A Multi-Scale Exploration into the Spatial Patterns of a Three Dimensional Urban Tree Infrastructure (UTI): integrating landscape connectivity, network resilience, and social deprivation

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Abstract

A functionally connected urban tree infrastructure (UTI) contributes to ecosystem function, resilience, and the provision of Ecosystem Services (ES). Variation in tree height is an important attribute influencing movement of passerines, habitat quality and landscape patterns. UTI provided ES are particularly beneficial in the most deprived areas of a city. Presented in this thesis is an exploration into the social-ecological shape of a UTI using a holistic, multi-scale and dimensional, landscape approach.

The potential landscape connectivity of a UTI in the City of Salford, UK was quantified and compared using the integral index of connectivity (IIC) across vertically stratified canopies existing in 2005, 2009, and 2013. System resilience was assessed through landscape graph network analysis and by the identification of canopies critical in maintaining connectivity (dIIC). The index of multiple deprivation (IMD) was related to UTI landscape composition and configuration through a series of statistical tests.

The connectivity of Salford's vertically stratified UTI was low (IIC = <0.000001 – 0.0045), besides this the temporal change in connectivity was complex with no discernible overall pattern. The rate of connectivity increase decreased after a 90-120m gap-crossing threshold. The resilience of Salford's UTI relies on the connectivity of canopies within 4 to 5 sub-connected regions, depending on passerine perception, increasing to 10 -16 smaller regions for canopies above 17.1m. The resilience and stability of these sub-connected regions were often reliant on a central canopy patch. UTI composition is related negatively with deprivation, UTI configuration is related positively, while structural diversity of canopy heights revealed no correlation with deprivation.

The research in this thesis contributes to the debates on how to best manage the UTI for both people and nature. The findings of this thesis have a number of important implications for future urban landscape management, especially as previously unknown landscape patterns have been identified.

Chapter 1 : Introduction

The urban tree infrastructure (UTI) is a major component of the infrastructure needed to make urban environments better places to work and live and is recognised as being part of the solution to modern day challenges such as climate change, declining human health and well-being, and the need to promote and establish sustainable communities (Forestry Commission, 2012; TDAG, 2012). Furthermore, the composition, patterns of configuration, and degree of connectivity of the UTI at the landscape scale are principal factors impacting the ability of urban birds to successfully move through urban environments (Fernández-Juricic, 2001a; Tremblay & St. Clair, 2011). Landscape connectivity is defined as the degree to which the landscape facilitates or impedes movement among resource patches (Talyor *et al.* 1993) and a loss in connectivity is seen as a major threat for both biodiversity conservation and the maintenance of ecological functions operating across landscapes (Pascual-Hortal & Saura, 2006). Movement is a behavioural mechanism of critical ecological and evolutionary importance (Paradis *et al.*, 1998; Tremblay & St. Clair, 2009) and many important ecosystem services are provided by the foraging behaviour of moving organisms, especially birds (Kremen *et al.,* 2007; Wenny *et al.*, 2011). Through foraging, birds transfer energy both within and between ecosystems, contributing to ecosystem function and resilience (Lundberg & Moberg, 2003). Many birds, particularly passerines from the order Passeriformes (known as perching birds or songbirds), have a strong relationship with trees and hence the UTI plays a fundamental role in both the facilitation of urban bird movement and the provision of ecosystem services in urban areas (Fernández-Juricic, 2001a; Tremblay & St. Clair, 2011; TDAG, 2012).

Urbanisation causes the permanent alteration and fragmentation of natural habitats, often negatively affecting bird movement capabilities by influencing the size, shape, composition, and connectivity of natural habitat patches (Alberti, 2005; Tremblay & St. Clair, 2009). In addition to the importance of the landscape structure of the UTI is the intra-patch structure of its constituent canopy patches (Spies, 1998). Such structure can differ significantly from patch to patch and the spatial arrangement of habitat elements within canopy patches have a strong influential effect on bird foraging behaviour and habitat suitability (Dolman *et al.*, 2007a). Specifically, variation in tree height is an important structural attribute of canopy patches; those containing a high degree of height diversity are likely to contain a variety of tree ages and species, and thus provide a diversity of micro-habitats especially for species which are sensitive to habitat fragmentation (Goldstien, 1986; McElhinny *et al.*, 2005). Despite this, landscape connectivity studies disregard landscape patterns occurring in the third dimension (height), often considering habitat patches as homogenous two-dimensional elements in the landscape (e.g. Metzger & Decamps, 1997; Tischendorf & Fahrig, 2000; Pascual-Hortal & Saura, 2006; Uuemaa *et al*., 2009; Foltête *et al.,* 2014). By overlooking the importance of vertical structure in connectivity models (especially for canopy patches) landscape patterns are overlooked while potentially critical landscape components, for maintaining connectivity and overall system resilience, remain unidentified.

Identifying the core structure of the UTI, made up of canopy patches important for maintaining connectivity, will reveal insights into the resilience of the UTI (Janssen *et al.,* 2006; Minor & Urban, 2008). By using a resilience approach (Holling, 1973; Folke, 2006; Garmestani *et al.*, 2009) to investigate and understand social-ecological systems, the goals of sustainable development and the continual provision of ecosystem services become more attainable (Walker *et al.*, 2004; Ernstson *et al.*, 2010; Biggs *et al.*, 2012). Ecological resilience can be defined as the capacity of a system to absorb disturbances, re-organise itself while undergoing change and still maintain a similar function and structure (Holling, 1996; Folke, 2006). It is also an important concept which is increasingly used to underpin the understanding, managing, and governing of linked social-ecological systems (Folke *et al.*, 2004). Therefore, resilience can also be thought of as an approach, a way of thinking, that can guide and organise how we perceive systems and, more specifically, provide a valuable context for analysing the current state of social-ecological systems (Folke *et al.*, 2002; Folke, 2006). An understanding of the state of a system can be derived from the values of the variables that constitute the system (Walker *et al.*, 2004). In relation to the resilience of social ecological systems, it is landscape structure and the structural change exhibited within a system which can provide insights into function (Janssen *et al.*, 2006). More specifically, it has been shown that *connectivity* and *centrality* are comprehensive variables of landscape structure that can be used to assess the function and resilience of a system when described as a network (Albert *et al.*, 2000; Janssen *et al.*, 2006; Rhodes *et al.*, 2006; Minor & Urban, 2008; Opsahl *et al.*, 2010; Biggs *et al.*, 2012). Ecologically relevant data must be incorporated into any model which attempts to effectively measure landscape connectivity and centrality, the values of which will change with spatial and dimensional scale.

Any analyses of UTI heterogeneity for the connectivity of organisms, especially birds, need to consider the pattern of structural elements over various heights. As urban passerines are important ecosystem service providers and play a major part in the underpinning of the stability and resilience of ecological systems, they are ideal organisms to integrate into potential landscape connectivity models. Furthermore, to be ecologically relevant, a description, or rather a component of organism perception, needs to be built into any model. In this thesis the ability of passerine birds to cross gaps, of varying distances, in the canopy is considered a component of perception, if only a single component. Therefore, the incorporation of varying spatial scales and gap-crossing capability thresholds into landscape connectivity models in order to understand the levels of connectivity exhibited by an UTI, over time, is one of the main aims and major novelties of this thesis. The models employed through the thesis research approach should also facilitate the description of UTI resilience to change; especially important in an urban landscape which is constantly in a state of development.

In addition to ecological fragmentation, urban areas also present a societal disconnect with nature. Since 2008 there are more people, globally, that live in urban areas than in rural areas, with 54% of the world's population residing in urban areas in 2014 (United Nations, 2008, 2012, 2014). Within the UK at least 80% of the population is now urban (Pateman, 2011). Urbanite lifestyles are being satisfied at the expense of land use, climate, biogeochemical, and species distribution change (MA, 2005). Biodiversity is declining globally, raising concerns about the consequences of such losses on ecosystem structure and function, the provision of ecosystem services, and human well-being (Balvanera *et al.*, 2006). Many ecosystem services and goods must be imported into cities from the urban hinterlands and global commons, however having access to nature within urban environments is an important criteria for quality of life and human well-being (O'Brien, 2005; Forestry Commission, 2012). Therefore, the ability to benefit from the cultural services provided by natural environments within urban areas may represent the closest relationship between people and ecosystem services. Research suggests that those living within the most deprived urban areas benefit the most from green space access and the presence of urban trees (Gilchrist, 2011). Despite this, a social – ecological disconnect often exists in urban areas as residents of higher socio-economic status use and live within areas richer in green infrastructure than those inhabiting more deprived areas (Matsuoka & Kaplan, 2008). However, it is not readily known whether the horizontal and vertical landscape composition and configuration of green infrastructure vary across areas with differing levels of deprivation.

The most important canopy patches, for maintaining levels of UTI landscape connectivity across differing vertical planes, are to be identified. How these canopies are composed and configured within the landscape and how they relate to neighbouring canopy patches will be evaluated through a landscape network approach and related to system resilience. Furthermore, this multi-scale, hierarchal approach to describing the UTI will hopefully reveal further insights into the relationship between area deprivation and green infrastructure; specifically, if presence also means structural quality, underpinning ecological function. Describing the current state and shape of the urban fabric and understanding how organisms respond to the permeability of the occurring spatial patterns is of critical importance for urban conservation strategies, urban greenspace design, urban ecosystem resilience, and human well-being.

In Chapter 2 the results of a critical literature review of the links between urban humans, urban biodiversity, and urban landscape patterns and processes are presented. A focus on urban ecology and ecosystem services can be found in Section 2.1. In Section 2.2 the concepts surrounding the urban tree infrastructure are discussed, followed by a critical appraisal of landscape structure and connectivity in Section 2.3. How landscape structure and connectivity is related to ecological system resilience is reviewed in Section 2.4. Finally, how structural patterns affect ecological processes such as bird movement and ecosystem service provision is reviewed in Section 2.5.

Within Chapter 3 the project aims and objectives, general methods, and study area are presented. In Section 3.1 the overall research aim, research questions, and the research objectives used to address the research questions are established. In Section 3.2 the City of Salford is introduced as the area of study, followed by a description of the general methods employed in the description and mapping of Salford's UTI (Sections 3.3 and 3.4).

The temporal changes in the vertical and horizontal connectivity of Salford's UTI are addressed in Chapter 4. The aim is to quantify the potential level of UTI connectivity within the Salford landscape and how this connectivity changes across spatial and temporal dimensions. By further incorporating passerine gap-crossing abilities into landscape connectivity models it is proposed that the results will represent functional as well as structural connectivity. The inclusion of canopy height and the subsequent vertical mapping of the UTI offer an innovative approach to landscape connectivity assessment.

Chapter 5 provides an investigation into the system resilience of Salford's UTI. The aim is to understand how resilient the UTI's provision of functionally connected habitat is to potential changes in the landscape by identifying key structural landscape components. As the UTI exists in three dimensions it is appropriate to assess the resilience of canopy networks across differing heights. Furthermore, passerine perception of UTI gaps can influence the values of key structural variables. However, the landscape connectivity literature reduces vegetation into two dimensions while network approaches to system resilience have, as of now, not included organism perception in the description of networks, thus presenting gaps in research. These gaps are filled by the research presented in Chapter 5, which investigates how the variables of landscape networks change when varying vertical dimensions and passerine perceptions are included in the analysis.

An exploration into the relationship between a societal metric and UTI landscape pattern is presented in Chapter 6. The aim of the research presented in this chapter is to expose any socio-ecological disconnects and potential environmental injustices within the City of Salford in relation to area deprivation and UTI quantity and quality. Area deprivation is described based on the UK Government's index of multiple deprivation (IMD). UTI quantity is described as the percentage of canopy cover while quality is specifically related to ecological quality as described by the UTI's vertical structural diversity and canopy fragmentation. It is well cited that more deprived areas have less green spaces and tree canopy cover, the research gap lies in whether this relationship extends to canopy cover configuration and structure in both two and three dimensions.

Finally, a general discussion on the research undertaken throughout the thesis is presented in Chapter 7. This is followed by more specific discussions on UTI mapping (Section 7.1), landscape connectivity and system resilience (Section 7.2), and the relationship between UTI and social structure (Section 7.3). Research limitations and future opportunities are discussed within each section of Chapter 7 while the concluding remarks are presented in section 7.4.

Chapter 2 : Literature Review

2.1 Urban Ecology and Ecosystem Services – people and nature

The urbanisation of natural environments is set to continue into the foreseeable future, a trend which profoundly affects the way we understand, connect with, and use nature and its resources (Niemelä *et al.*, 2011). Through the continual development of urbanisation the science of urban ecology has emerged, and ecologists have formed new ecological paradigms relevant to the urban environment. Ecologists are now moving away from placing 'pristine nature' on a pedestal - while vilifying the acts of humankind - to the understanding that it is the union of cultural and biological diversity which underpins, both social and ecological, resilience and sustainability (Berkes *et al.*, 2003; Niemelä *et al.*, 2011). This social-ecological relationship is most relevant where urbanisation has had the greatest effect – cities. Cities are associated with one species – humans – and can be characterised by the existence of large human populations in extraordinary densities (Rees, 2003). In addition, cities can be defined by the physical effect humans have on the design, creation, and control over their local environmental conditions as well as the subsequent patterns of landscape structure defined by a mosaic of land-use and land-cover types (Pickett *et al.*, 2001; Alberti *et al.*, 2003; Andersson, 2006). Cultural selective forces (i.e. societal decisions and human behaviour) are the direct drivers of land-use change within a city, influencing the ecological selective forces (e.g. hydrological cycles, nutrient cycles, and species interaction networks) and therefore influencing ecological patterns and processes (Grimm *et al.*, 2000; Pauleit & Breuste, 2011). A feedback mechanism is then instigated as the form of these ecological patterns and processes can influence those societal decisions which drive land-use change (Figure 2-1).

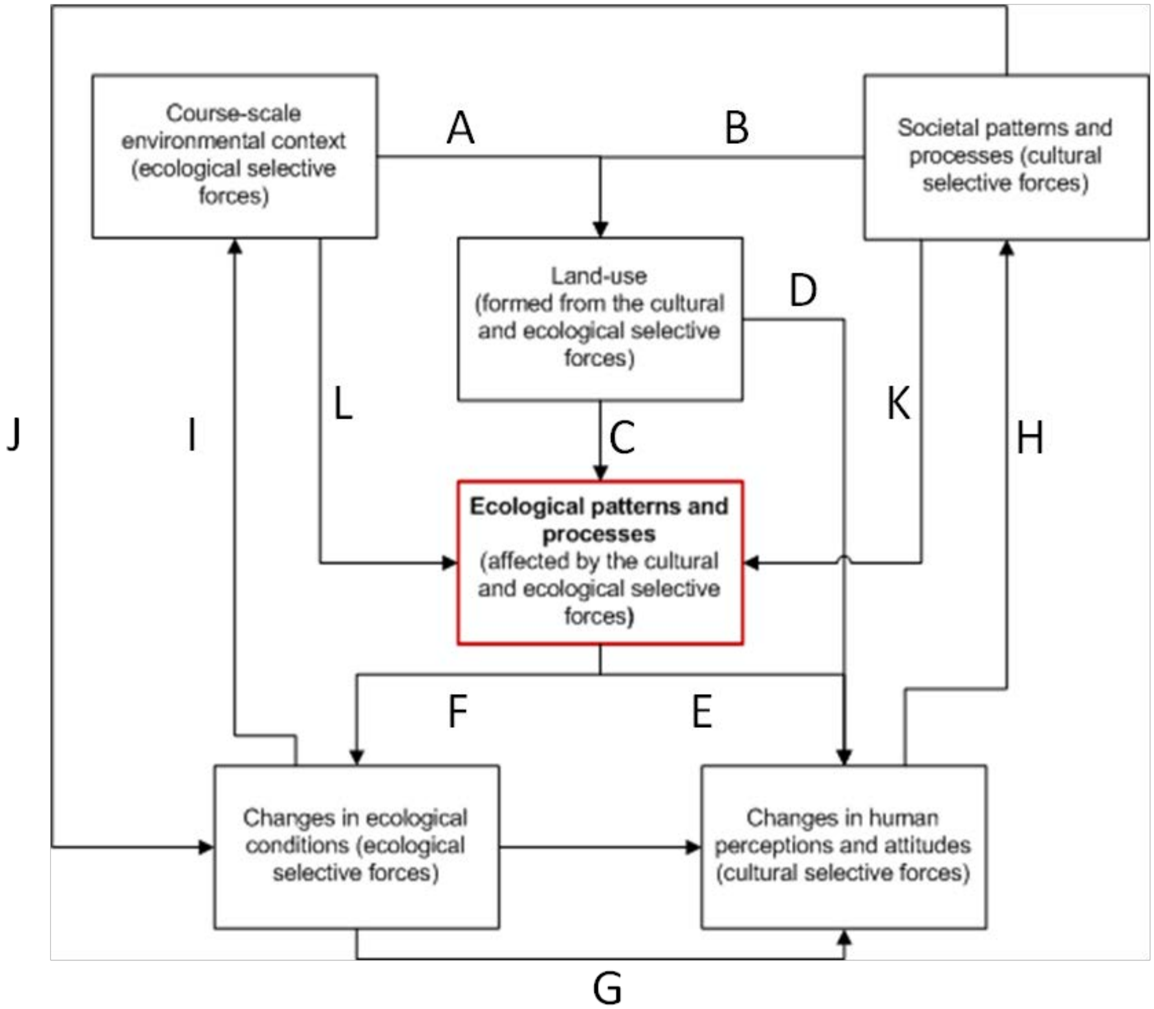

Figure 2-1: Conceptual scheme for integrating ecological and social systems in urban environments with emphasis on ecological patterns and processes (Grimm *et al.*, 2000).

Variables in boxes; interactions and feedbacks represented by arrows: **A**, environmental context sets the range of possibilities for land use and land-cover; **B**, societal decisions and human behaviour are the direct divers of land-use change; **C**, the pattern of land-use determines ecological patterns (of land-cover) and processes, it becomes the variable of most interest as it sits in the middle of the conceptual network; **D**, human perception of land-use changes (independent of any ecological understanding):**E**, humans also perceive and react to ecological patterns and processes, better understanding of these patterns will influence this perception; **F**, ecological patterns and processes, affected by land-use change, result in a change in ecological conditions; **G**, changes in ecological conditions may result in changes in attitude as these conditions are judged as good or bad by humans; **H**, changes in perception and attitude feed back into the societal patterns, influencing decision making and this part of the cycle begins a new; **I**, changed ecological conditions can alter the environmental context, resulting in feedback relatively independent of human response; **J**, society can act directly on the change in ecological conditions; **K**, society can act directly on the underlying ecological patterns and processes producing the change in ecological conditions; **L**, environmental context also directly influences ecological patterns independent of land-use and the cultural selective forces acting upon land-use. Note; land-use variable and the ecological patterns and processes variable are products of an amalgamation of the external cultural selective forces and the internal ecological selective forces maintained by society and nature respectively.

Such relationships between the physical, ecological land-cover complexities, and the social-economic and social-ecological land-use complexities of urban ecosystems are explored in the concept of 'ecosystem services'. The Millennium Ecosystem Assessment (MA) appraised ecosystems in terms of the services they provide to society, how humanity benefits from these services, and how the actions of humans alter these services and the ecosystems providing them (MA, 2005). By doing so, the MA 'introduced a new framework for analysing social-ecological systems that has had wide influence in the policy and scientific communities' (Carpenter *et al.*, 2009, pg. 1305). Fisher *et al* (2009), defined ecosystem services as the aspects of ecosystems which are utilized (actively or passively) by humans to provide well-being. Thus, services must be ecological and functions or processes can only become services when humans benefit from them; without beneficiaries there are no services (Fisher *et al.*, 2009). The authors also consider infrastructure as an important component of ecosystems as some configuration of structure and process is needed for *healthy* service provision (Fisher *et al.*, 2009). The United Kingdom National Ecosystem Assessment (UKNEA) (2011) also purposed that the interaction between the living and physical environments (ecosystems) deliver necessary services to humanity. This simple definition is expanded on by providing a breakdown of service definitions and examples (Figure 2-2). The links between human well-being and ecosystem services as depicted by the MA (2005) can be incorporated into the conceptual framework of the UKNEA, which depicts the influential role of social feedbacks and future UK scenarios, to create a cyclical process than constantly shapes social-ecological relationships (Figure 2-2). The MA (2005) highlighted that biodiversity underpins supporting services which in turn underpins the provisioning, regulating, and cultural services linked to the four constituents of well-being; security, basic material for good life, health, and good social relations. These four constituents then feed into freedom of choice and action, which is essential for well-being. In addition to ecosystem services, other factors influence human well-being, including cultural, economic, social, and technological factors. The social feedbacks depicted by the UKNEA, emphasise that societal drivers of change influence human action, future social, economic, and ecological UK scenarios, and finally ecosystem services; thus the cycle ends and begins again (Figure 2-2).

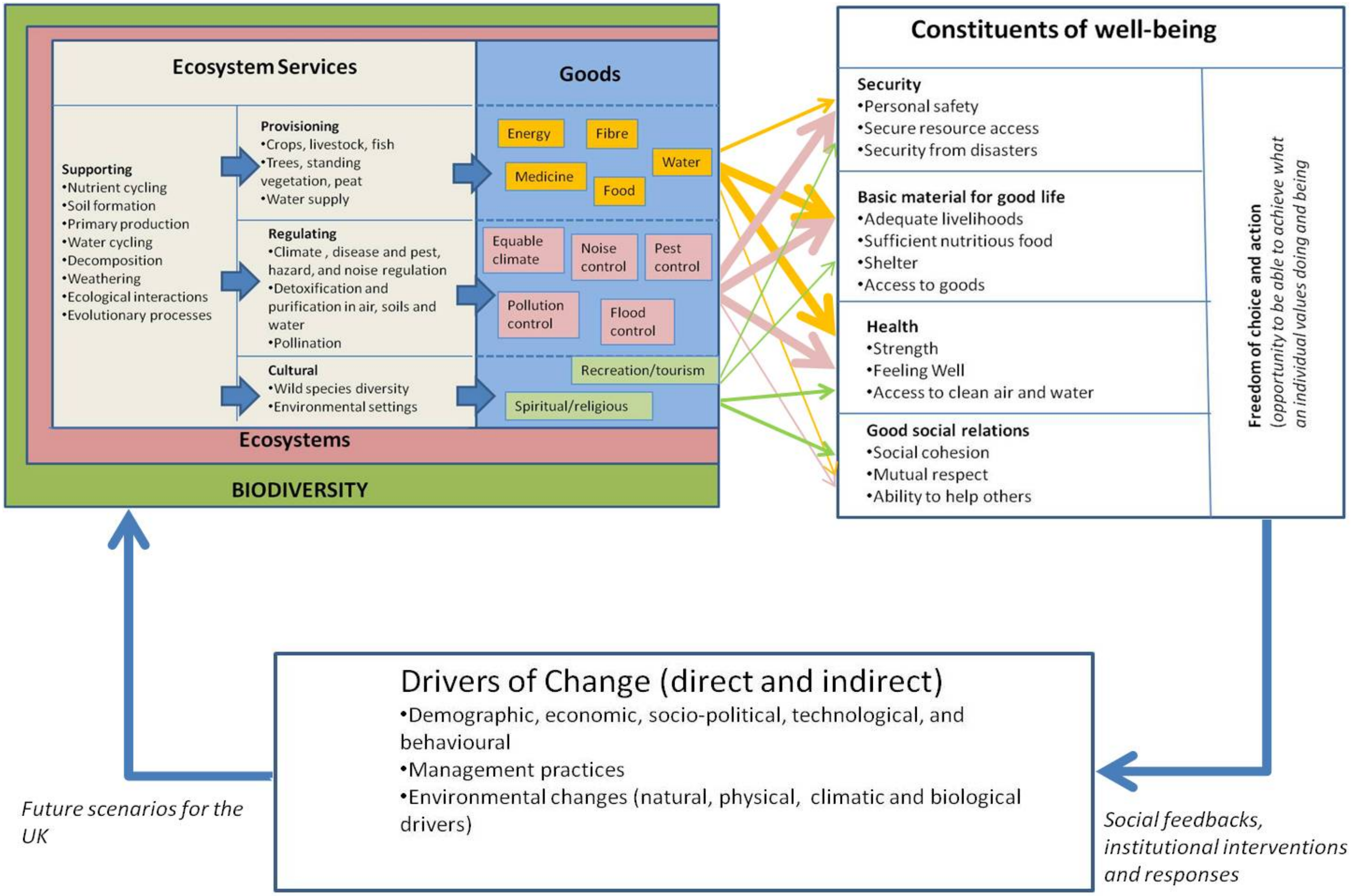

Figure 2-2: The links between ecosystem services, human well-being, and drivers of change.

Biodiversity underpins ecosystems which provide ecosystem services. Those services are underpinned by ecosystem processes (intermediate/supporting services). Final ecosystem services are then provided as a result of these processes, from which goods or benefits can be enjoyed. These goods have weighted influence on human well-being; the thicker the lines the stronger the influence on the relative well-being category. Yellow goods are provided by provisioning services, pink by regulating services, and green by cultural services. The constituents of well-being influence freedom of choice and action which in turn influences the drivers of change through social feedbacks, institutional interventions, and responses. These drivers of change can be direct or indirect and will inform future scenarios for the UK and the associated outcomes these may have on biodiversity, ecosystems and ecosystem services. Adapted from MA (2005) and UKNEA (2011)

The role of biodiversity in underpinning ecosystem service provision is further discussed in The Economics of Ecosystems and Biodiversity (TEEB) (2010). 'Biodiversity reflects the hierarchy of increasing levels of organisation and complexity in ecological systems' (TEEB, 2010: Pg. 5). This hierarchy exists at the level of genes, individuals, populations, species, communities, and ecosystems (TEEB, 2010a). For the greater levels of organisation and complexity to be reached, a functioning level of spatial connectedness (structural connections over space) is needed to maintain links and genetic interchange between individuals of a certain population; thus underpinning ecosystem function through physical connections that facilitate organismal movement (TEEB, 2010a). The ability of an organism to underpin ecosystem service provision through movement is therefore influenced by how connected a landscape is (see section 2.3 for a discussion on landscape connectivity). Just how the spatial connectedness of a landscape facilitates organismal movement depends on how it is structured, thus pattern affects process.

There is a considerable need to describe, map, and analyse these structural patterns in a robust and ecologically meaningful way so that social perception and attitudes towards land-use and green spaces can be influenced, new paradigms formed, strategic decisions made, and effective maintenance developed. Such an approach started in the 1970s in Germany as biotope mapping made its way from the rural to the urban landscape (Lachmund, 2004). Termed as 'urban habitat mapping', land-use and land-cover patterns were described, mapped, and analysed; a research development which has led to a change in ecological perception as the 'artificial' city is reconstructed as an ecological space (Pauleit & Breuste, 2011). The importance of mapping urban land-use and land-cover is still recognised (e.g. by journals such as *Landscape and Urban Planning* and through scientific programmes such as UNESCO's Man and the Biosphere (MAB) programme). Societal decisions and cultural perceptions greatly affect the urban fabric, urban landscape, and urban quality of life and therefore these decisions and perceptions should be formed from comprehensive information and the appraisal of what exists should be based on appropriate, complete, and ecologically relevant data (Jarvis & Young, 2005). Furthermore, the systemic issues which occur within cities (e.g. urban heat island effect, altered hydrological systems, and complex and highly fragmented vegetation cover) arise due to its complex and unique social and environmental characteristics (Gill *et al.*, 2008). Assessing the ecology and resilience of urban areas provides understanding on ways to effectively manage such systemic issues (Niemelä *et al.,* 2011).

Urban ecology and the resilience of the social-ecological system of cities can be addressed, and should be recognised, at two distinctive scales; 1) the ecology and resilience *in* cities and 2) the ecology and resilience *of* cities (Ernstson *et al.*, 2010; McDonnell, 2011). Resilience can be basically defined as the capacity of a system to absorb disturbance and reorganise itself despite change while still maintaining the same structure and function (Holling, 1973; Walker *et al.*, 2004). The study of ecology *in* cities is usually located within a city, at the local to landscape scale, and of single discipline, similarly resilience *in* cities concerns operations at the city/landscape scale, deals with sustaining local-to-regional ecosystem services and is linked to urban form, land-use patterns and local and spatial ecological processes (Grimm *et al.*, 2000; Pickett *et al.*, 2001; Ernstson *et al.*, 2010). The ecology *of* cities studies are interdisciplinary and incorporate both the ecological and social aspects of urban ecosystems, equally the resilience *of* cities operates at the scale of a 'system of cities' tied together through social and economic relations that sustain the flow of energy, matter, and information between cities (Grimm *et al.*, 2000; Pickett *et al.*, 2001; Batty, 2008; Ernstson *et al.*, 2010).

This 'system of cities' relies on the flow of energy, matter, and information permitted through trade, anthropogenic migration, and technological innovation and held together by a shared culture (Batty, 2008; Ernstson *et al.*, 2010). Similarly, the resilience of ecological systems *in* the city depends on the connectivity of the ecological networks and green infrastructure which facilitates the exchange and movement of energy, matter, and information (i.e. genetic information) between ecological communities (Ricotta *et al.*, 2000; Crooks & Sanjayan, 2006b; Biggs *et al.*, 2012). However, Crooks and Sanjayan (2006) argue that as human life becomes more connected, non-human life becomes increasingly disconnected. Specifically, urbanisation and the systemic cultural selective forces of the social-ecological system influence the size, shape, interconnectivity, and composition of natural habitats (Alberti, 2005; Andersson, 2006). This in turn influences the movement abilities of urban organisms, the vectors of energy and matter transferral and therefore important ecosystem service providers (Whelan *et al.*, 2008; Tremblay & St. Clair, 2009; Kunz *et al.*, 2011; Wenny *et al.*, 2011). In particular, many of the ecosystem services provided by birds are solely the result of their foraging behaviour (Whelan *et al.*, 2008; Wenny *et al.*, 2011) (Table 2-1). Birds are highly mobile, occur globally throughout many ecosystems and landscapes, fill many ecological roles, and respond rapidly to environmental change (Wenny *et al.*, 2011).

Table 2-1: A description of various ecosystem services provided by birds and bats.

Ecosystem Service	Examples	Key Studies
Pest Control	Insectivory of herbivorous arthropods, potential rodent control, and weed suppression by insectivorous, carnivorous, and granivorous birds in agroecosystems	(Kay *et al.*, 1994; Holmes & Froud-Williams, 2005; Johnson *et al.*, 2010)
Pollination & seed Dispersal	85% of the oaks (*Quercus robur* and *Q. petrea*) within the Stockholm National Urban Park most likely result from seed dispersal provided primarily by the Eurasian Jay (*Garrulus glandarius*)	(Hougner *et al.*, 2006)
Scavenging & Nutrient Cycling	Scavenging by birds such us raptors, gulls, shorebirds, woodpeckers, herons, and passerines contribute to waste removal, disease regulations, and nutrient cycling	(DeVault *et al.*, 2003)
Cultural Services	Birds play a major role in creating meaningful places and socially valued landscapes. Large, enigmatic bird species as well as garden birds hold a special fascination and attraction for people. Birdsong is now being recognised as having profound psychological effects, - positively influencing productivity, anxiety, and concentration	(Crocker & Mabey, 2005; Norris *et al.*, 2011; Winterman, 2013)

Birds provide benefits within each of the ecosystem service categories – regulating services (pest control), provisioning services (pollination and seed dispersal), supporting services (nutrient cycling), and cultural services. Within urban environments it is the cultural services that provide the most direct benefits for people.

Furthermore, through foraging birds transfer energy both within and among ecosystems, contributing to ecosystem function and resilience (Lundberg & Moberg, 2003). Therefore, the ability of these organisms to move throughout a landscape influences the persistence of populations (and therefore overall biodiversity), the provision of ecosystems services, and the resilience and overall functioning of ecosystems within the landscape. In addition, as movement is impacted on, due to habitat loss and fragmentation, so are species-interaction networks which in turn influence the robustness of ecological networks (Evans *et al.*, 2013).

It is at the ecology and resilience *in* cities scale that the research in this thesis is focused. Although the 'system of cities' approach could reveal insights into how cities survive within a nation– or global–wide network, it ignores the fact that a city's inhabitants interact with their city on a more local to landscape scale. Although, it must also be recognised that many ecosystem services are imported from the urban hinterlands and global commons, while waste is exported (Rees & Wackernagel, 1996; Rees, 2003). Through such large scale interactions, urban ecosystems appropriate a large proportion of the earth's carrying capacity from other, more natural regions by importing resources and exporting waste (Alberti, 2005). To understand the effect of this appropriation on ecosystem function as well as the nature of the 'system of cities' the resilience and ecology *of* cities approach should be taken. However, it can be argued that for the urban population the greatest interaction with nature happens within their neighbourhoods (Natural England, 2010) and the benefits of ecosystem services for urbanites, on a day-to-day basis, are utilised, directly or indirectly, at a local to landscape scale via green infrastructure (Landscape Institute, 2011, 2013). Green infrastructure can be defined as the 'structure, position, connectivity and types of green spaces which together enable delivery of multiple benefits as [ecosystem] goods and services' (Forestry Commission, 2010, pg. 4). Well-connected components of green infrastructure (e.g. urban trees and pond networks) generate infrastructure that is resilient as well as help strengthen ecosystem services provision (Landscape Institute, 2013). As the function and provision of ecosystem services greatly depends on the functional interactions and the ability of organisms (and therefore energy) to move through the landscape, the green infrastructure supporting movement and ecosystem service function should be studied at the *in* cities scale (i.e. within the city landscape).

Research suggests that the greatest beneficiaries of urban nature interaction and ecosystem service utilisation are the most deprived of a population, particularly as green infrastructure supports the amelioration of deprivation (Maas *et al.*, 2006; Mitchell & Popham, 2008; CABE, 2010a; Marmot, 2010). In contrast, a study by Mitchell and Popham (2007) ascertained that although overall health (a component of deprivation) in higher income areas shows no significant association with greenspace, health in low income suburban areas is negatively affected by greenspace. The authors suggest that quality and not just quantity of greenspace is integral to the enhancement of health and specifically that poor quality greenspace (e.g. greenspace that is not accessible and/or is aesthetically poor) may not be sufficient to negate the health problems of more deprived areas, where it is likely that there will be greater amounts of poor quality greenspace (Fairburn *et al.*, 2005; Mitchell & Popham, 2007). Regardless of this lack of quality and due to the combination of increasing urbanisation, urban densification, and past planning policies, residents of low socioeconomic groups and/or ethnic minorities still face the likelihood of living in areas with few green assets, without the resources to move to greener areas (Maas *et al.*, 2006; Zhou & Kim, 2013), thus potentially leading to environmental injustice. Environmental justice is a term, originating from the United States of America, which describes the 'inter-relationship between geographical space and conceptions of equity and justice' (Agyeman & Evans, 2004 pg. 155). Specifically, environmental justice combines the principles of environmental protection and social justice and is concerned with how negative (e.g. pollution) and positive (e.g. greenspace) aspects of the environment are distributed across landscapes in relation to society (Fairburn *et al.*, 2005). To understand environmental justice, or in other terms the levels of environmental equity within a landscape, it is essential to first determine how unevenly distributed aspects of environmental quality are in relation to social structure (Walker *et al.*, 2003). As green infrastructure is a key determinate of urban environmental quality (Bell *et al.*, 2008) the distribution of integral green infrastructure assets, in relation to social structure, is a focal study area within the environmental justice/equity framework (Van Herzele *et al.*, 2005; Barbosa *et al.*, 2007; Jones *et al.*, 2009).

Urban trees can be described as 'flexible' green infrastructure assets as they can be used to form green corridors and avenues through neighbourhoods and along streets thus providing green space outside of fixed areas such as parks (Zhou & Kim, 2013). Therefore, urban trees provide opportunities to increase the level of green space in neighbourhoods that may not have access to fixed/formal greenspace due to physical barriers such as distance or perceived barriers such as negative perceptions towards greenspace (Maas *et al.*, 2006; Jones *et al.*, 2009; TDAG, 2012). A study by Gilchrist (2011) also ascertained that it is the most deprived and vulnerable section of the urban population that benefit the most from urban trees. Despite this, the level of canopy cover in urban areas has been found to be lower in ethnic minority, low-income, and more deprived neighbourhoods (Landry & Chakraborty, 2009; Kendal *et al.*, 2012; Zhou & Kim, 2013). Thus the provision of urban trees is not equal across society. However, it has not been established whether the configuration of urban tree canopies, both vertically and horizontally, follows a similar pattern. Although some work has been done by Duncan *et al.* (2014) on the density of trees occurring in differing neighbourhoods, how these trees were configured within these neighbourhoods was not investigated. Furthermore, previous studies only focus on a single or at best a few aspects of social structure (in regards to deprivation) such as ethnicity, income level, and or educational level. It would be more advantageous to understand how canopy cover and configuration is related to neighbourhoods/areas structured based on a plethora of deprivation values. Hence a research question arises; does the relationship between canopy cover and deprivation extend to canopy landscape structure and overall deprivation?

Describing and assessing the state and shape of the social-ecological urban fabric, in terms of environmental equity, and understanding how organisms respond to the permeability of the spatial patterns occurring in urban areas is of critical importance for urban conservation strategies, urban greenspace design, urban ecosystem resilience, and human well-being. Identifying areas for improving, expanding and maintaining the extant infrastructure and highlighting areas for the development of new infrastructure is a research and strategic planning criteria which champions the notion of the social and the ecological combining to ensure resilience and sustainability. Uncovering patterns and spatial relationship can also inform targeted landscape management projects. Therefore, quantitative methods for measuring landscape structure and the patterns and connectivity of landscape elements are needed. Furthermore, to incorporate the best of both worlds the ecology *in* cities approach taken within this thesis should, unlike past studies, undertake a more interdisciplinary approach as taken by ecology *of* cities studies; ecological and sociological approaches should be combined. In regards to measuring landscape pattern and connectivity, appropriate landscape elements must be identified. If the benefits of ecosystem services are routinely appropriated by the urban population via the green spaces and green infrastructure around them, these landscape elements must represent the most noticeable and arguably beneficial aspects of such green spaces (i.e. critical green infrastructure asset/s).

2.2 The Urban Tree Infrastructure (UTI)

Trees exist throughout the urban fabric - within remnant woodlands, urban parks, private gardens, abandoned land and brown field sites, as well as on roads and highways in the form of street trees (Britt & Johnston, 2008; Jim, 2011; Johnston & Percival, 2011; TDAG, 2012). The resulting 'urban forest' has developed through an interrelated set of ecological, social, and economic processes (Nowak *et al.*, 1996; James *et al.*, 2009; Chen & Wang, 2013). While ecological conditions inherently underpin the existence of urban forests the establishment and management of urban trees are often the result of human intervention and institutional frameworks, as standards for the planting, care, and protection of trees are embedded into local policy documents, tree establishment policies, and urban forestry management techniques (Nowak *et al.*, 1996; Pauleit *et al.*, 2002; Jim & Chen, 2008; Dobbs *et al.*, 2011; TDAG, 2012). In addition, and as stated in section 2.1, the establishment of urban tree canopy cover is related to social deprivation and the economic condition of a given area (Landry & Chakraborty, 2009; Kendal *et al.*, 2012; Zhou & Kim, 2013).

Evidence from a study throughout Chinese cities has illustrated the nature of this social-ecological relationship, highlighting that ecological forces (e.g. natural conditions and local biophysical factors) are the most statistically significant factors in determining urban forest cover, followed by cultural forces (e.g. economic development and institutional capacity) (Chen & Wang, 2013). However, the authors also state that the results of their study are specifically important and useful for informing urban forestry decisions in China (and other transitional economies) and therefore may not be transferable to UK cities. Nevertheless, the study effectively expresses the strong social-ecological nature of urban forests. However, the term 'urban forest' itself does not reflect the exerted influence cultural perceptions and governmental policies have on the structure of urban tree communities. Therefore, it is proposed that the term 'urban tree infrastructure' (UTI) is to be used when discussing trees within an urban area. This term encompasses all trees, including (but not limited to) urban woodlands, public and private trees (maintained and unmaintained), singular scattered trees, and street trees. The term also reflects the need for urban trees to be equally considered alongside other types of infrastructure, especially when it comes to those decisions which will shape the places in which people live (TDAG, 2012). Therefore, the UTI is a construct of the social and the ecological, a restorer of the 'environmental and social balance' within urban areas, and thus the UTI contributes to the conditions for economic, ecological, and societal success (TDAG, 2012).

The UTI as a whole has an important role in creating sustainable communities, providing the urban population with several aesthetic, social, and health benefits (O'Brien, 2005; Britt & Johnston, 2008; Dobbs *et al.*, 2011). Trees and woodlands themselves, regardless of being in an urban environment, are being promoted in the UK as 'nature's health service' (O'Brien, 2005). Urban woodlands in particular present one of the most complex and the most valued natural areas within urban environments (Tregay, 1979; Kunick, 1987). They generate societal benefits by providing restorative environments (Gilchrist, 2011) while the ability to simply view urban trees has been shown to reduce mental and physical stress and ameliorate emotional and physiological states (Ulrich, 1986; Parsons *et al.*, 1998). Dandy *et al.* (2011) investigated the societal benefits of another component of the UTI – street trees – and, in summary, found that they have the capacity to generate social interaction through promoting greater use of public areas; have an aesthetic value that is influenced by canopy size and tree height; provide restorative environments; improve the feeling of security through increased ownership and surveillance; and improve social cohesion and sense of community. In addition to these cultural services providing social benefits, the UTI also provides environmental benefits such as carbon sequestration and the removal of air pollutants, thus mitigating the environmental quality problems characteristic of urban environments (Yang *et al.*, 2005; Jim & Chen, 2009; Dobbs *et al.*, 2011; Escobedo *et al.*, 2011). The ecosystem functions underpinning these benefits can lead to societal well-being benefits as, for example, the presences of street trees are associated with the reduction in the risk of childhood asthma (Lovasi *et al.*, 2008). However, it is unlikely that many urban residents are aware of or interested in the ecological functions supporting such benefits (Escobedo *et al.*, 2011). In contrast, urban woodlands and individual trees often hold specific meaning for people and are seen as representing nature within urban environments (O'Brien, 2005). Therefore, it could be argued that the cultural services provided by the UTI are more actively experienced by urbanites than the more passively experienced environmental services.

The UTI also produces ecosystem disservices – the costs associated with urban trees which negatively affect well-being (Lyytimäki & Sipilä, 2009). These associated negative services can be financial (e.g. management costs), environmental (e.g. volatile organic compound emissions), and/or social (e.g. fear of crime, provide habitat for vector-based diseases) (Lyytimäki & Sipilä, 2009; Escobedo *et al.*, 2011). While the majority of urbanites are unaware of the, specifically environmental, benefits of the UTI they do generally recognise their disservices due to personal experiences (Agbenyega, 2009). However, this is also an issue of perception and why the use of well-informed management practices and public engagement with the UTI is necessary (Escobedo *et al.*, 2011). Furthermore, the social benefits associated with the UTI often outweigh the environmental and economic costs of maintaining it (Dobbs *et al.*, 2011).

The governance of the UTI and the public attitudes towards trees in cities greatly determines how successful the provision of benefits will be. For example, individual tree survival and the resilience of the UTI as a whole is influenced by both social attitudes (whether trees are perceived as beneficial or not) and organisational governance through the use of various tree planting and establishment criteria, conservation schemes, and educational programmes (Pauleit *et al.*, 2002; Lohr *et al.*, 2004; Soares *et al.*, 2011; TDAG, 2012). In addition, Soares *et al.* (2011) also suggest increased diversity would ensure the sustainable provision of UTI benefits. Therefore, diversity, such as biodiversity (within the UTI and the organisms it supports) and structural diversity, is needed so that ecological interactions increase, internal structures and processes strengthen, and resilience is maintained. The resilience of ecological systems in cities is affected by the connectivity of the green infrastructure which facilitates the exchange and movement of energy, matter, and information (i.e. genetic information) between ecological communities (Ricotta *et al.*, 2000; Crooks & Sanjayan, 2006b; Biggs *et al.*, 2012) (Chapter 2.1). More specifically, the movement and foraging behaviour of urban birds contributes to ecosystem function and resilience as well as the provision of several ecosystem services (Lundberg & Moberg, 2003; Wenny *et al.*, 2011) (Table 2-1).

The fragmentation of the whole UTI can have the same adverse effects on urban bird diversity as exhibited in the fragmentation of natural ecosystems (Fernández-Juricic & Jokimäki, 2001). The UTI strongly influences the species richness of birds in urban landscapes. Fernández-Juricic (2001a) indicated that the number of bird species in Madrid increased from streets without trees (least suitable habitat) to wooded urban parks (most suitable habitat), with street trees being the intermediate landscape element. Furthermore, the study revealed that tree lined streets connecting urban parks, along with the complexity of vegetation structure, positively influences the number of bird species. This relationship was explained by the notion that street trees provide alternative habitat for feeding and nesting, allowing urban birds to supplement their resource needs (Fernández-Juricic, 2001a) (see Figure 2-3 for information about resource supplementation). Complex habitat structure ensures the provision of niche- and micro-habitats which favours colonisation by new species (Fernández-Juricic, 2000). Therefore, tree lined streets, and more specifically those which connect wooded urban parks, can increase urban landscape connectivity (Fernández-Juricic & Jokimäki, 2001). Street trees have also been shown to enlarge and change the shape of urban parks; as population density increases in the parks, individuals begin to occupy street trees as alternative habitats (Fernández-Juricic, 2001b). Consequently, structurally complex tree lined streets have an influential role in augmenting the connectivity of urban environments for the suitability of bird species (Savard *et al.*, 2000).

It has been argued that compared to natural ecosystems, urban ecosystems exhibit unique ecological and social dynamics as a result of different ecological patterns, processes, and disturbances (Pickett *et al.*, 1997). However, it has also been argued that the ecological patterns and processes between urban and other 'natural' ecosystems are essentially the same, the only difference being the importance and frequency of certain disturbances and processes (Niemelä, 1999). Therefore, existing ecological theories can be applied when studying urban ecology (Niemelä, 1999). Basic ecological theory suggests that it is not just the size of habitat patches but also the quality or structural complexity of habitat patches, within a landscape, that affects biodiversity (MacArthur & MacArthur, 1961; Rosenzweig, 1995). This within patch structural complexity can be explained by the distribution of vegetation cover at different heights, in particular woodland patches and the influence of canopy layer cover on bird species richness (MacArthur & MacArthur, 1961; Huth &

Possingham, 2011). In fragmented environments such as urban areas (Luck & Wu, 2002) abundance of birds and their assemblages are influenced more by the three-dimensional structure of tree canopies than contiguous environments as structural diversity provides potential habitat for fragmentation sensitive species (Goldstein *et al.*, 1986). However, ecological studies assessing landscape connectivity often over simplify landscape models by ignoring this basic ecological observation. This simplification manifests itself in a number of ways; (1) by underestimating the importance of describing urban patterns occurring across multiple scales and dimensions; (2) by failing to recognise the importance of vertical complexity by providing only two-dimensional representations of landscape elements; and (3) by focusing on the whole landscape while ignoring within patch heterogeneity (Alberti, 2005; Dolman *et al.*, 2007a; Lesak *et al.*, 2011; Walz, 2011). Therefore, although connectivity between 2D representations of habitat patches can provide insightful results, important structural elements existing on a vertical plane are ignored. Therefore, structural patterns existing in nature are not represented in 2D landscape models.

This oversight is especially true for studies which analyse the configuration and composition of woodland patches within a landscape, as they are often considered as homogeneous patches within heterogeneous landscapes (Dolman *et al.*, 2007a). However, the intra-patch structure of wooded areas can differ significantly from patch to patch and the spatial arrangement of habitat elements within patches have a strong influence on foraging behaviour and habitat suitability (Dolman *et al.*, 2007a). Variation in tree height is an important structural attribute as patches containing a high degree of height diversity are likely to contain a variety of tree ages and species and thus provide for a diversity of micro-habitats (McElhinny *et al.*, 2005). Furthermore, tree heights are often incorporated into habitat suitability and habitat complexity models which maintain that bird species richness for a patch of a given size is relative to its structural diversity (Schroeder *et al.*, 1992; Huth & Possingham, 2011). Therefore, any analyses of landscape heterogeneity for the connectivity of organisms, especially birds, need to consider the pattern of landscape elements over various heights. A critical review on ways to quantify such spatial patterns is presented in section 2.3.

2.3 Landscape Structure and Connectivity – quantifying spatial patterns

Dunning *et al.* (1992) identified four ecological processes acting at the landscape scale that influence population dynamics and community structure: (1) landscape complementation; (2) landscape supplementation; (3) sources and sink; and (4) neighbourhood effects. These ecologically critical landscape processes are dependent on the *configuration* (spatial arrangement) and *composition* (distribution and relative abundance) of resources distributed throughout the landscape in habitat patches (Taylor *et al.*, 1993). The structure of the landscape can be measured directly by describing the distribution of these resource, or rather habitat patches. For the described processes to work organisms must move among these habitat patches (Dunning *et al.*, 1992; Taylor *et al.*, 1993) (Figure 2-3).

A critical evaluation of Figure 2-3 reveals that there was an assumption that the ability of an organism to move through the landscape, and subsequently undertake the described ecological processes, depended only on landscape configuration and composition. Such an assumption disregards the influential effect landscapes have on the ability of an organism to move as well as the intrinsic movement abilities of that organism. Tischendorf and Fahrig (2000a, 2000b) highlight that it is now recognised that the ability of an organism to move through a landscape not only depends on the structure of the landscape but also on the organism's behavioural traits and movement capabilities. This interaction, between species attributes and landscape structure, in determining the movement of an organism between resource patches was conceptualised as landscape connectivity by Merriam (1984). The importance of organismal movement as a component of landscape structure was recognised by Taylor *et al*, (1993) as the authors defined connectivity as 'the degree to which the landscape facilitates or impedes movement among resource patches' (p 571). With *et al.* (1997) further defined landscape connectivity as 'the functional relationship among habitat patches, owing to the spatial contagion of habitat and the movement responses of organisms to landscape structure' (p 151). Therefore, landscape structure should be measured in three ways - by describing the distribution of habitat patches in terms of their; (1) configuration (spatial arrangement in the landscape); (2) composition (distribution and relative abundance within the landscape); and (3) connectivity (the functional and structural links between patches within the landscape).

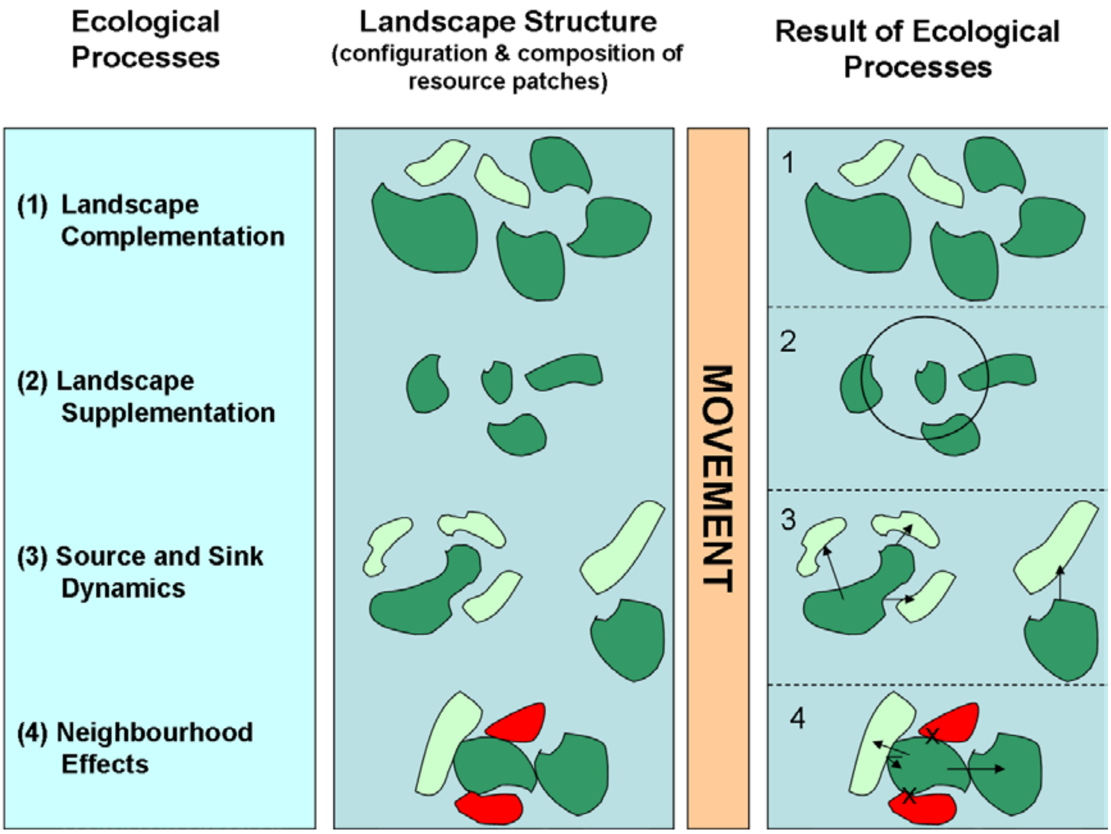

Figure 2-3: A representation of four important ecological processes acting at a landscape scale and affected by landscape structure.
(1) Organisms acquire a full complement of resources to meet their needs. These resources may be found in different habitats (in this case two habitat types i.e. light and dark green patches represent different habitats providing different resources). It is assumed that heterogenic areas where habitats are relatively close to each other support a greater amount of individuals than more homogenous areas; **(2)** Organisms supplement their existing resources with those in additional patches. Where a patch is too small to support a population, organisms may move to other patches within an accessible portion of the local landscape (dark ring); **(3)** Movement from source to sink populations is required to maintain sink populations. Rich habitat patches (dark green) produce excess individuals, and poor patches (light green) do not produce enough individuals to support a local population. Therefore, sink populations (light green) depend on dispersers (arrows) from the source population (dark green); **(4)** Organisms move through patches but focus on the permeability of boundaries between contiguous patches. An individual's movement is unhindered when crossing a permeable boundary (dark green), affected when crossing into a semi-permeable patch (light green), and completely restricted by habitats with an impermeable boundary (red) (adapted from Dunning *et al.* 1992 and Taylor *et al.* 1993).

Connectivity is the most important component of landscape structure as it requires an understanding of landscape configuration and composition as well as animal movement and behaviour. Furthermore, a loss of connectivity is seen as a major threat for both biodiversity conservation and the maintenance of ecological functions operating across landscapes (Pascual-Hortal & Saura, 2006). In addition, connectivity is particularly important for tackling the effects of climate change on species and ecosystems, as a well-connected landscape may allow species to adjust to the shifts in their natural ranges resulting from changes in environmental conditions (Opdam & Wascher, 2004).

As the concept of landscape connectivity is concerned with both the spatial and structural components of the landscape as well as the behavioural attributes of organisms, it has two theoretical components; 1) structural connectedness, and 2) functional connectivity (Pascual-Hortal & Saura, 2006). Structural connectivity can be quantified by analysing the two components of landscape structure, configuration and composition, which are not biased towards the attributes of an organism (Tischendorf & Fahrig, 2000b). Therefore, structural connectivity considers how the arrangement of different landscape elements and habitat patches form a mosaic of features that can either hinder or enhance organism movement (POST, 2008). Metzger and Decamps (1997) identified that structural connectivity can be defined and quantified on the basis of; (1) Inter habitat patch distance; (2) patch density; (3) patch complexity; (4) width and quality of corridor networks; (5) density of stepping stones; and (6) permeability of landscape matrix. However, they also highlight that measuring these components alone will not reveal a measure of landscape connectivity as specific levels of structure may or may not meet the requirements of a given species. These requirements depend on the species' capacity of movement, demographic potential, minimal area, and habitat requirements (Metzger & Decamps, 1997). In other words, by measuring functional connectivity which explicitly considers the behavioural responses of an organism to the landscape structure (Tischendorf & Fahrig, 2000b), or rather the ability of a species to move through a landscape mosaic (POST, 2008).

Metzger and Decamps (1997) inclusion of terminology such as '...quality of corridor networks' and 'permeability of landscape...' denotes species specific perceptions of landscape structure. That is to say, the permeability of the matrix is highly species specific and to quantify and model permeability would only be useful in regards to that species. This is the same for modelling the quality of a landscape component such as habitat patches; it depends on the species of study. Therefore, a Metzger and Decamps (1997) approach to landscape connectivity requires a knowledge of species perception, behaviour, capacity for movement, and specific habitat requirements as well as mapping structural patterns. Such an approach is therefore not only data intensive but limited in its use as it is extremely species or even individual specific as perceptions can change from one organism to the next (Farina & Belgrano, 2004; Farina & Belgrano, 2006). Results and recommendations stemming from such an approach can only be regarded as a study into behaviour, with movement being the catalyst to instigate the behaviour and the concept of landscape connectivity the framework to study behaviour. Therefore, how can stakeholders such as local councils and landscape managers implement best practice if recommendations from the scientific community are focused on a small set of species and whose requirements may differ from each other? An alternative approach would need to incorporate general organism perception and movement capacity into landscape connectivity models. In order to do this, it is necessary to use efficient and appropriate landscape connectivity metrics.

Quantification of spatial heterogeneity is needed to elucidate the relationships between ecological processes and spatial patterns (Turner, 1989, 1990). Consequently, a variety of landscape metrics have been developed to measure, analyse, and interpret the composition, configuration, and connectivity of landscapes (Uuemaa *et al.*, 2009). Landscape connectivity has the strongest potential to describe and highlight the links between processes and patterns due to the combination of geographic and biological thinking. Landscape connectivity studies can be broadly divided into two categories, empirical field studies and modelling studies (Tischendorf & Fahrig, 2000b). Field studies often use point count surveys and translocation or mark-release-recapture experiments, and therefore can only represent local environments on a small scale (Hedblom & Söderström, 2010; Tremblay & St. Clair, 2011). Modelling studies simulate habitats, often using remotely sensed data incorporated into a GIS, where habitat patch distribution and organismal movement capabilities are assessed mathematically using landscape connectivity metrics (Metzger & Decamps, 1997; Tischendorf & Fahrig, 2000a; Bierwagen, 2007).

Through comparative studies Pascual-Hortal and Saura (2006, 2007b) have emphasised that the most commonly used connectivity metrics fail to fulfil the desirable properties needed for effective conservation decision-making at a landscape scale. Namely, connectivity metrics need to not only describe how connected the landscape is for a focal species but also to identify those habitat patches which are the most important for maintaining connectivity and to be sensitive to the loss of different landscape elements (e.g. habitat patches, links between patches, and components made from linked patches) (Saura & Torne, 2009). The metrics that do stand up to these requirements have been identified as the integral index of connectivity (IIC) and probability of connectivity (PC) which are based on both graph theory and the habitat availability concept (Pascual-Hortal & Saura, 2006; Saura & Pascual-Hortal, 2007b).

Graph theory and network analysis (the assessment of graph topology) has existed for centuries and persists within several disciplines such as environmental sciences, social sciences, engineering, and mathematics (Urban *et al.*, 2009). Relatively recently, the use of graph theory to model the functional relationships between organisms and landscape patterns has increased (Galpern *et al.*, 2011). The inter-disciplinary nature of graph theory can be related to the fact that it is explicitly concerned with connectivity within systems (Urban *et al.*, 2009). Graphs can be used to model landscapes, describe underlying structure and ecological processes, which can therefore inform conservation practices, landscape planning, and design (Bunn *et al.*, 2000; Rhodes *et al.*, 2006; Minor & Urban, 2007, 2008; Zetterberg *et al.*, 2010; Foltête *et al.*, 2014). The construction of a graph relies on an understanding of the components which create it. On a basic level, a graph is a set of nodes which are connected by links. A link between two nodes indicates a functional connection between them. These basic components form the basis of more complicated structures. These structural elements can then be related to ecological processes (Table 2-2) (Urban & Keitt, 2001; Galpern *et al.*, 2011). The ecological processes which the structural elements represent depend on the research aim and the underlying conceptual model of the system under assessment (Urban *et al.*, 2009; Galpern *et al.*, 2011). With regards to the research presented within the thesis, the underlying concept is that landscape structure influences ecological function. More specifically, the structural patterns exhibited by tree canopy patches (the habitat patch of interest) influences the function of the UTI in regards to the provision of functionally connected habitat and resources for urban birds. As a result, landscape graphs constructed within the thesis will model the structural relationship between tree canopy patches – known as ‘patch-based graphs’.

Table 2-2: Summary of graph theory terminology and ecological relevance.

Graph Term	*Definition*	*Ecological Relevance*
Node	Representative of a habitat patch. Typically points at the centroid of the patch or have two-dimensional geometry.	The habitable, resource patch of a focal organism. Critical for organism survival.
Degree	A node attribute that measures the number of connected neighbours adjoining a focal node.	The connectivity and resilience of an ecological system strongly depends on node degree distribution. A high degree habitat patch may act as a population source or sink, while extinction may occur in low degree patches if neighbouring patches are removed.
Centrality	A node attribute which measures the influence a node has on its neighbours. Node degree can be considered a measure of centrality, along with betweenness and closeness centrality.	High degree patches, high betweenness patches, and low closeness patches are conservation priorities as they have the most influence over local movement (within sub-networks).
Hub	An important node for maintaining network connectivity.	While other habitat patches may be removed without affecting overall connectivity, hubs keep the landscape from fragmenting
Link	The existence of a link between two nodes implies that these nodes are functionally connected. Links can be measured from the centre or edge of a patch.	Represents movement between nodes (e.g. energy, information, disease, individuals)
Path	A sequence of links in a graph joining more than two nodes so that no node is visited more than once.	Represents potentially connected routes for an organism to take.
Graph diameter	An attribute of the network – measures the longest path between two nodes, where the path itself is the shortest possible distance.	Short graph diameters imply that movement is fast through the network – beneficial in regards to organism movement, detrimental in regards to the spread of disease.
Component	A group of connected nodes	Movement can happen throughout all patches within a component. There may be no movement between different components, or such movement may incur additional survival costs.
Sub-network	The connected nodes within a component	Represent the locally occurring routes and habitat patches an organism may move through. The existence of a sub-network implies there is the potential of local connectivity even if the whole landscape is unconnected.
Compartmentalisation	A graph which contains a number of hubs that are not directly connected to each other. In other words, high degree nodes will have low degree nodes as neighbours.	High compartmentalisation reduces or slows movement through the network - beneficial in regards to the spread of disease, detrimental in regards to organism movement.

Definitions and/or examples of the ecological relevance of landscape graph structural mechanisms adapted from Freeman, 1979; Albert *et al.,* 2000; Urban & Keitt, 2001; Minor & Urban 2007, 2008; Galpern *et al.,* 2011.

Within a patch-based graph, the focal habitat (i.e. UTI), distinguishable from the surrounding inhospitable matrix, serve as nodes. The links between these nodes represent potential for movement. When these links represent geographic distances, then a connection can only occur when the distance is below an ecologically relevant movement threshold (Galpern *et al.*, 2011). Such patch-based graphs therefore model potential functional connectivity as the links used to describe the graph represent an organism's perception of, or functional response to, the landscape (Galpern *et al.*, 2011). A consecutive series of these functional links between nodes creates a path (representing potential routes) and the longest path between two nodes (while taking the shortest route) is the graph diameter (Urban & Keitt, 2001). A component arises when a group of nodes are connected by links so that an individual inhabiting the component can potentially move to any node within the component (Urban & Keitt, 2001). Nodes critical for maintaining connectivity are known as hubs and if components are formed from hub nodes which are not directly connected to each other, a compartmentalised graph occurs (Melián & Bascompte, 2002; Minor & Urban, 2008). The connected nodes within these compartmentalised components represent sub-networks – areas of localised connectivity (Maslov & Sneppen, 2002). The central, influential node(s) within these sub-networks can be measured by its level of centrality*;* via node degree, betweenness, and or closeness (Freeman, 1979; Urban *et al.*, 2009; Opsahl *et al.*, 2010). Centrality refers to the influence a node has on its neighbours and within the network (or sub-network). Node degree measures the number of links and therefore number of neighbours a node has, betweenness centrality identifies which node(s) have the highest proportion of the shortest paths running through them, and closeness centrality considers the length of the paths from a focal node to all other nodes it is connected to (Freeman, 1979). Table 2-2 contains a description and example of the ecological relevance of these structures and attributes.

There are several strengths associated with using a graph theoretic approach in the assessment of complex ecological systems or networks existing across landscapes. To inform landscape planning and design, several studies have used graph theory in the analysis and visualisation of species-habitat interactions (Urban *et al.*, 2009; Zetterberg *et al.*, 2010). Graph theory is dynamic enough to be used as an initial, heuristic framework for conservation management (Bunn *et al.*, 2000) yet still have the same explanatory power as more data and time intensive methods (i.e. spatially explicit population models) (Minor & Urban, 2007).

A network analysis of a landscape graph is an effective tool for the assessment of complex systems as it focuses on the components of a system, how these components are structured, and how this structure can affect the performance and state of the system (Janssen *et al.*, 2006; Zetterberg *et al.*, 2010). Network analysis has been used to assess various systems across several scientific disciplines (e.g. the internet, social systems, food webs, protein networks) (Freeman, 1979; Maslov & Sneppen, 2002; Krause *et al.*, 2003). Therefore, several well developed and tested graph-theoretic metrics have been created to undertake network assessments. These traditional metrics have also been related to ecosystem and social-ecological system function (Bunn *et al.*, 2000; Urban & Keitt, 2001; Galpern *et al.*, 2011). However, there was an inherent problem with this inter-disciplinary crossover – a rich vocabulary that is sometimes not consistent across disciplines (Urban *et al.*, 2009). Specifically, the graph theoretic meaning of connectivity differed from the landscape ecologist's meaning of connectivity (Urban & Keitt, 2001). A graph theoretic approach only considered structural connectivity, not functional connectivity. However, since the ecological adoption of graph-theory, several functional connectivity graph-based metrics have been developed (Pascual-Hortal & Saura, 2006; Saura & Pascual-Hortal, 2007b).

Several landscape connectivity metrics based on graph theory also disregard habitats themselves as spaces where connectivity can occur. For example, the often used Harary index (Ricotta *et al.*, 2000; Jordan, 2003) only considers the shortest path (route along nodes where no node is visited twice) between two different patches in terms of the topological distance between them (i.e. the number of links). Graph diameter (Urban & Keitt, 2001) considers the maximum length of all the shortest paths between any two nodes in the graph (measured in distance units, not the number of links). Other more simplified graph based metrics quantify the number or size of specific elements within the landscape graph such as the total number of links, number of components (connected region of nodes which all have a path to each other), mean size of components, and largest component. Although such metrics are important in regards to understanding graph and network structure (and can be related to system resilience, see Chapter 5) they only consider landscape connectivity according to Taylor *et al.'s* (1993) strict definition: connectivity is the degree to which the landscape facilitates or impedes movement *among habitat patches.* Therefore, these metrics ignore the fact that movement also takes place *within habitat patches* (if the size or quality allows for it) and so habitat patches should be viewed as areas where connectivity occurs (Pascual-Hortal & Saura, 2006). In order to recognise habitat patches as areas of movement facilitation within the quantification of connectivity Pascual-Hortal and Saura (2006) developed the concept of 'habitat availability'. The concept integrates habitat area, or other habitat patch attributes (e.g. quality based on a numerical value), and between patch connections into graph based connectivity metrics. Therefore, the concept works on the assumption that for a habitat to be easily available it needs to be both abundant (and/or of a suitable quality) and well connected (Pascual-Hortal and Saura, 2006).

As mentioned, two landscape connectivity metrics adhere to the requirements discussed above. Both IIC (Pascual-Hortal & Saura, 2006) and PC (Saura & Pascual-Hortal, 2007b) have been used to quantify overall landscape connectivity and identify those habitat patches which are critical for maintaining connectivity (Neel, 2008; Ribeiro *et al.*, 2011; Saura *et al.*, 2011; Decout *et al.*, 2012; Crouzeilles *et al.*, 2013; Foltête *et al.*, 2014). PC is considered to have more descriptive power than IIC as it not only considers Euclidean distances between patches but also the strength of the links between patches, which is lost when connections are described as binary (i.e. either patches are connected or unconnected) (Saura & Pascual-Hortal, 2007b; Foltête *et al.*, 2014). However, the movement probabilities of organisms are complex to quantify and often are arbitrarily set (e.g. Foltête *et al.,* 2014). Furthermore, the complexities of movement probabilities and thresholds can be lost in the calculation of PC as the probability of movement between two patches is calculated as simply a negative exponential function of the internode distances (thus extreme thresholds are ignored). In addition, empirical studies demonstrate that different bird species follow different canopy gap-crossing probability curves (Desrochers & Hannon, 1997; St. Clair *et al.*, 1998; Creegan & Osborne, 2005; Awade & Metzger, 2008; Robertson & Radford, 2009) and therefore PC measurements can only be species or individual organism specific (Bodin & Saura, 2010) (Figure 2-4). In regards to this limitation, IIC has been considered a better choice of metric as it focuses on the topology and availability of landscape graphs which can allow for the long-term studies of potential species movement rather than individual movement (Crouzeilles *et al.*, 2013).

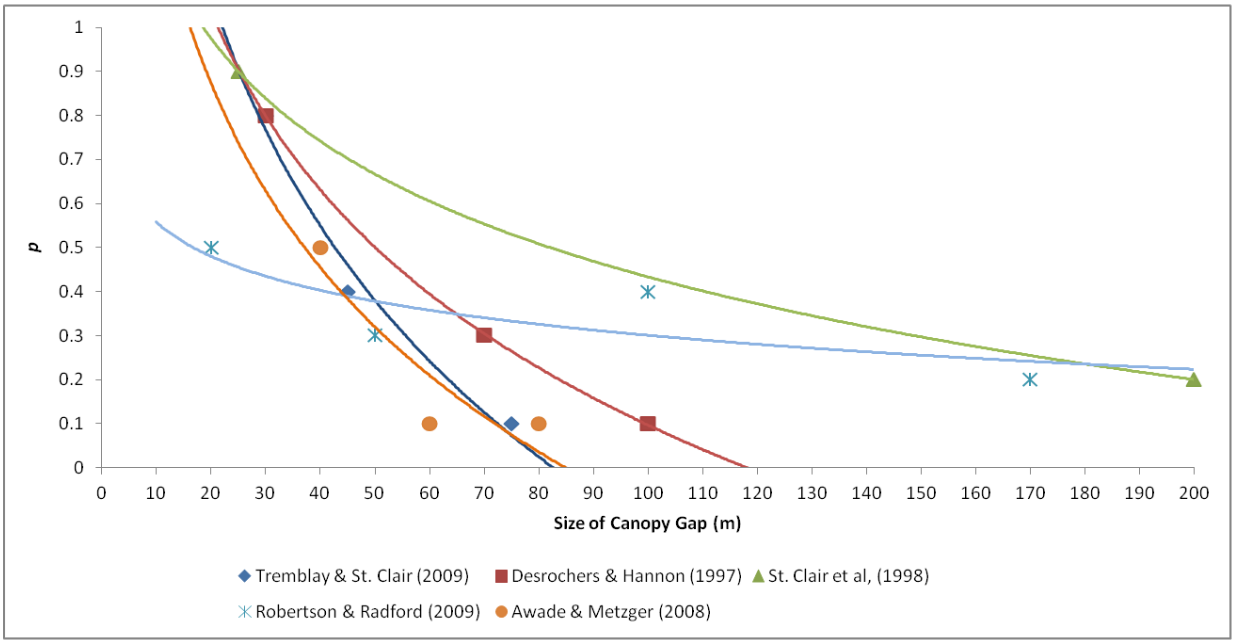

Figure 2-4: Passerine gap crossing probability.
The results from several studies, within a variety of landscapes and focusing on a variety of species, emphasise that gap-crossing probability is complex and species specific. What can be ascertained from the graph above is that as gap distance increases the probability of movement decreases and the maximum gap crossing distance is 200m.

Landscape metrics based on graph theory and the habitat availability concept can be combined with organism movement threshold data and gap-crossing abilities (i.e. perception of gaps and movement capacity) to measure the connectivity of landscapes for a particular species, genus, or order. Although dispersal and home range movement patterns can be summarised by assessing the probability of individuals moving a certain distance, the process behind these patterns are part of a more complicated chain of processes; namely dispersal propensity, ability to move outside of habitat, and perception of new habitat for immigration (Baguette & Van Dyck, 2007). Such processes are not always species-specific but should rather be considered as an individual trait as the ability and willingness to disperse, displace, and immigrate varies among populations according to a myriad of selective pressures (Van Dyck & Baguette, 2005). However, a graph model based only on habitat and dispersal and gap-crossing capability thresholds have been shown to make strong connectivity predictions very similar to spatially explicit population models which can be data intensive, requiring further life history and behavioural parameters (Minor & Urban, 2007). Furthermore, the strength or intensity with which nodes and components are connected has an influential effect on network resilience and the ecological processes acting at a landscape scale which underpin the provision of ecosystem services (Biggs *et al.*, 2012). Therefore, by recognising and assessing the dynamic, functional interactions of the landscape networks emerging from landscape graphs, conservationists, landscape planners and managers are better equipped to manage and enhance urban resilience and ecosystem services (Zetterberg *et al.*, 2010). The relationship between landscape network analysis and system resilience is critically assessed in Section 2.4.

2.4 Landscape Networks and System Resilience

Unlike other forms of traditional urban infrastructure, urban green infrastructures such as the UTI are ecological as well as societal constructs (see section 2.2). In social-ecological systems humans can intentionally and unintentionally change the resilience of the system as well as transform the entire system through and because of structural, behavioural, and cultural changes (Berkes *et al.*, 2003; Walker *et al.*, 2004). In this regards the UTI can be approached as a social-ecological system: an ecological system connected and fragmented by humans, the resilience of which can be directly managed by humans, and through its own ecological connections form social connections (Janssen *et al.*, 2006; Dandy *et al.*, 2011). By using ecologically relevant models to inform cultural perceptions and organisational policies, the spatial structure and resilience of the UTI can be competently adapted or transformed.

By using a resilience approach (Holling, 1973; Folke, 2006; Garmestani *et al.*, 2009) to investigate and understand social- ecological systems, the goals of sustainable development and the continual provision of ecosystem services become more attainable (Walker *et al.*, 2004; Ernstson *et al.*, 2010; Biggs *et al.*, 2012). As previously stated, ecological resilience can be defined as the capacity of a system to absorb disturbances, re-organise itself while undergoing change and still maintain a similar function and structure (Holling, 1996; Folke, 2006). It is also an important concept which is increasingly used to underpin the understanding, managing, and governing of linked social-ecological systems (Folke *et al.*, 2004). Therefore, resilience can also be thought of as an approach, a way of thinking, that can guide and organise how we perceive systems and, more specifically, provide a valuable context for analysing the current state of social-ecological systems (Folke *et al.*, 2002; Folke, 2006). Furthermore, Gunderson and Holling (2002) developed the general theory of panarchy as a conceptual approach to understand the state of social-ecological systems (Walker *et al.*, 2006b). The panarchy recognises that social-ecological systems are comprised of structures and functions that exist across a range of spatial and temporal scales (Walker *et al.*, 2006a). These structures and functions exist within nested subset of hierarchies known as adaptive cycles – changes in structure and function over time due to internal dynamics and external forces (Holling, 2001; Walker *et al.*, 2006a). At the smaller, faster levels of a panarchy exists individual organisms, functioning within time frames of days, months and years. Further up the panarchy level are habitat patches functioning within a temporal scale of tens to hundreds of years. Then finally, at the larger, slower level are landscapes, functioning within a temporal scale of thousands of years (Holling, 2001). It is at this larger level where the conditions are set for the smaller, faster levels to function by, while at the same time it is also open to influence from those lower levels (Holling, 2001). A healthy system can undergo change instigated from the lower levels, while being protected from system collapse by the upper levels of the panarchy (Holling, 2001; Gunderson & Holling, 2002). Therefore, panarchical relations develop as top-down, bottom-up interactions occur (Gunderson & Holling, 2002). It is because of this panarchical relationship that the dynamics of a system cannot be fully understood without considering the influence structures and functions within a focal scale have on the scales above and below it (Walker *et al.*, 2006a).

The description of a social-ecological system can therefore be undertaken within a panarchy approach. Within this approach the landscape can be 'broken down' into different spatial and temporal scales so that the relationships between the components or variables within these differing scales can be analytically described. Furthermore, Janssen *et* al (2006) argue that, from a resilience perspective, the dynamics and heterogeneity in the structure of a system need to be made explicit. Those authors suggest that comparing landscape structures under different land management regimes will reveal insights into how the landscape components (i.e. habitat patches) differ in function over time. However, structural change should not only be assessed temporally, for example by comparing landscape connectivity before and after a major land-use change (e.g. Foltête *et al.*, 2014). Rather, the structure of a landscape also changes with perception, in regards to how we describe components within the landscape (e.g. by changing scales and dimensions) and how organisms perceive these components (Farina & Belgrano, 2006; Baguette & Van Dyck, 2007). Therefore, a system, when its structure is presented as a network, may reveal different insights into structural resilience under different perception thresholds. Urban and Keitt (2001) recognised that by incorporating threshold behaviour within the description of landscape graphs, different structural patterns occur. However, when creating their habitat maps, the authors assumed that they only existed in two dimensions. Therefore, although the authors did recognise patch heterogeneity in regards to forest density (ranging between 0 = no trees and 1 = closed canopy) other aspects of intra-patch heterogeneity were ignored (Urban & Keitt, 2001). As a result, other structural patterns were overlooked within their description of the landscape, patterns which have ecological implications (section 2.3).

A network perspective focuses on the components of a system, how they are structured, and how this structure can affect the performance and state of the system (Janssen *et al.*, 2006). Network analysis uses graph theory, an effective tool for the analysis of complex systems such as the social-ecological system arising from urbanisation (Zetterberg *et al.*, 2010). Within the graph theory approach a landscape and its intricate network of functional connections can be assessed by quantitatively describing it as a set of interconnected habitat patches (Ricotta *et al.*, 2000; Pascual-Hortal & Saura, 2006; Saura & Torne, 2009). This network of patches can be considered as a set of nodes (e.g. tree canopies) joined together by links (direct movement between canopies), both of which are surrounded by an inhospitable matrix (non-habitat) (Urban & Keitt, 2001; Saura & Torne, 2009; Zetterberg *et al.*, 2010). A path is a connected route from one node, through intermediary nodes, to another (Minor & Urban, 2008). A cycle forms when a path made up of three or more nodes becomes closed, so that the first node is re-visited (Urban & Keitt, 2001). When a path does not become closed then a tree forms, if a tree includes every node in the graph then it becomes a spanning tree (to avoid confusion with trees that exist in nature, the term 'graph trees' will from now on be used) (Urban & Keitt*, 2001).* A network component arises when sets of nodes are connected to each other (via paths) but remain separate from the rest of the nodes in the network (Minor & Urban, 2008). The network becomes compartmentalised when the components are composed of hub nodes (structurally important nodes which maintain network connectivity) that are not necessarily connected to each other (Minor & Urban, 2008). These areas of high connectivity represent landscape hubs (Melián & Bascompte, 2002; Minor & Urban, 2008). The structures exhibited within network components (whether they be cycles or graph trees) are sub-networks and highly compartmentalised networks may have several sub-networks (Maslov & Sneppen, 2002).

An understanding of the state of a system can be derived from the values of the variables that constitute the system (Walker *et al.*, 2004). Specifically, the structure or topology of a network can provide insights into the functioning of the system that it represents (Strogatz, 2001). In relation to the resilience of social-ecological systems, it is landscape structure and the structural change exhibited within a system (both temporally and spatially) which can provide insights into function (Janssen *et al.*, 2006). More specifically, it has been shown that connectivity and centrality are comprehensive variables of landscape structure that can be used to assess the function and resilience of a system when described as a network (Albert *et al.*, 2000; Janssen *et al.*, 2006; Rhodes *et al.*, 2006; Minor & Urban, 2008; Opsahl *et al.*, 2010; Biggs *et al.*, 2012).

The connectivity of a system depends on the ability of an organism to move between the nodes of a landscape graph (i.e. between tree canopy patches) and is structurally influenced by the levels of node accessibility and link density (Janssen *et al.*, 2006). However, connectivity is not only influenced by the spatial pattern of the nodes (habitat patches) within the landscape but also by the behavioural traits of the organism (Section 2.3). Therefore, a landscape graph needs to be created from functionally relevant data before a network analysis can take place. Centrality considers the amount of important nodes (i.e. keystone patches) within a system, which when removed would greatly affect the connectivity and resilience of the system (Albert *et al.*, 2000; Strogatz, 2001). Traditionally the centrality of a system focuses on the structural importance of links between nodes (Janssen *et al.*, 2006). A landscape graph with a high level of centrality will contain highly connected nodes (hubs) which are structurally important as they have a significantly higher amount of links than other nodes (Janssen *et al.,* 2006). However, this definition of centrality suggests that a landscape graph containing a hub node with several links is more connected than a landscape graph with one large node covering the whole landscape (as this graph will have no links). Therefore, paradoxically, to increase centrality within a landscape means fragmentation would have to occur.

Within the thesis, as to avoid such an illogical assumption, centrality is defined as the amount of important nodes within the landscape graph for maintaining connectivity. The level of importance should therefore be calculated not only by the amount of links attached to the node, but the area of the node itself – thus following the habitat availability approach to landscape connectivity (Section 2.3). Therefore, hub nodes are defined as nodes which are important for maintaining connectivity, calculated by its location within the landscape graph as well as its inherent attributes (i.e. area). By identifying, conserving, and maintaining these highly connected keystone patches (represented as hub nodes) landscape managers and decisions makers can contribute to the sustained and resilient provision of ecosystem services through a resilient system (Janssen *et al.*, 2006). However, behavioural properties such as organismal perception and movement capabilities can strongly influence the values of these structural variables (Urban & Keitt, 2001; Crooks & Sanjayan, 2006a; Farina & Belgrano, 2006) – a relationship often overlooked in the network analysis of system resilience (e.g. Albert *et al.*, 2000; Janssen *et al.*, 2006). In addition, while the graph theory approach to landscape connectivity literature demonstrates this relationship, the use of network analysis has reduced the landscape to two dimensions, thus ignoring other structural patterns existing in the third dimension (e.g. Bunn *et al.*, 2000; Ricotta *et al.*, 2000; Urban & Keitt, 2001; Rhodes *et al.*, 2006; Minor & Urban, 2008; Biggs *et al.*, 2012).

The movement and foraging behaviour of urban birds contributes to ecosystem function and resilience as well as the provision of several important ecosystem services (Lundberg & Moberg, 2003; Wenny *et al.*, 2011) (Table 2-1). The amount of trees, the variation in tree age and height (aspects of structural complexity), and the overall connectivity of the UTI have been shown to positively influence bird species richness and can increase urban landscape connectivity (Fernández-Juricic, 2000, 2001a; Fernández-Juricic & Jokimäki, 2001). However, gaps in canopy cover may be perceived as inhospitable by forest birds if they are more detectable by predators there (Rodríguez *et al.*, 2001) and thus gaps may restrict normal daily movements (MacIntosh *et al.*, 2011). Yet, it must also be kept in mind that no landscape is inherently fragmented or connected, the level of connectivity can only be assessed in the context of an organism's willingness (Harris & Reed, 2002) or rather capability to cross gaps between patches and the scale at which the organism interacts with the landscape (e.g. D'Eon *et al.*, 2002; Andersson, 2006; Baguette & Van Dyck, 2007). Therefore, the landscape is not only a static, physical, construct; it changes depending on how it is perceived (Farina & Belgrano, 2006). The capability of movement can change temporally and behaviourally and thus the variables of connectivity and centrality will also change, a relationship often excluded from previous network studies.

By considering the UTI under changing dimensional scales, different structural networks may be revealed and therefore different insights into the resilience of the UTI. At the same time, landscape connectivity, although important to system resilience, is only effective when considering functional connectivity. The capability of forest passerines to cross tree canopy gaps (a representative aspect of organism perception of the environment) varies between species; being shown to be between 30m and 200m (section 2.5). By including and changing the gap crossing capability threshold within a network analysis of the UTI, it should be possible to reveal different patterns in the structural composition of the UTI. This may subsequently uncover further insights into the resilience of the UTI.

A network approach to assess the resilience an UTI should therefore consider an organisms capability of movement across habitat gaps, how this capability affects the values of key structural variables (connectivity and centrality), and consider the UTI network in three dimensions by including tree height data. In order to achieve this goal a critical analysis of organism movement in general and passerine gap-crossing capabilities in specific is presented in section 2.5.

2.5 Passerines and Connectivity – movement through the urban fabric

Dispersal and gap-crossing ability are of critical ecological and evolutionary importance as they support the utilisation of multiple small habitat patches, gene flow among populations, and the colonisation of patches which have become vacant (Paradis *et al.*, 1998; Tremblay & St. Clair, 2009). Landscape connectivity facilitates such movement and strongly influences the persistence of metapopulations (Crooks & Sanjayan, 2006a) – a set of local subpopulations dispersed throughout a larger area and linked together by occasional dispersal and immigration (Hanski, 1999). It has been reported that in highly fragmented landscapes organisms moving between habitat patches undertake faster and straighter trajectories compared to the slow, indirect trajectories associated with foraging movement within habitat patches (Baguette & Van Dyck, 2007). These straighter movements provide the best solution when energy resources are limited and moving individuals incur predation risks (Zolliner & Lima, 1999).

The connectivity of discrete patches, or rather patch connectivity (Tischendorf & Fahrig, 2001) depends on the ability of an organism to disperse throughout the network of patches, subsequently sustaining migration, colonisation, and gene flow (Crooks & Sanjayan, 2006a). Patch connectivity works on the same underlying process as landscape connectivity – the facilitation or impediment of movement across landscapes – yet on a different spatial scale (Tischendorf & Fahrig, 2001). This explains the major difference between the two disciplines of landscape ecology and metapopulation ecology, the former considers connectivity as an attribute of an entire landscape, whereas the latter understands connectivity as an attribute of a habitat patch (Moilanen & Hanski, 2001). However, a graph-theoretic perspective can be applied to understand connectivity at a landscape scale or applied to localised ecological fluxes, such as those concerned with metapopulation ecology (Van Langevelde & Van der Knaap, 1998; Urban & Keitt, 2001). Zetterberg *et al*. (2010) further highlighted the strength of graph theory to link metapopulation ecology and landscape ecology scale investigations via a life-cycle based approach and the metapatch concept (Figure 2-5).

Similar to the idea that smaller sub-populations form together to create a metapopulation, when each habitat patch within the available landscape becomes linked together by an organism's ability to move, the accessible habitat patches, the links between the habitat patches, and the traversable regions between the habitat patches eventually come together and form a metapatch (Zetterberg *et al.*, 2010). The size and type of a metapatch also varies on a temporal scale so that its functional definition depends on the movement type being considered and the time frame within which this movement type occurs (Theobald, 2006; Zetterberg *et al.*, 2010) (Figure 2-5). What does not change however is that movement has to occur at each spatial and temporal scale.

The metapatch concept is similar to graph theory's component, or connected region. A component is a set of nodes (habitat patches) in which a path (a route connected by links so that no node is visited more than once) exists between every pair of nodes (Pascual-Hortal & Saura, 2006). Therefore, there is no functional relation between nodes belonging to different components and a component itself could be considered as a functional patch (containing habitat patches and the links between these patches created via direct movement).

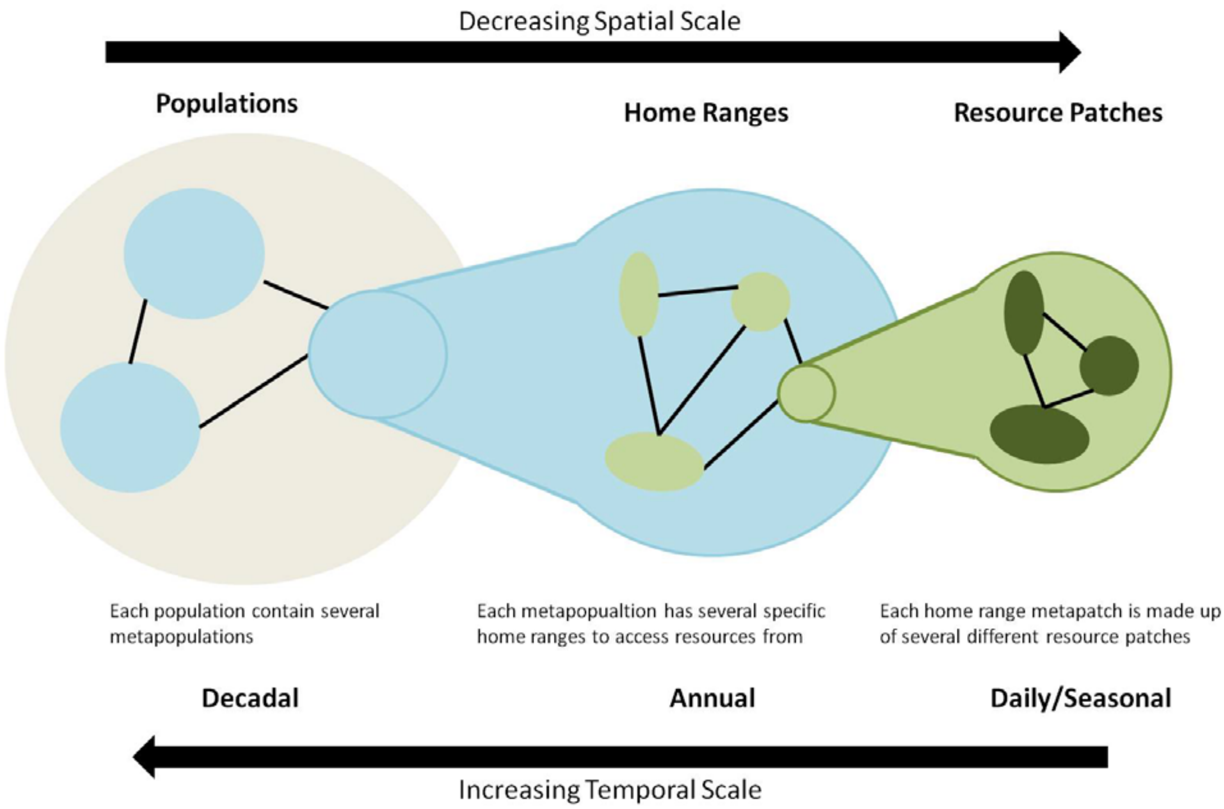

Figure 2-5: Metapatch concept.
The size and type of resource patches change on a spatial as well as on a temporal scale. Populations contain several metapopulations which exist within a landscape. Each metapopulation access resources by moving throughout a home-range and each home-range contain several resource/habitat patches. Movement between populations happens at a greater temporal scale compared to movement between resource patches within a home-range. A metapatch is created by describing not only the scattered distribution of vegetation cover but also the areas in-between the vegetation cover that are moved within when crossing gaps (from Zetterberg, *et al.,* 2010).

Interactions between organisms and landscapes should depend on whether or not the grain of habitat patches match the spatial scale of the perceptual range (Baguette & Van Dyck, 2007). The grain size is the smallest spatial scale at which an organism recognises spatial heterogeneity according to its perceptual range, which is the basic limitation of animal perception (Wiens, 1989; Kotliar & Wiens, 1990; Lima & Zollner, 1996). Baguette and Van Dyck (2007) argue that when the landscape grain is smaller than the perceptual range of the individual, there is no real difference between movements within and between habitats. On the other hand, if the grain of resources is larger than the perceptual range of the individual, time outside of habitat increases, as do predation or other mortality risks, and therefore dispersal has a higher cost. According to this hypothesis, the spatial scale determining the functional landscape grain depends on the perceptual range of the individual. The perceptual range of the individual may be restricted by a variety of factors, such as vision, olfaction, body size/energy reserves, environmental conditions, and/or predation risk (Zollner & Lima, 2005). The trade-off between perception and predation has been shown to be sensitive to the search strategy employed by a moving individual so that, in summary, a moving organism with low energy reserves and employing start/stop type movement would benefit from dispersal when predation risk is high and they have a high perceptual range, while the opposite is true for organisms moving straighter and faster and with high energy reserves (Zollner & Lima, 2005). However, the results come from a simulated study using a general model and actual relationships between perception and predation are rare and highly species specific. Without knowing the specific predator population of a landscape or the amount of risk associated with movement, it may be more beneficial to assume direct movement within a perceptual range.

Dispersal distances and habitat gap-crossing capabilities/probabilities can be used to quantify and understand the potential perceptual threshold of an organism (St. Clair *et al.,* 1998; Smith *et al.,* 2013). Long–distance dispersal (i.e. as exhibited in breeding and post-natal dispersal) across a landscape has important consequences for the spatial and genetic structuring of populations (Ibrahim *et al.*, 1995; Paradis *et al.*, 1998; Dolman *et al.*, 2007b), as such species specific dispersal distances and thresholds are needed to be known. In this study it is assumed dispersal has already taken place and that for an organism to survive post-dispersal, both resource supplementation and complementation are of importance. To maintain home-ranges and to successfully forage in an urban environment organisms are likely to cross gaps in habitat patches at a local scale. In this case, the willingness to cross gaps of various sizes between available habitat patches is important. Therefore, gap-crossing ability will be incorporated into the connectivity analysis within this thesis. That is to say, within this thesis the home-range/resource patch scale of connectivity is focused on (Figure 2-5).

Passerine birds, and their gap-crossing abilities, have been chosen as the focal organism of this study. This decision was made as the Passeriformes include about 60 per cent of all living bird species, are extremely diverse and conspicuous, yet all exploit the same ecological niche of the small, perching bird (Tudge, 2000). Furthermore, many urban passerines provide an important cultural ecosystem service – bird song (Winterman, 2013), and there are several available studies providing passerine gap-crossing ability data. However, there are no studies which explicitly provide gap-crossing capability for British, urban passerines. Therefore, the following studies analysing gap-crossing capability within various environments in North America and Europe will be used as proxies.

Tremblay and St. Clair (2009) investigated the permeability of an urban environment for songbirds (another term for passerines). The study took place in the city of Calgary, Canada and used mobbing calls as a lure to test the willingness of forest songbirds to cross linear features within the city (Tremblay & St. Clair, 2009). Species tested included the black-capped chickadee (*Poecile atricapillus*), red-breasted nuthatch (*Sitta canadensis*), white-breasted nuthatch (*S. carolinensis*) and downy woodpecker (*Dryobates pubescens* - not a Passiforme). The types of urban linear features were roads, railways, transportation bridges, and rivers. The results showed that gap size was the most important determinant of movement and, more specifically, if gaps exceeded 30m the likelihood of movement started to decrease dramatically and by 45m birds were half as likely to cross a gap. By 75m birds were less than 10 per cent likely to move across a gap in the canopy cover (Tremblay & St. Clair, 2009).

Desrochers and Hannon (1997) also undertook a study of gap-crossing decisions made by forest songbirds in Canada. However, the study took place near Quebec City, within an agricultural and forested landscape. Playbacks of mobbing calls were again used to lure birds across gaps of various sizes within the canopy cover. The five species included in the study were the black-capped chickadee, red-breasted nuthatch, red-eyed vireos (*Vireo olivaceus*), golden-crowned kinglet (*Regulus satrapa*), and yellow-rumped warbler

(*Dendroica oronate*). Overall, the study showed that the birds strongly preferred to move through woodland than in the open when given the choice. Thus, it can be concluded that woodland links significantly facilitate the movement of birds in fragmented landscapes. In particular, the study found that the birds were willing to cross gaps up to 30m while at 70m the probability of gap-crossing reduced to 30% and then 10% at 100m (Desrochers & Hannon, 1997).

St. Clair *et al.* (1998) also studied the movement of forest song birds near Quebec City, as well as in north central Alberta, Canada. The willingness of the black-capped chickadee, white – breasted nuthatch, downy woodpecker and hairy woodpecker (*Leuconotopicus villosus* – not a Passiforme) to cross various gaps in forest cover was assessed. The results showed that the study birds would use forested detours over short distances, keep within 25m of the forest edge, and were increasingly less likely to cross gaps as distance increased to 200m (St. Clair *et al.*, 1998). Further analysis was undertaken on the black-capped chickadee, resulting in a maximum gap-crossing capability threshold of 200m (20 per cent likely to cross) if there was no forested alternative, while a gap of 25m had little effect on movement (90 per cent likely to cross). Similarly, Bélisle & Desrochers (2002) concluded that their study birds wouldn't stray 25m from the forest edge.

The gap-crossing decisions of forest songbirds within the forests of West Scotland were assessed by Creegan and Osborne (2005). The authors conducted gap-crossing experiments with and without mobbing call playbacks as a lure. Species included in the study were the common chaffinch (*Fringilla coelebs*), European robin (*Erithacus rubecula*), coal tit (*Periparus ater*), and goldcrest (*Regulus regulus*). Maximum gap-crossing capability thresholds were obtained for each of the study species and each species was less than 5 per cent likely to cross its corresponding maximum gap-crossing capability threshold (Table 2-3).

Table 2-3: Maximum gap-crossing capability thresholds exhibited by a sample of UK passerines (Creegan & Osborne, 2005).

	Non-Playback	Playback using song thrush (*Turdus philomelos*) mobbing calls
Species	**Maximum Distance (m)**	**Maximum Distance (m)**
Goldcrest (*R. Regulus*)	**No gap-crossing recorded**	**46 (p = 0.05)**
Robin (*E. Rubecula*)	**50 (p = 0.05)**	**60 (p = 0.05)**
Coal Tit (*P. Ater*)	**50 (p = 0.05)**	**92 (p = 0.05)**
Chaffinch (*F. Coelebs*)	**120 (p = 0.05)**	**150 (p = 0.05)**

150m was the maximum gap-crossing capability threshold exhibited by a sample of UK passerines (chaffinch). However, the willingness to actually cross such gaps is low (p = 0.05). Therefore, the chance that each of the surveyed species would actually cross their associated maximum gap distance is 5%.

Gap-crossing abilities of forest specialists and habitat generalists within a mature forest in Canada were assessed by Rail *et al.* (1997). Again the authors used playback trails to determine if passerines would cross treeless gaps (control tests used playbacks within contiguous forest). The aim of the study was to identify if there were gap-crossing thresholds presented by the study passerines, after which the probability of crossing the gap becomes greatly affected. Passerines included in the study were Swainson's thrush (*Catharus ustulats*), golden-crowned kinglet, black throated green warbler (*D. virens*) (Forest specialists), white-throated sparrow (*Zonatrichia albicollis*) and dark-eyed junco (*Junco hyemalis*) (habitat generalists). Gaps within the woodland were caused by hiking or skiing trails, unpaved roads, power-lines, and clear-cuts (vegetation within the gaps was less than 1.5m high). Results of the study revealed that the probability of forest specialists crossing gaps in the canopy was most abruptly affected (negatively) when gaps reached between 25 and 40m. The habitat generalists on the other hand revealed a gap-crossing probability threshold of 65-70m for the white-throated sparrows, while the dark-eyed juncos' response to the playback trails did not seem to vary with gap-crossing capability threshold.

Robertson & Radford (2009) also undertook playback trails to determine willingness to cross gaps in forest cover within southern Victoria, Australia. However, unlike Rail *et al.* (1997) two types of forest gaps were included in the study: cleared gaps containing 'paddock' or scattered trees and cleared gaps with field vegetation (i.e. no trees). Both of the study species, the grey shrike-thrush (*Colluricincla harmonica*) and white-throated treecreeper (*Cormobates leucopaeus*), were affected by canopy gaps. Furthermore, Robertson and Radford (2009) demonstrated that both species were more likely to move through continuous forest then cross a gap of the same distance. Gap-tolerance thresholds (i.e. the willingness to cross gaps abruptly reduced after a given distance) were found to be 85m for the grey shrike-thrush and 65m for the white-throated treecreeper. However, these thresholds were only indentified within the cleared gaps containing field vegetation study areas. When scattered trees were present grey shrike-thrushes would cross distances of up to 260m.

In a North American agriculturally dominated landscape with forest fragments Grubb & Doherty (1999) observed the gap-crossing probability of various birds (including non-passerines). The authors used transects and directly observed gap-crossing without the use of playback trails. The results revealed that the majority of gap-crossing was between 50 and 200m for the passerine species song sparrow (*Melospiza melodia*), Carolina chickadee (*Poecile carolinensis*), tufted titmouse (*Baeolophus bicolour*), white-breasted nuthatches (*S. carolinensis*) and northern cardinals (*Cardinalis cardinalis*). The maximum gap crossed was undertaken by the red-bellied woodpecker (*Melanerpes carolinus*) which is within the order Piciformes rather than Passeriformes (i.e. not a passerine). The study also established that birds were more likely to move from small woodland fragments than larger ones and if the fragment had a large perimeter or extent of shrub cover. Furthermore, larger birds were found to be more likely to cross gaps than smaller birds. In a similar North American landscape MacIntosh *et al.* (2011) used radio telemetry to study the gap-crossing behaviour of wood thrushes (*Hylocichla mustelina*). Their results revealed that most forays were greater than 150m in distance and that foray rate to adjacent woodland fragments decline with the increase in gap width. In addition, male wood thrushes would typically cross gaps less than 100m wide (14 out of 20 forays), however the maximum gap was recorded at 615m. For female wood thrushes the majority of gaps crossed were less than 200m (3 out of 4) while the maximum gap was 300m.

In a South American (Brazil) agricultural landscape with fragmented woodland Awade and Metzger (2008) studied the gap-crossing probabilities of the variable antshrike

(*Thamnophilus caerulescnes)* and golden-crowned warbler (*Basileuterus culicivorus*); both forest dependent birds. Again, playbacks were used to simulate territorial invasion and provoke gap-crossing. For both species gap-crossing probability decreased as distance increased. Specifically, both species exhibited a 50% change of crossing a 40m gap and this probability dropped to 10% at 60m for the warbler and 80m for the variable antshrike. The warbler also exhibited a gap-crossing probability threshold between 25 and 55m, after which the gap-crossing probability abruptly dropped. No such threshold was identified for the variable antshrike.

Finally, Hinsley (2000) used a theoretical model to investigate the time and energy costs of a female great tit (*Parus major*) in crossing gaps between habitat patches. An estimated daily energy expenditure (DEE) was used to ascertain the likelihood of crossing various gaps under two lifestyle criteria; foraging involving low amounts of gap-crossing and foraging involving larger brood size and/or proportion of gap-crossing. When gap-crossing was low, the model proposed that gaps of between 300-550m could be crossed without exceeding the maximum DEE of the great tit. However, when foraging trips involved/required higher amounts of gap-crossing (i.e. in a fragmented landscape) potential gap-crossing capability threshold decreased to between 50-100m.

2.6 Conclusion

As a result of the critical literature review presented above a set of research gaps have been identified. Namely, these are: 1) previous assessments of potential landscape connectivity only consider patches of tree cover in two dimensions, disregarding vertical structure; 2) network analysis of ecological system resilience overlook the importance of organismal perception and movement capabilities in the quantification of critical structural variables such as connectivity and centrality; 3) studies into the relationship between UTI landscape patterns and societal deprivation patterns only consider UTI landscape composition and not configuration or vertical structure.

It has been established that a functionally connected UTI contributes to ecosystem function, resilience, and the provision of ecosystem Services. However, to effectively quantify UTI landscape connectivity, models should integrate the various dimensional scales present in nature– spatial, perceptual, and temporal. Specifically, there has been a distinct lack in the use of vertical scales in the analysis of connectivity. It is therefore proposed that tree height, along with passerine perception (through gap-crossing ability), is to be integrated into the connectivity models used in this thesis. To understand how resilient a UTI is, in regards to providing continual levels of connectivity, model results should also be compared over time – thus recognising the temporal dimension. Further insights into the resilience of the UTI can then be gleaned from identifying critical components in the landscape for maintaining connectivity alongside a network analysis of the UTI. By including changing vertical and perceptual dimensions (via movement, or rather gap crossing capability) a holistic analysis of UTI resilience is made possible – compared to two dimensional, single distance threshold network analyses of landscapes. Understanding how the UTI is structured, with regards to both landscape composition and configuration, in relation to societal structure can inform a targeted UTI management approach. Until now only the composition (i.e. amount of tree cover) has been related to social structure. By including UTI landscape configuration and vertical structure in the exploration of social and UTI landscape pattern relationships, the thesis research contributes towards the ways in which we understand the social-ecological nature of UTIs.

Within the following chapter a set of research aims and objectives are presented – in order to address the emerging research issues highlighted above (3.1). In section 3.2 the study areas within the City of Salford are described. The general methods employed throughout the thesis (i.e. used within all the research chapters) are presented in section 3.3, 3.4, and 3.5.

Chapter 3 : Study aims, Study Area, Sample Plots & General Methods

3.1 Aims and Objectives

The research in this thesis makes a contribution to knowledge by describing, in new ways, the structure of a UTI and then to understand the emerging patterns within this structure in relation to UTI function, UTI system resilience, and societal structure. The study UTI for the research presented in this thesis existed within the City of Salford, UK, in the years 2005, 2009, and 2013/14. However, the outputs of this research will have relevance to other cities as the methods used are transferable. Research has been undertaken using an inductive, explorative approach. Inductive research investigates empirical evidence using specific data to reach conclusions and develop new theories (Grix, 2004). This involves seeking out patterns within the data without the need of a guiding hypothesis (Grix, 2004). However, clear links need to be established between the research objectives and the findings from the raw data. These links can be subsequently used to develop a model or theory about the underlying structure or processes evident in the data (Thomas, 2003). A diagrammatical representation of the inductive research approach within this thesis is in Figure 3-1.

The research aim is to critically evaluate the vertical and horizontal structural patterns of a UTI and to understand how these emerging patterns relate to UTI functionality, system resilience, and societal structure (Figure 3-1). In order to accomplish the research aim four research questions were addressed: (1) How is Salford's UTI structured in terms of the size and shape of tree canopy patches? (2) What is the change in potential landscape connectivity of Salford's three dimensional UTI for passerines over time and space? (3) How resilient is Salford's UTI network, in regards to providing functional levels of connected habitat? (4) What is the relationship between societal metric patterns and structural UTI patterns in Salford? Each research question has a related set of emerging objectives, the response to which may influence the response to the following research question/objective. Specifically, the objectives of question 1 need to be completed before any other research question can be addressed. Objectives 1.2 and 1.3 will generate data for questions 2 and 4.

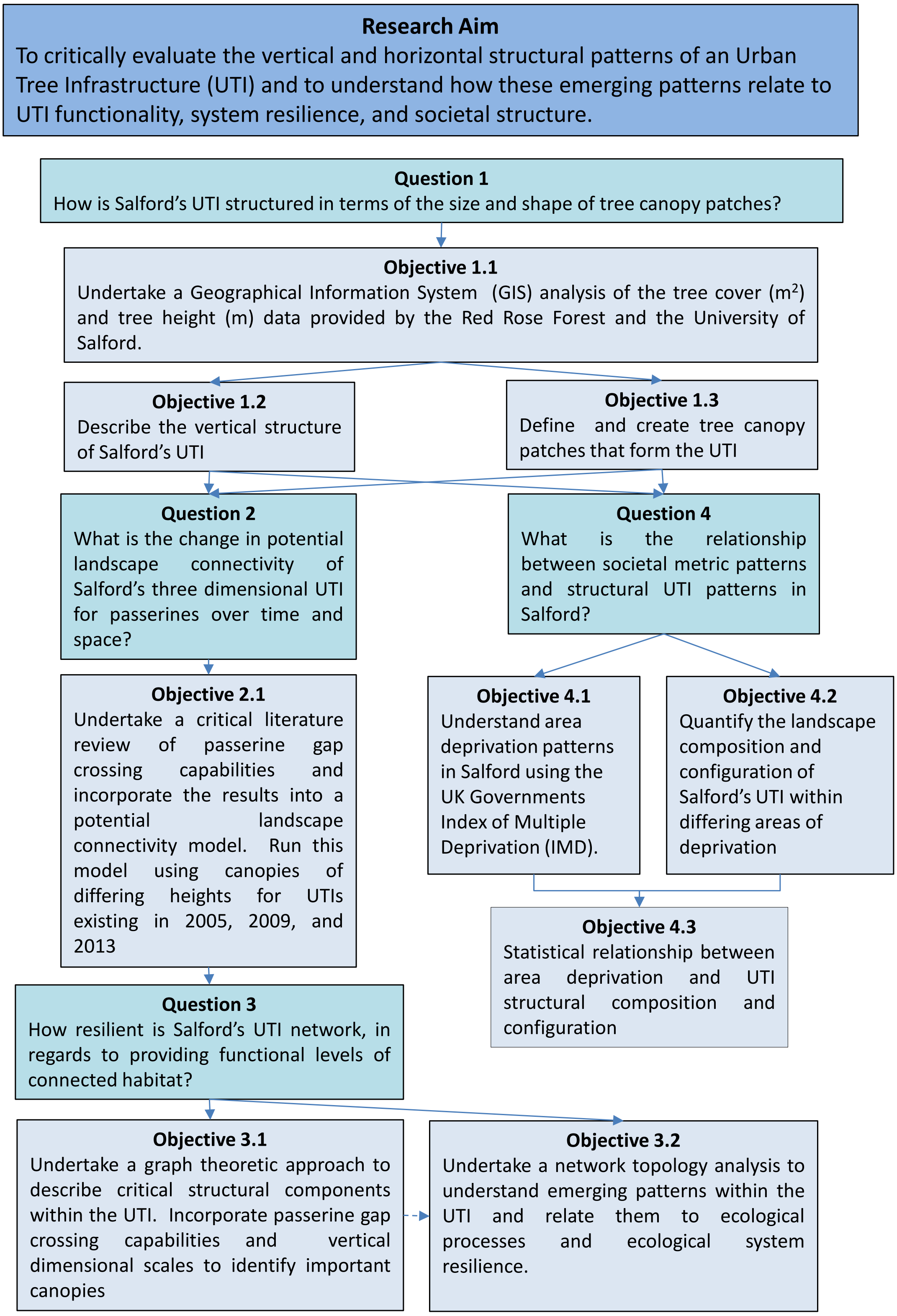

Figure 3-1: A diagrammatic representation of the project aims and objectives.

In order to complete the research aim a set of four questions must be answered. The results of question 1 will inform the response to all subsequent questions. Question 2 will provide key data and therefore strongly influence the outcome of question 3. Question 4 does not influence and is not influenced by questions 2 and 3. Responses from the completed objectives will be related to the research questions and the overall research aim. The formation of new theories and further questions should arise through the analyses and discussion of the objective results.

An explanation of the of the original, raw tree canopy data used to answer research questions 1 to 4 is presented in section 3.2. A description of the City of Salford and the various study areas used within the thesis research are provided in section 3.3. Before the tree canopy data within these study areas could be used they were taken through a series of transformations as described in sections 3.4 and 3.5. Finally the software used to implement a graph theoretic approach to quantify landscape connectivity and describe landscape networks are described in section 3.6.

3.2 Tree Canopy Data

In 2007 Red Rose Forest, one of the twelve community forests in the UK, responsible for developing well-wooded, multipurpose landscapes within central and western Greater Manchester (Red Rose Forest, 2012), commissioned a tree audit and canopy survey of Greater Manchester (Red Rose Forest, 2008; TDAG, 2012). The tree canopy survey was undertaken by ecoscape, led by Chris Senior, and used 2005/6 aerial photography to map trees above 3m with canopies larger than 1.5m (referred to as the 2005 data) (Red Rose Forest, 2008). The Manchester tree audit proved to be so successful (as well as cost effective) that Salford City Council commissioned Red Rose Forest to undertake a similar study within Salford (TDAG, 2012). In 2009/11 (referred to as the 2009 data) another aerial survey was flown, mapping further information about the trees of Salford. Consultants BlueSky conducted the survey using the tool ProximiTREE™, which uses stereo imagery and a Digital Terrain Model (DTM) to accurately locate the position, height, and canopy size of trees (BlueSky, 2012). Red Rose Forest provided the vector polygons representative of both 2005 and 2009 trees within the City of Salford.

The 2005 data did not contain information on canopy height. Furthermore, on reviewing the data, there seemed to be no intention of identifying individual canopies. Instead, identified tree cover within the aerial photography were 'drawn' around – meaning canopies were considered as homogenous components of the landscape, represented as irregular polygons within a GIS. The 2009 data, on the other hand, contained information on canopy height and did attempt to identify individual canopies which were represented as circular polygons. Where individual canopies could not be identified, tree cover was kept homogenous, represented by irregular polygons termed as *tree lines*. Yet, these tree lines still had height data associated with them, as height was calculated, horizontally, every 20m.

The 2013/14 data (referred to as the 2013 data) was creating using the 2009 canopy footprint and field surveys (the methods of which are discussed in detail within Chapter 4). The 2005, 2009, and 2013 tree canopy data were used to map changes in tree canopy cover. Subsequently, the differences in potential landscape connectivity exhibited by the different levels of tree canopy cover could then be assessed (Research Question 2, Figure 3-1). The 2009 tree canopy data were used to assess UTI system resilience (Question 3, Figure 3-1) and also in the exploration of the relationship between societal metrics and UTI landscape patterns (Question 4, Figure 3-1).

3.3 Study Areas and the City of Salford

The whole of the City of Salford was used as a study landscape as well as various smaller scale study areas within the more urban regions of the city (scale of study area changes with the research question and/or objective being assessed). The City of Salford is located within Greater Manchester, North West of England, and covers a large western part of the Manchester conurbation (Figure 3-2). This conurbation can be characterised by the presence of dense urban and industrial development, commercial, financial, retail and administrative centres, commuter suburbs and residential areas, interspersed with a network of green infrastructure (Natural England, 2013). The resilience of these green areas will be challenged as development pressures within the conurbation begin to increase. These pressures are underpinned by the drive towards economic growth which will require further built infrastructure and associated services (Natural England, 2013). Salford itself suffers from significant deprivation with particular problems in regards to obsolete housing, derelict and underused land and buildings, and poor environmental quality (Salford City Council, 2006, 2007, 2009) (Table 3-1). In addition to the complex mosaic of urban areas at various stages of regeneration, Salford also contains rural, urban fringe, and green belt areas (Salford City Council, 2006, 2007), reflecting the heterogeneous complexity and ecologically fragmented characteristics of urbanised environments (Table 3-1). Table 3-1 contains a brief introduction to the landscape characteristics and key features of the rural, urban fringe, and urban areas of Salford.

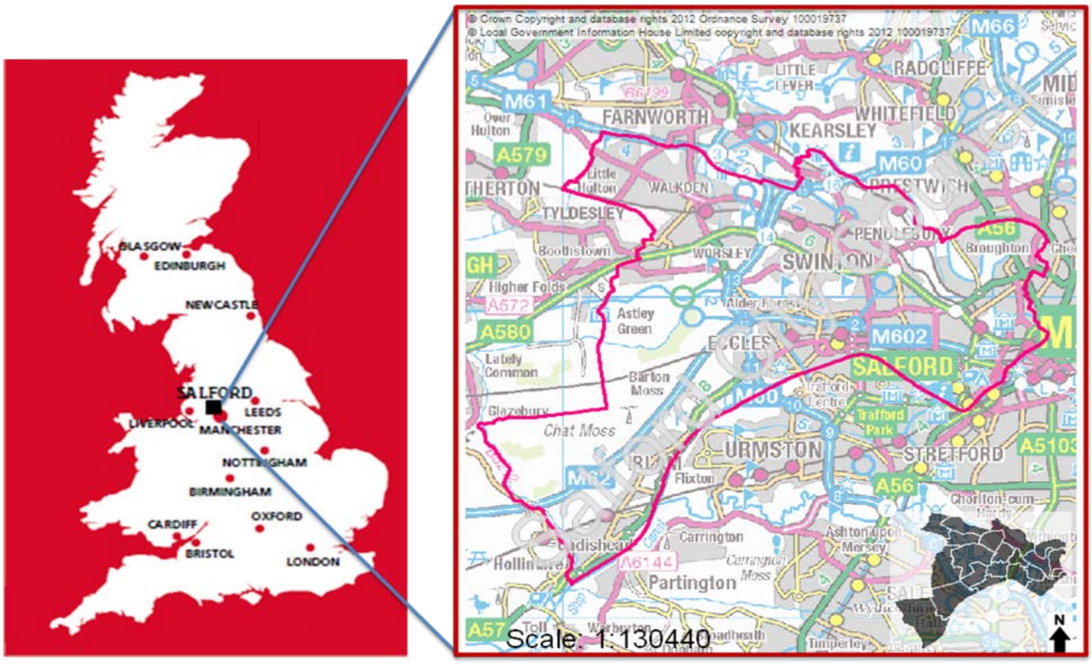

Figure 3-2: The location of Salford within the North West of England.
Salford exists within the Greater Manchester conurbation, characterised by dense urban and industrial development, commercial and residential centres, and a network of green infrastructure. Salford also suffers from high levels of deprivation and the urban areas exhibit various levels of regeneration.

The whole of the Salford landscape (study landscape) was used to assess the relationship between societal metric patterns and UTI landscape structural patterns (Research Question 4, Figure 3-1). To assess changes in potential landscape connectivity over time and space (Research Question 2, figure 3-1) a smaller study area and four sample plots within the valley of the River Irwell were used (Figure 3-3 and 3-4). This same ‘river valley study area’ (but not the four sample plots) was used to assess UTI network resilience (Research Question 3, Figure 3-1).

Table 3-1: The City of Salford's landscape character and key landscape features.

Key Features	Landscape Character Classifications and Descriptions				
	Rural Mosslands	**Urban Mosslands**	**Urban Fridge Lowland**	**Urban River Valley**	**Urban Areas**
Topography	Low lying, low relief topography.	Low relief topography.	Lowland area crossed by small streams forming narrow valleys.	**Medium scale, U shaped valley.**	**Varied. The western half of the City of Salford is characterised by largely low-lying, low relief topography. The eastern half of the city is characterised by a south facing ridge running from the north west to the south east. Small streams have carved out narrow valleys running generally north to south.**
Land Use and Land Cover	Network of deep drainage ditches running alongside private roads and between large fields. Arable agricultural land use with large scale fields. Small patches of peat bogs. Some birch and scrub woodlands as well as planted woodland for forestry, game, public access and amenity value. Relatively little built environment. Railway line and canal.	M60, M61, and M62 Motorway interchanges. A pattern of large fields (some disused) and evidence of past mining activities. A small area of scrub and planted woodland containing a few hedges has reclaimed the mossland and a former colliery respectively. Small woodland and grassland areas create a local nature reserve. Disused farmlands now contain two ponds and a small area of planted woodland. Relatively little built environment.	Two major roads with interchanges and a railway line. Areas of farm land and arable disused farmland. Wooded slopes and narrow streams. Relatively large areas of woodland including wet woodland. Relatively large amounts of water cover including a lake, small ponds and a canal. Golf courses, areas of open land, and little built environment.	**Principally open land following the River Irwell. Poor quality farmland. Extensive areas of good quality woodland some of which have been planted over former landfill and sewage works. A large lake and areas of marsh/swamp. Areas of lowland heath. Unimproved acidic and neutral grassland and wildflower meadows. Encroachment of urban development (although there are few buildings on the valley floor). Major roads and railway line.**	**Transport infrastructure such as railway lines, the metrolink, and a canal. Large parts of central Salford suffer from significant levels of deprivation and there are particular land use/cover problems with regards to obsolete housing, derelict and underused land/buildings and poor environmental quality. Residential neighbourhoods containing older housing areas (some of which are of low quality and have a lack of open space) and newer housing areas (some of which are of poor design and layout). Large and older industrial areas which suffer from dereliction and traffic congestion. The City District shopping centres consist of relatively poor quality precinct developments; suffering from environmental problems such as vandalism, low standards of land and property maintenance, and traffic congestion.**
Vegetation	Trees, scrub, grasses, mosslands, hedges.	Trees, scrub, grasses, hedges.	Trees, grasses.	**Trees, reed beds, grasses, wild flowers.**	**Trees, garden vegetation.**
Water	Relatively little although there are wetter patches in areas where drainage is poor.	Ponds.	Narrow streams, small ponds, a relatively large lake and a canal.	**Lake, ponds, marshes, swamps, river and streams.**	**Canal.**

Table 3-1 presents a brief outline of Salford's landscape character. The sections in bold represent the areas of particular research interest and explain the topology, land use and land cover types, vegetation, and water cover that can be found within the river valley study area (Salford City Council, 2006, 2007, 2009)

The valley of the River Irwell which runs through Salford contains the greatest diversity of vegetation and land cover types within the city (Salford City Council, 2007). Therefore, the river valley is considered as a very important wildlife corridor, helping to bring biodiversity right into the centre of a major conurbation (Salford City Council, 2007). The river valley study area lies within the southern section of the valley, within the east of Salford, and consists of the Broughton, Kersal, and Irwell Riverside electoral wards (Figure 3-3). The southern section of the river valley study area represents part of the river Irwell's flood basin (Salford City Council, 2007) and contains highly urbanised areas and various types of tree cover (i.e. linear street trees, woodland patches, and singular, scattered trees). Therefore, the river valley study area is reasoned to be a suitable place to investigate the levels of UTI landscape connectivity across various heights and UTI network resilience.

To undertake an up-to-date tree inventory and to compare potential connectivity levels over time, four tree canopy sample plots (Figure 3-4) were selected within the river valley study area (Figure 3-3). These sample plots contained varying levels of tree cover, composition, and configuration (Figure 3-4). A circular window with a diameter of 500m was used to select these four sample plots located within north-eastern Kersal, southern Irwell Riverside, Higher Broughton (north Broughton), and Lower Kersal (south-western Kersal, Figure 3-3). The north-eastern Kersal sample plot (referred to as 'Kersal sample plot) is located in the least deprived area of Kersal (Salford City Council, 2008c) and exhibited around 42% tree canopy cover in 2009 (Figure 3-4). The sample plot within southern Irwell Riverside is located within Peel Park and the University of Salford's Peel Park Campus and therefore will be referred to as the 'Peel Park sample plot'. The Peel Park sample plot contained a high canopy cover in 2009 (33%) (Figure 3-4) and is within the 7-10% most deprived areas nationally (Salford City Council, 2008b). The 'Higher Broughton sample plot' is located within one of the most deprived areas of Broughton as well as within the 3-7% most deprived areas nationally (Salford City Council, 2008a). The Higher Broughton sample plot also contained 6% canopy cover in 2009 (Figure 3-4). The 'Lower Kersal sample plot' is located in the most deprived area of Kersal and is in the top 3% most deprived areas nationally (Salford City Council, 2008c). Furthermore, the Lower Kersal sample plot exhibited the lowest canopy cover value of around 3% in 2009 (Figure 3-4).

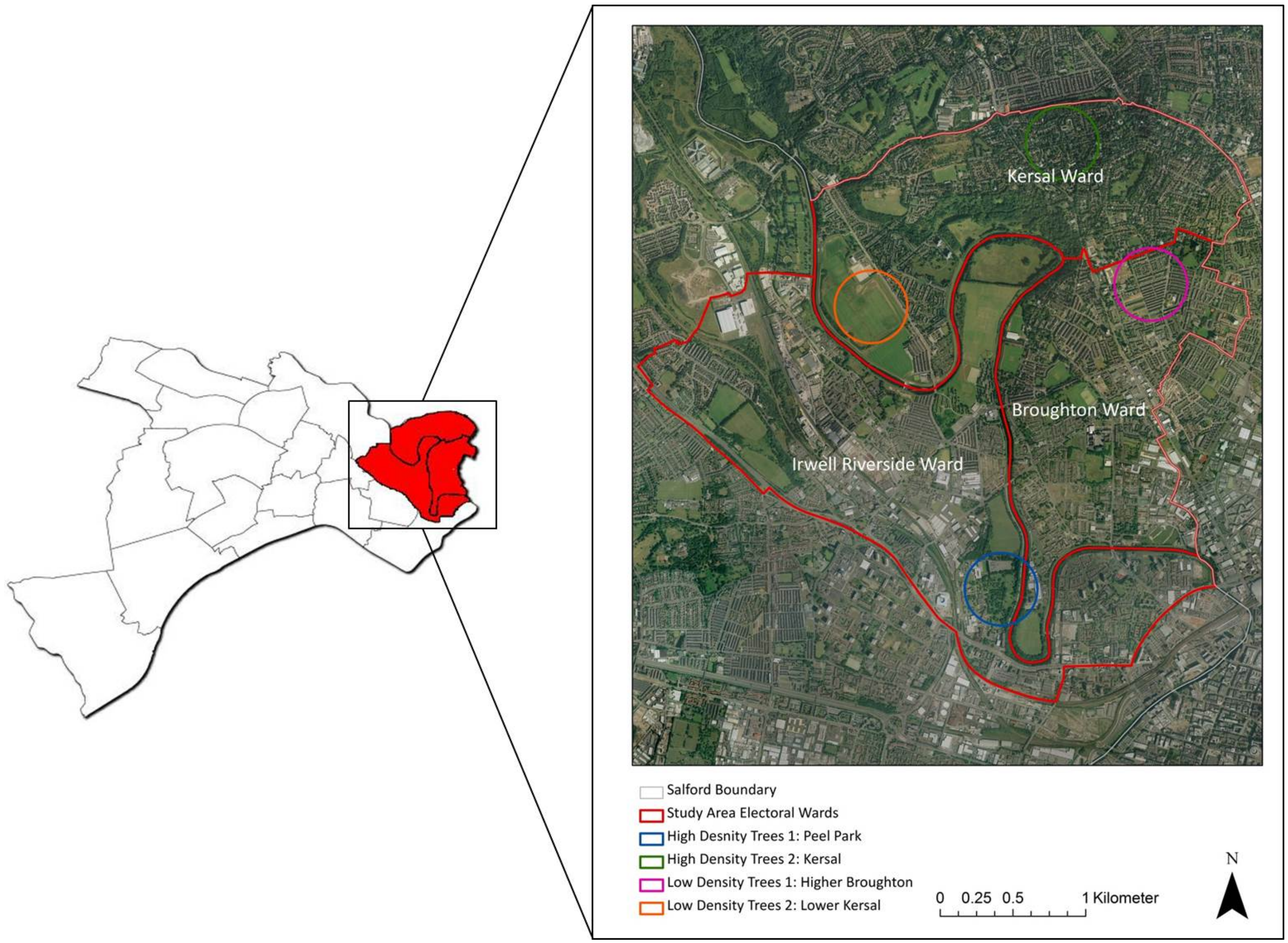

Figure 3-3: River valley study area in East Salford.

The river valley study area sits within the southern part of the Irwell River Valley, Salford. The boundary of the river valley study area is represented by a red line and contains the electoral wards of Kersal, Broughton, and Irwell Riverside. Four sample plots have also been selected within the river valley study area. The four tree canopy sample plots can be divided into two categories – high density canopy cover, and low density canopy cover. High density tree cover 1 (blue) is located primarily in Peel Park while high density tree cover 2 (green) exists in Kersal. Low density tree cover 1 (pink) can be found in Higher Broughton and low density tree cover 2 (orange) is located in Lower Kersal.

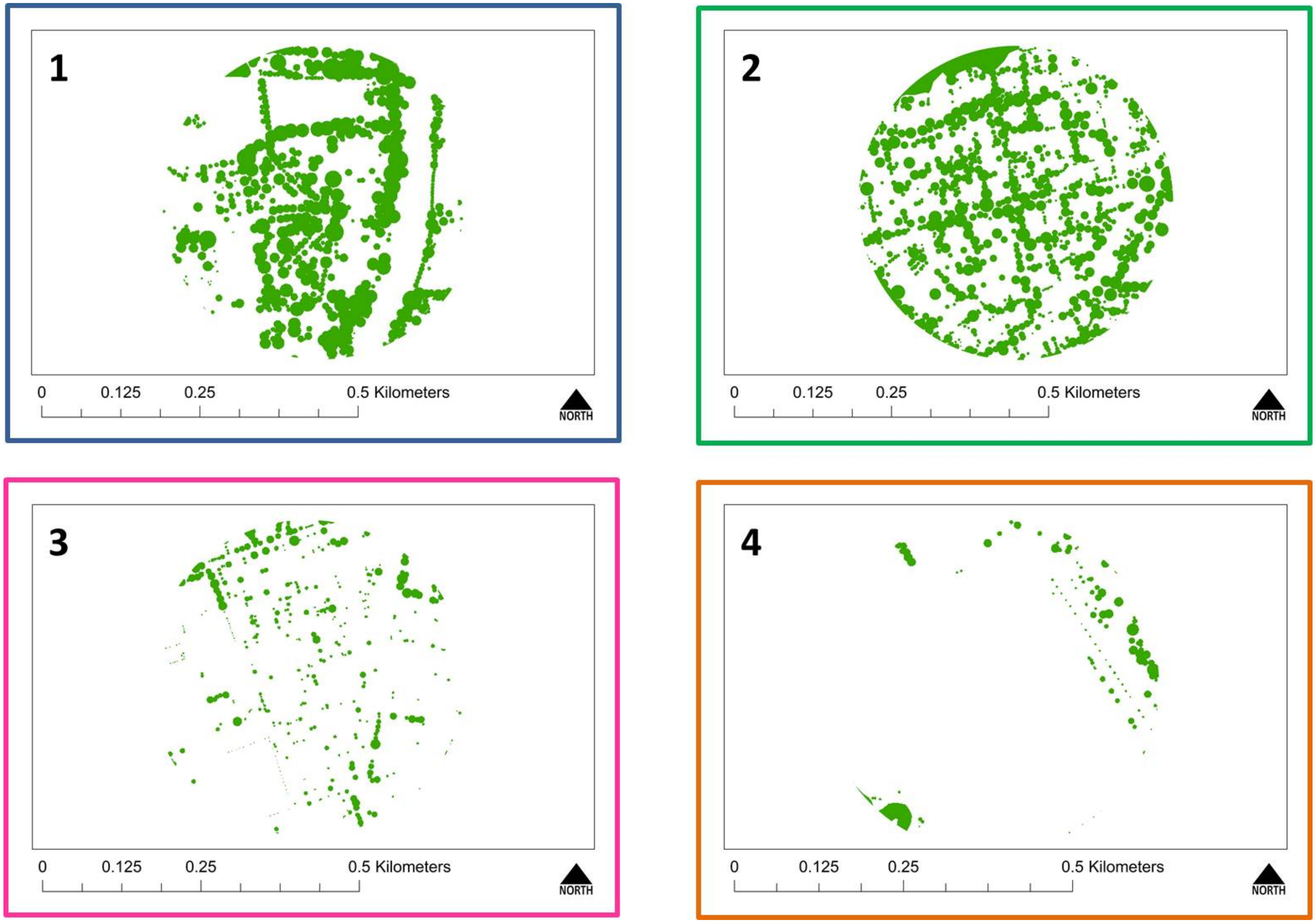

Figure 3-4: Tree Canopy Sample Plots.

Each tree canopy sample plot is within a 500m diameter circle and contains varying tree cover, composition and configuration. Sample plots 1(~33% tree cover, Peel Park) and 2 (~42% tree cover, Kersal) represent high canopy density landscapes. Sample plots 3 (~6% tree cover, Higher Broughton) and 4 (~3% tree cover, Lower Kersal) represent landscapes with relatively little amounts of tree cover. Tree cover percentage taken from 2009 data.

3.4 Vertical Stratification

3.4.1 Interpolating height data

Within the 2009 data, individual polygons (representing tree canopies) contained a location, height (m), and canopy area (m^2) measurement. Where individuals trees could not be identified (i.e. tree lines) height measurements were calculated at 20m, horizontal, intervals (Figure 3-5). To gain a representation of the 'vertical shape' of these tree lines, the spatial analyst tool 'Interpolation' was used (Figure 3-6). Specifically, 'inverse distance weighted' (IDW) interpolation was calculated within ESRI ArcGIS's ArcMap, version 9.3. IDW interpolation works on the assumption that objects that are spatially close to one another are more alike than those that are farther apart (Watson & Philip, 1985). Therefore, to predict the most likely value for an unmeasured location IDW considers the values surrounding the prediction location. The values under consideration here come from a point data set where height has been measured every 20m (Figure 3-5) and therefore treelines were effectively made up of 20x20m cells. These sample points were considered sufficiently dense and evenly spread enough to simulate the vertical shape of the treelines (Watson & Philip, 1985).

To be able to use IDW interpolation, treeline polygons were first converted to a raster data set of 1x1m cells. Therefore, the 20m sample points (which could be considered as 20x20m cells) were divided into twenty 1x1m cells so that a clearer description of the vertical shape of the treelines could be achieved (Figure 3-6). Based on the measurement of the 8 nearest neighbour cells, 1x1m cells within the treelines were assigned a value relating to height (m) (determined by its proximity to an original sample point height). This provided treelines with a new spatial structure which incorporated height (m). As a result, post interpolated treelines, as well as individual tree canopy polygons, could be stratified into canopy height classes in order that tree canopy connectivity could be assessed vertically as well as horizontally (section 3.4.2). Interpolation of treelines was only necessary for the 2009 tree canopy data.

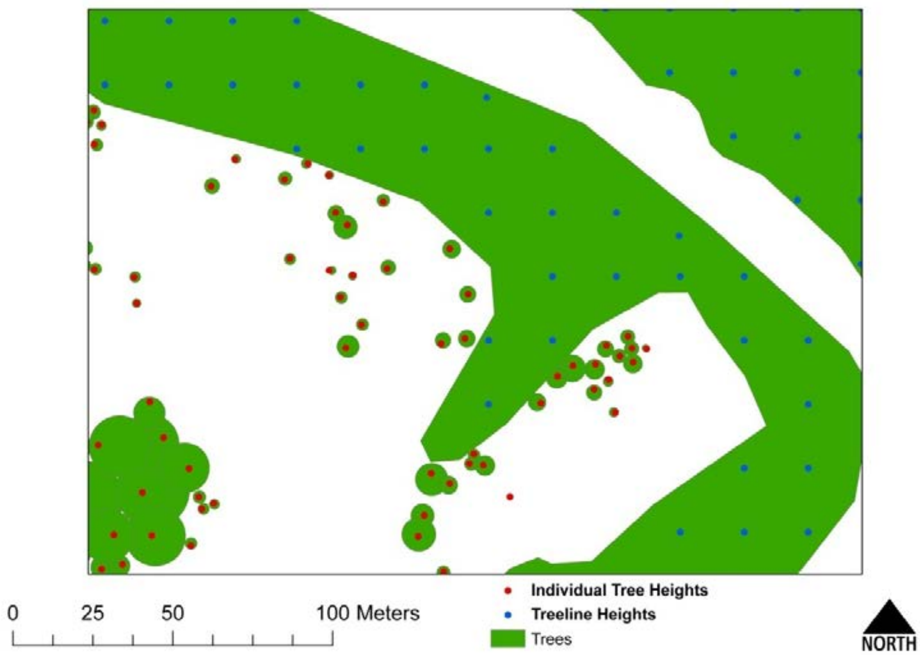

Figure 3-5: The distribution of individual and treeline heights (m) within a sample area of Salford.
Height (m) was calculated for each individual tree canopy and represented by a red point. Treeline heights (continuous tree cover where individual trees could not be identified) were calculated at 20m horizontal intervals and are represented by a blue point. Map created from the 2009 spatial data.

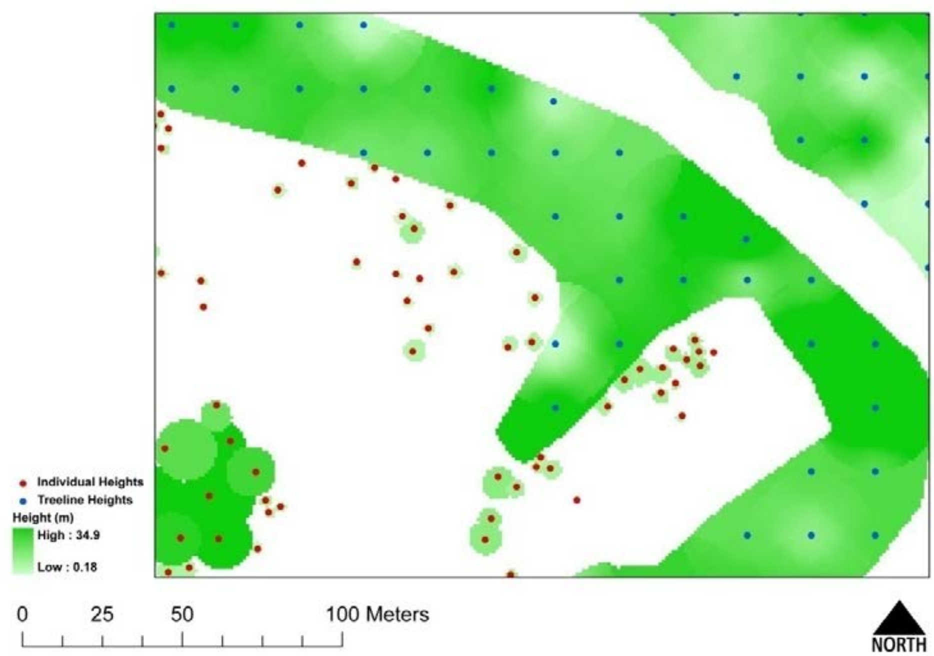

Figure 3-6: Treeline after inverse distance weighted (IDW) interpolation.
To gain a better representation of how treelines were vertically structured IDW interpolation was used. IDW assumes that the closer cells are to each other the more similar they are. The heights of the 8 nearest neighbours of a focal cell were used to calculate its height. As the actual height of the treeline canopies was measured every 20m it was assumed that this method would result in a fair representation of the canopy structure. Map created using the 2009 spatial data.

3.4.2 Stratification of height data

The post interpolation raster dataset representing all trees in Salford were stratified based on the natural breaks within the height data (Figure 3-7). The natural breaks were calculated within ArcMap (version 9.3) which can automatically identify breaks in data and create classes (or natural groups) by grouping similar values together while maximising the differences between groups of similar values. A lower limit of 3m was applied to the height data, corresponding with the definition of a 'standard tree' (2-3m in height with an established, sub-divided branch structure above 1.5m) (Miller, n.d.) as well as with the 2005 data (which will be used to assess the rate of change in tree cover/connectivity). Three natural breaking points were selected in ArcMap in order to correspond to traditional ecological stratification of woodlands (i.e. shrub layer, understorey layer, and canopy layer) (MacArthur & MacArthur, 1961). However, as the UTI data were interpreted from stereo-aerial photography, actual understorey and shrub layers, within tree patches, were not known. Instead, a description of the canopy cover within the heights corresponding to these traditional layers has been described. This approach provides a more straightforward way to characterise tree patch structure, and the amount of canopy cover and the variation of cover produced by gaps are important attributes of woodland structure (McElhinny *et al.*, 2005).

The stratification of the river valley study area's UTI, based on the three natural breaks in height, resulted in the lower canopy including trees between 3 – 7.24m, the middle canopy between 7.25 – 17.1m, and the upper canopy between 17.11 – 34.9m in height. These canopy layers are similar, if slightly smaller, to the definition of large (>18m), medium (10-18m), and small (<10m) trees (Hiller and Coombes, 2007). These natural breaks were used for the landscape connectivity analysis (Chapter 4) and UTI network resilience analysis (Chapter 5). The stratification of Salford's entire UTI resulted in slightly different natural break heights which were pre-standard trees (0-2.99m), small trees (3 – 7.19m), medium trees (7.2 – 17.49m), and large trees (17.5 – 34.9m). These break heights were used for the societal metric and landscape pattern analysis (Chapter 6).

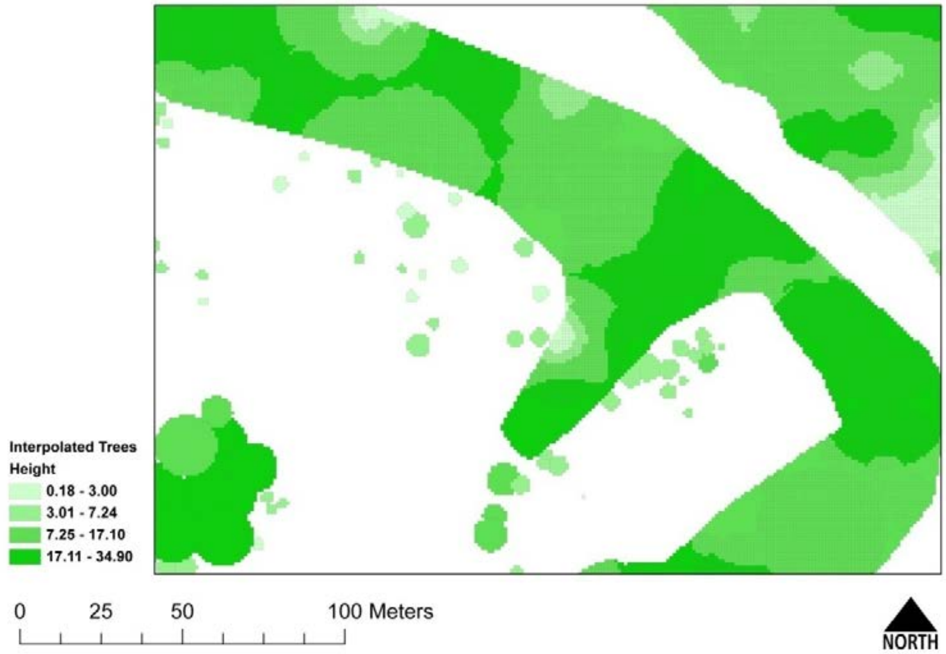

Figure 3-7: The stratification of the tree height (m) data after the interpolation of treelines.
The tree canopy cover was stratified based on the natural breaks within the height (m) data. The figure shows all trees, including those below 3m. However, when creating the natural break height (NBH) canopies a lower limit of 3m was applied (Figure 3-8). Map created from the 2009 spatial data.

3.5 Describing the Urban Tree Infrastructure (UTI) – creating Natural Break Height (NBH) Canopies

The features that constitute the UTI, represented within the tree canopy data, need to be created in a meaningful and useable way. A traditional, widely accepted approach in the assessment of spatial data is to describe the landscape as a lattice containing values or measurements which represent key regions within the landscape (Cressie, 1993). Two conceptual models can then be used in consideration of the lattice landscape: (1) a field view which uses raster grids to form continuous surfaces, defined by a given variable, which can be measured at any point(s); (2) a features view which uses a vector of co-ordinates to describe polygons representing discrete entities within the landscape (Goodchild, 1994). Depending on the application of the spatial data either of the landscape models (field or feature) and data structures (raster or vector) can be used, or preferably a combination of the two (Urban & Keitt, 2001).

Collectively, the trees that form the basis of UTIs create tree canopy patches. Describing these patches is therefore critical to the thesis research. The 2005 and 2009 datasets were provided as vector datasets, thus the landscape was represented as a set of features. Polygons within the 2005 data represented tree canopy patches and not the individual trees which comprised those patches. However, when incorporating height data, individual canopies need to be known so that the vertical shape of the tree canopy patches can be described (methods for applying height values to the 2005 data are in Chapter 4). Polygons within the 2009 data represented both individual tree canopies and tree canopy patches (in the form of treelines - however these were 'deconstructed' via interpolation so that the canopy could be stratified, see section 3.4.1). As the 2009 dataset comprised individual canopies, these would need to be converted into features representing tree canopy patches.

It was assumed that the spatial structure of the tree canopy patches emerged from neighbourhood contacts (Oborny *et al.*, 2007). In other words, tree canopy patches arise when there are no more possible connections between an individual canopy and its neighbour. When the landscape is represented as a grid made up of cells which denote habitat then neighbourhood can be defined in several ways. One of the most common ways is to define the eight connecting cells of a focal cell as the immediate neighbours (Stauffer & Aharony, 1994; Metzger & Decamps, 1997). This method of defining habitat patches based on neighbourhood contact comes from percolation theory. Percolation theory has often been used to create neutral landscape models – landscapes formed from an expected pattern in the absence of a specific ecological process (Caswell, 1976; Gardner *et al.*, 1987). When the landscape is represented as a grid, a two-dimensional percolating network (equitable to a random map model) can be defined by the density of habitable and inhabitable cells (Turner, 1989; Oborny *et al.*, 2007). A cluster (e.g. habitat patch) occurs when a group of habitable cells are in contact with their nearest neighbour, e.g. share at least one cell edge (Turner, 1989). Following Oborny *et al.*'s 2007 definition of percolation theory in ecology, when the density of habitable cells is low then the area for movement is confined, as a habitable cell is part of a finite sized patch (if we assume crossing into the matrix is impossible). Therefore, if the habitable cell density is high then it is more likely that the lattice (i.e. the landscape) will contain a patch which spans, or rather percolates across it (Oborny *et al.*, 2007).

Landscape connectivity models, based on percolation theory, assume that landscapes with high amounts of habitat are strongly connected because it is easier for moving organisms to complement and supplement their habitat requirements (Goodwin & Fahrig, 2002). In essence, this assumption is correct. However, structurally connected habitat patches within a landscape may not be *functionally* connected and even in a non-percolating or contiguous landscape functional connectivity could exist; it depends on the species' movement capabilities (Tischendorf & Fahrig, 2000b). However, a percolation approach can be used to construct the initial structural maps within which functional connectivity can be assessed. Therefore, percolation theory provides a useful framework within which habitat patches can be constructed.

To create a structural map for Salford the post-stratified UTI landscape was considered as a field, a lattice of cells which either represented tree canopy cover at varying heights (e.g. habitat cells representing trees ≥3m, trees ≥7.2, or trees ≥17.1m) or no tree cover (i.e. the matrix). It was assumed that percolating, contiguous habitat cells represented homogenous tree cover and therefore constituted a tree patch. To 'percolate' the landscape (i.e. join touching habitat cells together) habitat cells were assigned the number 1, while the surrounding matrix was assigned the value 0. The spatial analysis tool 'region group' (which generalises data) was then used to join tree canopy cells which were touching. The number of neighbours was set to 8 so horizontal, vertical, and diagonal connections were taken into consideration. An excluded value of 0 was set so as to ignore the matrix when describing cell connectivity. The resulting raster dataset was converted to a vector dataset so that polygons of contiguous tree cover could be represented. As a result, the final landscape consisted of features – tree canopy patches – existing within a matrix void of tree canopy cover. As trees within the higher natural breaks were present throughout the rest of the vertical distribution of the river valley study area's UTI, the natural break heights were cumulated. This resulted in the formation of the natural break height (NBH) canopies, canopy ≥3m, canopy ≥7.2m, and canopy ≥17.1m (Figure 3-8).

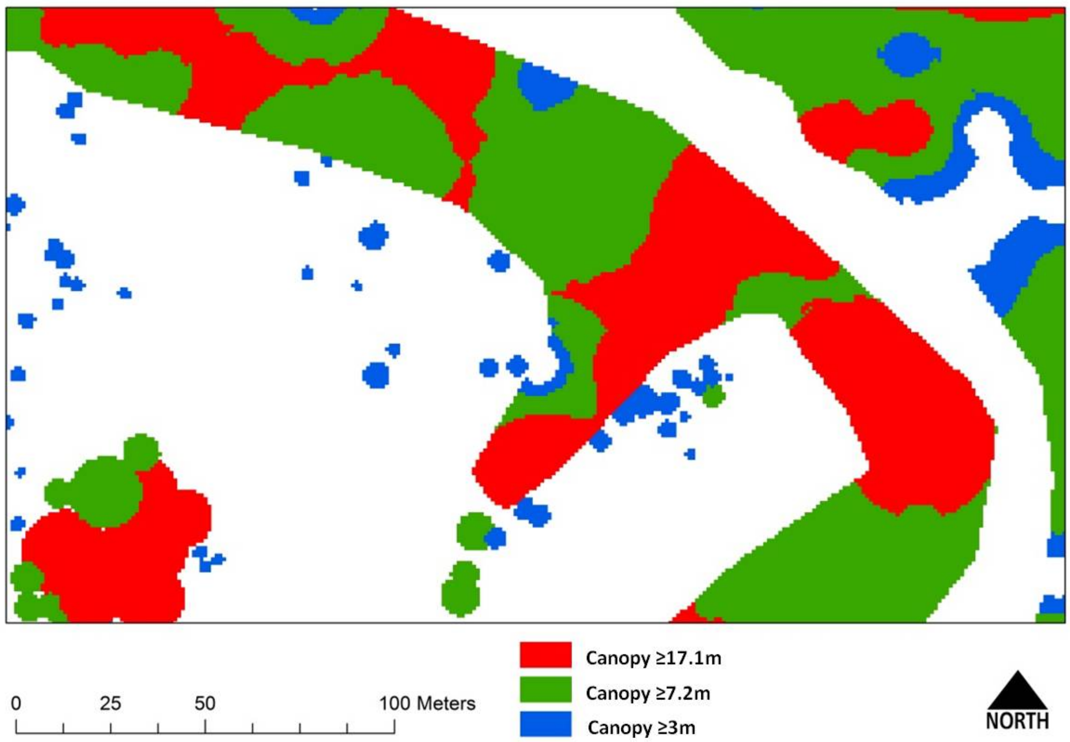

Figure 3-8: The formation of the natural break height (NBH) canopies in 2009.
The NBH canopies were creating by vertically stratifying and then region-grouping cells that represented tree cover. The canopy ≥17.1m (red) represents the upper tree canopies above 17.1m. The canopy ≥7.2m (green) represents all middle and upper tree canopies above 7.24m. The canopy ≥3m (blue) represents all lower, medium, and upper tree canopies above 3m. Map created using 2009 data.

The NBH canopies form the basis of the vertical and horizontal connectivity analysis in Chapter 4 and the UTI resilience analysis in Chapter 5. The same process was applied to the Salford UTI for the societal and landscape pattern analysis in Chapter 6, minus the cumulating of canopy heights as the subsequent canopy patches are to be used to calculate canopy height diversity and so individual tree heights are needed. Therefore, for Chapter 6 the entirety of Salford's tree canopies were percolated, after interpolation, to create canopy patches which were categorised into the height classes pre-standard trees, small trees, medium trees, large trees (detailed methodology in section 6.2).

3.6 Graph Theory

The development of graph theory and habitat availability based metrics has led to the creation of graph theory based landscape connectivity software such as ‘Conefor Sensinode 2.2’ (Saura & Torne, 2009) and ‘Graphab 1.1’ (Foltête *et al.*, 2012). These software assess levels of connectivity using functional-connectivity, habitat availability metrics and network analysis (these issues are discussed in detail in sections 2.3 and 2.4). In addition, Graphab can also be used for the visualisation of landscape graphs.

To assess the structural patterns of the River Valley Study area’s UTI, both of these specialist software were used. Conefor and the ArcGIS extension ‘Conefor Inputs’ (Jenness, 2011) were used to calculate the overall landscape connectivity (Chapter 4) and individual canopy patch importance for maintaining connectivity within the river valley study area (Chapter 5). Graphab was used to create a set of river valley study area graphs to undertake a network topology analysis (Chapter 5). Conefor’s primary function is to quantify the importance of habitat patches for maintaining landscape connectivity through graph structures and habitat availability indices (Saura & Pascual-Hortal, 2007a). Similarly, Graphab is a software application for modelling ecological networks using landscape graphs (Foltête *et al.*, 2012). Both programmes can be used to identify nodes and links within landscape data and therefore be used to construct landscape graphs. Connectivity metrics can then be computed from these graphs. Unlike Conefor, however, Graphab can also be used to create species distribution models and has a module for allowing visual and cartographic interfacing.

Conefor uses numerical files (ASCII text or DBF formats) to calculate landscape connectivity. In these files numbers represent the nodes and links of a landscape graph. Therefore, Conefor does not produce actual landscape graphs that can be visualised or mapped. Graphab however uses raster grids to calculate connectivity, within which cells represent a given landcover category (e.g. tree canopy cover or no canopy tree cover). Similar cells that are touching are defined as patches and therefore landscape nodes. As raster grids can be geo-referenced the resulting landscape graphs constructed by Graphab can be mapped using a GIS.

It is graph theory's versatility and mathematically based pattern-describing-powers that have led to its use within the thesis. Temporal and spatial changes in landscape connectivity within the river valley study area's UTI is analysed within Chapter 4. The relationship between the same UTI's structural patterns and its resilience for maintaining function is assessed in Chapter 5. As these two chapters are both ecologically focused, while the UTI is a social-ecological system, Chapter 6 contains a presentation of research into the relationships between spatially explicit societal patterns and ecological structure.

Chapter 4 : Temporal Changes in the Vertical and Horizontal Connectivity of an Urban Tree Infrastructure (UTI)

4.1 Introduction

In urban areas the size, shape, composition, and connectivity of habitat patches have been influenced by land use and land cover changes which often negatively affect bird movement abilities (Alberti, 2005; Tremblay & St. Clair, 2009, 2011). The UTI plays a fundamental role in ameliorating these effects through facilitating urban passerine movement which in turn influences the provision of ecosystem services and ecosystem function and resilience in urban areas (Fernández-Juricic, 2001a; Lundberg & Moberg, 2003; Tremblay & St. Clair, 2011; Wenny *et al.*, 2011; TDAG, 2012). However, gaps in the UTI can impede such movement, the level of which depends on passerine gap perception (Tremblay & St. Clair, 2009). The vertical structure of the UTI is also important for urban passerine assemblages and abundance (Goldstein *et al.*, 1986) and gaps in the UTI can occur across various canopy heights (McElhinny, 2002; McElhinny *et al.*, 2005).

As an expression of landscape structure, spatial heterogeneity can be used to indicate the variability of an ecological system's properties and therefore quantification of spatial heterogeneity is needed to understand relationships between pattern and process (Turner, 1990; Uuemaa *et al.*, 2009; Walz, 2011). However, attempts to describe the spatial heterogeneity of landscape elements and levels of connectivity between them have previously only been developed in two dimensions. As such they ignore landscape elements which exist in three dimensions, such as the UTI. Research that maps the UTI in three dimensions and uses this map to model landscape connectivity by assimilating a passerine's ability to cross gaps existing across the vertical structure of the UTI has not yet been undertaken. The work presented in Chapter 4 addresses this research gap.

Chapter 4 addresses the following research question; 'what is the change in potential landscape connectivity of Salford's three dimensional UTI for passerines over time and space?' To answer this, research objective 2.1 must be completed (Figure 3-1). Research objective 2.1

can only be accomplished once a thorough passerine gap-crossing capability threshold analysis is incorporated into potential landscape connectivity models which focus on assessing connectivity across spatial and temporal dimensions. In order to achieve this goal, the river valley study area's UTI should be described in three dimensions and modelled using data collected at different times. Furthermore, these connectivity models need to incorporate relevant passerine perceptions and behaviours: ones that affect their potential movement through the UTI such as ability to cross gaps. Methods employed in order to achieve these research goals (underpinning research objective 2.1) are described in section 4.2. The subsequent results and analysis of the data are presented in section 4.3. The meaning and impact of these results are discussed in section 4.4, followed by the conclusions in section 4.5.

4.2 Methods

Figure 4-1 provides a diagrammatical outline of the data, processes, and methods used to achieve chapter 4's research objectives. The river valley study area's UTI was vertically stratified based on tree height (m) and described as it existed in 2005/6, 2009, and 2013/14. Therefore, potential landscape connectivity models were constructed from spatial datasets that have been generated at different points in time. The description of the UTI canopies from 2005, 2009, and 2013, were incorporated into a connectivity analysis model with passerine gap-crossing capability threshold data. Section 4.2.1 contains an outline of the literature review methods used to establish passerine perceptual behaviour, with a focus on the capability of passerines to cross canopy gaps of varying sizes. The methods used to describe the river valley study area's UTI in 2005 and 2009 are outlined in section 4.2.2 and 4.2.3 respectively. A representation of the UTI in 2013 was created using field survey data (section 4.2.4) and spatial data pre-processing (section 4.2.5). Finally, the connectivity analysis methods are explained in section 4.2.6.

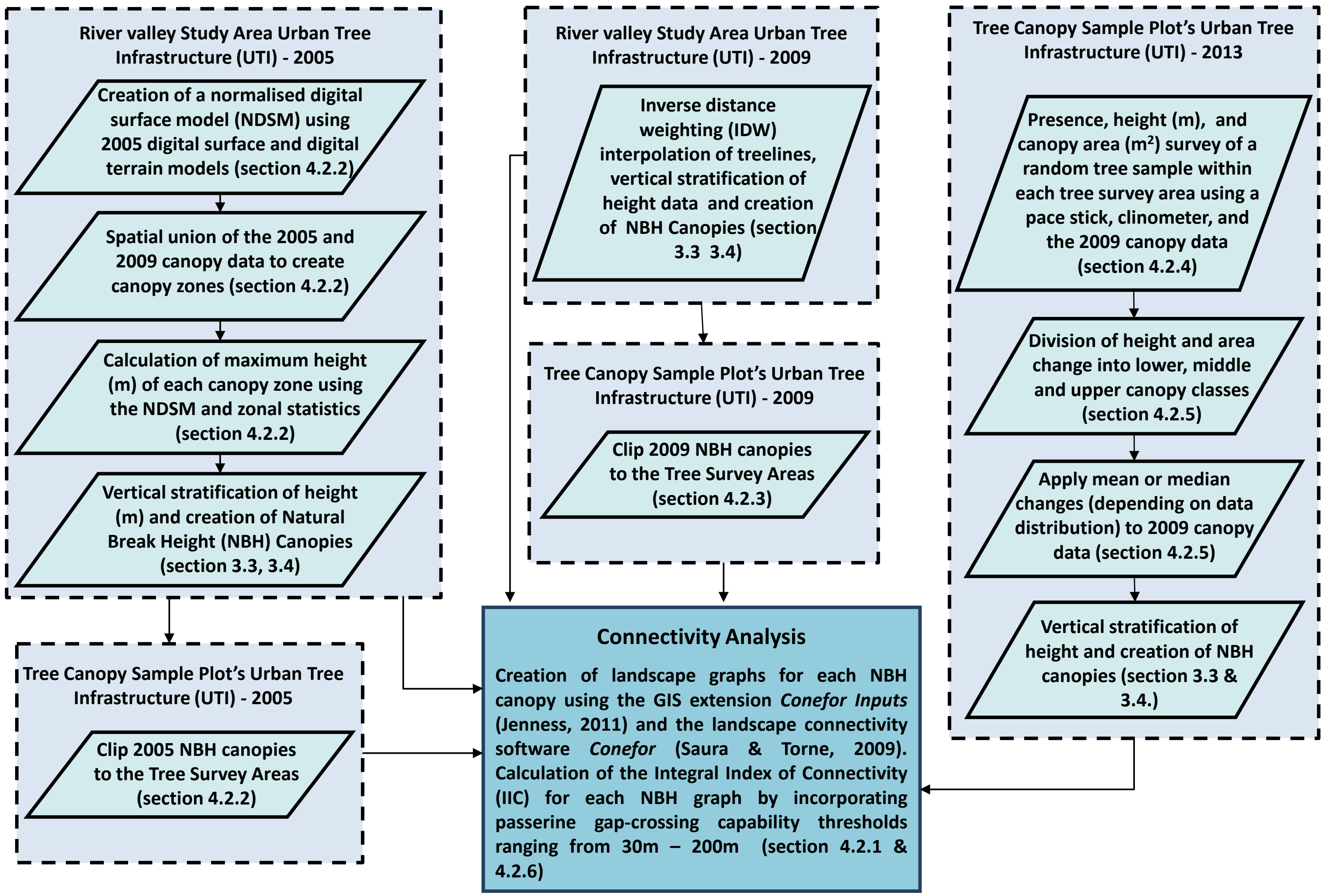

Figure 4-1: A diagrammatic representation of the processes (parallelogram), data (dashed rectangle) and analysis (rectangle) used in Chapter 4.

A total of five dataset (dashed rectangles) were included in the potential landscape connectivity analysis (blue rectangle). In order to create useable data a series of processes were undertaken (parallelograms). By completing these processes a 2005 and 2009 river valley study area dataset and a 2005/9/13 tree canopy sample plot dataset were created. By following the methodological approach outlined in Figure 4-1 a multi-scale and multi-dimensional analysis of potential UTI landscape connectivity was achieved. The vertical stratification and creation of all NBH canopies followed the methods set out in section 3.3 and 3.4.

4.2.1 Passerine gap-crossing capability thresholds

A critical literature review on the gap-crossing abilities of forest passerines in both highly fragmented and relatively contiguous landscapes was undertaken. Using the literature databases *Web of Science* and *Scopus,* papers containing the key words *passerines*, *forest songbirds, connectivity*, *canopy gaps*, *gap-crossing*, *patch, urban*, *movement, dispersal,* and *dispersal threshold* from 1980 to 2013 were searched for. Eleven papers provided usable data on the gap-crossing abilities of passerine birds and only one paper provided data on passerine gap-crossing abilities in an urban environment (Table 4-1). The literature indicated that gap-crossing capability thresholds ranging from 30m to 200m should be used within the connectivity analysis.

Within the least fragmented, forested landscapes gap-crossing capability thresholds are shown to be between 25m and 150m (Rail *et al.*, 1997; Creegan & Osborne, 2005). In a forested landscape containing large, cleared gaps maximum gap-crossing capability threshold was shown to be 260m (Robertson & Radford, 2009). However, this distance was traversed by individuals moving through cleared gaps containing scattered, single trees which were potentially aiding movement by providing stepping stones (Robertson & Radford, 2009). Therefore, it can be assumed that the actual maximum gap (i.e. no tree cover within gap/matrix) crossing distance recorded was 80m (Robertson & Radford, 2009). As fragmentation increases (agricultural environment with remnant woodland) passerines are, on average, shown to be willing to cross gaps of up to 200m (St. Clair *et al.*, 1998; Grubb & Doherty, 1999; Hinsley, 2000; Awade & Metzger, 2008; MacIntosh *et al.*, 2011). Furthermore, as gap distances increase towards 200m, the probability of an individual to move across the gap will dramatically decrease (Awade & Metzger, 2008). Interestingly, it has also been shown that individuals will only cross gaps of up to 50m when a forested alternative is available (even if this alternate route is further than 50m), if this alternative is unavailable, passerines would cross gaps of up to 200m (St. Clair *et al.*, 1998). Furthermore, passerines are shown to typically move within 25m of the forest edge, even if they are able to do otherwise (Bélisle & Desrochers, 2002). As the landscape becomes ever more urbanised, passerine gap-crossing probabilities start to become affected negatively after 30m (Desrochers & Hannon, 1997). Similarly, within an urban environment, ability to cross gaps in tree cover decreases after 30m (Tremblay & St. Clair, 2009) (Table 4-1).

Table 4-1: The gap-crossing capability thresholds and gap-crossing probabilities of passerines in forested, agricultural, and urban landscapes.

Author/s	Landscape	Gap Crossing Distances
Rail *et al.,* 1997	Forest	25m – 70m
Creegan & Osborne, 2005	Woodland	50m - 150m
Robertson & Radford, 2009	Forest	Up to 260m if scattered trees were present in gaps Up to 80m when no trees were present
St. Clair *et al.*, 1998	Agriculturally dominated with fragmented woodland	50m if forested alternatives. 200m when no choice
Grubb & Doherty, 1999	Agriculturally dominated with fragmented woodland	50 – 175m
Hinsley, 2000	Agriculturally dominated with fragmented woodland	50 – 100m when gap-crossing occurs regularly during foraging
Belisle & Desrochers, 2002	Agriculturally dominated with fragmented woodland	No more than 25m from forest edge
Awade & Metzger, 2008	Agriculturally dominated with fragmented rainforest	50% chance of crossing at 40m 10% chance of crossing at 60 - 80m
MacIntosh *et al.,* 2011	Agriculturally dominated with fragmented woodland	Male thrushes typically < 100m (max. 615m) Female thrushes typically <200m (max. 300m)
Desrochers & Hannon, 1997	Rural and forested landscape near city	<30m no impact 30% chance of crossing at 70m 10% chance of crossing at 100m
Tremblay & St. Clair, 2009	Urban	30m no impact 50% chance of crossing at 45m

Gap-crossing capability thresholds (m) and movement characteristics exhibited by passerines in North America, South America, Australasia, and Europe. Typically, passerines would cross gaps in the canopy up to a maximum of 200m. In urbanised areas gaps below 30m have little impact on movement.

4.2.2 Describing NBH canopies in 2005

The shapefile representing tree canopies within Salford in 2005 was provided by Red Rose Forest. This shapefile contained irregular polygons representing groups of trees referred to as 'canopy polygons'. To ascertain the heights of the 2005 canopy polygons a '2009 PROFILE digital terrain model' (DTM) (EDINA, 2009) was obtained from Digimap (http://digimap.edina.ac.uk) while a 'Manchester/Liverpool 2005 digital surface model' (DSM) (Landmap, 2006) was acquired from Landmap (http://learningzone.rspsoc.org.uk). Both models were created using light detection and ranging (LiDAR) data. The DTM comprised of 10x10m cells relating to terrain height only and hence void of any surface objects. Conversely, the DSM contained 2x2m cells relating to the heights (m) of objects on the surface of a given area (e.g. buildings, trees, structures etc.). However, the DSM made no distinction between the heights of objects and terrain, thus the DSM cells contained cumulative values. Therefore, height values associated with surface objects were not the actual height of the object. To rectify this both models first needed to be resampled so that their pixel sizes were the same as each other and that of the tree canopy cells.

Using the raster processing tool 'Resample' the DTM and DSM were resampled so that their pixel sizes were reduced to 1x1m. The 2005 canopy polygons were then used as a mask to extract the cells relating to trees ('tree cells') within the resampled DTM and DSM. This processes created a canopy terrain model (CTM) and canopy surface model (CSM). To determine the actual height of the tree cells the CTM height values were subtracted from the CSM height values. Thus, a 'normalised canopy surface model' (NCSM) was created. The NCSM therefore contained 1x1m cells relating to the height of canopies in 2005, independent of the terrain/ground height. The river valley study area footprint was then used as a mask to extract the relevant tree cells from the NCSM.

The NCSM consisted of single, 1x1m cells and therefore did not associate a single height measurement with a given tree canopy as canopies were not described as distinct entities. To define individual canopy height a single, maximum height value would have to be applied to a distinct tree canopy polygon representing an individual tree (i.e. not treelines). Collectively, these polygons would then act as the spatial 'zone' within which each maximum height could be calculated using the height values from the NCSM (i.e. the NCSM acts as the focal layer) and ArcMap's zonal statistics tool. However, the original, 2005 shapefiles could not, initially, be used as the spatial zone as no attempt was made to distinguish individual tree canopies within large areas of tree cover. If it were to be used, large canopies would only be assigned a single height value, thus creating homogenous tree canopy patches which could not be meaningfully stratified. Therefore, a new spatial zone (i.e. discrete canopy polygons) had to be created.

The interpolated 2009 canopy polygons (e.g. treelines have been interpolated based on the 20m horizontal height measurements, see Chapter 3.3) were spatially merged with the 2005 canopy polygons. This merge created a new spatial zone consisting of a mixture of the 2009 and 2005 canopy polygons (Figure 4-2). As the NCSM was cut to the 2005 canopy polygons, the zonal statistics would only consider pixel heights within the 2005 footprint (Figure 4-3). However, the 2009 canopy polygons also provided a footprint that effectively described canopy borders within the homogenous 2005 canopy polygons (especially for large treelines).

This merged dataset (referred to as the '05/09 zone') was then used to calculate maximum canopy zone height (Figure 4-4). The canopy zone heights were then converted to vector data – thus creating canopy polygons containing height data. The canopy polygons could then be vertically stratified and transformed into natural break height (NBH) canopies (Figure 4-5) (see section 3.4.2). The canopies were stratified based on the same natural breaks in the height data as the 2009 canopies in order for direct comparisons to be made (maximum height in 2005 was 38.5m). The resulting NBH canopies were termed 'canopy ≥3m', 'canopy ≥7.2m', and 'canopy ≥17.2m'. So that temporal changes in connectivity could be assessed the NBH canopies were also clipped to the tree canopy sample plot boundaries (Figure 3-4) using the ArcGIS v9.3 geoprocessing tool 'clip'. Therefore, fifteen 2005 NBH canopy shapefiles were created in total - three for the River Valley study area and three for each of the four tree canopy sample plots.

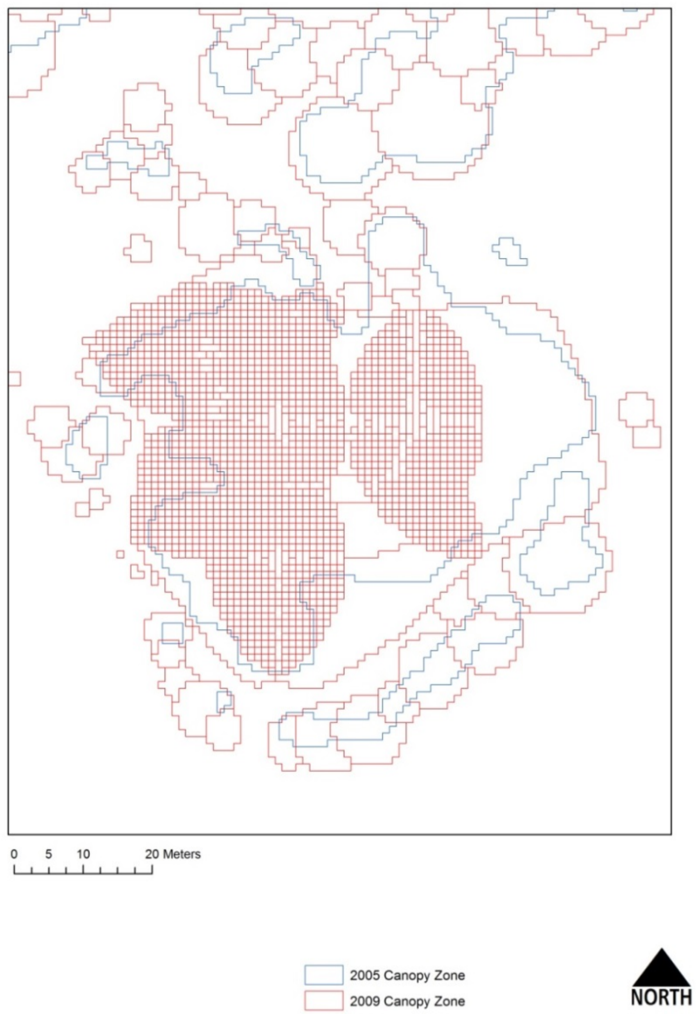

Figure 4-2: Canopy zones for calculating maximum height (m).
Canopy zones were created by combining the 2005 and 2009 canopy footprints. The 2005 footprint is the maximum canopy boundary while the 2009 footprint represents the intra- patch canopy boundaries.

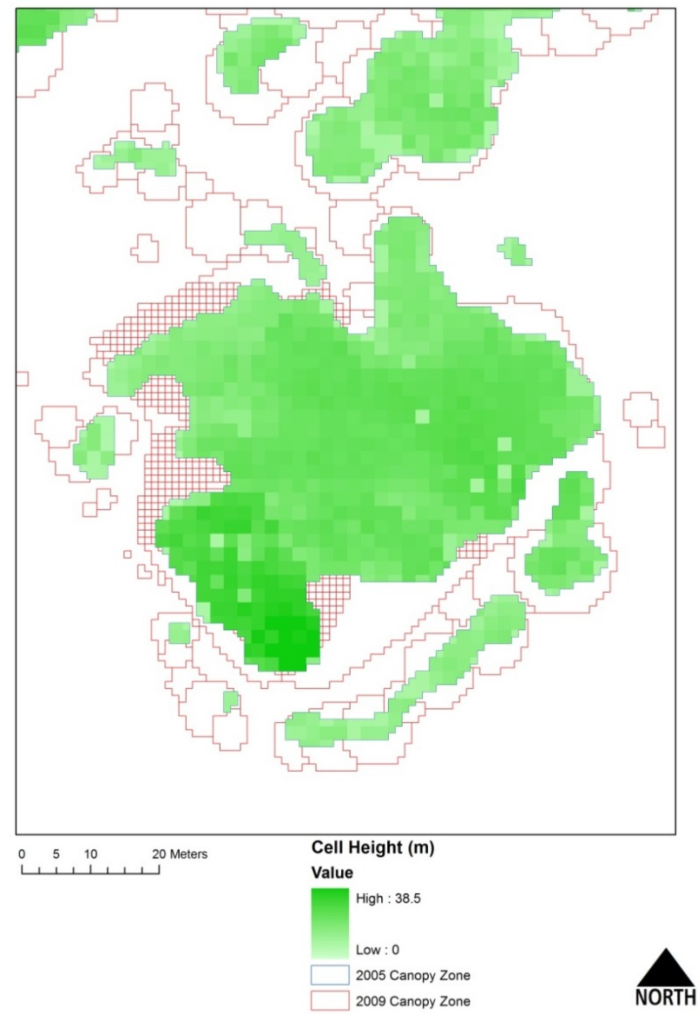

Figure 4-3: Canopy zones and normalised canopy surface model (NCSM) cell height (m).
The NCSM illustrates the heights of the 2005/9 canopy zone tree cells; the darker the green, the higher the tree pixel value.

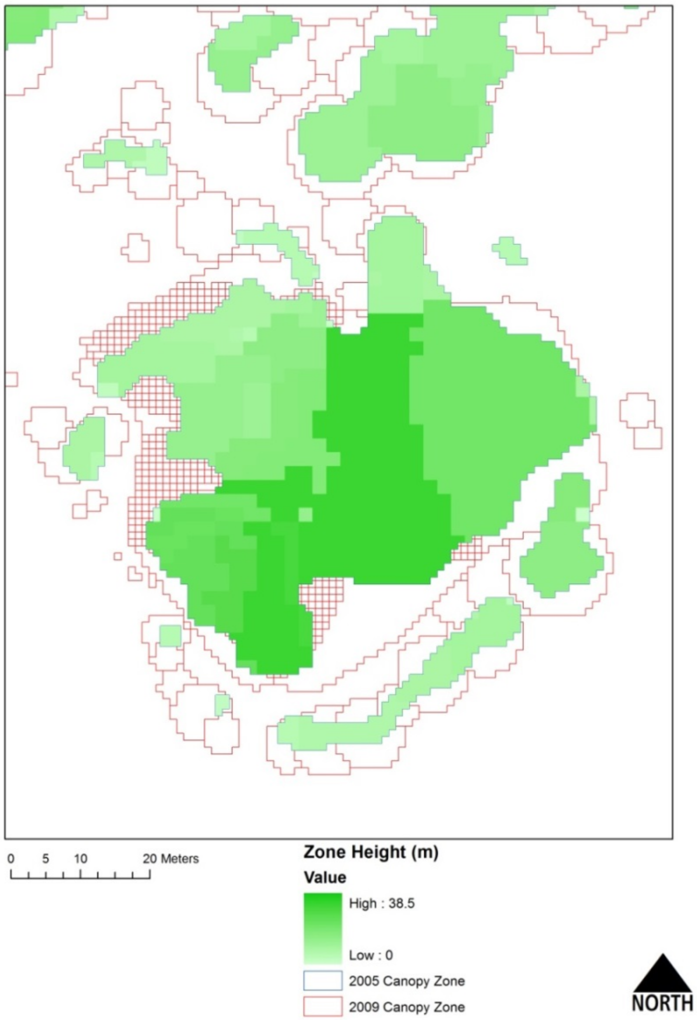

Figure 4-4: Canopy zone height (m).
The vertical shape of the 2005 canopies was spatially expressed by calculating the maximum height of tree cells within the canopy zones.

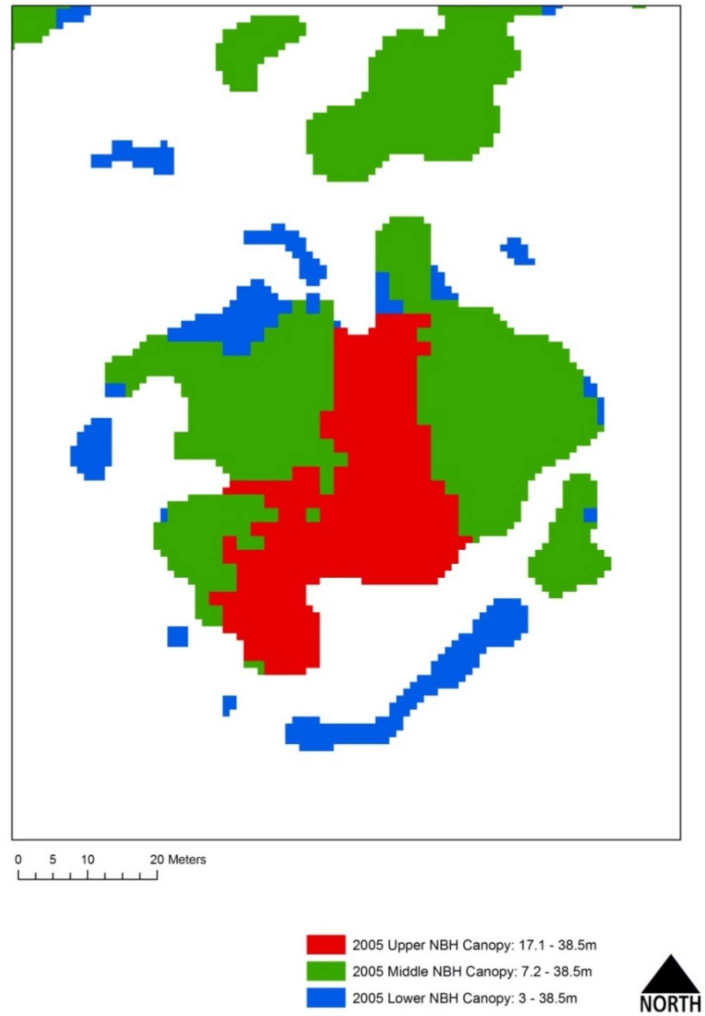

Figure 4-5: 2005 Natural Break Height (NBH) canopy sample.
Using the 2009 natural break heights the 2005 UTI canopies were vertically stratified and grouped together to create 2005 NBH canopies.

4.2.3 Describing NBH canopies in 2009

The creation of the 2009 NBH canopies followed the same methods described in sections 3.3 and 3.4. As a height value was already associated with the polygons representing 2009 tree canopies, there were no other data pre-processing requirements. Therefore, the 2009 tree canopy data were interpolated, vertically stratified, percolated and grouped, and finally converted to a vector dataset representing the vertically stratified canopies of the 2009 UTI. As for the 2005 data the resulting vertically stratified canopies were termed 'canopy ≥3m', 'canopy ≥7.2m', and 'canopy ≥17.1m'. The NBH canopies were also spatially clipped to the four 500m diameter tree canopy sample plots. Therefore, fifteen NBH canopies were created in total.

4.2.4 Collecting tree height and canopy area data

The 2009 data were used to inform and guide the creation of a completely new UTI dataset representing tree canopies within the river valley study area in 2013. Relevant variables needed to be sampled in the field and the values of which applied to the 2009 canopy data. However, before a field survey could be undertaken guide maps of the four tree canopy sample plots (Chapter 3) had to be made. These guide maps were created from the 2009 tree canopy data (individual trees >3m, except for Lower Kersal which also had <3m trees included) and Ordnance Survey data.

Tree canopies were labelled with a unique identification number and grids made up of 100m x l00m cells were applied to each tree canopy sample plot. The grid cells were labelled horizontally by letters and vertically by numbers so that each grid cell had a point of reference (Figure 4-6, Figure 4-7, Figure 4-8, Figure 4-9). Finer scale maps, describing landcover within each grid cell, were also used in the field.

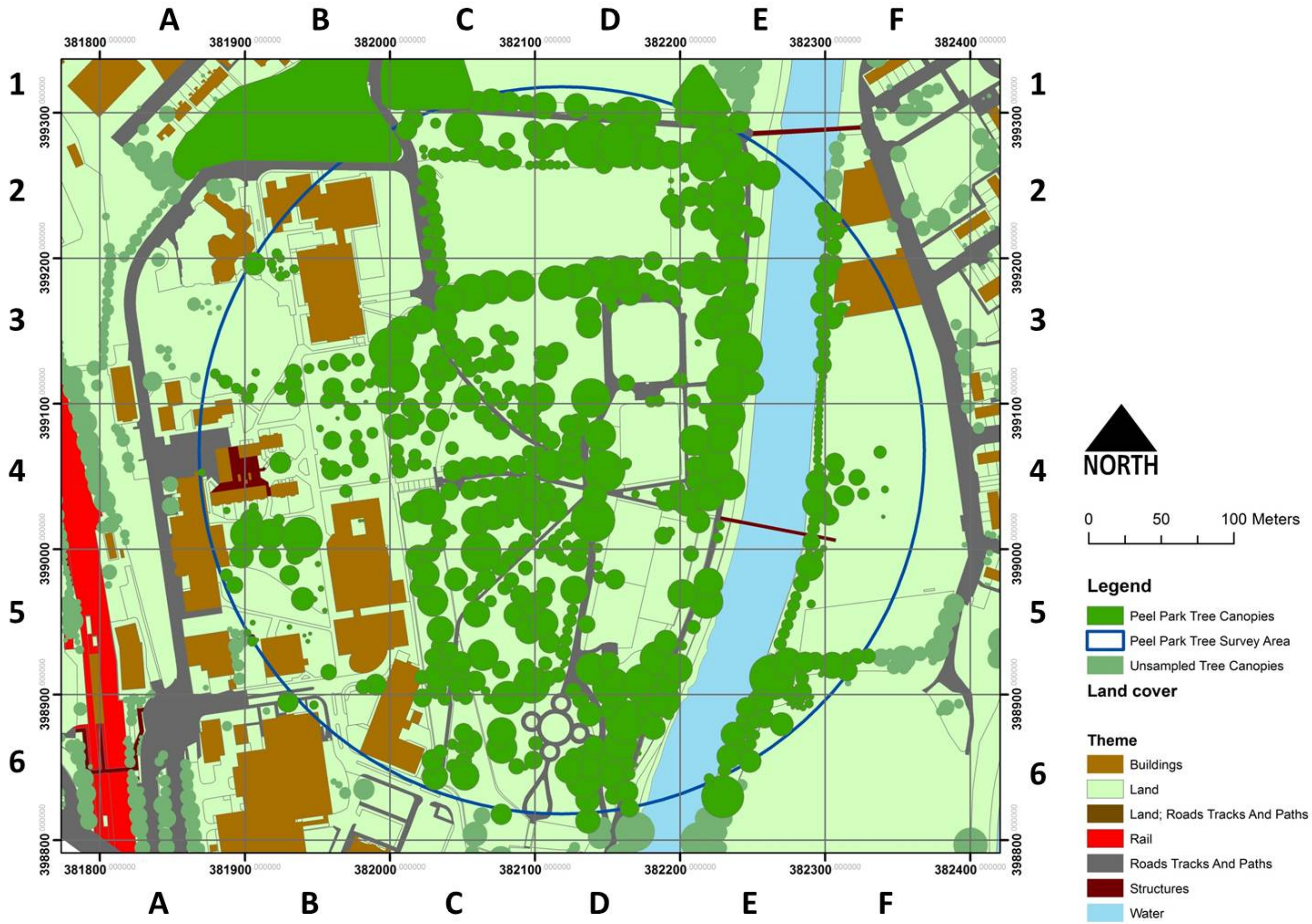

Figure 4-6: Field sampling guide map for the Peel Park tree canopy sample plot.

The Peel Park sample plot comprises a section of The University of Salford's Peel Campus (west), Peel Park (centre), the River Irwell (east), and ex-University land and now brown field site (east). The majority of trees are within the park as well as the University's arboretum.

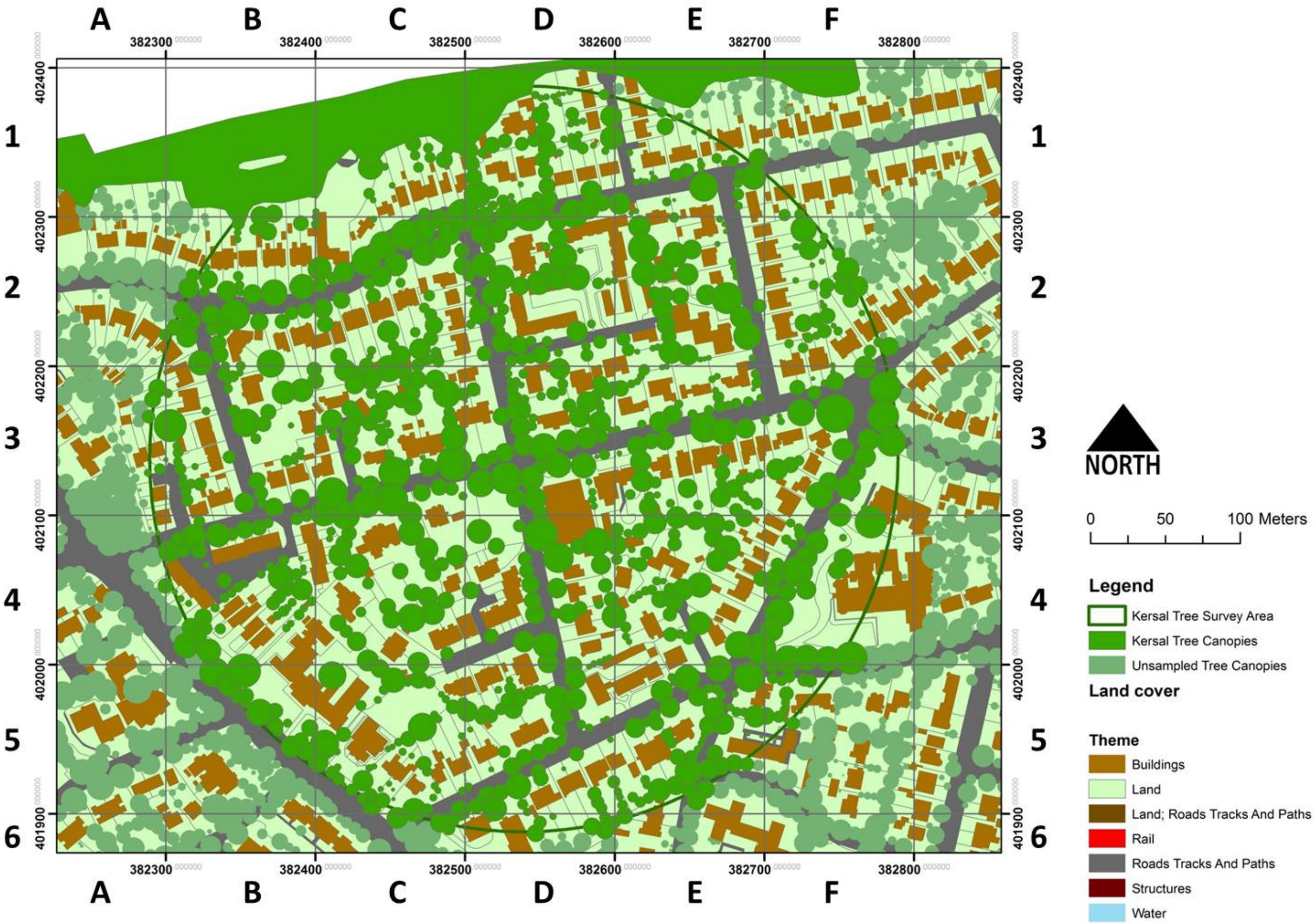

Figure 4-7: Field sampling guide map for the Kersal tree canopy sample plot.

The Kersal sample plot exists at the very northern part of Salford, next to the boarder to Prestwich. The majority of trees in the study area are street trees or privately owned trees in gardens.

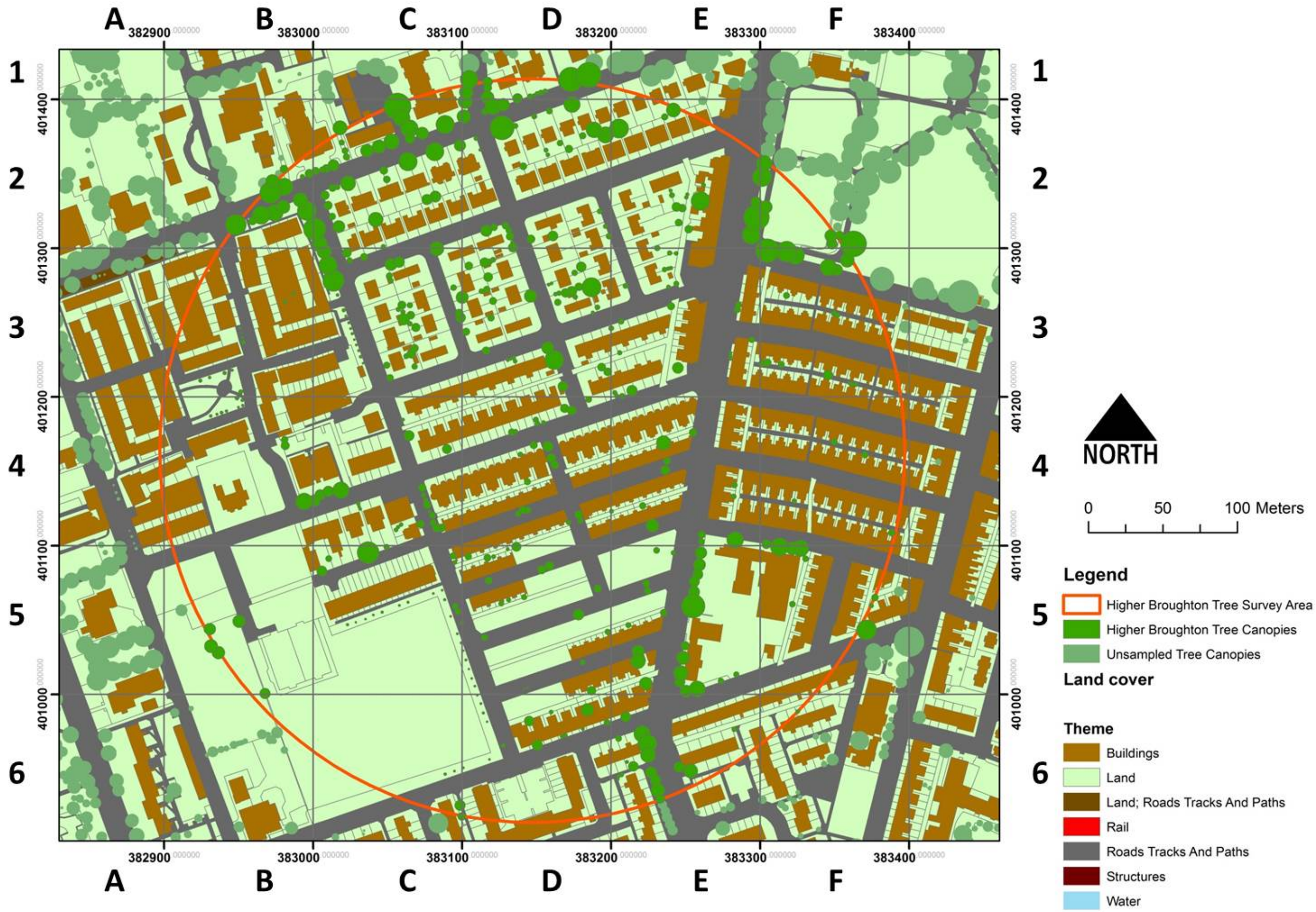

Figure 4-8: Field sampling guide map for the Higher Broughton tree canopy sample plot.

The Higher Broughton sample plot contains a section of public park (north east), playing fields (south west), and residential buildings. Extensive development and construction work has been undertaken towards the centre and southern parts of the map. This has lead to the loss of a great number of trees as well as a change in the spatial arrangement of buildings and roads.

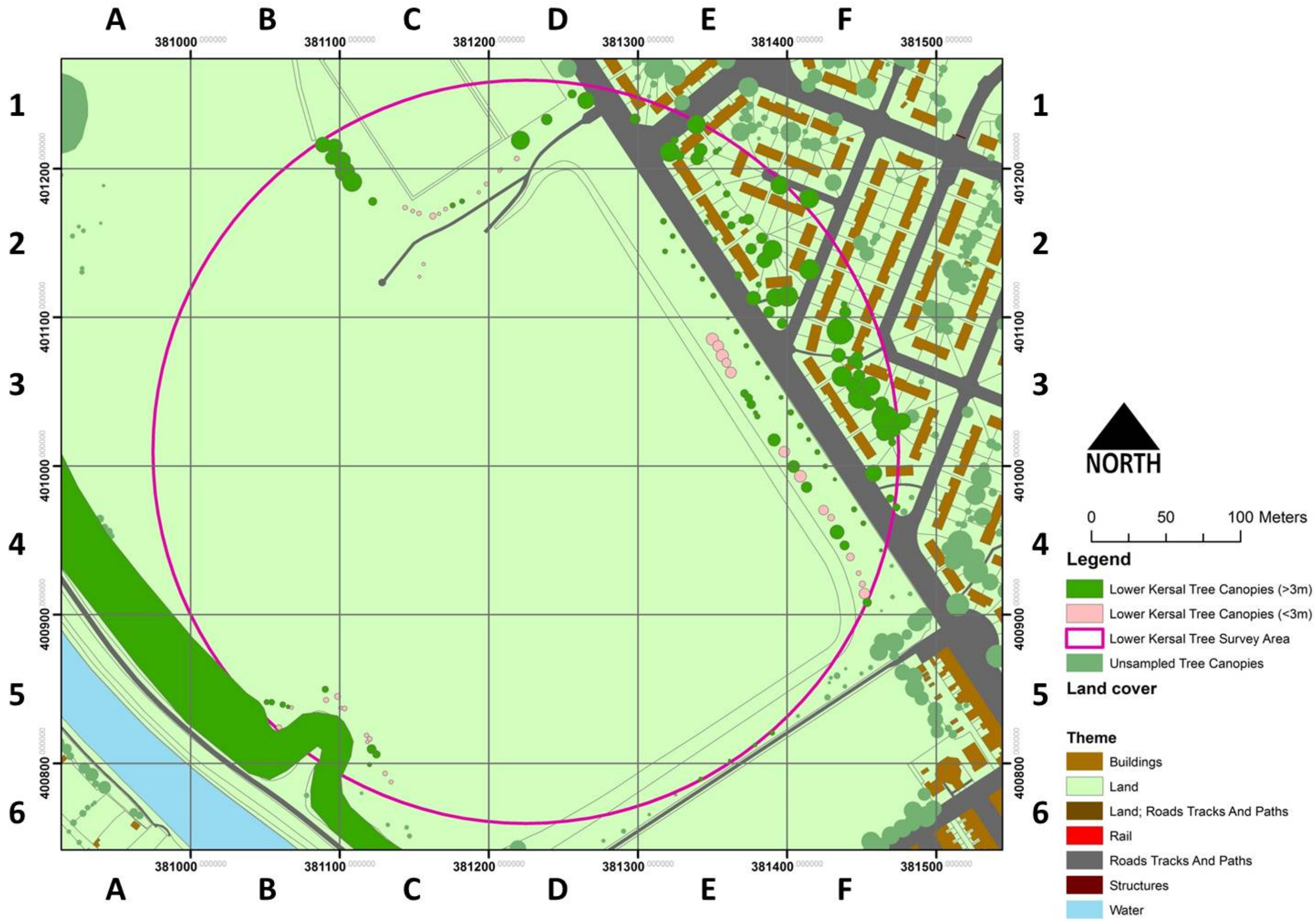

Figure 4-9: Field sampling guide map for the Lower Kersal tree canopy sample plot.

The majority of the Lower Kersal sample plot is made up of playing fields and flood plains. The north eastern part of the study area contains residential buildings and privately owned trees. The rest of the trees in the study area are either planted street trees (east, following the main road), small saplings and patches of scrubland/regenerating woodland (north, east), or form a small patch of woodland (south west).

Two tree surveys were performed within the four tree canopy sample plots in order to collect the relevant data needed for the creation of a 2013 UTI dataset. The first survey (primary survey) was undertaken at the end of the 2013 growing season, while leaves were still present (October/November 2013). The second survey (secondary survey) was conducted in early spring, 2014 while leaves were off (February/March). For the primary survey only a sample of the tree population within each of the tree canopy sample plots were studied while 100% of the tree population was studied for the secondary survey. The secondary survey was undertaken in order to record the presence/absence of all trees within each of the tree canopy sample plots. Using the 2009 guide maps, the location of each tree was visited and if the tree was not there (i.e. had been removed) the canopy was crossed of the map. This information could then be used to remove tree canopies from the 2009 UTI dataset before the creation of the 2013 UTI dataset.

To define tree population sample size for the primary survey, a scoping study within each of the tree canopy sample plots was undertaken, revealing that the variability of tree species and tree height (within vertical layers) was low. Therefore, intra-survey area homogeneity existed. Many trees, especially street trees, seemed to be planted at the same point in time as they appeared to be of similar age as well as of the same species. For example in the Kersal sample plot several streets exhibited lines of lime (Tilia spp.) and beech (Fagus sylvatica), while the Higher Broughton sample plot contained several recently planted Chaenomeles spp. The only exception was the lower Kersal sample plot which contained a few tall, inaccessible, trees as outliers which would not be included in the field survey. Due to these observations the use of small sample sizes for the primary survey was established. Such sample sizes could be surveyed relatively quickly and effectively. Furthermore, the majority of trees in the sample plots (minus Peel Park) were on private land or unsuitably located (slopes, barriers to sight lines etc.) which meant using small sample sizes represented the most realistic option. Therefore, it was decided that either 100 trees or 20% of the overall tree population were to be sampled (which ever value was reached first). Surveying 20% of the population meant that no less than 30 trees were sampled within any one sample plot, while 100 trees represented the maximum survey number.

The sample trees were selected randomly (using Excel random calculator to select canopy identification numbers) and located in the field by using the guide maps. When a randomly selected tree could not be surveyed due to inaccessibility, the closest accessible tree was surveyed instead. At each tree several variables were recorded onto the relevant guide map next to the tree's ID number. The initial variables included in the survey were: (i) the location of the tree in regards to whether it was on public land, private land, set within a park or greenspace, or a street tree; (ii) the genus of each tree and if the tree genus could not be discerned it was recorded as unknown; (iii) the height of the tree (m); and (iv) the approximate canopy area of the tree (m^2).

To record tree height and area further variables were needed. These extra variables were then used to calculate tree height and canopy area. To be specific, a pace stick, set to a metre, was used to measure the approximate radius of a given tree canopy. To do this the distances from the centre of the trunk to two points on the edge of the canopy, at a right angle to each other, were recorded. Therefore, two canopy radius measurements were recorded (r1 and r2) (Figure 4-11). The pace stick was also used in conjunction with a clinometer to measure tree height. The distance from the centre of the tree trunk to a point where the top of the canopy was clearly visible was recorded as was the angle from the recorders eye level to the top of the canopy (Figure 4-11). The clinometer was used by the same recorder throughout the field sampling to maintain eye level height as well as to preserve the judgement of canopy zenith. If a tree 'sat' above ground level (e.g. within a raised garden) the height from the ground to the tree base was noted so that it could be subtracted from the calculated height.

To maintain the greatest accuracy, distance from a focal tree should be approximately the same as the height of the tree so that the angle measured is 45˚ or below (Williams *et al.*, 1994). However, in urban areas space is limited and being able to walk 15 to 20m and still maintain a visual of the tree being measured as well as maintaining level footing can be difficult. Consequently, angles above 45˚ were recorded in the field. Nevertheless, the majority of measurements (60-70%) were 45˚ or below. The accuracy of the clinometer used was also tested by measuring the height of a 3.5m tree on flat ground using a tape measure. The clinometer was used to measure the angle of the tree at eye height (same recorder as in the field surveys) from a distance of 1 to 4m. At 3 and 4m (i.e. around the height of the tree) the difference in results was +/-0.2m (Table 4-2). When the angle measured was over 45˚ the margin of error doubled to +/-0.4m (Table 4-2). These values are similar to the results of a simulated tree height study (Figure 4-10). Using a distance of 10m and angles ranging from 20˚ up to 75.5˚ (increasing every 0.5˚) simulated tree heights were calculated in Excel. The height difference between the 0.5˚ increments was then calculated (Figure 4-10).

The results suggest that from 20˚ to 40˚ the difference in height, if the clinometer has an error of plus or minus 0.5°, is 0.2-0.3m, increasing to around 1m difference at 65˚. The highest angle recorded in the field was 65˚ meaning the maximum error for the clinometer is potentially 1m. This is considered acceptable as the vertical accuracy for the 2009 height data is also +/- 1m. As the majority of angles measured were below 60˚, and 60-70% were below 45˚, clinometer accuracy is between +/-0.2m and +/- 0.5m.

Table 4-2: Testing the accuracy of the clinometer used in the 2013 tree height survey.

Clinometer Accuracy Testing						
Distance from tree (m)	**Angle (˚)**	**Radians**	**Eye Level (m)**	**Clinometer Height (m)**	**Actual Height (m)**	**Difference (+/-m)**
1	53	0.93	1.75	3.08	3.5	0.4
2	35	0.61	1.75	3.15	3.5	0.3
3	27	0.47	1.75	3.28	3.5	0.2
4	21	0.37	1.75	3.29	3.5	0.2

Using a tape measure the height of a small, 3.5m tree on flat ground was measured. The angle from eye height to the top of the tree was then measured using a clinometer from a distance of 1 to 4m. Using these variables the height of the tree was calculated using a trigonometric equation (Equation 4-1). The difference between the calculated height and the actual height was recorded. The accuracy of the clinometer increases as distance increases and angle decreases. The optimal distance is around the actual height of the tree (3-4m) which provides an accuracy of +/- 0.2m.

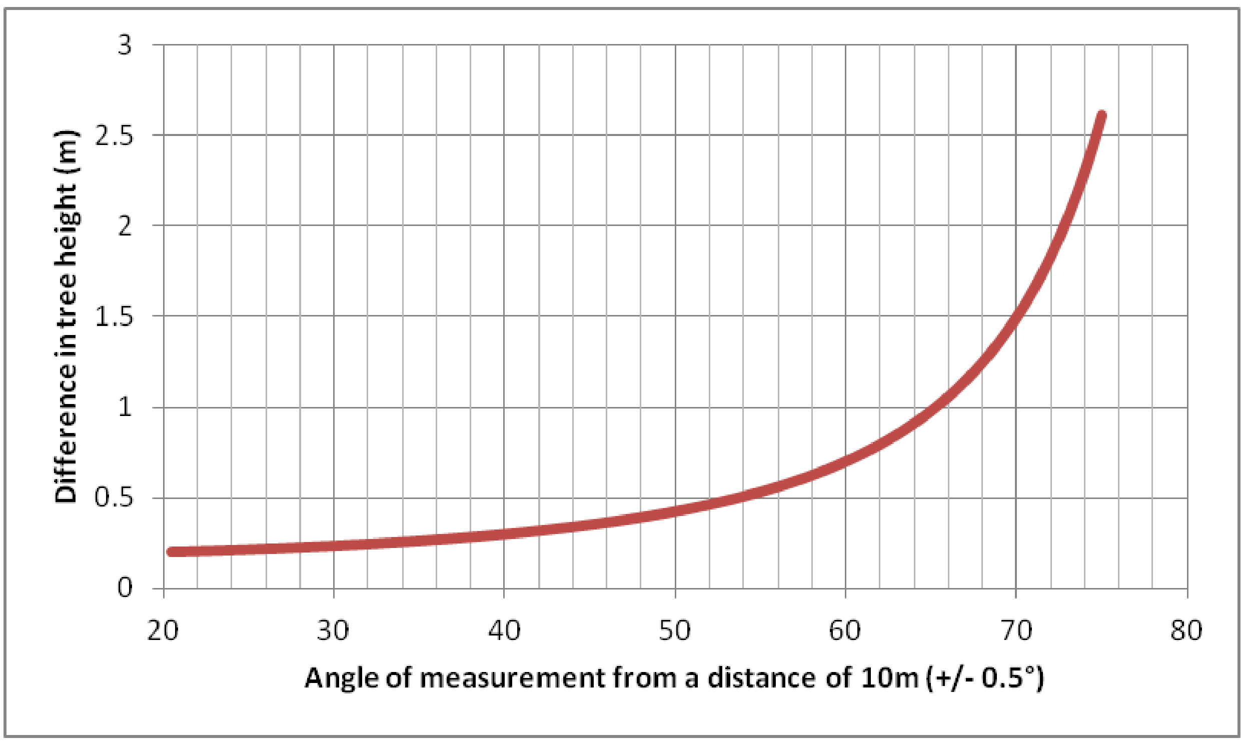

Figure 4-10: Assessing the difference in tree height (m) in relation to clinometer angle change (+/-0.5°). Using a set distance of 10m angles ranging from 20° to 75.5° were used to calculate simulated tree heights. The difference in calculated height by using two different angles, increasing by 0.5°, was then quantified. This difference was used to indicate the potential accuracy of a clinometer to measure tree height. Below 40° the difference in calculated height is less than +/-0.3m while over 65° the difference in height is over +/-1m.

The resulting number of trees surveyed in Higher Broughton was 87 (20% of the total number of trees) and 17 of those trees had been removed (24% of the sample). In Lower Kersal the variables of 30 trees were recorded (20% of the total number of trees included those <3m in 2009) and none of these trees had been removed. The total number of trees surveyed in Peel Park was 100 and 14 had been removed (15% of the sample). Finally, in Kersal only 92 trees were surveyed (instead of 100) due to the large number of trees in private gardens and/or unsuitably located. Out of the 92 trees recorded 10 had been removed (11%) (see Appendix A).

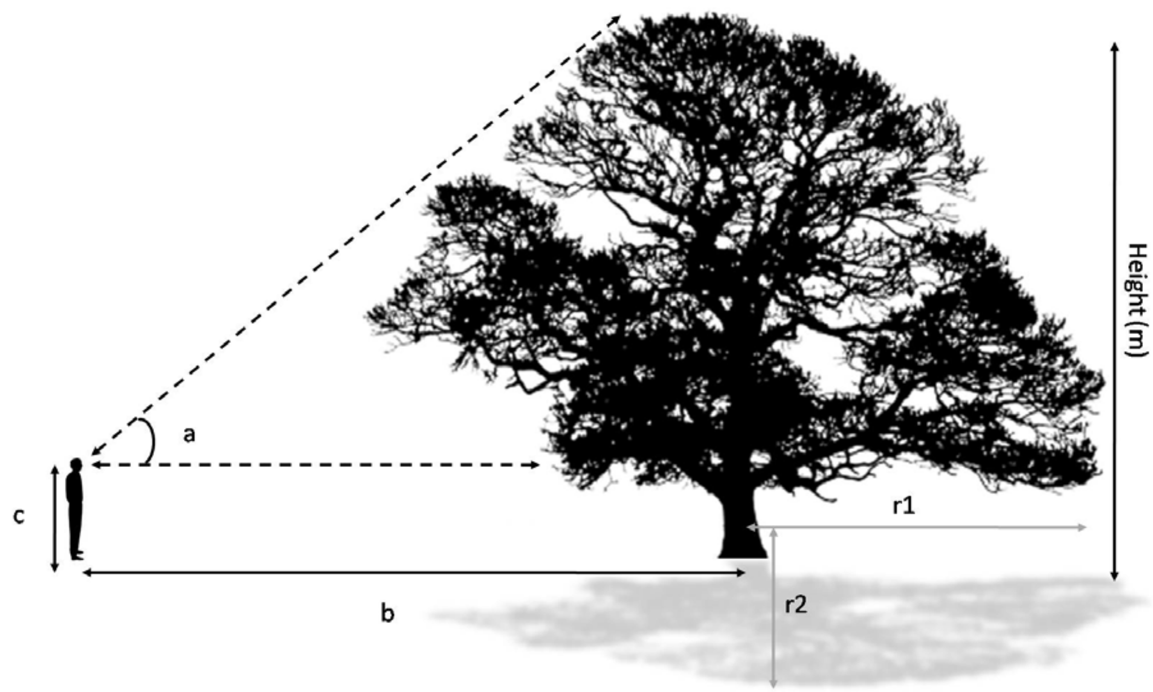

Figure 4-11: Diagram of the variables recorded in the field to calculate tree height (m) and area (m²).
To calculate the height of a tree 3 variables are needed – the angle from eye height to the top of the canopy (*a*) measured using a clinometer, the distance from the recorder with the clinometer to the centre of the tree trunk (*b*) measured using a 1m pace-stick, and the eye level height of the recorder (*c*). To gain an approximate measure of tree canopy area two radius measurements are needed, measured at right angles to each other (r1 and r2).

The field sample data were imported into an Excel spreadsheet and separated based on the natural breaks in the 2009 tree height data thus creating the ‘tree height classes’ shrub layer (<3m, only present in Lower Kersal), small trees (3m – 7.2m), medium trees (7.2m-17.1m), and large trees (>17.1m). The data for each tree height class included tree ID, location, genus, 2009 canopy area, 2009 height, recorder’s distance from tree, recorded angle, the radian of the angle (used to calculate equation 4-1 within Excel), recorder’s eye height (adjusted to incorporate tree substrate), 2013 height, radius measurement 1, radius measurement 2, radius average, 2013 canopy area, difference in height, and finally difference in area. The following trigonometric equation was used to calculate the 2013 height;

$$H = (tan\, a \times b) + c$$

Equation 4-1: Trigonometric equation to measure tree height (m).

where *a* is the angle from the recorders eye to the top of the canopy (converted to radians), *b* is the distance (m) from the recorder to the centre of the tree trunk, and *c* is the recorder’s eye height (m) (also see Fig 5-2). To calculate the approximate area of the tree canopy the average measurement of the radiuses recorded in the field were used in the equation;

$$A = \pi r^2$$

Equation 4-2: Area of a circle.

The mean and median values of the difference between the 2009 and 2013 height and area results were then calculated (referred to as the ‘change data’). As the mean or median changes were calculated for small, medium, and large trees a better understanding of how the UTI develops and changes at various stages of growth and for different species can be gained. Therefore, changes were applied depending on what vertical classification the tree fell under in 2009.

To decide whether the mean or median values should be applied to the 2009 canopies (and thus create the 2013 canopies), the change datas normality were assessed. By using the statistical software SPSS 20 the Shapiro-Wilk test was applied to each of the tree height classes for each tree canopy sample plot. If the data was shown to exhibit a normal distribution (i.e.

parametric data), the mean value of the change data would be applied to the 2009 canopy data. Conversely, if the data were shown to not follow a normal distribution (i.e. non-parametric) the median values were used.

The Higher Broughton small trees height change data were not normally distributed (SW = 0.227, $p \leq 0.001$) whereas the medium and large trees height change data were normally distributed (SW = 0.964, p = 0.332 and SW = 0.964, p = 0.637). The area change data for Higher Broughton also followed the same pattern (small SW = 0.620, $p \leq 0.001$; medium SW = 0.955, p = 0.182; Large SW = 0.917, p = 0.443). For Lower Kersal no large trees and only two medium trees were sampled (due to limited access to private land). Therefore, only the small trees change data were assessed for normality. The small trees height change data were not normally distributed (SW = 0.577, $p \leq 0.001$) as were the small trees area change data (SW = 0.612, $p \leq 0.001$). The Peel Park small and medium trees height change data were not normally distributed (SW = 0.879, p = 0.021 and SW = 0.930, p = 0.002), yet the large trees height change data were normally distributed (SW = 0.937, p = 0.612). The Peel Park area change data followed the same pattern (small SW = 0.817, p = 0.002; medium SW = 0.905, $p \leq 0.001$; large SW = 0.925, p = 0.505). Finally, the Kersal small and large tree height change data were normally distributed (SW = 0.820, p = 0.064 and SW = 0.932, p = 0.07) and the medium trees height change data were not normally distributed (SW = 0.939, p = 0.017). Like the Higher Broughton and Peel Park change data, the Kersal area change data followed the same pattern as the height change data (small SW = 0.906, p = 0.366; medium SW = 0.930, p = 0.008; large SW = 0.951, p = 0.214).

Table 4-3: Average change in tree height (m) and area (m^2) exhibited by trees from 2009 to 2013.

	Shrub Layer (<3m)		*Small trees (3m-7.2m)*		*Medium trees (7.2m-17.1m)*		*Large Trees (>17.1m)*	
	Height (m)	*Area (m^2)*	*Height (m)*	*Area (m^2)*	*Height (m)*	*Area (m^2)*	*Height (m)*	*Area (m^2)*
Higher Broughton	0[a]	0[a]	1 (median)	1.3 (median)	2.1 (mean)	3.2 (mean)	3.7 (mean)	14.1 (mean)
Lower Kersal	3.7 (median)	4.6 (median)	1 (median)	1.5 (median)	2.1[c]	-14[c]	3.7[b]	14.1[b]
Peel Park	0[a]	0[a]	2 (median)	2.7 (median)	2.8 (median)	-5.4 (median)	3.9 (mean)	27.4 (mean)
Kersal	0[a]	0[a]	1.3 (mean)	-14.4 (mean)	0.4 (median)	-24 (median)	0.4 (mean)	-57.2 (mean)

[a]trees recorded in 2009 as less than 3m were shrubs and hedges and none were more than 3m in 2013

[b]no large trees could be sampled in the field and therefore the Higher Broughton results were used as a proxy

The changes in height and area of the UTI within the four tree canopy sample plots were calculated using data collected in the field. All trees increased in height while there are five instances of tree canopies decreasing in size over time. The data generated in table 4-2 will be applied to the 2009 UTI in order to create a new 2013 UTI dataset. The distribution of the tree height and area change data dictated whether a mean or median value should be applied.

The results of the 2013 tree survey and the normality tests of the change data provided various negative and positive values which were to be applied to the 2009 canopies (Table 4-3). The 2013 NBH canopies could then be created by implementing those values in Table 4-3 within ArcGIS. Once these canopies were formed a 2013 connectivity analysis could be undertaken.

4.2.5 Describing NBH canopies in 2013

The 2009 vector polygons representing individual or small groups of trees in the river valley study area were used as a base-line to create a 2013 UTI dataset. The polygons were not edited or transformed to represent contiguous habitat patches (i.e. the original data were not interpolated, stratified, or percolated to create the NBH canopies as in Chapter 3.3). Each polygon had a unique ID number, area value, and height value joined to it (as used in the field sampling, Chapter 4.2.4). Trees less than 3m high were removed from the dataset except for the Lower Kersal sample plot. This was due to the fact that, while conducting the second field survey (early spring 2014), it was found that only the trees in Lower Kersal which were reported to be less than 3m high in 2009 were actually young, sapling trees. Within the other tree canopy sample plots the overwhelming majority of trees less than 3m high in 2009 were actually shrubs, hedges, and or bushes.

The polygons that were within the tree canopy sample plot buffers (circles with a 500m diameter) were selected and exported as new shapefiles (this method of selecting was used rather than clipping the data to the 500m diameter as any polygons at the edge needed to be whole so that the relevant changes could be applied). Therefore, four shapefiles were created representing trees within and touching the tree canopy sample plot boundaries.

The attribute tables of the tree canopy sample plot canopy data were joined to an Excel spreadsheet containing the presence or absence of a focal tree in 2013. Under the field ‘State’ a value of 1 next to a tree’s ID number meant it was present; the value 0 meant it had been removed. The polygons with the value 1 were then selected and eventually exported as a new shapefile representing trees that were extant in 2013. As the change in area and height were to be applied to shrub, small, medium, and large trees the selection process had to also take height into consideration. Therefore, selection algorithms had to be used within the ‘select by attribute’ tool in ArcGIS. To find and select large trees the algorithm *selection = ‘State = 1’ AND ‘Height >= 17.1’* was used. For medium trees the algorithm was *selection = ‘State = 1’ AND ‘Height >=7.1 AND <17.1’.* For small trees within Higher Broughton, Peel Park, and Kersal the algorithm used was *selection = ‘State = 1’ AND ‘Height >=3 AND <7.2’* while for Lower Kersal the algorithm was simply *selection = ‘State = 1’ AND ‘Height<7.2’* (as trees below 3m high – the shrub layer – are to be included in further analysis). When exported, the selected polygons created new shapefiles representing the small, medium, and large trees existing within the tree canopy sample plots in 2013 (referred to as the ‘existence shapefiles’).

To apply the 2013 area and height changes, the attribute tables of each of the newly created existence shapefiles had to be exported to an Excel spreadsheet. New fields were then added to the attribute tables, namely ‘Area_13’, ‘Radius_09’, ‘Radius_13’, ‘Buffer’, and ‘Height_13’. The relevant area and height changes were added or subtracted from the 2009 values (under the fields ‘Area’ and ‘Height’) to calculate Area_13 and Height_13 (see Table 4-3 for actual values). In regards to area, if the subtracted area value was larger than the 2009 value, thus essentially eliminating the canopy, no change was applied. The radius of the circular polygons representing trees in 2009 and 2013 were then calculated using the equation:

$$R = \sqrt{\frac{A}{\pi}}$$

Equation 4-3: Radius of a circle.

The 2009 radiuses were subtracted from the 2013 radiuses to calculate the difference between the two, thus providing a value for the buffer field. This value could then be used to create a buffer around the 2009 polygons thus creating 2013 polygons. A negative buffer would decrease the size of the 2009 polygons while a positive value would increase them (if no change in area was applied as doing so would remove the polygon, the buffer value was set to 0). The 2013 area, height, and buffer values were calculated for the small, medium, and large trees within each of the tree canopy sample plots, plus for the shrub layer in Lower Kersal. Therefore a total of 13 tables were created which were then joined to the corresponding existence shapefile.

The tool Union was then used to spatially join the existence shapefiles within each tree canopy sample plot back together (with the relevant trees removed and the buffer value and 2013 height attached to the attribute table). The canopies were then stratified based on the NBH values, selecting the height value from the field Height_13. The selected polygons were then exported as '2013 trees ≥3m', '2013 trees ≥7.2m', and '2013 trees ≥17.1m' shapefiles (2013 maximum height was 32.1m). A buffer was applied to each polygon within each of these shapefiles using Hawth's Tools 3.27 (Bayer, 2004). The value of the buffer was taken from the field Buffer. The subsequent shapefiles represented the UTI present in 2013 which had been vertically stratified and increased or decreased in area based on field observations. These '2013 NBH buffers' were then used to create the '2013 NBH canopies' using the methods outlined in Chapter 3 (e.g. converted the various 2013 NBH buffer shapefiles to raster, used region group to join touching cells, and then converted back to vector). Each tree canopy sample plot contained the 3 NBH canopies ‘canopy ≥3m’, ‘canopy ≥7.2m.’ and ‘canopy ≥17.1m’ totalling in 12 NBH canopies to be used in the 2013 connectivity analysis.

4.2.6 Connectivity analysis

The NBH canopies of the river valley study area in 2005 and 2009 as well as the NBH canopies of the tree canopy sample plots (Peel Park, Kersal, Higher Broughton, and Lower Kersal) in 2005, 2009, and 2013 were incorporated into a potential landscape connectivity model (Figure 4-12). Due to the strength of graph structures in assessing landscape connectivity (section 2.3) the NBH canopies of the river valley study area (2005/9) and tree canopy sample plots (2005/9/13) were redefined as a set of nodes (tree patches) and links (direct, Euclidean dispersal from one node to another) using the ArcGIS extension Conefor Inputs (Jenness, 2011). All tree patches within each NBH canopy were assimilated into a node text file. This text file contained two columns; column 1 contained node ID number, column 2 contained node area (m^2) (both synonymous with NBH canopy polygon values). Simultaneously, the Euclidean distance (m) from one node to another, limited to a corresponding gap-crossing capability threshold (i.e. 30 – 200m in 20m intervals), was incorporated into a link text file. This text file contained 3 columns; column 1 related to node *A’s* ID number, column 2 related to node *B’s* ID number, and column 3 contain the distance between the two. When making the node and link files, analysis was restricted to a set distance based on passerine gap-crossing capabilities. Therefore, only the polygons (i.e. tree patches) within a specified distance from a focal polygon were included in the files. In other words, for each NBH canopy only the polygons within 30, 60, 90, 120, 150, 180, and 200m of each other were incorporated into a node file. The distances between these nodes were therefore always limited to a given distance. As a result, for each NBH canopy 7 node and 7 link files were created, which meant there were 7 graph structures created for each NBH layer. This resulted in 21 graph structures to be used within the landscape connectivity analysis, referred to as the NBH landscape graphs (Figure 4-12).

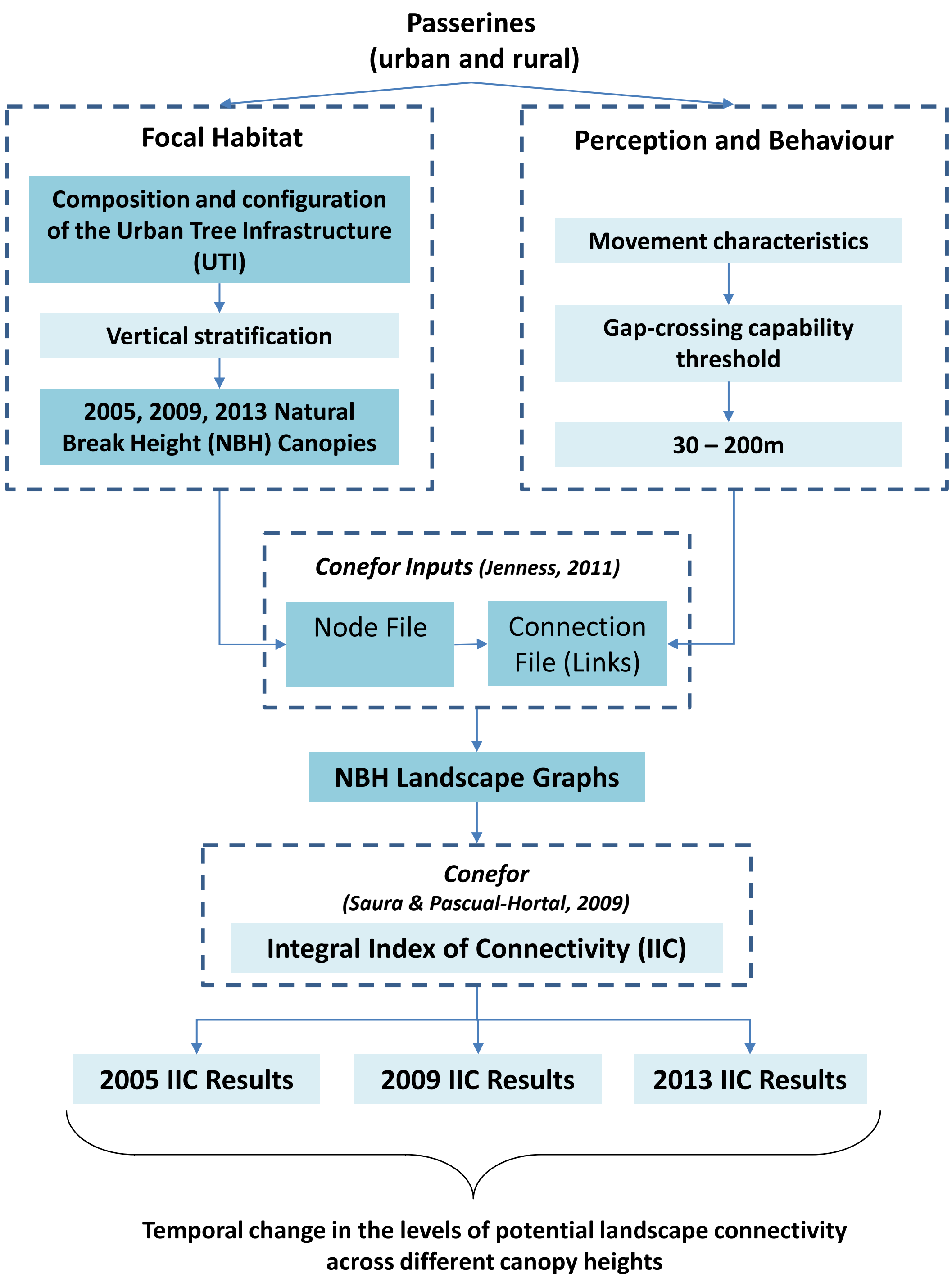

Figure 4-12: Schematic outline of the landscape connectivity analysis.

For the potential landscape connectivity model to work two main inputs are need; 1) a useful description of the focal organism's habitat (i.e. the UTI), and 2) the focal organism's movement characteristics (characterised here as ability to cross canopy gaps). These inputs can then be used to create a landscape graph by generating nodes representative of habitat and links representative of movement. A connectivity index (e.g. integral index of connectivity) can then be used to calculate the potential level of connectivity exhibited by this landscape graph.

Landscape connectivity indices which only require a description of landscape composition and movement distances have been shown to provide useful data on potential landscape connectivity (Chapter 2.3). Pascual-Hortal and Saura (2006) compared 10 graph-based landscape connectivity indices in terms of their usefulness and efficiency for informing landscape conservation planning and concluded that the Integral index of connectivity (IIC) performed the best in this regard. IIC is a graph theory and habitat availability concept based metric (see sections 2.3 and 3.5) and it has the ability to provide data on the importance of particular habitat patches for connectivity as well as providing an overall landscape connectivity measure (Pascual-Hortal & Saura, 2006).

However, in Saura and Pascual-Hortal's (2007) paper another index was compared to IIC – probability of connectivity (PC). Both indices are based on habitat availability and graph theory yet PC is a probability based index while IIC is a binary index. Due to its probabilistic approach, in which the probability of movement among patches is modelled as a decreasing function of interpatch distance, PC provides a more detailed representation of interpatch connections, (Saura and Pascual-Hortal, 2007). However, for the research presented in this thesis IIC was chosen over PC for several reasons. Firstly, Saura and Pascual-Hortal (2007) suggest that IIC should be used instead of PC when data needed to define a probable dispersal value (P_{ij}) is scarce. The research in this thesis concentrates on passerines and therefore not on individual species. Therefore, specific data on probable movement or dispersal characteristics isn't needed, instead general movement capabilities are. The reason for using passerines to inform the landscape connectivity model is that the UTI is a multifunctioning social-ecological system and although landscape models need to be ecologically relevant (i.e. incorporate organism perceptions and relate to ecological function) they cannot be too species specific so that they become operationally constricted and biased. In other words, landscape models which are biased to a given focal species could only provide useable information, in regards to landscape conservation and planning, on that species. Furthermore, Harris and Reed (2002) found in their review of bird gap-crossing studies (both passerines and non-passerines) that there were no apparent thresholds in gap-crossing behaviour, instead the species within the various studies exhibited a willingness to cross gaps up to a maximum distance – beyond which they wouldn't cross. Consequently, the use of a binary index which sharply connects or disconnects patches depending on a maximum distance can be justified. Furthermore, the literature review on passerine gap-crossing capability thresholds in this thesis (section 2.5) revealed that there was no consensus on gap-crossing probabilities (Figure 2-4), or rather on the relationship with which gap-crossing willingness decreases in line with gap distance. Consequently, any attempt to apply a dispersal probability value in the potential connectivity model would be arbitrary.

Therefore, the landscape connectivity index ‘Integral Index of Connectivity’ (IIC) was used to assess the level of potential connectivity for passerine birds within each NBH graph. IIC ranges from 0 to 1, increases with improved connectivity and is given by the equation:

$$IIC = \frac{\sum_{i=1}^{n}\sum_{j=1}^{n}\frac{a_i . a_j}{1 + nl_{ij}}}{A_L^2}$$

Equation 4-4: Integral Index of Connectivity (IIC).

where n is the total number of nodes in the landscape graph, a_i and a_j are the area (m^2) of node i and j, nl_{ij} is the number of links (direct, topological distance limited to a gap-crossing capability threshold) in the shortest path between nodes i and j, and A_L is the overall landscape area (m^2). When $I = j$ then $nl_{ij} = 0$ as no links are needed to reach a given patch from itself. When IIC = 1 then the whole landscape is occupied by dense tree cover (Saura & Pascual-Hortal, 2007b).

Using the landscape connectivity software Conefor (Saura & Torne, 2009), IIC was run using the node and link files associated with each NBH canopy (i.e. to create an NBH graph, Figure 4-12). From the list of binary landscape connectivity indices within Conefor IIC was selected. For the 2005 and 2009 NBH graphs representing the UTI of the river valley study area, A_L was set to 10,326,990m^2 – the whole area of the river valley study area. For the 2005, 2009, and 2013 NBH graphs representing the tree canopy sample plot’s UTI, A_L equalled 1,96,247m^2 (the area of a 500m circle). As landscape connectivity was assessed across spatial and temporal dimensions, a large number of IIC models were ran. For the river valley study area (limited to 2005 and 2009) seven distance threshold graphs for each of the three NBH canopies were used in the IIC analysis (totalling 42 IIC calculations). In addition, seven distance threshold graphs for the three NBH canopies within each of the four tree canopy sample plots across three points in time (2005, 2009, and 2013) were used in the IIC analysis (totalling in 252 IIC calculations). Therefore, 294 models were run in total, providing a sufficient amount of data to assess connectivity change across spatial and temporal dimensions.

4.3 Results

4.3.1 UTI connectivity 2005

The vertical stratification of the river valley study area's 2009 UTI revealed three canopy layers. These canopies were defined using the lower natural break height (NBH) of 3m, the middle NBH of 7.2m, and the upper NBH of 17.1m. The river valley study area's 2005 UTI was also stratified using the same breaks in height data (for direct comparisons to be made), thus creating the 2005 canopy ≥3m, canopy ≥7.2m, and canopy ≥17.1m (Figure 4-13). As a consequence, the 2005 canopy ≥3m contained all trees above 3m and covered an area of approximately 161ha (~ 16% of total landcover). The 2005 canopy ≥7.2m canopy contained all trees above 7.2m and covered around 13% of the study landscape (~136ha), making up around 85% of the overall tree canopy cover. The 2005 canopy ≥17.1m contained all trees above 17.1m and covered around 4% of the study area (~36ha), comprising around 23% of the overall tree canopy cover.

The canopy ≥17.1m was found to be least connected (IIC = 0.00003 – 0.00016) followed by the canopy ≥7.2m (IIC = 0.0010 – 0.0031) and then the canopy ≥3m (IIC = 0.0014 – 0.0044) (Figure 4-14). There was a strong correlation between an increase in the gap-crossing capability threshold (m) and an increase in potential landscape connectivity (IIC) (R^2 = 0.98 – 0.99). The greatest rate of increase in potential connectivity was exhibited by the canopy ≥17.1m as the IIC value at a 200m gap-crossing capability threshold is around 5 times higher than at a 30m gap-crossing capability threshold. Comparatively, both the ≥3m and ≥7.2m canopy IIC values are only 3 times higher at 200m then 30m. Furthermore, the rate of increase in the connectivity of the canopy ≥17.1m began to decrease after 120m compared to the ≥3m and ≥7.2m canopies which began to decrease after 90m (Figure 4-14). The biggest increase in connectivity can be seen between the 30m and 60m gap-crossing capability thresholds for the canopy ≥17.1m (116% increase), the canopy ≥7.2m (54% increase), and the canopy ≥3m (60% increase) (see Appendix B).

At the 30m gap-crossing gap-grossing capability threshold, the canopy ≥7.2m was around 30 times more connected then the canopy ≥17.1m (canopy ≥17.1m IIC = 0.00003, canopy ≥7.2m IIC = 0.0010). The rate of difference decreases with gap-crossing capability threshold so that at 200m, the canopy ≥7.2m was around 19 times more connected then the canopy ≥17.1m (canopy ≥17.1m IIC = 0.00016, canopy ≥7.2m IIC = 0.0031). This rise in connectivity did not occur between the canopy ≥3m and canopy ≥7.2m. The canopy ≥3m was around 1.5 times more connected than the canopy ≥7.2m at 30m and only 1.4 times more connected at a gap-crossing capability threshold of 200m (see Appendix B).

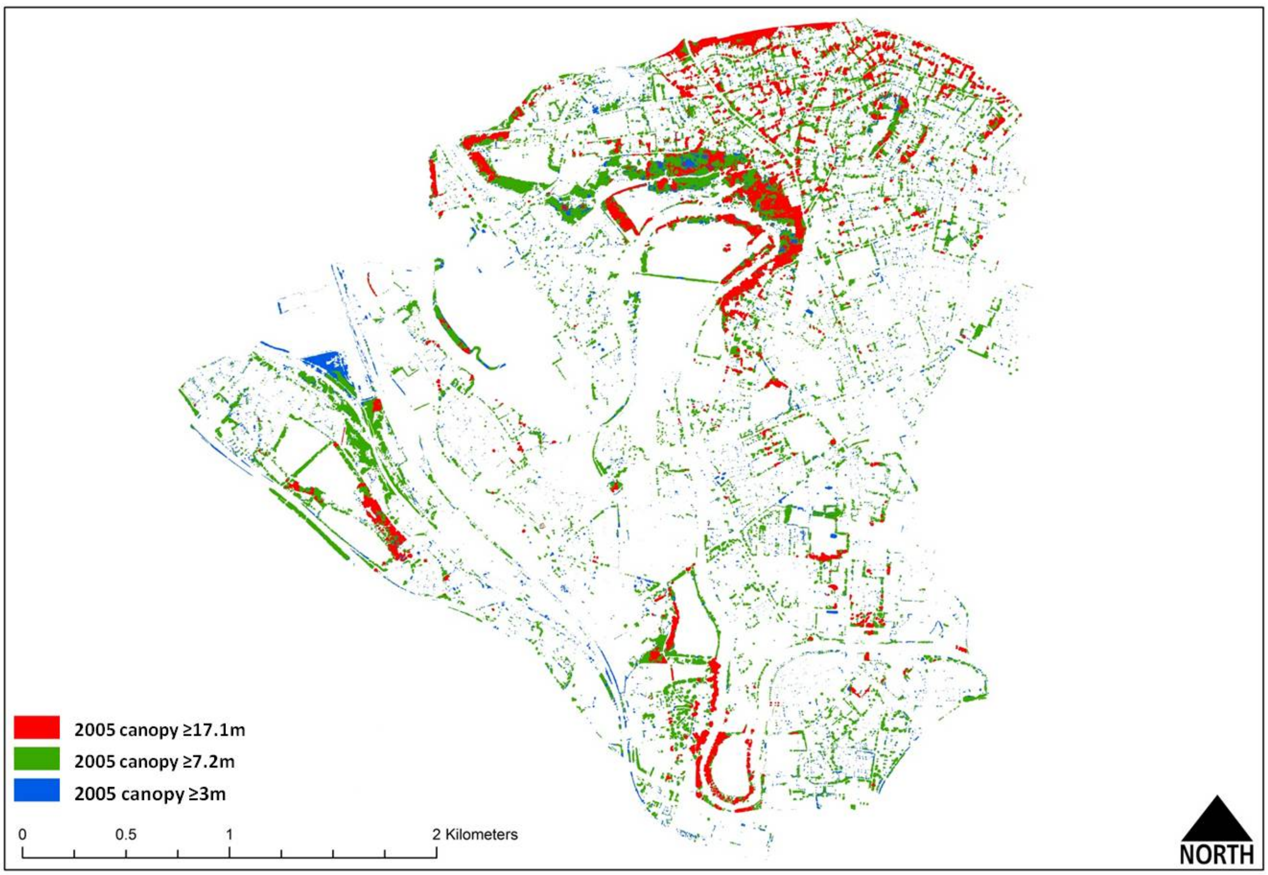

Figure 4-13: The river valley study area's 2005 natural break height (NBH) canopies.
The canopy ≥17.1m contains all trees above the upper natural break in the height data (17.1m), the canopy ≥7.2m contains all trees above the middle natural break in the height data (7.2m), and the canopy ≥3m contains all trees above the lower break in the height data (3m – manually set so not natural break but termed as such to maintain nomenclature).

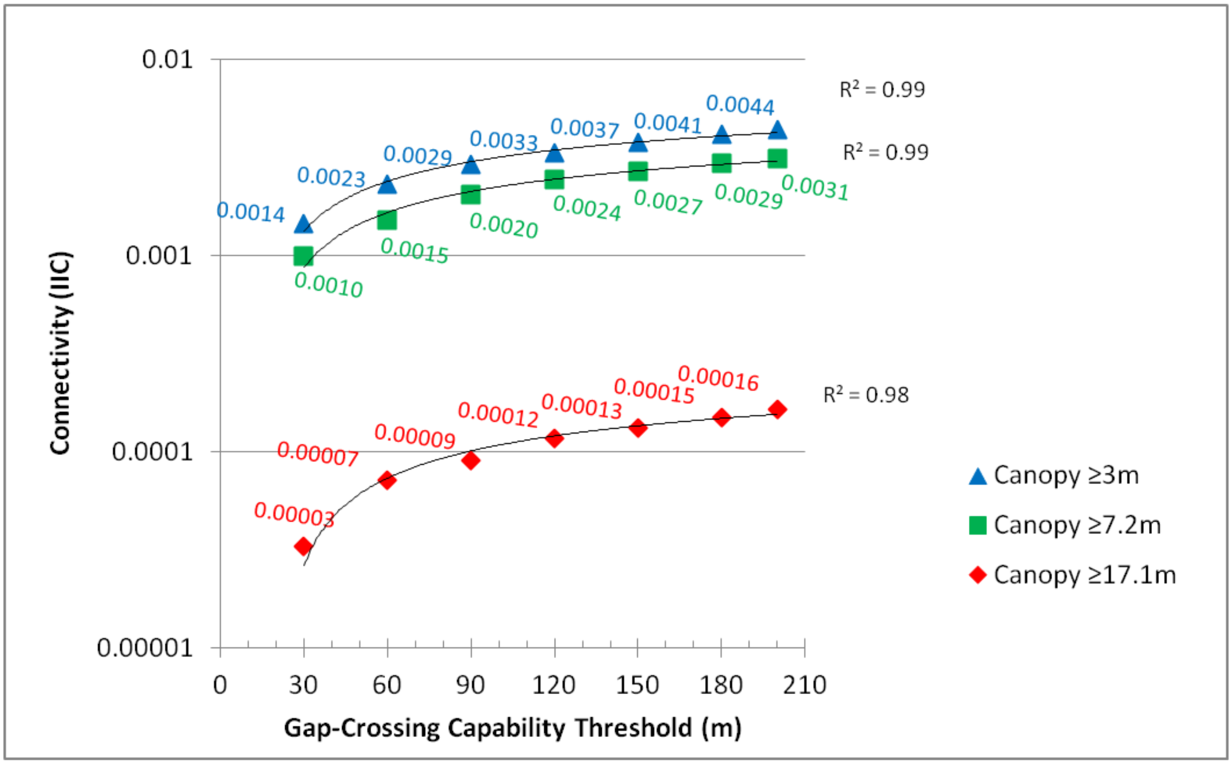

Figure 4-14: Levels of connectivity (IIC) exhibited by the 2005 natural break height (NBH) canopies.
The canopy ≥17.1m (trees above 17.1m) is the least connected canopy as IIC = 0.00003 at 30m, increasing by 397% to 0.00016 at 200m. The canopy ≥7.2m (trees above 7.2m) is the second least connected canopy as IIC = 0.0010 at 30m, increasing, slower than the canopy ≥17.1m, by 214% to 0.0031 at 200m. The most connected canopy is the canopy ≥3m (all trees above 3m) as IIC = 0.0014 increasing, slower than the other canopies, by 201% to 0.0044.

The connectivity of the NBH canopies of each tree canopy sample plot followed similar patterns as the river survey area's NBH canopies (Figure 4-15). However, due to the finer spatial scale of the analysis, further detailed results and patterns were revealed. The highest connectivity values were calculated for the Kersal sample plot's canopy ≥3m (IIC = 0.052), while the lowest IIC scores were attained for the Lower Kersal sample plot's UTI (max IIC = 0.000075). The NBH canopies of the river valley study area were more connected than both the low density canopy cover sample plots Higher Broughton sample plot and Lower Kersal sample plot. In contrast, the high density canopy cover areas, Peel Park sample plot and Kersal sample plot, were much more connected than the river valley study area (Appendix B and C, Figure 4-15).

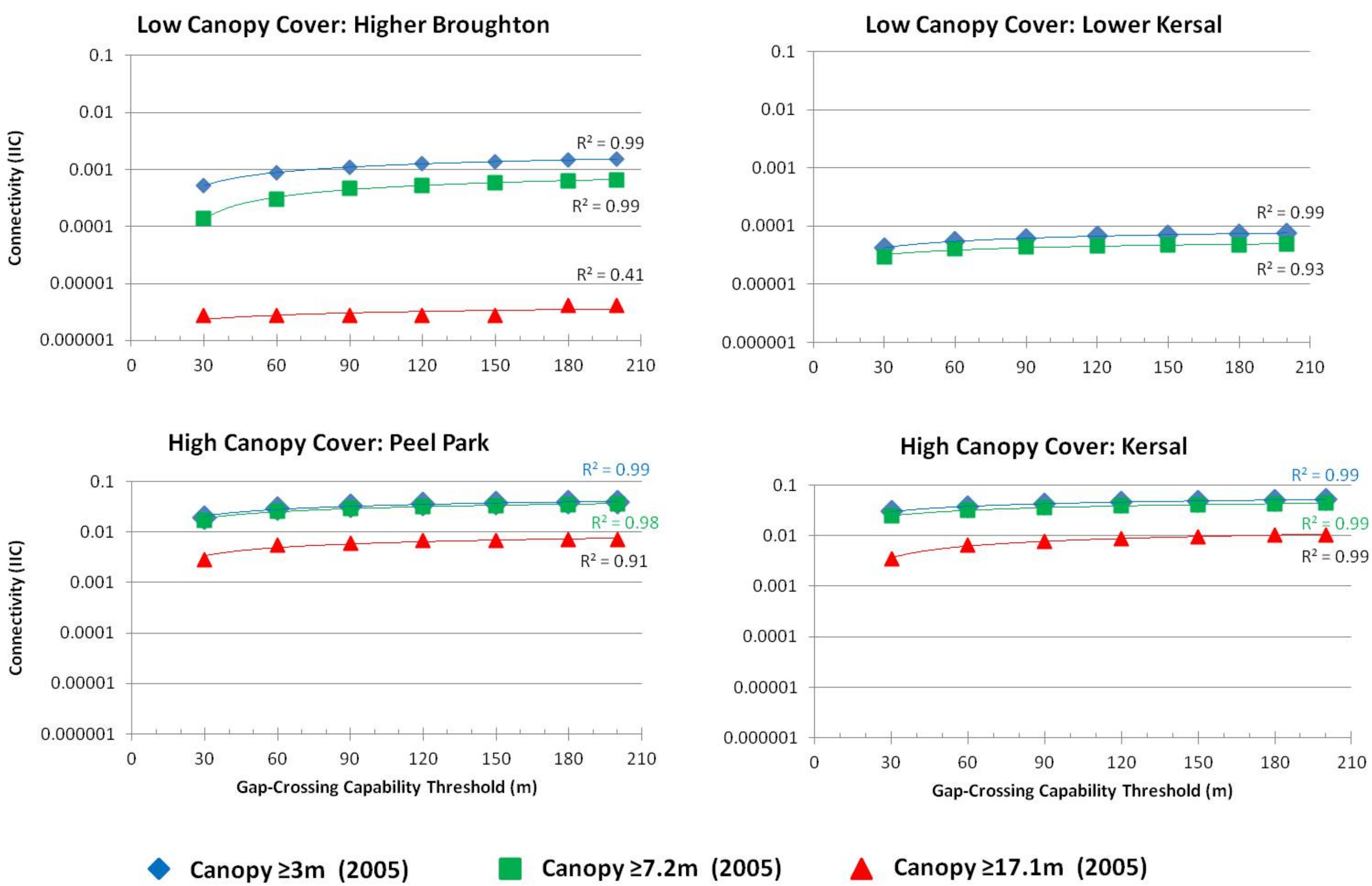

Figure 4-15: Connectivity results (IIC) for the tree canopy sample plots in 2005.

For all tree canopy sample plots the canopy ≥17.1m is the least connected canopy. As the distance a passerine is able to cross increases so does connectivity. The rate at which this happens begins to decrease after a 60, 90, or 120m threshold is reached – the majority of cases being between 90 and 120m. The biggest increase in connectivity occurs as ability to cross gaps increases from 30 to 60m (minus the canopy ≥17.1m for the Higher Broughton and Lower Kersal sample plots). The percentage change in connectivity between a 30 and 60m gap-crossing capability threshold, depending on NBH canopy, ranges from 23-122%. No canopy ≥17.1m was present in the Lower Kersal sample plot in 2005.

The Higher Broughton survey area's 2005 UTI covered approximately 6% (~12,000m^2) of the study landscape. The canopy ≥17.1m was the least connected canopy in Higher Broughton (IIC = 0.000003 – 0.000004) and as gap-crossing capability threshold increased the rate of increase in IIC was negligible so that the canopy was only 52% more connected at 200m then 30m . Therefore, willingness to cross gaps did not especially affect connectivity. The IIC results for Higher Broughton's canopy ≥7.2m were between 0.00014 and 0.00066, which meant it was between 45 (30m gap-crossing capability threshold) and 164 (200m gap-crossing capability threshold) times more connected than the canopy ≥17.1m. The rate of increase in connectivity began to slow after probability to cross gaps passed 90m. The same threshold existed within the canopy ≥3m (IIC = 0.0005 – 0.0015) which was between 2 (200m gap-crossing capability threshold) and 4 (30m gap-crossing capability threshold) times more connected than the canopy ≥7.2m (Figure 4-15).

In 2005, Lower Kersal's UTI covered around 2% of the landscape (~3,000m^2). There were no trees 17.1m or above within the sample plot in 2005. The connectivity scores for Lower Kersal's canopy ≥7.2m ranged from 0.000030 (30m) to 0.000048 (200m). As gap-crossing capability threshold increased so did IIC, however the rate of increase for the canopy ≥7.2m began to decelerate after 60m – after which willingness/ability to cross gaps did not notably affect connectivity. The canopy ≥3m revealed IIC scores between 0.000042 to 0.000075, increasing with gap-crossing capability threshold, and the rate of change of which decreasing after 120m. Therefore, the canopy ≥3m was between 1.4 (30m gap-crossing capability threshold) and 16 (200m gap-crossing capability threshold) times more connected than the canopy ≥7.2m (Figure 4-15).

The first, high-canopy-cover sample plot, Peel Park, had an UTI that covered around 29% of the landscape (~56,000m^2). In 2005, Peel Park had very high IIC results, higher than the river valley study area, so much so that even the canopy ≥17.1m of Peel Park was more connected than all of the trees in the river valley study area (except for at the 30m gap-crossing capability threshold) (Figure 4-14 and Figure 4-15). The canopy ≥17.1m connectivity results ranged from 0.0028 to 0.0071 and increased in line with gap-crossing capability threshold. The rate of change in IIC score began to slow down after the 60m gap-crossing capability threshold. A 90m threshold was observed in Peel Park's ≥7.2m (IIC = 0.0173 – 0.0356) and ≥3m (IIC = 0.0196 – 0.0395) canopies. The biggest increase in connectivity for all canopies was identified between the 30m and 60m gap-crossing capability thresholds. The canopy ≥7.2m was between 5 (200m) and 6 (30m) times more connected than the canopy ≥17.1m while the canopy ≥3m was only ever around 1.1 times more connected than the canopy ≥7.2m (Figure 4-15).

The final tree canopy sample plot, Kersal, had a high-canopy comprising approximately 34% of the landscape (~66,000m^2). Kersal had the highest connectivity values of all tree canopy sample plots as well as the river valley study area (only the canopy ≥17.1m at 3m had a lower IIC value than the river valley study area's canopy ≥3m) (Figure 4-14). The IIC scores of Kersal's canopy ≥17.1m were valued between 0.0034 and 0.0104, increasing with gap-crossing capability threshold. The canopy ≥7.2m was between 4 (200m) and 7 (30m) times more connected than the canopy ≥17.1m (IIC = 0.024 – 0.044). The rate of increase in connectivity for both canopies decelerates after the gap-crossing capability threshold of 90m. Kersal's canopy ≥3m was only ever between 1.1 (200m) and 1.2 (30m) times more connected than the canopy ≥7.2m as IIC results were between 0.030 and 0.052. A rate of increase threshold for the canopy ≥3m was also identified at the 90m gap-crossing capability threshold. For all canopies, the biggest increase in connectivity was between the 30m and 60m gap-crossing capability thresholds (Figure 4-15 and see Appendix C).

4.3.2 UTI connectivity 2009

The vertical stratification of the 2009 UTI revealed three NBH canopy layers, distributed throughout the river valley study area landscape (Figure 4-16). Approximately 21 % (~213ha) of the river valley study area was covered by tree canopies above 3m (canopy ≥3m). Trees above 7.2m (canopy ≥7.2m) covered around 17% (~175ha) of the river valley study area, making up 82.3% of the overall tree cover. The tallest trees - canopy ≥17.1m - comprised 4% (~38ha) of the total landcover, which was 18% of the overall canopy cover. The maximum tree height was 34.9m.

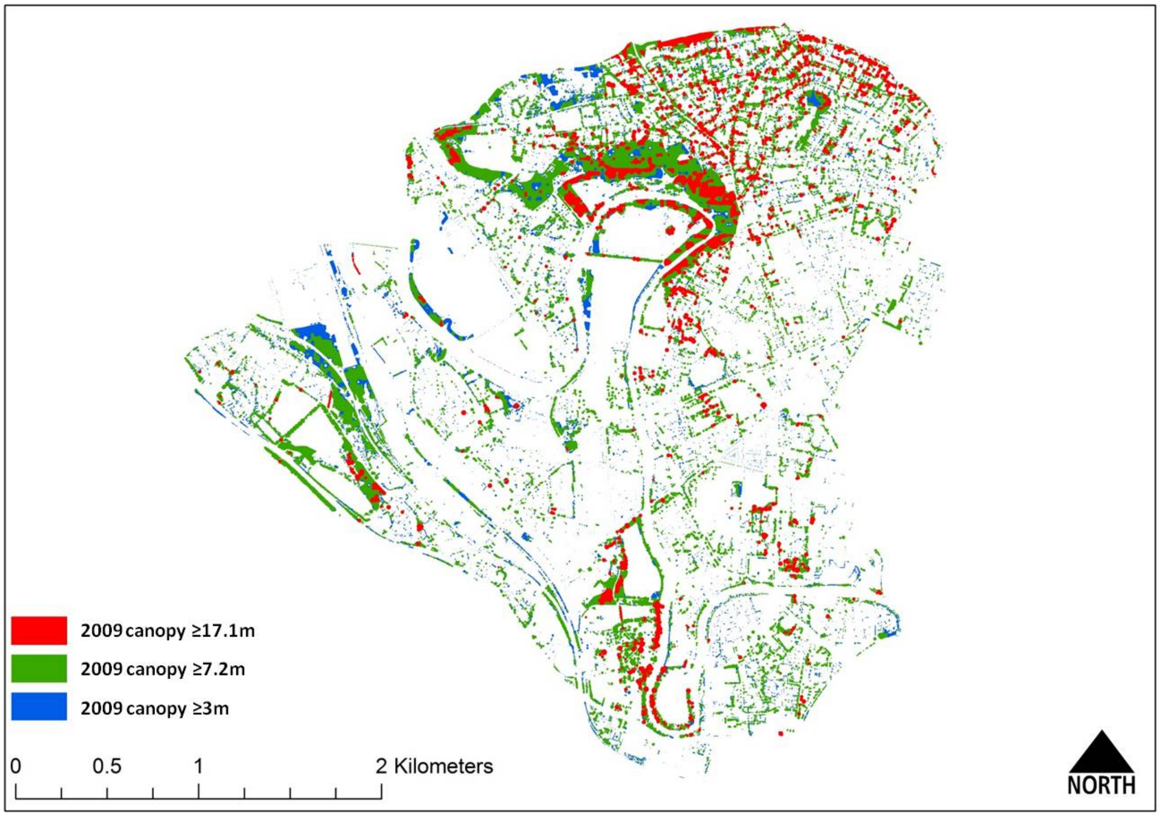

Figure 4-16: The 2009 Natural Break Height Canopies.

The canopy ≥17.1m contains all trees above the upper natural break in the height data (17.1m), the canopy ≥7.2m contains all trees above the middle natural break in the height data (7.2m), and the canopy ≥3m contains all trees above 3m. The majority of trees above 17.1m exist towards the north of the river valley study area in Kersal as well as a few patches in the south within the University of Salford's Peel Park Campus and around the River Irwell. Trees above 7.2m are evenly distributed across the study area. Trees between 3m and 7.2m also exist throughout the study area as well as in larger patches to the west in Pendleton/Lower Kersal, to the north in Kersal and around the River Irwell which cuts through the river valley study area.

The canopy ≥17.1m was the least connected canopy (IIC = 0.00002 – 0.0002) followed by the canopy ≥7.2m (IIC = 0.0021 – 0.0057) then the canopy ≥3m (IIC = 0.0043 – 0.0089) (Figure 4-17). There was a strong correlation between an increase in the gap crossing capability threshold (m) and an increase in connectivity (IIC) (R^2 = 0.94 – 0.99). The greatest rate of increase in connectivity was exhibited by the canopy ≥17.1m as the IIC score at the 200m gap-crossing capability threshold was around 11 times more connected than at the 30m gap-crossing capability threshold. The canopy ≥7.2m was around 3 times more connected while the canopy ≥3m was around 2 times more connected at the 200m gap-crossing capability threshold that at the 30m gap-crossing capability threshold. However, the rate of increase within the canopy ≥17.1m began to decrease after the 90m gap-crossing capability threshold while the canopy ≥7.2m and canopy ≥3m decreased at around 120m (Figure 4-17).

An upsurge in connectivity existed between the canopy ≥17.1m and the canopy ≥7.2m. Specifically, the canopy ≥7.2m was between 20 and 50 times more connected then the canopy ≥17.1m (canopy ≥17.1m IIC maximum = 0.00021, canopy ≥7.2m IIC maximum = 0.0057). Conversely, at the 30m gap-crossing threshold the canopy ≥3m was around 2 times more connected then the canopy ≥7.2m (canopy ≥7.2m IIC = 0.0021, canopy ≥3m IIC = 0.0043). This gap in connectivity began to decrease so that at the 200m gap-crossing threshold the canopy ≥3m was 1.1 times more connected than the canopy ≥7.2m (canopy ≥7.2m IIC = 0.005, canopy ≥3m IIC = 0.006) (see Appendix B).

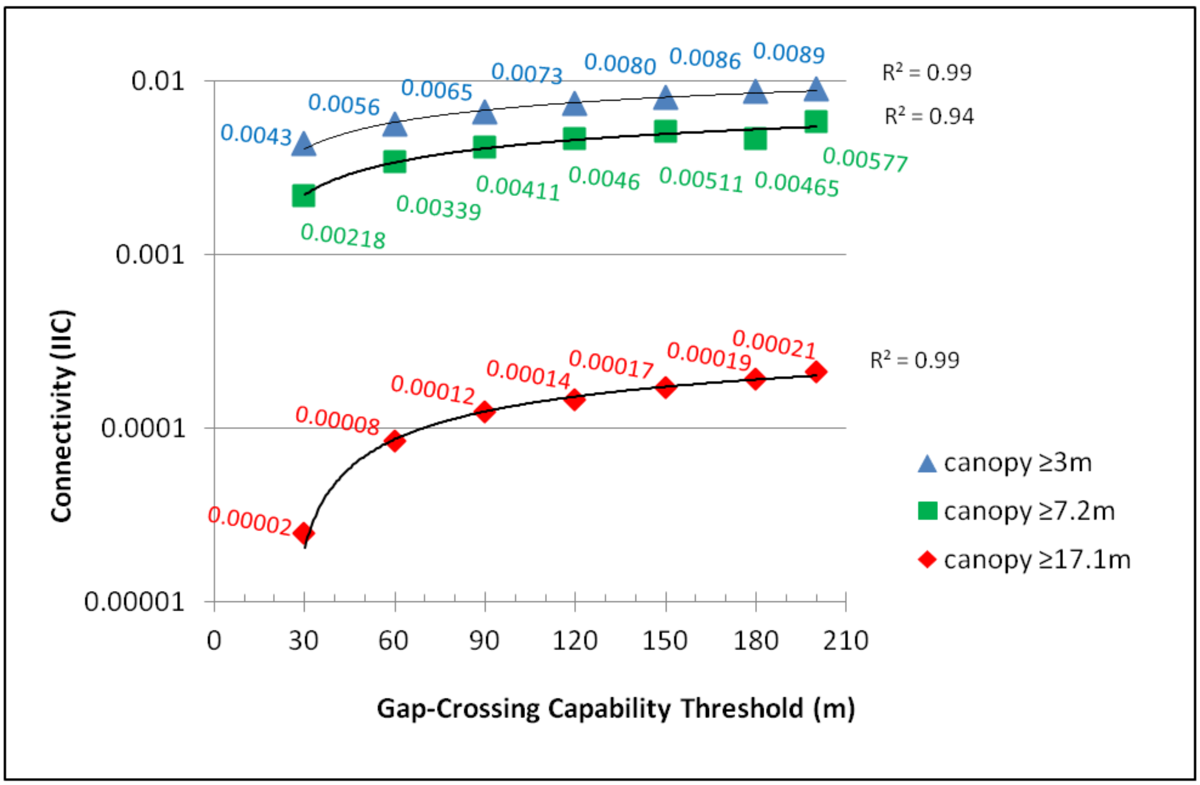

Figure 4-17: Levels of connectivity (IIC) exhibited by the 2009 natural break height (NBH) canopies.
The canopy ≥17.1m (trees above 17.1m) is the least connected canopy as IIC = 0.00002 at 30m, increasing by 741% to 0.00021 at 200m. The canopy ≥7.2m (trees above 7.2m) is the second least connected canopy as IIC = 0.002418 at 30m, increasing, slower than the canopy ≥17.1m, by 165% to 0.00577 at 200m. The most connected canopy is the canopy ≥3m (all trees above 3m) as IIC = 0.0043 increasing, slower than the other canopies, by 108% to 0.0089.

In 2009, the Higher Broughton survey area's UTI covered roughly 7% of the landscape (~12,000m^2). Connectivity values were lower than the river valley study area except for the river valley's canopy ≥17.1m which revealed lower connectivity than Higher Broughton's canopy ≥7.2m and canopy ≥3m. The connectivity scores for Higher Broughton's canopy ≥17.1m ranged from 0.000005 to 0.000008 and there was a negligible rate of increase in connectivity up until the 150m gap-crossing capability threshold when there was a 36% increase in connectivity. The canopy ≥7.2m IIC scores were between 0.00019 and 0.00085 and even though connectivity increased in line with gap-crossing capability threshold the rate of change reduced after the 90m mark. The canopy ≥7.2m was also between 38 (30m) and 107 (200m) times more connected than the canopy ≥17.1m. The canopy ≥3m was between two (200m) and three (30m) times more connected than the canopy ≥7.2m, as its IIC scores ranged between 0.00058 and 0.00165. Like the canopy ≥7.2m, the rate of increase in connectivity also reached a threshold, yet in the canopy ≥3m this was at around 90 to 120m gap-crossing capability threshold (Figure 4-18).

The UTI of lower Kersal covered around 3% of the landscape (5,800m^2), resulting in lower connectivity values than the river valley study area (expect Lower Kersal's canopy ≥3m and canopy ≥7.2m which were more connected than the river valley study area's canopy ≥17.1m). The IIC scores for lower Kersal's canopy ≥17.1m ranged from 0.0000008 – 0.0000012, and as a consequence there was only a minor rate of increase in connectivity at the 120m gap-crossing capability threshold (50% increase), after which rate of increase stopped. The canopy ≥7.2m (IIC = 0.00007 – 0.00012) in lower Kersal was between 88 (30m) and 100 (200m) times more connected than the canopy ≥17.1m. The increase in connectivity with gap-crossing capability threshold began to slow down after the 90m gap-crossing capability threshold. This threshold moved to between the 90 and 120m gap-crossing capability thresholds for the canopy ≥3m which was around 2.5 times more connected than the canopy ≥7.2m (IIC = 0.00018 – 0.00029) (Figure 4-18).

All canopies in the Peel Park sample plot were more connected than the river valley study areas UTI (except Peel Park's canopy ≥17.1m at 30 ad 60m which resulted in lower IIC values than the river valley study area's canopy ≥3m). The entire UTI of Peel Park in 2009 covered approximately 35% of the sample plot (~70,000m^2). The canopy ≥17.1m IIC results were between 0.0028 and 0.0077, increasing with gap-crossing capability threshold, and slowing down in the rate of connectivity change after 90m. Peel Park's canopy ≥7.2m was between 8 (200m) and 14 (30m) times more connected than the canopy ≥17.1m (IIC = 0.04 – 0.058) and had a rate of change threshold of 60m. The canopy ≥3m was around a further 1.3 times more connected than the canopy ≥7.2m (IIC = 0.053 – 0.067) and exhibited a low rate of connectivity change, slowing down to 2% change at 90m (Figure 4-18).

The final tree canopy sample plot was Kersal which, in 2009, had a UTI which covered around 39% of the landscape (~75,000m^2). The IIC results for Kersal were higher than all canopies in the river valley study area, with the only exceptions being Kersal's canopy ≥17.1m at 30 – 120m providing lower IIC results than the river valley study area's canopy ≥3m. Kersal's canopy ≥17.1m had IIC results ranging from 0.0015 – 0.0073, the rate of change in IIC began to reduce after the 90m gap-crossing capability threshold. This threshold reduced to 60m for the canopy ≥7.2m which was between 8 (200m) and 25 (30m) times more connected than the canopy ≥17.1m (IIC = 0.038 – 0.057). The canopy ≥3m in Kersal was only ever around 1.3 times more connected than the canopy ≥7.2m (IIC = 0.049 – 0.072) and the rate of connectivity increase reduced after the 90m gap-crossing capability threshold (Figure 4-18 and see Appendix C).

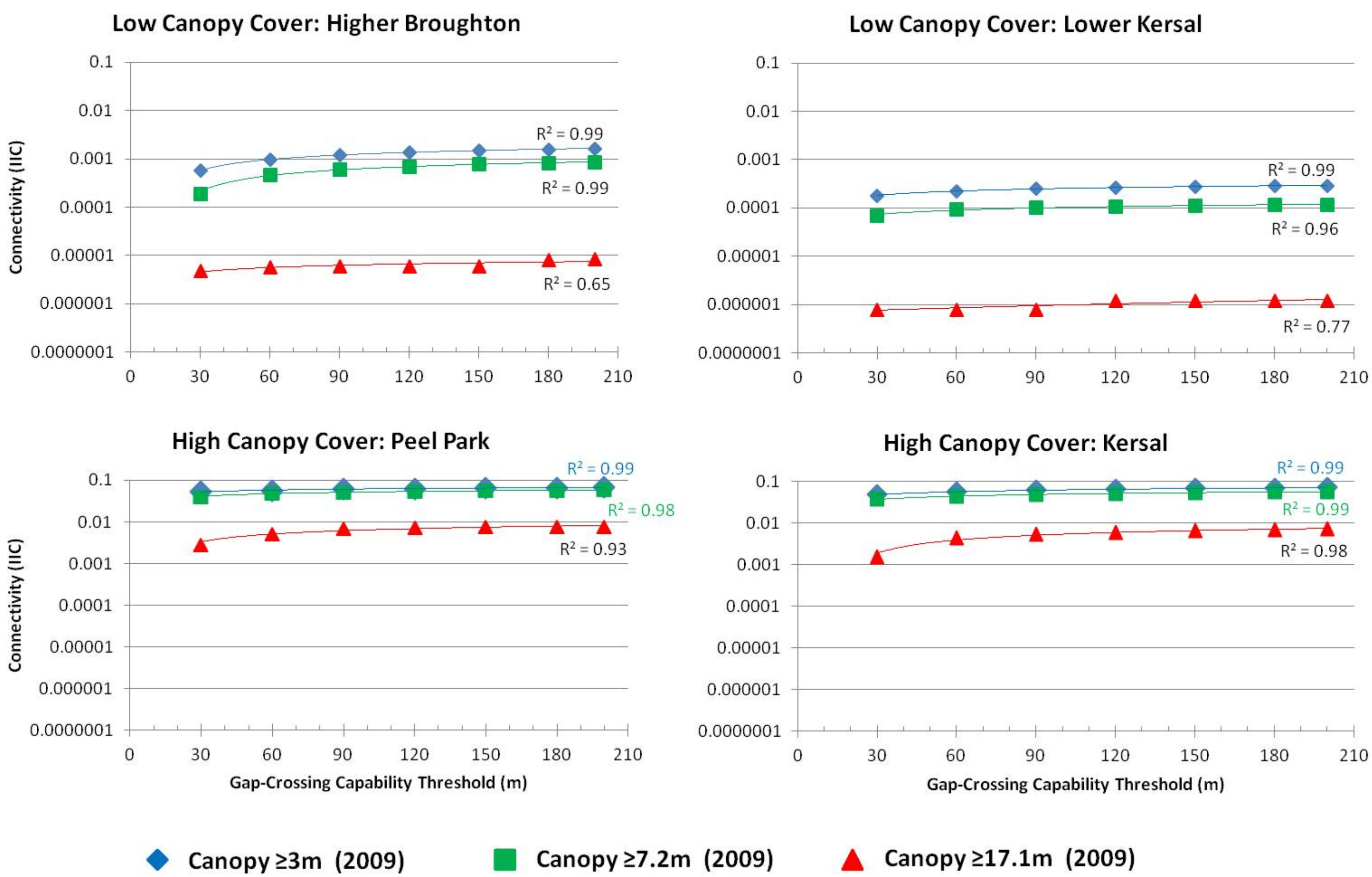

Figure 4-18: Connectivity results (IIC) for the tree canopy sample plots in 2009.

For all tree canopy sample plots the canopy ≥17.1m is the least connected canopy. The difference in connectivity is most prevalent between the canopy ≥17.1m and the ≥7.2m/≥3m canopies. The Peel Park and Kersal sample plots exhibit small differences in connectivity between the canopy ≥7.2m and canopy ≥3m. Connectivity increases in line with an increase in gap-crossing capability threshold. The rate at which this happens begins to decrease after a 90, 120, or 150m threshold is reached – the majority of cases being between 90 and 120m. The biggest increase in connectivity occurs as ability to cross gaps increases from 30 to 60m (minus the canopy ≥17.1m for the Lower Kersal sample plot and the canopy ≥3m for the Peel Park sample plot). The percentage change in connectivity between the 30 and 60m gap-crossing capability thresholds, depending on NBH canopy, ranges from 13-184%.

4.3.3 UTI connectivity 2013

Each of the four tree canopy sample plot's UTI were vertically stratified using the same natural breaks found in the 2009 canopy data and as such three NBH canopies, distributed throughout the study landscapes, were identified for Higher Broughton, Lower Kersal, Peel Park, and Kersal (Figure 4-19, Figure 4-20, Figure 4-21, and Figure 4-22). In Higher Broughton the canopy ≥3m covered around 6% of the landscape (~11,000m^2). The canopy ≥7.2m covered around 5% (~9,000m^2) of the sample plot, making up approximately 81% of the total canopy cover. Approximately 18% of the total canopy cover in Higher Broughton was found in the canopy ≥17.1m, covering only 1% of the landscape (~2,000m^2) (Figure 4-19).

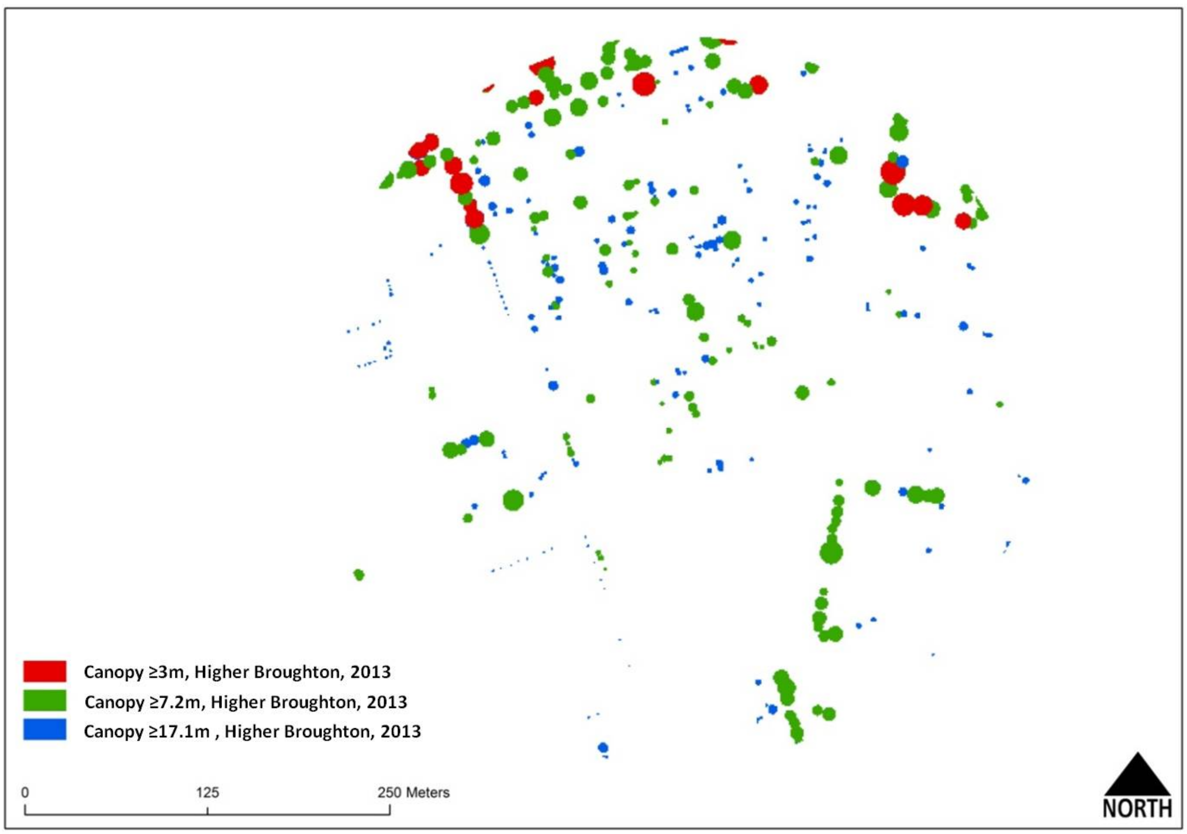

Figure 4-19: The 2013 NBH canopies for Higher Broughton.
The canopy ≥17.1m is clearly limited to the north of the sample plot. Reasons for this pattern are 1) there exists a public park to the north east edge of the sample plot and 2) the northern border of the sample plot encroaches onto a more affluent, less deprived area of Salford. There is a distinct lack of trees to the south of the study due to the presence of playing/sports fields and recent development of housing estates.

In Lower Kersal the canopy ≥3m only covered roughly 3% (~6,000m^2) of the study landscape. Around 43% of the total canopy cover was comprised of the canopy ≥7.2m, which only covered approximately 1% (~3,000m^2) of the landscape. The least amount of canopy cover for all tree canopy sample plots was found in Lower Kersal's canopy ≥17.1m which covered only 0.4% (~800m^2) of the landscape (approximately comprising 13% of total canopy cover) (Figure 4-20).

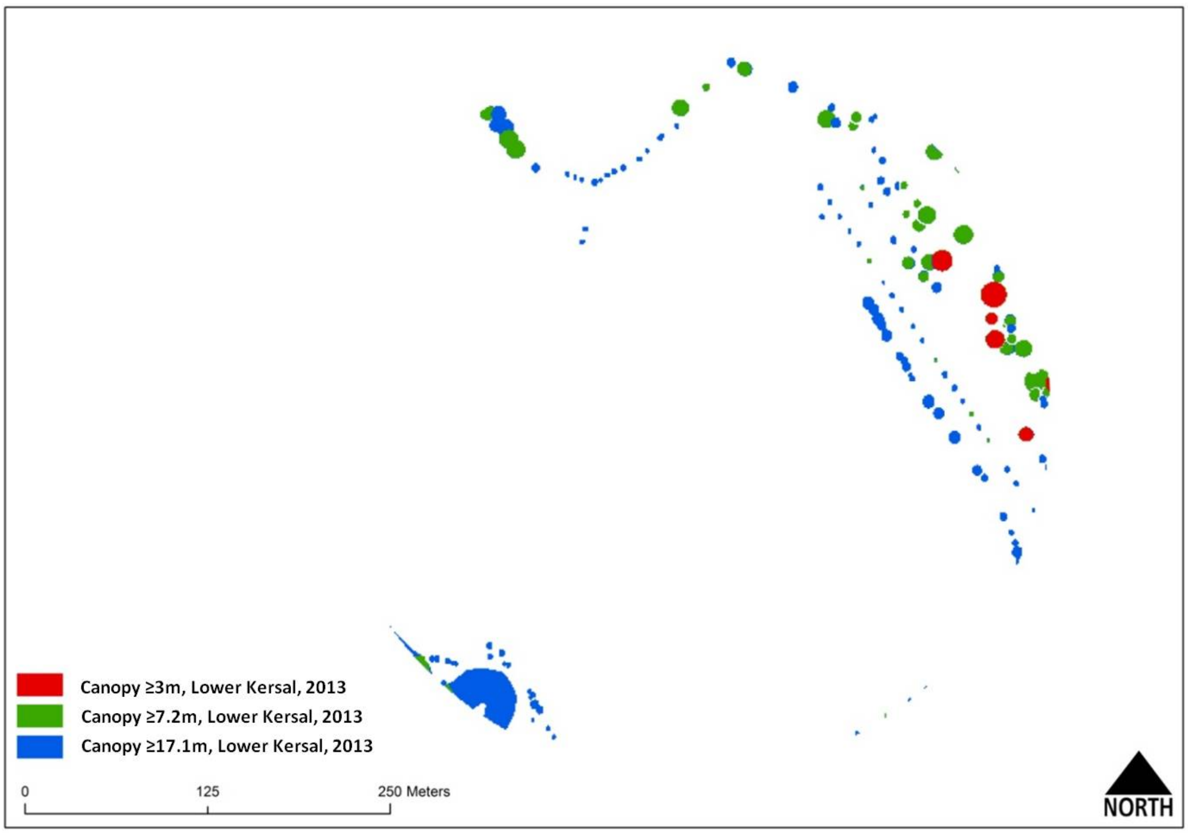

Figure 4-20: The 2013 NBH Canopies for Lower Kersal.
The great absence of trees within the central and western parts of the Lower Kersal sample plot is due to the presence of sport fields and flood plains. The majority of trees are below 7.2m, with the canopy ≥17.1m limited to the east of the sample plot.

The greatest amount of canopy cover was found in Peel Park as its canopy ≥3m covered around 34% (~67,000m^2) of the landscape. Approximately 97% of this tree cover was found in the canopy ≥7.2m as it covered around 33% (~65,000m) of the Peel Park landscape. Peel Park's canopy ≥17.1m covered a greater area than Higher Broughton and Lower Kersal's entire UTI put together (~39,000m^2). Furthermore, over half (58%) of the tree cover was comprised of the canopy ≥17.1m, covering around 20% of the landscape (Figure 4-21).

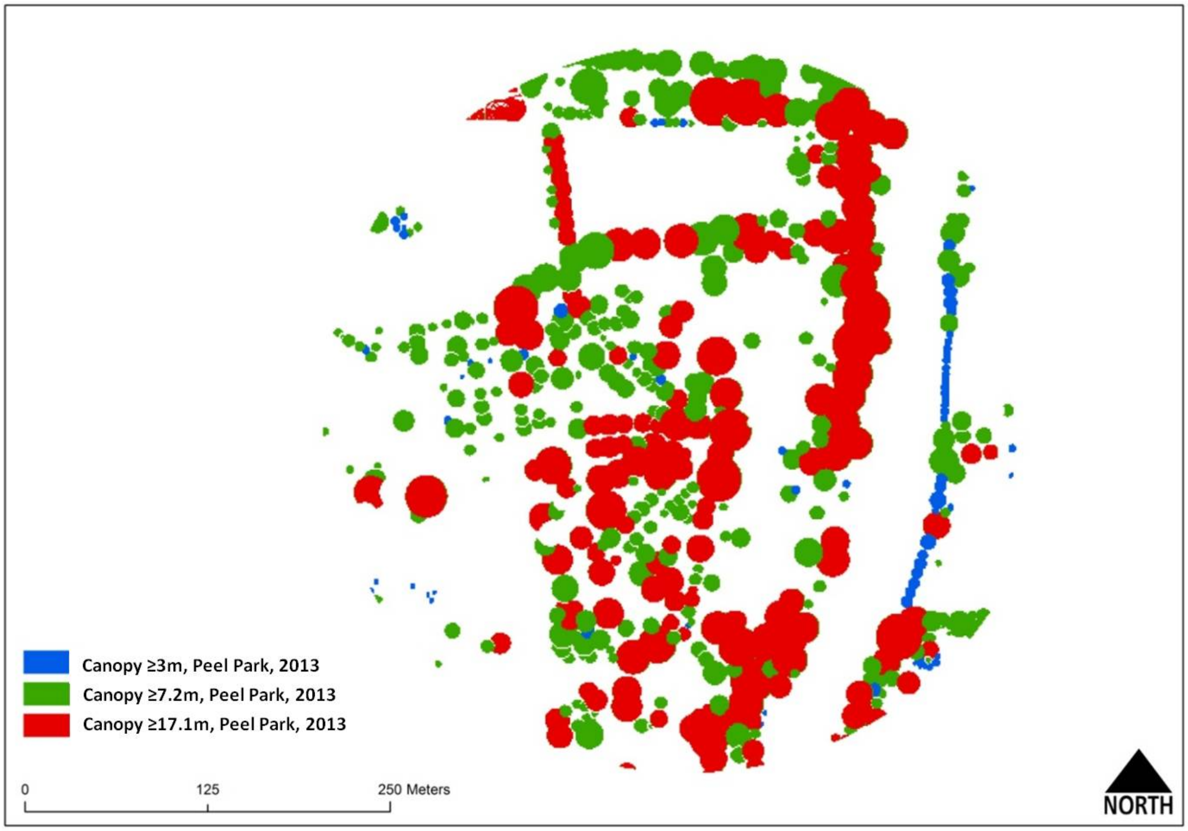

Figure 4-21: The 2013 NBH Canopies for Peel Park.
The Peel Park sample plot contained the highest amount of canopy ≥17.1m cover (58% of the tree cover was above 17.1m). Large gaps in the canopy occur due to the River Irwell to the east, playing fields to the north and north west, and the buildings/sealed surfaces of the University of Salford to the south west.

Finally, the canopy ≥3m in Kersal covered roughly 26% (~52,000m^2) of the land surface. Kersal's canopy ≥7.2m represented around 94% of total canopy cover as it covered around 25% of the landscape (~49,000m^2). The canopy ≥17.1m covered approximately 10% (19,000m^2) of Kersal, which was around 37% of total canopy cover (Figure 4-22).

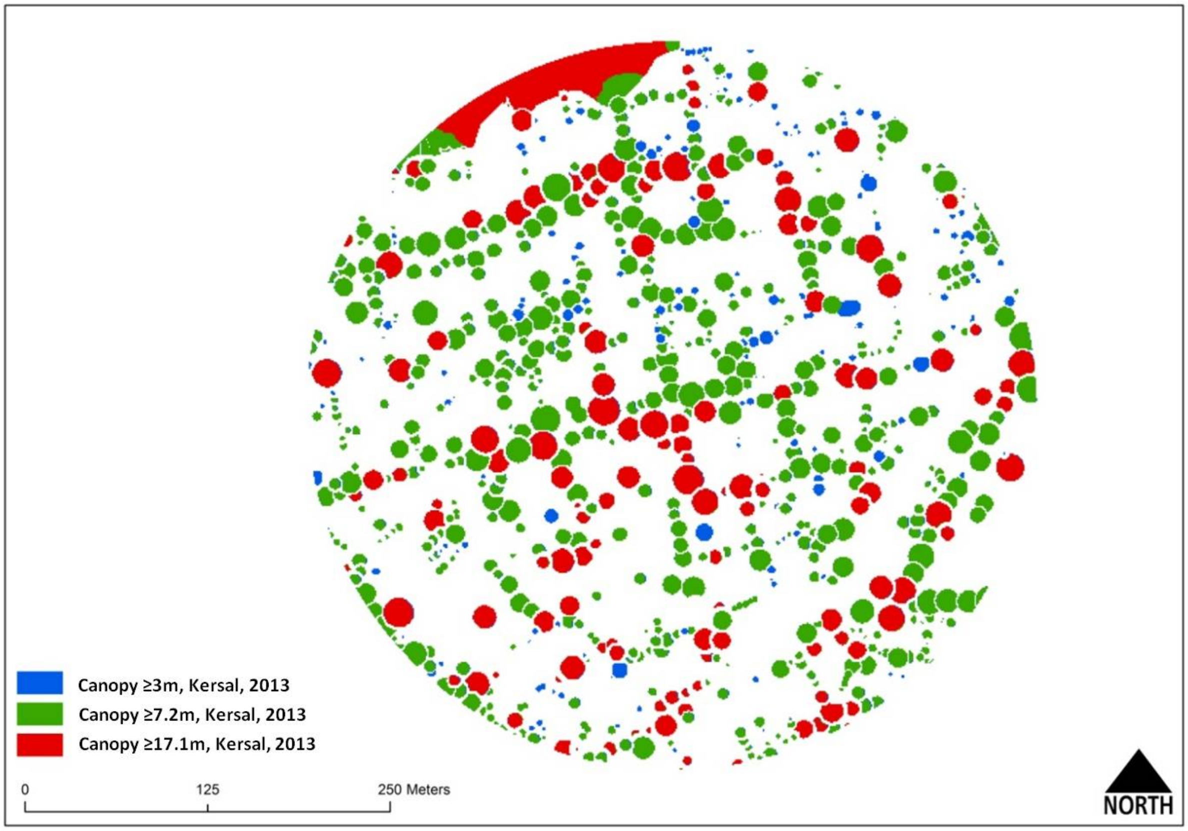

Figure 4-22: The 2013 NBH canopies for Kersal.
Kersal is the least deprived of the tree canopy sample plots and represents a suburban residential area. The gaps in the canopy are mainly caused by roads and houses. To the north of the sample plot is a dense woodland patch which forms part of the border to Prestwich. The canopy ≥3m and canopy ≥17.1m are equally distributed across the sample plot.

The Higher Broughton sample plot's canopy ≥17.1m revealed IIC values between 0.00002 and 0.00005 and the rate of connectivity increase began to decelerate after the 120m gap-crossing capability threshold. The canopy ≥7.2m of Higher Broughton was between 11 (30m) and 17 (200m) times more connected than the canopy ≥17.1m (IIC = 0.00022 – 0.00085) and the rate of change in connectivity began to slow down after 90m. A similar threshold was also observed for the canopy ≥3m (90-120m) which was between 1.5 (200m) and 2 (30m) times more connected than the canopy ≥7.2m (IIC = 0.0004 – 0.0013) (Figure 4-23).

The Lower Kersal sample plot's IIC scores for the canopy ≥17.1m were only ever between 0.000006 and 0.000009 and the rate of change in connectivity stopped after the 120m gap-crossing capability threshold, after which distance no longer affected connectivity. The canopy ≥7.2m only had a slight rate of connectivity increase, slowing down at 150m, and was between seven (30m) and nine (200m) times more connected than the canopy ≥17.1m (IIC = 0.00004 – 0.00008). The canopy ≥3m was then a further three (200m) to four (30m) times more connected than the canopy ≥7.2m (IIC = 0.00016 – 0.00027). The canopy ≥3m was the only canopy to reveal a rate of connectivity increase threshold, which was at the 90m gap-crossing capability threshold (Figure 4-23).

In 2013, Peel Park was the most connected tree canopy sample plot. The canopy ≥17.1m's IIC scores were between 0.01 and 0.02 and the rate of increase in connectivity slowed after the 90m gap-crossing capability threshold. The canopy ≥7.2m was around three (200m) and four (30m) times more connected than the canopy ≥17.1m (IIC = 0.039 – 0.057) with the same rate of change threshold of 90m. This threshold was also identified for the canopy ≥3m canopy which was only ever around 1.1 times more connected than the canopy ≥7.2m canopy (IIC = 0.04 – 0.06) (Figure 4-23).

Kersal was also well connected in 2013. The IIC scores for Kersal's canopy ≥17.1m were between 0.0007 and 0.004 and the rate of increase in connectivity began to reduce after the 90m gap-crossing capability threshold. The canopy ≥7.2m IIC scores ranged between 0.01 and 0.026, which meant it was six (200m) and 14 (30m) times more connected than the canopy ≥17.1m. The rate of change threshold was also at the 90m gap-crossing capability threshold, the same as the canopy ≥3m canopy which was only ever around 1.1 times more connected than the canopy ≥7.2m canopy (0.01 – 0.03) (Figure 4-23).

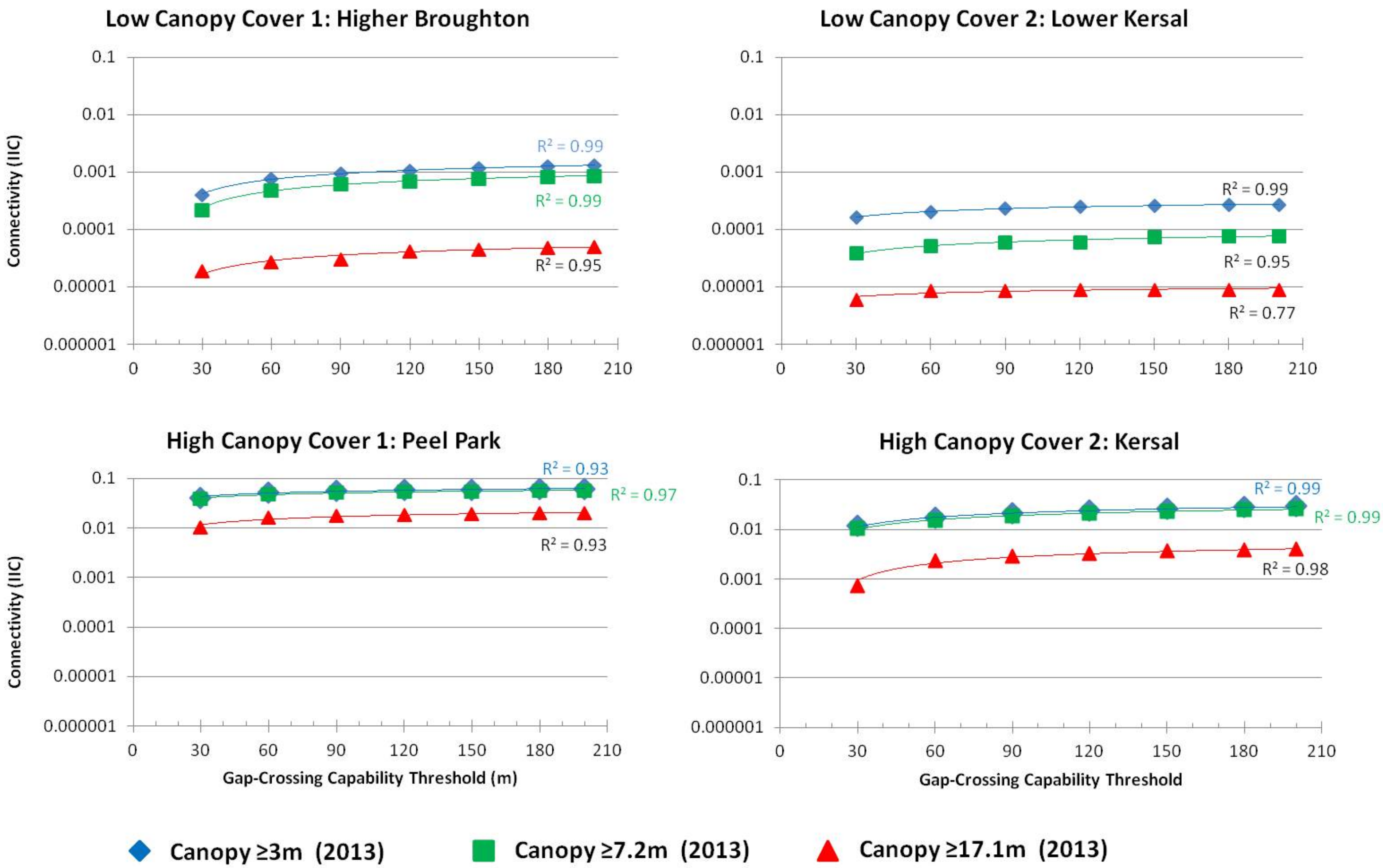

Figure 4-23: Connectivity results (IIC) for the tree canopy sample plots in 2013.

The canopy ≥17.1m is the least connected for all tree canopy sample plots. The difference in connectivity is most prevalent between the canopy ≥17.1m and the middle/canopy ≥3m canopies. The Peel Park and Kersal sample plots also exhibit negligible differences in connectivity between the middle and canopy ≥3m canopies. Connectivity increases in line with an increase in gap-crossing capability threshold. The rate at which this happens begins to decrease after a 90, 120, or 150m threshold is reached – the majority of cases being between 90 and 120m. The biggest increase in connectivity for all NBH canopies occurs as gap-crossing ability increases from 30 to 60m. The percentage change in connectivity between a 30 and 60m gap-crossing capability threshold, depending on NBH canopy, ranges from 23-226%.

4.3.5 Temporal change in connectivity

The connectivity results of the NBH canopies for the four tree canopy sample plots were compared over three points in time (See Appendix C) for raw results). The temporal change in connectivity for the canopy ≥3m canopy can be viewed in Figure 4-24, the canopy ≥7.2m canopy in Figure 4-25, and the canopy ≥17.1m in Figure 4-26.

For the canopy ≥3m, connectivity reached its peak in 2009 in all tree canopy sample plots (Figure 4-24). In the Lower Kersal and Peel Park sample plots, the canopy ≥3m was least connected in 2005. Therefore, connectivity increased from 2005 to 2009 and then decreased again in 2013 but not so much as to be lower in connectivity than 2005. Specifically, in Lower Kersal, the canopy ≥3m connectivity increased by between 286% (200m gap-crossing capability threshold) and 323% (30m gap-crossing capability threshold) in 2009 and then decreased by between 6% (2000m) and 9% (30m) in 2013. The canopy ≥3m connectivity results for Peel Park followed the same pattern, with a less severe increase in connectivity in 2009 (between 70% at 200m and 168% at 30m) but a greater decrease in 2013 (between 8% at 200m and 22% at 30m). The canopy ≥3m canopy in the Higher Broughton and Kersal sample plots were both least connected in 2013 – meaning there was an increase in connectivity from 2005 to 2009 and then an even greater decrease in 2013. In regards to Higher Broughton, canopy ≥3m canopy connectivity increased from 2005 to 2009 by between 7% (200m) and 20% (30m) and then decreased from 2009 to 2013 by between 19% (220m) and 33% (30m). Kersal's canopy ≥3m canopy increased in connectivity from 2005 to 2009 by between 40% (200m) and 62% (30m) and then greatly decreased from 2009 to 2013 by between 60% (200m) and 80% (30m) (Figure 4-24).

The canopy ≥7.2m canopy was most connected in 2009 for all tree canopy sample plots except for Higher Broughton which was at its most connected in 2013 (Figure 4-25). However, there wasn't much difference between the 2009 and 2013 canopy ≥7.2m canopy IIC values in Higher Broughton and an inverse relationship was also evident in Peel Park where the 2009 canopy ≥7.2m IIC scores were only very slightly more than the 2013 results. In the Higher Broughton, Lower Kersal, and Peel Park sample plots, the canopy ≥7.2m was also least connected in 2005. Therefore, in Higher Broughton connectivity increased from 2005 to 2009 by between 30% (200m) and 40% (30m) and then increased again from 2009 to 2013 by between 0.2% (200m) and 15% (30m). In Lower Kersal, canopy ≥7.2m connectivity increased from 2005 to 2009 by between 131% (30m) and 142% (200m) and then decreased from 2009 to 2013 by between 34% (200m) and 44% (30m). In Peel Park, canopy ≥7.2m connectivity increased by between 62% (200m) and 132% (30m) in 2009 and then decreased by between 1% (200m) and 3% (30m) in 2013. Kersal was the only tree canopy sample plot where the canopy ≥7.2m was least connected in 2013. Therefore, connectivity increased from 2005 to 2009 by between 30% (200m) and 53% (30m) before following a decrease in connectivity from 2009 to 2013 of between 55% (200m) and 73% (30m)(Figure 4-25).

Unlike the other two NBH canopies, the connectivity of the canopy ≥17.1m was at its highest in 2013 for all tree canopy sample plots except Kersal, which was at its highest in 2005 (Figure 4-26). In the higher Broughton, Lower Kersal, and Peel Park sample plots the connectivity of the canopy ≥17.1m was at its lowest in 2005. An inverse relationship was observed in Kersal, as the lowest IIC scores for the canopy ≥17.1m were calculated for 2013. More specifically, in the Higher Broughton sample plot canopy ≥17.1m connectivity increased by between 67% (30m) and 100% (200m) in 2009 and then increased again by between 280% (30m) and 513% (200m) in 2013. In the Lower Kersal sample plot there was no canopy ≥17.1m in 2005, therefore connectivity only increased from 2009 to 2013 by 650% (however IIC values were still extremely low). In the Peel Park sample plot canopy ≥17.1m connectivity increased from 2005 to 2009 by between 3% (30m) and 9% (200m) and then increased again in 2013 by between 162% (200m) and 264% (30m). Finally, the canopy ≥17.1m of the Kersal sample plot reached its connectivity peak in 2005 and its trough in 2013. Explicitly, the Kersal sample plots' canopy ≥17.1m IIC values decreased by between 30% (200m) and 56% (30m) in 2009 and then decreased again by between 46% (200m) and 53% (30m) in 2013 (the only canopy and survey area to follow such a pattern) (Figure 4-26).

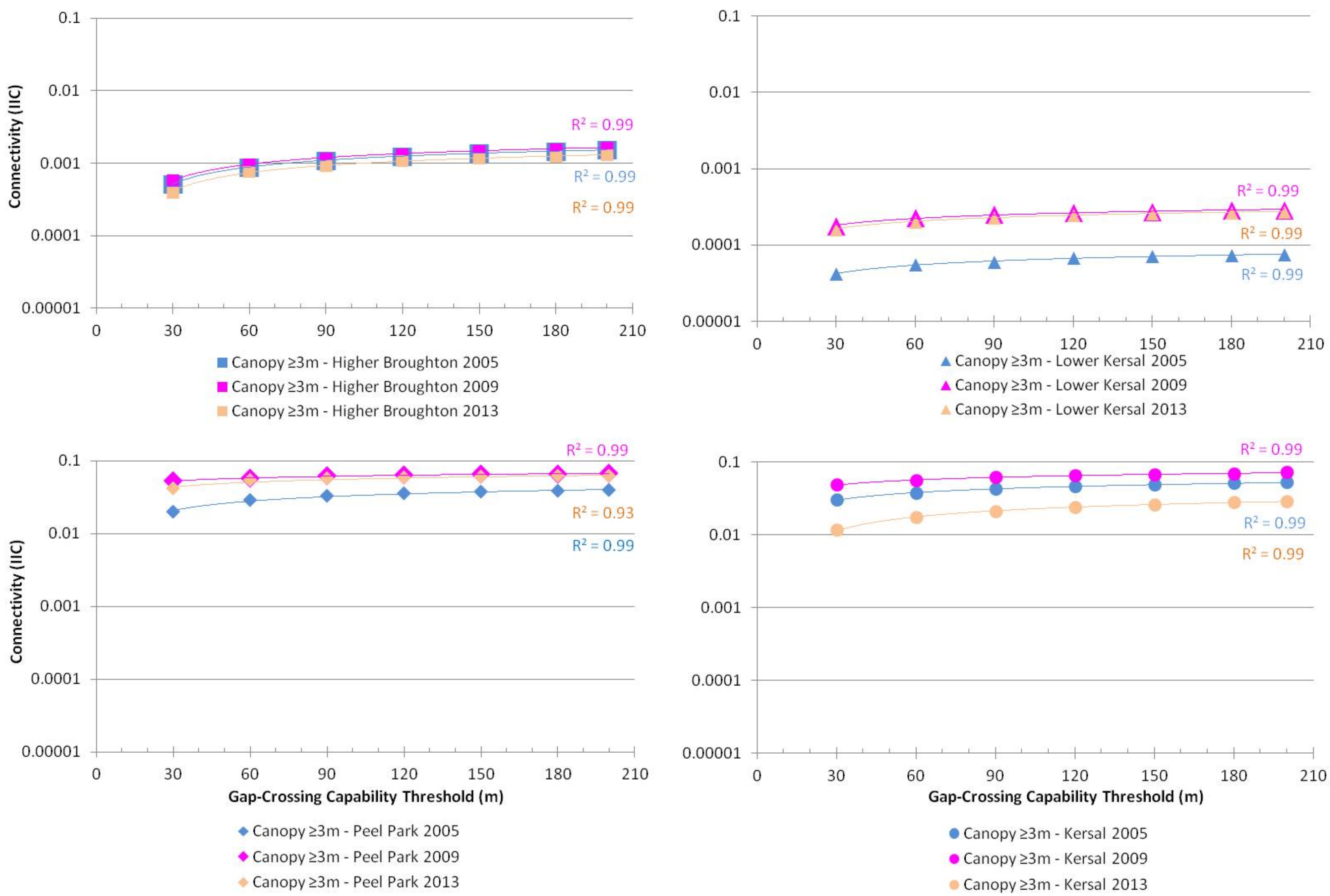

Figure 4-24: Temporal change in the connectivity (IIC) of the tree canopy sample plot's canopy ≥3m.

The canopy ≥3m for all tree canopy sample plots were most connected in 2009. The canopy ≥3m of the Kersal and Higher Broughton sample plots were least connected in 2013. Conversely, the Peel Park and Lower Kersal sample plots exhibited their least connected canopy ≥3m in 2005.

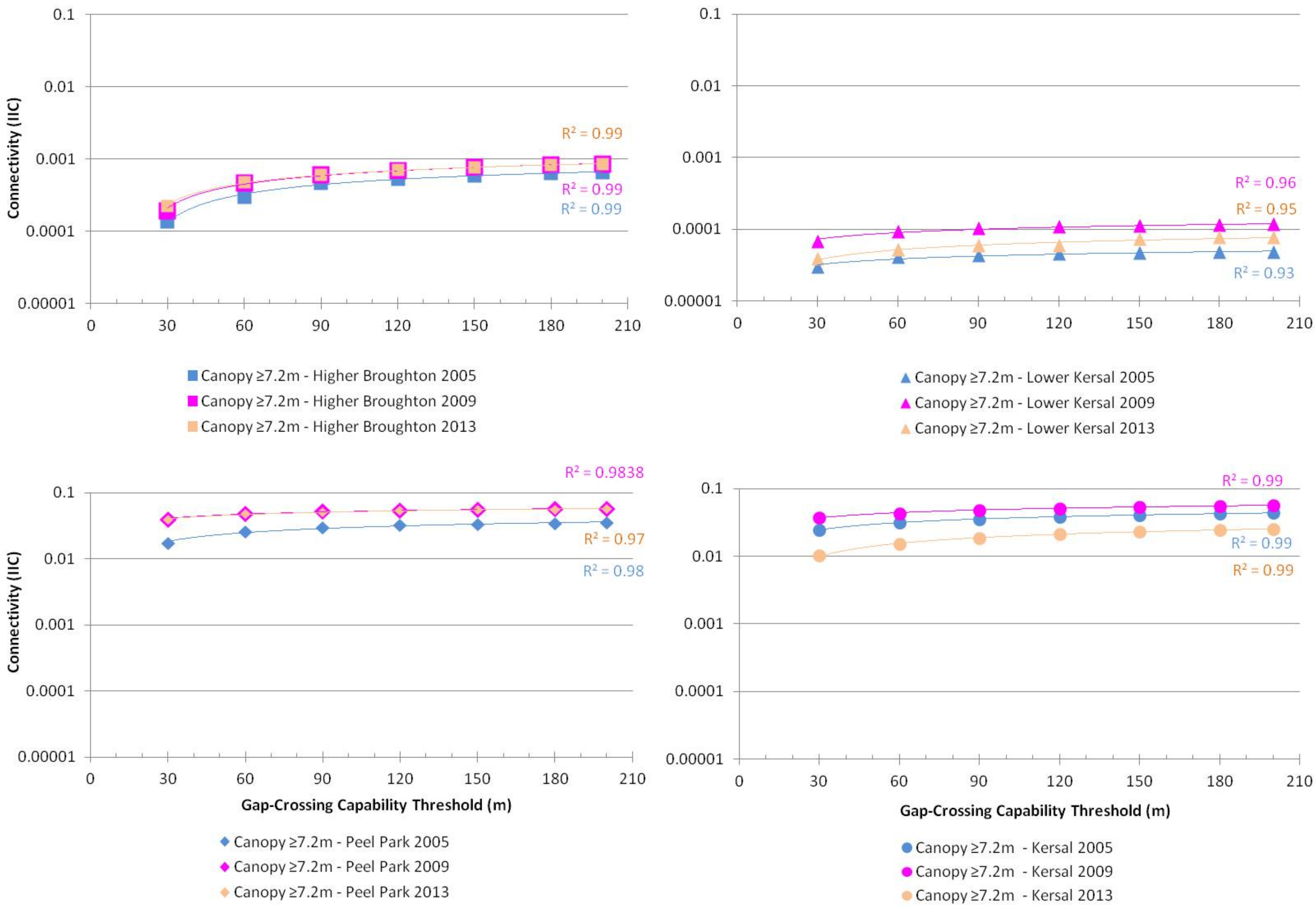

Figure 4-25: Temporal change in the connectivity (IIC) of the tree canopy sample plot's canopy ≥7.2m.

The canopy ≥7.2m was most connected in 2013 for the Higher Broughton sample plot (by a few percent). The Lower Kersal, Peel Park, and Kersal sample plots exhibited their most connected canopy ≥7.2m in 2009. The canopy ≥7.2m was least connected in 2005 within the Higher Broughton, Lower Kersal, and Peel Park sample plots. In 2013 the Kersal sample plot exhibited its least connected canopy ≥7.2m.

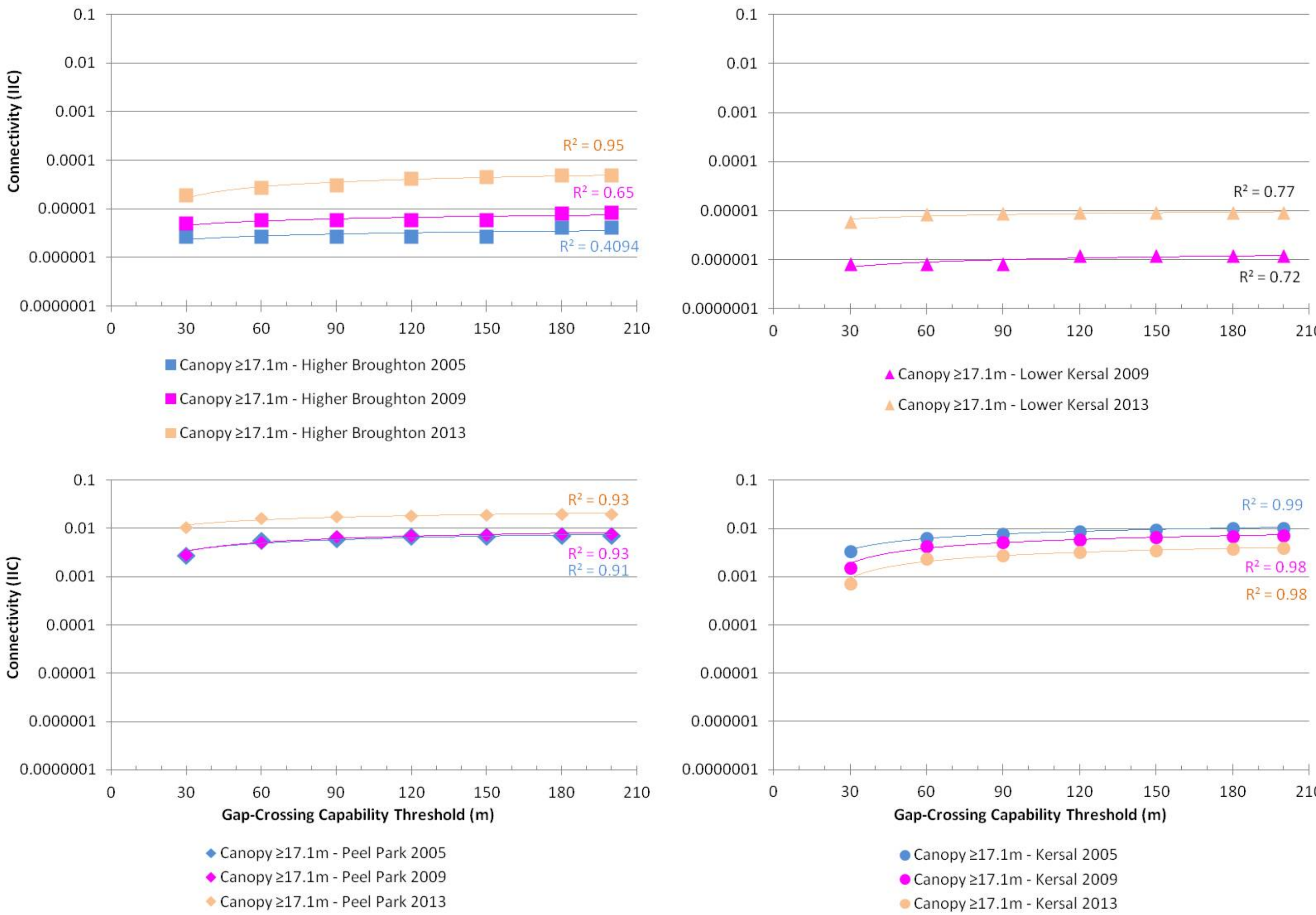

Figure 4-26: Temporal change in the connectivity (IIC) of the tree canopy sample plot's canopy ≥17.1m.
The canopy ≥17.1m was connected most in 2013 for all the tree canopy sample plots except the Kersal sample plot. For the higher Broughton, Lower Kersal, and Peel Park sample plots the canopy ≥17.1m was least connected in 2005 (non-existent within the Lower Kersal sample plot in 2005). For the Kersal sample plot the canopy ≥17.1m followed a unique pattern of continual reduction in connectivity from 2005 to 2013.

4.4 Discussion

To quantify and evaluate the levels of potential landscape connectivity exhibited by the river valley sample plot's UTI, its vertically stratified canopies have been transformed into a set of landscape graphs. The UTI canopy patches, defined by its structural neighbourhood, are represented as nodes while the functional links between these nodes have been defined using passerine gap-crossing capability thresholds (a proxy for perceptual threshold). The graph and habitat availability based, binary metric - the integral index of connectivity (IIC) - has been utilised in order to gain a representative value of vertical canopy landscape connectivity. The consequential findings of this approach are varied, novel, and interesting. An overview of the results, focusing on the methodological impacts towards understanding, is presented below. This is followed by a discussion on the pertinent patterns identified in section 4.3 (section 4.4.2, 4.4.3).

4.4.1 Overview

There are two major characteristics of the integral index of connectivity (IIC) which need to be acknowledged in order to understand the results in chapter 4.3. The first is that IIC measures both intra- and inter-patch connectivity simultaneously. The second attribute of IIC, like all landscape pattern indices, is that its absolute values are not as empirically useful as its relative values. In regards to the first attribute of IIC low scores in the river valley sample plot can be attributed to the large amount of small canopy patches and as a result low intra-patch connectivity. Locally connected regions do exist within the study landscape, for example the Peel Park and Kersal sample plots, yet the large amount of small, scattered patches that are present throughout the landscape reduces the overall landscape connectivity score. As stated by Neel (2008), the 'dependence on the ratio between area of the focal habitat and total landscape extent makes [IIC] potentially problematic for situations in which patches are very small relative to the total landscape'(p. 951) . However, although the inclusion of intra-patch connectivity within the calculation of IIC can cause low values in highly scattered landscapes, it does not necessarily mean it is 'problematic'. Rather it must be kept in mind that IIC attempts to calculate *functional* connectivity not just *structural* connectivity.

Neel (2008) held intra-patch connectivity as negatively responsible for the low IIC values of Cushenbury/silvery—white milkvetch (*Astragalus albens*) patches within southern California. From this position, the author added a caveat stating that, in fact, the majority of *A. albens* patches had a high proportion of neighbouring patches within a critical distance, thus meaning higher levels of connectivity actually existed, if only locally. By adding this stipulation, the highly descriptive powers of IIC (in regards to landscape connectivity) were overlooked for the simpler descriptive powers of neighbourhood – in other words the implications of functional connectivity were overshadowed by structural connectivity. Although low IIC scores can be attributed to the presence of smaller sized patches, this does not mean there is an explanatory flaw within the metric but rather an explanatory strength – it highlights the lack of functional habitat. For instance, If a given species cannot effectively use the patches within a landscape, i.e. patches are no more than just stepping stones which can be used to move through or out of the landscape or only provide supplementary resources, then the landscape isn't *functionally* connected. Such a landscape, like the river valley study area, contains small habitat patches with structurally connected neighbours but it does not mean they are areas within which functional connectivity can occur. Therefore, the low IIC results of the river valley study area suggest that the most effective way to increase connectivity would be to increase the number of large area canopy patches. In urban areas such an approach is evidently difficult – practically impractical – as free land is highly contested and preference is afforded to other forms of infrastructure plus built surfaces. Another approach would be to identify where closing gaps in the canopy would provide the greatest benefit to overall landscape connectivity. Before such an approach is discussed, the second main attribute (absolute values are not as useful as relative values) of IIC should be addressed.

Measuring the relative importance of canopy patches for maintaining landscape connectivity would be more insightful than a single IIC score. Furthermore, if large area canopy patches are identified as important for maintaining connectivity this would also support the claim that an increase in large patches, within the river valley study area, is necessary (analysing patch importance is undertaken in Chapter 6). In addition, the study into the temporal and vertical changes of landscape connectivity provides the opportunity to assess relative IIC values.

By incorporating the temporal scale, levels of connectivity exhibited by the river valley study area and tree canopy sample plot's UTI's can be compared. However, the results in section 4.3 reveal no pertinent pattern - in regards to connectivity change. As a result, UTI management processes cannot be influenced by research insights such as 'connectivity is decreasing/increasing at a given rate and therefore this project plan should be followed'; rather connectivity is in a state of flux. Certainly, and in general, the river valley study area's UTI increases in connectivity from 2005 to 2009, yet the results from the tree canopy sample plots reveal more complex and sometimes opposing results.

If a panarchy approach is taken towards the results in section 4.3 then the IIC results of the smaller tree canopy sample plots are set and restrained by the processes acting in the larger river valley study area while at the same time the river valley study area is a construct of the smaller tree canopy sample plots thus its own IIC results are influenced by the processes acting in the smaller survey areas. Changes to, development in, and losses of cover can greatly affect the ecological function - when related to connectivity - of the UTI over time. However, the results reveal that the UTI is always in flux; it is not set to an equilibrium which can be identified and aimed for. For example, the canopy ≥17.1m canopy decreases across 2005 to 2013 in the Kersal sample plot yet increases from 2005 to 2009 and then decreases to a level lower than that of 2005 in the Peel Park sample plot. Therefore, temporal changes in connectivity cannot provide useful management information. Instead, connectivity results should be incorporated into understanding how resilient to change the UTI is (see Chapter 5).

Despite the lack of pertinent temporal patterns there were other evident patterns emerging from the analysis in Chapter 4 – general rules that can be applied to the effective management, maintenance, and creation of UTIs. The benefit of using a temporal connectivity analysis is that any patterns, or apparent 'rules', which occur across sample plots in different points in time are not a product of that time but rather an inherent property of UTI landscape patterns and passerine perception (i.e. through gap-crossing capabilities). These research outputs embody new theories and highlight potential avenues of future research.

4.4.2 Connectivity increase threshold

Landscape connectivity (IIC) increased by means of gap-crossing capability threshold; however the rate of this increase began to decelerate after a specific distance. A total of 41 canopies were subjected to the connectivity model – 14 study areas with three canopies each, minus the Lower Kersal sample plots absent canopy ≥17.1m in 2005. A total of 40 canopy connectivity models exhibited a rate of connectivity increase distance threshold. Out of the 40 canopy connectivity models which exhibited such a threshold (95% of all canopy connectivity models), 60% were set at 90m and 17.5% set at 120m and therefore, 77.5% of all canopy connectivity results revealed a connectivity increase threshold of 90-120m. When including those canopies which exhibited an actual threshold of between 90-120m (a total of 4 canopies) this value increases to 85% (10% set at 60m, 5% set at 150m), consequently, a rate in connectivity increase threshold emerges. Hence, it can be stated that the influence of a passerine's ability to cross gaps on potential landscape connectivity begins to decrease after 90 -120m.

Such a threshold has not been identified before. Zollner and Lima (2005) did state that 'as perceptual ranges approach a large fraction of the width of the landscape an increase in perceptual range does little to increase dispersal success' (p 226). However, the authors did not specifically identify a connectivity increase threshold; nonetheless, this comment therefore raises a point of concern in regards to the credence of the 90-120m connectivity threshold. In particular, the tree canopy sample plots are 500m in diameter and therefore the landscape would become close to maximum connectivity levels nearer the 200m gap-crossing capability threshold, thus the rate in connectivity would naturally reduce as distance increased. However, the same 90-120m threshold was identified within the larger river valley survey area, which has a maximum width of around 4,000m. The 90-120m threshold, identified over time, in differing canopies, and in different study areas, therefore seems to be an inherent property of the connectivity of Salford's UTI when described using passerine gap-crossing capability thresholds.

This finding has a number of implications for future practice. Due to IIC measuring both intra- and inter- patch connectivity, this threshold suggests that after 90-120m intra-patch connectivity is more important than inter-patch connectivity. That is to say, area of habitat patch is more influential to connectivity after 90-120m gap-crossing capability threshold because although a passerine may cross such a gap there is still insufficient access to functionally sized habitat. Therefore, if overall canopy area cannot be increased within a given area, UTI management practice should target gaps in the canopy below 90-120m for overall connectivity to increase. Furthermore, the implication of this threshold also affects future studies in the way of selecting focal species. For example, to undertake the research within this thesis no focal species or genus were selected as this was considered a too narrow approach and results could potentially be biased towards that focal species/genus and therefore of little use if methods/results were to be transferable. However, due to the more general approach the 90-120m threshold suggests that future connectivity studies – which aim to improve the connectivity of an area for a certain species – should focus less on passerine habitat generalist, which potentially will move across bigger canopy gaps, and instead consider passerine habitat specialists who may resist crossing large canopy gaps (see section 2.5 and 4.2.1). To clarify, after a gap-crossing capability threshold of 90-120m is reached, UTI connectivity does not increase as much as below 90-120m. It is therefore logical to assume that studies which achieve improving landscape connectivity for passerine species, which only have a perception of, or capability to cross gaps less than 90-120m, will increase overall connectivity greater than focusing on more generalist species which are capable of crossing 200m gaps. Therefore, improving connectivity for specialist species may improve overall connectivity.

If this theoretical approach was to be adopted by future studies than it would be remiss not to mention that connectivity increased the greatest when the gap-crossing capability threshold increased from 30 to 60m. This increase was as big as 240% (Figure 4-17 and Appendix B). Therefore, focusing on closing 60m gaps would increase overall connectivity and studies that improved connectivity for even more forest specialist species (which will only cross small 30m gaps) would have a greater impact on overall landscape connectivity then trying to close 200m gaps. Of course, it must be made explicit that the greatest way to increase connectivity is to increase habitat area; however urban landscapes can make this difficult. The research in Chapter 4 therefore highlights an alternative approach, providing the hypothesis that the most beneficial increase in connectivity is ascertained by targeting and closing canopy gaps below 90-120m and specifically those gaps between 30 and 60m. This hypothesis could then guide future research.

4.4.3 Connectivity of the vertically stratified UTI canopies

The vertical connectivity analysis reveals that there is a need to reduce the connectivity inequality between the vertically stratified UTI canopies. In the main, the canopy ≥3m and canopy ≥7.2m have relatively similar associative connectivity values. The canopy ≥3m is only around 1.1-16 times more connected than the canopy ≥7.2m canopy, with 16 being the exception to the rule and a 2 time increase being the norm. As such the influential effect of including trees between 3m and 7.2m in the potential landscape connectivity model is minor. In contrast, trees 17.1m and above were severely disconnected and unrepresented across a large amount of the river valley study area and as such the difference in connectivity between the canopy ≥3m and ≥7.2m and the canopy ≥17.1m is great. For the entire river valley study area the canopy >3m is between 27 and 173 times more connected than the canopy ≥17.1m while the canopy ≥7.2m is between 19 and 88 times more connected. The tree canopy sample plots reveal even greater variation, depending on the original abundance of canopy cover. In particular, the canopy ≥17.1m is between 3 and 511 times less connected than the canopy ≥3m and between 3 and 219 times less connected than the canopy ≥7.2m. The lower difference values are found in the higher canopy cover sample plots Peel Park and Kersal, where the vertical distribution of the canopies are more equal than both the river valley study area and the lower canopy cover areas (i.e. Higher Broughton and Lower Kersal). Furthermore, in the Peel Park and Kersal sample plots the canopy ≥17.1m is better represented, reducing these connectivity differences are therefore a potential UTI management criteria.

Although the canopy ≥17.1m was the least connected of all the canopies it was often most influenced by an increase in gap-crossing capability threshold (minus the low density tree canopy sample plots). The greatest increase in connectivity from 30m to 200m was calculated for the canopy ≥17.1m of the river valley study area in 2009 at 741%. Therefore, the canopy ≥17.1m would benefit the most from decreasing canopy gaps (especially between 30 and 60m and up to 90-120m). In regards to passerine perception a greater perception of gaps in taller trees may improve connectivity. The results also suggest that, as the canopy ≥17.1m displays low total canopy area, the greatest possible increase in connectivity can only be achieved by establishing links between existing canopies.

By vertically stratifying the UTI another significant finding emerged – there is a great need to Increase new tree stock. The canopy ≥17.1m canopy became more connected in the sample areas in 2013 (minus Kersal) while the other canopies suffered. In all sample areas except Lower Kersal, no new trees were identified –trees had only been removed. Yet as trees increased in height they were reclassified into a different NBH canopy, thus effectively increasing the canopy ≥17.1m canopy stock.

4.5 Conclusion

Urban ecosystems are dynamic – not homogenous patches of vegetation set to equilibrium. There is constant change in the structure and function of the UTI. These changes occur not only across temporal dimensions and scales but also across spatial dimensions and scales. Due to this change, the UTI is open to influence – as it is constantly put under pressures that cause functional and structural shifts. Therefore, external forces need to recognise the panarchal, ever changing nature of urban green infrastructure such as the UTI and manage it accordingly. This is inherently difficult. From the research conducted in Chapter 4 general rules and hypotheses (which can be tested in future studies) can be applied in order to approach managing this difficulty. In summary; 1) the canopy ≥17.1m is most affected by an increase in gap-crossing capability threshold and therefore increasing connectivity in this canopy will provide the most statistically noticeable increase in overall connectivity than the other canopies; 2) connectivity increases the most when gap-crossing capability thresholds increases from 30m to 60m and so if a UTI practitioner can reduce a 60m gap to 30m then it would be more beneficial than closing a, for example, 150m gap; 3) after 90-120m an increase in gap-crossing capability threshold does not appreciably affect an increase in connectivity, therefore studies should focus on species that are more forest specialised and unable to cross large canopy gaps.

Chapter 5 : **Network Analysis and Urban Tree Infrastructure (UTI) Resilience**

5.1 Introduction

Network analysis focuses on the components of a system, how they are structured, and how this structure can affect the performance and state of the system (Jansen *et al.* 2006). Network analysis uses graph theory, an effective tool for the analysis of complex systems such as social-ecological systems arising from urbanisation (Zetterberg *et al.*, 2010). Graphs can be used to model landscapes, describe their underlying structure and functional relationships, and therefore inform conservation practices, landscape planning, and design (Bunn *et al.*, 2000; Rhodes *et al.*, 2006; Minor & Urban, 2007, 2008; Zetterberg *et al.*, 2010; Foltête *et al.*, 2014). The construction of a graph relies on understanding the components which create the landscape or network: that is to say the nodes and links. These basic components form the basis of more complicated structures which can then be related to ecological processes (Table 2-2) (Urban & Keitt, 2001; Galpern *et al.*, 2011). How the ecological processes, which the structural elements represent, are assessed depend on the research aim and the underlying conceptual model of the system under assessment (Urban *et al.*, 2009; Galpern *et al.*, 2011). With regards to the research presented within this thesis, the underlying concept is that landscape structure influences ecological function. More specifically, the structural patterns exhibited by tree canopy patches influences the function of the UTI in regards to the provision of functionally connected habitat and resources for urban passerines.

The variables of these structural components can be related to system resilience (Table 2-2). However, behavioural properties such as organismal perception (i.e. perception towards gaps in the canopy) can strongly influence the values of these structural variables (Urban & Keitt, 2001; Crooks & Sanjayan, 2006a; Farina & Belgrano, 2006)– a relationship often overlooked in the network analysis of system resilience (e.g. Albert *et al.*, 2000; Janssen *et al.*, 2006). In addition, while the graph theory approach to landscape connectivity literature demonstrates this relationship, the use of network analysis has reduced the landscape to two dimensions, thus ignoring other structural patterns existing in the third dimension which may also influence the values of key structural variables (e.g. Bunn *et al.*, 2000; Ricotta *et al.*, 2000; Urban & Keitt, 2001; Rhodes *et al.*, 2006; Minor & Urban, 2008; Biggs *et al.*, 2012). A network approach to assess the resilience of an ecological system should therefore consider organism perception via passerine gap-crossing capability thresholds, describe the focal ecological system in three dimensions by including height data, and describe how changes in gap-crossing capability and vertical structure alter the values of key structural variables (e.g. connectivity and centrality, section 2.4). The research reported in Chapter 5 has hence been undertaken in order to resolve this emerging research gap.

Chapter 5 addresses research question 3 – 'how resilient is the UTI in regards to providing functional levels of connected habitat?' – by means of completing research objective 3.1 (Fig. 3-1). Research objective 3.1 can be achieved by first implementing the connectivity analysis of Chapter 4 (incorporating vertical dimensions and gap-crossing capability thresholds) and then by identifying the canopies which are most important for maintaining connectivity. A network analysis, related to ecological resilience, can then be applied upon the occurring landscape graphs, comprised of the most important canopies. The methods used in order to achieve these research goals are explained in section 5.2. The results of these implemented methods are presented in section 5.3 followed by their meaning and significance in section 5.4. The final research conclusions are presented in section 4.5.

5.2 Methods

The aim addressed in Chapter 5 is to assess the resilience of a UTI by first mapping its horizontal and vertical structure and then incorporate this vertical structure, in conjunction with passerine gap-crossing capability thresholds, into landscape graphs (Figure 5-1). A connectivity analysis and identification of priority habitat patches for maintaining or improving connectivity then follows (Figure 5-1). These identified canopies, critical for maintaining connectivity, are then used to create new landscape graphs upon which a series of network analysis techniques can be applied and related to system resilience (Figure 5-1).

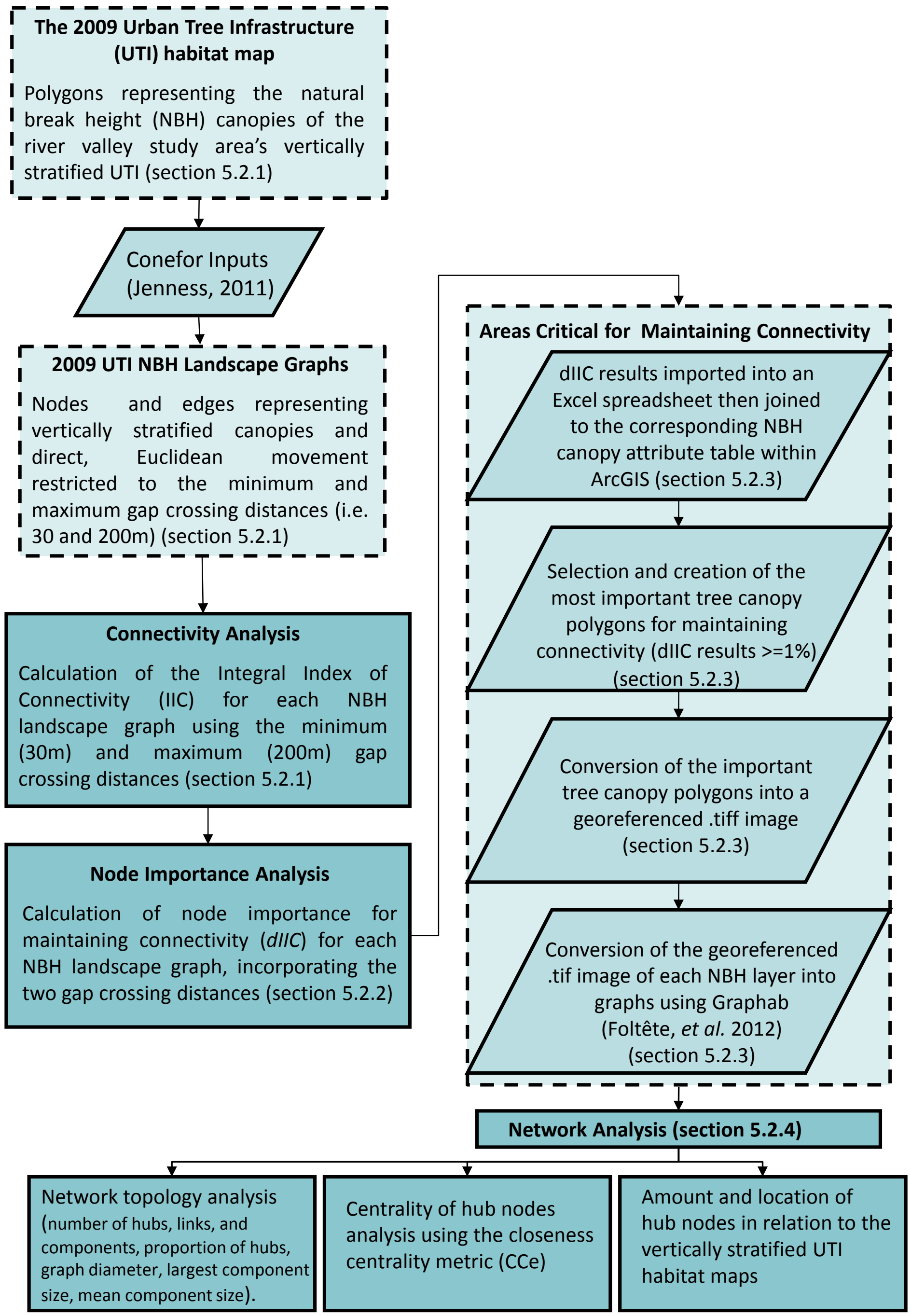

Figure 5-1: A graphical representation of the data (dashed rectangles), processes (parallelogram), and analysis (rectangle) used in Chapter 5.

A total of three data sets (dashed rectangles) were created to conduct the chapter 5 analysis (rectangles). A series of processes (parallelograms) were employed to create these data sets. The creation of the NBH canopies and landscape graphs followed the same methods outlined in Chapter 3 and 4 and summarized in section 5.2.1. The connectivity analysis also follows the same processes as described in Chapter 4 with only the 30m and 200m gap-crossing capability thresholds being incorporated into the connectivity model (section 5.2.1). The node importance analysis uses data created by research undertaken in Chapter 4 and then employs the methods set out in section 5.2.2. The set of processes employed to create connectivity importance graphs are presented in section 5.2.3. Finally, the network analysis methods are presented in section 5.2.4.

Section 5.2.1 contains an outline of the methods used to create habitat maps and landscape graphs. The methods used to calculate UTI canopy importance are presented in section 5.2.2 followed by a description of the methods used to define and map areas critical for maintaining connectivity in section 5.2.3. Finally, the network analysis approach is explained in section 5.2.4

5.2.1 Creating habitat maps and landscape graphs

The methods outlined in Figure 5-1 use the spatial dataset that represents the 2009 river valley study area's UTI as described in Chapter 3. The shapefile containing polygons representing eastern Salford's UTI were stratified based on the natural breaks in the height data and percolated into tree patches using the spatial analysis tool region group (section 3.3 and 3.4). This stratification of the data created the three natural break height (NBH) canopies, canopy ≥3m, canopy ≥7.2m, and canopy ≥17.1m. The ArcGIS extension Conefor Inputs (Jenness, 2011) was used to produce NBH graphs from the NBH canopies by converting tree patches into nodes and creating links between these nodes by incorporating the minimum (30m) and maximum (200m) passerine gap-crossing capability thresholds. The methods employed to do so are the same as those described in section 4.2.6, however the other five gap-crossing capability thresholds were omitted. This exclusion was implemented with the intention of focusing on and exemplifying the differences in extreme gap-crossing capability thresholds. The minimum and maximum gap-crossing capability thresholds can be related to passerines characterised as either forest specialists (30m) or generalists (200m) (section 2.5). The landscape graphs created from using these two gap-crossing capability thresholds, and the resulting analyses of them, can then be related to how the UTI is structured from the differing organism perceptions.

Therefore, in total, 6 NBH graphs were constructed – the canopy ≥3m 30m and 200m graphs, the canopy ≥7.2m 30m and 200m graphs, and the canopy ≥17.1m 30m and 200m graphs. The landscape connectivity software Conefor (Saura & Torne, 2009) was used to calculate the level of landscape connectivity within each of the NBH landscape graph using the integral index of connectivity (IIC) (section 4.2.6)

5.2.2 Identifying important tree canopies

A calculation of tree patch importance for maintaining connectivity is needed in order to assess the levels of centrality exhibited within the UTI system. Furthermore, important tree patches need to be identified so that further network analysis can be undertaken. A network analysis will concentrate on the keystone patches within the river valley study area as it is assumed that it is these patches which are most influential to the resilience of the system. Centrality is an important conceptual tool for exploring networks and identifying those patches which are the most influential within the network (i.e. the most important) (Freeman, 1979). Traditionally, there are three methods for calculating centrality; degree, closeness, and betweenness (Wasserman & Faust, 1994; Opsahl *et al.*, 2010). However, these measurements solely concentrate on the location of the focal patch within the network (i.e. betweenness), how many links there are to other patches in the network (i.e. degree), and the distance of the focal patch to other patches within the network (i.e. closeness). Although these measurements have their benefits, they ignore an intrinsic quality of the patches – patch area. As patch importance, within this thesis, is related to the amount of influence a patch has on landscape connectivity, the actual size of the patch is also important (as larger patches allow for intra-patch movement). Therefore, when initially identifying the central or most important nodes within a network (level of centrality) patch area needs to be incorporated into the calculations used.

Habitat availability indices such as the integral index of connectivity (IIC) incorporate patch size within the calculation of overall landscape connectivity (Saura & Torne, 2009) (Chapter 4). By using the removal method (Urban & Keitt, 2001; Saura & Torne, 2009; Foltête *et al.*, 2014) the overall connectivity value (IIC) of a landscape is compared to the connectivity value calculated after a focal patch is removed from the landscape. The variation between these two values is subsequently calculated for each patch. If the difference between the two values is great, then the focal patch is important for maintaining connectivity. Each patch gains a value of importance given as a percentage. The removal method (Urban & Keitt, 2001) was therefore used to identify the most important patches within the stratified canopies of the river valley study area when described as landscape graphs. Due to the use of landscape graphs, the important habitat patches are represented as nodes. If a node is identified as being critical to maintaining connectivity it is termed a hub node.

The connectivity software Conefor (Saura & Torne, 2009) was used to implement the removal method using the equation:

$$dIIC(\%) = 100.\frac{IIC - IIC_{remove}}{IIC}$$

Equation 5-1: Node importance for maintaining connectivity (*dIIC*)

where *IIC* is the overall connectivity index value when all nodes are present within the NBH layer graph and IIC_{remove} is the overall index value after the removal of a focal node from the graph. Node importance (*dIIC*) was calculated for the 6 NBH landscape graphs incorporating the minimum and maximum gap-crossing capability threshold thresholds – canopy ≥3m at 30m and 200m; canopy ≥7.2m at 30m and 200m; canopy ≥17.1m at 30 and 200m. It was conceded that only the nodes with a *dIIC* value of 1% or more were to be used in the creation of the ‘areas critical for landscape connectivity’ maps and graphs. These nodes were therefore acknowledged as being at least 1% influential to maintaining connectivity and therefore representative of a hub node. Nodes with dIIC values less than 1% provide negligible influence over landscape connectivity.

5.2.3 Creating habitat maps and landscape graphs of areas critical for connectivity

To create an ‘areas critical for connectivity map’, the dIIC (node importance) results for each NBH graph were imported into an Excel spreadsheet and joined to the corresponding NBH canopy shapefile attribute table. Habitat patches with a dIIC value of 1% or more (i.e. corresponding to a landscape graph hub node) were then selected within the attribute table and exported to create the ‘NBH patch importance canopies’; canopy ≥3m important canopy patches, canopy ≥7.2m important canopy patches, and canopy ≥17.1m important canopy patches. A new field, named Canopy, was added within the attribute table of these shapefiles so that each tree canopy was given the habitat value of *1* and non-habitat (i.e. the matrix) was equal to *0*.

This habitat map was then used to create an 'areas critical for connectivity graph' by converting the vector data into raster data. The raster datasets were saved as a .tiff image so that it would be compatible within the ecological network analysis and visualisation software Graphab (Foltête *et al.,* 2012). Graphab can also be used for cartographic interfacing and therefore the .tiff image of important tree canopy patches needed to be georefferenced. The important canopy patches image was therefore exported as a geotiff (.tfw) extension using the ArcMap image and raster data management tool 'export raster world. This extension allowed the important canopy patches image to be converted into a landscape graph while retaining its spatial reference.

The landscape graphs were created within Graphab by creating both a node and link dataset. To identify which components of the landscape should be transformed into nodes, a habitat value must be entered. Therefore the Canopy value of 1 was entered and NoData was ignored. Neighbourhood was defined by setting patch connectivity to 8 (although patches were already defined for each NBH canopy when they were created – see section 3.3). To create links a distance threshold must be entered. For each NBH patch importance canopy two link datasets were created corresponding to the smallest and largest gap-crossing capability thresholds used in the *dIIC* analysis (i.e. for the *dIIC r*esults created from the 30m threshold connectivity analysis, the link threshold was set to 30m). This resulted in six NBH node importance graphs - three NBH canopies relating to two gap-crossing capability thresholds (30m and 200m).

5.2.4 Network analysis of the areas critical for connectivity graphs

An assessment of each NBH node importance graphs structure was undertaken within Graphab (Foltête *et al.*, 2012), using a series of calculations and topological metrics. The number of hub nodes within the landscape graph provides an indication of the level of centrality exhibited within the stratified canopies of the river valley study area's UTI. The number of these hub nodes within each NBH node importance graph was summed and then used to calculate the proportion of hub nodes within each graph compared to the overall number of nodes. The total number of links between the hub nodes was then counted.

In order to understand the greatest distance a passerine could travel without crossing a gap more than its perceptual range (in this case either 30m or 200m), the overall graph diameter (GD) was calculated. A graph's diameter is the longest possible path between two nodes in the graph (with no node being visited more than once), where the path length is itself the shortest possible path (Urban & Keitt, 2001). A short diameter therefore implies that fast movement is possible for a focal organism (Minor & Urban, 2008), on the other hand a longer graph diameter means greater connectivity within the graph. The greatest distance between two nodes within the graph (GD) was calculated using the equation:

$$GD = \max_{ij} d_{ij}$$

Equation 5-2: Graph diameter (*GD*).

where *dij* is the shortest possible distance between two focal nodes (i.e. least cost distance).

The number of components represents the number of areas critical for connectivity. Within this chapter, components arise when a set of hub nodes are connected to each other but remain separate from the rest of the hub nodes in the network – thus creating sub-networks. Therefore, the total number of components was simply counted once the graphs were created. As movement can occur between any two patches (i.e. nodes) within a component, it is assumed that the larger the size of the component, the more movement that can occur within it. More movement means more connectivity, which influences resilience. Furthermore, within Graphab, area is related to capacity – the ability of a patch or component to accommodate a given number of individual organisms (Foltête *et al.*, 2012). That is to say, a habitat patch or component with a high capacity can accommodate a large population and vice versa (Foltête *et al.*, 2012). Therefore, the size of the largest component (SLC) and the mean size of the components (MSC) can be used to compare differences in potential connectivity and capacity between each NBH node importance graph. The SLC metric is calculated using the equation:

$$SLC = \max\{ac_k\}$$

Equation 5-3: Size of largest component (*SLC*).

where ac_k is the capacity of the component calculated by summing the area of the nodes which comprise that component. The MSC metric is calculated using the equation:

$$MSC = \frac{1}{nc}\sum_{k=1}^{nc} ac_k$$

Equation 5-4: Mean size of the components (*MSC*).

where *nc* is the number of components within the network.

Within this thesis the level of centrality is considered as the total number of structurally important nodes within the landscape – calculated using the removal method (chapter 5.2.2). Furthermore, importance was not only assessed by considering the number of links a node has (i.e. inter-patch connectivity of a habitat patch) but also its area – as intra-patch movement increases connectivity (section 2.3). This approach was useful to identify the areas critical for maintaining connectivity and the key habitat patches within them. However, centrality is also traditionally considered as the number of nodes that are connected to a focal node (Freeman, 1979). Under this definition, nodes with high centrality have the most influential links to other nodes within the landscape graph that they produce (Wasserman & Faust, 1994; Opsahl *et al.*, 2010). By using this definition, centrality measurements can also be used to uncover the spatial relationship between the sub-networks within landscape graphs that are formed from compartmentalised hub nodes. Therefore, a further topological assessment was undertaken in order to quantify the level of centrality of each hub node within their corresponding component. This structural relationship was quantified by using the closeness centrality metric (*CCe*) (Freeman, 1979; Urban *et al.*, 2009; Foltête *et al.*, 2012). The metric considers the length of the paths from a focal node to all other nodes within its component using the equation:

$$CCe_i = \frac{1}{n_k - 1}\sum_{\substack{j=1 \\ j \neq i}}^{n_k} d_{ij}$$

Equation 5-5: Closeness centrality metric (*CCe*).

where n_k is the number of nodes in the component and d_{ij} is the distance between the focal node and another node within the component.

The NBH node importance graphs were also exported as ArcGIS shapefiles and layered upon the NBH canopy maps, thus integrating habitat maps with landscape graphs. This integration of graph and map, along with the network analysis results, allowed for a qualitative assessment of the relationship between network centrality, hub node connectivity, and the overall resilience of the river valley study area's UTI.

5.3 Results

5.3.1 Network analysis and landscape graphs

Table 5-1 contains the results from the network analysis of the 6 NBH node importance graphs. The network analysis results of the canopy ≥3m node importance graphs reveals minor differences between graphs incorporating the minimum and maximum gap-crossing capability thresholds. The number of hub nodes (NH) decreased by 8 from the canopy ≥3m 30m graph to the 200m graph (NH 30m = 29, NH 200m = 21). Therefore, the proportion of hub nodes (PH) within the entire network, while depleted, increased slightly within the canopy ≥3m 30m graph (PH 30m = 0.28%, PH 200m = 0.20%). The two gap-crossing capability threshold graphs contained the same number of links (NL = 28) and graph diameter (GD) was only slightly longer by 144m in the canopy ≥3m 30m graph (GD 30m = 2,152m, GD 200m = 2,008). The number of components (NC) decreased from the canopy ≥3m 30m graph to the 200m graph by 1 (NC 30m = 5, NC 200m = 4) and the size of the largest component (SLC) in the canopy ≥3m 30m graph was only 1.06 times greater in size than the 200m graph (SLC 30m = 590,082m^2, SLC 200m =

554,468m^2) . Conversely, the mean component size (MCS) of the canopy ≥3m 200m graph was 1.2 times higher than the 30m graph (MSC 30m = 163,481m^2, MSC 200m = 200,099m^2).

Table 5-1: Network analysis results for each natural break height (NBH) node importance graphs.

NBH Canopy	No. of Hubs (NH)		Proportion of Hubs (PH)		No. of Links (NL)		Graph Diameter (GD)(m)		No. of Components (NC)		Size of Largest Component (SLC) (m²)		Mean Size of Components (MSC) (m²)	
	30m	*200m*	*30m*	*200m*	*30m*	*200m*	*30m*	*200m*	*30m*	*200m*	*30m*	*200m*	*30m*	*200m*
Canopy ≥3m	29	21	0.28%	0.20%	28	28	2152	2008	5	4	590082	554468	163481	200099
Canopy ≥7.2m	18	19	0.38%	0.41%	9	21	29	453	9	3	317400	469067	64645	205199
Canopy ≥17.1m	40	32	4.88%	3.90%	25	49	159	563	15	10	35434	83428	8468	12556

A total of seven calculations and landscape metrics were implemented in the network analysis of the NBH node importance graphs. The results of the node importance graphs network analysis are altered as gap-crossing capability threshold (i.e. organism perception) changes. The results or the canopy ≥3m are in the blue cells, the canopy ≥7.2m in the green cells, and the canopy ≥17.m in the red cells. The influence gap-crossing capability threshold has on the network analysis results is dependent on the NBH canopy under assessment.

The network analysis for the canopy ≥7.2m node importance graphs revealed a variance of results which were dependant on the change in gap-crossing capability threshold (Table 5-1). The total amount of hub nodes was greater within the canopy ≥7.2m 200m graph by only 1 (NH 30m = 18, NH 200m = 19). The proportion of hub nodes within the entire network was low, with the canopy ≥7.2m 30m graph comprising 0.38% and the 200m graph 0.41% of the total network. The amount of links within the two gap-crossing capability threshold graphs differed greatly, with the canopy ≥7.2m 200m graph having more than double the amount of links (30m = 9, 200m = 21). Therefore, graph diameter within the canopy ≥7.2m 200m graph was around 16 times greater than the graph diameter exhibited within the 30m graph (GD 30m = 29m, GD 200m = 453m). The canopy ≥7.2m 30m graph contained triple the number of components (NC 30m = 9, NC 200m = 3) yet the largest component was around 1.5 times smaller than the largest component within the 200m graph (SLC 30m = 317,400m^2, SLC 200m = 469,067m^2). Furthermore, the mean size of components within the canopy ≥7.2m 30m graph was 3 times smaller than those within the 200m graph (MSC 30m = 64,645m^2, MSC 200m = 205,199m^2).

The greatest number of hub nodes, proportion of hub nodes within the network, links, and components were calculated for the canopy ≥17.1m node importance graphs. However, the largest and mean sizes of components were the smallest for the canopy ≥17.1m node importance graphs. The canopy ≥17.1m 30m graph contained 8 more hub nodes than the 200m graph (NH 30m = 40, NH 200m = 32). The proportion of hub nodes within the network was nearly 5% within the canopy ≥17.1m 30m graph, decreasing to nearly 4% in the canopy ≥17.1m 200m graph. The number of links within the canopy ≥17.1m 200m graph was nearly double that of the canopy ≥17.1m 30m graph (NL 30m = 25, NL 200m = 49). There was an increase in graph diameter of 404m from the canopy ≥17.1m 30m graph to the 200m graph, meaning the 30m graph diameter was around 4 times longer than the 200m graph diameter (GD 30m = 159m, GD 200m = 563m). There was found to be five more components within the canopy ≥17.1m 30m graph than the 200m graph (NC 30m = 15, NC 200m = 10) and the largest of the canopy ≥17.1m 30m graph components was just over two times smaller than the largest component of the 200m graph (SLC 30m = 35,434m^2, SLC 200m = 83,428m^2). There was a minor difference in the mean size of components between the two canopy ≥17.1m graphs as the 200m graph exhibiting a larger mean component size only 1.5 times larger than the 30m graph (SLC 30m^2 = 8,468, SLC 200m = 12,556m^2).

Visual representations of the results within Table 5-1 were created in order to understand how these differences in network results are rendered within the landscape graphs (Figure 5-2, Figure 5-3, Figure 5-4, Figure 5-5, Figure 5-6, Figure 5-7). The components of the landscape graphs presented below contain sub-networks (localised areas of connectivity within the overall network). Graph trees occur within these sub-networks when they are comprised of more than three connected nodes. If the path of these graph trees becomes closed (i.e. all nodes are connected to each other so that no node s visited twice) then the graph tree, in the sub-network, becomes a cycle. Therefore, sub-networks can embody graph trees or cycles but not solitary or single link nodes (i.e. two nodes connected together).

In the canopy ≥3m 30m graph the largest component (component 1) contains a sub-network composed of an 18 node graph tree (Figure 5-2). A 6 node graph tree produces the sub-network in component 2 and a 3 node graph tree can be found within component 3 (Figure 5-2). Components 4 and 5 contain only a single hub node and therefore no sub-network. As gap-crossing capability threshold increase to 200m the canopy ≥3m landscape graph begins to change shape (Figure 5-3). Components 1 and 4 of the canopy ≥3m graph (Figure 5-2) join to create component 2 of the 200m graph (Figure 5-3). Nodes were lost in the process of this change so that the sub-network of component 2 is made up of a graph tree containing 10 nodes (Figure 5-3). Another major change between the two gap-crossing capability threshold graphs is that the other two tree graphs in the 30m graph (Figure 5-2) become a single links away from becoming cycles in the 200m graph (Figure 5-3). Specifically, in component 1 in the canopy ≥3m 200m graph the sub-network is composed of a 6 node tree graph and only one node is a link short to creating a cycle. A similar 4 node tree graph exists in component 3 where an extra link would transform it into a cycle. Component 4 however contains only a single hub node (Figure 5-3).

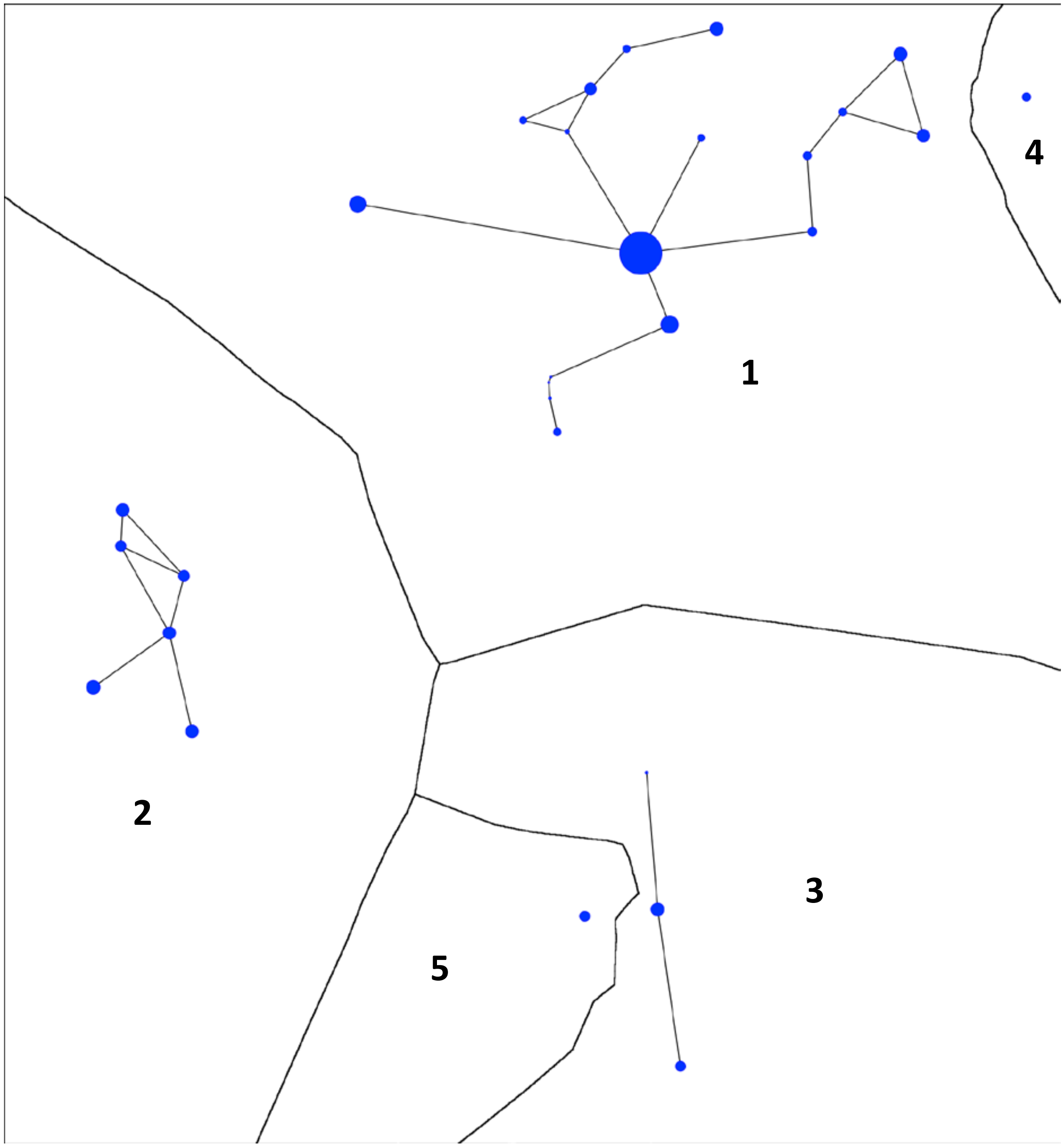

Figure 5-2: The canopy ≥3m node importance graph created using the 30m gap-crossing capability threshold. A total of 5 components were identified via the node importance (dIIC) analysis. In these components three sub-networks were identified, composed of either an 18 node graph tree (component 1), a 6 node graph tree (component 2), or a 3 node graph tree (component 3). Single hub nodes were identified in components 4 and 5. The larger the node the greater its area (i.e. tree canopy capacity). Polygons represent landscape components (size of component calculated by the total area of the nodes, not the area of the polygon).

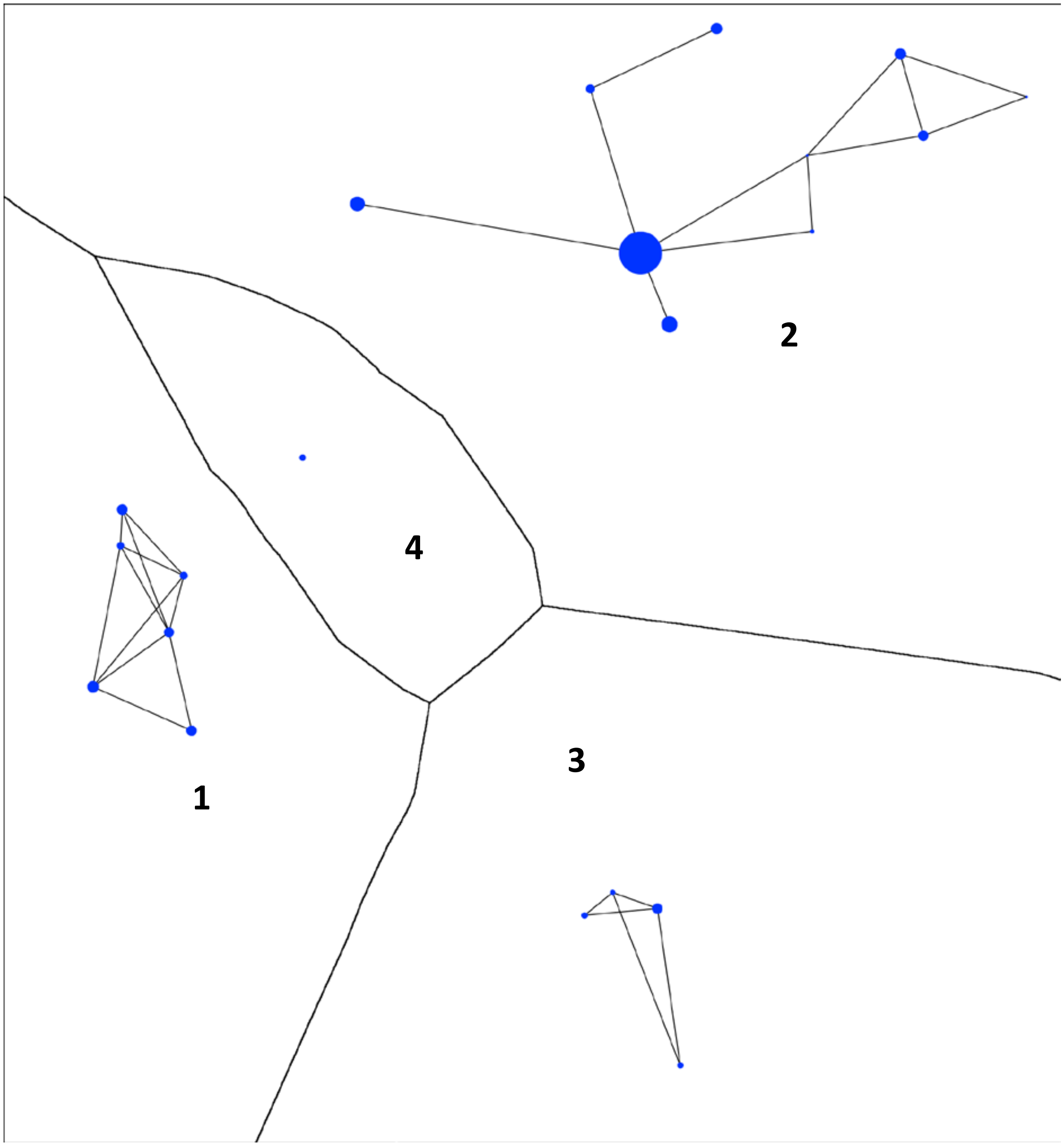

Figure 5-3: The canopy ≥3m node importance graph created using the 200m gap-crossing capability threshold. The node importance analysis (dIIC) revealed 4 components within the landscape graph. Out of the four components three contained graph trees which were either composed of 10 (component 2), 6 (component 1), or 3 nodes (component 3). Components 1 and 3 were one link away from forming a cycle. Component 4 contained an individual hub node. The larger the node the greater its area (i.e. tree canopy capacity). Polygons represent landscape components (size of component calculated by the total area of the nodes, not the area of the polygon).

Even though there was only one extra hub node within the canopy ≥7.2m 200m graph then the 30m graph (Table 5-1) there was a substantial difference in graph structure between the two (Figure 5-4 and Figure 5-5). Unconnected, isolated hub nodes can be seen in over half of the components (5 out of 9) within the canopy ≥7.2m 30m graph (components 3, 5, 7, 8, and 9 in Figure 5-4). Component 2 contained the most connected graph tree in canopy ≥7.2m 30m graph, made up of 4 nodes with only three links between them (Figure 5-4). The other graph trees within the canopy ≥7.2m 30m graph contained 3 nodes (components 1, 4, and 6) and there were no cycles present (Figure 5-4). As gap-crossing capability threshold increased to 200m the number of components decreased to three and each component contained a sub-network, all of which were made up of graph trees - although the graph tree in component 2 was one link away from becoming a cycle (Figure 5-5). The greatest number of connected nodes in a graph tree was exhibited in component 3, made up of 11 nodes and connected by 12 links between them (Figure 5-5)

The canopy ≥17.1m node importance graphs were smaller and more fragmented than the other two node importance graphs (Figure 5-6 and Figure 5-7). Furthermore, the canopy ≥7.2m and canopy ≥3m 200m node importance graphs share similarities in graph structure, particularly in terms of the graph trees that were revealed within components (Figure 5-2, Figure 5-3,Figure 5-4, and Figure 5-5). The canopy ≥17.1m 30m and 200m node importance graphs reveal rather different graph structures. The graph trees within the canopy ≥17.1m 30m graph were all linear, forming no node clusters or cycles, and the maximum possible number of paths within the trees was 2 (components 1, 3, 6, and 7 in Figure 5-6) . The majority of the components in the canopy ≥17.1m 30m graph contained a single node (components 2, 5, 9, 11, 12, 13, 14, and 15) while the rest contained 2 connected nodes (8, 4, and 10, Figure 5-6). However, in the canopy ≥17.1m 200m graph component 6 contained a large sub-network made up of a highly connected 19 node graph tree (Figure 5-7). This graph tree was the most complex of all the components described for all NBH node importance graphs, containing 48 links and exhibiting localised cycles within the graph tree (component 6 in Figure 5-7). However, as the component absorbed other smaller components, which previously existed within the canopy ≥17.1m 30m graph (Figure 5-6) and the graph trees and hub nodes within them, there is only one other, 3 node, graph tree in the network (component 9 in Figure 5-7). The rest of the components in the canopy ≥17.1m 200m graph contained either 2 connected nodes (components 2 and 4) or a single hub node (components 1, 3, 5, 7, 8, and 9, Figure 5-7).

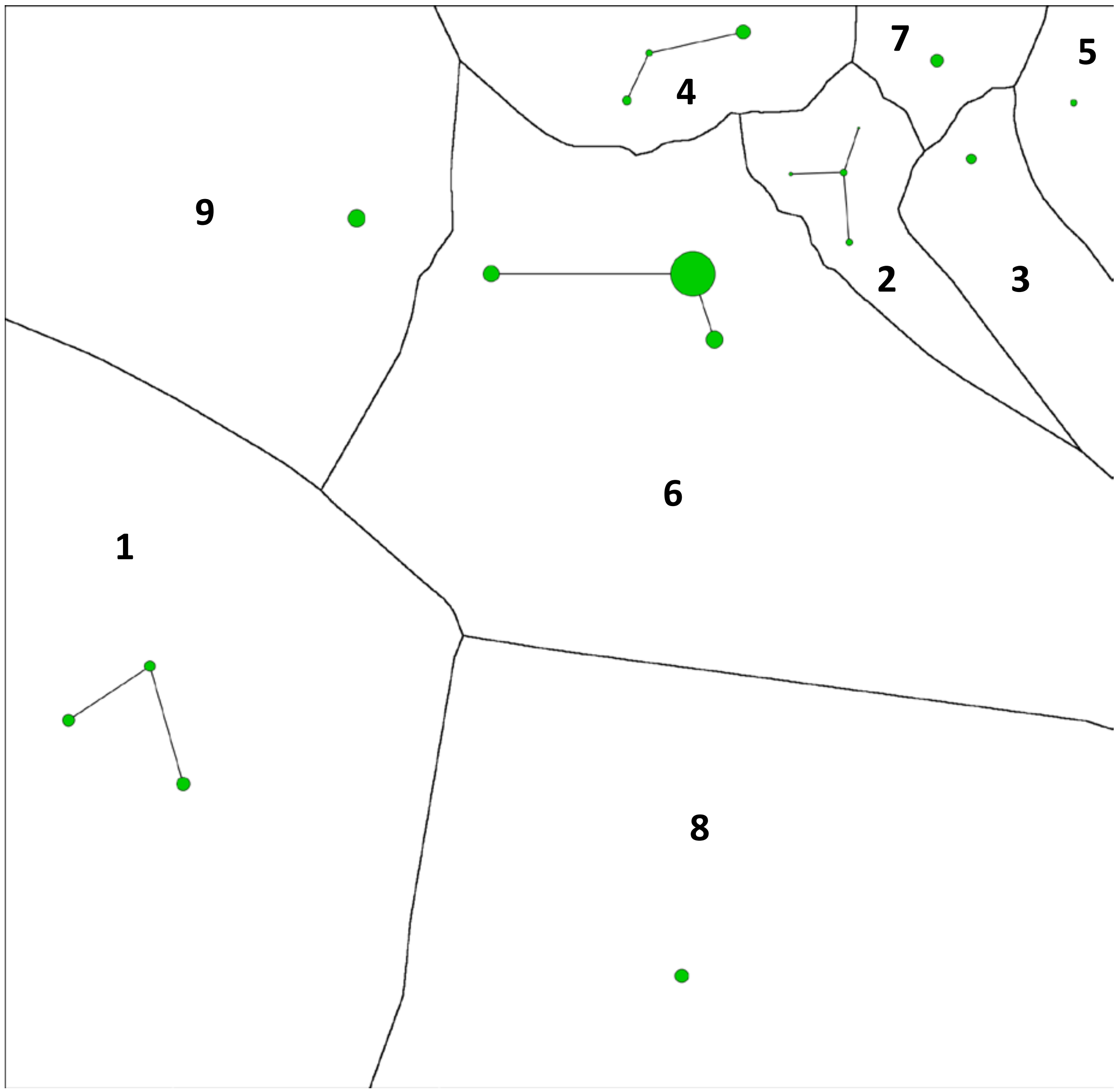

Figure 5-4: The canopy ≥7.2m node importance graph created using the 30m gap-crossing capability threshold. The node importance (dIIC) analysis revealed 9 components containing 4 sub-network composed of either 3 (components 1, 4, and 6) or 4 (component 2) node graph trees. The other five components contained only a single hub node and therefore no sub-network existed. The larger the node the greater its area (i.e. tree canopy capacity). Polygons represent landscape components (size of component calculated by the total area of the nodes, not the area of the polygon).

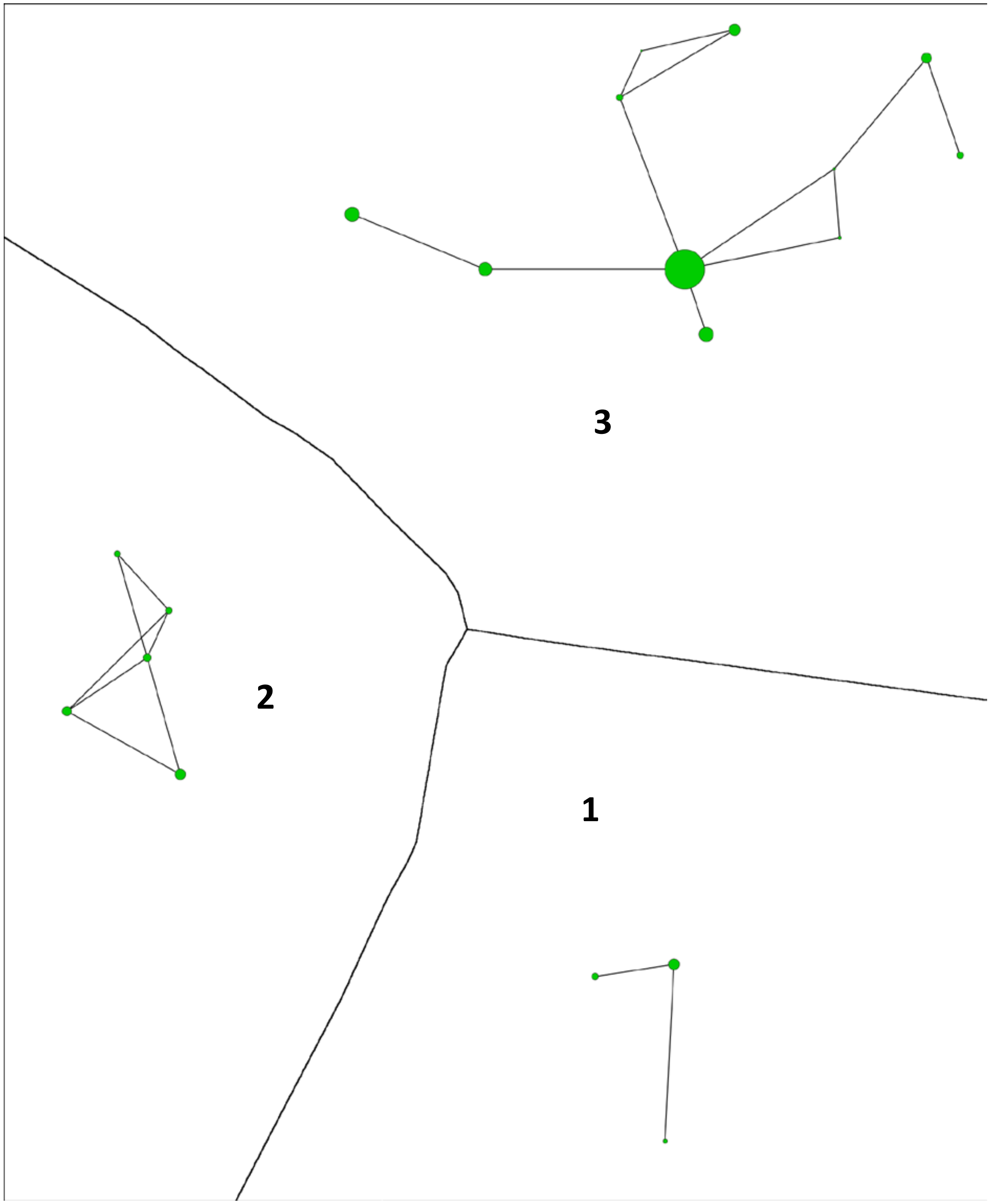

Figure 5-5: The canopy ≥7.2m node importance graph created using the 200m gap-crossing capability threshold. Three components were identified through the node importance (dIIC) analysis. All three components contained a graph tree. The number of nodes comprising these graph trees were either 11 (component 3), 5 (component 2), or 3 (component 1). The graph tree in component 2 is close to forming a cycle (one link missing). The larger the node the greater its area (i.e. tree canopy capacity). Polygons represent landscape components (size of component calculated by the total area of the nodes, not the area of the polygon).

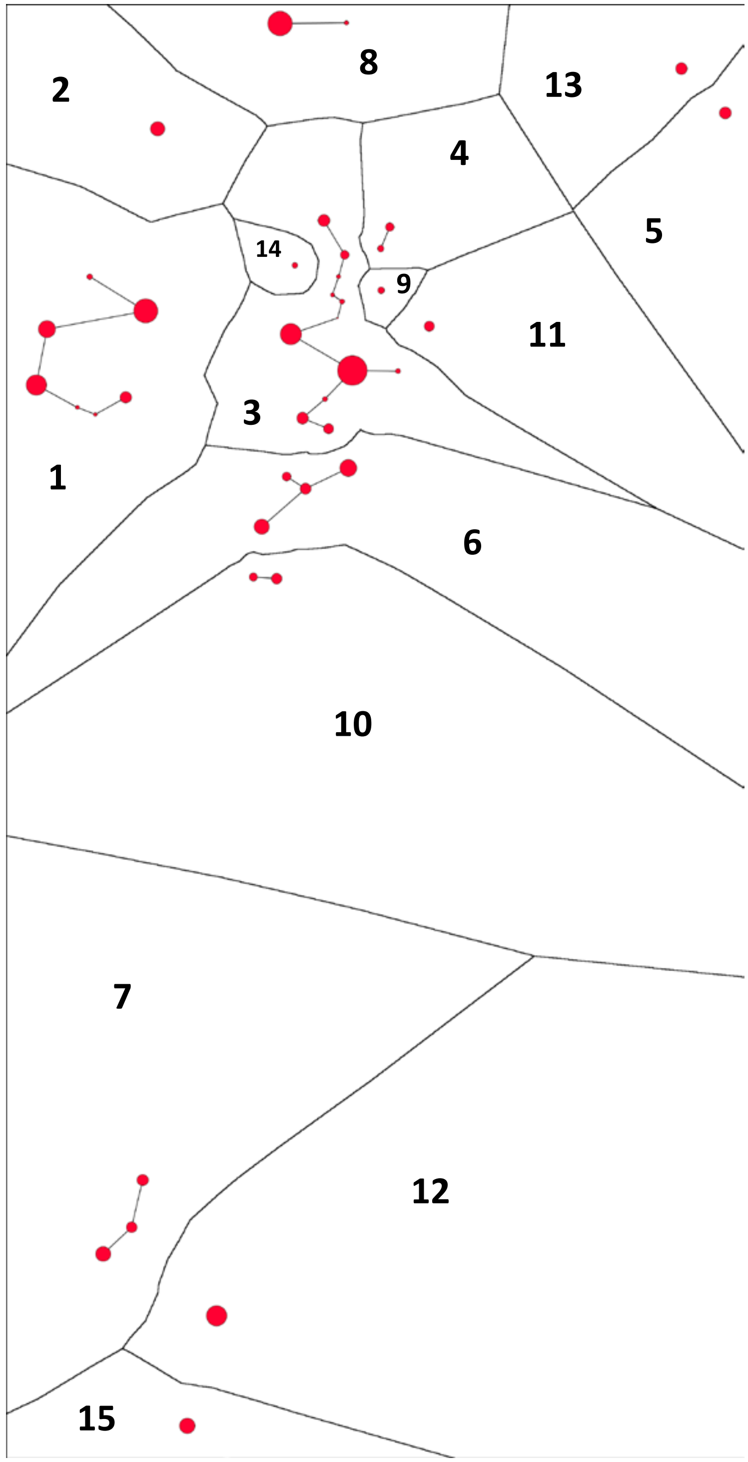

Figure 5-6: The canopy ≥17.1m node importance graph created using the 30m gap-crossing capability threshold. A total of 15 components were described due to the node importance (dIIC) analysis. A total of 4 graph trees were identified within 4 of these components, all of which were linear (components 1, 3, 6, 7). Three of the components contained just 2 connected hub nodes (components 4, 8, and 10) while the rest of the components only contained an individual hub node (components 2, 5, 9, 11, 12, 13, 14, and 15). The larger the node the greater its area (i.e. tree canopy capacity). Polygons represent landscape components (size of component calculated by the total area of the nodes, not the area of the polygon).

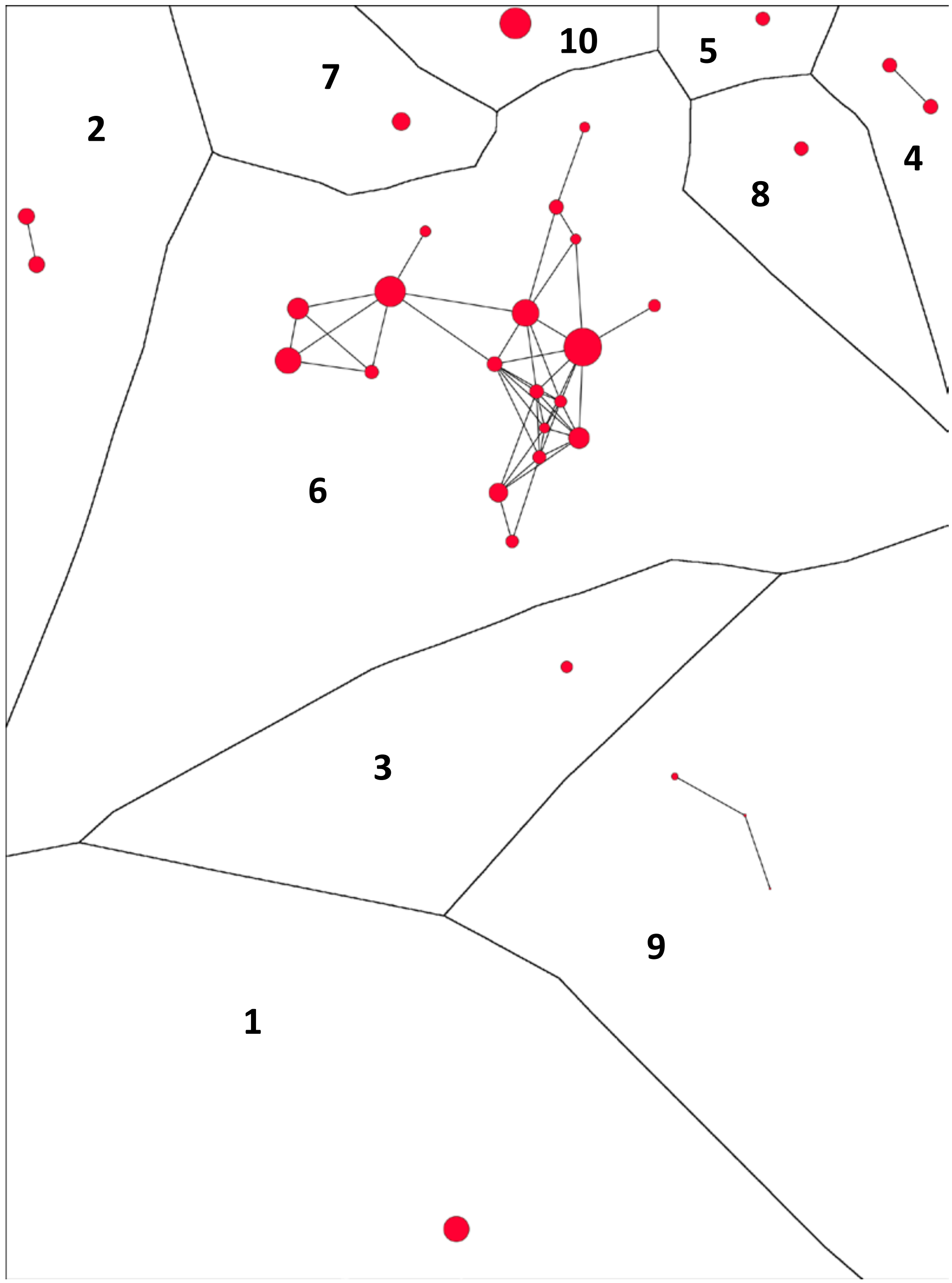

Figure 5-7: The canopy ≥17.1m node importance graph created using the 200m gap-crossing capability threshold.

The node importance (dIIC) analysis revealed that the landscape graph is composed of 10 components, two of which contained sub-networks composed of graph trees contained either 19 (component 6) or 3 nodes (component 9). The rest of the components contained single hub nodes (components 1, 3, 5, 7, 8, and 9) or 2 connected nodes (components 2, and 4). The larger the node the greater its area (i.e. tree canopy capacity). Polygons represent landscape components (size of component calculated by the total area of the nodes, not the area of the polygon).

5.3.2 Closeness centrality (CCE) of sub-networks

The closeness centrality (CCe) results demonstrate the varied influence hub nodes have for maintaining component sub-network structure and overall component connectivity. The larger and redder the nodes in the NBH node importance graphs the less influence they have for maintaining connectivity and structure (Figure 5-8, Figure 5-9, Figure 5-10, Figure 5-11,Figure 5-12, and Figure 5-13). Within the canopy ≥3m node importance graphs component 1 of the canopy ≥3m 30m graph (Figure 5-8) and component 2 of the canopy ≥3m 200m graph (Figure 5-9) were the only components which contained graph trees exhibiting noticeably varied CCe values. The other components contained graph trees with a relatively uniform level of centrality or single hub nodes with a CCe value of 0. This result can be attributed to the existence of sub-networks that have nodes forming linear graph trees, or, more effectively, have graph trees that are close to forming a cycle (Figure 5-8 and Figure 5-9). Component 1 of the canopy ≥3m 30m graph contained a central, important node which had outwardly spiralling ‘satellites’ made up of less important, high CCe nodes (Figure 5-8). Component 2 of the canopy ≥3m 200m graph also contained the same central node, however the ‘satellite’ nodes which surround it did not show as high CCe results as component 1 of the canopy ≥3m 30m graph due to the increase in gap-crossing capability threshold (Figure 5-9).

The canopy ≥7.2m 30m and the canopy ≥7.2m 200m node importance graphs reveal noticeably varied CCe values (Figure 5-10 and Figure 5-11). The canopy ≥7.2m 30m graph contains 9 components, of which 5 contain a single hub node, therefore these nodes have a CCe value of 0 (Figure 5-10). Within the 4 components that contain connected nodes in the form of graph trees, the CCe results reveal, intuitively, that the centre node has the lowest CCe value (components 1, 2, 4, and 6 in Figure 5-10). The 3 node graph structure in component 4 contained the overall highest CCe values and as such the hub nodes comprising the graph tree maintain relatively little influence over each other (Figure 5-10). The entire landscape graph then changes as gap-crossing capability threshold increases to 200m and as a result there were only 3 components and no single hub nodes in the canopy ≥7.2m 200m graph. Consequently, the lowest CCe value exhibited by the canopy ≥7.2m graph trees was 25.7 (component 2 in Figure 5-11). The overall lowest CCe values were calculated for the nodes that make up the graph tree in component 2 as they very nearly formed a cycle (Figure 5-11). The result of this structure means the graph tree nodes were nearly all connected to each other so that they would all be central to the sub-network and influential for maintaining connectivity within the component (Figure 5-11). There was also little variation in CCe results within component 1 with the centre hub node having a slightly lower CCe values (Figure 5-11). The greatest variation in CCe can be seen in component 3 (Figure 5-11). A central hub node was connected to four 'satellites' which increased in CCe the further they were from the centre hub node (Figure 5-11). The highest CCe results can be seen to the furthest right of component 3 (CCe= 309.7), thus that hub node has least influence on maintaining component 3 sub-network connectivity.

Most components within the canopy ≥17.1m node importance graphs contained hub nodes with equal levels of centrality (Figure 5-12 and Figure 5-13). The only exceptions were components 1 and 3 within the canopy ≥17.1m 30m graph (Figure 5-12), and components 9 and 6 within the canopy ≥17.1m 200m graph (Figure 5-13). Components 1 and 3 contained sub-networks structured as linear tree graphs with relatively little centrality (hub nodes sitting in the centre of the graph tree had a CCe score of around 46) (Figure 5-12). The nodes of the graph trees had little influence over each other or maintaining component connectivity. However, as gap-crossing capability threshold increased to 200m, components 1 and 3 merged together (while losing and gaining hub nodes) (Figure 5-13). This merge created the larger component 6 which contained the most complex graph tree described within this analysis (Figure 5-13). To the right of component 6 lie the central hub nodes which form an almost complete cycle, as highlighted by the relatively equal distribution of CCe results. This central 'pseudo-cycle' also has high CCe nodes attached to it, 4 towards the top of the graph tree and 2 towards the bottom (component 6 in Figure 5-13). Furthermore, to the left of graph tree lays a low centrality cluster of 5 hub nodes with high CCe values (component 6 in Figure 5-13). In Figure 5-7 it seemed as if this cluster formed a localised cycle within component 6's graph tree. However, the CCe values show that this was not the case but rather the cluster to the left of the graph tree was too far removed from the more important central nodes and therefore not as influential in maintaining sub-network connectivity.

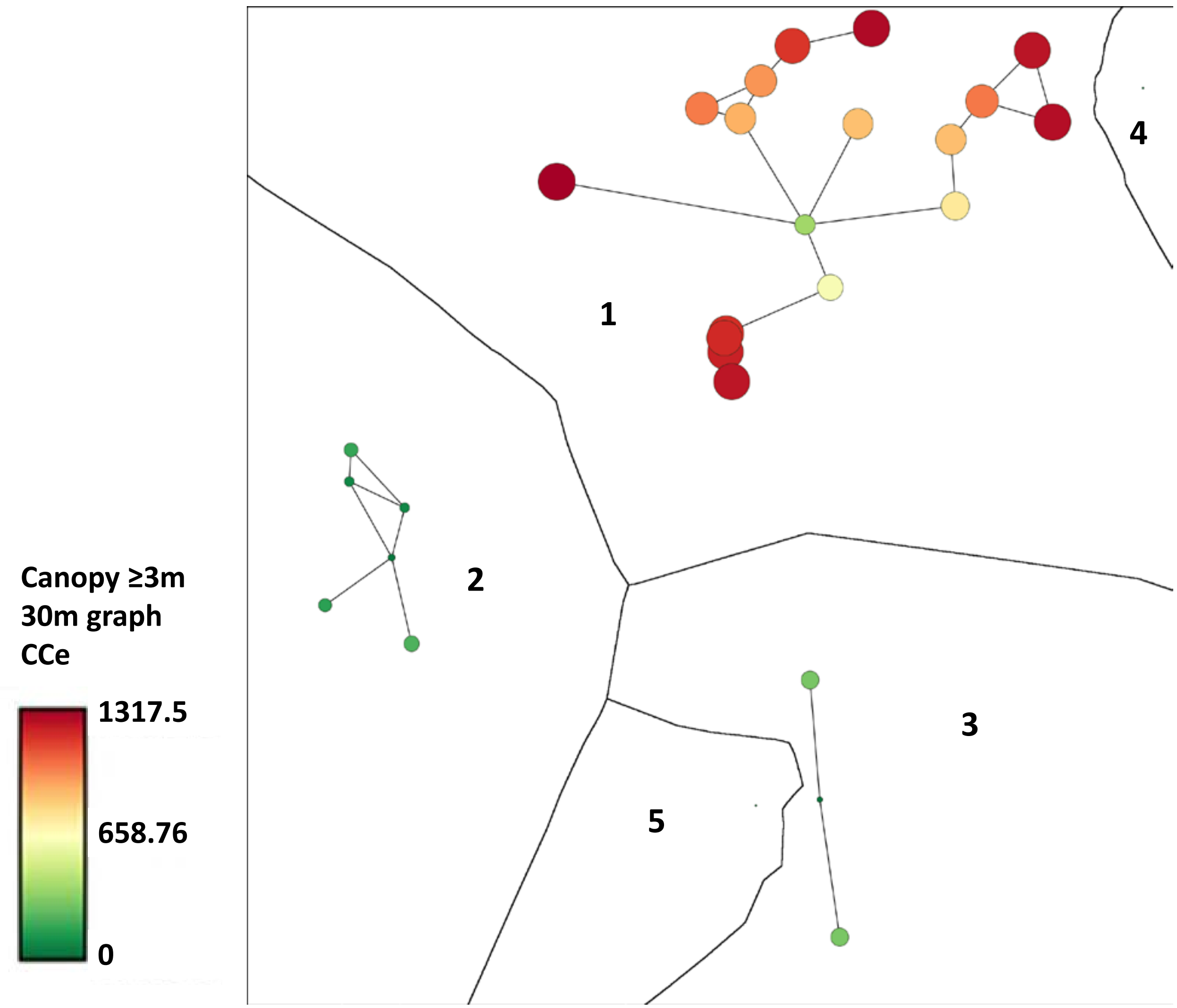

Figure 5-8: Closeness centrality results of the canopy ≥3m node importance graph created using the 30m gap-crossing capability threshold.
Component 1's graph tree CCe values range from 388.14 (centre node) to the maximum value of 1317.51 (far left node). In component 2 the graph tree exhibits CCe values from 44.76 (centre node) to 223.15 (bottom right node). Component 3's graph tree CCe values range from 27.26 (centre node) to 306.24 (bottom node). The solitary hubs nodes in both components 4 and 5 exhibit CCe values of 0.

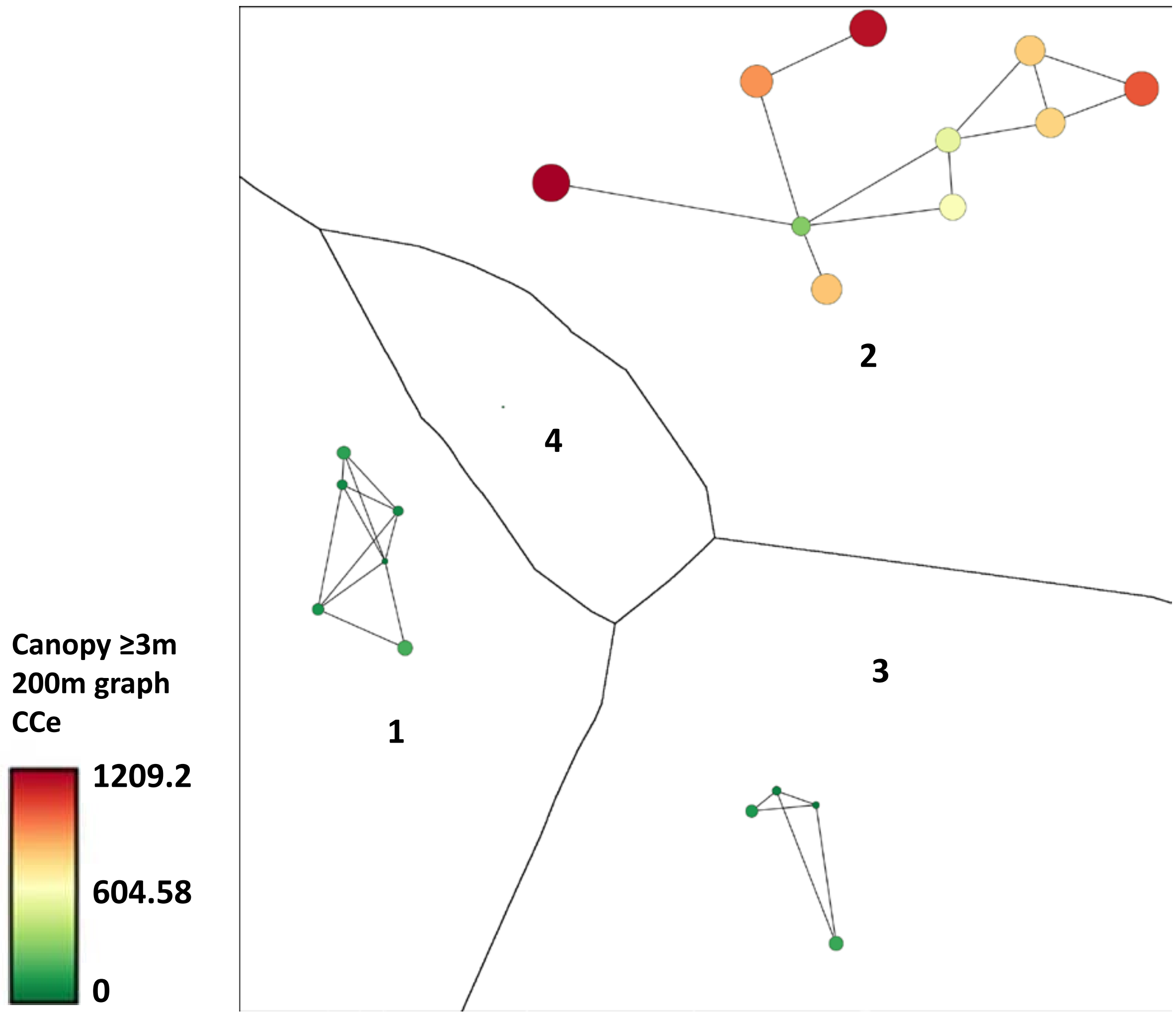

Figure 5-9: Closeness centrality results of the canopy ≥3m node importance graph created using the 200m gap-crossing capability threshold.
In component 1 the sub-network's graph tree exhibits CCe values ranging from 28.45 (centre node) to 191.04 (bottom right node). In component 2 the lowest CCe value was calculated for the centre node (301.23) while the highest value was calculated for the node farthest left (1209.17). The sub-network's graph tree in component 3 exhibits CCe values that range from 38.67 (top-right node) to 170.62 (bottom node). The single hub node in component 4 has a CCe value of 0.

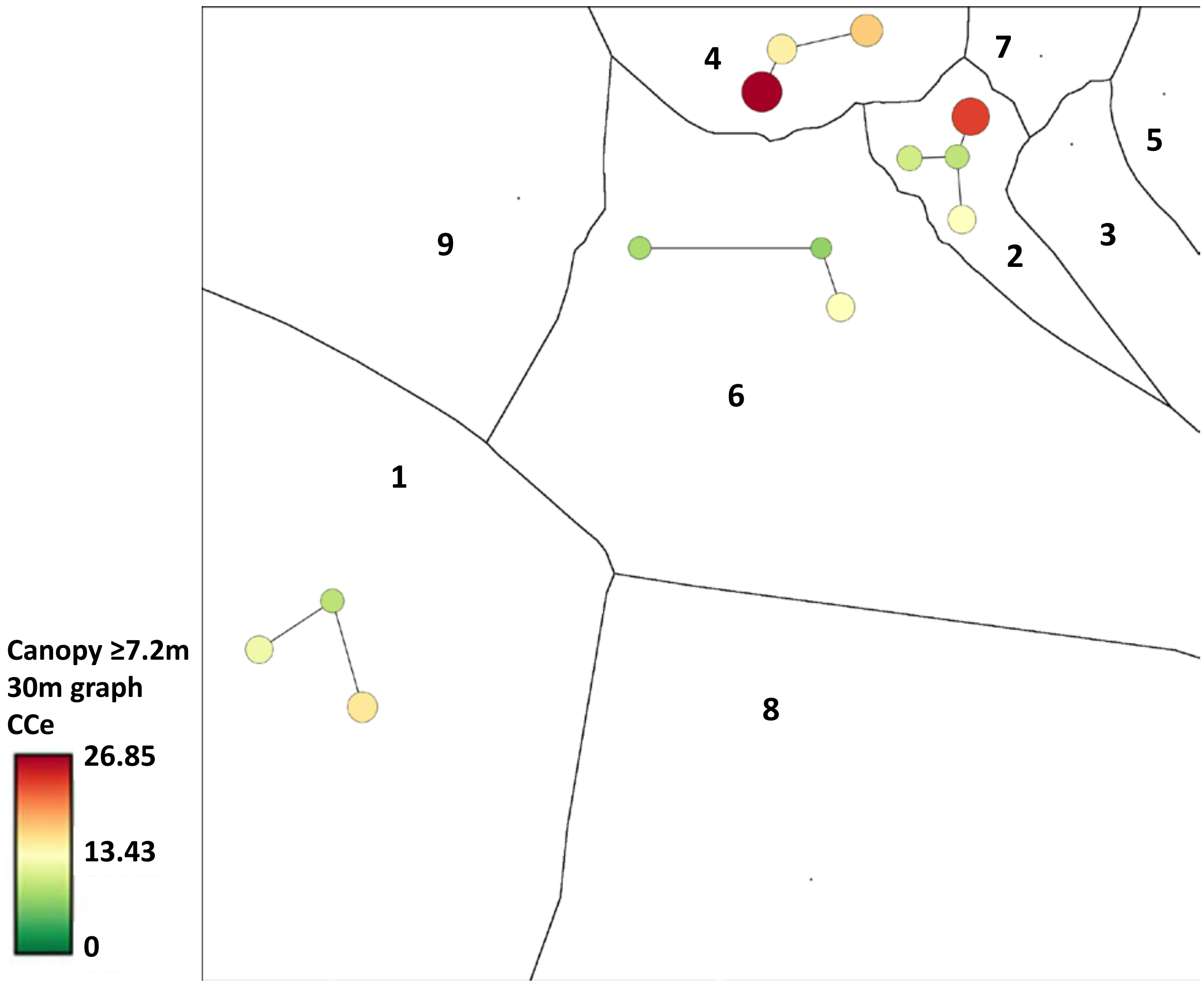

Figure 5-10: Closeness Centrality results of the canopy ≥7.2m node importance graph made using the 30m gap-crossing capability threshold.
The sub-network's graph tree in component one has CCe values ranging from 9.27 (centre node) to 15.37 (bottom right node). In component 2 the sub-network's graph tree exhibits CCe values of 9.47 (centre node) to 23.47 (top node). The centre node in the graph tree within component 4 has a CCe value of 14.68 which increases to 26.85 for the bottom node. The last graph tree of the network is in component 6 and its CCe values range from 7.3 (centre node) to 13.49 (bottom node). The single hub nodes in components 3, 5, 7, 8, and 9 have CCe values of 0.

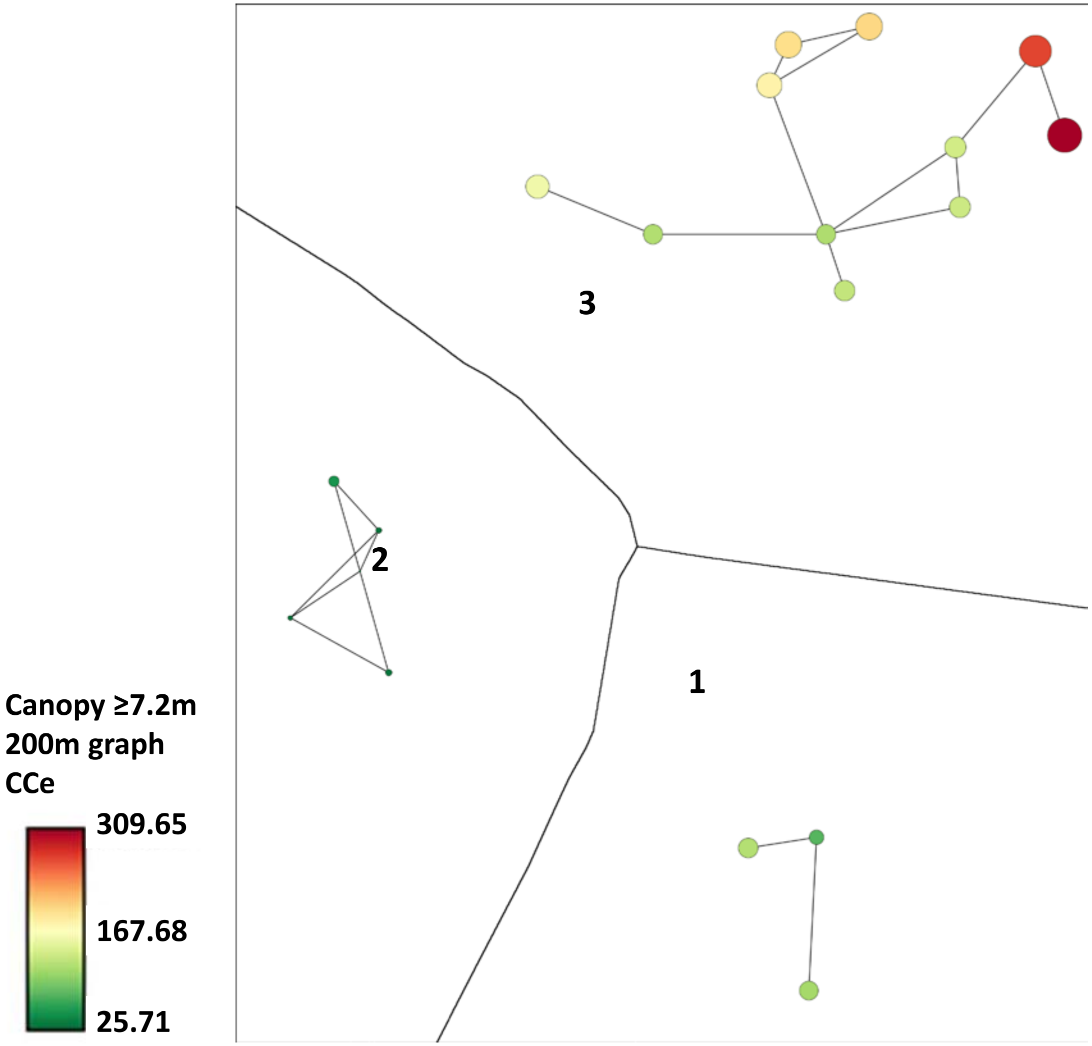

Figure 5-11: Closeness Centrality results of the canopy ≥7.2m node importance graph made using the 200m gap-crossing capability threshold.
Each of the three components in the canopy ≥7.2m, 200m graph contains a graph tree. In component 1 the lowest CCe value was calculated for the top-right node (76.68) while the highest value was calculated for the bottom node (111.12). Component 2's sub-network graph tree contains the hub node with the lowest CCe value of 25.71 (centre node) which increases to 52.19 (top node). The largest graph tree in component 3 exhibits CCe values that range from 116.01 (centre node) to 309.65 (far-right node).

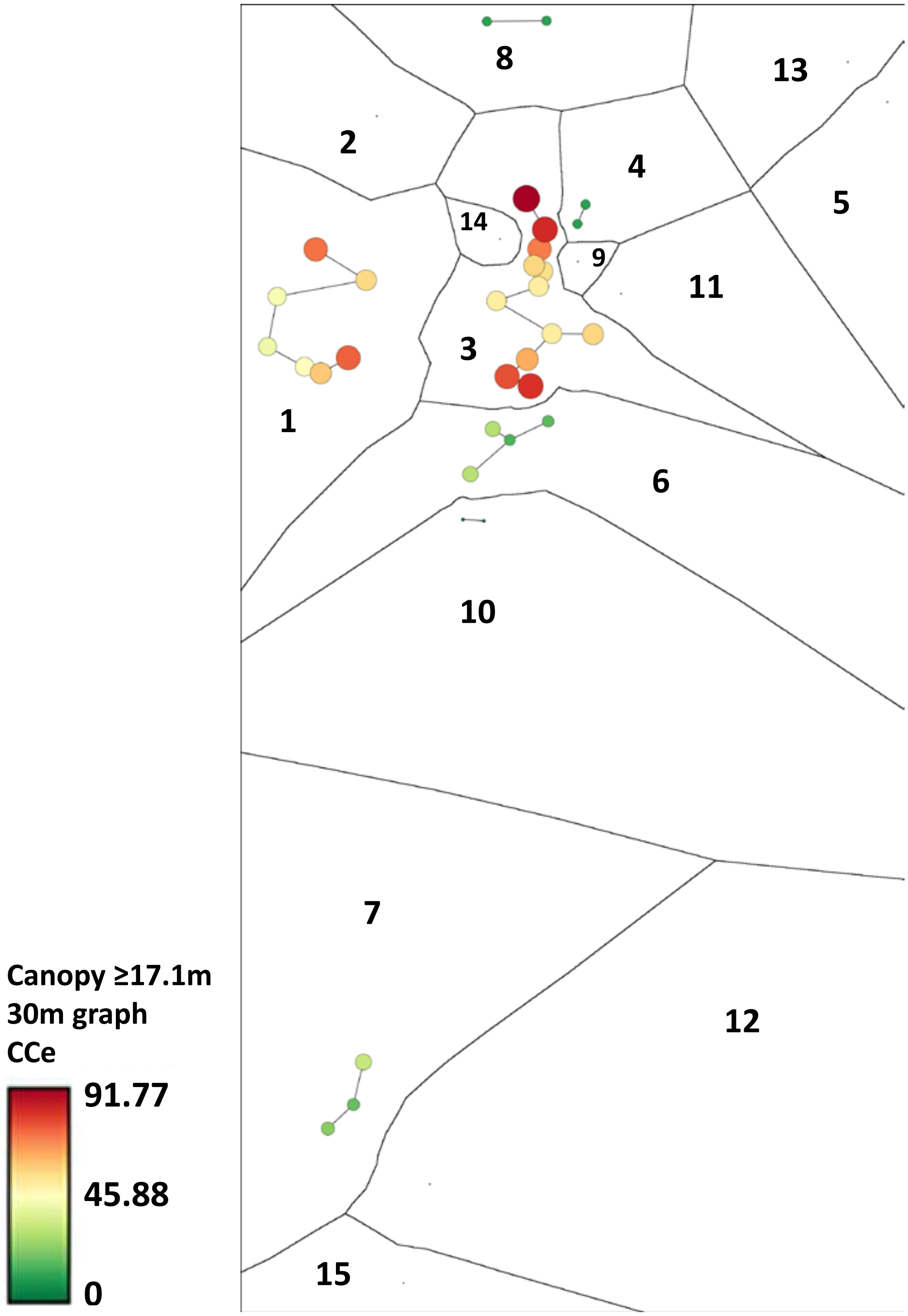

Figure 5-12: Closeness Centrality results of the Canopy ≥17.1m node importance graph created using the 30m gap-crossing capability threshold.

The graph tree in component 1 exhibit CCe values ranging from 41.96 (centre-left) to 75.19 (bottom-right). The maximum CCe value was calculated for the very top node (91.77) in the graph tree within component 3; the minimum value for this tree was calculated for the centre-left and centre-right nodes (50.89). Component 4's two node graph tree exhibited equal CCe values of 11.4. Component 6's graph tree's CCe values range from 15.7 (centre node) to 30.78 (bottom node) while component 7's CCe values range from 19 (centre node) to 33.5 (top node). Component 10 revealed equal CCe values of 1 for the two hub nodes in the graph tree. Components 2, 5, 9, 11, 12, 13, 14, and 15 all contained solitary hub nodes and therefore revealed the lowest CCe value – 0.

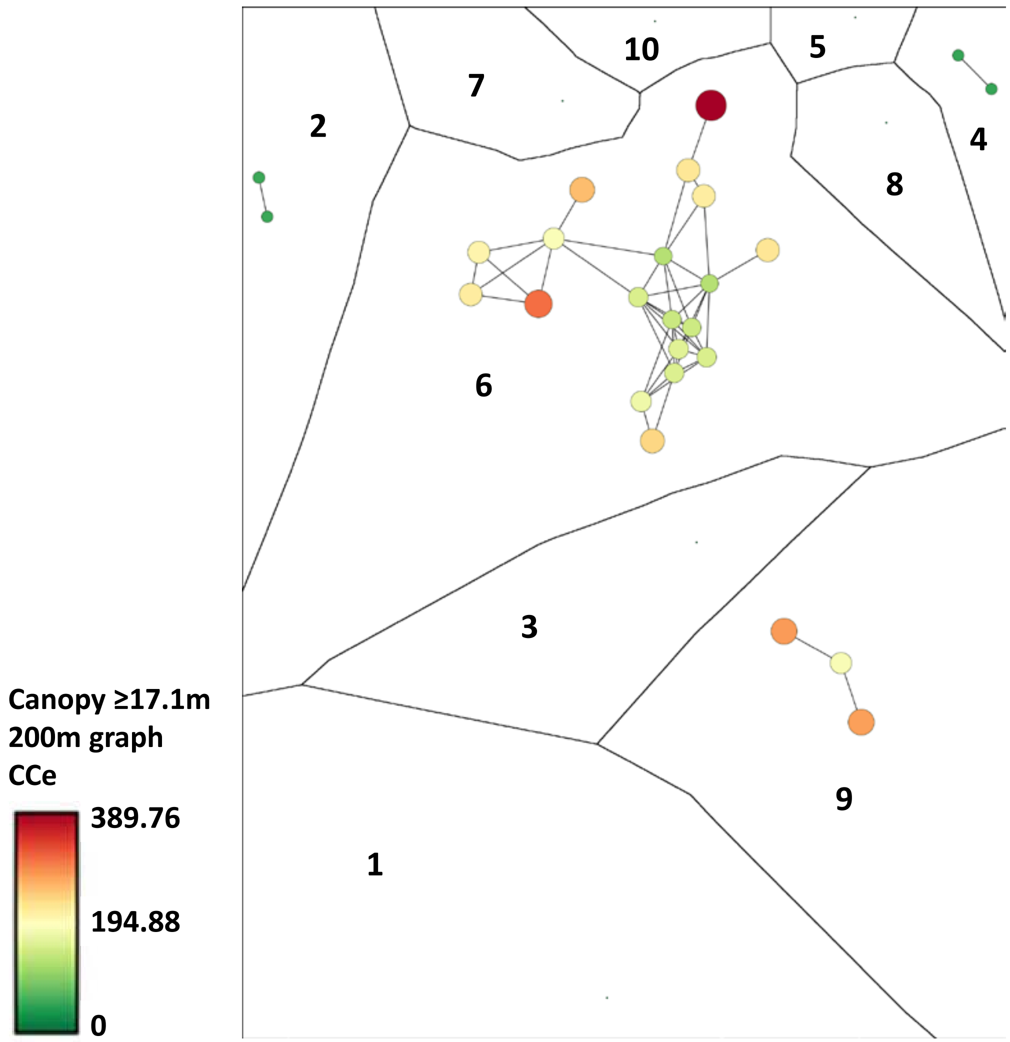

Figure 5-13: Closeness Centrality results of the Canopy ≥17.1m node importance graph created using the 200m gap-crossing capability threshold. Component 2 and component 4 contained graph trees composed of only two hub nodes and therefore the CCe values were the same for each node (component 2 CCe = 56.08, component 4 CCe = 53.54). The largest graph tree (for all NBH canopy graph) is in component 6, revealing CCe values ranging from 129.35/129.46 (centre-top/centre-right nodes) to 389.76 (top node). In component 9 the graph tree's CCe values range from 188.35 (centre node) to 283.88 (top node). Components 1, 3, 5, 7, 8, and 10 contain single hub nodes and therefore the CCe value of 0.

When comparing the CCe results between the NBH node importance graphs, it can be seen that the minimum level of centrality (i.e. least influence in maintaining connectivity depicted by max CCe score) also changes. The canopy ≥3m node importance graph contained the largest CCe results, with the canopy ≥3m 30 graph having a maximum CCe value of 1317.5 (Figure 5-8); decreasing to 1209.2 within the canopy ≥3m 200m graph (Figure 5-9). The canopy ≥7.2m and canopy ≥17.1m node importance graphs exhibited smaller CCe results and were closer to each other than they were to the canopy ≥3m node importance graphs. The canopy ≥7.2m node importance graphs presented a maximum CCe value of 26.8 for the 30m graph (Figure 5-10) and 309.6 for the 200m graph (Figure 5-11). The canopy ≥17.1m node importance graph's CCe values were slightly higher with a maximum of 91.7 for the 30m graph (Figure 5-12) and 389.7 for the 200m graph (Figure 5-13). Therefore, CCe increased with gap-crossing capability threshold within the canopy ≥7.2m and canopy ≥17.1m node importance graph while the canopy ≥3m node importance graphs revealed the opposite relationship.

5.3.3 Integration of habitat maps and landscape graphs

For Figure 5-14, Figure 5-15, and Figure 5-16 the NBH node importance graphs were layered over the river valley study area UTI habitat maps. In Figure 5-14 the most important tree canopies for maintaining UTI landscape connectivity (i.e. hub nodes) lay within the northern, southern, and western parts of the river valley study area. The larger tree patches to the north correspond to Broughton Cliff Nature Reserve and Kersal Vale (Figure 5-14). The other, smaller tree patches to the north are embodiments of Broughton Park, Clowes Park, as well as public street trees and private garden trees within the suburb of Broughton (Figure 5-14). The large tree patches to the south of the map represent Salford University's Peel Campus, Peel Park, and David Lewis Sports Ground. To the west lay canopy patches surrounding and within (in the form of an avenue of trees) Bolton Road/Duchy Playing fields as well as surrounding railway lines, brownfield sites, and industrial units (Figure 5-14). The northern component contained the largest number of important tree patches and as willingness to cross gaps increased the northern and western components were bridged together by the occurrence of a new hub node and therefore a new component (Figure 5-14, B). This tree patch was located next to the River Irwell, upon a flood plain which also forms part of Salford Sports Village. There was no change in network size between the two gap-crossing capability threshold graphs, both of which covered most of the river valley study area (Figure 5-14, A and B).

The integration of canopy ≥7.2m node importance graphs and canopy ≥7.2m UTI habitat maps demonstrate similar patterns of those discussed above (Figure 5-15). Particularly, important tree canopies for maintaining canopy ≥7.2m connectivity also exist within the northern, southern, and western parts of the river valley study area (Figure 5-15). However, the trees in Peel Park were not included within the overall importance network until gap-crossing increased to 200m (the south of Figure 5-15, B). The trees present in the network in the canopy ≥7.2m 30m form part of the David Lewis Sport Grounds (Figure 5-15, A). In contrast, various street and garden trees to the north-east of the Figure 5-15, A, were removed from the network as gap-crossing capability threshold increased to 200m (Figure 5-15, B). Compared to the canopy ≥3m node importance graphs, (Figure 5-14) the northern part of the canopy ≥7.23 30m graph was more compartmentalised (Figure 5-15, A). This compartmentalisation dissipates as willingness to cross gaps increased; consequently the landscape graph became comprised of just 3 components (Figure 5-15, B). These three components shared similar borders to the 3 wards which make up the study area, Broughton to the north and north east, Irwell Riverside to the south, and Kersal to the west (see Figure 3-4). Both the canopy ≥7.2m node importance graphs covered slightly less of the river valley study area then the canopy ≥3m node importance graphs (Figure 5-14 compared to Figure 5-15).

The canopy ≥17.1m 30m node importance graph was limited to the eastern side of the river valley survey area (Figure 5-16, A), increasing its coverage to the west as willingness to cross gaps increased to 200m (Figure 5-16, B). The canopy ≥17.1m important tree patches were localised within the north and south of the river valley study area (Figure 5-15). The trees to the north shape the canopy ≥17.1m of Broughton Cliff Nature Reserve, Kersal Vale, and various street and garden trees (Figure 5-16). The southern trees form part of David Lewis Sports Ground and Peel Park (Figure 5-16). As gap-crossing capability threshold increased to 200m, three important tree patches were identified to the south east of the study area (Figure 5-16, B). These tree patches represent urban street trees and they were not identified in the other two NBH node importance graphs.

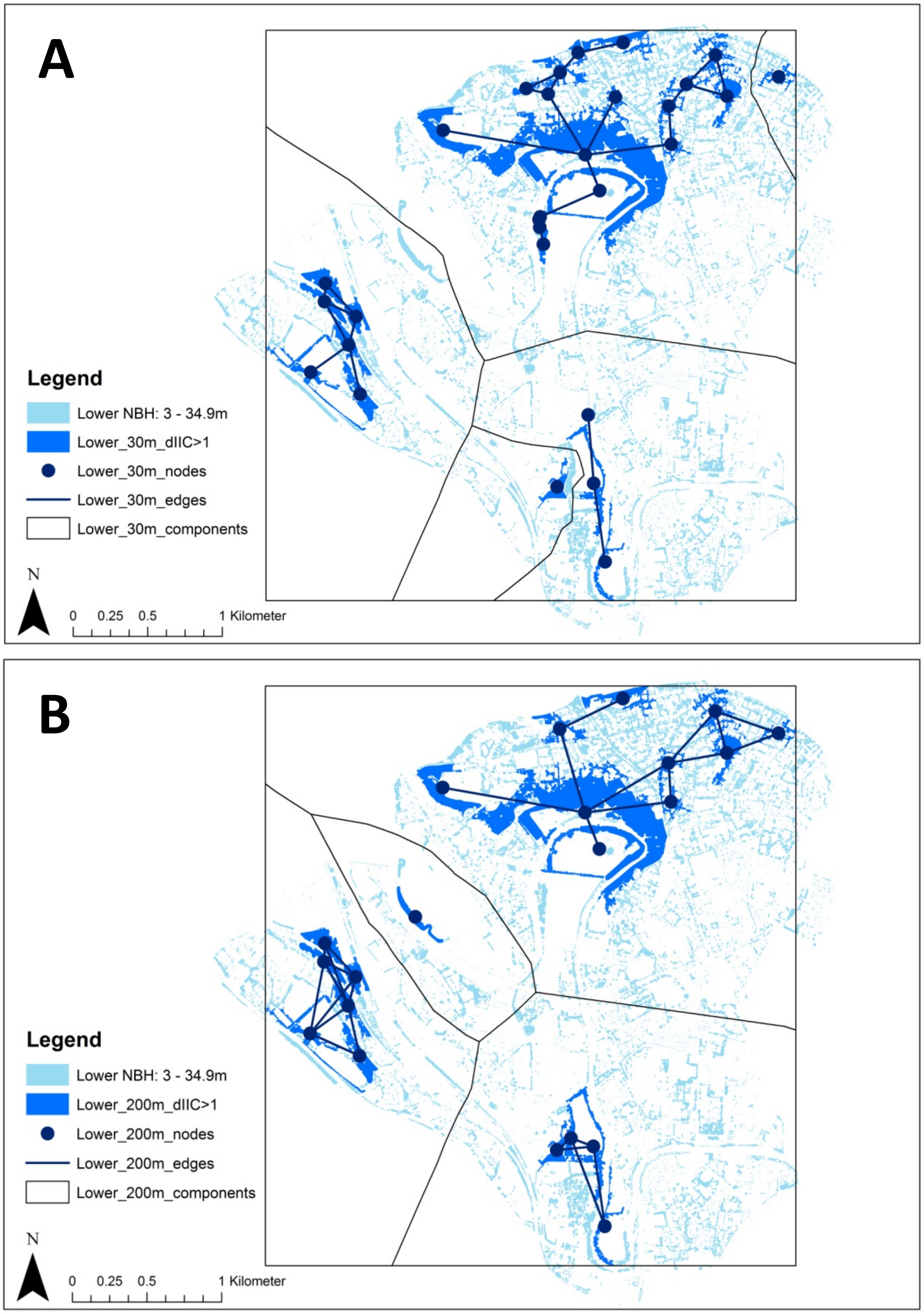

Figure 5-14: Integration of the canopy ≥3m habitat maps and landscape graphs.

For both gap-crossing capability thresholds the important canopy patches exist to the north, south, and west of the river valley study area. However, the 200m node importance graph reveals the location of another important canopy patch in Lower Kersal.

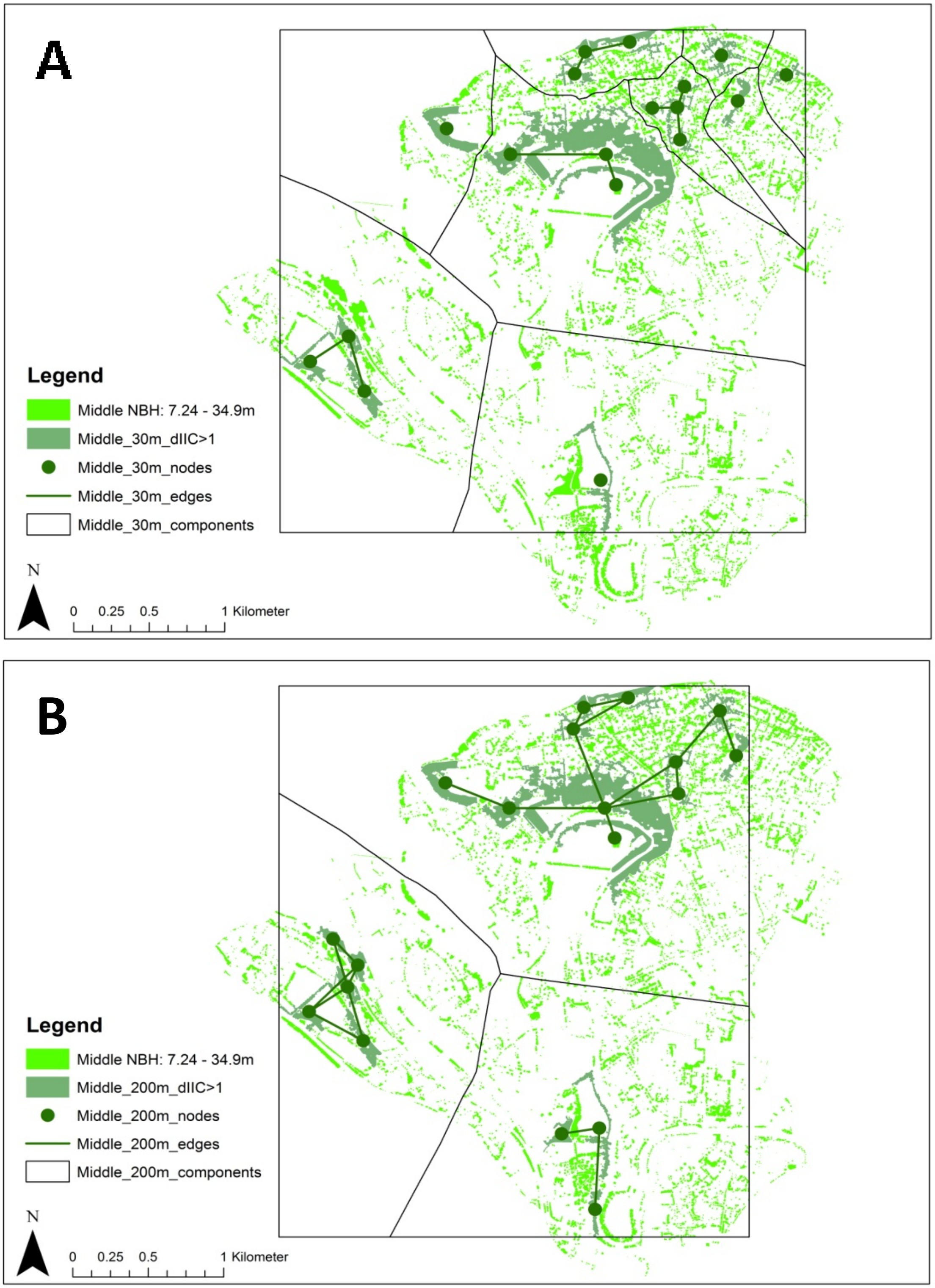

Figure 5-15: Integration of the canopy ≥7.2m habitat maps and landscape graphs.

At the 30m gap-crossing capability threshold, the importance graph network is, relatively, highly compartmentalised – especially to the north of the river valley study area. As gap-crossing increases to 200m compartmentalisation decreases reveal only 3 components.

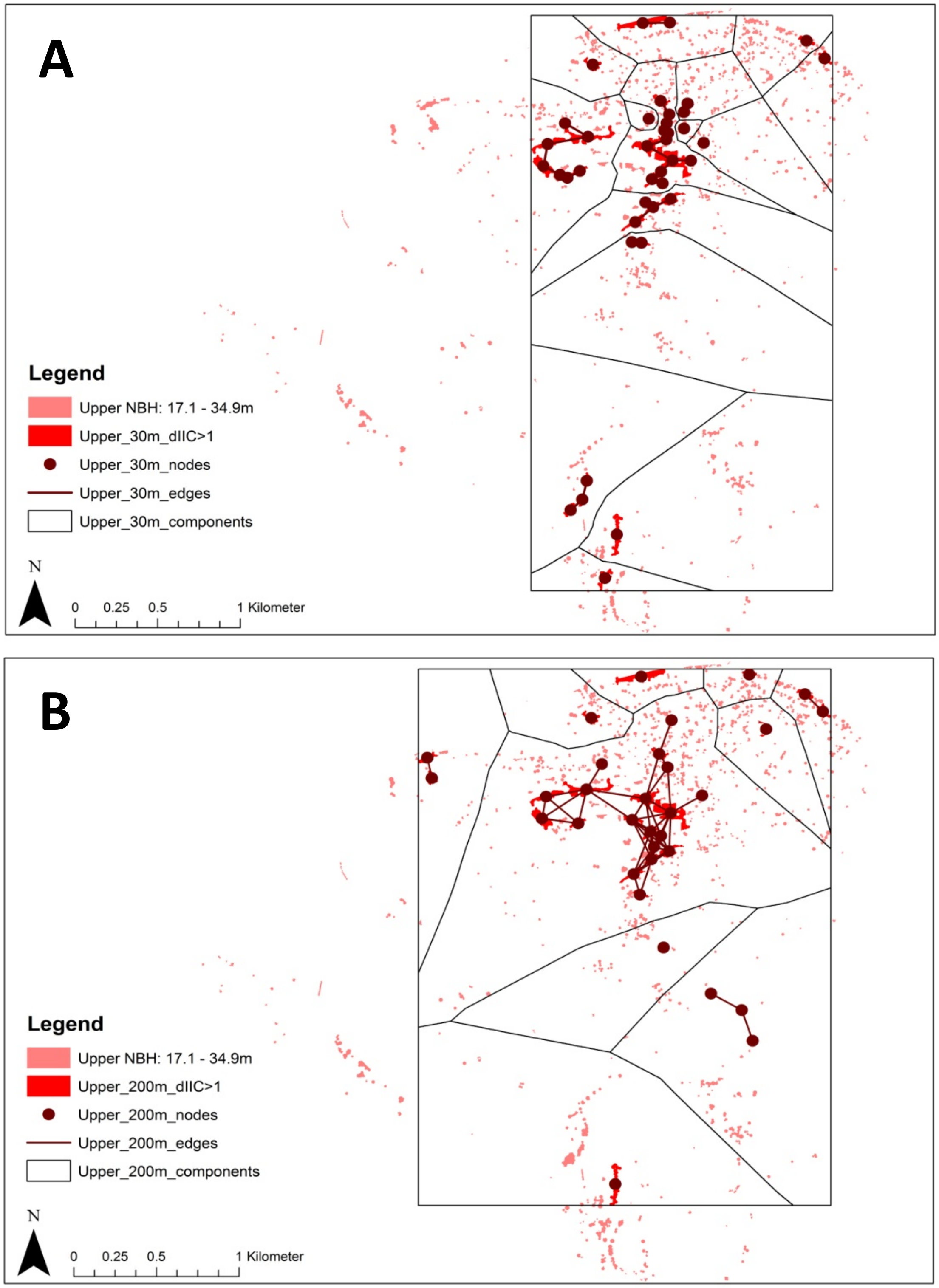

Figure 5-16: Integration of the canopy ≥17.1m habitat maps and landscape graphs.

Both importance graph networks are highly compartmentalised. At the 30m gap-crossing threshold, the importance network is restricted to the west of the river valley study area. The network increases westerly as gap-crossing capability threshold increases.

5.4 Discussion

A comparable landscape connectivity analysis and network topology assessment was applied upon Salford's river valley study area's UTI. Important canopy patches have been identified (keystone canopy patches) as have areas important for maintaining connectivity (landscape hubs) (section 5.3.1). The relationship between the important nodes comprising the sub-networks within these landscape hubs has also been uncovered (section 5.3.2). The location of the landscape hubs, sub-networks, and hub nodes have been identified by combining landscape graphs with habitat maps (section 5.3.3). The general structural patterns emerging from the network analysis, related to system resilience, are discussed below. A more detailed discussion of the UTI's stratified tree canopies is then presented in sections 5.4.2, 5.4.3, and 5.4.4.

5.4.1 Overview

The advantage of using a network approach to assess the resilience of a system is that important structural properties can be revealed (Janssen *et al.*, 2006). The structure or topology of a network is an emergent property that affects the qualities of a system such as the movement of organisms, energy, information, and disease, vulnerability to disturbance, and stability (Melián & Bascompte, 2002; Gastner & Newman, 2006). The structure of a network can be effectively described by incorporating the variables of landscape connectivity and network centrality. However, the results above highlight that network topology can change when the components of landscape graphs, used to describe the network components (i.e. nodes and links), are described through the lens of varying structural scales and by incorporating differing gap-crossing capability thresholds.

The importance for maintaining connectivity (dIIC) analysis revealed that the river valley study area's UTI contained only a few keystone, central patches (Figure 5-2, Figure 5-3, Figure 5-4, Figure 5-5, Figure 5-6, Figure 5-7). These patches represent the core structure of the UTI; they are the largest and or most connected canopy patches. All three of the NBH canopies revealed low central patch density (the proportion of hub nodes within the landscape), fluctuating in accordance with canopy height and gap-crossing capability threshold. In respect to network topology, the NBH canopies represent scale free networks; characterised by the presence of a few hub nodes surrounded by a landscape made up of less important, often described as expendable, nodes (Strogatz, 2001; Minor & Urban, 2008). The potential connectivity of the UTI landscape depends on the central patches established in section 5.3.1. In scale-free networks, landscape connectivity would show little change if most of the smaller/unconnected low dIIC patches were removed yet would rapidly diminish if these central patches were somehow lost (i.e. through urban development or tree diseases) (Minor & Urban, 2008). Such networks are therefore resilient to random impacts on the system yet vulnerable to targeted attacks to the central, most important tree patches (Albert *et al.*, 2000; Strogatz, 2001; Barabási & Bonabeau, 2003). However, it must be stated that the results in section 5.3 should not be used to conclude that all other trees within the river valley study area's UTI are expendable.

Central patch density in the river valley study area is extremely low (0.2% to around 5%) meaning if most of the below 1% dIIC tree patches were to be removed, the resulting UTI would be extremely restricted and potentially insignificant in regards to providing functional habitat. What remained would be localised to the north, south, and west of the river valley study area except the canopy ≥17.1m which would only exist to the north and to the south. These smaller, less connected trees may not maintain overall landscape connectivity or provide primary resources, yet they may be important providers of supplementary resources (Dunning *et al.* 1992 and Taylor *et al.* 1993). Furthermore, a loss in the bulk of canopy patches will cause the UTI system to become even less resilient since any disturbance or attack could only be targeted towards the central, most important tree patches. Moreover, the UTI is a social-ecological system – a construct of ecological and cultural selective forces and provider of an array of ecosystem services (section 2.1 and 2.2). The smaller canopy patches and individual trees may have little influence on overall landscape connectivity, yet they may act as *cultural* ecosystem service providers (aesthetic quality, sense of place, representative of nature etc. Parsons *et al.*, 1998 (O'Brien, 2005) Dandy *et al.* (2011)). In general, it has been found that urban residents feel extremely positively towards trees in cities (Lohr *et al.,* 2004) and the loss of the more 'structurally unimportant' trees can therefore affect quality of urban living and be viewed as a negative environmental change by local residents. Therefore, the results in this chapter should not be considered in isolation from the other issues surrounding Salford's UTI. However, the aim addressed in chapter 5 is to identify and describe important tree patches and structural properties within the UTI system and relate them to ecological resilience. Therefore, it is these relationships that will be discussed here.

In order to acquire an understanding of the relationships between the central canopy patches that create the river valley study area's sub-networks, a structural network analyses was undertaken (section 5.3.2). In addition, the resilience of the important canopy patch network was assessed by calculating the levels of centrality (CCe) within areas of localised connectivity (landscape hubs) (Figure 5-8, Figure 5-9, Figure 5-10, Figure 5-11, Figure 5-12, Figure 5-13). Therefore, if a targeted attack upon the sub-networks comprising important canopy patches did occur, the findings of section 5.3.2 could inform landscape practitioners on which canopy patches are fundamental in the maintaining both local and overall connectivity and system resilience. There are two important factors affecting the connectivity and resilience of the river valley study area's sub-networks – the level of compartmentalisation and centrality (section 2.4).

A compartmentalised network consists of highly important hub nodes which are not directly connected to other highly important hub nodes (Melián & Bascompte, 2002; Minor & Urban, 2008). The components that make up these compartmentalised networks can be regarded as landscape hubs – areas of high, localised connectivity within a highly fragmented landscape (Minor & Urban, 2008). Therefore compartmentalisation can also be correlated to connectivity (Melián & Bascompte, 2002) or, more specifically, the connectivity of structurally important hub nodes. Compartmentalised networks exhibit landscape hubs containing sub-networks – areas of highly connected nodes, often organised around a central node (Maslov & Sneppen, 2002). In regards to the river valley study area, the keystone canopy patches are not all connected to each other but rather they form sub-networks existing within several landscape hubs (section 5.3.2). However, it must be stated that even though these sub-networks and keystone canopy patches are not directly connected to each other (as they are compartmentalised), there is still a network of ‘expendable’ tree patches for organisms to move through to access them. However, if these sub-networks and hub nodes that produce them were lost, the entire system would become ever more unconnected and the functioning of the UTI would be greatly affected.

A compartmentalized network can be seen in each of the NBH node importance graphs. As landscape hubs exist locally (almost following the ward boundaries for the canopies ≥3m and ≥7.2m) a heterogeneous landscape exists. The UTI networks are therefore not homogenous but not entirely fragmented; instead there are focal areas of high connectivity which are critical to maintain overall connectivity. This compartmentalised pattern of important canopy patches may increase the overall resilience of the UTI by isolating any damaging disturbances (Melián & Bascompte, 2002). In other words, the important canopy patches of the river valley study area’s UTI reveal a pattern that is related to the resistance of system collapse. This is because, if, for some deleterious reason, a number of the tree patches were removed, or potential outbreak of disease, within one landscape hub, the other landscape hubs would not be affected or the disease is less likely to spread, as they are unconnected (Melián & Bascompte, 2002). Therefore, the overall network will not be as affected as it would be if all of the important tree patches were connected to each other. However, it must be kept in mind that the proportion of important tree patches within the study area’s UTI is low to begin with, so any removal of the identified hub nodes would reduce overall system resilience – the capacity of the system to function as normal. The sub-networks are also important for maintaining local connectivity. The dIIC results were used to identify the important canopy patches and therefore, through network topology analyses, they reveal the level of compartmentalisation within the NBH node importance graphs. Further network topology analysis was used to identify the landscape hubs and the sub-networks that maintain connectivity within these landscape hubs. If a disturbance did happen within these sub-networks which tree patches are relatively expendable and which need to be prioritised in any management decision?

The closeness centrality metric (CCe) was used to identify which canopy patches are holding the landscape hub's sub-networks together (i.e. critical for maintaining sub-network function and local connectivity). When landscape hubs are made up of just one tree patch (a single hub node) then centrality is zero. Furthermore, these landscape hubs will not be resilient to any targeted attack as no sub-network exists, just a single tree patch. The CCe results are therefore only relatable to sub-networks within landscape hubs. The higher the CCe score, the least important that tree patch is for maintaining the structure of the sub-network. Tree patches assigned with a low CCe score are regarded as the central device of the sub-network, to which the rest of the sub-network is connected. The removal of such a canopy patch would mean that the sub-network would collapse, the landscape hub would become fragmented, and connectivity and function would decrease. Therefore, the hub nodes within the NBH node importance graphs with a low CCe score (green nodes) are key to the resilience of the sub-network system, and therefore in turn, the system as a whole. Within the canopy ≥3m and canopy ≥7.2m node importance graphs there are sub-networks which are close to forming cycles – termed here as 'pseudo-cycles'. These can be regarded as the most resilient form of sub-networks as centrality is equal among nodes. That is to say, one characteristic of a resilient system is the ability to replace a node or link which is removed from the system (Walker *et al.*, 1999). If one of the tree patches within a pseudo-cycle were to be removed from the sub-network then there are still other nodes with connected links to the rest of the sub-network. However, any random attack can only be targeted towards a functionally important node. On the other hand, the sub-networks which contain variations in centrality will be resilient to random attacks (such as those in the canopy ≥17.1m node importance graphs, (Figure 5-12, Figure 5-13). Yet the structural importance of a central canopy patch will not be replaced, or filled by any of the other, less central patches. Instead, as stated the sub-network would collapse – thus such sub-networks exhibit low levels of resilience.

All of the important structural patterns, in regards to maintaining connectivity and system resilience, have been identified within the vertically stratified canopies of the river valley study area's UTI. However, as the inputs of canopy height and passerine gap-crossing capability threshold changed, so did the emerging patterns. Therefore, there is no single network pattern that emerges from the analysis. The general patterns described above can be applied to each of the NBH canopies, yet there are also NBH canopy specific structural patterns which need to be assessed. These emerging patterns would not have been identified if a two dimensional, single distance threshold approach was applied to the study area. Once the canopy specific structure and resilience is discussed, a network comparison and the implications of the comparison on the perception of system resilience, is presented in section 5.4.5.

5.4.2 The structure and resilience of the canopy ≥3m trees

At the canopy ≥3m level, the scale free network is highly resilient to random attacks, as most of the landscape is made up of ‘expendable’ canopy patches (Figure 5-14). However, the total amount of keystone canopy patches (important for maintaining connectivity) is relatively low – therefore any attack will be detrimental for the system (Figure 5-2, Figure 5-3). As gap-crossing capability threshold increases to 200m two of the sub-networks become more connected and thus landscape-hubs increase in size (Figure 5-2, Figure 5-3). Furthermore, a keystone canopy patch is revealed in the 200m graph (Figure 5-3). As a result, a change in perceptual threshold increases the observable resilience of the canopy ≥3m network.

At the 30m gap-crossing capability threshold only two landscape hubs (areas of high connectivity) do not contain sub-networks (graphs within the landscape hub), decreasing to one as gap-crossing capability threshold increases to 200m (Figure 5-8, Figure 5-9). As perceptual threshold increases the level of centrality also decreases, as sub-networks either transform into pseudo-cycles or reduce their number of distant ‘satellite’ canopy patches (Figure 5-9). The largest sub-network (component 1, Figure 5-8) is fully dependent on the central node, which if removed would cause the sub-network to collapse. This means the landscape-hub may be resilient to attacks on the ‘satellite’ canopy patches because if they were removed or damaged the ‘core’ of the sub-network, which maintains landscape-hub connectivity, is not affected. However, the current structure and connectivity of the sub-network displays a pattern relatable to low resilience, given that if the central canopy patch were removed the landscape-hub would potentially break into 4 smaller, less connected hubs. As gap-crossing capability threshold increases to 200m, the importance of the central patch to maintain local levels of connectivity is lessened, as its ‘responsibility’ is shared with the other surrounding canopy patches. As such resilience is increased.

The largest landscape-hub is located to the north of the river valley study area in Kersal (Figure 5-14). The central canopy patch for the sub-network existing in the ‘Kersal landscape-hub’ is Broughton Cliff Nature Reserve (level of centrality decreases as gap-crossing increases) (Figure 5-14). As gap-crossing capability threshold increases to 200m, two landscape-hubs to the south of the river valley study area transform into a single hub containing a pseudo-cycle made up of canopy patches from Salford University’s Peel Campus, Peel Park, and David Lewis Sports Ground (Figure 5-14, B). Similarly, as gap-crossing increases to 200m the sub-network within the landscape hub to the west also becomes a pseudo-cycle composed of canopy patches existing around and within Bolton Road Playing Fields (also known as Duchy Playing Fields), railway lines, brownfield sites, and industrial units. Furthermore, a keystone canopy patch is also identified within Lower Kersal’s floodplain (Figure 5-14, B). This increase in connectivity and structural resilience, in line with gap-crossing capability threshold, is exhibited within all of the river valley study area’s wards; Kersal (north), Broughton (north west and south west), and Irwell Riverside (east and south). However, the centre of the Broughton ward has a marked absence of sub-networks or even single keystone canopy patches.

5.4.3 The structure and resilience of the canopy ≥7.2m trees

The highly compartmentalised network of the river valley study area markedly decreases as the total number of components, or rather landscape-hubs, reduce from 9 to 3 as gap-crossing capability threshold increases to 200m (Figure 5-4 and Figure 5-5). The levels of local connectivity increase as all landscape-hubs are composed of sub-networks; hence network resilience increases as gap-crossing capability threshold increases.

Levels of centrality are low when considering the 30m gap-crossing capability threshold, mainly due to the fact that the landscape-hubs contain more single keystone canopy patches than sub-networks (Figure 5-10). The important canopy patch network is therefore not very resilient when described using the 30m gap-crossing capability threshold and local connectivity is dependent on only a single canopy patch. When a sub-network occurs it is mainly made from three keystone canopy patches (3 out of 4 sub-networks) however the levels of centrality within these sub-networks are relatively equal (except for the sub-network in component 2) (Figure 5-10). Although, due to these sub-networks being composed of 3 keystone canopy patches, the removal of the central patch would always result in the breaking apart of the sub-network and therefore landscape hub. Therefore, the resilience of the sub-networks for maintaining local connectivity is reliant on only a single canopy patch (a total of 4 for the entire network). This resilience increases in line with gap-crossing capability threshold as the three landscape hubs are all composed of sub-networks (Figure 5-11). The most connected sub-network is in component 2, thus underpinning a highly resilient landscape-hub; if any of the canopy patches were removed the sub-network would decrease in size but not break apart. The sub-network is component 1 is dependent on the central node as is the sub-network in component 3. However, the sub-network in component 3, underpinning the landscape hub, is more resilient than the one in component 1 since if the central canopy patch in component 3 were to be removed, the sub-network would break apart and create two new sub-networks, a single keystone canopy patch, and two connected keystone patches. On the other hand, if the central canopy patch were removed from component 1, there will be no more sub-networks, just 2 unconnected, solitary keystone patches (Figure 5-11).

Similar to the canopy ≥3m, the sub-networks and keystone canopy patches of the canopy ≥7.2m exist to the north in Kersal, the south in Lower Broughton, and the west in Pendlebury. At the 30m gap-crossing capability threshold the keystone canopy patches in the west exist in and around Bolton Road Playing fields (Figure 5-15, A) and the canopy patches around the more industrial land-cover/use are not included in the sub-network until gap-crossing capability threshold increases to 200m (Figure 5-15, B). Similarly, the central, keystone canopy patch in Kersal (Broughton Cliff Nature Reserve) is not connected to the street and garden trees to the north, Broughton Park to the east, or Kersal Vale to the west (Figure 5-15, A) until gap-crossing capability threshold increases to 200m (Figure 5-15, B). To the south of the river valley study area, only the canopy patches of David Lewis Sports Ground are recognised as essential for maintaining connectivity (Figure 5-15, A). Local connectivity and UTI resilience increases with gap-crossing capability threshold as the canopies in Peel Park and Salford University's Peel

Campus become included in the sub-network (Figure 5-15, B). The 'flood plain' patch is not recognised as important for the connectivity of the canopy ≥7.2m, and therefore it is only influential to connectivity when smaller trees (i.e. 3-7m) are included in the landscape connectivity model.

5.4.4 The structure and resilience of the canopy ≥17.1m trees

The canopy ≥17.1m contains a higher amount of hub density (nearly 5% of nodes are hub nodes) and therefore is more vulnerable to random attacks. Furthermore, landscape hubs within the canopy ≥17.1m are often made up of only single hub nodes, thus these are the central nodes for maintaining connectivity (Figure 5-6, Figure 5-7). Therefore, resilience is low. The landscape hubs that contain sub-networks either have high centrality values, as in the 30m graph (i.e. no single hub node is key to maintaining connectivity; there is no central node. Figure 5-6), or centrality (i.e. importance) is concentrated on only a few nodes, as in the 200m graph (Figure 5-7). Where centrality is low (i.e. high CCe values with low amounts of variability), a loss in a hub node may not have such a great impact on the maintenance of connectivity, as the hub nodes are relatively independent of each other (e.g. components 3 and 1 in Figure 5-12). Although the sub-network would break apart, it would not be a deleterious as if, for example, the sub-network was a well-connected cycle (e.g. like the centre of the sub-network in component 6 in Figure 5-13). If the highly central hub nodes within the high centrality sub-networks (i.e. high variability in CCe) were to be removed, then the whole component would become fragmented, breaking apart into more sub-graphs, thus connectivity is reduced and the system loses its structure and function. Therefore, the low centrality sub-networks may represent more resilience as function would not change drastically if a change occurred, however the sub-networks with high centrality are open to targeted attacks to the central node. On the other hand, the sub-network in component 6 of Figure 5-13 is more resilient to random attacks as there are a number of 'satellite' canopy patches which, if removed, have little overall effect on the structure of the sub-network and therefore local connectivity. As for the other NBH canopies, an increase in gap-crossing capability threshold increases overall resilience as sub-networks become larger, more connected, and less dependent on a central canopy patch. However, centrality is never equal in any sub-network as no cycles exist.

The level of compartmentalisation and centrality is higher, while connectivity is lower within the canopy ≥17.1m graph compared to the other NBH graphs. This is due to the increased number of landscape-hubs within the landscape and the larger proportion of the landscape comprised of hub nodes. This high compartmentalisation means that the canopy ≥17.1m canopy of the UTI contains a high number of locally connected areas containing functionally important groups or single nodes (i.e. landscape hubs). These landscape hubs exist predominately in the eastern side of Salford, to the north and south (Figure 5-16). At the 30m gap-crossing capability threshold, the highly important Broughton Cliff Nature Reserve is disconnected, becoming highly compartmentalised (Figure 5-16, A). However, previously unidentified keystone canopy patches are also described (Figure 5-16, A). The canopy patches of the Broughton Cliff nature reserve form a sub-network as gap-crossing increases to 200m, along with canopy patches around the river Irwell (Figure 5-16, B). Keystone canopy patches are also identified westerly of Broughton cliff Nature Reserve, in Kersal Vale, as gap-crossing increases to 200m. The landscape hub to the south east of the river valley study area, containing 3 canopy patches, is not identified in any of the NBH canopy graphs and therefore would be overlooked by landscape connectivity analysis that did not consider tree height (Figure 5-16, A). The recognition of this sub-network is also due to the increase in gap-crossing capability threshold, however this increase also sees the removal of key stone patches in the South of the study area (Figure 5-16, A). Therefore, these canopy patches, of the David Lewis sports Ground, are important for passerines with a 30m perceptual threshold but not for a more generalist passerine with a 200m threshold. Therefore, the location of canopy patches ≥17.1m, that are important for maintaining connectivity, is highly dependent on the perceptual threshold of the species of study.

5.4.5 Network comparison

The level of compartmentalisation and the influential relationship within sub-networks fluctuated between the landscape graphs as well as within each graph as gap-crossing capability threshold changed. Thus highlighting that emerging patterns of critical structural variables will change depending on vertical scale and perceptual thresholds. The canopy ≥17.1m may be less connected than the canopy ≥7.2m and canopy ≥3m in regards to the number of components it contains, yet it has more important canopy patches which need prioritisation. The canopy ≥17.1m contains a lot of unconnected keystone patches and if one is lost then the connectivity of the canopy ≥17.1m is greatly reduced and so initially it is seems as though it is less resilient than the other NBH canopies. Furthermore, the canopy ≥17.1m layer is smaller in size than the other NBH canopies, constrained to the eastern side of the river valley study area.

When the term resilience is used to refer to the number of nodes (i.e. canopy patches) that can be removed without altering network connectivity (Minor & Urban, 2008) there seems to be a large amount of expandable nodes and therefore resilience. Therefore, the river valley study area could be considered rather resilient seeing as the critical, keystone patches are at such low density (0.28 – 5% of the overall patches) that a random attack would be unlikely to affect them. However, because they are at such a low density it would be illogical to reach the conclusion that the rest of the 95 to 99.72% of patches are expendable. Rather, the potentially 'expendable' patches should be considered as supporting rather than directly influencing overall landscape connectivity. Networks, such as those representing the river valley study area's UTI, with significant variance in node connectivity are most robust to random removal of nodes (Albert et al., 2000). For the effective governance of ecological networks it is important to maintain connectivity of the overall system by conserving , for example, canopy patches with high centrality/importance (Janssen *et al.*, 2006). Similarly, the UTI network would quickly break apart if these hubs were removed and consequently conservation efforts would be best spent on hub patches (Minor & Urban, 2008). However, identifying these central patches can be difficult as their location differs in line with structural and perceptual change. As the canopy ≥17.1m contains the greatest number of keystone canopy patches it is in need of the most protection and conservation prioritisation. Furthermore, the canopy patches within the canopy ≥17.1m are often unconnected to other keystone canopies (i.e. do not form sub-networks) and therefore if one were to be removed there is no other patch to replace it – thus the canopy can actually be considered as less resilient than the other NBH canopies.

5.5 Conclusion

Due to the many layers of structure, and the feedbacks between structure and function, the task of uncovering the relationship between structure and function is difficult (Pickett et al., 2004). One of the key insights to emerge from contemporary ecology is that spatial heterogeneity can govern the functioning of a system at any scale (Pickett & Cadenasso, 1995). By using a network approach, incorporating varying structural layers (change in dimensional scale), and passerine gap-crossing capability thresholds to understand system resilience, various important structural patterns have been discovered.

A resilient social–ecological system, has the ability to create opportunity from disturbance (Folke, 2006). Due to the cross-scale dynamics of key landscape structures, which are sources of resilience, it has been argued that the resilience of complex systems is not simply about resistance to change or the conservation of existing structures (Folke, 2006). However, there are limitations to such a theoretic approach towards urban systems; especially systems which only contain a few critical areas for maintaining resilience (e.g. the river valleys study area's UTI). Even though it is true that resilient systems should be open to change, if a non-resilient system is identified, by assessing its basic, critical structure, then a resilience approach towards that system should be about conserving existing structures and avoiding deleterious change. These existing structures have been identified within the river valley study area's UTI, described at the habitat patch and landscape scale, and related to system resilience. In summary 1) the proportion of central, keystone canopy patches within the vertically stratified canopies is low and therefore the UTI is mainly composed of 'expendable' patches; 2) overall landscape connectivity and system resilience is dependent on only a few sub-networks within landscape hubs, thus a scale free network occurs; 3) where sub-networks occur they are often dependent on a single central canopy patch which underpin landscape-hub connectivity; 4) system resilience increases as gap-crossing capability threshold increases; 5) where landscape-hubs are composed of only a single keystone canopy patch resilience is low, as there no other canopy patches to fill the functional gap that would occur if it were removed; and 6) the perceived structure of the UTI network is dependent on vertical scale and organism perception when modelled using gap-crossing capability thresholds.

Chapter 6 : Exploring the Relationship between Societal Metrics and Urban Tree Infrastructure (UTI) Landscape Patterns

6.1 Introduction

The UTI is a social as well as an ecological construct (section 2.1 and 2.4). The ability to connect with nature in everyday life can positively influence the quality of urban living (O'Brien, 2005). Urban trees can provide this connection as well as other ecosystem services (e.g. carbon sequestration, climate regulation, air pollution control, and aesthetic quality) (Dobbs *et al.*, 2011; Johnston & Percival, 2011). It is reported that those in the most deprived section of society benefit most from access to green space in general and urban canopy cover in particular (Maas *et al.*, 2006; Cohen *et al.*, 2007). For example, health inequalities related to income deprivation are lower in greener areas (Forestry Commission, 2010). However, residents of higher socio-economic status often use and live within areas richer in green infrastructure than residents of low socio-economic status (Matsuoka & Kaplan, 2008). A relationship thus exists between the level of societal deprivation and the degree of the benefit derived from green infrastructure, such as the UTI (Forestry Commission, 2010). This relationship, however, often reveals itself as a negative representation of environmental justice (Agyeman & Evans, 2004). That is to say, the amount of canopy cover in areas of deprivation is less than in areas of provision (Landry & Chakraborty, 2009; Kendal *et al.*, 2012; Zhou & Kim, 2013). Studies that support this finding focus only on one aspect of UTI landscape structure – composition – and therefore overlook the importance of the other feature of structure – configuration. Furthermore, the landscape composition of the UTI has not been described in three dimensions and related to societal structure.

Landscape structure has been identified as influential in the provision of ecosystem services (section 2.1 and 2.3). Specifically, how habitat patches are structured within the landscape strongly influences animal-habitat associations (MacArthur & MacArthur, 1961; Vierling *et al.*, 2008) which in turn underpins the provision of ecosystem services particularly if they are provided or supported by mobile organisms (Kremen *et al.*, 2007; TEEB, 2010a). One of the most influential aspects of this relationship is spatial heterogeneity (i.e. patchiness) which includes both the characteristics of habitat patches and its variation across the landscape (Wiens, 1976; Rotenberry & Wiens, 1980). Structural heterogeneity also exists in three dimensions and the diversity of canopy heights – an important aspect of structural complexity - influences the overall habitat quality of a given habitat patch (MacArthur & MacArthur, 1961; Rosenzweig, 1995; McElhinny *et al.*, 2005). Research that relates these structural patterns to societal patterns existing across the landscape is missing.

The research described within Chapter 6 was undertaken in an attempt to answer research question 4; 'what is the relationship between the landscape structure of Salford's UTI and Salford's societal structure?' Research objectives 4.1, 4.2, and 4.3 were implemented In order to address research question 4 (Figure 3-1). To accomplish these objectives first the patterns of social deprivation should be mapped followed by a description of Salford's UTI landscape composition and configuration. Finally, the strength of the statistical relationship between the two will be assessed. The methods used in order to achieve these research objectives are described in section 6.2 while the results of these implemented methods are presented in section 6.3. The meaning and significance of these results are explained in section 6.4, followed by the final research conclusions in section 6.5.

6.2 Methods

The aim addressed in Chapter 6 is to assess the relationship between societal and UTI landscape patterns in Salford (Figure 6-1). To achieve this, the UK government's Index of Multiple Deprivation (IMD) was used to map area deprivation levels in Salford (Figure 6-1). Following this, Salford's UTI landscape structure was described, within differing areas of deprivation, in terms of exhibited levels of canopy cover, influence of fragmentation based on canopy cover, canopy height diversity (CHD), number of canopy patches (NP), patch density (PD), and levels of structural contagion (CONTAG) (Figure 6-1). The strength of the relationship between area deprivation structure and UTI landscape structure was then tested using a series of statistical calculations in SPSS (v20) (Figure 6-1).

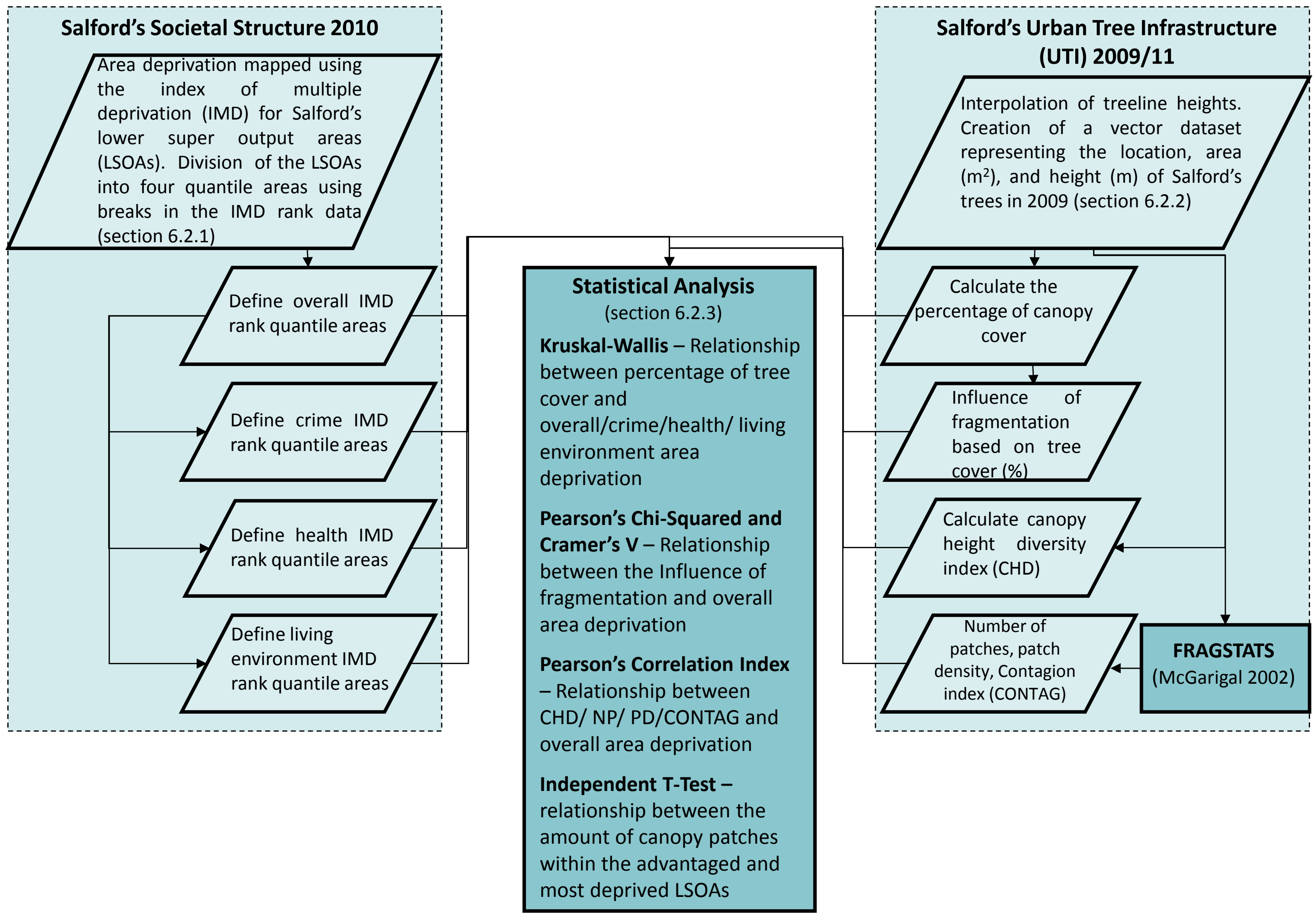

Figure 6-1: A graphical representation of the processes (parallelogram), data (dashed rectangle), and analysis (solid rectangle) used in Chapter 6. A total of two datasets (dashed rectangles) were created using a series of processes (parallelograms) so that a statistical relationship analysis (solid rectangle) could take place. The City of Salford was described and mapped based on the levels of area deprivation present within its lower super output areas (LSOAs – section 6.2.1). The UTI within these LSOAs were also described and mapped based on its landscape composition and configuration. The landscape analysis software FRAGSTATS (v4.2) (McGarigal *et al.*, 2002) was used to described certain aspects of landscape structure. The relationship between UTI and social structure was statistically assessed using the statistical software SPSS (v20).

In section 6.2.1 a description of the IMD ranking system and the methods used to map area deprivation are presented, with a particular focus on overall, crime, health, and living environment IMD rank. The UTI landscape structure analysis methods are described in section 6.2.2. The statistical analysis used in order to elucidate the social-ecological relationship between area deprivation and UTI structure is presented in section 6.2.3.

6.2.1 Describing Societal Landscape Patterns - Index of Multiple Deprivation (IMD)

Area deprivation (at the lower super output area [LSOA] level) was used to map societal patterns at the landscape scale. To discern area deprivation, the UK's Index of Multiple Deprivation (IMD) was used. The IMD is a measure of anthropogenic deprivation at the small area level (McLennan *et al*. 2011). People within these areas can experience several and distinct domains of deprivation (McLennan *et al.*, 2011). The area in which an individual lives is then ranked based on an overall IMD score, calculated as a weighted area level aggregation of these domains of deprivation (McLennan *et al.*, 2011). The seven domains of deprivation, as defined by the UK Government, are (1) income deprivation; (2) employment deprivation; (3) health deprivation and disability; (4) education, skills, and training deprivation; (5) barriers to housing and services; (6) crime; and (7) living environment deprivation (McLennan *et al.*, 2011). The smallest geographic areas which can be related to a statistic such as IMD are defined as Lower Super Output Areas (LSOAs). LSOAs are small areas of even size containing approximately 1,500 people (McLennan *et al.*, 2011).

IMD has also been used to describe the socio-economic category of public health by Tzoulas *et al.* (2007). The authors illustrated that the IMD describes social, economic, and some environmental conditions of communities. Moreover, as living and working conditions are determinants of public health (WHO., 1998; Paton *et al.*, 2005) the IMD's domains of deprivation (specifically living environment, educational level, access to services such as health care and housing) are also important determinants of public health (Tzoulas *et al.*, 2007). Ecological sustainability and human health are not seen as mutually exclusive and therefore decisions about the management of urban green space can have important consequences for health and vice versa (WHO., 2005). Therefore, the IMD can also be used to unravel the relationships between green infrastructure and public health (Tzoulas *et al.,* 2007).

The 2010 IMD rank of Salford's LSOAs were obtained and used as an indicator of the societal structure of Salford (obtained from data.gov.uk/dataset/index-of-multiple-deprivation). Each LSOA was assigned a corresponding code and IMD rank. The IMD rank for each LSOA was then stratified into four quartiles. Quartile 1 contained the least deprived LSOAs, quartile 2 contained the second least deprived LSOAs, quartile 3 contained the second most deprived LSOAs, and quartile 4 contained the most deprived LSOAs. As a result, the terms 'advantaged', 'less advantaged', 'deprived', and 'most deprived' were appointed to the IMD quartile areas; representing the level of relative deprivation experienced by the inhabitants of the IMD quartile area's LSOAs.

The ecosystem service and urban green infrastructure literature make explicit that the societal beneficial influences of urban trees (and green infrastructure in general) are strongest in the realms of crime (directly reducing crime or increasing community cohesion potential), health (mental and physical), and living environment (i.e. enhancing landscape quality) (e.g. NUFU, 2005; O'Brien, 2005; Tyrväinen *et al.*, 2005; Marmot, 2010; O'Brien *et al.*, 2010). Therefore, Salford's LSOAs were also categorised using the four quartile breaks in the health, crime, and living environment rank data. In regards to the crime rank data, quartile 1 LSOAs represented the 'lowest crime' areas, quartile 2 represented 'low crime' areas, quartile 3 represented 'high crime' areas, and quartile 4 represented the 'highest crime' areas. Using the health rank data, LSOAs were categorised as the 'healthiest' areas (quartile 1), 'healthy' areas (quartile 2), 'unhealthy' areas (quartile 3), or the 'unhealthiest' areas (quartile 4). The quartile breaks in the living environment rank data resulted in the four categories 'best environment' (quartile 1), 'good environment' (quartile 2), 'bad environment' (quartile 3), and 'worst environment' (quartile 4). In regards to the weight each domain of deprivation had within the calculation of overall IMD score (and therefore subsequent IMD rank), the health domain had the highest weight of 13.5%, followed by crime and living environment with an equal weight of 9.3%.

After using the overall and pertinent IMD ranks to classify and divide Salford's landscape into quartile areas, the structural patterns of the UTI within these quartiles were then explored.

6.2.2 UTI landscape structure analysis

The percentage of tree cover (landscape composition) and the range of tree heights (m) (landscape configuration) were calculated within each of Salford's IMD Quartile Areas (using tree patches defined in Chapter 3 and tree heights after interpolation, section 3.3.1). The percentage of tree cover was also calculated within the health, crime, and living environment quartile areas. The number of tree patches (NP) were calculated using the spatial pattern analysis programme FRAGSTATS (v4.2) (McGarigal *et al.*, 2002) while patch density per hectare (PD) was calculated in Excel (both aspects of landscape composition). The landscape configuration metric Contagion index (CONTAG) was used to assess the *structural* connectivity of tree patches within each IMD Quartile Area. FRAGSTATS was again used to calculate the Contagion score. CONTAG is based on the probability of finding cell type *i* next to cell type *k* and was first proposed by O'Neill *et al* (1988). FRAGSTATS uses one of the reviewed contagion index proposed by Li and Reynolds (1993) (McGarigal *et al.*, 2002) and is given by the equation:

$$CONTAG = \left[1 + \frac{\sum_{i=1}^{m}\sum_{k=1}^{m}\left[P_i \frac{g_{ik}}{\sum_{k=1}^{m} g_{ik}}\right]\left[ln\left(P_i \frac{g_{ik}}{\sum_{k=1}^{m} g_{ik}}\right)\right]}{2\ln(m)}\right](100)$$

Equation 6-1: Contagion (CONTAG).

where P_i is the proportion of the landscape occupied by a tree canopy patch, g_{ik} is the number of joins between tree canopy cells and the surrounding inhospitable matrix cells (each cell side is counted twice and only the four nearest orthogonal neighbours are considered), and *m* is the number of patch types in the landscape (i.e. two – tree canopy and matrix).

This contagion index is based on raster cell adjacencies and consists of the sum of the product of two probabilities (summed over all habitat patch types). In particular in this study those probabilities are: (1) the probability that a randomly chosen raster cell belongs to the patch type 'tree canopy' (i.e. *i* represents tree cover), and (2) the conditional probability that given a cell is of the patch type tree canopy, one of its neighbouring cells belongs to the patch type 'inhospitable matrix' (i.e. *k* represents the areas outside of tree cover, in the landscape matrix) (McGarigal & Marks, 1995). The product of these two probabilities equals the probability that if two adjacent cells are randomly chosen they will be of patch type tree canopy and inhospitable matrix. Therefore, CONTAG measures both patch type interspersion (i.e. the intermixing of canopy cover and non-canopy cover) and patch dispersion (i.e. the spatial distribution of tree patches) (McGarigal & Marks, 1995; McAlpine & Eyre, 2002; Xiao & Ji, 2007).

The results are given as a percentage and therefore CONTAG ranges from 0 to 100 and increases with structural connectivity. CONTAG approaches 0 when patches are maximally disaggregated (i.e. every cell is a different patch type). When CONTAG is 100, all patch types are maximally aggregated and therefore the landscape consists of a single patch (i.e. one singular canopy). Therefore, landscapes with large, contiguous canopy patches will have higher CONTAG scores than landscapes with several smaller, fragmented tree patches. Furthermore, the level to which the inhospitable matrix is interspersed throughout the UTI can also be described. Consequently, as habitat fragmentation leads to the concomitant processes of decreasing habitat suitability and increasing remaining patch isolation within the matrix (Joly *et al.*, 2003), CONTAG can also be used to understand the levels of tree canopy fragmentation within Salford. As a result, CONTAG should not be used in order to understand functional connectivity as organism movement distances are not considered. Instead, CONTAG scores can be used to ascertain where the UTI is most fragmented and where a greater ability or willingness to cross into and through the matrix is of importance for organism survival.

The Canopy Height Diversity index (CHD) was created to describe the vertical, structural diversity of Salford's UTI (three dimension configuration). Similar to the work of MacArthur and MacArthur (1961) the CHD index is an adapted version of the Shannon-Wiener diversity index. However rather than grouping the heights of intra-patch foliage into layers (i.e. the vegetation layers present within an area of tree cover) (MacArthur & MacArthur, 1961) only the heights of the upmost canopy were stratified and grouped. These stratified 'canopy height classes' were then used to calculate the diversity of canopy patches within Salford's UTI at differing heights. Thus, a measure of structural diversity was calculated which incorporates the third dimension. To measure CHD, the tree polygons of Salford were firstly stratified based on the natural breaks within the height data. This stratification created four canopy height classes; pre-standard trees (0-2.99m - the maximum height of this class was manually set to represent pre-standard trees (Miller, n.d.), small trees (3 – 7.19m), medium trees (7.2 – 17.49m), and large trees (17.5 – 34.9m). The tree polygons that fell within these height classes for each IMD quartile area were selected and exported so that they could be transformed to represent canopy patches (using methods described in section 3.4). Therefore, vertically stratified canopy patches within Salford's advantaged, less advantaged, deprived, and most deprived LSOAs were created. Using these canopy patches, an adapted Shannon-Wiener diversity index was calculated using the equation;

$$CHD = -\sum p_i(l_n p_i)$$

Equation 6-2: Canopy height diversity index (CHD).

where p_i is the proportion of total canopy patches which lay within the i^{th} of the chosen canopy height class (i.e. pre-standard, small, medium, or large).

The theoretical level of influence that fragmentation has on organism richness, abundance, and distribution was used with the aim of relating landscape structure to ecological function. Fragmentation is defined as the breaking apart of habitat (Fahrig, 1998; Fahrig, 2003). As a result, fragmentation increases the number and alters the spatial arrangement of habitat patches within a landscape, thus increasing habitat patch isolation and disrupting structural (and potentially functional) connectivity (With *et al.*, 1997; Fahrig, 2003). It has been theorised that organism 'survival' (that is richness, abundance, and or distribution of a species population) only becomes significantly affected by fragmentation within and below a critical habitat cover threshold (Andrén, 1994; Fahrig, 1998; Flather & Bevers, 2002; With *et al.*, 2002; Radford *et al.*, 2005; Swift & Hannon, 2010). Ultimately, within this chapter, the level of influence fragmentation has on organism survival, in relation to percentage of canopy cover, will be statistically assessed in line with area deprivation to explore the relationship between canopy structure function and deprivation. Therefore, the LSOAs of each IMD quartile area were grouped based on the percentage of canopy cover they exhibited. This grouping used the fragmentation threshold set out by Andrén (1994) who first demonstrated that the strength of influence fragmentation has on species population is dependent on habitat cover - specifically 10-30%. However, the overall amount of habitat within the landscape explains population size variance more than fragmentation (e.g. Flather & Bevers, 2002). Consequently, there are a greater number of studies on the effect of critical thresholds in habitat/vegetation loss on organisms than fragmentation thresholds (see Swift & Hannon, 2010). However, an actual habitat or fragmentation threshold value, as well as the need for one, is disputed by Parker & MacNally (2002) and Lindenmayer *et al.* (2005b). Nonetheless, Fischer & Lindenmayer (2007) highlight that although, what they term, the ‘30% rule’ does not apply to all species or ecosystems, they do propose that the vegetation threshold literature illustrates that extinction cascades are likely to occur at low levels of vegetation cover. Likewise, sharp declines in woodland bird species richness have been empirically demonstrated when habitat cover is at 10% (Radford *et al.*, 2005). Ergo, classifying landscapes based on habitat cover can be useful in identifying areas where species have been, or may become, negatively affected. Besides, the most recent review of critical thresholds associated with both habitat loss and habitat amount, concludes that although the exact values of a threshold depend on several factors, most empirical values fell near Andren’s (1994) proposed range of 10–30% habitat cover (Swift & Hannon, 2010).

As a result, Salford’s LSOAs were divided into three canopy cover groups based on whether they had less than 10%, between 10% and 30%, or greater than 30% canopy cover. Using the literature discussed above as a guide, these groups were labelled dependent (<10%), influential (10-30%), and independent (>30%), thus representing the potential influence fragmentation has on organism survival.

6.2.3 Statistical Analysis

A series of statistical tests were used to assess the significant relationships between area deprivation and UTI structural landscape patterns. The relationship between percentage of tree cover and area deprivation was tested using the overall, crime, health, and living environment IMD ranks. Percentage of tree cover was used as an initial assessment in order to reveal if any likely differences between the overall IMD quartile areas and the three domains of deprivation quartile areas existed. The results of this analysis subsequently showed that all IMD quartile areas, regardless of which IMD rank was used to describe them, followed a similar tree cover (%) pattern (see Chapter 4.3). As a result, further statistical analysis between deprivation and canopy structure only considered overall IMD quartile areas. Therefore, the relationships between UTI landscape structural patterns (e.g. NP, PD, CONTAG, and CHD) and the four IMD quartile areas (e.g. advantaged, less advantaged, deprived, most deprived) were assessed.

A Kolmogorov – Smirnov test (K-S Test) was used to assess the distribution of the percentage of canopy cover data. The null hypothesis had to be rejected for the canopy cover data (K-S Test=0.313, df=144, P-Value = <0.05) and therefore the Kruskal-Wallis test (H) was chosen to compare averages of percentage canopy cover between each IMD quartile area (i.e. overall, crime, health, and living environment). Pearson's product-moment correlation (r) was used in an effort to reveal significant association between NP, PD, CONTAG, CHD and overall IMD rank. An independent t-test (t-test) was undertaken to assess the difference in means between the number of post stratified canopy patches within the advantaged and most deprived IMD quartile areas. The distribution of the stratified canopy patches were assessed for normality and variance using the Shapiro-Wilk test (SW) and Levene's test for equality of variances (F) respectively. Number of canopy patches within the advantaged and most deprived quartile areas were normally distributed (SW = 0.92 and 0.95, p = 0.56 and 0.7) and equal variances were assumed (F = 4.813, p = 0.071). Finally, a Pearson's Chi-Square Test for Independence (X^2) was used to assess whether, statistically, influence of fragmentation on organism survival was independent from area deprivation. Using the Cramer's V coefficient, the strength of the observed results from the X^2 test were assessed. All statistical analysis was undertaken within SPSS (v20) statistical software.

6.3 Results

6.3.1 Salford's societal structure

Each of Salford's LSOAs were ordered based on overall, crime, health, and living environment IMD rank. The distribution of Salford's overall area deprivation can be seen in Figure 6-2 (A). The most deprived areas (red LSOAs) are predominantly within the north-west, east, and south central parts of Salford, often flanked by the deprived areas (orange SLOAs). The less advantaged areas (yellow LSOAs) exist mostly across the northern, extending both to the east and west, and south-western borders of Salford. The advantaged areas of Salford (green SLOAs) lay to the west of the city. The other domains of deprivation maps follow similar patterns, with a few exceptions – specifically in the crime deprivation map (Figure 6-2, B).

While the west of Salford is largely defined as advantaged (Figure 6-2, A) the use of the crime rank data also reveals that it is an area of high crime (orange LSOAs, Figure 6-2, B). As a result, low crime areas (green LSOAs) are scattered across Salford while the highest crime LSOAs (red LSOAs) are, in the main, also the most deprived areas of Salford (Figure 6-2, A and B). The health deprivation map follows a similar pattern to the overall deprivation map, with only minor differences (Figure 6-2, C). Hence, for the most part, the advantaged LSOAs are the healthiest and the most deprived LSOAs are the unhealthiest (Figure 6-2, A and C). In regards to living environment deprivation the worst environments exist to the north, east, and south of Salford (Red LSOAs, Figure 6-2, D). There are no worst environments in the north-west of Salford where, in the overall IMD map, there is a cluster of most deprived LSOAs (Figure 6-2, A and D). This pattern shift is the main difference between the overall and living environment deprivation maps.

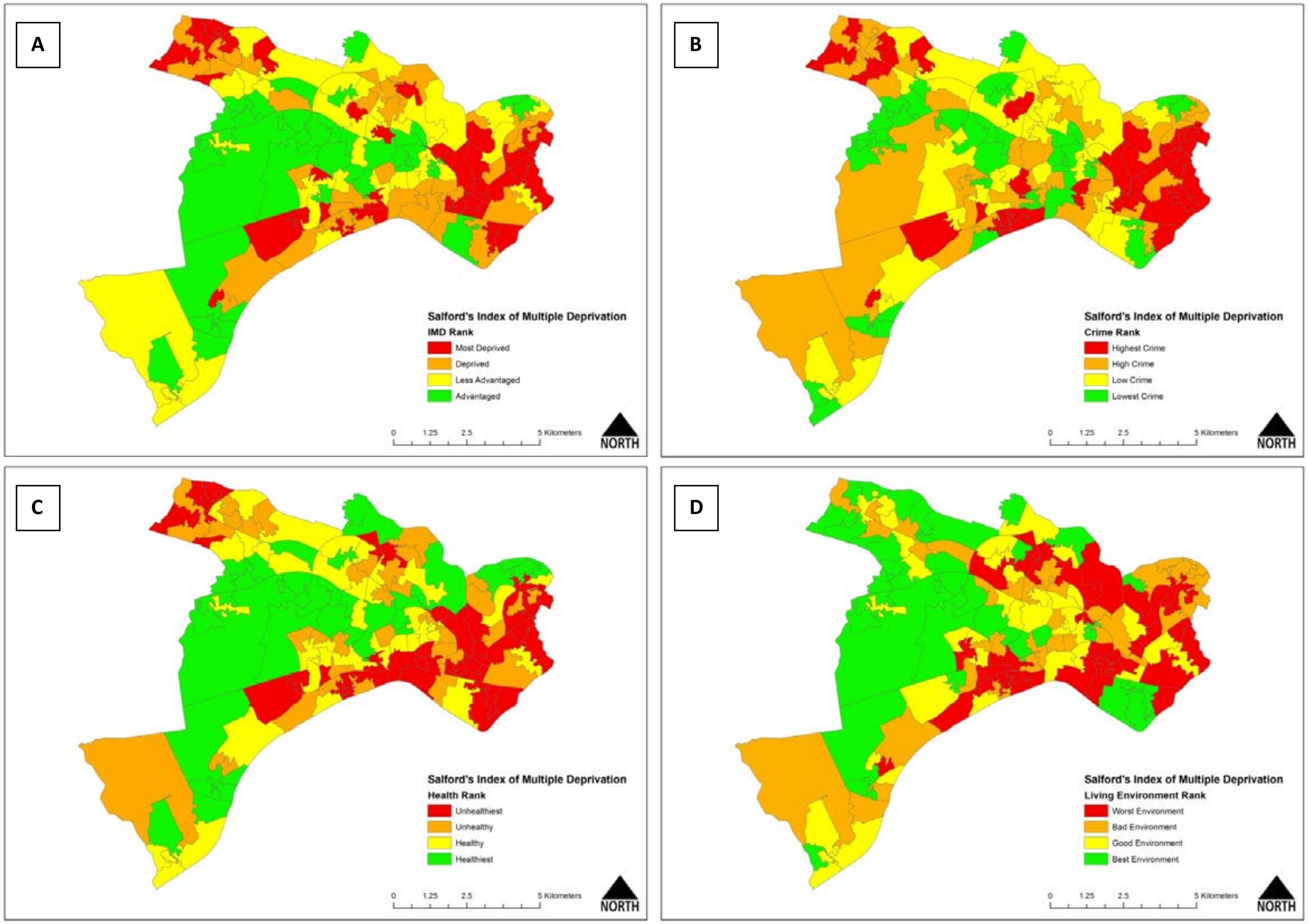

Figure 6-2: The levels of deprivation in Salford.

The IMD ranks of Salford's LSOAs were divided into 4 quartiles based on the overall (A), crime (B), health (C), and living environment (D) rank data. The more urbanised areas of Salford in the north and the east are the most deprived. The more rural fringe areas to the west, containing greater amounts of green space and agricultural land, are the least deprived areas (with the exception of crime derivation).

6.3.2 UTI landscape patterns and statistical relationships

The structural landscape patterns of Salford's UTI within the IMD quartile areas were described by focusing on a set of key structural components; namely percentage of canopy cover, theoretical influence of fragmentation to organism/population survival, number of canopy patches (NP), canopy patch density (PD), structural contagion, and canopy height diversity (CHD.

The highest canopy cover values (30 – 45% cover) were calculated for 13 advantaged LSOAs (Figure 6-3). Within the less advantaged quartile area only 5 LSOAs contained high canopy cover, which reduced to 3 LSOAs in the deprived quartile area (Figure 6-3). No high amounts of canopy cover were exhibited within the most deprived LSOAs (Figure 6-3). The high canopy cover LSOAs form a 'green belt' which runs across Salford from the western boundary, through the centre of the city, to the eastern boundary. However, there is a break in this pattern towards the eastern part of Salford, where clusters of highly deprived LSOAs exhibit low to medium amounts of canopy cover (Figure 6-3). For all the LSOAs comprising the IMD quartile areas the majority of canopy cover fell between 10% and 30% (Advantaged = 21 LSOAs, Less Advantaged = 28 LSOAs, Deprived = 28, LSOAs, Most Deprived = 27 LSOAs, Figure 6-3). All IMD quartile areas contained LSOAs with low amounts of canopy cover (5 – 10%). The advantaged quartile area contained 4 low canopy cover LSOAs while the less advantaged contained 2 and the deprived 5. The majority of the low canopy cover LSOAs (8) were classified as most deprived.

Salford's advantaged quartile area contained the highest percentage of tree cover and as deprivation increased tree cover decreased (median tree cover; advantaged = 21.6%, less advantaged = 19.4%, deprived = 13.5%, most deprived = 13.5%) (Figure 6-4). This relationship was found to be significant ($H = 20.53$, $df = 3$, $P \geq 0.001$) thus there is a negative association between deprivation and tree cover (Figure 6-4). The strength of this associative relationship develops as the other domains of deprivation are considered (Figure 6-5).

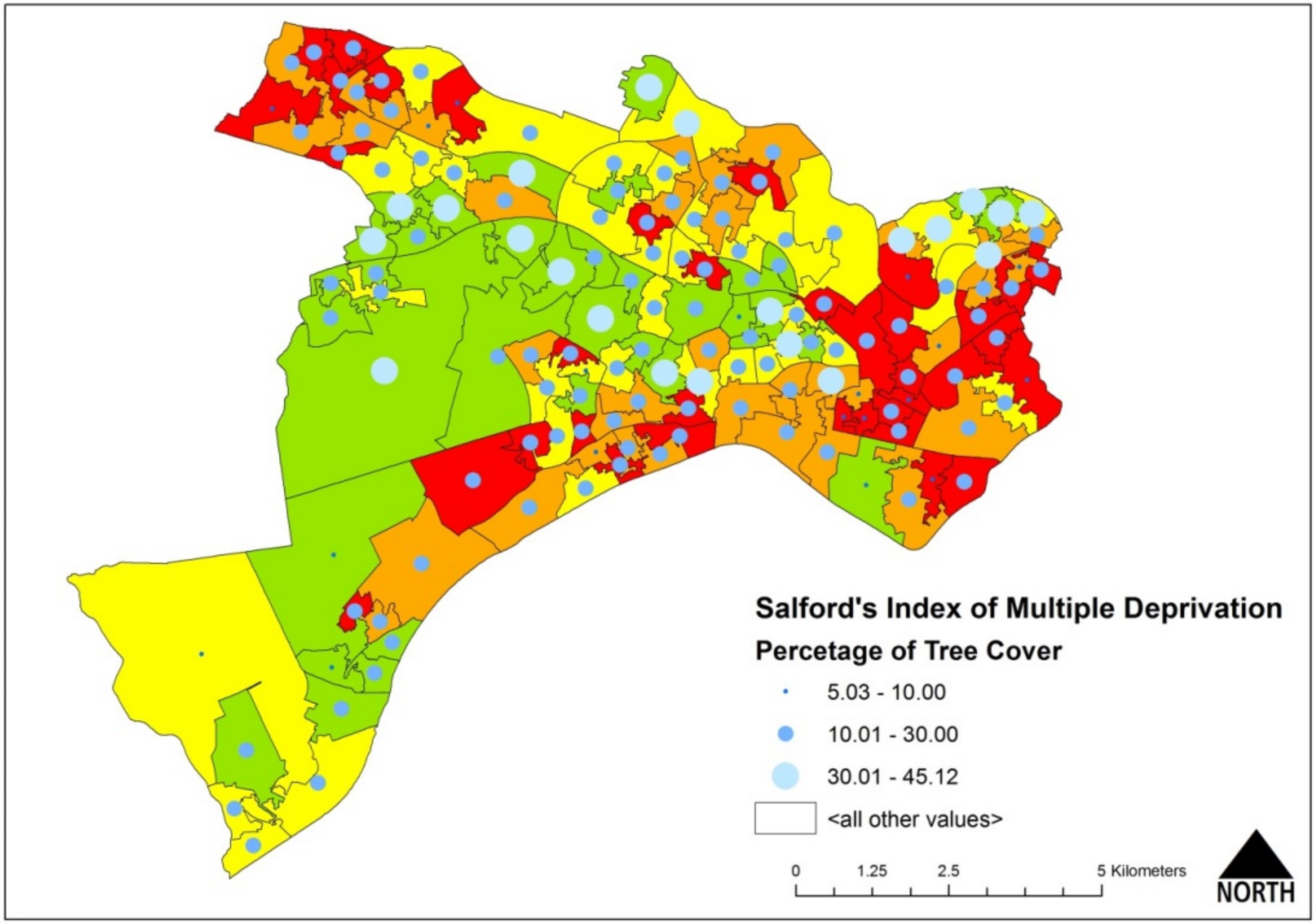

Figure 6-3: Percentage of Tree cover within each IMD quartile area.

In the advantaged quartile area (green) a total of 13 LSOAs contain high canopy cover (30.01% – 45.12%), 21 LSOAs contain a medium amount of canopy cover (10.01 – 30%), and 4 LSOAs contain low canopy cover (5.03 – 10%). The less advantaged quartile area (yellow) contains 5 LSOAs exhibiting high canopy cover, 28 exhibiting medium canopy cover, and 2 exhibiting low canopy cover. In the deprived quartile area (orange) only 3 LSOAs contain high amounts of canopy cover, 28 contain medium amounts of canopy cover, and 5 contain low amounts of canopy cover. The most deprived quartile area (red) contained no high canopy cover LSOAs, 27 medium canopy cover LSOAs, and 8 low canopy cover LSOAs.

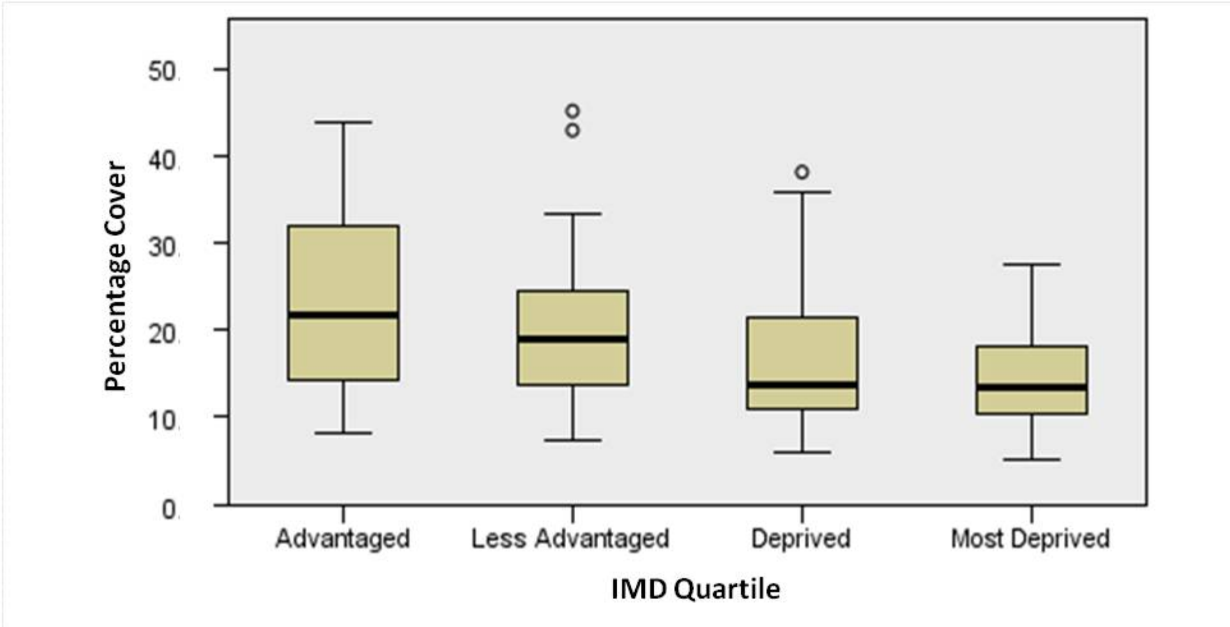

Figure 6-4: The percentage of tree cover within the IMD quartile areas.
As deprivation increases, the percentage of tree cover present within the IMD quartile area decreases. There was a significant difference in the percentage of tree cover between the IMD quartile areas (H=20.53, df=3, P≤0.001).

Crime deprivation follows a similar pattern as overall deprivation – percentage of canopy cover decreases as deprivation increases. However the only clear difference in median percentage of tree cover between the IMD quartile areas was calculated for the highest crime quartile area (median tree cover; lowest crime = 18.3%, low crime = 18.2%, high crime = 18.2%, highest crime = 13.4%). Nonetheless, the negative relationship between crime deprivation and tree cover was found to be statistically significant (H = 16.48, df = 3, P = 0.001) (Figure 6-5). In regards to health deprivation, the healthiest quartile area contained the greatest amount of tree cover, which decreased as health deprivation increased (median tree cover; healthiest = 27.7%, healthy = 16.4%, unhealthy = 13.8%, unhealthiest = 13.4%). This relationship was found to be statistically significant (H = 26.29, df = 3, P ≤ 0.001) (Figure 6-5). Finally, as living environment deprivation increases, tree cover decreases (median tree cover; best environment = 18.8%, good environment = 17%, bad environment = 16.3%, worst environment = 12.8 %). Yet again, this negative relationship between living environment deprivation and tree cover was statistically significant (H = 10.46, df =3, P = 0.015) (Figure 6-5).

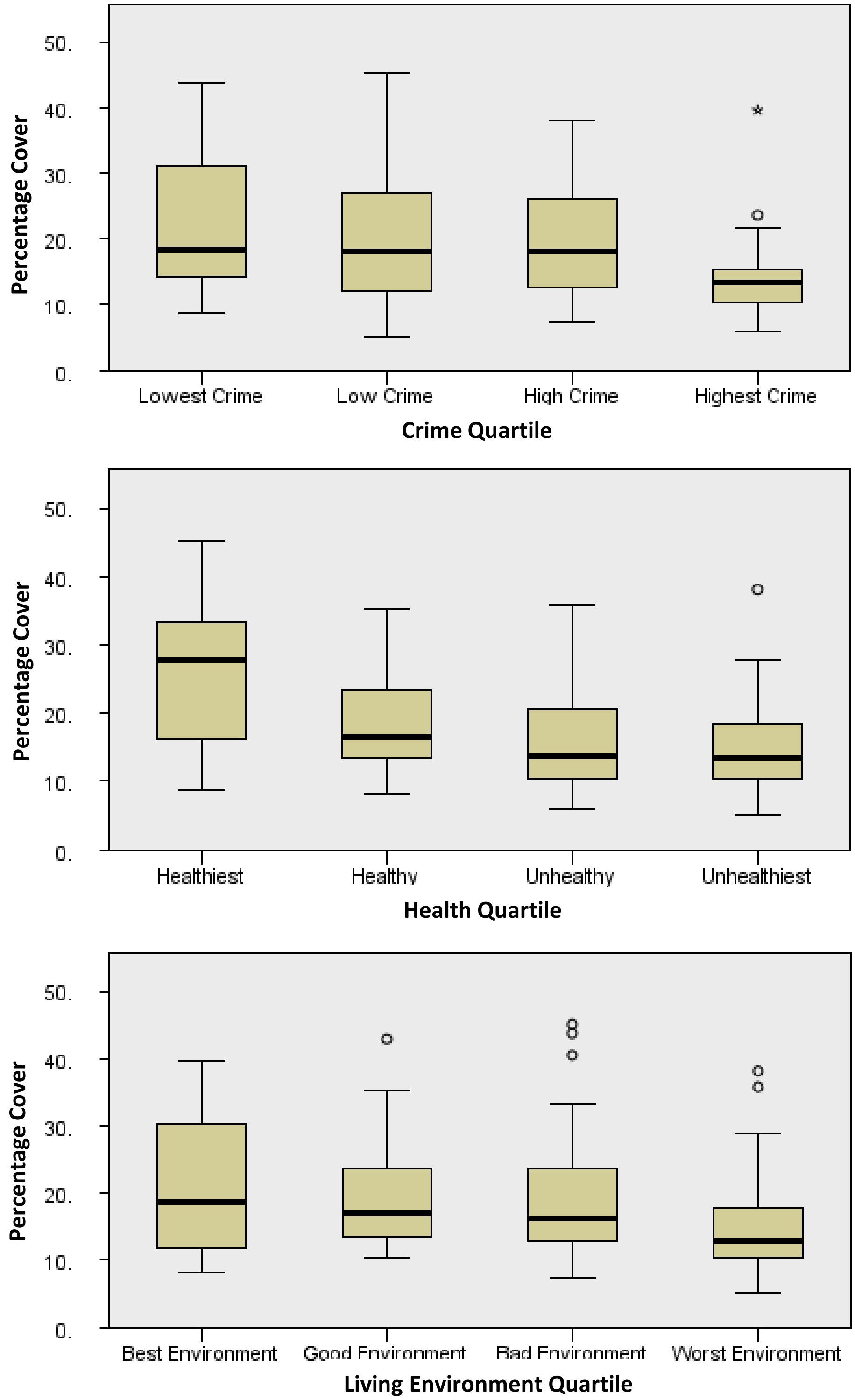

Figure 6-5: Percentage of tree cover within the crime, health, and environment quartile areas.
Percentage of tree cover decreases in line with an increase in the different domains of deprivation. This relationship is statistically significant (Crime deprivation quartiles, H = 16.48, df 3, P = 0.001; Health deprivation quartiles, H = 26.29, df = 3, P ≤ 0.001; Environmental deprivation quartiles, H = 10.46, df =3, P = 0.015)

Percentage of canopy cover within the LSOAs of each IMD quartile area was categorised based on the theoretical strength-of-influence fragmentation has on species survival. Using the vegetation threshold described by Andrén (1994), it was assumed that organisms inhabiting a landscape with more than 30% canopy cover were relatively independent from fragmentation effects (thus the LSOAs with 30% or more canopy cover were classified as independent). Fragmentation effects were regarded as influential to organism survival within landscapes which exhibited 10-30% canopy cover while organisms occupying areas with less than 10% canopy cover were considered as being comparatively dependent on the degree of canopy fragmentation. Therefore, LSOAs exhibiting percentage of canopy cover values that fall within these categories were termed influential or dependent.

It was found that the influence of fragmentation categories (Frag-Influence; dependent, influential, and independent) correlated significantly with area deprivation ($X^2 = 22.37$, df = 6, p = 0.001) (Figure 6-6). The number of LSOAs which were defined as influential remained relatively constant for all IMD quartile areas, with only a noticeable increase between the number of influential LSOAs within the advantaged and less advantaged areas (number of influential LSOAs within each IMD quartile area; advantaged = 21, less advantaged = 28, deprived = 28, most deprived = 27) (Figure 6-6). The dependant LSOAs increased in number as deprivation increased, with an exception laying between the advantaged and less advantaged quartile areas (number of dependant LSOAs within each IMD quartile area; advantaged = 4, less advantaged = 2, deprived = 5, most deprived = 8) (Figure 6-6). In contrast, the number of independent LSOAs decreased as deprivation increased, to such an extent that there were no LSOAs classed as independent within the most deprived quartile area (number of independent LSOAs within each IMD quartile area; advantaged = 13, less advantaged = 5, deprived = 3, most deprived = 0) (Figure 6-6). The strength of these associations was found to be strong (Cramer's V = 0.3, p = 0.001) (Figure 6-6).

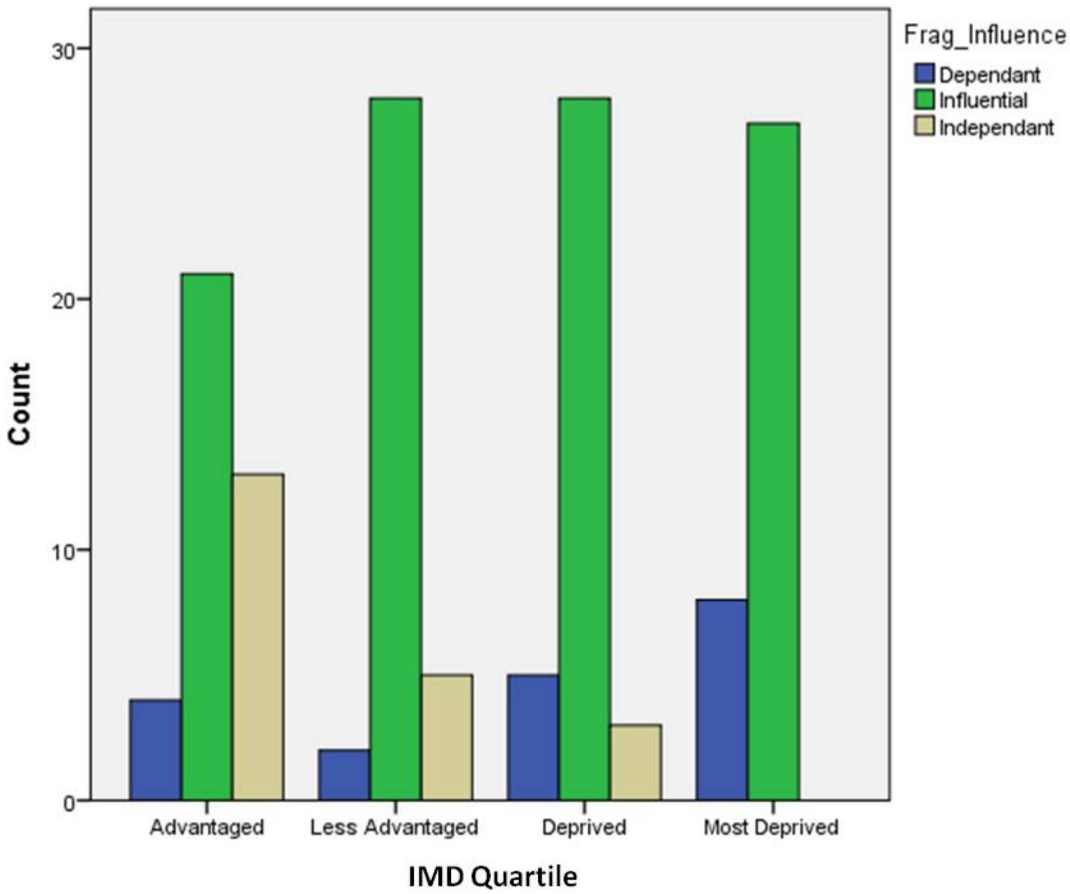

Figure 6-6: Levels of fragmentation influence exhibited within the LSOAs within the IMD quartiles.
There was a statistically significant relationship between deprivation (IMD_Quartile) and the influence of fragmentation towards organisms survival (Frag_influence) within the LSOAs in each IMD quartile area (count) (Pearson's Chi-Square = 22.37, df=6, p=0.001). The strength of this association between the variables was strong (Cramer's V = 0.3, p = 0.001)

The number of overall tree patches (NP), patch density (PD), and contagion (CONTAG) within each IMD quartile area was calculated using Fragstats (v4.2) and Excel (for PD only). As area deprivation increased the NP value began to decrease (Q1 NP = 32463, Q2 NP = 29762, Q3 = 23292, Q4 NP = 18382) (Figure 6-7). The resulting negative correlation between NP and area deprivation is statistically significant (r = -0.989, n = 4, p = 0.005) (Figure 6-7). However, an opposite relationship was revealed when assessing the spatial composition of these patches as PD (per hectares) increases in line with deprivation (Q1 PD = 9.7, Q2 PD =10.2, Q3 PD =12.1, Q4 PD =11.8) (Figure 6-8). This associative relationship is also statistically significant, yet only at the

0.05 level (r = 0.903, n = 4, p = 0.048). The statistical assumption that UTI landscape composition relates positively to an increase in area deprivation is supported by the contagion (CONTAG) results in Figure 6-9 (Q1 CONTAG = 45.8, Q2 CONTAG = 52.8, Q3 CONTAG = 53.3, Q4 CONTAG = 56.9). Therefore, it can be proposed that as area deprivation increases so does the structural connectivity of the UTI; which is to say the level of fragmentation within the UTI decreases as deprivation increases. Such a proposition is statistically supported as the relationship between CONTAG and area deprivation is significant (r = 0.939, n = 4, p = 0.031) (Figure 6-9).

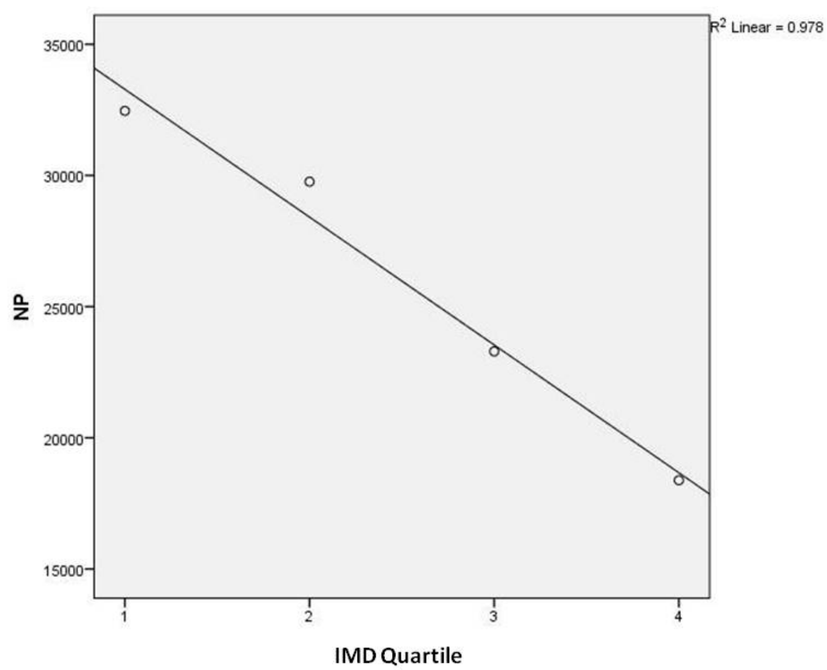

Figure 6-7: The number of tree canopy patches (NP) within each IMD quartile area.
There is a statistically significant negative correlation between number of tree canopy patches (NP) and area deprivation (IMD_Quartile) (r = -0.989, n = 4, p = 0.005).

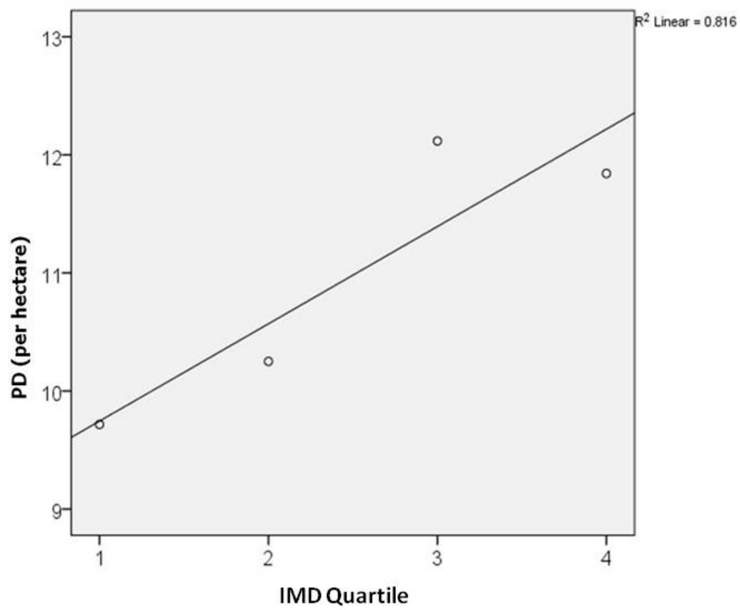

Figure 6-8: The patch density (PD) of each IMD quartile area.
There is a statistically significant positive correlation between area deprivation (IMD_Quartile) and patch density (PD) per hectare (r = 0.903, n = 4, p = 0.048). Therefore, the correlation is significant at the 0.05 level (1-tailed).

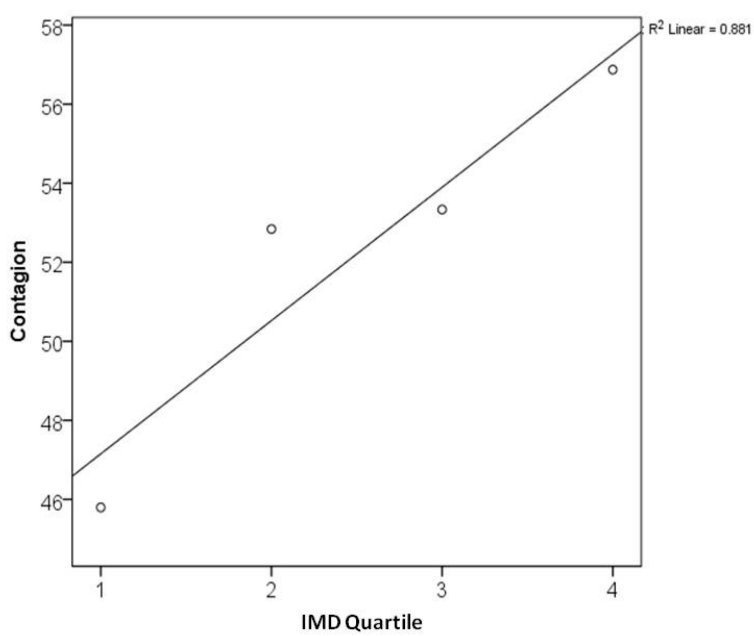

Figure 6-9: The Contagion Index (CONTAG) score for each IMD quartile area.
There is a statistically significant positive correlation between area deprivation (IMD_Quartile) and structural connectivity (Contagion) (r = 0.939, n = 4, p = 0.031). Correlation is significant at the 0.05 level (1-tailed).

Tree canopies were vertically stratified based on the natural breaks in the height data (with the 3m break set manually). The number of tree canopy patches within the height break categories was calculated for each IMD quartile area (Figure 6-10). For all quartile areas the greatest numbers of canopy patches were found to be 3-7.19m high, followed by 0-2.99m, 7.2-17.49m, and then 17.5-34.9m (Figure 6-10). The total number of canopy patches for all height break categories increased as deprivation decreased (Figure 6-10). However, the CHD results did not significantly correlate with an increase in area deprivation (Q1 CHD = 1.123, Q2 CHD = 1.114, Q3 CHD = 1.126, Q4 CHD = 1.128, r = 0.569, n = 4, p = 0.215). Therefore, no relationship was revealed between structural diversity, based on canopy height, and area deprivation, thus vertical structure diversity is relatively equal across Salford, regardless of deprivation (Figure 6-11). The results in Figure 6-11 are supported by the independent t-test results which reveal that there is no statistically significant difference between the number of canopy patches described within each canopy height class in the advantaged and most deprived LSOAs (t = 1.2, df=6, p=0.26).

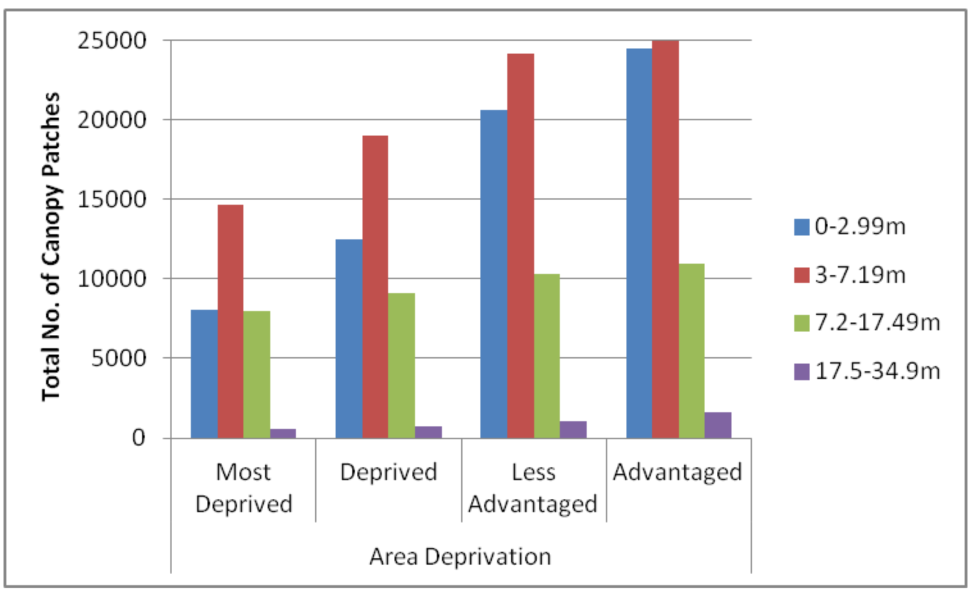

Figure 6-10: Total number of canopy patches within each canopy height class.
The total number of canopy patches within the vertically stratified canopy classes increase as deprivation decreases. All quartile areas followed the same pattern in regards to the number of canopies within each vertical class. Specifically, the greatest amounts of canopy patches were between 3-7.19m, then 0-2.99m, 7.2-17.49m, and finally 17.5-34.9m (maximum tree height).

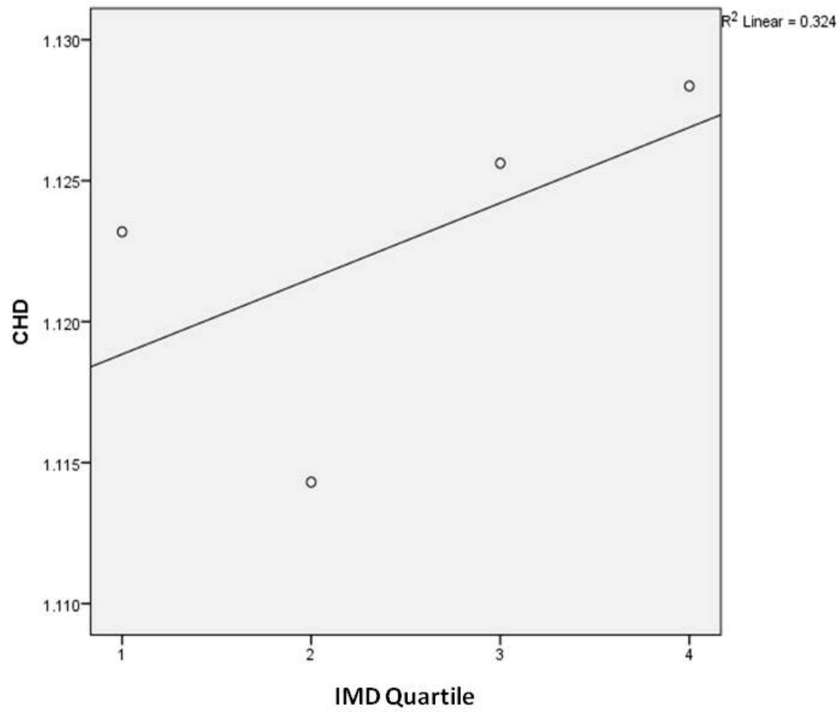

Figure 6-11: Relationship between canopy height diversity (CHD) and area deprivation (IMD_Quartile). There is a positive correlation between an increase in area deprivation and an increase in CHD but it is not statistically significant ($r = 0.569$, $n = 4$, $p = 0.215$). Vertical structural diversity is homogenous across Salford, regardless of area deprivation.

6.4 Discussion

The results presented in section 6.3 are formed from a quantitative assessment of both ecological landscape patterns and the societal metric IMD landscape patterns. Although there was no guiding research hypothesis, due to the utilisation of an explorative research approach, work undertaken from the position that an area's level of social deprivation (at LSOA level) is indicative of the amount of canopy cover within that area. Such a relationship is well known (see section 2.1 and 6.1) and therefore a major impetus behind the research undertaken in Chapter 6 was to explore new relationships by incorporating innovative approaches. An overview of the research originality is presented below followed by a discussion on the relationship between UTI landscape structure and area deprivation (section 6.4.2) and then the potential impact of these relationships on greenspace management (section 6.4.3).

6.4.1 Overview

A study by the Commission for Architecture and the Built Environment (CABE, 2010a) found that residents of deprived inner city areas within the UK have access to five times fewer parks and good-quality green space than people residing in more affluent areas. A follow up study concluded that improving urban green space is an important, cost-effective, way of improving local neighbourhoods and resident's quality of life (CABE, 2010b). The apparent novelty of this second study was that previously only a few studies tackled the relationship between green space, deprivation (or ethnicity), and health (CABE, 2010b). The research presented in Chapter 6 supports previous research claims that the UTI is disproportionately represented across communities in urban areas. The novelty of the Chapter 6 research is that it improves on current knowledge by relating pertinent, vertical and horizontal, structural landscape patterns to urban societal structure (in regards to IMD patterns across the Salford landscape). Although the research presented in Chapter 6 has focused on only a single aspect of green space – the UTI (a key component of green infrastructure and green spaces) – it has been borne from a unique scale and focus. Specifically, a multi-scale analysis has been employed; incorporating the small area scale of single LSOAs to create larger scale IMD quartiles in order to describe the city of Salford at a landscape scale. Furthermore, at the small area scale canopy patches were also described using the third dimension, by incorporating height. The use of these scales in conjunction with a comprehensive description and assessment of landscape structure has led to a significant number of newly established relationships between the UTI and urban social deprivation (section 6.4.2).

A purely spatial, quantitative approach was used to infer relationships between the UTI and urban communities (delineated by the levels of area deprivation). It was assumed that the potential ecosystem services provided by urban trees are higher in areas exhibiting significant amounts of canopy cover, structural diversity, and structural connectivity (see section 2.2 and 2.3). However, perception of green infrastructure (including the UTI) also plays a pivotal role in the ways individuals receive urban ecosystem services (Jones *et al.*, 2009; Nisbet & Zelenski, 2011). When such a qualitative approach has been undertaken (e.g. CABE, 2010b) the size of the areas being sampled cannot be at the same scale as a spatial, landscape scale assessment. Additionally, little is known about how the urban population precisely interact with green infrastructure on a larger landscape scale (Moseley *et al.*, 2013) and they are mostly unaware of the ecological processes providing them with services, especially the more environmental services (Agbenyega, 2009; Escobedo *et al.*, 2011). Therefore, a quantitative, landscape approach was selected over a more qualitative, social approach to examine the area deprivation – UTI relationship set out in Chapter 6. Moreover, such an approach facilitates an ecological assessment of the results presented above (section 6.3), thus multi-scale structural landscape patterns can be assessed in terms of their relationship with both area deprivation and ecological function (section 6.4.2).

In addition to perception, access and use of green spaces and green infrastructure influence how ecosystem services are utilised (Moseley *et al.*, 2013). Previous studies that have attempted to measure levels of greenspace access have provided standards, mostly based on size of greenspace in relation to travel distance, which can be used by landscape planners (e.g. Box & Harrison, 1993; Harrison *et al.*, 1995). A study by Moseley *et al.* (2013) highlighted that these standards are often arbitrary and do not take into account variations in the landscape configuration of greenspaces. The authors' developed a new methodology to effectively measure greenspace access by describing greenspace use profiles and by relating levels of landscape configuration to greenspace access. The aim addressed in Chapter 6 was not to identify or measure levels of access but describe structural landscape patterns within different areas of deprivation. By doing so, any discovered significant relationships can be used to inform future landscape planning, based not on arbitrary distances or complicated movement and resource use probabilities/patterns but on ecological and socially relevant spatial patterns. Furthermore, highlighting that an area needs more green space and or access should only be the beginning of greenspace planning. Understanding the structural characteristics of green space in differing areas of deprivation would allow for targeted and specific management or planning criteria to be enforced. In other words, if the composition and configuration of the UTI reveals a characteristic pattern in relation to social deprivation, landscape management and greenspace planning can employ techniques which target and adapt those patterns (section 6.4.3).

6.4.2 The spatial relationship between UTI landscape composition, configuration, and structural connectivity and area deprivation

The relationship between UTI landscape composition and Salford's societal deprivation is a negative one. As overall deprivation exhibited within an area increases, the percentage of canopy cover decreases (Figure 6-3). As a result, people inhabiting deprived areas of Salford do not have as much direct access to the benefits provided by urban trees compared to those within the more advantaged areas. The deprived, low canopy cover areas exist to the east of Salford in Broughton, S. Kersal, Irwell Riverside, E. Ordsall, and Langworthy; to the south in S. Weaste and Seedley, N.E. and S.E. Irlam, Barton, S. Eccles, and N. and S. Winton; to the north in S. Pendlebury, E. Swinton North, and N. Swinton South; and to the north west in Little Hulton and N.W. Walkden North (Figure 6-12). These wards (and specific LSOAs within them) represent Salford's urbanised areas, comprising residential neighbourhoods of low quality housing, large industrial areas, and poorly developed shopping centres (Table 3-1). The, relatively, advantaged areas of Salford exist to the north east in N. Kersal; to the south east in S. Ordsal; to the south west in N.E. and S.E. Irlam, and Cadishead; and to the West and Central parts of the city in Walkden South, Boothstown and Ellenbrook, Worsley, E. and S. Swinton South, and N. Eccles (Figure 6-12). These wards (and certain of their LSOAs) contain areas characteristic of rural and urban mosslands and urban fringe lowlands containing relatively little built environment yet relatively large amounts of greenspace (and UTI) (Table 3-1).

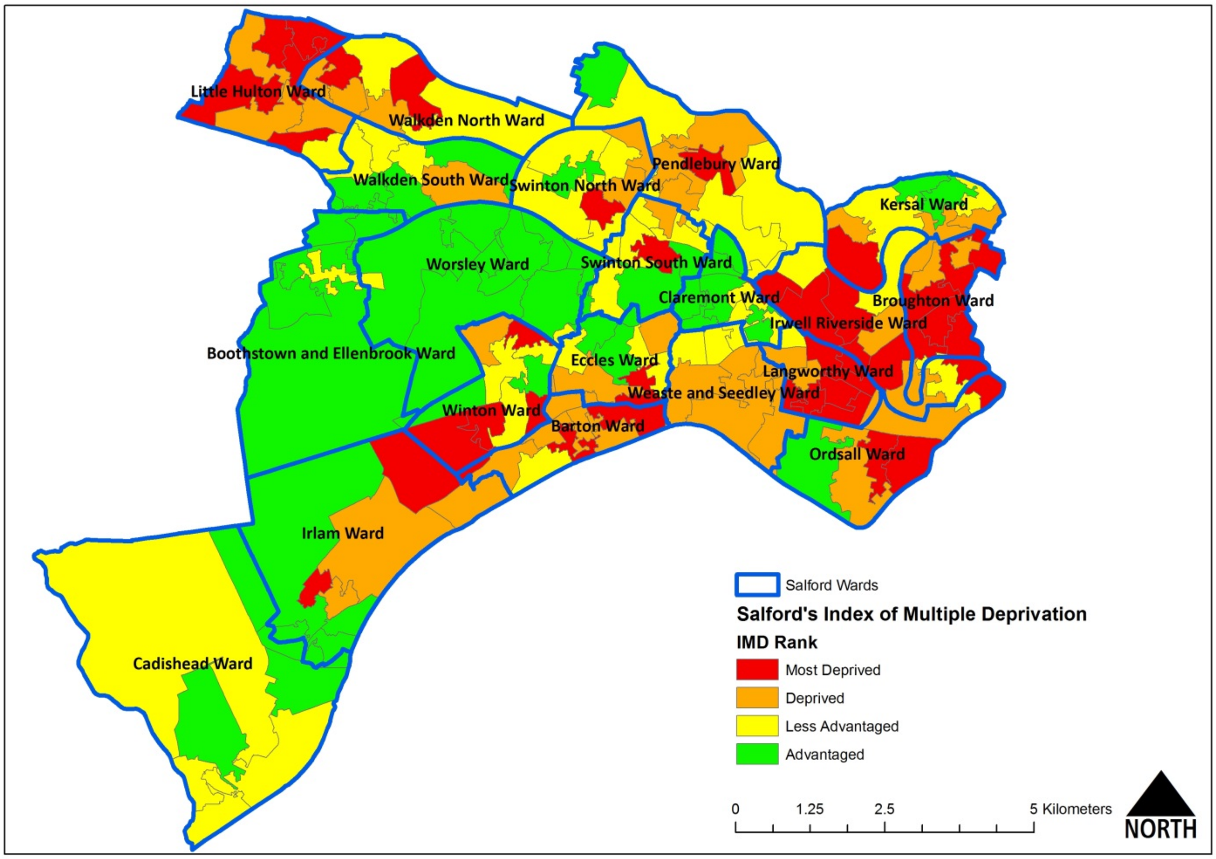

Figure 6-12: Salford wards and IMD rank.
Several of Salford's wards are made up of LSOAs of varying deprivation. However, certain wards such as Worsley, Boothstown and Ellenbrook, and Cadishead contain LSOAs of homogenous levels of low deprivation. On the other end of the spectrum wards such as Little Hulton and Broughton are a rather homogenous landscape with regards to levels of high deprivation.

Fragmentation of the UTI is more influential to organismal survival (in regards to species/population richness, abundance, and or distribution) within the areas of Salford exhibiting high levels of deprivation (when using the 10-30% habitat threshold) (Figure 6-6). Therefore where deprivation is high, the configuration of the UTI canopy patches is more influential to a population than in the less deprived areas which have higher amounts of canopy cover. Interestingly, the density of habitat patches is higher within the more deprived areas meaning fragmentation may be less in these more deprived areas (Figure 6-8). This relationship was proven to be true as structural connectivity, measured using the contagion metric (CONTAG), is greater in the more deprived areas compared to the less deprived areas

(Figure 6-9). In other words, UTI canopy patch fragmentation decreases as deprivation increases and for that reason the inter-dispersion of the urban matrix is less in the more deprived areas. However, it must be kept in mind that these deprived areas also contain little habitat cover; an ecological condition which is more influential to species survival than habitat isolation and connectivity. However, structural connectivity can be an indicator of functional connectivity and when taken together (i.e. landscape connectivity) these phenomena underpin the movement of organisms and energy within an area (see section 2.3). Therefore, connectivity is an important structural quality affecting habitat quality at a landscape scale. Consequently, the more deprived areas of Salford reveal higher structural quality than the less deprived areas. To understand if this relationship transfers to the habitat patch scale, levels of canopy patch structural complexity was investigated.

A key component of structural complexity, measured as canopy height diversity (CHD), is equal across Salford's LSOAs (Figure 6-11). Despite the level of deprivation within an area the proportion of small, medium, and large trees within a canopy patch are equal. Therefore, although the percentage of canopy cover and the number of canopy patches are higher in more advantaged areas, the level of vertical canopy configuration complexity – a measure of canopy patch quality – is the same in all areas of deprivation. This is a new finding. By describing diversity of canopy heights, canopy patches are no longer regarded as homogenous areas of vegetation but vertically heterogeneous and structurally complex (Dolman *et al.*, 2007a). The diversity of canopy heights within tree canopy patches is influential to the overall structural complexity of that patch (however it is only one component of complexity, others being mean diameter at breast height, amount of dead standing trees, and diversity of understorey vegetation) (McElhinny *et al.*, 2005). These complementing components of complexity were outside the scope of the research presented here. High diversity of canopy heights can be related to high bird species richness, overall structural and ecological quality of a habitat patch, and therefore potential higher ecosystem service provision (which is depended not only on quantity but quality of green space) than less vertically diverse canopy patches (MacArthur & MacArthur, 1961; Rosenzweig, 1995; Savard *et al.*, 2000; TEEB, 2010b). Therefore, it is interesting to reveal that canopy height diversity (CHD) is independent of area deprivation and as such similar levels of canopy patch structural quality are seen across the whole of Salford.

Results from chapter 6 also provide further support for the conceptual framework linking green infrastructure, ecosystem health, and human health developed by Tzoulas *et al.,* (2007) (Figure 6-13). This framework was formulated to provide an interdisciplinary 'conceptual meeting point' between practitioners in urban ecology and public health professionals (Tzoulas *et al.*, 2007, p. 175). Green infrastructure (such as the UTI), the ecosystem services it provides, and the varied aspects of ecosystem health these components influence represent the environmental setting of public health. These environmental settings are then influential to, and in turn influenced by, aspects of human health (Figure 6-13). However, not all of the ecosystem and human health elements described in Figure 6-13 were evaluated in Chapter 6.

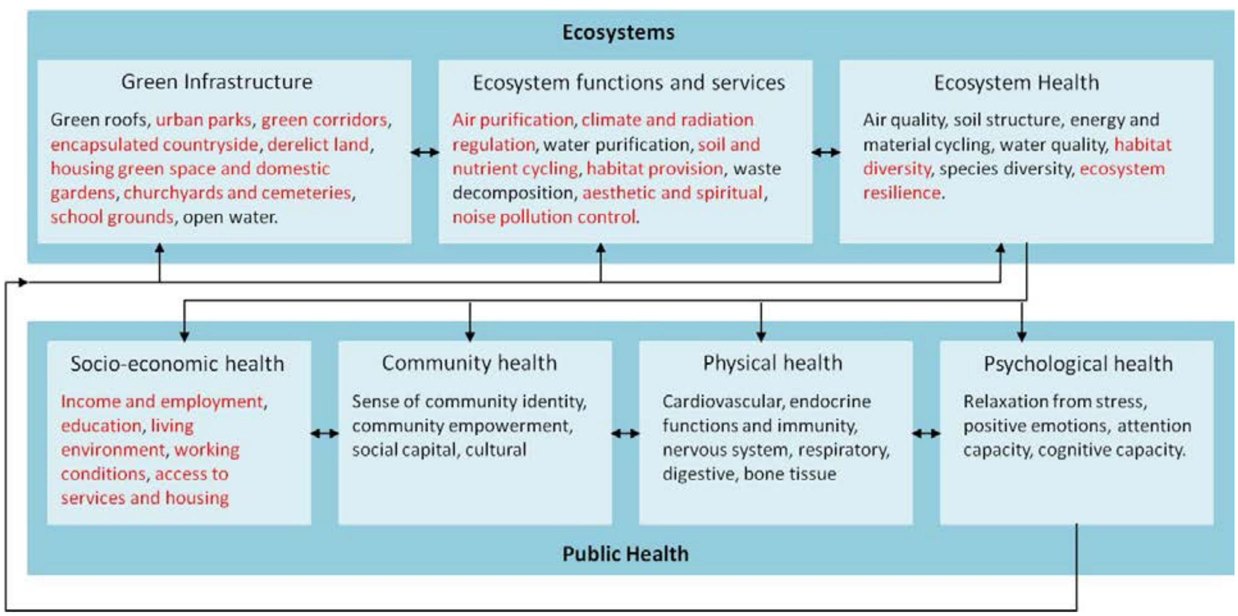

Figure 6-13: The conceptual links between green infrastructure, ecosystems service and health, and public health (adapted from Tzoulas *et al.*, 2007).

The three aspects of urban ecosystems – green infrastructure, function and services, and health – are all represented by the UTI (specifics in red text). These components set the environment for the aspects of public health – socio-economic, community, physical, and psychological – however only socio-economic health has been described in chapter 6. The two-way arrows represent the interrelationship between the urban ecosystem and the public health components (and between the components themselves).

Firstly, research presented in Chapter 6 focused only on the UTI and no other aspects of green infrastructure. However, it is argued that, due to the presence of canopy cover throughout the study landscape, the UTI is most likely present throughout all urban green spaces (minus monoculture fields, green roofs, and open water). Secondly, although there are suites of associated ecosystem services generated by urban trees none were explicitly identified in Chapter 6 as this was not the research aim. Rather, research was guided by the position that pattern affects process. Therefore, it was believed that the presence of structurally diverse and structurally connected canopy patches provided sufficient amounts of ecosystem services. Thirdly, as a landscape approach was adopted only structural diversity and UTI landscape patterns were used to describe ecological health (in regards to UTI). Finally, and in regards to the public health section of Figure 6-13, only socio-economic health was considered. Consequently, more work needs to be done to reveal if there are any further associative links between public health and the UTI (i.e. UTI landscape patterns with community, physical, and psychological health). However, clear relationships have been identified between Salford's UTI and societal structure which can be used to support the assertion that green infrastructure and public health are strongly linked to an extent that relatable structural patterns can be identified at the landscape scale.

When the IMD map presented in Figure 6-3 is considered alongside the results of Chapter 4 and 5 further structural relationships with area deprivation emerge. The river valley study area is located in the east of Salford and contains all four of the IMD quartile areas (i.e. LSOAs are either most deprived, deprived, less advantaged, or advantaged). To the north of the river valley study area exist the more advantaged LSOAs (advantaged in the north, less advantaged to the north west and a single advantaged LSOA to the north east) while the southern part of the study area contains the more deprived LSOAs (most deprived to the east and west and deprived to the north east). Interestingly, the largest keystone canopy patches for maintaining connectivity and the resulting sub-networks also exist to the north of the river valley study area, within the less deprived, more advantaged LSOAs. However, sub-networks do exist to the east and south of the river valley study area, in the most deprived LSOAs, they are much smaller than the one/s in the 'advantaged-north' LSOAs. In regards to localised connectivity, the low canopy cover sample plots of Lower Kersal and Broughton also exist in the most deprived LSOAs and exhibit extremely low levels of connectivity. In comparison, the well connected, high canopy cover sample plot of Kersal exists in the more advantaged LSOAs. The Peel sample plot which exhibits high levels of connectivity and canopy cover does not follow this pattern as it exists in the 'southern-deprived' LSOAs. However, the sample plot comprises not only trees from public land (i.e. Peel Park) but also from private land owned by the University of Salford. Therefore, the sample plot could be considered isolated from the effects of the surrounding deprivation. Therefore, in summary it seems as if the local connectivity and UTI resilience of the river valley study area is relatable to area deprivation. The river valley study area could therefore be used as a future case study to assess if this relationship is significant. If it is found to be so then another potential research output from this thesis is to assess the relationship between area deprivation, landscape connectivity, and system resilience (using a landscape graph and network analysis approach) across the whole of Salford.

6.4.3 Greenspace management implications

The use of and local residents concerns over greenspace has been shown to be a factor in what is perceived to be high-quality maintenance and aesthetics (Tzoulas & James, 2010). Furthermore, residents are also the best placed to recognise the benefits provided by good quality greenspace and what aspects of quality are of most importance (CABE, 2010b). However, such an anthropomorphic approach can neglect the importance of ecological quality, as urbanites are often unaware of the ecological processes (such as habitat structure) underpinning the provision of the services they benefit from (Escobedo *et al.*, 2011). The findings emerging from Chapter 6 can inform large and small scale greenspace planning, in regards to the UTI, with focus being on the structural aspects of canopy patches.

In particular, the least deprived areas have higher canopy cover than the more deprived areas, however the canopy patches are significantly more fragmented. Therefore, greenspace managers targeting the maintenance of the UTI in such areas should focus their efforts on reducing this fragmentation. Conversely, the more deprived areas exhibit significantly more structurally connected canopy patches and therefore focus on UTI improvement should be on increasing the over canopy cover. Targeted UTI maintenance/improvement in regards to structural diversity (as described by canopy height diversity) is independent of social deprivation and should be undertaken wherever necessary.

A potential green network has also been identified running fairly laterally through Salford (Figure 6-3). This green network is composed of LSOAs exhibiting higher canopy cover (~30-45%) than other areas in Salford. However, highly deprived LSOAs within Irwell riverside and Broughton (two of the three wards that make up the river valley study area) break this potential network as they contain lower levels of canopy cover (5-30%). Therefore, if a Salford green network were to be created the tree stock of these areas would first have to be increased. Future research could then be targeted towards the green network footprint.

6.5 Conclusion

Social-ecological inequality does exist in Salford. However, the relationship between societal and ecological landscape patterns is multifaceted. An environmental justice issue has been raised as higher deprived areas contain lower canopy cover; Salford exhibits an ecological deficit. However, fragmentation is lower in more deprived areas and therefore so is structural connectivity. This infers that the canopy patches within these areas may be lacking in quantity but not necessarily in quality. Therefore, the relationship between area deprivation and landscape composition is not necessarily the same as area deprivation and configuration. Salford also contains a homogenous UTI in terms of height diversity and as such intra-patch structure is independent of social structure.

Chapter 7 : General Discussion

Through an explorative, multi-scale approach the research presented in this thesis has enabled the systematic description and understanding of the structural patterns emerging from an Urban Tree Infrastructure (UTI) and of how these patterns relate to the provision of functional habitat, UTI system resilience, and to IMD landscape patterns. The research aim set out in Chapter 3 (Figure 3-1) has been successfully achieved by means of effectively resolving the four research questions (Figure 7-1). Although the study UTI existed in Salford, UK, the methods used are transferable to other cities. Furthermore, although the results gleaned are spatially constrained to Salford they reveal patterns, thresholds, and relationship which can be tested using other UTIs from differing cities.

Research question 1 has been responded to by effectively quantifying Salford's UTI, at differing spatial scales, via the application of a suite of spatial data pre-processing and management methods upon both remotely sensed and field survey data (Chapter 4). The UTI has also been described as a landscape graph in order to answer research questions 2 and 3 (Chapter 4). Through using a graph theoretic and habitat availability based approach towards measuring landscape connectivity, the gap-crossing abilities of passerines were incorporated into a three dimensional landscape connectivity model (Chapter 4). As a result, research question 2 has been sufficiently answered and the potential modelled functional connectivity of Salford's UTI, from 2005 to 2013, is low – specifically the canopy ≥17.1m (Figure 7-1). Additional patterns and relationships have also been identified due to the thesis' explorative approach. The greatest increase in connectivity occurs as gap-crossing capability threshold increases from 30m to 60m. In addition, after a 90-120m gap-crossing threshold, the willingness or ability to cross gaps has less influence on the overall level of potential landscape connectivity (Figure 7-1). An increase in gap-crossing capability threshold is also most effectual for the connectivity of the canopy ≥17.1m (Figure 7-1).

Arising from this graph theoretic and habitat availability approach the key, central canopy patches of Salford's UTI have been identified (Chapter 5). The importance of each individual canopy patch was identified by determining the impact removing that patch would have on potential landscape connectivity (Chapter 5). Canopy patch importance was calculated for each canopy layer using the minimum (30m) and maximum (200m) gap-crossing capability thresholds (200m) (Chapter 5). Using network analysis it was revealed that the system resilience of Salford's river valley study area's UTI is reliant on only a few sub-connected regions due to low central canopy patch density (Figure 7-1). As a result, the UTI system represents a scale-free network – resilient to random attacks and disturbances to the overall system but vulnerable to system collapse when central patches are attacked, disturbed, or removed (Figure 7-1). The central, keystone canopy patches also form sub-networks representing landscape hubs; hence the UTI system is also a highly compartmentalised network (Figure 7-1). As a result, a potential disturbance within one landscape hub would not necessarily affect the sub-networks of the other hubs. Consequently, it can be concluded that research question 3 has been resolved (Figure 7-1). In addition, further insights into the UTI system have also been identified. For example, in the majority, the sub-networks are dependent on a single central patch which, if removed, would cause the landscape hub to dissipate (Figure 7-1). In addition, the actual amount of identified central patches and the location of the landscape hubs are dependent on the canopy heights and passerine perception values incorporated into the creation of the landscape graphs (Figure 7-1). Finally, the largest sub-networks also exist in the least deprived areas of the river valley study area (Figure 7-1).

Research was undertaken at the city scale to answer research question 4. An assessment of the relationship between UTI composition and area deprivation successfully reveals that socio-ecological inequality, leading to potential environmental injustice, is present across Salford (Figure 7-1). This result supports the findings of previous studies which have focused on both the UTI and greenspaces in general (Barbosa *et al.*, 2007; Taylor *et al.*, 2007; Zhou & Kim, 2013). More detailed analysis of UTI configuration and three dimensional structures reveal that in more deprived areas, where habitat fragmentation is influential to organism survival, habitat contiguity is higher than less deprived areas. On the other hand, the structural diversity of canopies has no statistical relationship with area deprivation (Figure 7-1).

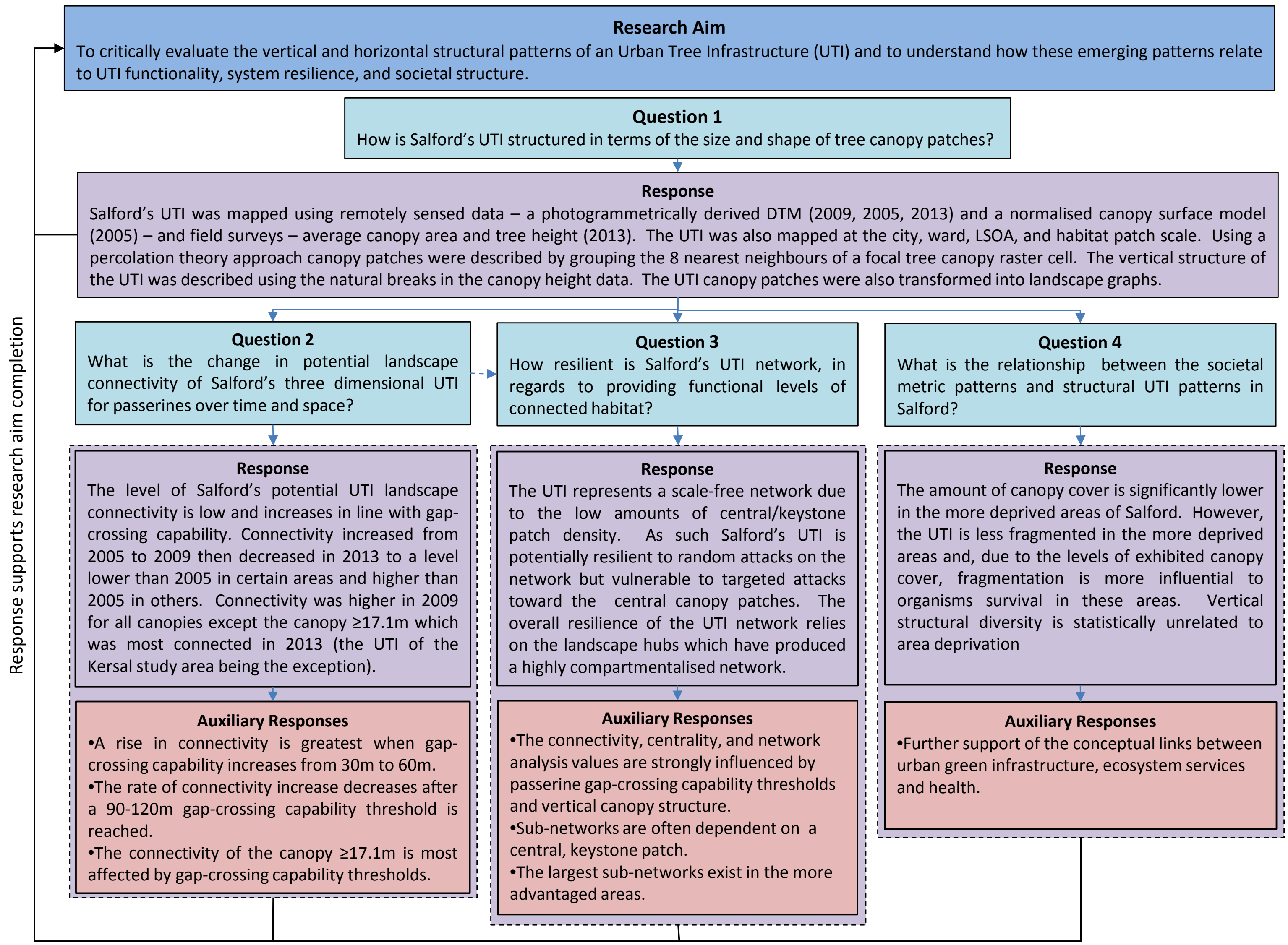

Figure 7-1: Outline of the research question responses and auxiliary responses supporting the completion of the research aim.

To achieve the research aim (dark blue) four research questions were answered (light blue). The responses (purple) and auxiliary responses (pink) to these questions support the completion of the research aim. Response to question 1 underpins the other research question responses.

These original findings can be used to both enhance the current knowledge on UTI social-ecological function as well as open interesting avenues of discussion; specifically how UTIs, which are forever in a state of flux, should be mapped and used to inform effective greenspace/green infrastructure planning, enhancement, and maintenance. Nevertheless, although the research questions have been answered and the research aim completed there are still research issues which need to be addressed. Research limitations, methodological influence, potential oversights, and future avenues of research are therefore discussed in the following sections.

7.1 Mapping the UTI

The UTI was vertically stratified based on the natural breaks within the 2009 height data, as these were the original height data (Chapter 3). It has been argued that the heights chosen to stratify tree canopies have often been arbitrary and not necessarily transferable – as the assignment of a tree to a canopy layer category (i.e. overstorey, understorey, shrub layer etc.) is relative to the neighbouring trees in which it shares a patch and the scale of the height class categories (Parker & Brown, 2000; McElhinny *et al.*, 2005). The seminal MacArthur & MacArthur (1961) paper is an exception to this as the heights used to define canopy layers were related to the observed diversity of birds within those layers, thus stratification and structure were related to function. However, both these points relate to mapping distinct vertical strata *within* canopy patches as well as being based on leaf mass as opposed to the alternative options of stratifying canopies based on species composition or individual trees (Smith, 1973).

The landscape scale approach of the research presented in this thesis meant that using species composition to stratify canopies was unviable. Likewise, the original 2009 canopy data did not include intra-patch canopy layers, thus the three dimensional structure beneath the canopy ≥17.1m canopy was unknown. Consequently, vertical stratification based on vegetation mass or distribution of foliage in disbursed vertical strata (Parker & Brown, 2000) was impossible. Therefore, the *accumulated* vertical distribution of UTI foliage was mapped. In other words, the trees of the upper most canopy were vertically stratified and mapped (n.b. some polygons representing a single canopy were in fact made up of more than one individual tree. However, the canopies were so close together and of the same height that it was deemed a single canopy; Chapter 3). In addition, natural breaks were used to define vertical strata rather than the height categories set out by MacArthur & MacArthur (1961) as those categories are non-transferable since vegetation in the field, shrub, or understorey layers were not being mapped for the thesis research but rather the vertical spatial arrangement of overstorey trees. Therefore, identifying the natural breaks in the height data was the least arbitrary and most useful and transferable method of stratification. In summary, the adopted UTI descriptive/mapping methods combine canopy cover mapping with accumulative vertical stratification plus relate the spatial arrangement of the amount of vertical canopy cover to ecological function (e.g. connectivity, habitat availability, system resilience) and local environment (e.g. societal distribution). Through using such an approach a better understanding of the canopy is obtained (Parker & Brown, 2000).

However, the arbitrariness of the UTI stratification methods is exposed as the natural breaks identified in the 2009 UTI data are transferred to the 2005 and 2013 data (Chapter 4). The distribution of heights in the study areas are likely to vary over time, therefore, for example, the canopy ≥17.1m canopy of 2009 was not likely to be the same as the canopy ≥17.1m canopy in 2005. In other words, the natural breaks in the height data would change, thus the spatial arrangement of the Natural Break Height (NBH) canopies would change (the 3m minimum being the exception). It was decided that all UTIs (2005, 2009, and 2013) would use the 2009 NBH categorisation so that comparisons in potential landscape connectivity could be made and any obvious patterns of change identified (Chapter 3 and 4). For this reason the NBH categorisation can be justified, however they are not necessarily transferable to other city's UTIs and therefore any future studies should undertake the methods outlined in Chapter 3 in order to identify their own specific natural breaks within their own UTI.

The methods used in the creation of the NBH canopy patches were primarily based in percolation theory; using the structural, physical connections between individual canopies to define neighbourhood and patch dimensions (Chapter 3). An alternative approach would have been to describe the canopy patches from a more perceptual or functional perspective, using approaches derived from the eco-field paradigm or the concept of umwelt (von Uexküll, 1926; Farina & Belgrano, 2004; Manning, 2004; Farina & Belgrano, 2006). The Umwelt is the 'self-world' of an organism, it refers to how an organism perceives and uses the landscape rather than the physical components (i.e. habitat) of that environment as perceived by humans (Manning, 2004). Similarly, the eco-field paradigm moves away from the landscape being defined as a neutral matrix, in which organisms exist, towards the landscape being borne from a combination of an organism's perceptions, altered by its functional traits, and vegetation/habitat cover; thus the landscape is a cognitive construct but not only 'a product of the human mind' (Farina & Belgrano, 2004, p. 107). Therefore, defining the Salford landscape and its UTI canopy patches from this approach would be extremely complex and species, if not individual, time, and space specific.

The creation of a perceptual patch – which included the proportion of canopy cover within a perceptual threshold buffer in the mapping of actual canopy cover – was experimented with while developing the research methods. Such an approach would improve cost-distance analysis as movement is related to perception and levels of habitat cover. These purposed perceptual patches would also need to incorporate data on organism movement through variations in proportional canopy cover. As the research aim was to describe the structure of the UTI and relate patterns to function rather than to create habitat suitability maps or assess population dynamics this approach to describing habitat, or rather canopy patches was unnecessary. Furthermore, the complexities of creating a perceptual patch are greater than simply describing proportional habitat within a perceptual threshold (e.g. functional traits over time and situation, and perception of matrix). Therefore, the creation of a perceptual patch presents an avenue for future research which needs more study.

Finally, both 'cognitive landscape' concepts can be used to argue that the human perception of landscape is not necessarily the same as those for other organisms and, therefore, physically mapping the landscape is inherently futile as we cannot effectively describe the landscape from a focal organism's perspective. However, incorporating highly specific organism perspectives into habitat mapping would also mean highly species – if not individual – specific results; this contrasts with the more integrated ecosystem approach that promotes a move away from concerns over species and area protection towards ecosystem protection and integrated land use/land cover planning (Hartje *et al.*, 2003). It could also be argued that the urban landscape is a human construct: the anthropocentric habitat. Therefore, mapping its components (e.g. habitat patches) can only come from a human perspective, for it is the human that will use the resultant map to alter, enhance, and/or create new components. After all, any human action in such an environment is done, essentially, for human benefit; it is a social-ecological system with the accentuation positioned on the *social*. As once stated -

> ''The picture we get throughout is of a world that seems created exclusively for this animal. And so we are justified in assuming that there are as many surrounding worlds as there are animals''
> (von Uexküll, 1926, p. 176).

Though this may be the case, we can only effectively describe one – ours.

7.2. Connectivity and system resilience

Through the analysis of spatial heterogeneity the relationships between ecological processes and spatial patterns can be explained (Turner, 1989, 1990). Landscape structure can influence organism behaviour and by doing so generate spatial patterns at both the habitat patch and landscape scales (Bélisle, 2005). Research on the heterogeneity of Salford's landscape structure has focused on the UTI canopy patches, evaluated through the lens of landscape connectivity (Chapter 4). Landscape connectivity involves an acknowledgement and understanding of both the spatial and structural components of the landscape as well as the behavioural attributes of organisms, hence it has two theoretical components; structural and functional connectivity (Tischendorf & Fahrig, 2000b). The structural aspect of Salford's UTI has been described at both the horizontal and vertical scale and transformed into landscape graphs. Passerine perceptual thresholds – indicated by recorded gap-crossing capability thresholds – were used to describe functional connectivity. By integrating these two aspects of connectivity into a model, the overall levels of potential landscape connectivity have been calculated using the integral index of connectivity (IIC) (Chapter 4).

In terms of the graph theoretic landscape connectivity models used in the research, pertinent findings have shown that both the third dimension and the variability in organism perception should not be overlooked when considering landscape connectivity (Chapter 4). However, calculation of connectivity could be more species specific to reveal even more accurate connectivity results (e.g. use of the probability of connectivity – PC – to describe connectivity using species movement probabilities). On the other hand, as discussed in section 7.1, any meaningful findings would be biased to that focal species. Therefore, the five emerging key outcomes of the research presented in Chapter 4 are more widely applicable. These outcomes are: 1) urban connectivity is low, especially for the uppermost canopy, 2) temporal landscape connectivity patterns are stochastic, 3) an increase in gap-crossing capability threshold from 30m to 60m has the greatest impact on connectivity, 4) the impact of gap-crossing capability threshold decreases after 90m-120m threshold, and 5) the connectivity of the uppermost canopy is most effected by gap-crossing capability thresholds. From these research outcomes a question arises - *How can this new understating be applied to a social-ecological system at a landscape scale*?

To answer this, future research would first need to understand what spatial criteria or structural landscape designs are required to create functional, healthy systems. In other words, *how should the urban landscapes be organised to ensure healthy urban ecosystems*? In regards to UTI research, this question can be reduced to *how should the UTI be designed to ensure healthy functioning?* Research asking this question can use the occurring 90m -120m gap-crossing-connectivity threshold, as a UTI management parameter, and investigate the threshold's impact on UTI spatial patterns. This threshold can be viewed as an important managerial device for the enhancement of UTIs. Although increasing the entire UTI within an urban landscape would increase connectivity, this cannot always happen due to high densities of buildings, sealed surfaces, and other urban infrastructures. However, by using the connectivity threshold, landscape connectivity can be best enhanced by identifying and closing canopy gaps that are less than 90m-120m. A landscape connectivity model employing the 90m-120m threshold can be compared to a random connectivity model (i.e. a control model). Specifically, hypothetical canopy patch creation can be targeted towards specific locations – sensitive to the surrounding urban environment – in order to fill 90m-120m gaps. The connectivity and system resilience models set out in Chapter 4 and 5 can then be run and their results compared to a landscape map which has had the same number of canopy patches added randomly (i.e. not targeted to the 90-120m parameter). Furthermore, the 90m-120m model can also be compared to a landscape model in which only the size of important canopy patches are increased (as identified in chapter 5). By replacing the 90m-120m parameter with one which targets reducing the number of 60m gaps, as the greatest increase in connectivity was found when gap-crossing capability threshold increased from 30m to 60m, another connectivity model comparison can be made. It would be interesting to see the outcome of such research so that the findings in this thesis can be justified or opposed.

An understanding of the state of a system can be derived from the values of the variables that constitute the system (Walker *et al.*, 2004). Specifically, the structure of a network can provide insights into the functioning of the system that it represents (Strogatz, 2001). The structure of Salford's UTI system has been investigated by describing it as a landscape graph and calculating the values of key structural variables (Chapter 5). Through the research presented in this thesis, UTI landscape connectivity and levels of centrality have been described and spatial patterns have been related to system resilience. The principal landscape components have been identified using transferable methods and as a result other UTIs, exhibiting different landscape patterns to that of Salford, can be similarly assessed. A future comparable study can subsequently occur; an evaluation of how diverse patterns of landscape design can affect the connectivity, function, and resilience of UTIs. Furthermore, the research presented in Chapter 5 has resulted in the identification of target study areas; highlighting a future opportunity to collect further ecological data from the identified sub-networks within the river valley study area.

A potential weakness of the presented research is that the majority of gap-crossing capability thresholds are based on mobbing behaviour; thus threat perception drove bird movement (e.g. Desrochers & Hannon, 1997; Creegan & Osborne, 2005; Tremblay & St. Clair, 2009). Consequently perception of canopy gaps and the probability of a bird to cross them are influenced by a particular functional trait – a need to defend from predators. In other words, the gap-crossing capability thresholds used in the connectivity and network analysis models were derived from a single, specific behaviour, thus the landscape was perceived through a single cognitive field (Farina & Belgrano, 2004). Other behaviours (i.e. foraging, mating, avoiding disturbances) may alter the perception of canopy gaps and, in turn, the distance a passerine is capable of moving. This highlights the complexities involved in integrating organism perception into landscape connectivity analysis. Movement ability is not only underpinned by the physiological nature of an organism (e.g. Zollner & Lima, 2005; Hodgson *et al.*, 2007; MacIntosh *et al.*, 2011) but different behaviour will alter perception of the landscape and therefore movement (Harris & Reed, 2002; Bélisle, 2005). Furthermore, as perception is an individual phenomenon, idiosyncratic behaviours cause more complexities.

Farina and Belgrano (2004, 2006) state that physical descriptions of the landscape and species behavioural integration into connectivity models is a redundant approach – as the landscape perceived by species differ from each other, between individual organisms, and from the anthropocentric perception of landscapes and landscape components. Landscape managers, community groups, and other greenspace practitioners cannot alter perception however, yet they can influence, enhance, and even diminish the physical landscape. The work undertaken by such landscape managers can only try to increase or maintain *potential* landscape connectivity – as *true connectivity* (due to the functional connectivity concept – i.e. incorporating behaviour and perception) can never really be known. In order to achieve this, potential connectivity models should reveal useful thresholds and landscape patterns which can be used to inform best practice. Although different behaviours may have resulted in different maximum gap-crossing capability thresholds, it was also demonstrated that the rate of increase in connectivity began to decrease after 90m-120m. Even if larger gap-crossing capability thresholds were identified, which was the case for Piciformes (Grubb & Doherty, 1999) and individual wood thrushes (MacIntosh *et al.*, 2011) it is unlikely that this threshold would change (see connectivity curves in sections 4.3.1 to 4.3.5).

Another limitation to the connectivity analysis is that the potential landscape connectivity model uses a binary landscape to quantify connectivity; hence the matrix was not mapped or considered. Studies have shown that both anthropogenic and natural linear features (e.g. roads and rivers) can impede bird movements: an affect ameliorated by well-managed, feature adjacent canopies (Tremblay & St. Clair, 2009, 2011). In contrast, a study on the movement responses of birds to different housing densities established that not only do birds move from canopy edges into the matrix, there is no significant difference in the total number of crossings between two different matrix types – high and low density housing (Hodgson *et al.*, 2007). Harris & Reed (2002) argue that gaps in habitat act as behavioural, rather than physical barriers to movement. In addition, avoidance of roads by birds is more likely a function of low quality nearby habitat rather than reticence of movement (Reijnen, 1997). Furthermore, in terrestrial environments the matrix has been shown to be rarely hostile and the perception and usage of the matrix will depend on the organism (Manning, 2004). As passerines are the focal organism within this thesis (and gap-crossing capability threshold taken from a variety of species) specific matrix use was unviable. However, further small scale studies (within the areas critical for maintaining connectivity) can be undertaken in the future in order to assess the effect of Salford's urban matrix on the gap-crossing capability thresholds of specific species. By describing the UTI as a graph its links can then be removed based on a resistance value comprising response to physical and perceptual barriers. This would also provide an alternative to the least cost approach to measuring connectivity.

7.3. The relationship between UTI and social structure

Societal decisions and cultural perceptions greatly affect the urban fabric, urban landscape, and urban quality of life and therefore these decisions and perceptions should be formed from comprehensive information (Jarvis & Young, 2005). The social-ecological issues arising from urban areas are due to its complex and unique social and environmental characteristics (Gill *et al.*, 2008). To understand and effectively manage these systemic issues the social- ecological nature and resilience of urban areas needs to be better understood.

The current literature on the benefits of urban greenspace, green infrastructure, and specifically urban canopy cover states that the most deprived of a given population can benefit more from urban ecosystem services then less deprived areas (section 2.1 and Chapter 6). Less deprived residents are more likely to reside in areas with higher quality as well as quantity greenspace (Matsuoka & Kaplan, 2008) and even when accessibility to greenspace has been found to be better in more deprived areas the negative perceptions of the residents towards greenspaces has meant they are less likely to use them (Jones *et al.*, 2009). However, the results in Chapter 6 demonstrate that structural diversity (an aspect of habitat quality and function) does not statistically relate to area deprivation. An increase in deprivation does follow an increase in canopy height diversity (CHD) so that the more deprived areas of Salford contain higher UTI structural diversity, but the relationship is not statistically significant. The strength of this relationship may increase in the future through the planting projects and other UTI projects from organisation such as the Red Rose Forest. The upward trend of CHD with deprivation (although not significant) could be due to a result of planning policies targeting more deprived areas and increasing the quality of greenspace through the implementation and/or maintenance of structurally diverse canopy patches.

Furthermore, in more fragmented environments the three-dimensional structure of habitat patches is potentially more influential on passerine abundance than in more contiguous environments (Goldstein *et al.,* 1986). However, the deprived areas of Salford are less fragmented than the more advantaged areas meaning levels of CHD would be more beneficial to organisms, for example bird populations, in the more advantaged areas. If this line of logic was followed then, hypothetically, landscape managers would focus on increasing CHD in the more advantaged areas which would in turn increase habitat quality, potentially increase bird population and therefore higher greenspace quality. As these areas are already more 'green' than the deprived areas, such an approach would prove to be folly and any improvements should be focused on the more deprived areas of Salford. However, problems also lie in such an approach. For instance, increasing canopy cover in the deprived areas of Salford can improve an inhabitant's ability to interact with and benefit from nature. However, 'greening' areas has also had negative effects due to gentrification and the displacement of the very residents in need of green spaces (Wolch *et al.*, 2014).

In terms of the social aspects of green space, the research presented in Chapter 6 adds to the body of evidence that cities exhibit environmental injustice in terms of the varying amount of canopy cover within differing areas of deprivation. However, the relationship between on-the-ground conservation and design work and overall landscape composition and configuration it affects is still to be articulated. The canopy patches mapped using raster cell neighbourhood effects and region group were representative of Salford's UTI. Aspects of this UTI's landscape composition and configuration were quantified and then statistically related to area deprivation in order to understand the relationship between UTI landscape patterns and societal landscape patterns. Salford revealed an aspect of social-environmental inequality as the amount of canopy cover (%) decreased with deprivation. This means that the most deprived LSOAs of Salford have less canopy cover and potentially less access to the ecosystem services provided by urban trees. The levels of canopy cover within the more deprived areas were also at a level at which habitat fragmentation becomes influential to species survival (i.e. <10%). Interestingly, although the number of canopy patches decreased with deprivation, the structural connectivity (CONTAG and patch density) increased, thus fragmentation decreased, with deprivation. In other words, in the areas of Salford where fragmentation is theoretically influential it is less than the areas where fragmentation is of secondary importance to habitat cover. Therefore, the nature of the apparent social-environmental inequality within Salford becomes more complex as UTI configuration is assessed along with UTI composition. In this regard, UTI quantity does not necessarily mean UTI structural quality.

Further research questions arise from these findings, these being; 1) are such patterns a consequence of planning (both intentional and non-intentional) or ecological self-organisation? 2) Can we correctly assume that certain urban landscape patterns are related to either biological or cultural selective forces or are the relationship between the two selective forces too complex? 3) Are the larger canopy patches self-organised while the smaller patches fully reliant on cultural selective forces for their survival (maintenance, planting policies etc.)? Evidently, future avenues of research have developed from the research presented in Chapter 6.

To explore the questions here briefly, a quote from Norberg (1999) is presented:

> "The ability of biological systems to follow [the] path of succession is of utmost importance for the ability of ecosystems to perform certain functions and thus to provide goods and services. To sustain healthy ecosystems, one has to understand what criteria are needed for biological organization to take place" (p 199).

The concept of biological organisation referred to by Norberg (1999) does not consider the influence of social criteria and cultural selective forces on biological organisation (Figure 7-2). The ecological selective processes require continuous inputs of energy (such as solar energy in the case of ecosystems) (Norberg, 1999) and similarly cultural selective forces require input from people and social infrastructure (Wu, 2014). The future research questions presented above are basically concerned with which inputs of energy are more influential to the observed landscape patterns observed in Chapter 6. For example, there is a significant relationship between social deprivation and UTI configuration and composition and the location of connectivity landscape hubs closely follow administrative boundaries (e.g. wards, Chapter 5). Therefore non stochastic patterns emerge from ecological and cultural organisation (through ecological and cultural selective forces, section 2.1). In ecological thinking, non stochastic patterns represent a move away from organisational disorder (Nordberg, 1999). Accordingly, the two selective forces (ecological and cultural) work together to maintain order. For example, ecological selective inputs need energy to function and landscape fragmentation may reduce the movement of energy through loss in connectivity. Cultural selective forces are then needed to reduce disorder by increasing connectivity yet not only at a local scale (which seems to be the case from the results in Chapter 4, 5, and 6) but organisation needs to be applied on a landscape scale.

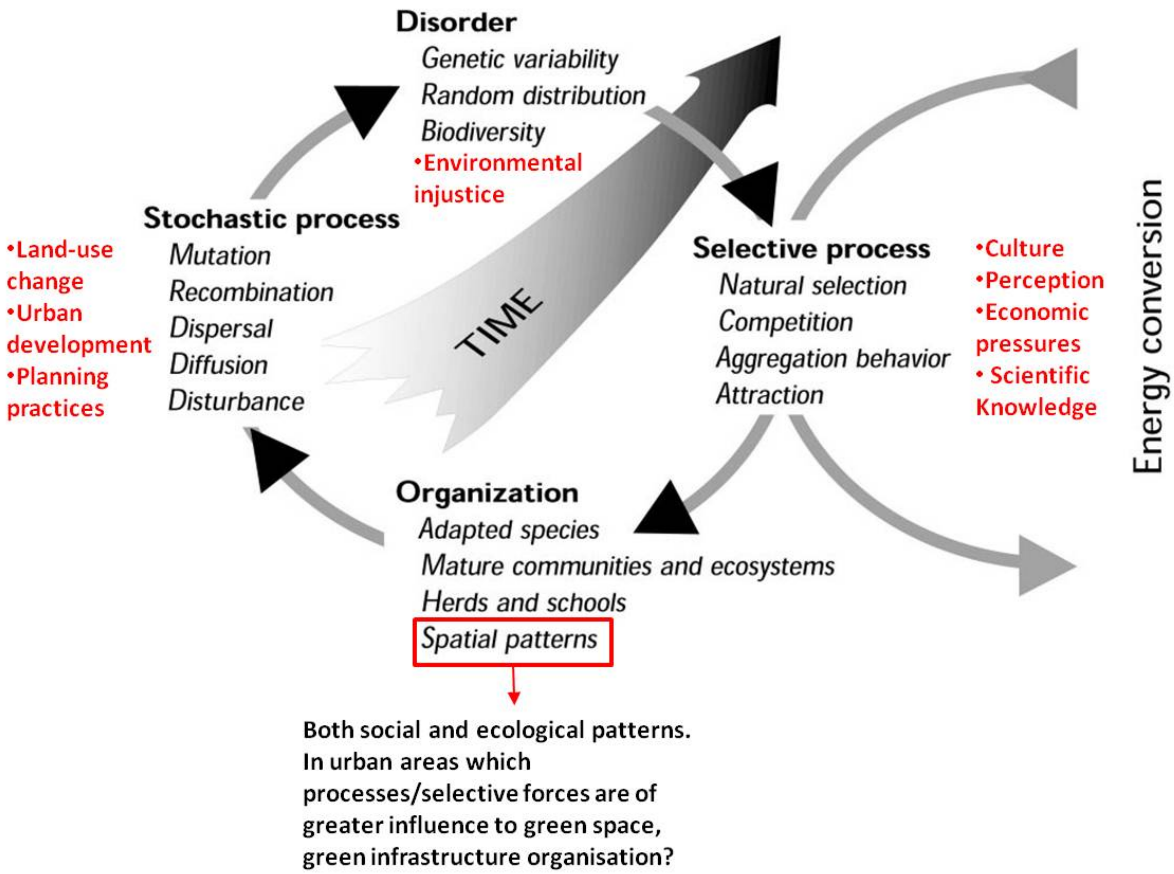

Figure 7-2: Biological organisation and the influence of cultural selective forces (based on Norberg, 1999). Biological organisation occurs over time by consuming energy. However, for organisation to occur disorder must happen. Organisation can be recognised at the landscape scale due to spatial patterns. A disturbance (stochastic process) may cause disorder (reduction of biodiversity). An ecological selective process will then occur, for example natural selection, which will lead back to organisation and potentially a new spatial pattern. However, in a social-ecological system (such as the UTI) these ecological processes and selective forces do not act in isolation. Therefore, cultural selective forces and process (red text) also influence biological organisation.

The Landscape Institute, 2009 recognised that;

> "There is still a widespread lack of awareness of how important [green infrastructure] assets are, demonstrated by the frequent failure to plan, design and manage them appropriately. Natural assets are often seen as separate entities ...this approach fails to recognise the symbiosis between the quality and connectivity of natural assets with local environmental and economic performance. Overcoming this failure ... relies on an understanding that these

> functions are multiplied and enhanced significantly when the natural environment is planned and managed as an integrated whole; a managed network of green spaces, habitats and places" (p 1)

With regards to the research presented in Chapter 6, a potential green network in Salford was identified (Figure 7-3). Further research which utilised a graph theory and network approach could also investigate the proficiency of this potential green network. Further data can also be incorporated into the green network assessment, such as local greenspace use (e.g. Moseley *et al.*, 2013), species diversity and abundance data, as well as organisms' movement probabilities. Thus a social and ecological connectivity analysis of the Salford UTI green network could be undertaken; resolving the issues identified by the Landscape Institute (2009). Furthermore, as social distribution has a statistically significant relationship with UTI pattern and structure, the results in this thesis can provide guidance for targeted landscape management. For example, in more deprived areas focus should be targeted towards increasing the amount of trees while maintaining or even increasing the higher structural connectivity values (relative to less deprived areas). On the other hand, work in the more advantaged areas should focus on reducing fragmentation as to create potentially higher functioning habitat. However, with gentrification, policy changes, life style changes, etc. deprivation values of Salford's SLOAs may change over time, thus current social patterns and the relationships with UTI patterns will change. However, targeted work which focuses on a specified UTI landscape composition and configuration, based on deprivation, will result in a more homogenous landscape pattern; potentially reducing social-ecological inequality. In other words, it would be a positive outcome if no relationship existed between UTI patterns and area deprivation as a result of the amount and structure of tree canopies becoming spatially equal across the city.

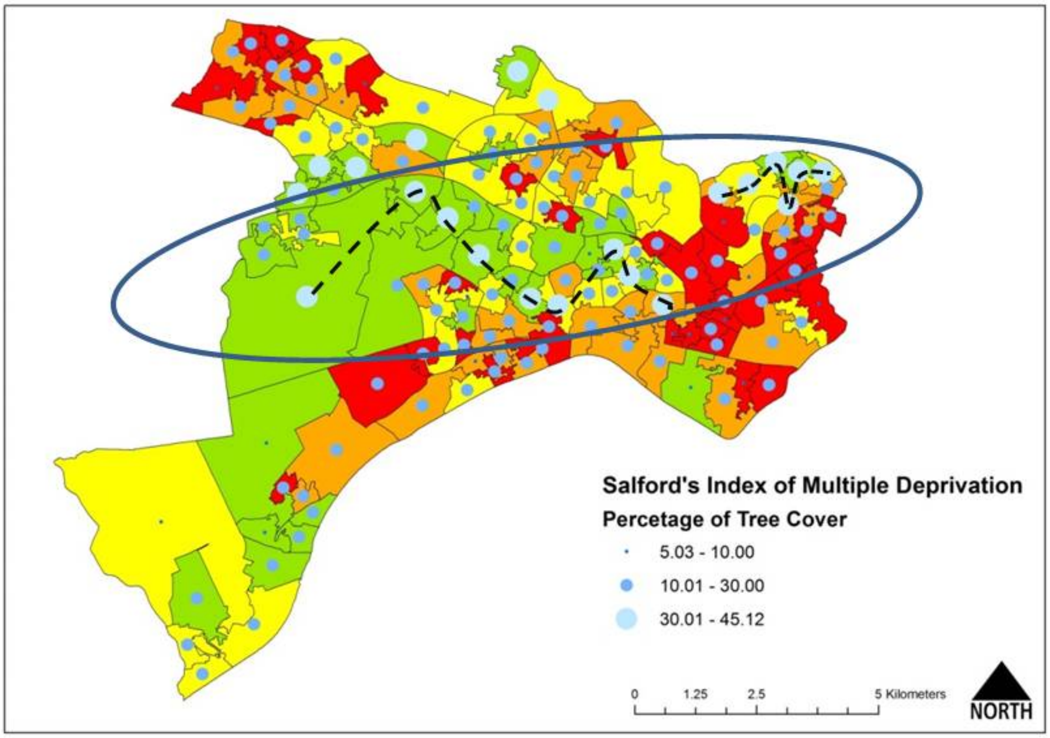

Figure 7-3: Potential green network for future studies, identified by proportion of canopy cover in LSOAs. The higher levels of canopy cover (30-45%) within Salford form to create a potential green network moving laterally across the city. This network is broken in the deprived LSOAs of the Irwell riverside and Broughton wards. Further work to assess the viability of creating this green network is needed.

As a final note, the benefits of the UTI are seen as more prevalent than the associative problems (Lohr *et al.*, 2004). However while collecting field data a total of four local residents left their house to actively complain about the trees in their areas (personal observation). These complaints ranged from "are you from the council? I have been asking for these trees to be cut back," to "are you finally here to remove these trees?" and even "trees don't belong in an urban setting" (personal observation). These comments were all made in the Kersal sample plot, the least deprived of all the tree canopy sample plots (classified as 'advantaged' in Figure 6-12) and exhibit high levels of canopy cover. No such comments were made in any of the other sample plots.

7.4. Concluding Remarks

The multi-scale hierarchical view of the landscape presented in this thesis, represented by individual trees, percolating canopy patches, and landscape components, across vertical space has allowed for a holistic analysis of the urban tree infrastructure. The vertical spatial arrangement of the canopies and gap-crossing capability thresholds of the organisms under study affect how the UTI is structurally perceived. In turn, the professed function and resilience of the UTI is a product of both anthropogenic and organismal perception – a point which should be kept in mind when studying other urban green infrastructures. Through the research presented in this thesis it has been shown that UTI connectivity is low and can be improved by targeting gaps 30-60m or 90-120m in distance. The connectivity of the upper-most canopy, while being lower than any other canopy, is most affected by a passerine's gap-crossing capability. The Salford UTI represents a scale free network, resilient to random attacks on canopy patches but not to targeted attack on keystone canopy patches. The overall resilience of the Salford UTI is dependent on only a few locally connected canopy patches that form sub-networks. The more deprived population of Salford live in areas with less canopy cover that is less fragmented than the greater canopy cover in the less deprived areas. To understand the UTI is to therefore comprehend spatial and dimensional landscape and patch scale structure, an aspect of organismal perception, and social-ecological association.

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Appendix A) 2013 Tree Survey Results

Appendix Table 1: Higher Broughton tree canopy sample plot results.

ID	Location	Species	Area 09 (m²)	Height 09 (m)	Distance (m)	Angle (°)	Radians	Eye Level (m)	Height 13 (m)	Radius 1 (m)	Radius 2 (m)	Average (m)	Area 13 (m²)	Height Δ (m)	Area Δ (m²)
2188415	Park	Carpinus	69.60	12.19	15	38	0.66	1.75	13.47	5.0	5.5	5.25	86.59	1.28	16.99
2188416	Park	Fraxinus (1 X Betula Removed)	235.20	15.08	18	38	0.66	1.75	15.81	6.5	8.0	7.25	165.13	0.73	-70.07
2188417	Park	Sorbus x intermedia	46.76	7.42	10	41	0.72	1.75	10.44	3.0	4.0	3.50	38.48	3.02	-8.28
2188418	Park	Fraxinus	98.50	15.70	20	34	0.59	1.75	15.24	6.5	7.0	6.75	143.14	-0.46	44.64
2188419	Park	Laburnum	114.36	8.55	15	30	0.52	1.75	10.41	7.5	7.5	7.50	176.71	1.86	62.35
2188422	Park	Fraxinus	126.26	15.00	10	62	1.08	1.75	20.56	6.5	6.5	6.50	132.73	5.56	6.47
2188425	Park	Prunus	58.95	6.20	10	39	0.68	1.75	9.85	7.0	3.0	5.00	78.54	3.65	19.59
2188427	Park	Betula	39.33	11.96	10	52	0.91	1.75	14.55	4.5	4.0	4.25	56.75	2.59	17.42
2188440	Park	Metasequoia	87.33	14.84	10	54	0.94	1.75	15.51	4.0	4.0	4.00	50.27	0.67	-37.06
2188927	Public/Street Tree	Removed	42.22	10.31	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-10.31	-42.22
2188941	Public/Street Tree	Removed	58.03	10.79	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-10.79	-58.03
2188951	Private	Fraxinus	195.89	14.75	15	47	0.82	1.75	17.84	6.0	6.0	6.00	113.10	3.09	-82.79
2188954	Public/Street Tree	Removed	18.11	6.90	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-6.90	-18.11
2188955	Public/Street Tree	Removed	19.32	7.94	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-7.94	-19.32
2188958	Public/Street Tree	Removed	56.33	8.32	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-8.32	-56.33
2188959	Private	Removed	77.20	9.36	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-9.36	-77.20
2188960	Private	Removed	58.56	10.10	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-10.10	-58.56

2188969	Private	Removed	36.06	13.49	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-13.49	-36.06
2188970	Public/Street Tree	Acer	96.21	10.29	9	41	0.72	1.75	9.57	6.5	6.5	6.50	132.73	-0.72	36.52
2188972	Public/Street Tree	Acer	78.18	12.45	14	47	0.82	1.75	16.76	6.0	6.0	6.00	113.10	4.31	34.92
2188975	Public/Street Tree	Tilia	46.81	9.58	10	39	0.68	1.75	9.85	3.5	4.0	3.75	44.18	0.27	-2.63
2188976	Public/Street Tree	Acer	42.27	8.67	8	51	0.89	1.75	11.63	3.0	4.0	3.50	38.48	2.96	-3.79
2188977	Public/Street Tree	Tilia	68.52	10.14	8	41	0.72	1.75	8.70	2.5	2.5	2.50	19.63	-1.44	-48.89
2188978	Public/Street Tree	Tilia	26.45	8.93	9	47	0.82	1.75	11.40	3.0	2.5	2.75	23.76	2.47	-2.69
2188988	Public/Street Tree	Betula	67.89	6.92	10	55	0.96	1.75	16.03	4.0	4.0	4.00	50.27	9.11	-17.62
2189143	Private	Removed	38.54	10.92	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-10.92	-38.54
2189148	Private	Removed	12.56	7.34	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-7.34	-12.56
2189149	Private	Removed	56.25	10.49	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-10.49	-56.25
2189152	Public/Street Tree	Removed	10.29	6.55	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-6.55	-10.29
2189156	Public/Street Tree	Removed	41.96	11.34	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-11.34	-41.96
2189157	Public/Street Tree	Removed	20.57	7.04	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-7.04	-20.57
2189531	Private	Acer	116.11	12.42	10	58	1.01	1.75	17.75	6.0	5.5	5.75	103.87	5.33	-12.24
2189570	Public/Street Tree	Crataegus	14.36	5.43	5	28	0.49	1.75	4.41	3.0	1.5	2.25	15.90	-1.02	1.54
2189571	Public/Street Tree	Sorbus	4.17	5.41	6	36	0.63	1.75	6.11	1.0	1.5	1.25	4.91	0.70	0.74
2189572	Public/Street Tree	Sorbus	20.61	8.10	6	50	0.87	1.75	8.90	2.5	2.5	2.50	19.63	0.80	-0.98

2189573	Public/Street Tree	Unknown	8.42	6.58	6	41	0.72	1.75	6.97	1.5	2.0	1.75	9.62	0.39	1.20
2189574	Public/Street Tree	Unknown	8.06	6.31	6	41	0.72	1.75	6.97	1.5	2.0	1.75	9.62	0.66	1.56
2189575	Public/Street Tree	Alnus	21.37	11.82	7	60	1.05	1.75	13.87	3.5	2.5	3.00	28.27	2.05	6.90
2189585	Public/Street Tree	Carpinus	12.61	7.93	8	38	0.66	1.75	8.00	2.0	2.0	2.00	12.57	0.07	-0.04
2189586	Public/Street Tree	Carpinus	12.31	8.07	8	41	0.72	1.75	8.70	2.0	2.0	2.00	12.57	0.63	0.26
2189587	Public/Street Tree	Carpinus	6.15	6.77	6	39	0.68	1.75	6.61	2.0	2.0	2.00	12.57	-0.16	6.42
2189588	Public/Street Tree	Chaenomeles	1.67	3.99	3	28	0.49	1.75	3.35	1.0	1.0	1.00	3.14	-0.64	1.47
2189589	Public/Street Tree	Chaenomeles	1.92	3.94	5	34	0.59	1.75	5.12	1.0	1.0	1.00	3.14	1.18	1.22
2189592	Public/Street Tree	Chaenomeles	1.75	3.89	5	30	0.52	1.75	4.64	1.0	1.0	1.00	3.14	0.75	1.39
2189599	Public/Street Tree	Chaenomeles	1.60	4.15	5	34	0.59	1.75	5.12	1.0	1.0	1.00	3.14	0.97	1.54
2189602	Public/Street Tree	Chaenomeles	1.38	4.14	4	31	0.54	1.75	4.15	0.5	1.0	0.75	1.77	0.01	0.39
2189606	Public/Street Tree	Chaenomeles	1.76	3.84	5	22	0.38	1.75	3.77	0.5	0.5	0.50	0.79	-0.07	-0.97
2189607	Public/Street Tree	Chaenomeles	2.79	3.84	3	25	0.44	1.75	3.15	0.5	0.5	0.50	0.79	-0.69	-2.00
2189686	Public	Prunus	21.34	5.41	8	42	0.73	1.75	8.95	3.0	2.5	2.75	23.76	3.54	2.42
2189687	Public	Prunus	38.27	5.94	8	36	0.63	1.75	7.56	4.5	4.0	4.25	56.75	1.62	18.48
2189758	Public	Chaenomeles	2.16	4.15	5	29	0.51	1.75	4.52	1.0	1.0	1.00	3.14	0.37	0.98
2189759	Public	Chaenomeles	1.90	3.93	5	26	0.45	1.75	4.19	1.0	1.0	1.00	3.14	0.26	1.24
2189760	Public	Chaenomeles	2.52	3.76	5	27	0.47	1.75	4.30	1.0	1.0	1.00	3.14	0.54	0.62

2189761	Public	Chaenomeles	1.79	3.57	5	28	0.49	1.75	4.41	1.0	1.0	1.00	3.14	0.84	1.35
2189762	Public	Chaenomeles	1.38	3.51	5	28	0.49	1.75	4.41	0.5	0.5	0.50	0.79	0.90	-0.59
2189763	Public	Chaenomeles	1.31	3.54	5	42	0.73	1.75	6.25	1.0	1.0	1.00	3.14	2.71	1.83
2189764	Public	Chaenomeles	2.57	3.57	5	39	0.68	1.75	5.80	1.0	1.0	1.00	3.14	2.23	0.57
2189792	Public/Street Tree	Malus	1.64	3.55	4	45	0.79	1.75	5.75	1.0	1.0	1.00	3.14	2.20	1.50
2189793	Public/Street Tree	Malus	3.28	3.82	4	45	0.79	1.75	5.75	1.0	1.5	1.25	4.91	1.93	1.63
2189794	Public/Street Tree	Malus	2.30	3.75	4	50	0.87	1.75	6.52	1.0	1.5	1.25	4.91	2.77	2.61
2189795	Public/Street Tree	Malus	2.03	4.09	4	46	0.80	1.75	5.89	1.0	1.0	1.00	3.14	1.80	1.11
2189796	Public/Street Tree	Malus	1.56	4.33	4	42	0.73	1.75	5.35	1.0	1.0	1.00	3.14	1.02	1.58
2189797	Public/Street Tree	Unknown	2.67	4.21	4	51	0.89	1.75	6.69	1.5	1.5	1.50	7.07	2.48	4.40
2189798	Public/Street Tree	Carpinus	2.08	4.29	3	53	0.93	1.75	5.73	1.0	1.0	1.00	3.14	1.44	1.06
2189799	Public/Street Tree	Carpinus	2.63	4.01	4	43	0.75	1.75	5.48	1.0	1.0	1.00	3.14	1.47	0.51
2189800	Public/Street Tree	Fraxinus	4.68	5.87	5	44	0.77	1.75	6.58	1.5	1.5	1.50	7.07	0.71	2.39
2189801	Public/Street Tree	Carpinus	1.95	3.91	5	39	0.68	1.75	5.80	1.0	1.0	1.00	3.14	1.89	1.19
2189802	Public/Street Tree	Fraxinus	3.42	4.29	6	35	0.61	1.75	5.95	1.0	1.0	1.00	3.14	1.66	-0.28
2189803	Public/Street Tree	Fraxinus	160.84	14.79	20	40	0.70	1.75	18.53	8.0	8.0	8.00	201.06	3.74	40.22
2189804	Public/Street Tree	Fraxinus	119.51	17.19	12	60	1.05	1.75	22.53	7.0	8.0	7.50	176.71	5.34	57.20

2189805	Public/Street Tree	Robinia	75.04	16.97	12	62	1.08	1.75	24.32	4.0	8.5	6.25	122.72	7.35	47.68
2189927	Private	Acer	85.88	15.36	10	60	1.05	1.75	19.07	5.0	5.5	5.25	86.59	3.71	0.71
2189928	Private	2 X Acer	63.57	13.99	8	59	1.03	1.75	15.06	3.5	3.0	3.25	33.18	1.07	-30.39
2189929	Private	Carpinus	58.39	9.94	8	57	0.99	1.75	14.07	4.5	4.0	4.25	56.75	4.13	-1.64
2189931	Private	Removed	13.50	6.73	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-6.73	-13.50
2189932	Private	Removed	10.88	6.89	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-6.89	-10.88
2191251	Public/Street Tree	Prunus	88.89	11.67	10	54	0.94	1.75	15.51	5.0	6.0	5.50	95.03	3.84	6.14
2191252	Public/Street Tree	1 X Fraxinus, 1 X Robinia	175.40	17.19	12	55	0.96	1.75	18.89	6.0	9.0	7.50	176.71	1.70	1.31
2191253	Public/Street Tree	1 X Prunus, 1 X Removed	128.66	16.46	10	50	0.87	1.75	13.67	4.0	6.0	5.00	78.54	-2.79	-50.12
2191254	Public/Street Tree	2 X Prunus	73.08	11.57	10	44	0.77	1.75	11.41	4.5	7.0	5.75	103.87	-0.16	30.79
2191255	Public/Street Tree	Aesculus	106.18	16.40	15	45	0.79	1.75	16.75	5.0	6.0	5.50	95.03	0.35	-11.15
2191256	Public/Street Tree	Fagus	159.49	17.37	14	55	0.96	1.75	21.74	6.0	7.5	6.75	143.14	4.37	-16.35
2191257	Public/Street Tree	Removed	60.12	16.71	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-16.71	-60.12
2191259	Public/Street Tree	Alnus	101.17	15.73	12	60	1.05	1.75	22.53	7.0	7.5	7.25	165.13	6.80	63.96
2191261	Public/Street Tree	2 X Prunus	114.88	11.22	10	41	0.72	1.75	10.44	6.0	6.0	6.00	113.10	-0.78	-1.78
2191262	Public/Street Tree	Betula	21.55	10.78	10	52	0.91	1.75	14.55	3.0	4.0	3.50	38.48	3.77	16.93
2191263	Public/Street Tree	1 X Prunus, 1 X Sorbus (dead)	150.74	11.15	14	37	0.65	1.75	12.30	6.0	9.5	7.75	188.69	1.15	37.95

Appendix Table 2: Lower Kersal tree canopy sample plot results.

ID	Location	Genus	Area 09 (m^2)	Height 09 (m)	Distance (m)	Angle (°)	Radians	Eye Level (m)	Height 13 (m)	Radius 1 (m)	Radius 2 (m)	Average (m)	Area 13 (m^2)	Height Δ (m)	Area Δ (m^2)
2195475	Public	Fraxinus	6.42	5.33	4	52	0.91	1.75	6.87	2.0	1.5	1.75	9.62	1.54	3.20
2195476	Public	Fraxinus	2.54	6.83	5	49	0.86	1.75	7.50	1.5	1.5	1.50	7.07	0.67	4.53
2195478	Public	Fraxinus	8.70	5.78	5	46	0.80	1.75	6.93	1.5	2.0	1.75	9.62	1.15	0.92
2195479	Public	Fraxinus	5.72	5.42	5	42	0.73	1.75	6.25	1.5	1.5	1.50	7.07	0.83	1.35
2195592	Public/Street	Corylus	4.13	5.19	6	35	0.61	1.75	5.95	1.5	1.5	1.50	7.07	0.76	2.94
2195593	Public/Street	Corylus	9.40	5.55	7	35	0.61	1.75	6.65	2.0	2.0	2.00	12.57	1.10	3.17
2195596	Public/Street	Tilia	6.35	5.06	6	36	0.63	1.75	6.11	1.5	1.5	1.50	7.07	1.05	0.72
2195597	Public/Street	Tilia	8.70	5.72	7	33	0.58	1.75	6.30	1.5	2.0	1.75	9.62	0.58	0.92
2195598	Public/Street	Unknown	4.70	6.73	7	40	0.70	1.75	7.62	1.0	1.5	1.25	4.91	0.89	0.21
2195600	Public/Street	Prunus	20.65	5.08	7	34	0.59	1.75	6.47	3.0	2.5	2.75	23.76	1.39	3.11
2195601	Public/Street	Prunus	18.93	5.02	6	38	0.66	1.75	6.44	3.0	3.0	3.00	28.27	1.42	9.34
2195740	Public/Street	Unknown	12.83	5.37	6	40	0.70	1.75	6.78	1.0	1.5	1.25	4.91	1.41	-7.92
2195741	Public/Street	Tilia	12.26	5.13	6	38	0.66	1.75	6.44	2.0	1.5	1.75	9.62	1.31	-2.64
2195849	Public/Street	Crataegus	40.28	6.52	8	40	0.70	1.75	8.46	3.5	4.0	3.75	44.18	1.94	3.90
2195853	Public/Street	Tilia	6.92	4.92	6	38	0.66	1.75	6.44	1.5	1.5	1.50	7.07	1.52	0.15
2195855	Public/Street	Tilia	8.64	4.96	6	35	0.61	1.75	5.95	1.5	2.0	1.75	9.62	0.99	0.98
2195857	Public/Street	Corylus	3.37	4.32	4	45	0.79	1.75	5.75	1.0	1.5	1.25	4.91	1.43	1.54
2195867	Public/Street	Unknown	9.21	6.15	4	56	0.98	1.75	7.68	2.0	1.5	1.75	9.62	1.53	0.41
2195868	Public/Street	Corylus	9.47	4.97	4	50	0.87	1.75	6.52	2.0	1.5	1.75	9.62	1.55	0.15
2195869	Public/Street	Corylus	6.42	4.77	4	52	0.91	1.75	6.87	2.0	1.5	1.75	9.62	2.10	3.20
2195987	Public	Prunus	41.29	7.60	5	57	0.99	1.75	9.45	4.0	3.0	3.50	38.48	1.85	-2.81
2195989	Public/Street	Poplus	25.41	5.41	10	41	0.72	1.75	10.44	4.5	3.5	4.00	50.27	5.03	24.86
2195990	Public/Street	Acer	96.69	10.72	11	46	0.80	1.75	13.14	4.0	5.5	4.75	70.88	2.42	-25.81
2198589	Public	Quercus and Fraxinus	7.55	2.16	2	64	1.12	1.75	5.85	2.0	2.0	2.00	12.57	3.69	5.02
2198590	Public	5 X Fraxinus	5.18	2.28	2	65	1.13	1.75	6.04	2.0	2.0	2.00	12.57	3.76	7.39

2198591	Public	Faxinus and Corylus	7.98	2.48	3	50	0.87	1.75	5.33	2.0	2.0	2.00	12.57	2.85	4.59
2198690	Public	Quercus	8.63	1.77	5	47	0.82	1.75	7.24	1.5	2.0	1.75	9.62	5.47	0.99
2198691	Public	Fraxinus	3.23	1.75	5	45	0.79	1.75	6.75	1.5	1.5	1.75	7.07	5.00	3.84
2199011	Public	Corylus	11.07	2.28	5	20	0.35	1.75	2.46	2.0	2.5	2.25	15.9	0.18	4.83
2199018	Public	Alnus	7.76	2.31	5	18	0.31	1.75	2.19	1.5	1.5	1.5	7.07	-0.12	-0.69

Appendix Table 3: Peel Park tree canopy sample plot results.

ID	Location	Genus	Area 09 (m^2)	Height 09 (m)	Distance (m)	Angle (°)	Radians	Eye Level (m)	Height 13 (m)	Radius 1 (m)	Radius 2 (m)	Average (m)	Area 13 (m^2)	Height Δ (m)	Area Δ (m^2)
2432820	Park	Removed	32.56	8.56	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-8.56	-32.56
2432821	Park	Removed	123.25	11.24	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-11.24	-123.25
2432822	Park	Prunus	159.19	9.24	15	38	0.66	1.75	13.47	8.0	7.0	7.50	176.71	4.23	17.52
2432823	Park	2 X Tilia	149.79	16.07	15	51	0.89	1.75	20.27	6.0	8.0	7.00	153.94	4.20	4.15
2432825	Park	Crataegus	136.68	16.73	10	45	0.79	1.75	11.75	3.5	4.5	4.00	50.27	-4.98	-86.41
2432834	Park	Fraxinus	202.92	13.74	20	39	0.68	1.75	17.95	7.0	7.0	7.00	153.94	4.21	-48.98
2432843	Park	Sorbus	33.79	6.28	10	33	0.58	1.75	8.24	2.0	1.0	1.50	7.07	1.96	-26.72
2432846	Park	Fraxinus	113.46	10.32	15	38	0.66	1.75	13.47	6.0	7.0	6.50	132.73	3.15	19.27
2432847	Park	Unknown /Dead	54.15	9.48	10	38	0.66	1.75	9.56	3.0	3.0	3.00	28.27	0.08	-25.88
2432853	Park	Crataegus	19.16	6.83	10	35	0.61	1.75	8.75	3.0	3.0	3.00	28.27	1.92	9.11
2432871	Park	Fraxinus	75.25	11.32	15	39	0.68	1.75	13.90	6.0	6.5	6.25	122.72	2.58	47.47
2432873	Park	Fraxinus	192.39	15.03	20	38	0.66	1.75	17.38	8.0	6.0	7.00	153.94	2.35	-38.45
2432877	Park	Sorbus	50.31	6.39	10	37	0.65	1.75	9.29	2.0	4.0	3.00	28.27	2.90	-22.04
2432879	Park	Removed	52.50	12.77	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-12.77	-52.50
2432880	Park	Removed	24.28	4.34	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-4.34	-24.28
2432881	Park	Crataegus , Fraxinus	723.98	20.48	20	50	0.87	1.75	25.59	13.1	13.8	13.43	566.21	5.11	-157.77

2432891	Park	Acer	322.23	14.30	15	47	0.82	1.75	17.84	8.0	7.0	7.50	176.71	3.54	-145.52
2432897	Park	Alnus	86.99	12.32	10	50	0.87	1.75	13.67	5.0	3.5	4.25	56.75	1.35	-30.24
2432898	Park	Unknown /Dead	110.76	13.05	11	50	0.87	1.75	14.86	1.0	1.0	1.00	3.14	1.81	-107.62
2432902	Park	Betula	75.57	13.91	15	45	0.79	1.75	16.75	5.0	5.0	5.00	78.54	2.84	2.97
2432905	Park	Alnus	70.40	12.46	15	40	0.70	1.75	14.34	5.0	5.0	5.00	78.54	1.88	8.14
2432909	Park	Alnus	31.12	10.37	10	50	0.87	1.75	13.67	2.0	3.0	2.50	19.63	3.30	-11.49
2432912	Park	Crataegus	33.41	7.59	10	37	0.65	1.75	9.29	5.0	3.5	4.25	56.75	1.70	23.34
2432921	Park	Acer	203.09	11.21	16	40	0.70	1.75	15.18	7.0	8.0	7.50	176.71	3.97	-26.38
2432922	Park	Fraxinus	26.06	5.16	6	38	0.66	1.75	6.44	3.0	3.0	3.00	28.27	1.28	2.21
2432923	Park	Fraxinus	40.75	5.24	6	39	0.68	1.75	6.61	3.0	4.0	3.50	38.48	1.37	-2.27
2433318	Public	Removed	141.88	9.02	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-9.02	-141.88
2433319	Public	Removed	49.78	5.99	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-5.99	-49.78
2433686	Park	Fraxinus, Prunus, Acer, Ulmus	26.24	6.20	10	48	0.84	1.75	12.86	5.0	6.0	5.50	95.03	6.66	68.79
2433687	Park	Removed	13.18	3.90	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-3.90	-13.18
2433707	Park	Prunus	113.08	10.03	11	30	0.52	1.75	8.10	2.0	7.0	4.50	63.62	-1.93	-49.46
2433708	Park	4 X Prunus	812.05	15.76	16	40	0.70	1.75	15.18	17.0	10.0	13.50	572.56	-0.58	-239.49
2433711	Park	Prunus	34.62	10.34	11	49	0.86	1.75	14.40	5.0	3.0	4.00	50.27	4.06	15.65
2433724	Park	Acer	76.19	12.97	15	43	0.75	1.75	15.74	7.0	3.0	5.00	78.54	2.77	2.35
2433726	Park	Acer	276.73	15.40	20	40	0.70	1.75	18.53	8.0	9.0	8.50	226.98	3.13	-49.75
2433729	Park	Prunus	129.41	10.24	11	48	0.84	1.75	13.97	4.0	7.0	5.50	95.03	3.73	-34.38
2433731	Park	1 X Acer, 1 X Prunus	190.47	14.53	15	45	0.79	1.75	16.75	7.0	7.0	7.00	153.94	2.22	-36.53
2433733	Park	Crataegus	51.77	6.57	10	35	0.61	1.75	8.75	4.5	4.5	4.50	63.62	2.18	11.85
2433734	Park	Acer	86.14	11.42	12	46	0.80	1.75	14.18	5.0	5.5	5.25	86.59	2.76	0.45

2433735	Park	Acer	223.45	13.74	20	45	0.79	1.75	21.75	9.0	10.5	9.75	298.65	8.01	75.20
2433751	Park	Fraxinus	344.68	18.08	20	42	0.73	1.75	19.76	9.0	11.0	10.00	314.16	1.68	-30.52
2433756	Park	Laburnum	74.90	10.37	16	33	0.58	1.75	12.14	5.0	4.0	4.50	63.62	1.77	-11.28
2433758	Park	Crataegus	77.74	11.01	12	43	0.75	1.75	12.94	6.0	5.0	5.50	95.03	1.93	17.29
2433761	Park	Prunus	103.31	9.33	15	33	0.58	1.75	11.49	7.0	6.5	6.75	143.14	2.16	39.83
2433762	Park	Acer	16.60	6.11	10	45	0.79	1.75	11.75	5.0	4.0	4.50	63.62	5.64	47.02
2433787	Park	5 X Acer, 4 X Prunus	467.60	18.10	20	48	0.84	1.75	23.96	20.0	12.0	16.00	804.25	5.86	336.65
2433791	Park	4 X Acer, 4 X Prunus	437.29	14.16	18	51	0.89	1.75	23.98	11.0	12.0	11.50	415.48	9.82	-21.81
2433792	Park	4 X Acer, 3 X Prunus	434.81	15.16	13	50	0.87	1.75	17.24	10.0	11.0	10.50	346.36	2.08	-88.45
2433795	Park	4 x Acer	359.51	16.59	20	38	0.66	1.75	17.38	11.0	11.0	11.00	380.13	0.79	20.62
2433823	Park	Fraxinus	196.05	13.74	16	43	0.75	1.75	16.67	7.0	8.0	7.50	176.71	2.93	-19.34
2433824	Park	Removed	76.82	8.19	0	0	0.00	1.75	1.75	0.0	0.0	0.00	0.00	-6.44	-76.82
2433904	Public/University	3 X Chamaecy paris	15.44	7.17	10	39	0.68	1.75	9.85	2.5	2.0	2.25	15.90	2.68	0.46
2433906	Public/University	Taxodium	41.32	13.17	10	50	0.87	1.75	13.67	3.5	4.0	3.75	44.18	0.50	2.86
2433908	Public/University	Prunus	39.50	10.30	10	40	0.70	1.75	10.14	4.0	3.5	3.75	44.18	-0.16	4.68
2433910	Public/University	Prunus	132.45	8.38	12	37	0.65	1.70	10.74	6.0	5.0	5.50	95.03	2.36	-37.42
2433911	Public/University	Sorbus	58.35	8.35	10	39	0.68	1.75	9.85	5.0	5.0	5.00	78.54	1.50	20.19
2433912	Public/University	Crataegus	23.36	7.09	7	45	0.79	1.75	8.75	3.5	3.0	3.25	33.18	1.66	9.82

2433913	Public/University	Sorbus	40.47	10.69	10	50	0.87	1.75	13.67	4.0	3.5	3.75	44.18	2.98	3.71
2433920	Public/University	Pinus	16.25	4.87	7	33	0.58	1.70	6.25	3.0	2.0	2.50	19.63	1.38	3.38
2433926	Public/University	Sorbus	23.05	5.07	8	33	0.58	1.75	6.95	3.0	3.0	3.00	28.27	1.88	5.22
2433927	Public/University	Tilia	138.58	12.28	14	43	0.75	1.75	14.81	5.5	8.0	6.75	143.14	2.53	4.56
2433935	Public/University	Removed	329.47	16.32	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-16.32	-329.47
2433957	Public/University	Removed	218.28	13.94	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-13.94	-218.28
2433961	Public/University	Robinia	6.92	3.93	10	23	0.40	1.75	5.99	2.0	2.0	2.00	12.57	2.06	5.65
2433962	Public/University	Pinus	30.49	7.16	8	45	0.79	1.70	9.70	2.5	4.0	3.25	33.18	2.54	2.69
2433963	Public/University	Sorbus	75.44	8.67	10	48	0.84	1.75	12.86	5.0	6.0	5.50	95.03	4.19	19.59
2433970	Public/University	Prunus	36.22	4.63	7	39	0.68	1.70	7.37	3.0	3.0	3.00	28.27	2.74	-7.95
2433974	Public/University	Fagus	193.73	12.06	13	47	0.82	1.70	15.64	8.0	10.0	9.00	254.47	3.58	60.74
2433984	Park	Acer	158.89	13.48	17	42	0.73	1.75	17.06	8.0	7.0	7.50	176.71	3.58	17.82
2433985	Park	Populus	121.66	27.44	20	58	1.01	1.75	33.76	9.0	7.0	8.00	201.06	6.32	79.40
2624216	Park	Fraxinus	6.48	4.08	3	43	0.75	1.75	4.55	1.5	1.5	1.50	7.07	0.47	0.59
2624217	Park	Fraxinus	7.22	4.49	3	40	0.70	1.75	4.27	1.5	2.0	1.75	9.62	-0.22	2.40
2624232	Park	Tilia	377.29	19.51	20	47	0.82	1.75	23.20	8.0	9.0	8.50	226.98	3.69	-150.31
2624234	Park	Alnus	84.09	17.06	18	48	0.84	1.75	21.74	4.0	5.0	4.50	63.62	4.68	-20.47
2624241	Park	Fraxinus	405.97	19.21	29	36	0.63	1.75	22.53	11.0	12.0	11.50	415.48	3.32	9.51
2624243	Park	Crataegus	112.61	12.39	15	40	0.70	1.75	14.34	6.0	7.0	6.50	132.73	1.95	20.12
2624245	Park	Crataegus	30.63	5.66	8	33	0.58	1.75	6.95	3.0	4.0	3.50	38.48	1.29	7.85

2624250	Park	Acer	154.01	13.18	16	45	0.79	1.75	17.75	6.0	7.0	6.50	132.73	4.57	-21.28
2624255	Park	Acer	275.68	13.58	10	65	1.13	1.75	23.20	7.0	9.0	8.00	201.06	9.62	-74.62
2624257	Park	Acer	121.79	17.79	12	55	0.96	1.75	18.89	12.0	5.0	8.50	226.98	1.10	105.19
2624262	Park	Removed	272.71	17.99	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-17.99	-272.71
2624263	Park	Fraxinus	201.22	16.08	20	40	0.70	1.75	18.53	10.0	11.0	10.50	346.36	2.45	145.14
2624269	Park	Prunus	266.16	16.02	21	45	0.79	1.75	22.45	10.4	8.5	9.45	280.55	6.43	14.39
2624270	Park	Acer	251.70	13.73	18	41	0.72	1.75	17.40	9.0	10.0	9.50	283.53	3.67	31.83
2624272	Park	Acer	312.79	15.00	16	41	0.72	1.75	15.66	8.0	8.0	8.00	201.06	0.66	-111.73
2624280	Park	Alnus	68.15	13.50	13	40	0.70	1.75	12.66	3.0	3.0	3.00	28.27	-0.84	-39.88
2624283	Park	Crataegus	114.01	8.94	10	35	0.61	1.75	8.75	6.0	4.0	5.00	78.54	-0.19	-35.47
2624284	Park	Betula	56.21	13.05	15	44	0.77	1.75	16.24	5.5	5.0	5.25	86.59	3.19	30.38
2624286	Public/University	Prunus	169.86	14.62	9	37	0.65	1.75	8.53	5.0	6.0	5.50	95.03	-6.09	-74.83
2624295	Public/University	Removed	12.57	4.34	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-4.34	-12.57
2624298	Public/University	Removed	103.88	8.68	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-8.68	-103.88
2624299	Public/University	Removed	57.33	8.58	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-8.58	-57.33
2624880	Park	Fraxinus	92.81	8.40	16	35	0.61	1.75	12.95	6.0	6.0	6.00	113.10	4.55	20.29
2624883	Park	Betula	72.56	9.61	10	45	0.79	1.75	11.75	3.0	3.0	3.00	28.27	2.14	-44.29
2624893	Park	Acer	50.22	10.37	10	60	1.05	1.75	19.07	4.0	3.0	3.50	38.48	8.70	-11.74
2624898	Park	Crataegus	96.51	6.29	11	39	0.68	1.75	10.66	5.5	4.0	4.75	70.88	4.37	-25.63
2624899	Park	Fraxinus	100.97	10.70	10	49	0.86	1.75	13.25	5.5	5.0	5.25	86.59	2.55	-14.38
2624900	Park	2 X Prunus	216.39	12.10	18	46	0.80	1.75	20.39	10.0	8.0	9.00	254.47	8.29	38.08
2625193	Park	Fraxinus	360.05	16.00	15	55	0.96	1.75	23.17	10.0	11.0	10.50	346.36	7.17	-13.69
2625352	Park	Robinia	111.95	12.21	15	45	0.79	1.75	16.75	4.5	8.0	6.25	122.72	4.54	10.77

Appendix Table 4: Kersal tree canopy sample plot results.

ID	Location	Genus	Area 09 (m^2)	Height 09 (m)	Distance (m)	Angle (°)	Radians	Eye Level (m)	Height 13 (m)	Radius 1 (m)	Radius 2 (m)	Average (m)	Area 13 (m^2)	Height Δ (m)	Area Δ (m^2)
2065969	Private	Ilex	38.23	9.34	10	39	0.68	0.75	8.85	4.0	3.5	3.75	44.18	-0.49	5.95
2065975	Private	Fagus	151.79	12.50	14	46	0.80	1.75	16.25	8.0	8.0	8.00	201.06	3.75	49.27
2066006	Private	Tilia	215.39	14.85	12	58	1.01	1.75	20.95	7.0	8.0	7.50	176.71	6.10	-38.68
2068097	Private	Fagus	62.27	8.73	11	41	0.72	0.50	10.06	5.0	4.0	4.50	63.62	1.33	1.35
2068098	Private	Fraxinus, Acer	188.27	16.68	18	43	0.75	0.75	17.54	5.0	7.0	6.00	113.10	0.86	-75.17
2068125	Private	Acer	173.82	14.81	12	51	0.89	0.70	15.52	7.0	3.0	5.00	78.54	0.71	-95.28
2068129	Private	Acer	80.57	13.83	11	53	0.93	1.75	16.35	3.0	3.0	3.00	28.27	2.52	-52.30
2068138	Private	Acer	260.46	17.21	20	38	0.66	1.75	17.38	7.0	8.0	7.50	176.71	0.17	-83.75
2068161	Private	Acer	127.01	13.33	17	32	0.56	1.75	12.37	5.0	4.5	4.75	70.88	-0.96	-56.13
2068162	Private	Fraxinus	89.40	11.06	12	37	0.65	1.75	10.79	2.0	3.0	2.50	19.63	-0.27	-69.77
2068163	Private	2 X Fagus	140.94	16.72	19	40	0.70	0.75	16.69	7.0	7.0	7.00	153.94	-0.03	13.00
2068170	Private	Fraxinus	302.32	17.65	18	43	0.75	1.75	18.54	7.0	6.0	6.50	132.73	0.89	-169.59
2068171	Private	Fagus	239.36	21.29	21	45	0.79	1.75	22.75	7.0	9.0	8.00	201.06	1.46	-38.30
2068172	Private	Fagus	141.04	16.54	17	42	0.73	1.75	17.06	4.0	5.0	4.50	63.62	0.52	-77.42
2068173	Private	Fraxinus	29.40	6.54	10	40	0.70	0.75	9.14	2.0	2.0	2.00	12.57	2.60	-16.83
2068192	Private	Removed	22.26	5.53	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-5.53	-22.26
2068197	Private	Malus	32.29	4.27	8	23	0.40	1.00	4.40	2.5	2.0	2.25	15.90	0.13	-16.39
2068198	Private	Malus	33.79	6.01	9	33	0.58	0.75	6.59	2.0	3.0	2.50	19.63	0.58	-14.16
2068254	Private	Fagus	348.52	18.24	20	48	0.84	1.75	23.96	10.0	13.0	11.50	415.48	5.72	66.96
2068748	Private	Fagus	433.67	21.79	20	47	0.82	1.75	23.20	12.0	12.0	12.00	452.39	1.41	18.72
2068756	Private	Tilia	109.25	11.68	16	29	0.51	0.00	8.87	4.0	4.0	4.00	50.27	-2.81	-58.98
2068757	Private	Fagus	199.27	14.64	17	45	0.79	0.00	17.00	8.0	7.0	7.50	176.71	2.36	-22.56
2068760	Private	Removed	26.58	11.86	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-11.86	-26.58
2068761	Private	Removed	28.68	10.98	0	0	0.00	1.75	0.00	0.0	0.0	0.00	0.00	-10.98	-28.68
2068762	Private	Aesculus	152.68	13.22	15	30	0.52	0.75	9.41	4.0	6.0	5.00	78.54	-3.81	-74.14

2068764	Private	Aesculus, Acer	277.03	14.03	16	42	0.73	0.75	15.16	7.0	9.0	8.00	201.06	1.13	-75.97
2068765	Private	Tilia	72.10	14.40	16	33	0.58	1.75	12.14	3.0	4.0	3.50	38.48	-2.26	-33.62
2068766	Private	Tilia, Acer	211.71	17.86	20	36	0.63	0.75	15.28	7.0	5.0	6.00	113.10	-2.58	-98.61
2068767	Private	2 X Tilia	189.13	17.25	20	40	0.70	0.75	17.53	5.0	5.5	5.25	86.59	0.28	-102.54
2068768	Public	Tilia	125.80	19.68	11	44	0.77	0.75	11.37	1.5	1.5	1.50	7.07	-8.31	-118.73
2068769	Public	Tilia	353.24	20.94	11	49	0.86	0.75	13.40	2.0	2.0	2.00	12.57	-7.54	-340.67
2068770	Private	Tilia	84.99	16.80	20	37	0.65	1.70	16.77	3.0	4.5	3.75	44.18	-0.03	-40.81
2068771	Private	Fagus	407.73	21.83	20	45	0.79	1.75	21.75	14.0	11.0	12.50	490.87	-0.08	83.14
2068790	Private	Fagus	226.41	14.80	18	46	0.80	1.75	20.39	10.0	9.0	9.50	283.53	5.59	57.12
2068793	Public	Tilia	51.98	9.38	11	44	0.77	0.75	11.37	3.5	3.0	3.25	33.18	1.99	-18.80
2068807	Private	Acer	64.13	10.40	17	28	0.49	0.75	9.79	5.0	5.0	5.00	78.54	-0.61	14.41
2068808	Private	Acer	167.05	11.73	14	34	0.59	0.00	9.44	4.0	4.5	4.25	56.75	-2.29	-110.30
2068817	Private	Acer	153.43	11.58	17	35	0.61	0.75	12.65	5.0	6.0	5.50	95.03	1.07	-58.40
2068818	Private	Betula	77.36	10.32	11	47	0.82	1.75	13.55	4.0	4.0	4.00	50.27	3.23	-27.09
2068855	Private	Fagus	81.03	14.10	12	47	0.82	1.75	14.62	4.0	4.5	4.25	56.75	0.52	-24.28
2068862	Private	Betula	34.77	8.08	11	35	0.61	0.75	8.45	2.0	2.0	2.00	12.57	0.37	-22.20
2068863	Private	Acer	108.85	10.21	16	30	0.52	1.70	10.94	4.0	4.0	4.00	50.27	0.73	-58.58
2068864	Private	Acer	47.49	7.11	8	41	0.72	1.75	8.70	4.0	3.0	3.50	38.48	1.59	-9.01
2068865	Private	Unknown	48.68	6.38	8	31	0.54	1.75	6.56	3.0	3.0	3.00	28.27	0.18	-20.41
2068866	Private	Unknown	29.53	5.38	8	25	0.44	1.75	5.48	2.0	2.0	2.00	12.57	0.10	-16.96
2068867	Private	2 X Acer, Betula, Prunus	146.18	13.51	15	38	0.66	1.70	13.42	5.0	6.0	5.50	95.03	-0.09	-51.15
2068876	Public/Street Tree	Acer	284.39	20.22	20	48	0.84	1.75	23.96	5.0	7.5	6.25	122.72	3.74	-161.67
2068877	Public/Street Tree	Acer	154.59	13.57	11	59	1.03	1.75	20.06	6.5	4.0	5.25	86.59	6.49	-68.00
2068879	Public/Street Tree	Acer	96.04	14.17	12	48	0.84	1.75	15.08	4.0	7.0	5.50	95.03	0.91	-1.01
2068880	Public/Street Tree	Tilia	168.72	17.35	14	54	0.94	1.75	21.02	6.5	6.0	6.25	122.72	3.67	-46.00
2068887	Public/Street Tree	Fagus	381.03	22.27	22	50	0.87	0.75	26.97	11.0	10.5	10.75	363.05	4.70	-17.98
2068891	Public/Street Tree	Aesculus	149.91	17.26	15	49	0.86	1.75	19.01	9.0	5.5	7.25	165.13	1.75	15.22

2068892	Public/Street Tree	1 X Aesculus, 1 X Acer, 1 X Tilia	242.82	19.87	16	51	0.89	1.75	21.51	7.5	8.0	7.75	188.69	1.64	-54.13
2068893	Public/Street Tree	1 X Acer, 1 X Fagus, 1 X Quercus	393.75	18.99	20	46	0.80	1.75	22.46	6.5	11.0	8.75	240.53	3.47	-153.22
2068903	Public/Street Tree	Acer	55.80	13.55	9	50	0.87	1.75	12.48	4.0	5.0	4.50	63.62	-1.07	7.82
2068957	Private	Tilia	191.42	20.38	20	48	0.84	0.75	22.96	6.0	4.0	5.00	78.54	2.58	-112.88
2068978	Private	Removed	26.23	3.45	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-3.45	-26.23
2069623	Private	Crataegus	57.80	7.18	10	44	0.77	0.75	10.41	4.0	4.0	4.00	50.27	3.23	-7.53
2069684	Public/Street Tree	Acer	302.22	20.67	11	55	0.96	0.75	16.46	5.0	6.0	5.50	95.03	-4.21	-207.19
2069687	Public/Street Tree	Tilia	185.92	13.43	11	59	1.03	0.75	19.06	8.0	6.5	7.25	165.13	5.63	-20.79
2192305	Private	Fagus	154.83	20.50	20	43	0.75	0.00	18.65	8.0	9.0	8.50	226.98	-1.85	72.15
2192306	Private	Fagus	195.29	20.73	20	45	0.79	0.00	20.00	6.0	7.0	6.50	132.73	-0.73	-62.56
2192307	Private	Fagus	61.67	18.18	18	49	0.86	0.00	20.71	6.0	8.0	7.00	153.94	2.53	92.27
2192347	Private	Tilia	140.48	21.38	20	45	0.79	1.75	21.75	6.0	5.0	5.50	95.03	0.37	-45.45
2192356	Private	2 X Tilia	231.23	22.49	22	45	0.79	1.75	23.75	9.0	6.0	7.50	176.71	1.26	-54.52
2192632	Private	Removed	6.80	4.25	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-4.25	-6.80
2192658	Private	Tilia	116.95	20.64	20	44	0.77	0.75	20.06	8.0	5.0	6.50	132.73	-0.58	15.78
2192660	Private	Aesculus	97.68	16.87	17	35	0.61	0.75	12.65	5.0	5.0	5.00	78.54	-4.22	-19.14
2192661	Private	Tilia	96.63	19.38	20	43	0.75	0.75	19.40	6.5	5.0	5.75	103.87	0.02	7.24
2192663	Private	Ilex	40.11	7.27	12	39	0.68	0.00	9.72	3.0	4.0	3.50	38.48	2.45	-1.63
2192664	Private	Removed	19.01	5.17	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-5.17	-19.01
2192665	Private	Removed	74.24	7.00	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-7.00	-74.24
2192670	Private	Ilex	29.52	10.06	9	42	0.73	0.05	8.15	3.0	2.5	2.75	23.76	-1.91	-5.76
2192672	Private	Salix	72.03	8.76	17	23	0.40	1.75	8.97	4.0	5.0	4.50	63.62	0.21	-8.41
2192675	Private	Removed	12.12	3.37	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-3.37	-12.12

2192676	Private	Removed	7.24	3.63	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-3.63	-7.24
2192677	Private	Quercus	142.35	14.50	21	34	0.59	0.75	14.91	6.0	5.0	5.50	95.03	0.41	-47.32
2192678	Private	Betula	36.47	12.53	17	42	0.73	0.75	16.06	5.0	4.0	4.50	63.62	3.53	27.15
2192680	Private	Ilex, Prunus	44.41	10.62	10	35	0.61	1.75	8.75	3.0	4.0	3.50	38.48	-1.87	-5.93
2192681	Private	Tilia	141.02	24.16	13	55	0.96	1.75	20.32	6.0	6.0	6.00	113.10	-3.84	-27.92
2192693	Private	Fagus	236.74	17.37	17	50	0.87	0.75	21.01	7.0	8.0	7.50	176.71	3.64	-60.03
2192700	Private	Fagus	327.02	20.71	20	46	0.80	1.75	22.46	10.0	10.0	10.00	314.16	1.75	-12.86
2192701	Private	Fagus	242.46	16.07	19	48	0.84	0.75	21.85	11.0	11.0	11.00	380.13	5.78	137.67
2192707	Private	Ilex	44.43	7.98	8	34	0.59	1.75	7.15	2.0	3.0	2.50	19.63	-0.83	-24.80
2192708	Private	Ilex	36.10	9.91	9	31	0.54	1.75	7.16	1.5	2.5	2.00	12.57	-2.75	-23.53
2192709	Private	Acer	55.97	14.33	12	47	0.82	1.75	14.62	4.0	3.5	3.75	44.18	0.29	-11.79
2192710	Private	Acer	58.47	13.80	13	36	0.63	1.75	11.20	4.0	4.0	4.00	50.27	-2.60	-8.20
2192711	Private	Acer, Ilex	58.51	13.21	10	44	0.77	1.75	11.41	3.0	4.0	3.50	38.48	-1.80	-20.03
2192926	Private	Aesculus	155.05	16.63	14	54	0.94	0.70	19.97	4.0	6.5	5.25	86.59	3.34	-68.46
2192928	Private	Removed	148.19	20.68	0	0	0.00	0.00	0.00	0.0	0.0	0.00	0.00	-20.68	-148.19
2193020	Public/Street Tree	Acer	84.01	12.08	15	35	0.61	1.75	12.25	5.0	5.0	5.00	78.54	0.17	-5.47
2193021	Public/Street Tree	2 x Acer	265.48	15.15	17	40	0.70	1.75	16.01	8.5	9.0	8.75	240.53	0.86	-24.95

Appendix B) River Valley Study Area (IIC) Data

Appendix Table 5: IIC results for the 2005 river valley study area NBH canopies.

Distance Threshold (m)	Canopy ≥17.1m IIC	Rate of Change	% Change	Total % Change
30	0.0000330			
60	0.0000712	0.0000013	116%	116%
90	0.0000903	0.0000006	27%	174%
120	0.0001166	0.0000009	29%	253%
150	0.0001325	0.0000005	14%	302%
180	0.0001495	0.0000006	13%	353%
200	0.0001639	0.0000007	10%	397%
Canopy ≥7.2m IIC				
30	0.0009869			
60	0.0015184	0.0000177	54%	54%
90	0.0020376	0.0000173	34%	106%
120	0.0024280	0.0000130	19%	146%
150	0.0026759	0.0000083	10%	171%
180	0.0029489	0.0000091	10%	199%
200	0.0031006	0.0000076	5%	214%
Canopy ≥3m IIC				
30	0.0014451			
60	0.0023156	0.0000290	60%	60%
90	0.0028881	0.0000191	25%	100%
120	0.0033422	0.0000151	16%	131%
150	0.0037439	0.0000134	12%	159%
180	0.0041097	0.0000122	10%	184%
200	0.0043507	0.0000121	6%	201%

Appendix Table 6: IIC results for the 2009 river valley study area NBH canopies.

Distance Threshold (m)	Canopy ≥17.1m IIC	Rate of Change	% Change	Total % Change
30	0.00002			
60	0.00008	0.0000020	240%	240%
90	0.00012	0.0000013	45%	394%
120	0.00014	0.0000007	18%	484%
150	0.00017	0.0000009	18%	591%
180	0.00019	0.0000006	11%	667%
200	0.00021	0.0000009	10%	741%
Canopy ≥7.2m IIC				
30	0.00218			
60	0.00339	0.0000406	56%	56%
90	0.00411	0.0000240	21%	89%
120	0.0046	0.0000177	13%	113%
150	0.00511	0.0000155	10%	135%
180	0.00465	-0.0000155	-9%	113%
200	0.00577	0.0000562	24%	165%
Canopy ≥3m IIC				
30	0.0042971			
60	0.0055858	0.0000430	30%	30%
90	0.0065495	0.0000321	17%	52%
120	0.0073046	0.0000252	12%	70%
150	0.0080106	0.0000235	10%	86%
180	0.0086367	0.0000209	8%	101%
200	0.0089428	0.0000153	4%	108%

Appendix C) Tree canopy sample plot's Connectivity (IIC) Data

Appendix Table 7: IIC results for the canopy ≥17.1m of the four tree canopy sample plots.

Canopy ≥17.1m - Higher Broughton									
Gap Crossing Distance (m)	2005 IIC	2005 % change	2005 total % change	2009 IIC	2009 % change	2009 total % change	2013 IIC	2013 % change	2013 total % change
30	0.000003			0.000005			0.00002		
60	0.000003	0%	0%	0.000006	18%	18%	0.00003	43%	43%
90	0.000003	0%	0%	0.000006	2%	20%	0.00003	11%	59%
120	0.000003	0%	0%	0.000006	0%	20%	0.00004	38%	119%
150	0.000003	0%	0%	0.000006	0%	20%	0.00005	9%	139%
180	0.000004	52%	52%	0.000008	36%	63%	0.00005	7%	157%
200	0.000004	0%	52%	0.000008	5%	71%	0.00005	1%	159%
Canopy ≥17.1m - Lower Kersal									
Gap Crossing Distance (m)				2009 IIC	2009 % change	2009 total % change	2013 IIC	2013 % change	2013 total % change
30				0.000001			0.00001		
60				0.000001	0%	0%	0.00001	42%	42%
90				0.000001	0%	0%	0.00001	2%	45%
120				0.000001	50%	50%	0.00001	3%	50%
150				0.000001	0%	50%	0.00001	0%	50%
180				0.000001	0%	50%	0.00001	0%	50%
200				0.000001	0%	50%	0.00001	0%	50%
Canopy ≥17.1m - Peel Park									
Gap Crossing Distance (m)	2005 IIC	2005 % change	2005 total % change	2009 IIC	2009 % change	2009 total % change	2013 IIC	2013 % change	2013 total % change
30	0.002778			0.002849			0.010370		
60	0.005523	99%	99%	0.005198	82%	82%	0.016331	57%	57%
90	0.006094	10%	119%	0.006946	34%	144%	0.017679	8%	70%
120	0.006771	11%	144%	0.007159	3%	151%	0.018483	5%	78%
150	0.006892	2%	148%	0.007445	4%	161%	0.019203	4%	85%
180	0.007000	2%	152%	0.007640	3%	168%	0.019772	3%	91%
200	0.007092	1%	155%	0.007696	1%	170%	0.020144	2%	94%

Canopy ≥17.1m - Kersal									
Gap Crossing Distance (m)	2005 IIC	2005 % change	2005 total % change	2009 IIC	2009 % change	2009 total % change	2013 IIC	2013 % change	2013 total % change
30	0.003433			0.001525			0.000717		
60	0.006411	87%	87%	0.004331	184%	184%	0.002337	226%	226%
90	0.007803	22%	127%	0.005306	23%	248%	0.00283	21%	295%
120	0.008815	13%	157%	0.005928	12%	289%	0.003246	15%	353%
150	0.009566	9%	179%	0.006559	11%	330%	0.003565	10%	397%
180	0.010224	7%	198%	0.006981	6%	358%	0.003806	7%	431%
200	0.010429	2%	204%	0.007277	4%	377%	0.003967	4%	453%

Appendix Table 8: IIC results for the canopy ≥7.2m of the four tree canopy sample plots.

Canopy ≥7.2m - Higher Broughton									
Gap Crossing Distance (m)	2005 IIC	2005 % change	2005 total % change	2009 IIC	2009% change	2009 total % change	2013 IIC	2013 % change	2013 total % change
30	0.000136			0.000191			0.00022		
60	0.000301	122%	122%	0.000467	144%	144%	0.00048	116%	116%
90	0.000466	55%	243%	0.000604	29%	215%	0.00061	28%	175%
120	0.000531	14%	291%	0.000697	15%	264%	0.00070	15%	217%
150	0.000593	12%	336%	0.000771	11%	303%	0.00077	10%	248%
180	0.000634	7%	366%	0.00082	6%	328%	0.00082	7%	271%
200	0.000657	4%	384%	0.000853	4%	346%	0.00085	4%	287%
Canopy ≥7.2m - Lower Kersal									
Gap Crossing Distance (m)	2005 IIC	2005 % change	2005 total % change	2009 IIC	2009% change	2009 total % change	2013 IIC	2013 % change	2013 total % change
30	0.000030			0.00007			0.00004		
60	0.000041	39%	39%	0.00009	36%	36%	0.00005	37%	37%
90	0.000044	7%	48%	0.00010	12%	52%	0.00006	12%	53%
120	0.000046	5%	55%	0.00011	4%	58%	0.00006	0%	53%
150	0.000047	2%	57%	0.00011	3%	62%	0.00007	24%	89%
180	0.000048	2%	61%	0.00012	4%	68%	0.00008	5%	98%
200	0.000048	1%	63%	0.00012	1%	70%	0.00008	1%	100%
Canopy ≥7.2m - Peel Park									

Gap Crossing Distance (m)	2005 IIC	2005 % change	2005 total % change	2009 IIC	2009 % change	2009 total % change	2013 IIC	2013 % change	2013 total % change
30	0.0173			0.040127			0.039063		
60	0.0260	50%	50%	0.048495	21%	21%	0.048068	23%	23%
90	0.0297	14%	72%	0.052019	7%	30%	0.051781	8%	33%
120	0.0322	8%	86%	0.053875	4%	34%	0.053671	4%	37%
150	0.0334	4%	93%	0.055742	3%	39%	0.055389	3%	42%
180	0.0348	4%	101%	0.056802	2%	42%	0.056407	2%	44%
200	0.0356	2%	106%	0.057791	2%	44%	0.057105	1%	46%
Canopy ≥7.2m - Kersal									
Gap Crossing Distance (m)	2005 IIC	2005 % change	2005 total % change	2009 IIC	2009 % change	2009 total % change	2013 IIC	2013 % change	2013 total % change
30	0.024715			0.037678			0.010209		
60	0.031231	26%	26%	0.043411	15%	15%	0.015235	49%	49%
90	0.035595	14%	44%	0.047734	10%	27%	0.018577	22%	82%
120	0.038324	8%	55%	0.051424	8%	36%	0.021145	14%	107%
150	0.040904	7%	66%	0.053866	5%	43%	0.023076	9%	126%
180	0.042921	5%	74%	0.055744	3%	48%	0.02464	7%	141%
200	0.043744	2%	77%	0.057061	2%	51%	0.025592	4%	151%

Appendix Table 9: IIC results for the canopy ≥3m of the four tree canopy sample plots.

Canopy ≥3m - Higher Broughton									
Gap Crossing Distance (m)	2005 IIC	2005 % change	2005 total % change	2009 IIC	2009% change	2009 total % change	2013 IIC	2013 % change	2013 total % change
30	0.000520			0.000582			0.000395		
60	0.000887	71%	71%	0.000969	67%	67%	0.000765	94%	94%
90	0.001099	24%	111%	0.001187	22%	104%	0.000936	22%	137%
120	0.001253	14%	141%	0.001359	15%	134%	0.001073	15%	171%
150	0.001379	10%	165%	0.001488	9%	156%	0.001172	9%	196%
180	0.001469	6%	183%	0.001587	7%	173%	0.001250	7%	216%
200	0.001521	4%	193%	0.001649	4%	183%	0.001298	4%	228%
Canopy ≥3m - Lower Kersal									

Gap Crossing Distance (m)	2005 IIC	2005 % change	2005 total % change	2009 IIC	2009% change	2009 total % change	2013 IIC	2013 % change	2013 total % change
30	0.000042			0.000178			0.000162		
60	0.000055	31%	31%	0.000226	27%	27%	0.000204	25%	25%
90	0.000060	9%	43%	0.000250	11%	41%	0.000230	13%	42%
120	0.000069	15%	64%	0.000268	7%	51%	0.000249	8%	53%
150	0.000072	4%	70%	0.000277	3%	56%	0.000259	4%	60%
180	0.000074	3%	76%	0.000286	3%	61%	0.000269	4%	66%
200	0.000075	1%	78%	0.000289	1%	63%	0.000273	2%	68%
Canopy ≥3m - Peel Park									
Gap Crossing Distance (m)	2005 IIC	2005 % change	2005 total % change	2009 IIC	2009 % change	2009 total % change	2013 IIC	2013 % change	2013 total % change
30	0.019645			0.052627			0.040881		
60	0.028668	46%	46%	0.057147	9%	9%	0.053218	30%	30%
90	0.033016	15%	68%	0.062025	9%	18%	0.056853	7%	39%
120	0.035587	8%	81%	0.063363	2%	20%	0.058468	3%	43%
150	0.037030	4%	88%	0.064777	2%	23%	0.059674	2%	46%
180	0.038655	4%	97%	0.065636	1%	25%	0.060729	2%	49%
200	0.039556	2%	101%	0.067123	2%	28%	0.061638	1%	51%
Canopy ≥3m - Kersal									
Gap Crossing Distance (m)	2005 IIC	2005 % change	2005 total % change	2009 IIC	2009 % change	2009 total % change	2013 IIC	2013 % change	2013 total % change
30	0.030226			0.049039			0.01161		
60	0.037213	23%	23%	0.055314	13%	13%	0.017181	48%	48%
90	0.042609	15%	41%	0.060953	10%	24%	0.020937	22%	80%
120	0.045630	7%	51%	0.064343	6%	31%	0.023829	14%	105%
150	0.048657	7%	61%	0.067421	5%	37%	0.026016	9%	124%
180	0.051029	5%	69%	0.069426	3%	42%	0.027785	7%	139%
200	0.052030	2%	72%	0.071849	3%	47%	0.028856	4%	149%

Appendix D) Node Importance (dIIC) results

Appendix Table 10: The node importance (dIIC) results for the canopy ≥3m.

Canopy ≥3m 30m Graph		Canopy ≥3m 200m Graph	
Node ID	dIIC_30m	Node ID	dIIC_200m
3944	63.70	3944	45.92
3157	9.41	3157	6.34
2046	8.02	2046	5.31
2132	4.22	466	3.16
601	3.88	9516	3.08
1416	3.76	233	3.06
466	3.65	1416	2.90
233	3.50	601	2.73
1334	3.44	7904	2.64
9516	3.28	5213	2.21
678	2.87	6911	2.06
6911	2.21	7731	2.02
7904	2.16	1334	2.00
7712	1.87	5865	1.82
914	1.61	2132	1.61
1035	1.56	4646	1.41
294	1.50	5653	1.41
649	1.50	10175	1.35
7731	1.48	9108	1.30
10175	1.44	8944	1.15
5213	1.38	649	1.06
5865	1.38		
726	1.34		
4165	1.18		
5653	1.09		
9108	1.08		
3362	1.07		
3090	1.05		
3123	1.05		

Appendix Table 11: The node importance (dIIC) results for the canopy ≥7.2m.

Canopy ≥7.2m 30m Graph		Canopy ≥7.2m 200m Graph	
Node ID	dIIC_30m	Node ID	dIIC_200m
1967	69.49	1967	43.86
1301	11.63	1652	5.46
1652	9.58	1301	5.05
1113	6.73	1113	4.83
122	4.38	4262	3.69
275	4.15	122	3.50
788	3.26	275	3.21
1082	2.35	788	2.55
350	2.28	3523	2.09
838	2.12	350	1.99
3032	1.77	838	1.77
4262	1.73	3251	1.65
431	1.62	1082	1.43
3251	1.55	3032	1.41
171	1.47	4088	1.33
3523	1.47	2817	1.14
349	1.44	2525	1.06
744	1.03	171	1.04
		4598	1.03

Appendix Table 12: Node importance (dIIC) results for the canopy ≥17.1m.

Canopy ≥17.1m 30m Graph		Canopy ≥17.1m 200m Graph	
Node ID	dIIC_30m	Node ID	dIIC_200m
491	25.99	491	13.31
424	12.50	424	9.37
451	12.29	451	7.19
443	10.20	31	7.17
489	9.61	489	5.06
31	6.22	531	3.88
306	5.65	657	3.69
569	4.53	443	3.65
531	3.92	670	3.45
479	3.89	680	3.31
509	3.83	569	2.78
405	2.99	486	2.69
396	2.81	171	2.42
542	2.81	306	2.26
413	2.76	509	1.96
770	2.28	487	1.62
488	2.26	542	1.55
357	2.25	214	1.44
427	2.19	770	1.37
370	1.86	510	1.36
339	1.73	427	1.32
577	1.64	75	1.29
510	1.57	12	1.28
791	1.57	148	1.27
753	1.51	579	1.21
487	1.39	312	1.20
492	1.39	398	1.19
463	1.36	332	1.09
148	1.34	357	1.07
746	1.30	169	1.02
373	1.30	522	1.01
171	1.29	625	1
75	1.22		
740	1.17		
534	1.12		
292	1.12		
19	1.08		
579	1.08		
364	1.06		
399	1.04		