	*	*	✓
Tree is visually accessible to the public	✓	*	*	✓
People are present nearby	✓	*	✓	*
Large-scale tree cover across urban area	✓	✓	x	✓

A summary of the results of the scenario based resilience analysis. Those necessary conditions considered vulnerable within a particular scenario are marked with ‗X‘ in the corresponding column. ‗*‘ represents those considered partially vulnerable, whilst ‗✓‘ is used to indicate where a condition is likely to be supported. PR = the Policy Reform scenario, MF = Market forces, FW = Fortress World, NSP = New Sustainability Paradigm. The scenario characteristics used to support this analysis are given in S3.

4. Discussion

The analysis presented here is based upon a generic proposal to plant a single street tree within an urban area in the UK, and would therefore need to be adapted for a more specific planting proposal and for a particular geographical location. The outcomes of the analysis are also sensitive to the variety of disciplines involved in identifying key benefits and their dependencies, and to how they interpret the scenario characteristics. Our aim was to consider the broad range of benefits that might be derived from urban street trees in the UK and to capture the diverse system conditions upon which they depend over time. There is evidence that studies of ecosystem services tend to focus on biophysical or economic dimensions and much less on socio-cultural services and drivers (Menzel and Teng, 2010, Martín-López et al., 2012). To a great extent this bias results from the difficulty in quantifying the latter. We argue that by ensuring a range of disciplinary ‗voices' were at the table and that by using highly contrasting future scenarios, we forced a broader questioning of the social, technological, economic, environmental and political dependencies. However, we acknowledge that our analysis may still be limited by deficiencies or biases within academic and practitioner knowledge. This is a fundamental problem in the study and synthesis of complex systems. This knowledge gap has been recognised and participatory research processes have been proposed as one mechanism for improving our understanding the social-cultural dimension of ecosystem services (Menzel and Teng, 2010). Despite these caveats, the outcomes are considered to be broadly indicative of how the benefits delivered by urban street trees might be vulnerable to loss over time.

4.1. Benefits, Necessary Conditions, Synergies and Tensions

The variety of necessary conditions, synergies and tensions identified in Table 1 illustrates both the diversity of factors that may influence street tree performance and the complexity of the urban system within which they are embedded. It raises questions about whether urban tree planting programs are able to realise such a broad range of intended benefits in the short term, and then to sustain them until 2050, a timeframe that is still significantly less than the potential lifespan of most tree species planted in urban areas.

We find that environmental benefits, which are often cited as the rationale behind urban greening programmes due to the relative ease of their monetisation, depend on system conditions which are not always mutually compatible. For example, using a street tree to reduce summertime air temperatures and to cool an adjacent building (via shading), may conflict directly with a desire to warm the building in winter, should the shading limit solar gain. Likewise, whilst it is desirable to block cool air flows in winter, the opposite may be desired in summer in continental climate zones. This clash has long been recognised in urban tree planting literature (McPherson et al., 2008), where it is recommended that deciduous trees be used for shade, and evergreen trees be used to provide wind shelter along the northern perimeter of a building (in northern hemisphere sites), providing an example of how tensions can be resolved by careful planning, as long as such benefit trade-off information is recorded and built into management plans. Spatial incompatibilities of intended benefits may also occur in areas such as busy street canyons, where street trees may be effective at providing useful shade and reducing perceptions of overcrowding, yet perform poorly in relation to air quality, where they can trap air pollutants emitted within that canyon, increasing population exposure (Gromke et al., 2008). This may be at least partially resolved by high levels of canopy thinning (Jin et al., 2014), although pruning, combined with street noise, lighting and moving vehicles would be expected to undermine many of the tree‘s potential biodiversity benefits (Forman and Alexander, 1998). Such synergies and tensions are however not universal, and careful analysis of the local context can reveal ways to reduce some potential conflicts. For example, in a busy street canyon where only electric vehicles were permitted or where trees are heavily pruned, conflicts between improving air quality and delivering shade would be much reduced.

What we broadly term as ‗social‘ benefits, such as: creating desirable environments for recreation and health, improving urban aesthetics, increasing residential and business property values, increasing inward investment in the area and decreasing perceptions of overcrowding, are often compatible with each other. However, the conditions that support social benefits are often incompatible with those necessary for ecological benefits, such as providing an effective feeding resources for urban bird and bat communities. This incompatibility is partly due to the more aggressive maintenance that trees in streets tend to be subjected to (e.g. the removal of insect-rich standing dead wood (Tyrväinen et al., 2003)), which is also linked to how local communities value different forms of tree cover (Dandy, 2010, Conway and Bang, 2014). In addition, trees in areas of high population density are at risk of being subjected to artificial lighting and noise, which are known to have negative impacts on urban invertebrates, birds and mammals (Forman and Alexander, 1998, Parris and Schneider, 2009, Gaston et al., 2013, Hale et al., 2015). As a result, tensions between social and ecological benefits would be expected to make street trees vulnerable to removal/functional simplification under certain future scenarios.

4.2. Scenario analysis and resilience implications

Scenarios have been used in a variety of ways to consider how ecosystems and the services they provide might change in the future (Carpenter et al., 2006, Metzger et al., 2008, Bateman et al., 2013, Deal and Pallathucheril, 2009) although this approach has rarely been applied at the city or sub-city scale (c.f. Perino et al., 2011). Their purpose here was to broaden the debate about urban street trees and resilience, to include not only threats that are expected to increase over time (e.g. climate change, pests and diseases (Tubby and Webber, 2010)), but also socio-political changes that may be much less predictable. Many of these scenario characteristics can be recognised in urban areas around the world. The contrasts between these scenarios can be used to highlight and question the implicit assumption within today's urban planting proposals that key urban conditions will persist. For example, the initiation of ‗Million Tree' planting schemes by politicians implies that there is currently broad public support for large-scale urban tree planting, yet public attitudes to urban trees are highly variable (Dwyer et al., 1991, Zhang et al., 2007, Pincetl, 2013, Conway and Bang, 2014). In this case, our scenario analysis helps provide a structure for questioning whether key benefits would be sustained if local people become less supportive of urban street trees in the future. Some planting initiatives also appear to depend on broad public involvement, as they rely in part on residential and other private land owners for providing planting sites and subsequent tree maintenance (Gillespie et al., 2012, Pincetl, 2013). In a future where the management responsibilities for urban street trees shifts further from the municipal to the individual level, long-term success may well depend on a sustained shift in public attitudes regarding responsibilities for urban tree stewardship (Moskell and Allred, 2013) and on improved participatory democracy in the form of a greater/more formal integration of volunteers into city management functions (Pincetl et al., 2013). Similarly, given that funding for the maintenance of urban trees is often considered inadequate (McPherson et al., 1997, Dandy, 2010) and likely to be further reduced in many areas in the context of fiscal austerity (Rotherham, 2010, Young, 2011, Pincetl, 2013), our methodology prompts the user to consider whether current planting strategies and techniques are sufficient to ensure that street trees planted today could survive in a future where maintenance budgets were virtually non-existent.

The vulnerabilities identified during this process might be addressed in a variety of ways; our analysis aims to initiate a broad-based discussion about potential risks to the long-term delivery of urban ecosystem services and to help structure the response. Essential to this is the recognition that urban trees are part of a complex social-ecological system (Pincetl et al., 2013) and that their delivery of benefits over the long-term relies on more than simply the persistence of the trees themselves. Broad strategies proposed for improving resilience in cities include maintaining high response diversity, multi-functionality, redundancy and decentralisation (Ahern, 2013). These themes are explored within the sections below, with suggestions for areas of future research. Our analysis highlights three classes of system conditions which appear to be particularly important: (i) retention and survival to maturity, (ii) large-scale planting and (iii) social context. We discuss vulnerabilities and the resilience implications for key benefits below.

4.3. Retention and survival to maturity

The presence of a tree is a self-evident requirement for the delivery of each benefit (Table 1) and it is therefore unsurprising that various forms of direct protection for individual street trees are commonly put in place within UK urban areas (Dandy, 2010, TDAG, 2014) and elsewhere. However, from this analysis street trees are still considered to be vulnerable to removal within their natural lifespan, either directly or via the degradation of key growing conditions; in some scenarios these trees may not be replaced (Table 2). Retention of tree cover over long time periods by the replacement of dead trees (MillionTreesNYC, 2013) might help to improve the resilience of some benefits. However, many benefits are most effectively delivered by large or mature trees, as they are often highly visible, can have disproportionally high ecological value (Le Roux et al., 2014), and exert a considerable influence over microclimates.

Long payback periods may be required before the benefits of urban trees have covered the initial costs of planting (McPherson et al., 1997), yet half of urban street trees may die before they reach 13 to 20 years old (Roman and Scatena, 2011). Those trees that do survive to maturity can generate a variety of tensions in urban areas (Table 1, Carpaneto et al., 2010, Dandy, 2010, Conway and Bang, 2014), creating pressure for their removal. There is a perception amongst practitioners that large street trees are already being lost in UK urban areas (Dandy, 2010, Rotherham, 2010) and such trees appear even more vulnerable under the MF and FW scenarios (Table 2). Key drivers for tree loss, restricted growth or periodic replacement include direct impacts from climate change (MF, FW), pests and disease (MF, PR), competition for space within high-density developments (PR, FW, NSP), and concerns about the costs of infrastructure damage (MF), litigation (MF), health impacts (MF, PR) and maintenance (MF). This implies that to reduce the risk of tree loss, careful attention should be given to the precise location of planting. Areas to avoid are sites where threats to public safety might reasonably be expected now or in the future, where high densities of utilities are present/expected, or where physical (re)development is a realistic risk during the natural life-span of the tree. Resilience to climate and disease impacts will also be increased by ensuring a diversity of genotypes, species and genera are planted (TDAG, 2014). This ‗response diversity‘ can be achieved through an approach that identifies ecological niche function and that selects several species that deliver similar benefits, but that respond differently to shifts in growing conditions, pests and anthropogenic pressures.

In some cases, more fundamental changes to how urban trees are valued, owned and managed may be required (Rotherham, 2010, Young, 2011). In a future where market forces dominate, high land-values combined with the risks posed to valuable built infrastructure may create pressures to remove urban street trees that social and governance systems are unable to resist. Resilience might therefore be improved through the development of market-based systems that enable the multiple functions of an individual street tree to be better captured within decision-making processes.

Payment for ecosystem services (Salzman, 2005) has been proposed as a tool for their management in urban areas (TEEB, 2011) and long-term funding for non-profit and community actors has been identified as an important route for securing the stewardship of urban trees (Young, 2011). However, we found no evidence that market-based systems have been considered as a tool for increasing the resilience of any of the large-scale urban tree planting initiatives listed in S1.

The availability of sufficient water and root space are also key conditions that must be sustained in order for urban street trees to reach maturity, both in terms of promoting tree growth and avoiding conflicts from the lateral expansion of their roots. Protecting these conditions may be challenging in a future where the local built density has broadly increased (PR, FW, NSP) or where the levels of protection and maintenance for urban trees have declined (MF, FW). However, technical solutions implemented at the time of planting could provide some resilience, such as the use of dedicated soil cells, suspended permeable pavements and the integration of planting sites with surface water drainage systems (TDAG, 2014, Mullaney et al., 2015). In addition, planting techniques that make the likelihood of future translocation more successful may provide useful flexibility. This decentralised approach may help to isolate individual trees from broader changes to the water table and pollutants within urban soils. Whilst such solutions may not be practical for all urban planting situations, they may be cost-effective for high-value trees in high-density locations where large numbers of people may benefit and where the potential costs of damage to built infrastructure are considerable.

4.4. Large-scale tree cover

Several benefits that are claimed for urban tree planting require the presence of large-scale canopy cover in order to be effective, e.g. CO2 assimilation, providing feeding resources for wildlife, and reduced stormwater runoff. In effect, the planting of a single street tree will have little impact on delivering these benefits if it is located within a landscape that is largely devoid of tree cover. Our futures analysis shows that although in most scenarios large-scale urban tree cover is maintained, considerable changes might be expected at the neighbourhood scale. Losses are likely in scenarios where infill development is common and where redevelopment typically occurs at higher built densities (PR, FW, NSP), and in poorer neighbourhoods where the maintenance of urban trees has become less of a priority (MF, FW). Large-scale planting also greatly increases the chances of drawbacks being realised (Table 1), with the potential for economic and social costs due to infrastructure damage, litigation and health impacts. Once again, improved methods for capturing the value of key benefits within economic and governance systems (James et al., 2009) may incentivise the retention of broad-scale tree cover, under scenarios where markets have greater power and where social attitudes to the environment are less supportive. High tree species diversity at the neighbourhood scale, as well as a large number of nodes within tree networks may also provide useful ecological redundancy, ensuring that alternative feeding resources and dispersal routes exist for bird and bat communities, should some be lost over time.

4.5. Social context

For many of their potential benefits to be delivered, urban trees need to be located near to people, yet trees can also result in a range of negative impacts on human wellbeing (Dwyer et al., 1991, Roy et al., 2012). Our analysis highlights the added complication that the magnitude of some social costs and benefits could vary over time and may be highly sensitive to changing social values (Ordóñez and Duinker, 2010). In a future scenario where market forces dominate (MF) or consumerist and individualist attitudes prevail (MF, PR, FW), street trees are vulnerable to removal if the (perceived) risks to health, safety or nuisance are high. Increased pressure to remove street trees might also be expected, as a response to litigation risks and infrastructure damage. One strategy to improve the resilience of urban tree planting could be to target parcels of private land that have relatively low densities of buried infrastructure, yet are still close to busy public streets. Gardens/yards and small amenity green spaces within high-density residential developments might therefore prove to be more resilient planting sites than paved areas immediately adjacent to roads, delivering higher rates of survival and growth (McPherson et al., 1997). However, homeowner support for such planting locations is not universal (Pincetl et al., 2013, Conway and Bang, 2014) and would be much reduced within the Market Forces scenario. This tension might therefore be reduced by making direct payments to residents/housing managers for hosting and managing urban street and yard trees and the ecosystem services provided; analogous to the practice of paying residents to create ‗rain gardens' for stormwater interception (Thurston et al., 2010). Such payments might be best targeted at economically deprived areas where urban tree cover is often poor (Landry and Chakraborty, 2009), and where its benefits may be most needed.

Large tree planting initiatives have had short-term success in engaging volunteers with site identification, planting and the management of urban trees (Young, 2011, MillionTreesNYC, 2013), often on private land. However, our analysis flags up the risk that public support might decline in the future. The active engagement of individuals, organisations, agencies and institutions in urban tree planting campaigns is already common practice in the UK (TDAG, 2014) (e.g. TreeBristol, The Mersey Forest, and Plymouth Tree Partnership). Yet meeting the different needs and aspirations of stakeholders within large planting projects can be challenging (Pincetl, 2010, Pincetl et al., 2013). Resolving these tensions could increase management ‗esponse diversity‘, by ensuring sufficient social capital is sustained over time (Folke et al., 2005) to adapt to the changing needs of urban street trees. Whilst resolving long-term funding issues is clearly important (Pincetl et al., 2013), co-management might also be strengthened by greater clarification of the roles and responsibilities‘ of different actors (Folke et al., 2005).

5. Conclusions

Our analysis makes explicit the conditions that are necessary to realise many of the potential benefits of urban street tree planting within a UK urban context. It identifies synergies and tensions between these benefits and questions the implicit high-level assumption within many planting campaigns that trees will survive and provide ecosystem services far into the future. We argue that by focusing on the system conditions upon which these benefits depend (rather than the tree itself), we are better able to examine the underlying mechanisms that drive their compatibility and resilience. Benefits are often dependent on conditions such as continued levels of tree maintenance, on public values and policies which are supportive and on a built form which remains largely unchanged. However, we have illustrated that these conditions may not be supported within plausible scenarios for future cities and that some changes to current practice are required in order to make the desired benefits of urban tree planting more resilient.

We suggest that large-scale urban tree planting projects should include explicit statements about which benefits will be prioritised and the timescales over which they are intended to be delivered. As with other pieces of urban infrastructure, risks to long-term performance should then be identified and holistically addressed. Ensuring the replacement of ‗lost' street trees is necessary but insufficient; tree survival to maturity is also vital for the delivery of many benefits. Although current best practice is to ensure urban tree planting is compatible with its social and built context, we suggest that this be broadened to consider the impact of plausible changes to the tree's context over its natural lifespan. The main aim of this paper is to illustrate how such an analysis could be undertaken. We also make some recommendations for how the resilience of urban street tree planting might be improved:

- Broaden planting locations to include private gardens and residential amenity green spaces immediately adjacent to streets, to reduce potential conflicts with people and built infrastructure and to reduce tree mortality due to environmental extremes.

- Introduce annual direct payments for local residents and business owners, to incentivise their involvement in the long-term protection and management of trees, in neighbourhoods where benefits are most needed.
- Develop more formal partnerships between the individuals, NGOs and municipal departments that are involved in the co-management of street trees in urban areas, to increase their legitimacy, accountability and ability to access and share resources.
- For planting in heavily developed areas such as urban centres, incorporate soil cells integrated with surface water drainage systems, and use planting techniques that facilitate the transplantation of trees at a later date if necessary.

Such changes would involve the broadening of current practice, requiring a greater integration of urban foresters and arboriculturists with the long-term spatial planning, funding, governance and infrastructure management processes within urban areas.

Supplementary Materials

Supplementary Information can be found in the Appendix at the end of this thesis.

S1: Identifying benefits and drawbacks of urban tree planting for use in the scenario-based resilience analysis, Supplementary Information S2: Justifications used to support the necessary conditions identified in Table 2 of the main manuscript, Supplementary Information S3: A scenario-based analysis of the vulnerability of the conditions required for an urban street tree to deliver its intended benefits.

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Author Contributions

Christopher Boyko, Julie Brown, Silvio Caputo, Maria Caserio, Richard Coles, Raziyeh Farmani, James Hale, Chantal Hales, Dexter Hunt, Joanne Leach, Rob MacKenzie, Thomas Pugh, Christopher Rogers and Jon Sadler generated the data on benefits and necessary conditions and initiated the resilience analysis. James Hale and Thomas Pugh led the analysis. James Hale, Rob MacKenzie, Thomas Pugh and Jon Sadler wrote the paper. Silvio Caputo, Richard Coles, and Russell Horsey provided additional text and substantial feedback on an early manuscript draft.

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APPENDIX

Supporting information and supplementary files for each of the five papers are given below.

PAPER I - SI

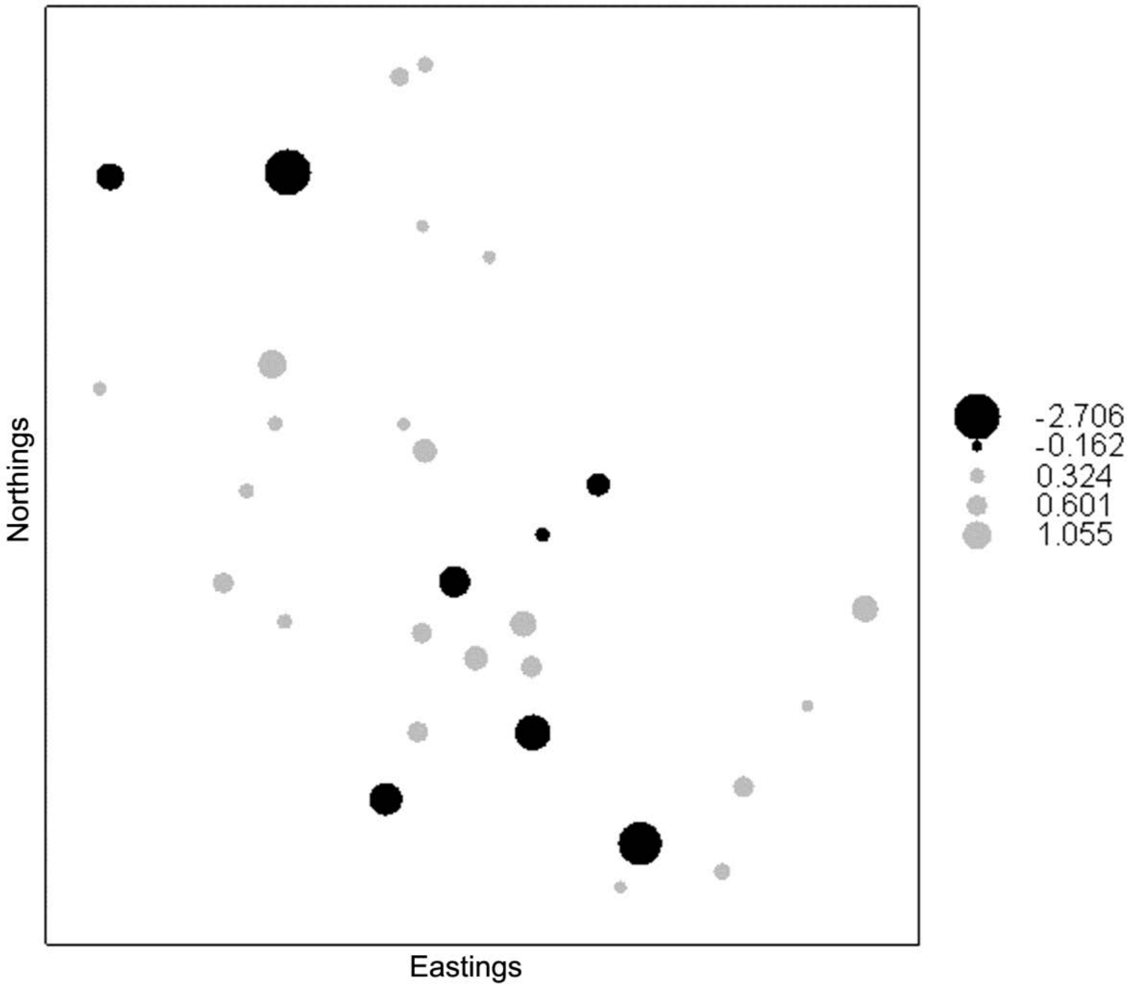

Figure S1. Residual bubble plot for NSL all-night Anabat data from logit binomial presence-absence data. The plot shows clumping of similar size positive residuals in the middle of the plot, indicative of spatial structuring in the data. Negative residuals in black and positive residuals are grey. The size of the circles indicates the size of the residuals.

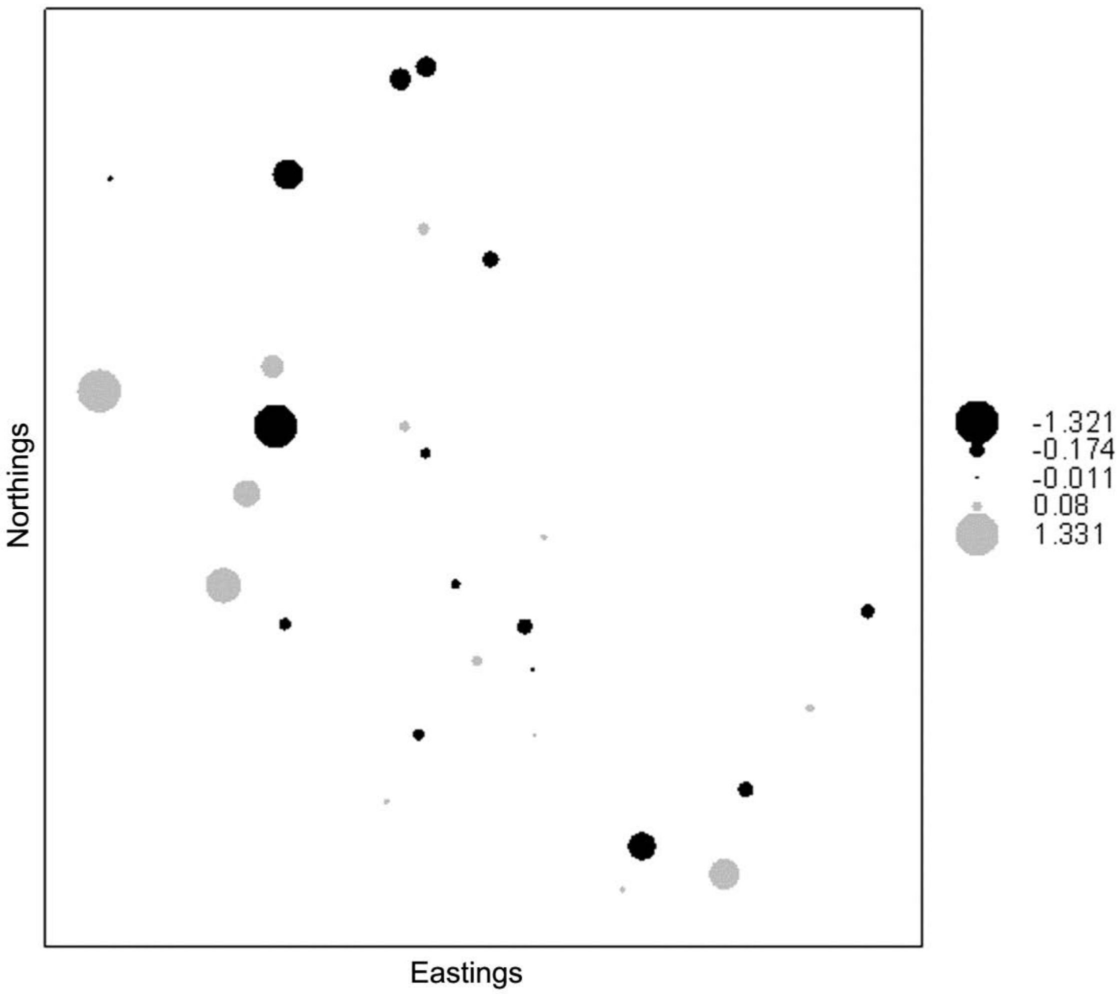

Figure S2. Bubble plot for *P. pipistrellus* all-night Anabat residuals from a GAM of bat activity minutes. The plot indicates no spatial structuring in the data. Negative residuals in black and positive residuals are grey. The size of the circles indicates the size of the residuals.

Class name	Class	Built structures		Gardens 0-400m²		Gardens>400 m²		Manmade (built) Open Space		Canals/rivers		Stillwaters	
		Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
Rural	1	5,227	4,726	6,007	9,076	17,552	12,281	14,370	9,107	709	1,161	1,313	1,407
Light suburban	2	14,266	11,282	18,680	20,991	37,781	33,742	24,876	15,240	2,011	1,837	7,659	9,107
Suburban	3	24,229	7,944	40,745	19,722	32,464	17,848	37,274	15,173	1,246	1,517	1,164	1,874
Dense suburban	4	37,502	8,858	76,756	18,492	17,820	10,781	39,164	9,631	562	942	370	961
Dense urban	5	51,307	15,910	30,202	17,521	9,573	9,035	65,137	17,939	3,440	2,240	597	1,338

Class name	Class	Motorways		A_roads		B_roads		Minor roads		Natural (vegetated) open spaces		Railways		Trees	
		Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
Rural	1	233	991	774	1,565	544	941	3,413	2,372	192,492	26,198	258	744	1,484	2,322
Light suburban	2	4,775	5,167	1,740	2,141	353	777	6,743	4,729	123,097	47,319	783	1,435	5,169	7,911
Suburban	3	52	435	1,653	1,919	753	1,140	11,794	3,641	94,461	31,104	789	1,664	941	2,008
Dense suburban	4	122	804	2,537	2,586	1,191	1,550	18,301	3,471	53,300	18,816	766	1,483	70	429
Dense urban	5	973	2,633	4,809	3,488	2,053	2,416	15,567	4,973	59,567	19,678	4,982	6,354	93	545

Table S1. Mean area (m^2) and standard deviation of Ordnance Survey (OS) land-cover type for each urban land class. Urban land classes were derived using a cluster analysis of OS land-cover data for 902 individual km squares.

Species	Summer roost preference	Emergence	Feeding preference
Pipistrellus pipistrellus (45kHz)	Built structures – within gaps and under cladding Preference for pre 1945 and damaged buildings	Average	Small flies Edge Half open areas Riparian vegetation and woodland edge
Pipistrellus pygmaeus (55kHz)	Predominantly built structures	Average	Small flies Edge Half open areas Strong riparian preference but also woodland edge
Myotis daubentonii	Structures such as stone bridges Tree hollows	Late	Aquatic insects Close to water Closed canopy Woodland edge
Eptesicus serotinus	Built structures – within gaps and under cladding	Early	Beetles + moths Open/Edge Parks and gardens Pasture Woodland edge
Nyctalus leisleri	Oak and ash trees that are larger than others locally available. 15-42m	Early	Flies, beetles + moths Open Wetlands Parkland Woodland edge
Nyctalus noctula	Woodpecker tree hollows. Beech and Oak trees, larger than others locally available. 18_44m	Early	Large insects - flies, beetles + moths Open Woodland Pastures Lakes

Table S2. Broad life history data for bat species recorded within the study area. A selection of information relating to land-cover preferences and flight behaviour of bats recorded in this study are presented, along with references for source data.

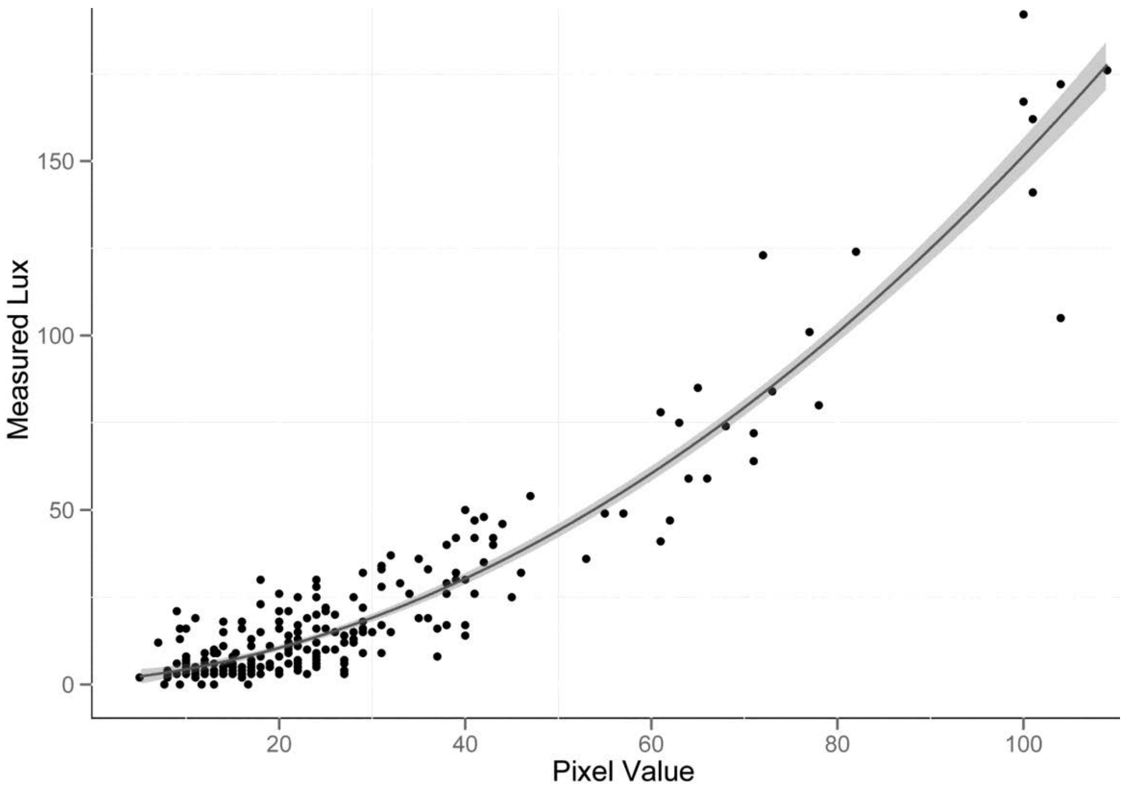

Figure S1. Ground incident lux plotted against corresponding greyscale pixel value for survey locations within Birmingham. The equation for the best fit line ($y = 0.0128X^2+0.2246X +0.8517$) was used to reclassify the greyscale raster. R2 = 0.9146. A 95% confidence interval is also indicated.

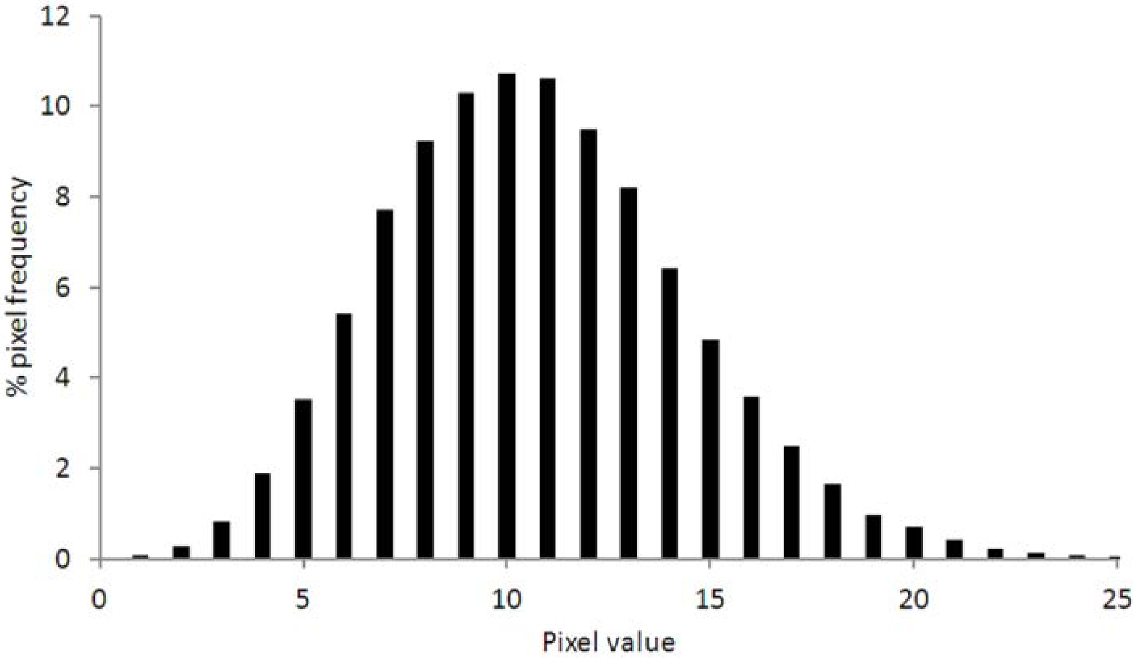

Figure S2. The distribution of greyscale pixel values for known —dark” locations (lit to <1lx).

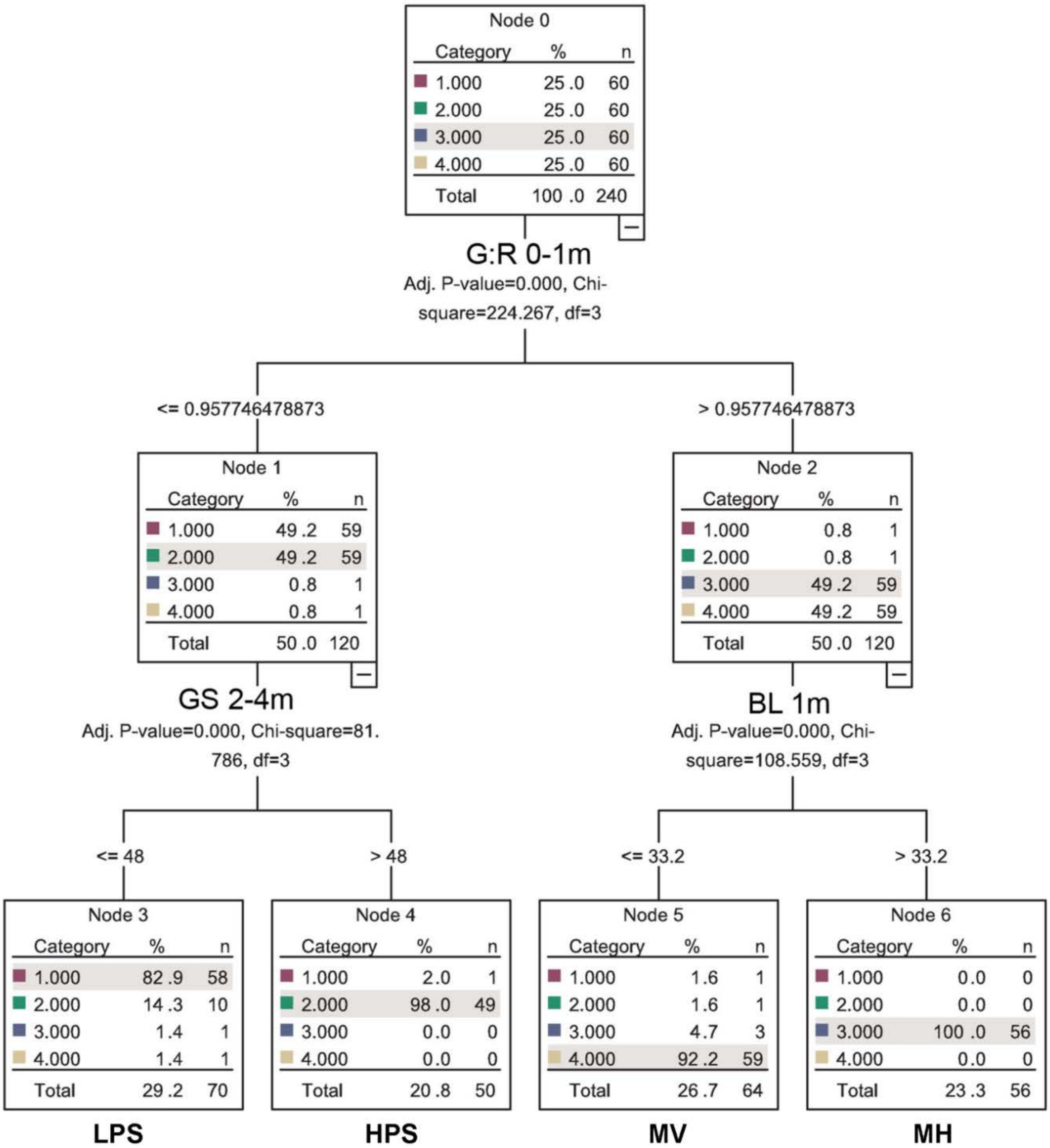

Figure S3. CHAID classification tree for lamp classes. 1 = low pressure sodium (LPS), 2 = high pressure sodium (HPS), 3 = metal halide (MH) and 4 = mercury vapour (MV). The first discriminating variable was the green to red ratio (G:R 0–1 m) for pixels up to 1 m from the lamp centre. LPS and HPS were then differentiated based on the maximum greyscale pixel value between 2 and 4 m (GS 2–4 m) from the lamp centre. MH and MV were differentiated based on the average blue pixel value up to 1 m from the lamp centre (BL 1 m).

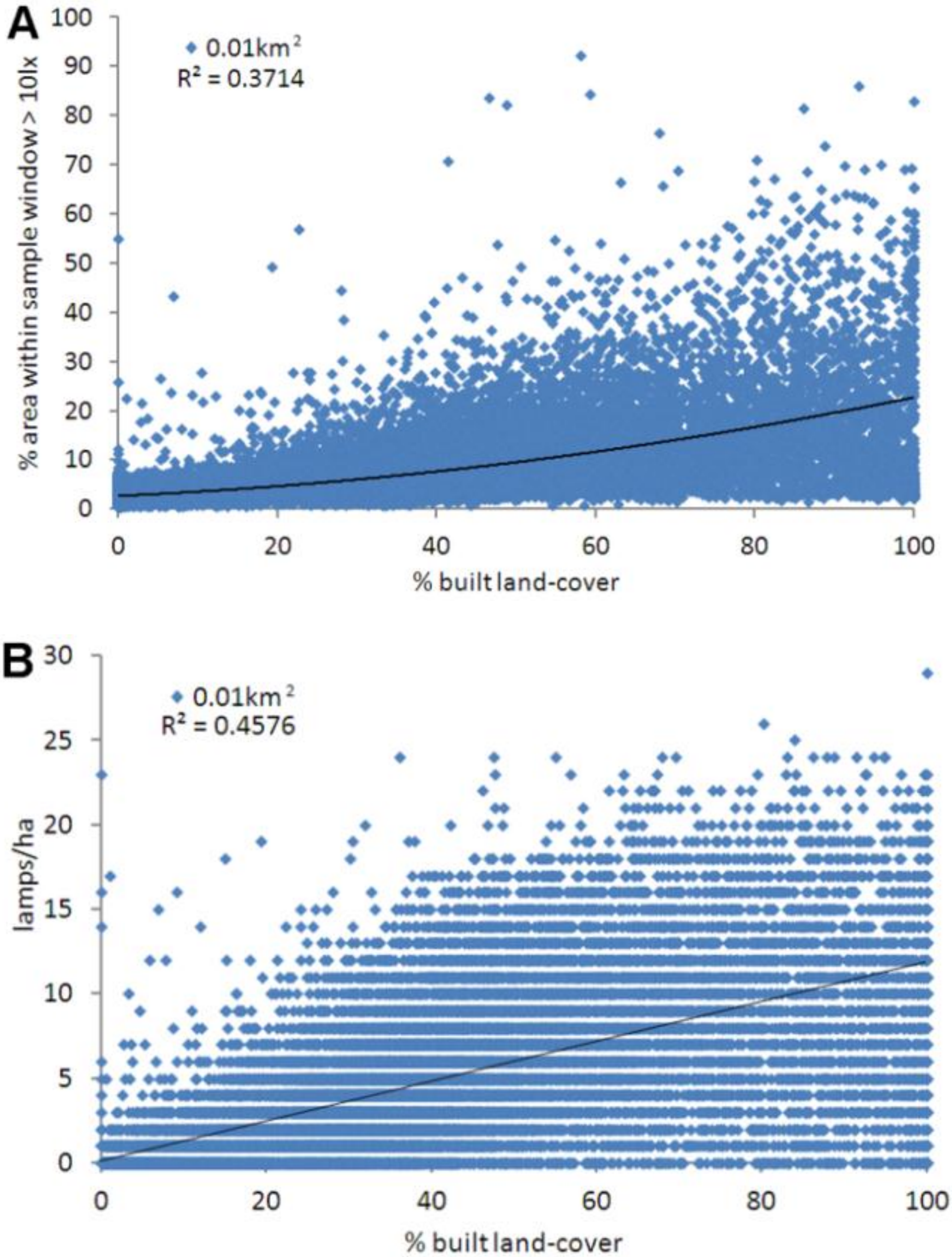

Figure S4. The results for sampling of lighting metrics at the 0.01 km^2 scale. (A) Percentage area ≥10lx and (B) density of lamps, both plotted against percentage built land-cover.

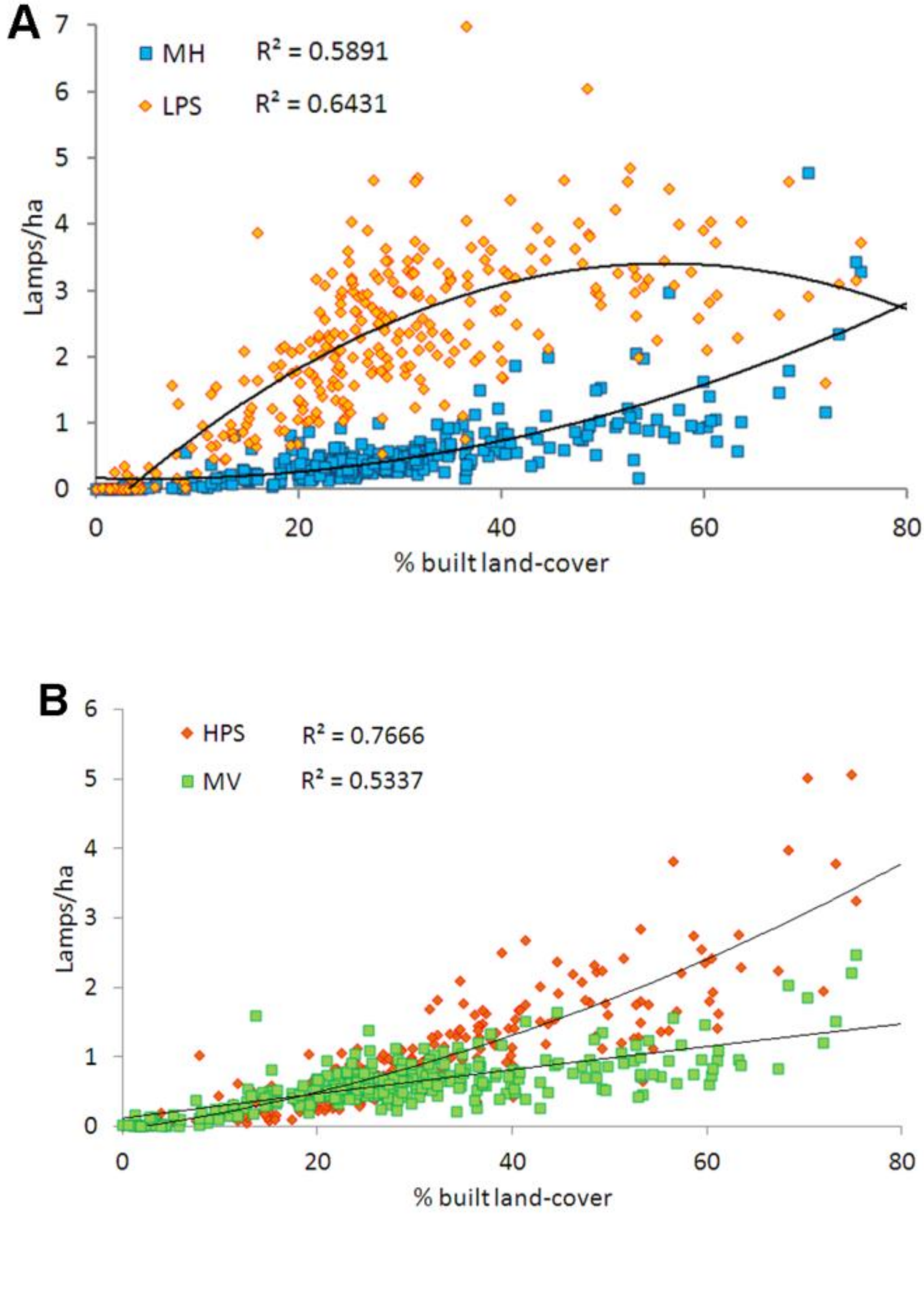

Figure S5. Changes in the density of lamp classes along the 1 km^2 urban gradient. (A) MH and LPS lamps and (B) MV and HPS lamps.

Classification type	Land-use class	Land-use as % of total city area	% contribution to total city lighting ≥ 30lx	% contribution to total city lamps
OSMM	Natural land-covers	33	13	9
	Gardens	29	5	11
	Roads & Pavements	15	29	53
	Buildings	14	13	11
	Other built surfaces	9	40	15
NLUD	Housing	52	23	55
	Leisure/recreational open space	16	4	4
	Manufacturing	7	23	11
	Agriculture	7	1	1
	Transport	4	12	6
	Education	4	5	5
	Utility services	3	5	1
	Retail distribution and services	3	11	6
	Unused land	2	4	2
	Community and health	2	6	4
	Wholesale	1	2	1
	Office	0.5	3	1
	Storage	0.2	1	0.3
	Defence	0.1	0.1	0.2

Table S1. Land-uses and lighting metrics for the city of Birmingham. Land-uses are given as a fraction of total city area, along with their contribution to the total city area lit ≥30lx and to the total number of city lamps. Two alternative measures of land-use are given; land-use parcels based upon the Ordnance Survey MasterMap (OSMM) (2008) and land-use zones based on the National Land Use Database (NLUD) categories (1995).

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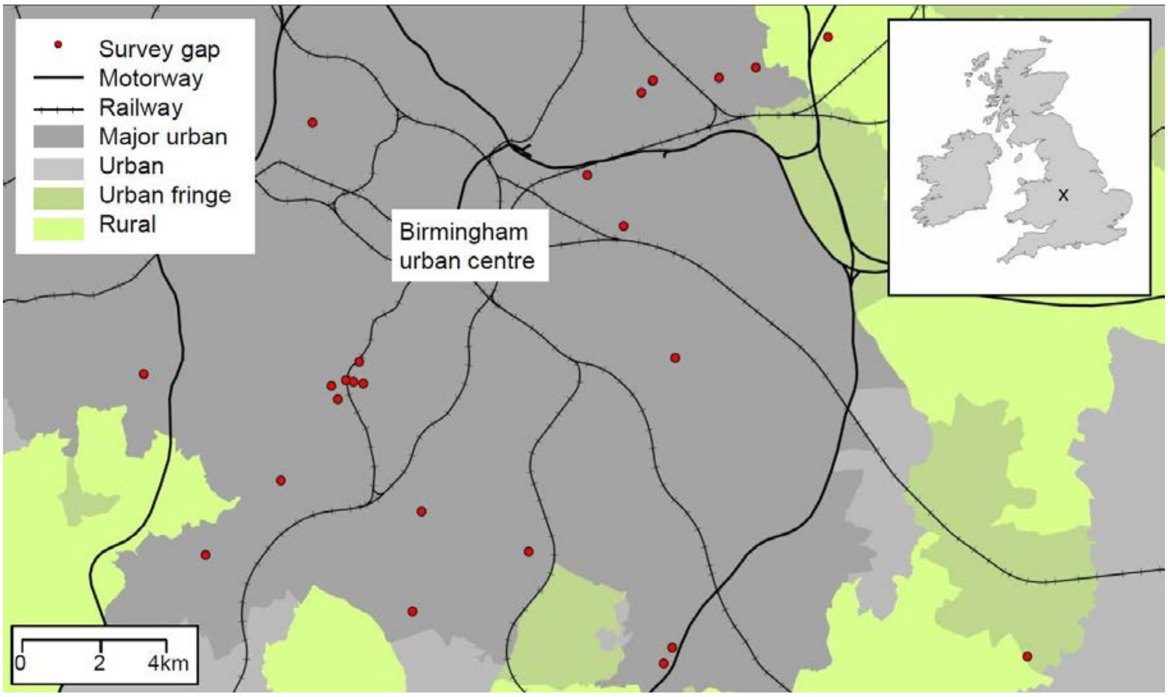
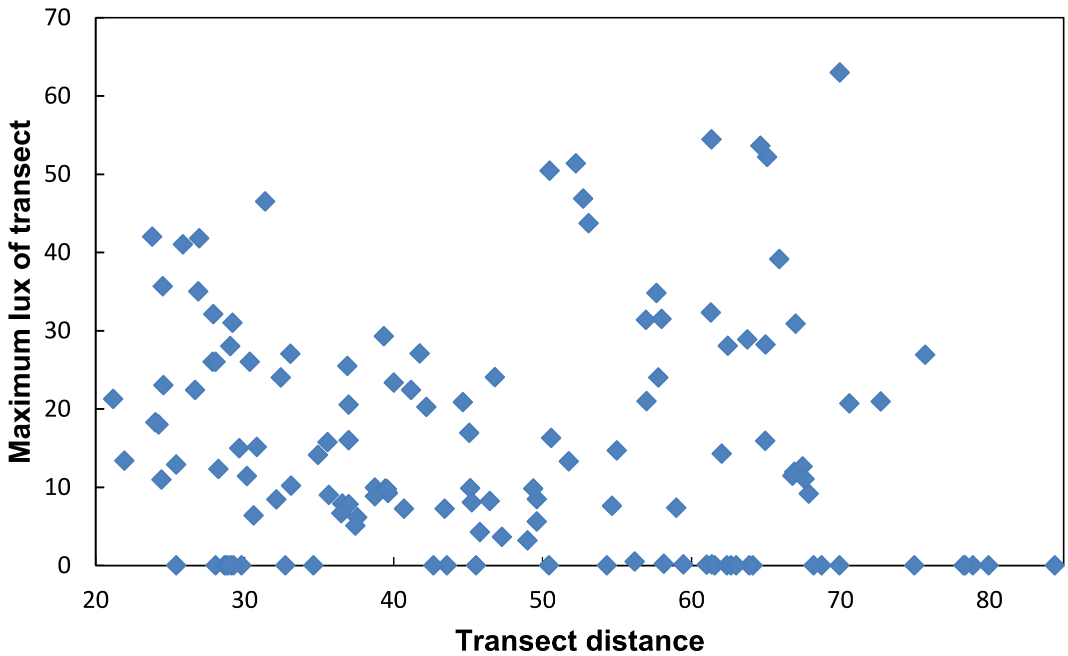
PAPER IV – SI

S1. Information on survey gap locations and characteristics.



The above figure indicates the position of the 27 survey gaps within the UK West Midlands (inset). The majority of sites are within the City of Birmingham, with some sites selected from adjacent urban areas and the urban fringe. Dense urban areas such as the centre of Birmingham were avoided. The percentage of built land-cover within a 350m radius of each site was <60%.

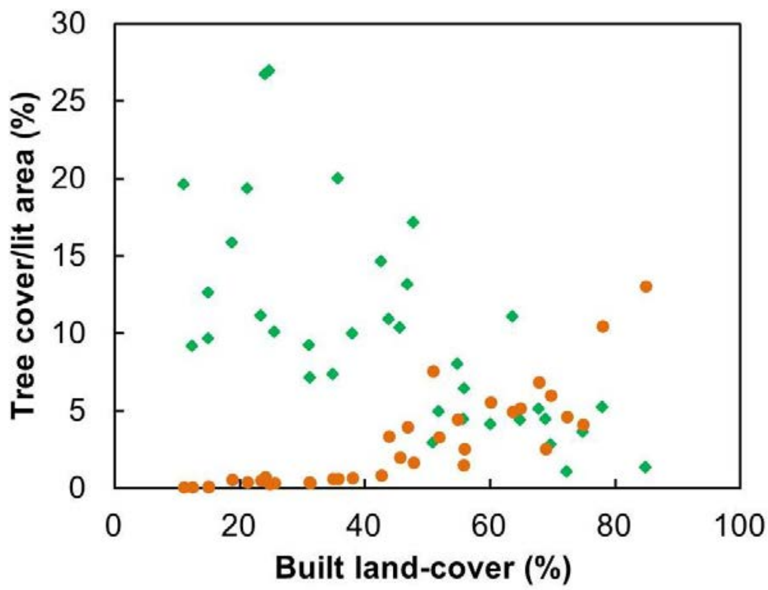


The above figure illustrates the variety of gap crossing conditions within the survey gaps. Although the 27 survey gaps were stratified by the median gap width and lux, an individual gap could vary considerably in lux (and to a lesser extent width). Lux and distance were measured using 5 transects at each candidate gap, to support the stratified selection of survey gaps. The values for all transect data from the final survey gaps are represented by the blue diamonds in the above graph.

		Gap lux			
		Dark	**0-20 lux**	**20+**	**Total**
Gap width (m)	**20-40**	6.5 (0.7)	4.4(2.2)	1.3(1.5)	48
	40-60	5(0)	4(1.4)	0.7(1.2)	20
	60-80	6(3)	0.0	0.0	18
	Total	41	38	7	

Survey gaps were stratified by 9 broad categories based upon their median gap width and lux. This table illustrates the variation in crossing frequency of *P. pipistrellus* within these broad categories. Numbers within the main body of the table represent the average crossing frequency for each gap category, along with the standard deviation in parentheses. Totals for number of crossing events within each width or lux category are given in the final column or row (respectively).

S2. Changes in tree cover and lighting along a built density gradient.



S2. Percentage tree cover (green diamonds) and percentage surface area >= 20 lux (orange dots) for a 350m radius buffer surrounding each sample pond, plotted against percentage built land-cover.

PAPER V - SI

Table S1. Benefits listed in support of large-scale urban greening programmes.

Benefit	**Number of citations**	**Programme ref**
Attract wildlife (biodiversity)	3	1, 3, 4
Mitigate against climate change	1	1
Carbon capture	2	3, 4, 6
Save energy	5	1, 2, 3, 5, 6
Reduce power plant emissions (through lower energy use)	1	3
Reduce greenhouse gases	1	4
Reduce summer air temperature	1	3
Shade	1	1
Shelter	1	1
Increased longevity of street surfaces	1	2
Reduce air pollution	6	1, 2, 3, 4, 5, 6
Reduce asthma and respiratory diseases	1	3
Reduce flood risk/capture stormwater	2	1, 5
Improve water quality (stormwater interception)	3	3, 4, 6
Reduce water consumption	1	4
Aquifer recharge	1	6
Reduce erosion	1	6
Reduce noise pollution	3	1, 2, 5
Improved aesthetics	3	1, 3, 4
Higher sales in business districts	3	2, 3, 5
Increase property value	3	2, 3, 5
Increased productivity among employees	1	2
Faster hospital recovery times	1	2
Lower crime	2	1, 2
Increased community pride	1	3, 4
Encourage physical activity	1	3
Create amenity spaces	1	4
Reduce stress	1	5
Calm traffic	1	5

1 The Big Tree Plant, UK, http://www.defra.gov.uk/bigtreeplant/ [Access: March, 2015].
2 Mile High Million, Denver, USA, http://milehighmillion.org [Access: March, 2015].
3 Million Trees NYC, USA, http://www.milliontreesnyc.org [Access: March, 2015].
4 Million Trees, South Australia, Adelaide, http://www.milliontrees.com.au [Access: March, 2015].
5 Million Trees LA, USA, http://www.milliontreesla.org [Access: March, 2015].
6 Million Trees, London, Ontario, http://milliontrees.ca [Access: March, 2015].

	Necessary conditions																										
Benefits to achieve and drawbacks to avoid	Consistent water supply for healthy growth[1]	Tree's access to light maintained	Low stress from soil pollution	Low stress from air pollution	Root growth not substantially impeded	A tree is still present	Tree is large or mature	People are present nearby	Tree is visually accessible to public	Tree is maintained for amenity[2]	Large-scale tree-cover across urban area	Tree is maintained for wildlife[3]	Surrounding area built to high density	Tree is physically accessible to public	Tree not in a street canyon with busy road	Tree does not overhang road or pavement	High canopy[4]	Tree blocks solar access to building	No artificial lighting	Tree is part of a densely-vegetated barrier[5]	No persistent noise	Tree is connected to a broader tree network	Species is native	Species is low VOC emitter	Lateral root spread not excessive	Tree is growing in a pervious surface	Species is evergreen
Provide feeding resource for native birds/bats	*a	*a	*a	*a	*a	✓	✓	!b		!c	✓	✓	!d	!b	✓				✓		✓	✓	✓e			*f	
Cool buildings (shade)	✓	✓	✓	✓	✓	✓	✓								!g			✓									*h
Reduce heating requirements during cold weather	✓	✓	✓	✓	✓	✓	✓						!j				!k	!m		✓							✓
Attenuate noise throughout the year	✓	✓	✓	✓	✓	✓	✓			!n							!k			✓						*o	✓p
Reduce net CO_2 emissions	✓	✓	✓	✓	✓	✓	✓			!q	✓r															*	
Reduce stormwater runoff rate/volume	✓	✓	✓	✓	✓	✓	*s				✓															✓	*t
Summertime cooling	*u	✓	✓	✓	✓	✓	✓				✓				!g	!g		✓m									
Reduce exposure to air pollutants (NO_2, O_3, PM)	✓w	✓	✓	✓	✓	✓	✓	✓		*	*x				✓y									✓			*p
Reduce psychological stress	✓	✓	✓	✓	✓	✓	*z	✓	✓					*			✓aa										
Decrease perceptions of overcrowding	✓	✓	✓	✓	✓	✓	*z	✓	✓			!bb	✓		!cc	!cc											
Create desirable environments for recreation	✓	✓	✓	✓	✓	✓	*z	✓	✓	✓		!bb		✓			✓					✓			✓		
Improve urban aesthetics	✓	✓	✓	✓	✓	✓	*z	✓	✓	✓		!bb													✓		
Increase property values	✓	✓	✓	✓	✓	✓	*z	✓	*dd	✓		!bb															

S2. Justifications used to support the necessary conditions identified in Table 2 of the main manuscript.
✓ = Condition is typically necessary for delivering intended benefit. * = Condition may be necessary in some contexts. ! = Potential conflict between condition and a particular benefit.

Benefits to achieve and drawbacks to avoid	Necessary conditions																										
	Consistent water supply for healthy growth[1]	Tree's access to light maintained	Low stress from soil pollution	Low stress from air pollution	Root growth not substantially impeded	A tree is still present	Tree is large or mature	People are present nearby	Tree is visually accessible to public	Tree is maintained for amenity[2]	Large-scale tree-cover across urban area	Tree is maintained for wildlife[3]	Surrounding area built to high density	Tree is physically accessible to public	Tree not in a street canyon with busy road	Tree does not overhang road or pavement	High canopy[4]	Tree blocks solar access to building	No artificial lighting	Tree is part of a densely-vegetated barrier[5]	No persistent noise	Tree is connected to a broader tree network	Species is native	Species is low VOC emitter	Lateral root spread not excessive	Tree is growing in a pervious surface	Species is evergreen
Reduce crime	✓	✓	✓	✓	✓	✓	![ee]	✓	✓	✓	✓	![bb]					✓		![ff]	![ff]							
Increase economic investment within surrounding area	✓	✓	✓	✓	✓	✓		✓	✓	✓	✓	![bb]							!		![gg]						
Stabilise cuttings/embankments						✓[hh]	✓[hh]							![jj]													
Avoid root interference with built infrastructure & paved surfaces[6]	✓[kk]	![mm]	![mm]	![mm]	!	!	!				!		!					!				!			✓		
Avoid shrink-swell damage to buildings & infrastructure[6]	✓[kk]	![mm]	![mm]	![mm]		!	!				!		!													✓	
Avoid public hazard due to leaf/fruit fall[6]	✓[nn]	!	!	!	!	!	!			*[oo]	!	!		!		*[oo]											
Avoid injury/damage due to branch/tree fall[6]	✓	✓	✓	✓	✓	!	!			✓	!	!	!	!		✓	!										

S2 ctn. Justifications used to support the necessary conditions identified in Table 2 of the main manuscript. ✓ = Condition is typically necessary for delivering intended benefit. * = Condition may be necessary in some contexts. ! = Potential conflict between condition and a particular benefit.

Notes

[1] May be some limited water stress in hot periods. [2] Tree pruned, leaf litter removed, pests controlled. [3] Dead wood retained, complimentary habitats protected. [4] Does not impede ground-level visibility. [5] Defined here as an optically opaque barrier. [6] Key drawbacks to avoid.

Justification

[a] Good tree health is not necessarily a requirement for ecological benefits, as dead wood can provide a variety of valuable habitats. [1,2].

[b] Human disturbance may be significant for some species [3-5].

[c] High levels of management can limit feeding opportunities for wildlife. e.g. Heavy pruning, pesticide use, removal of dead wood [2].

[d] Presence of some bat and bird species is negatively correlated with surrounding built density e.g. [6] .

[e] Tree species that have been present the longest in Britain tend to have high insect species richness [7]. Non-natives support few insect species.

[f] A vegetated area beneath the tree increases habitats for invertebrates.

[g] Shading of built and paved surfaces is important as they re-radiate solar radiation effectively. Avoiding planting in street canyons eliminates many shading opportunities.

[h] Only necessary when year-round cooling is required, rather than summer cooling alone.

[j] Windbreak effect greatest in low-density suburban-type areas. Trees are unlikely to provide significant windbreak in high density areas.

[k] High canopies may only provide a limited barrier effect.

[m] Blocking solar access will act to cool the building. Often trees are only used to block northerly winds which avoids this conflict [8]

[n] Pruning reduces canopy density, which would be expected to increase noise transmission.

[o] Soft ground surfaces have been shown to account for a significant portion of the sound attenuation by vegetation [9].

[p] Evergreen species will be effective all year-round [10].

[q] Pruning removes biomass, returning CO_2 to the atmosphere via decomposition or combustion. Maintenance can also have high carbon costs [11].

[r] Large-scale planting is required for a significant amount of CO_2 sequestration to occur and for broad savings to be accrued through summertime shade and wintertime insulation [11,12]

[s] Large trees will intercept substantially more rainfall and transpire more, thus being more effective [13].

[t] In-leaf trees are more effective due to interception of rainfall – consider seasonality of peak rainfall events.

[u] Air temperature reductions likely to be of most value during high temperature episodes when water supply is most likely to be limited.

[w] Water limitations will not affect particulate deposition but will reduce stomatal uptake of NO_2 and O_3. Thus effectiveness may be reduced under warm anticyclonic conditions which often exhibit low rainfall and high pollution episodes, or when supplementary watering ceases.

[x] Generally large-scale planting is necessary, but trees in street canyons may be an exception [14].

[y] Trees in street canyons may increase exposure to pollutants through reducing ventilation, when emissions are high enough to overwhelm the pollutant capture effect of the tree [14]. The level of emissions varies according to situation (*ibid.*). These impacts can be reduced via high levels of pruning [15]

[z] Mature trees highly valued [16,17], but that does not mean immature trees will not provide any benefit.

[aa] Good visibility increases feelings of safety (Kuo et al., 1998; Kuo and Sullivan, 2001) – an important aspect of reducing psychological stress.

[bb] Leaf, branch and fruit detritus may impede movement and reduce positive feelings about trees and the local area.

[cc] Roads and paved areas outside buildings are precisely the areas where trees may have to be placed to break up a dense city-scape.

[dd] Owning property in a neighbourhood with trees may be desirable, due to the benefits enjoyed by residents, customers or staff. However, such trees may not be welcomed by all [18]. Public access to these trees may cause problems for local property owners in relation to increased social use of the space and risk of litigation [19].

[ee] Tall trees may generate conflict with CCTV security cameras [20].

[ff] Reductions in visibility are popularly associated with an increased risk of crime, although research doesn't always support this [21,22].

[gg] Re-development and increased use of an area would likely be associated with high noise levels.

[hh] Root systems provide support to soil structure [23], although some stabilising function may still be preserved after the tree has died [24].

[jj] Public access, made more desirable by trees, might damage surface vegetation and encourage erosion of embankment.

[kk] Sufficient water supply may prevent large root expansion in search for water and may reduce the risk of shrink-swell damage to buildings and other structures for clay-based soils. See www.bgs.ac.uk/products/geosure/shrink_swell.html

[mm] The chances of root expansion may be higher for a healthy tree.

[nn] Drought may trigger early leaf and fruit fall as well as death of branches.

[oo] Tree litter over a vegetated surface is less likely to be a slip hazard for pedestrians. For litter falling on paved surfaces maintenance requirements are higher.

S2 References

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S3. A scenario-based analysis of the vulnerability of the conditions required for an urban street tree to deliver its intended benefits.

Details of the vulnerability of each condition within each scenario have been deduced, based upon the relevant scenario characteristics [in brackets]. Where a cell is shaded green, the condition is considered to be broadly supported within the scenario. Yellow shading indicates that it may be supported in some circumstances but not others, whilst red indicates that this condition is unlikely to be supported. A full set of scenario characteristics can be found at www.designingresilientcities.co.uk/downloads/Indicators-2.xls.zip

Necessary conditions	**Vulnerability of necessary conditions within each scenario**			
	Policy Reform	**Market Forces**	**Fortress World**	**New Sustainability Paradigm**
Species is native	Policy is likely to require the use of native replacement trees on publicly owned land, but the legacy from decades of mixed planting may mean that non-native street trees are still abundant within this scenario. Tensions may occur where non-native species are better able to deliver certain social benefits *[tree species; urban tree/hedge cover and arrangement, degree of policy protection for ecological*	Street tree species may be changed depending on cultural and architectural fashion. In any case, but the legacy from decades of mixed planting means that non-native street trees are still abundant. *[tree species, attitudes to consumerism]*.	Species suitable for coppicing are likely to be prioritised in urban woodlands and other treed areas with restricted public access. Practical concerns trump native species selection in poor areas, and aesthetic concerns predominate in rich areas *[tree species, attitudes to consumerism]*.	Native trees are particularly valued, yet historic planting is likely to be retained and supplemented, rather than replaced *[tree species]*.

	features].			
Species is low VOC emitter	Policy is likely to require the use of low VOC replacement species on public land, but has limited influence on the nature of tree cover on private land *[tree species; urban tree/hedge cover and arrangement]*.	Urban street tree species may be changed depending on cultural and architectural fashions. Risk of high VOC emitters being selected *[tree species]*.	Species suitable for coppicing are encouraged in urban woodlands and other treed areas. Low VOC emission species are a not a priority in rich areas and not a consideration in poor ones *[tree species]*.	Low VOC species are valued, but historic street tree planting is likely to be retained and supplemented, rather than replaced *[tree species]*.
Species is evergreen	Policy is likely to require the use of best practice (appropriate replacement species and planting methods) for street trees on public land, but replacements on private land may be less optimal. A mix of tree species is encouraged, with social benefits a high priority. Conifers are valued for year round aesthetics and their shelter effect when planted on the windward side of buildings However, winter shade may be a problem in high density areas and evergreens may therefore not be used *[tree species; planning policy; planning adherence]*.	Urban tree species may be changed depending on cultural and architectural fashion. Street trees may therefore be periodically replaced. No particular priority given to evergreen trees *[tree species]*.	Evergreen conifers are fast growing and may be preferred as a secure timber supply, but no clear preference for street tree type *[tree species]*.	Native tree species and mixed planting are valued, with evergreen species retained and planted in specific locations for winter shelter. Planting evergreen species is avoided in street canyons with high levels of particulates. However, much historic planting is retained rather than replaced *[tree species]*.

A tree is still present	The multiple potential benefits of trees are recognized by policy. However, tensions arise with conventional social values and development practice, where space is needed for other sustainability solutions such as built density gradients surrounding transport hubs, new public transport infrastructure or affordable housing. Lost trees are generally replaced and attempts are made to mitigate climate change related stresses. However, policy reach regarding trees on private land is minimal and ability to restrict tree imports (and associated pests) is limited *[total amount of green space; degree of policy protection for ecological features, attitudes to consumerism, civic activism]*.	Although tree cover at the city scale is likely to be broadly retained, street trees are particularly vulnerable to removal where this reduces the risk of: damage to surface and buried infrastructure, nuisance, maintenance costs, or litigation risk and where development pressures are high (e.g. from road widening). The replacement of lost trees is unlikely *[quality of strategic planning for biodiversity conservation; urban tree/hedge cover and arrangement; urban tree/hedge cover and arrangement; km of road networks, attitudes to consumerism].*	Trees are vulnerable in the majority of the urban area, felled for fuel and timber in areas of extreme poverty and not replaced. Climate change related impacts would be broadly evident, as the high density urban form increases the risk of water and heat stress. However, some trees are valued, retained and protected in areas controlled by the rich, and risks from pests and diseases are moderate due to a reduction in international trade and focus on resource security *[urban tree/hedge cover and arrangement, attitudes to consumerism].*	Retention or replacement is likely in most locations as there is a greater awareness and value placed on urban trees and a willingness to accept some negative impacts that are difficult to avoid. A focus on locally grown planting stock and on design solutions to minimise heat and water stress result in limited impacts from climate change and pests/diseases *[total amount of green space; degree of policy protection for ecological features, attitudes to consumerism]*
Lateral root spread is not excessive	Policy mandates the use of best practice (appropriate replacement	Most planting makes no provision to limit root growth, with trees simply removed if impacts	Root growth type is not a consideration, unless trees are adjacent to	Trees are valued, and historic planting is likely to be retained and

	species, planting methods and mitigation) for street trees on public land, but replacements and retrofit on private land may be less optimal. However, there is a strong legacy of street trees with roots that damage built infrastructure *[tree species; planting policy; planning adherence]*.	occur or are deemed to be high risk. As most species exhibit lateral shallow root growth, impacts are likely *[planning policy; planning adherence]*.	critical infrastructure. Water stress in poor areas is likely to encourage lateral root spread *[tree species; planning policy; planning adherence]*.	supplemented, rather than replaced. Soil cells and root barriers are used for new planting in high risk areas or are retrofitted *[tree species; planning policy]*.
Tree is connected to a broader tree network	Some strategic planting for social and ecological benefits takes place – improving functional connectivity, shading etc. However, the arrangement of trees may change locally, to accommodate shifts in the density of the built form. Coordination between planting on public and private land is poor *[urban tree/hedge cover and arrangement; urban dwelling density]*.	No particular spatial arrangement for tree planting is pursued, although there is a reduction in trees adjacent to built or buried infrastructure to reduce potential for conflict *[total amount of green space; urban tree/hedge cover and arrangement]*.	No particular spatial arrangement is pursued. Losses of tree in poor areas are widespread, as trees are felled for timber or fuel *[total amount of green space; urban tree/hedge cover and arrangement]*.	Spatial arrangements for delivering strategic social and ecological benefits - (connectivity, shading etc.) are protected and implemented. However, local arrangements of trees may be changed to accommodate significant shifts in the density of the built form *[urban tree/hedge cover and arrangement; urban dwelling density]*.
Trees are maintained for wildlife	Pesticide treatment of trees on public land is generally restricted, with active	Street trees are heavily managed for amenity, reducing the abundance of insects, fruits and microhabitats. Little	Maintenance for biodiversity only occurs if the tree is located in areas controlled by the rich,	Planning policies prohibit aggressive management practices that limit flowering,

	intervention to support the delivery of ecological goals. However, few controls are imposed on tree maintenance within private land *[urban tree/hedge cover and arrangement; degree of maintenance for ecological features; attitudes to consumerism]*.	management takes place to support wildlife *[management of public realm/open spaces; cultural and historical associations; attitudes to consumerism; degree of maintenance for ecological features]*.	although it is unlikely to be a priority *[management of public realm/open spaces; cultural and historical associations; attitudes to consumerism; degree of maintenance for ecological features]*.	fruiting or insect productivity. High levels of public volunteering takes place *[degree of maintenance for ecological features; degree of policy protection for ecological features]*.
Tree is not in a street canyon with a busy road	Street canyons are common, due to a strong policy push for higher built densities. However, road traffic and pollution in urban centres are much reduced *[urban dwelling density; settlement pattern; passenger road travel]*.	Street canyons are not ubiquitous but still common, and increases are seen in vehicle numbers *[passenger road travel]*.	An overall increase in busy road canyons due to high built densities in poor areas. Traffic in poor areas remains significant, although less abundant than the present *[passenger road travel; settlement pattern]*.	Reduced vehicle usage compared to present and vehicular usage kept at the border of neighbourhoods where possible *[passenger road travel; road and parking characteristics]*.
Trees are maintained for amenity	The removal of fruits and branches to improve public safety and amenity is permitted. Fallen leaves are generally cleared from pavements. However maintenance is balanced with the need to meet ecological goals and limited to areas where	Although maintenance budgets are down overall, remaining trees are likely to be heavily managed for amenity, reducing the abundance of insects, fruits and microhabitats. Strong canopy and root control is undertaken for visual amenity and to reduce pavement damage. Management is less	Maintenance for amenity only occurs if the tree is located in areas controlled by the rich, where the situation is as for market forces *[management of public realm/open spaces; cultural and historical associations; attitudes to consumerism;*	Unlikely. The public are willing to accept urban trees in a more natural state - untidy/dense canopy, fallen fruit and leaves *[degree of maintenance for ecological features; degree of policy protection for ecological features]*.

	negative social impacts are clear *[degree of maintenance for ecological features; attitudes to consumerism]*.	intensive in some poorer areas *[management of public realm/open spaces; cultural and historical associations; attitudes to consumerism; degree of maintenance for ecological features]*.	*degree of maintenance for ecological features]*.	
Consistent water supply for healthy growth	Variable. Policies to support trees for their environmental/social benefits generally succeed in protecting the infiltration of surface waters surrounding trees, with some supplementary watering undertaken for young or highly stressed trees. However, low soil moisture is still an issue in areas where built density has been increased and in locations where mains leakage has been reduced *[water efficiency and recycling measures]*.	Varies with land-use and social context. Permeable paving, the protection of soil cells and supplementary watering is not considered a priority, yet likely in some wealthier areas. However, this may be balanced by a relatively low built density within this scenario and by losses from poorly maintained drainage and water distribution infrastructure *[asset condition; water distribution system pattern at the city scale; impervious/pervious surfaces]*.	No maintenance budgets for trees in poor areas, combined with high levels of impervious surfaces and high built densities result in broad water stress. However, ageing infrastructure in poor areas may result in some gain in groundwater recharge and soil moisture from leaking mains water supplies and drainage systems. In the rich areas, some water will still be allocated for irrigation due to the aesthetic value of urban trees *[asset condition; impervious/pervious surfaces; degree of policy protection for ecological features]*.	Retaining access to water for existing street trees is included successfully as a design criteria for redevelopment. Soil cells are protected and incorporated into SUDS whenever built density is increased locally *[quality of strategic planning for biodiversity conservation; degree of policy protection for ecological features]*.
Root growth not	Often this is the case. However,	Not a planning priority, and root	Not a planning priority, and root	The potential impact of

substantially impeded	high-density development and strong market forces occasionally result in a lack of space for roots, limiting tree growth. There is also a legacy of urban tree planting in insufficient soil volumes *[planning policy; urban dwelling density; land recycling]*.	space may not be intentionally preserved. But this risk is limited as cities tend to expand outwards rather than infilling. Problems arise where land values encourage high density development. High levels of soil compaction may be a significant issue. *[land use; land recycling]*.	space may not be intentionally preserved. But risk is limited as cities tend to expand outwards rather than infilling. Informal developments unlikely to significantly modify available sub-surface space *[land use]*.	increased built density in some areas is mitigated through careful design and retrofit *[quality of strategic planning for biodiversity conservation; degree of policy protection for ecological features]*.
Tree's access to light maintained	Generally this is the case. However, high-density development and strong market forces occasionally result in vegetation losing optimal solar access *[planning policy; urban dwelling density; land recycling]*.	Not a planning priority, but generally a low risk due to a tendency for urban sprawl. Problems arise where land values encourage high density development *[land use; land recycling]*.	Not a planning priority. A low risk in areas controlled by the rich due to lower population densities, but vulnerable in poor areas *[land use]*.	Retaining access to light for trees is included successfully as a design criterion for redevelopment, particularly in high density areas *[degree of policy protection for ecological features; planning policy]*.
Tree is large or mature	A policy of protection and maintenance increases the likelihood of tree survival to maturity, although some losses would be expected due to conventional development pressures and the pursuit of high-density development. Tree canopy may	The low development density of urban form gives a good chance for some trees to grow to maturity. But this is offset by a lack of policy protection and increased pollution stresses. Positive management varies spatially (reflecting income). *[urban dwelling density; urban water*	Large, mature trees are valued in rich enclaves, where they have the space and resources to grow. But trees are unlikely to reach maturity in the majority of the city where environmental stresses are compounded by felling and coppicing *[degree of*	Tree health and survival is good in this scenario. Trees are strongly protected in policy and valued by the general population *[degree of maintenance for ecological features; degree of policy protection for ecological*

	still be heavily pruned to reduce trapping of particulates within street canyons. *[degree of policy protection for ecological features, urban dwelling density; settlement pattern]*.	*pollution levels; degree of policy protection for ecological features; degree of maintenance for ecological features]*.	*maintenance for ecological features, urban tree/hedge cover and arrangement]*.	*features]*.
High canopy	A policy of high protection and maintenance increases the likelihood of tree survival to maturity, yet removal and pollarding are common to reduce conflicts with infrastructure and to cope with the impacts of increased built density *[degree of policy protection for ecological features]*.	Low density of urban form gives a good chance for tree to grow to maturity. But this is offset by lack of policy on protection and increased pollution stresses. Compaction of soils may stunt tree growth and positive management varies spatially (reflecting income). *[urban dwelling density; urban water pollution levels; degree of policy protection for ecological features; degree of maintenance for ecological features]*	Large, mature trees are valued in rich enclaves, where they have the space and resources to grow. But trees unlikely to reach maturity in poor areas where environmental stresses are compounded by felling and coppicing *[degree of maintenance for ecological features, urban tree/hedge cover and arrangement]*.	Likely, as tree health is good in this scenario, therefore early death is unlikely. Historic planting is likely to be retained and supplemented, rather than replaced *[degree of maintenance for ecological features; degree of policy protection for ecological features]*.
Tree forms part of densely-vegetated barrier	Environmental policy and enforcement is strong, but contiguity may be counteracted by pressure for high-density development and concerns over heavy shading *[degree of policy protection for ecological*	Vegetated areas highly vulnerable to redevelopment if the market conditions are right, yet sprawl is the dominant development pattern *[degree of policy protection for ecological features]*.	Likely to be retained in rich areas where neighbourhood quality is considered important, but vulnerable to cutting for fuel or informal development in poor areas *[degree of policy protection for*	Wooded areas valued by the community and likely to be retained, especially if mature *[degree of policy protection for ecological features]*.

	features; dwelling density].		*ecological features]*.	
No persistent noise	Spatially variable, but reduced compared to present due to reduced private vehicle usage *[passenger road travel]*.	Spatially variable. Noisy in city centres and near major highways, but generally quiet in sprawling residential areas *[road and parking characteristics; passenger road travel; settlement pattern]*.	Spatially variable, with an increase around busy roads in poor areas. But overall decrease in poor areas due to reduced traffic, and the speed at which that traffic can travel. Ssimilar to present day in rich areas (more use, but spread over larger area) *[settlement pattern; passenger road travel]*.	Much less noise due to reduced private vehicle usage *[passenger road travel]*.
No artificial lighting	Spatially variable - no major changes compared to present. Focus is on lighting to improve safety and perception of safety - pavements, roads, road crossings, residential areas *[artificial external lighting quality; area of city that is artificially lit]*.	Spatially variable. Public street lighting provision is reduced in less affluent areas. The intensity and extent of lighting in private/more affluent areas is significantly higher for aesthetic and security reasons *[artificial external lighting quality; area of city that is artificially lit]*.	Spatially variable, with an overall reduction in lighting intensity and extent. Public street lighting provision for the poor is virtually non existent. The intensity and extent of lighting of areas controlled by the rich is significantly higher for security and to increase the perception of safety *[artificial external lighting quality; area of city that is artificially lit]*.	Lighting extent and intensity is much less than present, as shifts in social values make it easier to remove street lamps to reduce carbon and ecological impacts to *[artificial external lighting quality; area of city that is artificially lit]*.
Tree blocks solar access to building	High energy efficiency is widespread and mandated in	Solar gain not widely utilised for heating, with conventional	Solar gain not widely utilised for heating, with conventional	High energy efficiency is widespread, including

	policy, including recommendations for passive heating methods. In practice passive solar principles are not always used due to the constraints placed on design, and trees may be allowed to block solar access *[energy efficiency of building and urban morphology]*.	methods used instead. Therefore no pressure or policy from an energy perspective to avoid trees blocking solar access. However, loss of natural daylight is generally not tolerated and trees are often removed. *[energy efficiency of building and urban morphology]*.	methods used instead. Therefore no social pressure or policy to avoid trees blocking solar access in rich areas. The poor have little influence in policy but localised social pressures may force removal of trees blocking desired solar access as heating is expensive *[energy efficiency of building and urban morphology]*.	passive heating methods. Trees used to block solar access in summer, but solar access will not be impeded at those times of year when it is most needed, e.g. by using deciduous trees to allow solar access in winter *[energy efficiency of building and urban morphology]*.
Surrounding area built to high density	Increased density of the built form is pursued, with some negative implications for existing tree cover and its accessibility *[urban dwelling density; settlement pattern]*.	Spatially variable, depending on land values *[urban dwelling density; settlement pattern]*.	Spatially variable, with low densities in areas controlled by the elite and a high built density for the poor majority *[urban dwelling density; settlement pattern]*.	Spatially variable, increased density in some areas permits a reduction in total built surface *[urban dwelling density; settlement pattern]*.
Tree does not overhang a road or pavement	Some strategic planting for social and ecological benefits takes place - connectivity, shading etc. However, arrangements of trees may change locally, to accommodate changes in the density of the built form. Trees may be removed	A general reduction in trees adjacent to paved areas and roads to remove risks from litigation and also to reduce damage to adjacent built or buried infrastructure *[urban tree/hedge cover and arrangement]*.	No particular spatial arrangement is pursued, although an overall reduction in tree cover inevitably reduces tree associated with roads and pavements*[urban tree/hedge cover and arrangement]*.	Spatial arrangements for delivering strategic social and ecological benefits - (shading, connectivity, etc) are generally protected. An, increase in tree cover for wildlife and recreation leads to greater lining of roads

	where conflicts with social goals are inevitable (e.g. slipping hazard on pavement) *[urban tree/hedge cover and arrangement]*.			with trees (although overall numbers of roads are reduced). Social attitudes are more tolerant. *[urban tree/hedge cover and arrangement; road and parking characteristics]*.
Low stress from air pollution	Air pollution is generally low *[Particulate matter, NO_2, ozone].*	Spatially variable - generally higher than present and may be excessive in poor areas. May result in chronic stress, reducing tree growth, vitality and lifespan in most polluted areas, but acute stress leading to rapid deterioration of tree function unlikely *[Particulate matter, NO_2, ozone]*.	Spatially variable - generally higher than present and may be very high in poor areas where vehicles are poorly maintained. Chronic stresses reduce tree growth, vitality and lifespan in most polluted areas and periods of very high pollution resulting in acute stress leading to rapid deterioration of tree function may occur periodically *[Particulate matter, NO_2, ozone]*.	Air pollution is generally low *[Particulate matter, NO_2, ozone].*
Low stress from soil pollution	Soil pollution is generally low *[urban water pollution levels]*.	Spatially variable, depending mainly on land values *[urban water pollution levels; planning policy]*.	Soil pollution may be moderate in parts of poor areas, where regulation and enforcement is lacking. Low soil pollution in rich areas *[planning adherence; urban water pollution levels]*.	Soil pollution is generally low *[urban water pollution levels]*.

Tree is physically accessible to public	Street trees are generally accessible to the public, but not always. Policy prioritises protection of existing tree cover over protecting public access. Private land remains largely inaccessible *[urban dwelling density; settlement pattern; accessibility of public realm/open space]*.	Spatially variable, as access to private streets, parks and developments is increasingly restricted. Passive restriction occurs in public parks where path maintenance is much reduced *[management of public realm/open spaces; provision of public realm/open spaces; public land ownership]*.	Spatially variable, but poor overall. If trees are located in areas managed by the rich, the rich will have access (although the poor majority will be excluded). The few remaining street trees in poor areas are likely to be accessible. *[accessibility of public realm/open spaces; quality of public realm/open spaces]*.	Retaining public access to street trees is included successfully as a planning condition for redevelopment, particularly in high density areas *[accessibility of public realm/open spaces; total amount of greenspace]*.
Tree is growing in a pervious surface	No major changes to present day. Trees in green spaces have good access to soil and natural watering. Paving below street trees is removed where possible, as long as trip hazard to pedestrians is unlikely. Policy recognises the value of street trees as a means of mitigating surface water *[degree of maintenance for ecological features; degree of policy protection for ecological features]*.	Variable. In areas of high land value, gentrification results in the paving of surfaces below trees. Watering systems and root barriers are installed if required, with surface water being used for tree watering in areas of high flooding risk. Such retrofit technologies are not applied in less affluent areas *[degree of maintenance for ecological features; provision of public realm/open spaces]*.	Spatially variable. Rich areas as for market forces. In poor areas, the area of pervious surface will increase as informal developments are likely on compacted soil and therefore pervious, whilst a poor maintenance of existing infrastructure will increase the perviousness of the surface. In these areas, the surface will be vegetated where it is not in heavy use *[asset condition]*.	Street trees are generally located within unpaved areas. Trees are valued, given space to grow and recognised as a means of mitigating excess surface water *[degree of policy protection for ecological features; road and parking characteristics]*.
Tree is	Visual access is	Spatially variable.	Spatially variable.	Retaining public

visually accessible to public	generally good. Policy supports redevelopment that retains/enhances local identity. However, the form of high-density development may still reduce visual access, as other social goals are prioritised *[urban dwelling density; settlement pattern; accessibility of public realm/open spaces]*.	The business case for visual access to urban trees is rarely made. However, as urban expansion rather than densification is preferred, views of street trees are rarely obscured *[urban dwelling density; settlement pattern]*.	If trees are located in areas managed by the rich, the majority of the public (poor) will have limited visual access (although the rich will have excellent access) *[accessibility of public realm/open spaces; quality of public realm/open spaces]*.	visual access to street trees is included successfully as a design criteria for redevelopment, particularly in high density areas *[accessibility of public realm/open spaces; total amount of greenspace]*.
People are present nearby	A higher built density increases the probability of people being in close proximity to trees *[settlement pattern; urban dwelling density; land recycling]*.	Population density decreases. In some cases the market might result in a depopulation of an area or major change in land use *[urban population density; land recycling; planning policy]*.	Population density increases in most (poor) parts of the city. *[urban dwelling density;]*.	Spatially variable, as some areas increase significantly in built density whilst others are converted to semi-natural open space *[urban dwelling density; settlement pattern; road and parking characteristics]*.
Large-scale tree-cover across urban area	No major changes in tree cover at the city scale, but numbers of trees may change locally, to accommodate significant changes in the density of the built form *[total amount of green*	Little change in overall tree cover at the city scale, however this masks a reduction in planned tree coverage in streets/parks/gardens, and an increase in scrub woodland on abandoned brownfield sites *[total amount of*	Decrease at a city scale, as trees outside the enclaves are quickly cut for fuel as soon as they are large enough. Tree cover in areas controlled by the rich is similar to present *[total amount of green*	Increase in total tree cover at the city scale. However, it is possible that numbers of trees may change locally, to accommodate significant changes in the density of the

	space].	*green space; urban tree/hedge cover and arrangement].*	*space; urban tree/hedge cover and arrangement].*	built form *[total amount of green space].*